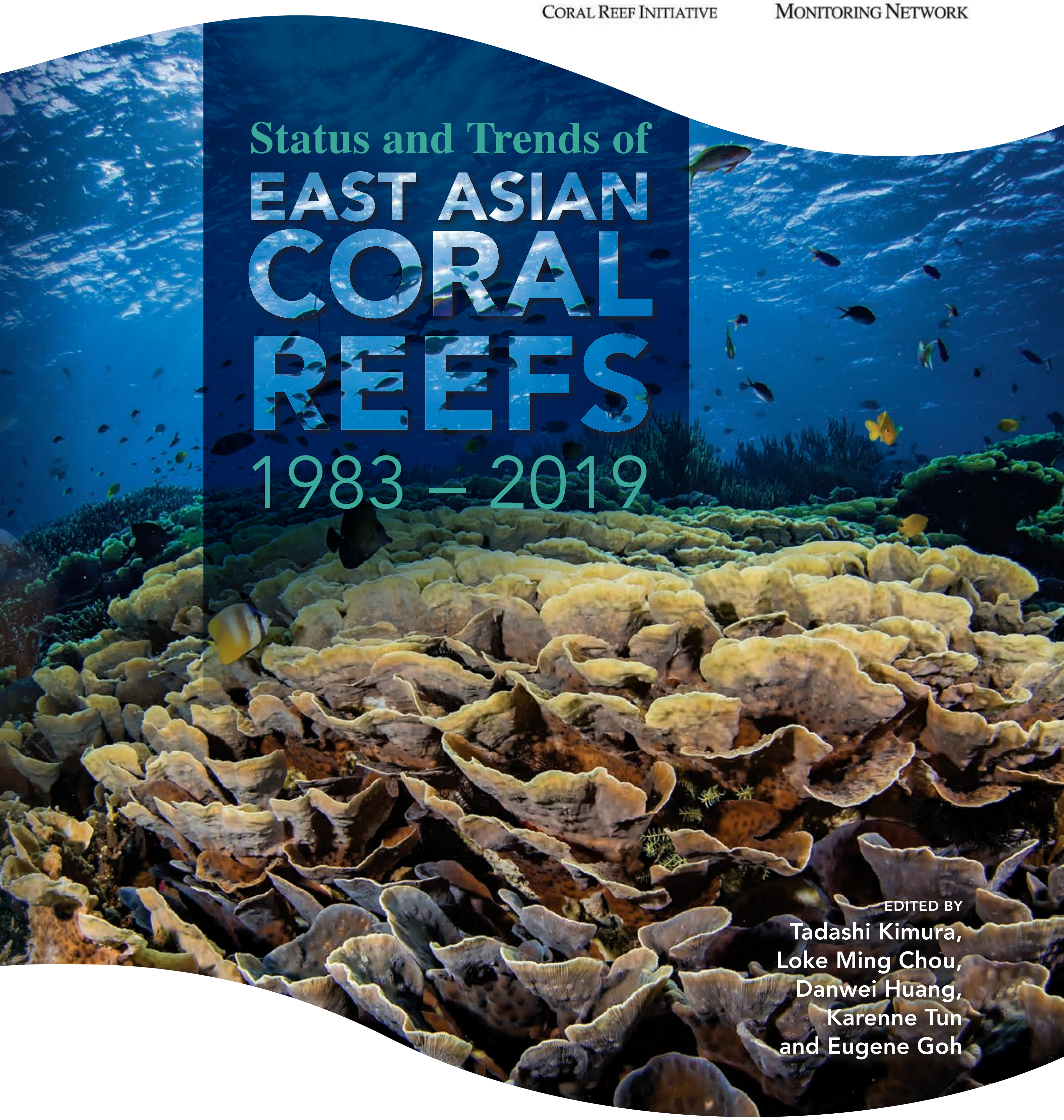
INTERNATIONAL CORAL REEF INITIATIVE

GLOBAL CORAL REEF MONITORING NETWORK

Status and Trends of EAST ASIAN CORAL REEFS 1983 – 2019



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

FRONT COVER: Shallow coral reef in Siaba Island, East Nusa Tenggara, Indonesia (© Nicholas Chew)

BACK COVER: Green sea turtle in Sipadan Island, Sabah, East Malaysia (© Nicholas Chew)

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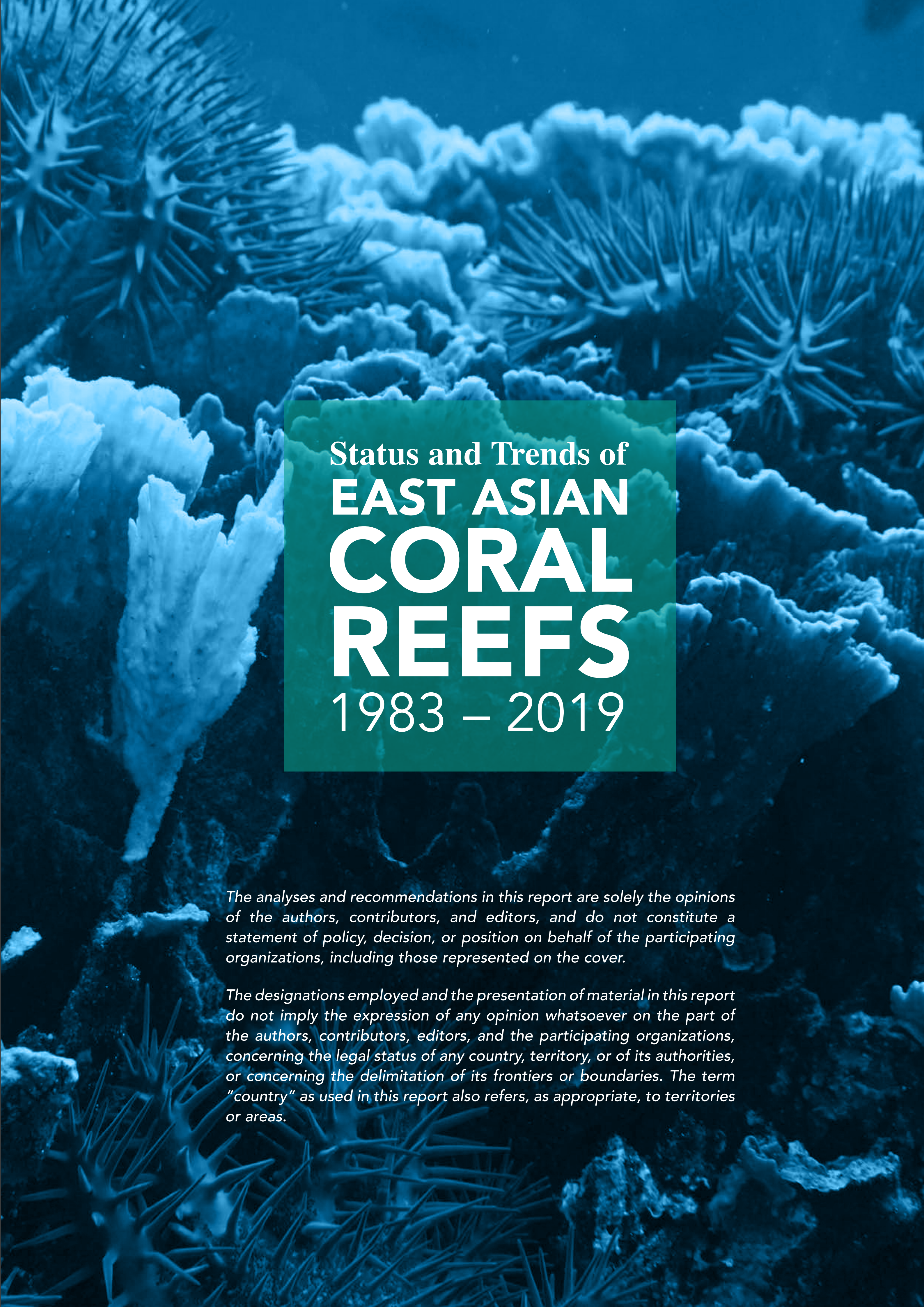

Status and Trends of EAST ASIAN CORAL REEFS 1983 – 2019

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FOREWORD

The International Coral Reef Initiative (ICRI) was established in 1994 to enhance the conservation and management of coral reefs and associated ecosystems of the world based on scientific data and information. To assess the status and trends of coral reefs, ICRI developed the Global Coral Reef Monitoring Network (GCRMN), which promotes cooperation among national research programs and monitoring networks all over the world. For this purpose, GCRMN has published a global report "Status of Coral Reefs of the World" since 1998 for providing scientific information to reef managers and decision-makers.

Coral reef scientists in the East Asia Region have been called upon to provide necessary information to the report under the global coordination of GCRMN. Through the contribution of the individual scientists for the first four global reports, the regional network has become stronger and started publishing its own status report since 2004 to provide more comprehensive data and information for effective management in the region.

This 2022 report is the latest publication prepared as a special issue on regional data analysis with support from the National University of Singapore. The report described decadal changes of coral reefs in the region to provide recommendations on their conservation for national governments and regional organizations into the future. I sincerely hope that it will be widely used.

I would like to express my gratitude to Professor Loke Ming Chou for his strong leadership in the region, Dr. Danwei Huang, and Samuel Chan for their technical support on data analysis, and all those involved in preparing this report. I would also like to especially thank Mr. Eugene Goh and Dr. Karenne Tun for their endless efforts on all the editorial works that made this regional report possible.

Tadashi Kimura

GCRMN Regional Coordinator of East Asia

PREFACE

In response to the International Coral Reef Initiative's (ICRI) "Call to Action" at its 1995 Dumaguete meeting, the Global Coral Reef Monitoring Network (GCRMN) was launched in 1996 with the purpose of collecting information on the state of coral reefs and raising awareness on the importance of coral reef conservation. The working arrangement involved establishing regional networks to provide effective coordination and cooperation among national reef research programs in maritime countries of each region.

The GCRMN Regional Coordinators took the lead for the East Asian Seas region and, in consultation with national coordinators, published the 'Status of Coral Reefs in the East Asian Seas Region' in 2004, 2010, 2014, and 2018. These reports aim to compile detailed information on reef conditions based on reef monitoring data generated by countries in a region that has the world's richest coral reef biodiversity and also the highest threats.

The previous publications were mainly a compilation of national reports on the status and management of coral reefs. In addition, trends across the region, such as coral species migration to higher latitudes in response to elevated sea surface temperature, and the impact of global bleaching events, were highlighted. The reports of 2014 and 2018 emphasized the need to go beyond national reporting and analyze reef status trends across the region in response to threats and management over the long term. The initiative to gather and integrate long-term monitoring data from national coordinators and other contributors began in 2017. The first regional analysis is completed and included in this report, thanks to the effort of a team of dedicated reef data analysts and modelers. This work is not over but has only just begun, which means more work for the data analysis and modeling team. The established regional database can now be further expanded on geographical and temporal scales.

Continued coral reef monitoring and reporting remain crucial as they build up the basis for the regional analysis. The national coordinators, all operating on a voluntary, informal, and collegial basis, have been extremely helpful and forthcoming with their contributions. They will be relied on to strengthen the regional database, which will eventually help to improve the relevancy and timeliness of a more appropriate management response to a specific challenge.

Loke Ming Chou
Chairman, Asia Pacific Coral Reef Society

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This current Status and Trends of East Asian Coral Reefs 1983-2019 report has benefitted from the unwavering support of many individuals and organizations who volunteered time and resources to support the regional network. The generous support has allowed the East Asia GCRMN network to remain actively engaged; produce and publish regular regional coral reef status reports since 2004; and organize yearly thematic workshops, including the four-yearly Asia Pacific Coral Reef Symposium (APCRS).

As regional coordinators, we are particularly grateful to the country coordinators who have worked tirelessly to secure and coordinate coral reef monitoring data for the regional analysis and also assisted in preparing their country status reports. Data contributors and supporting programs whose important work on the ground with community partners have likewise been vital to ensure that coral reef management and conservation are prioritized and continued. We are also thankful to the chapter reviewers for their scientific and technical inputs and the photo contributors* for sharing their beautiful coral reef images from the EAS region. We would also like to express our gratitude to Dr. Danwei Huang and Samuel Chan of the National University of Singapore for their long-term efforts on all the analysis to provide us with the essentials of this report.

Tadashi Kimura & Karenne Tun

GCRMN East Asia regional coordinators

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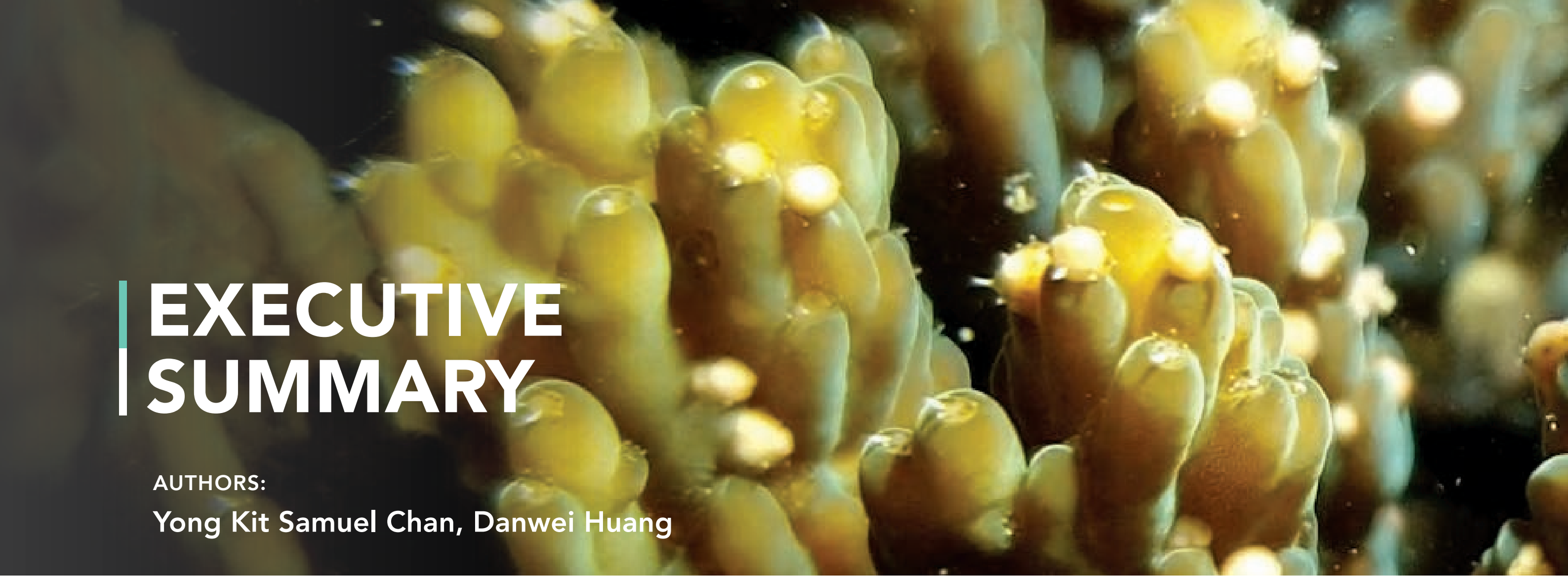
LIST OF ACRONYMS

ASEAN	Association of Southeast Asian Nations
COTS	Crown-of-thorns Starfish
DHW	Degree Heating Week
EA	East Asia
FAO	Food and Agriculture Organization of the United Nations
FFI	Fauna & Flora International
GCBE	Global-Scale Coral Bleaching Event
GCRMN	Global Coral Reef Monitoring Network
GPS	Global Positioning System
LIT	Line Intercept Transect
ICRI	International Coral Reef Initiative
MPA	Marine Protected Area
NEA	Northeast Asia
NGO	Non-Governmental Organization
NOAA	National Oceanic and Atmospheric Administration
PIT	Point Intercept Transect
SCS	South China Sea
SD	Standard Deviation
SEA	Southeast Asia
SST	Sea Surface Temperature
WWF	World Wide Fund for Nature



EXECUTIVE SUMMARY

AUTHORS:
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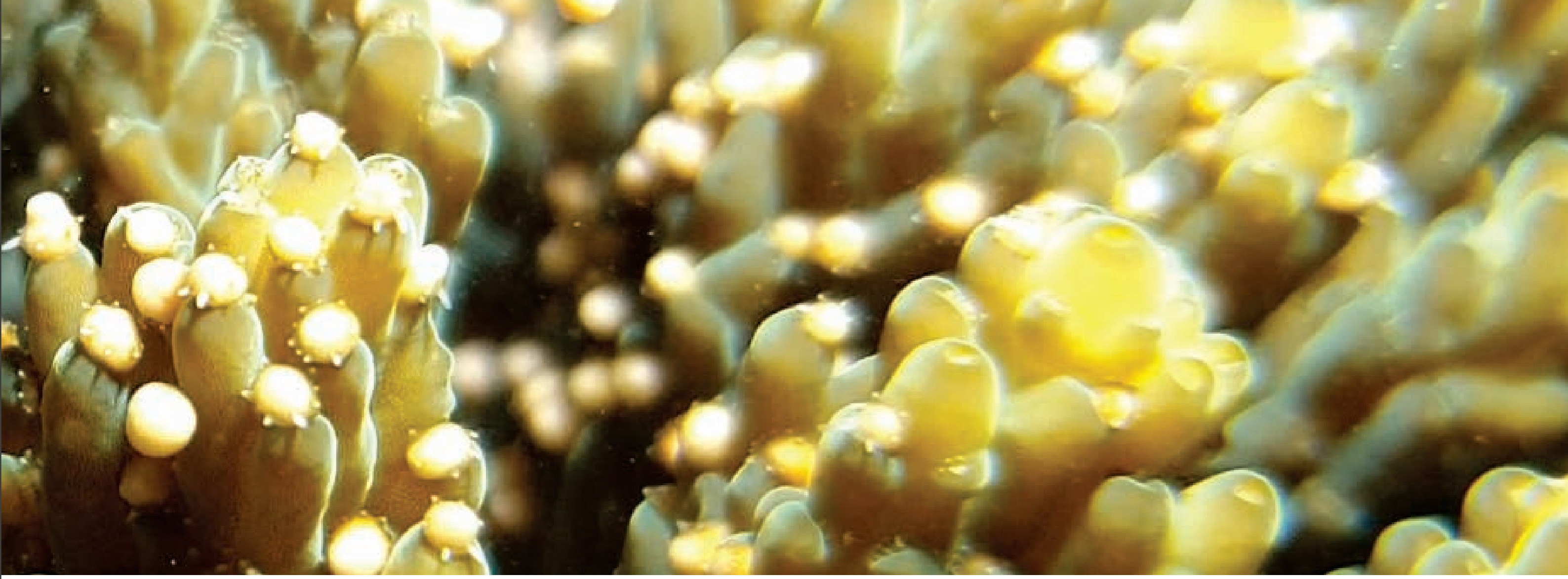
Introduction

The Northeast and Southeast Asia nodes of the Global Coral Reef Monitoring Network (GCRMN) regularly report on the status and trends of coral reefs in the East Asian Seas region under the framework of the International Coral Reef Initiative (ICRI). Regional and country-level reporting has lead to the Status of Coral Reefs in East Asian Seas Region series published in 2004, 2010, 2014 and 2018. The current report leverages the efforts of many individual countries and partner NGOs to study and monitor reef ecosystems in the East Asian Seas. It also builds upon past collaborative work in the region, such as the ASEAN-Australia Living Coastal Resources project in the 1990s (Wilkinson et al. 1992, Chou et al. 1994), and the associated data which have been consolidated for analysis here.

Following recent GCRMN reports from other regions, the Status and Trends of East Asian Coral Reefs examines the trends, status, threats, and management of reefs in the East Asian Seas. It first and foremost tests if and how reefs in the region have changed over time, building on more qualitative reports of the past (Kimura et al. 2014), and reports on trends and stressors such as mass coral bleaching events at the national level. While the regions' reefs have not been as prominent in academic research as the Caribbean, they represent many of the largest and most diverse reef ecosystems in the world and are situated within several endemism hotspots (Hughes et al. 2002, Veron et al. 2009, Huang et al. 2015). The region also suffers from a slew of global threats like climate-induced sea surface temperature rise and ocean acidification, as well as local impacts such as coastal pollution, development, and exploitation (Chou 1994, Hughes et al. 2003, Hoegh-Guldberg et al. 2007, Burke et al. 2012). The report thus aims to evaluate the status and trends of East Asian reefs in this context and identify potential environmental and socio-economic variables that are associated with reef change, applying a more quantitative approach compared to past GCRMN reports of this region. Results will better inform current and future management of the region's reefs, yielding a clearer understanding of the changes in coral cover and health over the last three and a half decades and building on regional efforts for continued reef monitoring into the future.



Strategy & Scope

In the Indo-Pacific region, large-scale studies examining coral cover change are not new. For example, Bruno and Selig (2007) identified a decline in Indo-Pacific reef coral cover between 1968 and 2004 based on a meta-analysis of reef surveys. Here, we curated local reef survey data provided by various GCRMN national coordinators from the Northeast Asia and Southeast Asia GCRMN nodes to generate a regional dataset of benthic, fish and invertebrates recorded on reefs extending from 1983 to 2019. Apart from long-term monitoring data obtained from GCRMN national coordinators, additional data were requested from partnering resource persons and organizations working throughout larger reef areas like Indonesia, Malaysia, and the Philippines, allowing for greater specificity and veracity of data as compared to using available but unverified data that may exist online. Some of these data, like in the Caribbean GCRMN report by Jackson et al. (2014), were also not within public domain, with older datasets only known to researchers who have worked to collect the data or in their native language (Nuñez and Amano 2021). While not all the data that were available to us could be used because of data sharing restrictions and lack of a common methodology, the current dataset still comprises one of the largest in the region, with 24,365 transects from 1,972 sites in 13 countries.

To maximize data representation through time, we applied stringent criteria to retain only reefs surveyed at a minimum of three time points over five years, resulting in a total of 21,107 transects to be analyzed. Despite the voluminous survey data obtained, coverage remains geographically and temporally limited. Japan, followed by Singapore, had the longest and most complete dataset, followed by other countries that were part of the ASEAN-Australia Living Coastal Resources project (i.e., Indonesia, Malaysia, Philippines, and Thailand). The remaining countries had much fewer and temporally restricted datasets. Macroalgal data were more limited, with only 15,697 transects. Similarly, data for fish and invertebrates were extremely variable and sparse, and thus were not used in this analysis. Our analyses characterized the trends of both coral and macroalgal cover across years and depths at both the regional and national scales. We also modelled these variations against environmental and socio-economic parameters to infer drivers of reef change in the East Asian Seas.

Regional Trends In Benthic Cover

Contrary to previous Indo-Pacific studies and other regional studies, there was limited evidence supporting a long-term decline in coral cover. Coral cover did vary across time, and analyses showed a slight trend of increasing cover over time, but the magnitude of the change was low. Macroalgal cover also showed minimal changes with a slight increase over time. Uncertainties around the trends were high from the late 1980s till about 1995 due to much less available data in the past. Across depths, both corals and macroalgae showed much higher cover nearer the sea surface, with a steep drop through about 6 m before flattening out, though coral cover continued to decline further past 10 m.

There were considerable country-level variations, with countries differing in means and spread of the data. In general, most countries had coral cover of around 0.2 to 0.4 with Myanmar registering the highest cover, likely an anomaly due to their recency of surveys. The more urbanized and developed countries like Japan and Singapore also appeared to have the lowest coral cover, compared to other Southeast Asian countries such as Indonesia, Malaysia, Philippines, and Thailand. Other countries had intermediate coral cover. Macroalgal cover was also generally low across the countries with markedly low spatial variation. Most countries appeared to converge in terms of coral cover towards the present, with different countries showing varying trends over time.

The lack of a strong decline over the last 3.5 decades can be attributed to several factors. Most importantly, it is likely that the limited surveys from the 1980s already represent a shifted baseline (Knowlton and Jackson 2008). Analogously, according to the Caribbean GCRMN analysis, coral cover had already declined to historical low levels by the late 1980s (Jackson et al. 2014). There is also anecdotal evidence showing that reefs prior to the 1980s had high coral cover and were in a more pristine state than today. While Bruno and Selig (2007) noted declines in their study, data prior to the 1980s were limited and yet showed higher coral cover compared to later years. Potentially, this means that there may have been a more drastic decline before the timeframe analyzed in the current study. However, without quantitative data from before the 1980s, it is impossible to ascertain the scale and confidence of this prediction.

The trends depicted here also represent a mean trend for all the reefs in the region, without accounting for the extent of individual reefs. Estimates of size and scale of the reefs in this region are rough because the coastlines and more turbid environmental conditions do not permit ease of remotely sensed reef area calculations. While estimates exist, variability is also high due to the large archipelagic islands of the various countries (Spalding and Grenfell 1997). Thus, it is difficult to use these areal data to predict change in total reef cover here.

Finally, the less severe decline in coral cover and lack of increase in macroalgal cover could potentially be due to the diversity of the reefs found in the region. The high reef diversity here may have helped to mitigate the effects of environmental and anthropogenic changes, encouraging recovery, or promoting resilience of reefs to avert coral cover losses from disturbance events (Nyström 2006, Hughes 2010). Coral community changes have also been observed at local scales, showing elevated abundances of stress-tolerant and weedy species compared to more competitive species in response to certain environmental pressures with limited change in coral cover (Darling et al. 2013). This may mean that as disturbance increases, species and functional redundancies are lost first before coral cover begins to decrease.

The low cover of macroalgae through time also supports the rarity of coral to macroalgal phase shifts in the region (Bruno et al. 2009). Aside from coral diversity, the variety of reef herbivores are likely controlling macroalgal growth as well (Lefcheck et al. 2019). While the different regions do host differing amounts of macroalgae, driven partially by seasonality, the overall trend of macroalgal increase is weak (Bruno et al. 2014). Sessile benthic animals such as corallimorpharians, soft corals and zoantharians also occupy the benthos and compete with macroalgae to limit their growth. Indeed, phase shifts in more nutrient-laden waters may also involve the dominance of these reef invertebrates (Norström et al. 2009).

National Trends In Coral Cover

At the national level, coral cover generally did not vary consistently over time and changes were variable between countries. Coral bleaching was recorded in all Southeast Asian countries, mostly coinciding with the El Niño-associated global-scale coral bleaching events (GCBE) in 1998, 2010 and 2016, whereas bleaching was more localized and less severe in Northeast Asian sites.

In summary, for Northeast Asia:

CHINA

Coral cover decreased from over 30% to less than 10% on average from 2004 to 2018, with offshore reefs registering greater declines compared to inshore reefs. Reefs were relatively unaffected by the global coral bleaching events, even in 2016, but localized thermal bleaching events had been recorded in Sanya, the Xisha Islands and Nansha Islands with increasing frequency since 2016.

HONG KONG

The extent of bleaching has been limited generally and no significant mortality of corals has been associated with bleaching. Severe coral bleaching was documented in 1997 and 2017, likely to have been caused by reduced salinity due to heavy rainfall.

TAIWAN

Coral cover had been generally stable at an average of about 33% from 1997 to 2017. There were isolated years of decline, such as between 2016 and 2017 when the mean cover dipped to about 25% due to the 2016 GCBE. During this event, bleaching was recorded in 9 of the 19 sites surveyed.

JAPAN

The initial years of reef surveys from 1983 to 1989 registered low coral cover at about 10% due to an outbreak of the crown-of-thorns starfish (COTS), recovering to 30% by 1995 but rapidly followed by degradation due to disturbances by typhoons and mass coral bleaching in 1998. A short period of recovery occurred between 2001 and 2004, but since then, there had been gradual degradation caused by chronic typhoon disturbances, COTS outbreaks, coral diseases, and extensive and severe coral bleaching in 2016.

SOUTH KOREA

Coral assemblages were not common, with lower coral cover at the Busan Metropolitan City region (10–18%) compared to Jeju Island (15–38%). Based on limited surveys from 2010 to 2017, coral cover appeared to exhibit a minor increase. Mass coral bleaching had never been recorded; coral colonies were not affected by the 2016 GCBE.

In summary, for Southeast Asia:

BRUNEI

Mean coral cover showed a slightly declining trend from the 1990s to 2011 but was maintained at 30–45%. Cover increased gradually from 2014 to 2016. The initial decline may be driven in part by the sparse data in the 1990s. Coral bleaching was last observed in 2016, with up to 60% of corals bleached at the site level.

CAMBODIA

Coral cover increased steadily from 2010 to 2018. Coral bleaching was recorded in 2010 and also in 2015–2017, being most severe in 2016 with 47% of sites registering moderate bleaching and 24% of sites severely bleached. However, these had no perceivable influence on the increasing coral cover trend.

INDONESIA

Coral cover was higher between 2014 and 2019 at a mean of about 30%, and stably ranging from 26% to 32%, compared to 20% cover during the late 1980s. There had been a slight upward trend from 2016 to 2019, after the last GCBE in 2016, with previous major events recorded in 1983, 1998 and 2010. Coral mortality during the 2016 GCBE was between 30% and 90%, recorded in the waters of East Nusa Tenggara, West Nusa Tenggara, South Java, West Sumatra, North Bali, Lombok, Karimun Java and Selayar. Reef recovery was faster in central Indonesia than in western Indonesia.

MALAYSIA

Coral cover was at nearly 50% prior to 1990, but it had declined by the early 2000s. Following the mass bleaching event in 2010, coral cover recovered to nearly 40% in 2016 but declined again to about 30% because of the GCBE that year, with limited recovery as of 2018. Peninsular Malaysia's coral cover has fluctuated between 30% and 45% from 2010 to 2018. The slight increase after the 2010 mass coral bleaching event was negated by the 2016 GCBE, and coral cover has remained relatively constant since then. In East Malaysia, coral cover ranged stably from 34% to 41% between 2010 and 2015, but declined to 29% in 2016, which saw the GCBE and COTS outbreak at Lankayan Island, Sabah. It had recovered to 34% by 2018.

MYANMAR

Data available between 2013 and 2018 showed an overall decline in coral cover, although it remained high at above 40% cover on average in 2018 and 2019. Throughout the survey period, sites were extremely heterogeneous, with cover as low as 0% and as high as 100%. Myeik Archipelago, for instance, showed increase in cover from 2014 to 2019 despite the national decline. More data are needed to infer trends and reef health.

PHILIPPINES

Over the last decade, mean coral cover showed strong declining trends in the South Philippine Sea and the Sulu Sea, with marginal declines in the North Philippine Sea, Visayas Region (inland seas) and the Celebes Sea. Coral cover increased in the West Philippine Sea from 10% in 2008 to 32% in 2018. Overall, the cover data reported here are higher than a recent report by Licuanan et al. (2019), mainly because most of the sites surveyed here are located within marine protected areas. Between 2015 and 2017, coral bleaching was reported in 36 (54%) of the 66 coastal and island provinces surveyed. Most of the confirmed bleaching reports were in 2016 (79%), when bleaching incidences were reported almost year-round, although most reports (81%) were of low to mild bleaching. Moderate to severe coral bleaching was reported between April and October in 2016. The following year also saw bleaching, but reports were generally of low to mild bleaching.

SINGAPORE

Mean coral cover declined drastically from 30% in 1986 to 10% in 1990, driven at least in part by coastal development and reclamation. Coral cover gradually increased to a mean of 25% in 2009, punctuated by a 4% drop coinciding with the 1998 mass bleaching episode. There were also clear but smaller effects of the 2010 and 2016 GCBE. Coral cover decreased considerably with increasing depth to less than 10% beyond the shallowest 3 m, indicating that reduced light availability had been driving losses at the deeper reef areas.

THAILAND

Coral cover decreased precipitously from a mean of about 50% before 1990 to <40% in recent years. The moderate coral bleaching events in the Andaman Sea in 1991 and 1993, as well as the severe bleaching event in the Gulf of Thailand in 1998 caused a reduction of coral cover by about 10% from 1991 to 2000. Subsequently, reefs continued to decline, driven mostly by the 2010 bleaching event, but coral cover eventually recovered to a mean of 38% in 2017. Bleaching severity was variable among sites during the 2016 GCBE, with mortality being much lower than the 2010 bleaching event.

VIETNAM

Between 2001 and 2017, mean coral cover fluctuated annually, with an overall increasing trend from 23% to 34%. Specifically, coral cover was stable from 2001 to 2005 (23.1–28.8%), increased to 33.3% in 2010, and notably declined to 27.5% in 2011 after the 2010 mass bleaching event. Following this decline, coral cover increased to 32% in 2016 and 34.2% in 2017. Coral bleaching occurred during 2015 and 2016, and was variable among sites, with the 2016 event being more serious than in 2015.

Potential Drivers Of Change

In relating benthic cover data with environmental and socio-economic covariates (Yeager et al. 2017), the averaged model revealed variable effects on both coral and macroalgal cover. The averaged model for macroalgal cover contained several variables and incorporated more similar-performing models compared to the coral cover model, indicating greater uncertainty in the tested associations. Anthropogenic impacts were evident for both cover types, with higher population correlating with higher macroalgal cover and lower coral cover, in addition to smaller distances from major markets for coral cover. Maximum net primary productivity (NPP), measured through chlorophyll a, sea surface temperature (SST) and photosynthetically active radiation (PAR) correlated with increasing coral and macroalgal cover but increasing standard deviation in NPP resulted in higher macroalgal cover at the expense of coral cover. Temperature variables in both models were different with only bleaching alert levels (BAA) negatively correlated with coral cover. For macroalgal cover, higher minimum, mean temperatures and degree heating weeks correlated with macroalgal cover positively while higher maximum SST and SST anomalies correlated negatively. Variables associated with temporal and spatial trends such as year, depth and country also featured in the averaged models, though not all individual country variables were significant.

The environmental and socio-economic correlates tested, while not altogether predictive of overall reef benthic cover, do independently highlight some of the potential drivers of different benthic categories through time. The clear influence of anthropogenic variables highlights that there is evidence for human pressures on reefs (Heery et al. 2018), while productivity and temperature have more complex effects on how benthic cover changes (Darling et al. 2019, McClanahan et al. 2019). The lack of significance of country variables reinforces the fact that marine ecosystems transcend political boundaries and monitoring efforts may need to be coordinated at the regional level to maximize biogeographic and ecological relevance (Veron et al. 2015).

RECOMMENDATIONS

Long-term monitoring programs yielding data-rich analyses are a necessity for adaptive management of coral reefs and serve to evaluate the effectiveness of past and ongoing management actions (Chou 2013, Hughes et al. 2017, Obura et al. 2019). Increasingly, the information gleaned from surveys presented here, such as coral and macroalgal cover alone, may not be adequate to discern the effects of anthropogenic impacts on reefs (Lam et al. 2017). Furthermore, while region-wide trends were apparent, not all countries had significantly different cover over time and trajectories were distinct. To keep up with the threats and to advance management based on enhanced knowledge of those threats against corals, monitoring must improve beyond current methods and be coordinated across geographical barriers for the results to benefit coral reef conservation (Flower et al. 2017). We also need to overcome various methodological challenges facing conventional surveys performed in the region, including accessibility of the more remote reefs, difficulty in taxonomic identification, greater potential diversity to distill, and comparatively lesser available resources for monitoring (Madin et al. 2019, Borja et al. 2020). For at least the historically and ecologically important sites, we suggest implementing standardized approaches incorporating a greater level of detail in data recording, with the collection of environmental data, more taxonomically detailed surveys, and inclusion of other reef organisms. While countries represent the easiest units to coordinate monitoring efforts, data need to be aggregated at geographically meaningful units for further analyses. Though difficult, any steps toward this ideal would yield data and analyses that would be beneficial to not just national but regional monitoring of reefs.

More generally, to aid current monitoring efforts in the region, capacity building remains important to level the skills of surveyors in terms of monitoring techniques and taxonomic identification (Madin et al. 2019). With the large number of reefs in the region, it would be difficult for any individual research or volunteer group to survey a considerable proportion of reefs within any country. Therefore, we recommend a central body to train, collate and vet data collected to ensure accuracy and precision in data recorded. This also eases collation of data for future analyses, ideally within a common data platform for collaborative use, research, and management. These strategies will lead to improved reef management and ensure that coral reefs in the East Asian Seas region continue to thrive into the future.

SUMMARY

In this report, we have analyzed and reported on the status and trends of East Asian reefs from 1983 to 2019. Results appear to paint a distinct picture when compared to other reef regions of the world, and the specific differences noted here are critical for evaluating management policies. In particular, our findings show:

1. There is a lack of clear decline in coral cover from the late 1980s to present day. This could be attributed to an already shifted baseline, the averaging of trends across different reef scales, mitigation by the diversity of reefs in the region, or more likely an interplay among all the above factors. To understand what and how community shifts have occurred, and the timescales involved, more past and community-level data are needed.

2. Correspondingly, there is no clear signal of macroalgal cover increase over time. Macroalgal dominance may be attenuated by resilience due to coral species diversity. Other benthic colonial organisms like corallimorpharians, soft corals and zoantharians are also typical of Indo-Pacific reefs, competing with macroalgae to limit their growth. Overall, there is no unambiguous evidence of phase shifts towards macroalgal dominance at the regional scale, though occasional and seasonal blooms of macroalgae do occur.

3. Benthic cover patterns are both environmentally and anthropogenically driven. Anthropogenic effects correlate negatively with coral cover and positively with macroalgal cover. In general, environmental variables can have distinct effects on corals and macroalgae. For example, increasing maximum productivity enhances both benthic components, but productivity variations result in distinct effects. Temperature variables are potential correlates of benthic cover, but only bleaching alert levels (BAA) are negatively correlated with coral cover. Macroalgal cover is affected by many more variables with more complex relationships. Variables associated with temporal and spatial trends such as year, depth and country also explain the data, though not all country variables are significant.

4. This report underscores the importance of data-driven, long-term monitoring programs. It is increasingly clear that monitoring requires more detailed data, beyond benthic cover and country-based classification. Higher taxonomic resolution of data recording and spatially resolved environmental data are necessary to provide better understanding of how different environmental and socio-economic factors, including anthropogenic drivers, affect reef communities locally and regionally. Critically, marine communities and their associated functions are not constrained by political or national boundaries, so analyses will be more effective when data are aggregated at more ecologically relevant scales. To this end, more capacity building and resources are needed to improve the scale and capability for long-term monitoring and analyses of East Asian reefs, and to foster collaborations within and between GCRMN regional nodes.

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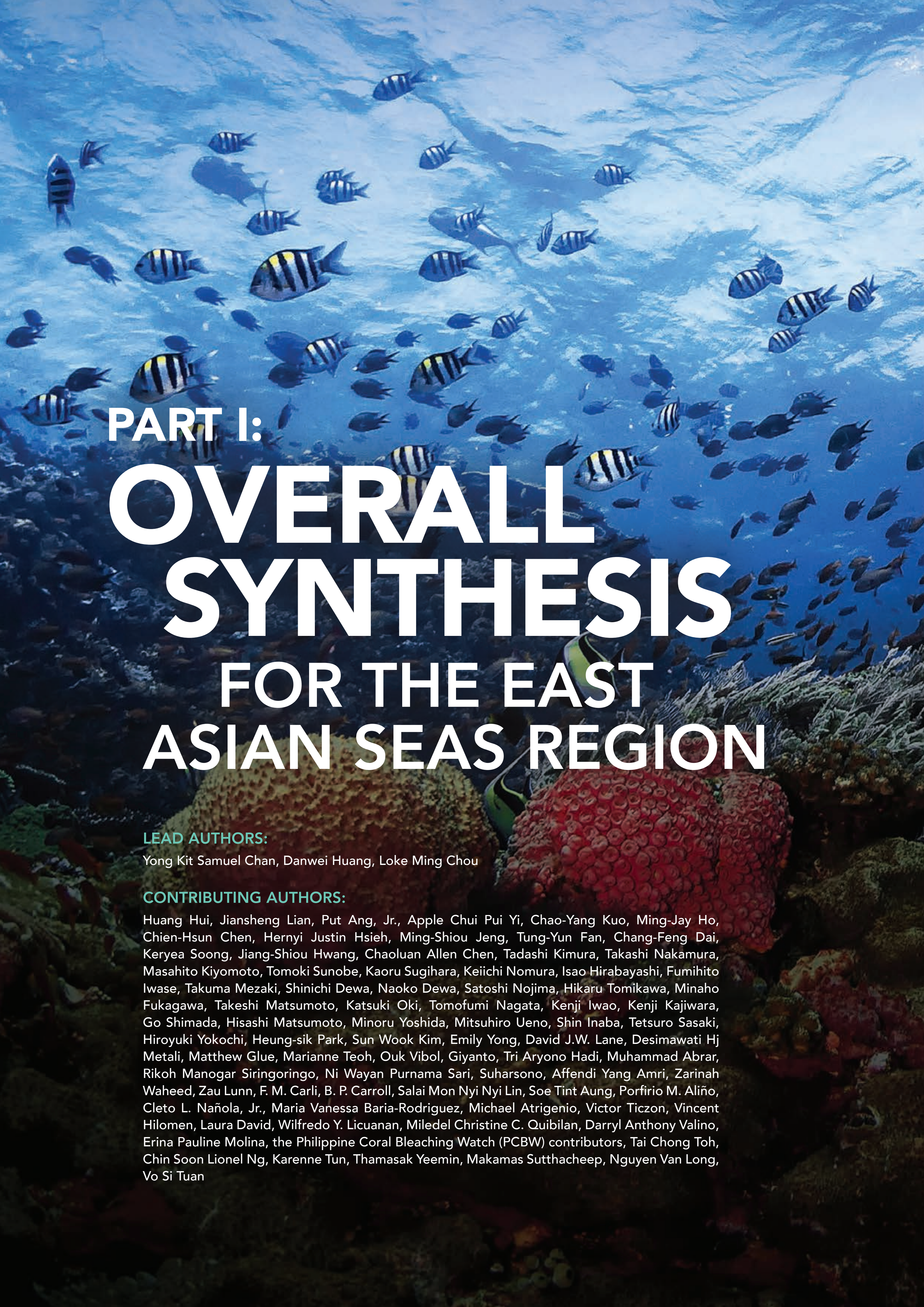

PART I: OVERALL SYNTHESIS FOR THE EAST ASIAN SEAS REGION

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Introduction

Long-term monitoring remains one of the most important means to track changes in ecosystems over time. It enables the early detection of threats affecting the health of ecosystems and allows for effective decision-making and intervention. Increasingly, the necessity for baselines to inform data-driven solutions (Pauly 1995, Knowlton and Jackson 2008) has led to the creation and implementation of many long-term monitoring programs across ecosystems. In part, this progress is assisted by greater available resources and technological advancement. The continual push for monitoring also adds to the available data for further downstream analysis through mechanisms such as the Global Coral Reef Monitoring Network (Wilkinson et al. 1992, Obura et al. 2019).

In the Indo-Pacific region, effective monitoring coverage is hampered due to the massive scale of the reefs present here (Smith 1978, Spalding and Grenfell 1997), the diversity of reefs present (Hughes et al. 2002, Huang et al. 2015), and the limited scientific monitoring resources available (Costello et al. 2010). Yet, impacts such as coastal development, pollution, nutrient runoff, sedimentation, and overfishing are important local-scale threats to reefs in the region (Chou 1994, Knowlton and Jackson 2008, Burke et al. 2012). Additionally, increasing sea surface temperatures have also seen the rise of coral bleaching intensity, while ocean acidification remains a threat to reef carbonate formation (Hoegh-Guldberg 1999, Hughes et al. 2003, 2018, Hoegh-Guldberg et al. 2017). Without monitoring efforts, it is difficult to identify and observe different reefs' responses and changes due to these threats. Global and regional meta-analyses in the past have shown that coral reefs are in decline around the world, and these findings are reflected in the prevalence of media portrayals of reefal collapses (Pandolfi et al. 2003, Côté et al. 2005, Bruno and Selig 2007, Ateweberhan et al. 2011, De'ath et al. 2012). This decline has driven the need for regional status reports to inform on the future of these reefs.

Various methods and metrics targeting different reef aspects have been used to monitor reef health and achieve specific monitoring aims (English et al. 1997, Hill and Wilkinson 2004). Benthic cover remains the most common metric in reef monitoring (e.g., coral and macroalgal cover) and is the quickest and easiest means to determine general reef health (Lam et al. 2017). Other common measures include coral species richness, abundance, and diversity; however, the latter can be difficult to determine in the tropical Indo-Pacific due to high species richness, some of which are difficult to differentiate underwater (Veron et al. 2015, Huang et al. 2015). The latest monitoring techniques consider resilience-based management as a whole-of-ecosystem approach to determine reef health. It includes various metrics such as habitat complexity, ecosystem functioning, and evolutionary divergence (Huang and Roy 2015, Darling et al. 2017, Denis et al. 2017, Williams et al. 2019, Woodhead et al. 2019; see also Lam et al. 2017 for a review). Studies have even begun looking at the correlation between the different metrics as a means to find a suitable suite of metrics for a comprehensive and quantitative description of reef health (Darling et al. 2017, Wong et al. 2018a, Donovan et al. 2018).

Aside from these reef-based metrics, other environmental parameters have been measured to determine environmental correlations of reef health for monitoring purposes. These include sea surface temperature (SST), salinity, turbidity, and oceanic productivity, measured both in situ and through satellite imagery (Yeager et al. 2017, Obura et al. 2019). Many of these have been known to impact corals directly or indirectly by affecting light availability, growth rates, and niche availability (Tanzil et al. 2013, Chow et al. 2019, Laverick et al. 2020). These have been investigated for the potential to determine the cause and effect of reef health decline and degradation in various regions. Defining the causal pathways of many of these stressors and their direct impact on corals in the region would have significant implications for management and conservation into the future.

The East Asian region of the Global Coral Reef Monitoring Network (GCRMN) comprises both coastal Northeast Asia (China, Hong Kong, Taiwan, Japan, South Korea) and Southeast Asia (Brunei, Cambodia, Indonesia, Malaysia, Myanmar, Philippines, Singapore, Thailand, Vietnam). The node has built upon past collaborative works with the ASEAN-Australia Living Coastal Resources project in the 1990s, which led to more qualitative reports (Kimura et al. 2014) and aimed to collate data for an East Asian regional reef analysis following the Caribbean report (Jackson et al. 2014), the Western Indian Ocean report (Obura et al. 2017), and the Pacific report (Moritz et al. 2018).

The East Asian data were provided by the various member countries and their contacts, comprising multiple academic, governmental, and non-governmental organizations. While the datasets provided are hardly exhaustive or comprehensive, they still represent the most complete, accurate, and long-term data presently available. Limitations of the data sources meant that species-level information was scarce due to the difficulty in identifying species and assimilating different data sources. Additionally, certain data were available but not within the public domain or in different languages that did not allow for ease of use (Nuñez and Amano 2021). Thus, only hard coral cover and macroalgal cover could be reliably compared across datasets.

A preliminary analysis was first conducted on a subset of the data (see Obura et al. 2019) before a second compilation of data was collected for further analysis. The data were updated, and the methods were improved upon following feedback, resulting in the final version of the analysis presented here. The current study puts together several datasets to identify the trends in the benthic cover of coral reefs in the East Asian region over time, including (i) changes in monitoring effort over time, (ii) the statuses and changes in coral and macroalgal cover over time, and (iii) links between benthic cover change and anthropogenic and environmental parameters.

Materials & Methods

DEFINITIONS

In this dataset and analysis, several descriptor terms are used in the following manner:

Transect – represents an individual replicate within a survey.

Survey – represents a collection of transects at one site at a fixed depth at one time point.

Depth – refers to a depth below sea level, measured at some point during the survey.

Site – represents an area where surveys are conducted, usually at the individual reef level, though larger reefs may have multiple sites.

Location – represents a collection of sites; this definition is largely arbitrary and furnished by the data providers, so the scale may differ between datasets.

Country – represents country or territory, political boundaries for ease of policy implementation.

Region – represents either the Northeast Asian (NEA) region, the Southeast Asian (SEA) region, or both together (East Asian – EA).

Hard coral – refers to scleractinian species that form a framework for other reef-dwelling organisms and includes the reef-forming non-scleractinians blue coral *Heliopora coerulea* and the fire coral *Millepora* spp.

Macroalgae – refers to fleshy algae species, including typical fleshy algae like *Sargassum* spp. and excludes calcareous coralline algae.

DATA COLLATION & CLEARING

Coral reef survey data obtained from the coordinators of the Global Coral Reef Monitoring Network East Asia (GCRMN EA) differed in methodology as well as the number of replicates and resurveys (Table 1). However, they were generally transect-based surveys and were relatively comparable for benthic coral cover (Leujak and Ormond 2007, Facon et al. 2016). Reef survey methods differed the most by the coordinators and regions than other factors; thus, it was ideal and necessary to include all data in the dataset to analyze a representative dataset. Several checks were conducted to ensure data accuracy, including verifying that benthos values and the total values were within range and summed correctly. As it was also possible to have full and no cover in some transects, values of 1 and 0 respectively, these values were still included in the dataset. Latitudes and longitudes were checked for each site, following which, sites were merged where possible across datasets and time points for a more consistent and longer-term inference. Inaccuracies were clarified with the various coordinators, and data with unresolved issues were discarded. Due to the coarse nature of GPS coordinates in the earliest surveys, many surveys were wrongly positioned on land. These were kept as sites with the coordinates updated if possible and upheld as valid if they could be resolved by the coordinators.

TABLE 1: *Collated coral reef benthic surveys with source information, methods, and description by country and dataset across the GCRMN East Asia.*

COUNTRY	SURVEY METHOD	LENGTH OF SURVEYS (CM)	NO. OF REPLICATES	SOURCE
Brunei	LIT	3,000	5-6	David Lane, Universiti Brunei Darussalam
Brunei	Reef Check PIT	2,000	4	Reef Check Brunei
Cambodia	PIT	2,000	4	Flora Fauna International Cambodia
China	-	-	-	None*
Hong Kong	LIT	3,000	3	Reef Check Hong Kong
Indonesia	PIT, LIT, Photo Transect	5,000	1-5	Various NGOs collated through Coral Triangle Center
Indonesia	PIT	5,000	3	Wildlife Conservation Society (USAID Sea Project)
Japan	Timed Swims	5,000	1	Japan Ministry Reports
Malaysia	Reef Check PIT	2,000	5	Reef Check Malaysia
Myanmar	PIT	2,000	5	FFI Myanmar
Philippines	PIT	2,000	5	Coral Cay Conservation
Philippines	LIT, Photo Transect	2,000	1-3	Various NGOs
Singapore	LIT	2,000	5	Reef Ecology Study Team, Reef Ecology Lab
South Korea	LIT	1,000	1	National Marine Ecosystem Survey
Taiwan	Reef Check PIT	2,000	4	Taiwan Environmental Information Association
Taiwan	Reef Check PIT	5,000	1	Taiwanese Coral Reef Society
Thailand	LIT	1,500	3-4	Marine Biodiversity Research Group, Ramkhamhaeng University
Vietnam	Reef Check PIT	2,000	4	Institute of Oceanography, Vietnam
Indonesia, Malaysia, Philippines, Singapore, Thailand	LIT	10,000	1	ASEAN-Australia Living Coastal Resources

* Data from China is not yet made available for analysis.

Coral reef benthic cover values were proportions and treated as such in the analysis; these were transformed according to Smithson and Verkuilen (2006) to remove values outside the end of the beta scale (i.e., 0 and 1), and the other values were scaled accordingly to allow for beta family analyses. Depths of coral surveys played an important role in explaining coral cover due to light availability but differed strongly across regions and countries and were not always measured. Transects were treated as replicates for the same site.

Sites with missing data were also managed; accordingly, sites with missing coral benthic cover were removed, while missing latitudes or longitudes were infilled if site names matched; else, they were also removed. Sites with missing depths were still included in the analysis but represented as not applicable (NA). Sites with missing years were also removed, but sites with missing months were treated as observed in December for further analysis. The final dataset was split into two for the trend analyses; one dataset containing all valid data and the second dataset containing only sites with at least three repeated surveys over different times. A third dataset was generated from the first dataset for environmental analyses, with all missing depth data removed.

DATA EXPLORATION

Several exploratory plots were constructed to visualize the large dataset. Histograms were used to visualize the datasets by countries and check for data patterns and potential non-normality, skew, or heterogeneity. Coral and macroalgal cover were also averaged yearly with standard deviations and plotted against time. Coral and macroalgal cover were then plotted against time, with LOESS (locally estimated scatterplot smoothing) smoothers plotted for the overall dataset, regions, and countries. Contour and density plots were also generated to observe potential patterns in the dataset across specific environmental and location variables.

SOCIO-ECONOMIC & ENVIRONMENTAL DATA

To better understand potential drivers of the change in the benthic cover through time, a series of environmental covariates were obtained from Yeager et al. (2017) using the latitude and longitude of survey sites, including net primary productivity (NPP), reef area, wave energy, human populations within 5 km and 100 km, and distance to the nearest market. As human population data was only available in 5-yearly intervals, data was modeled using both a linear model (LM) and a generalized additive model (GAM) with the R package 'mgcv' (v1.8-28; Pedersen et al. 2019). These were then combined, with the GAM values preferred over LM values as population change might not be linear. Still, in some instances where large declines in population happened abruptly between two data points, likely due to methodology changes (see Yeager et al. 2017 for details), the LM approximation was used instead. Values less than zero were replaced with zero. Distance to market was also extracted from Yeager et al. (2017), and this represented distance from economic centers of development that, aside from population numbers, have potentially greater effects on reefs.

Apart from socio-economic variables, we also extracted means and standard deviations of physical variables such as wave energy and net primary productivity (NPP). Reef area calculated within 5 km and 100 km of individual reef sites were also assessed by Yeager et al. (2017). These measures represent an easily accessible proxy for connectivity, which is otherwise difficult to assess on a large scale. Larger nearby reef areas represent potential source sites for reef repopulation and genetic diversity that can improve resilience of reefs (Schill et al 2015).

Temperature variables directly impact coral health and consequently have long-term implications for reef ecosystem health. Therefore, we also assessed sea surface temperature (SST) variations using the monthly composite temperature data extracted from the NOAA Coral Reef Watch (NOAA Coral Reef Watch 2019). Temperature variables included means, maxima, minima, and standard deviations of monthly composite degree heating weeks (DHW), bleaching alert area level (BAA), sea surface temperature (SST), and SST anomalies (SSTA). These represent different temperature variables that affect reefs differently, with DHW and BAA indicating bleaching potential, and SST and SSTA affecting reef growth and bleaching. Aside from monthly temperature variables, we also calculated annual composites from the monthly variables from one year before the surveys, including the sums of the DHW and BAA maxima, minima, means, and standard deviations of the various temperature components. A complete list of all model components is listed in Table 2.

TABLE 2: *Variable components, names, and presence in various models for model averaging and their predicted effects on the coral cover. Model A consists of only monthly variables while Model B includes monthly and annual estimations.*

COMPONENTS	VARIABLE	MODEL	PREDICTED EFFECT	EXPLANATION
Year	Year	Both	±	—
Country	Country	Both	±	—
Depth	Depth	Both	-	Light attenuates with depth, a similar effect for macroalgae
Estimated population within 5 km	pop_est_5km	A	-	Population density increases anthropogenic pressures
Estimated population within 100 km	pop_est_100km	Both	-	Population density increases anthropogenic pressures
Estimated reef area within 5 km	reef_area_5km	A	+	Reef area attenuates localized impacts
Estimated reef area within 100 km	reef_area_100km	Both	+	Reef area attenuates localized impacts
Intra-annual Mean NPP	npp_mean	Both	+	NPP increases with increased photosynthesis
Intra-annual Minimum NPP	npp_min	Both	+	NPP increases with increased photosynthesis
Intra-annual Maximum NPP	npp_max	Both	-	Much higher NPP may indicate algae blooms
Intra-annual NPP SD	npp_sd	Both	-	A large variation in NPP may indicate algae blooms
Inter-annual NPP SD	npp_interann_sd	A	-	A large variation in NPP may indicate algae blooms
Intra-annual mean wave energy	wave_mean	Both	-	High wave energy can damage reefs
Intra-annual wave energy SD	wave_sd	Both	-	High variation in wave energy can damage reefs
Inter-annual wave energy SD	wave_interann_sd	A	-	High variation in wave energy can damage reefs
Distance to nearest provincial capitals	dist_market	Both	+	Proximity to cities increases anthropogenic impacts
Monthly maximum bleaching alert levels	baa_max	Both	-	Bleaching alert levels highlight potential for bleaching

COMPONENTS	VARIABLE	MODEL	PREDICTED EFFECT	EXPLANATION
Monthly maximum Degree Heating Weeks	dhw_max	Both	-	DHWs highlight direct bleaching threat
Monthly maximum SST	sst_max	Both	-	SST increases highlight temperature threats
Monthly mean SST	sst_mean	Both	-	SST increases highlight temperature threats
Monthly minimum SST	sst_min	Both	+	Min SST increases the potential for greater reef growth
Monthly maximum SSTA	ssta_max	Both	-	SSTA (SST anomalies) increases highlight greater temperature variability
Monthly mean SSTA	ssta_mean	Both	-	SSTA increases highlight greater temperature variability
Monthly minimum SSTA	ssta_min	Both	-	SSTA increases highlight greater temperature variability
Annual sum of BAA maxima	sum_baa_max	B	-	Bleaching alert levels highlight potential for bleaching
Annual sum of DHW maxima	sum_dhw_max	B	-	DHWs highlight direct bleaching threat
Annual maximum of SST maxima	max_sst_max	B	-	SST increases highlight temperature threats
Annual mean of SST maxima	mean_sst_max	B	-	SST increases highlight temperature threats
Annual mean of SST means	mean_sst_mean	B	-	SST increases highlight temperature threats
Annual minimum of SST minima	min_sst_min	B	+	Min SST increases potentially allow for greater reef growth
Annual mean of SST minima	mean_sst_min	B	+	Min SST increases potentially allow for greater reef growth
Annual mean of SSTA maxima	mean_ssta_max	B	-	SSTA increases highlight greater temperature variability
Annual mean of SSTA means	mean_ssta_mean	B	-	SSTA increases highlight greater temperature variability
Annual mean of SSTA minima	mean_ssta_min	B	-	SSTA increases highlight greater temperature variability

TREND ANALYSES

A Bayesian hierarchical generalized additive model (HGAM) was used to identify trends in coral reef and macroalgal cover over time. The model applies smoothers to generate the trends across year and depth with a country effect and a random effect of location and site (Bürkner 2018, Pedersen et al. 2019). Weakly informative priors were used to constrain the coral cover values within the beta distribution. Models were visually checked for convergence with R-hat values ensured below 1.02. The best model was selected using Leave-One-Out (LOO) cross-validation and the Leave-One-Out Information Criterion (LOOIC), alongside Widely Applicable Information Criterion (WAIC) and model R^2, ensuring selection of the most parsimonious and effective model. All models and analyses were run using the R package 'brms' (v2.10.0; Bürkner 2017) and compared using package 'performance' (v0.4.4; Lüdecke et al. 2020).

MODEL EFFECTS & TREND VALIDATION

The most dissimilar survey methods were removed in phases to check if methodology differences affected the survey results, starting with removing Timed Swims and then removing either Line Intercept Transects or Point Intercept Transects. The model results were also compared to the original model to see if they remained similar. Likewise, method and transect length were included as random effects to determine if these inputs improved the model. Comparisons were made with the R package 'performance' (v0.4.4; Lüdecke et al. 2020), and effects plots were visually inspected.

A series of sensitivity analyses were conducted to determine if the model effects and effect sizes were precise and accurate. First, the survey dataset was subdivided to only sites with observations across time spanning at least 5, 10, 15, and 20 years to identify if long-term coral cover change remained consistent in separate analyses, despite the different monitoring period lengths. Second, the dataset was split into decades before and after major bleaching events (1998, 2008) to determine if new sites were selected based on a higher or lower cover than the mean. This split was done based on the intervals between the first and final survey dates and the interval between the last survey dates and the present day and determining if new survey sites were disproportionately higher or lower in reef cover, affecting trends across time. We also did the same analysis with the datasets selected after the first bleaching event (post-1998) and before the second bleaching event (pre-2008) to check if trends remained the same. Lastly, the rate of coral cover change was checked against the time of the survey to determine if sites were removed due to loss of coral cover and to confirm if reefs were no longer surveyed because no viable reef was present anymore. Such removal can potentially alter coral cover change over time and artificially inflate coral cover trends through time. Together, these ensure that the trends we have reported are valid.

SOCIO-ECONOMIC & ENVIRONMENTAL DATA CORRELATIONS

To better understand how the benthic variations have changed across time, we also ran correlations with the socio-economic and environmental data generated against the benthic cover. We first scaled the socio-economic and environmental variables and centered them before applying an information theory approach following Grueber et al. (2011). We set a maximum limit of 6 variables to include through the 'dredge' function in package 'MuMIn' (v1.43.17; Bartoń 2011) and included the adjusted R^2 of the models for exploratory analyses. As there were also many approaches to perform model averaging, we decided to select models using ΔAICc for the different models. As expected, the first few models had much higher weights than the rest. However, no specific model had a weight more than 0.9, indicating that model averaging would preserve many variables and effects. The top few models selected had similar variables, though additional variables were included at greater ΔAICc units, which have much smaller weights. We thus followed Grueber et al. (2011) and included all models within 6 ΔAICc units from the main model (Richards 2008) to obtain the conditional average model. While other model averaging measures exist, such as the use of ΔAICc of 2 or 10 or a 95% model weight, in the current context, all the variables appeared to limit or expand the model selection process considerably. Therefore, ΔAICc of 6 was used as the best compromise between complexity and explanatory power.

VERSION CONTROL

All analyses were conducted, and all plots were drawn with package 'ggplot2' (v3.2.1; Wickham 2016) in R (v3.6.1; R Core Team 2019) and R Studio (v1.2.5033; RStudio Team 2020) on a Windows 10 OS, with the seed set to 13.

Results

DATA EXPLORATION

In total, 24,365 transects were conducted over 1,972 sites in 219 locations from 13 countries and regions spanning 37 years from 1983 to 2019. For the long-term data, 21,107 transects had 3 observations spanning at least 5 years. However, 8,668 surveys had no macroalgal data, with most of these data coming from Japan, which were then excluded from the macroalgal analysis.

Sites had varying hard coral cover (Mean ± SD: 0.333 ± 0.232, Median: 0.300) and macroalgal cover (Mean ± SD: 0.057 ± 0.110, Median: 0.000), and ranged from 0–1 and 0–0.974 respectively. Most of the surveys were relatively shallow, with three-quarters of the dataset conducted between 0 m to 7 m. Japan has the most complete long-term dataset from 1983 to 2019, followed by Singapore from 1986 to 2019 with breaks in between, and followed by the other ASEAN-Australia Long-Term Monitoring Survey countries (Malaysia, Philippines, Indonesia, Thailand) with surveys in the late 1980s followed by more recent surveys.

Coral cover was generally unimodal and showed some positive skew in most datasets, except Myanmar (Figure 1). Macroalgal cover showed an even stronger positive skew, with a distinct peak at the means near 0 as many surveys had little to no macroalgae (Figure 2). Depths of the different surveys also varied greatly across countries, with the majority spread around shallow depths. However, certain countries had distinct depths where surveys were conducted, such as in Japan (6 m), Singapore (0 m, 4 m, 6 m, 10 m), Hong Kong (0 m, 3 m), Taiwan (5 m, 10 m) and South Korea (5 m, 10 m, 15 m) (Figure 3).

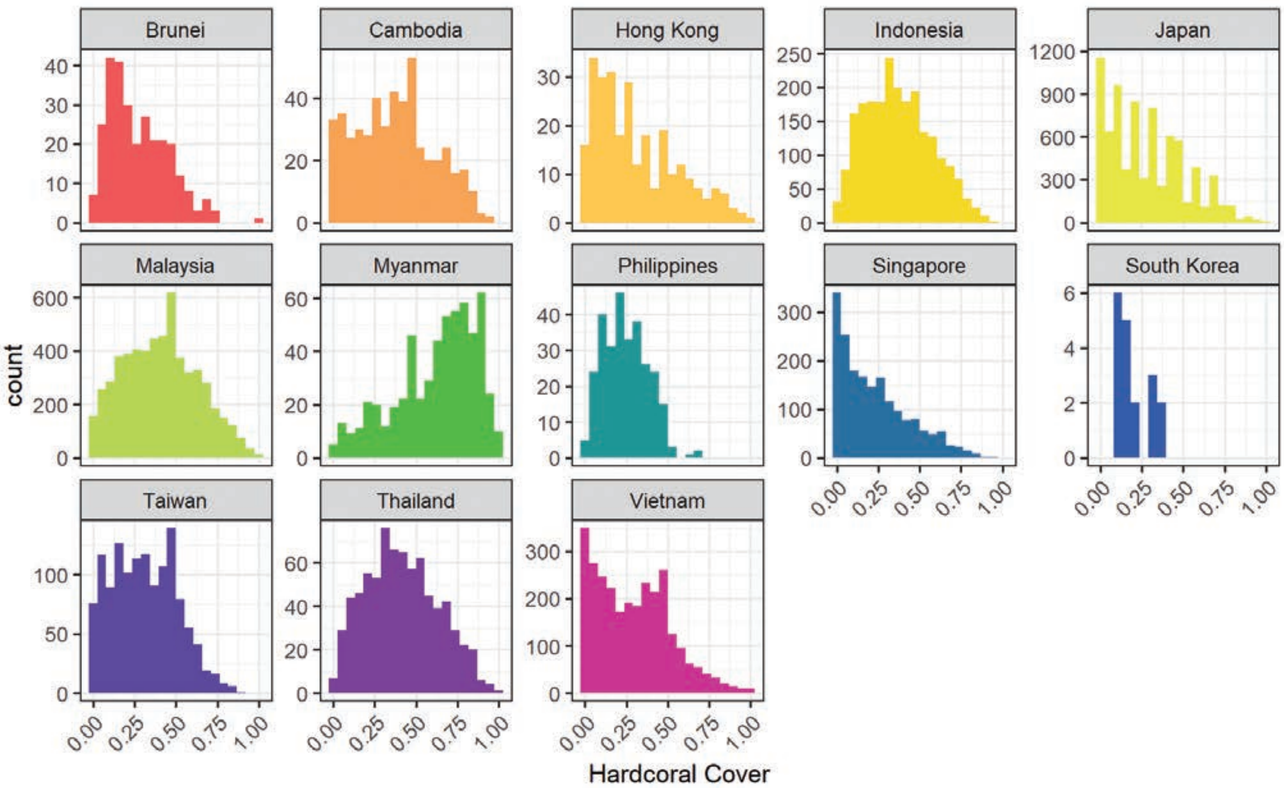

FIGURE 1: *Histogram of coral cover across countries, showing general unimodal and positively skewed data (Note the varying y-axis values).*

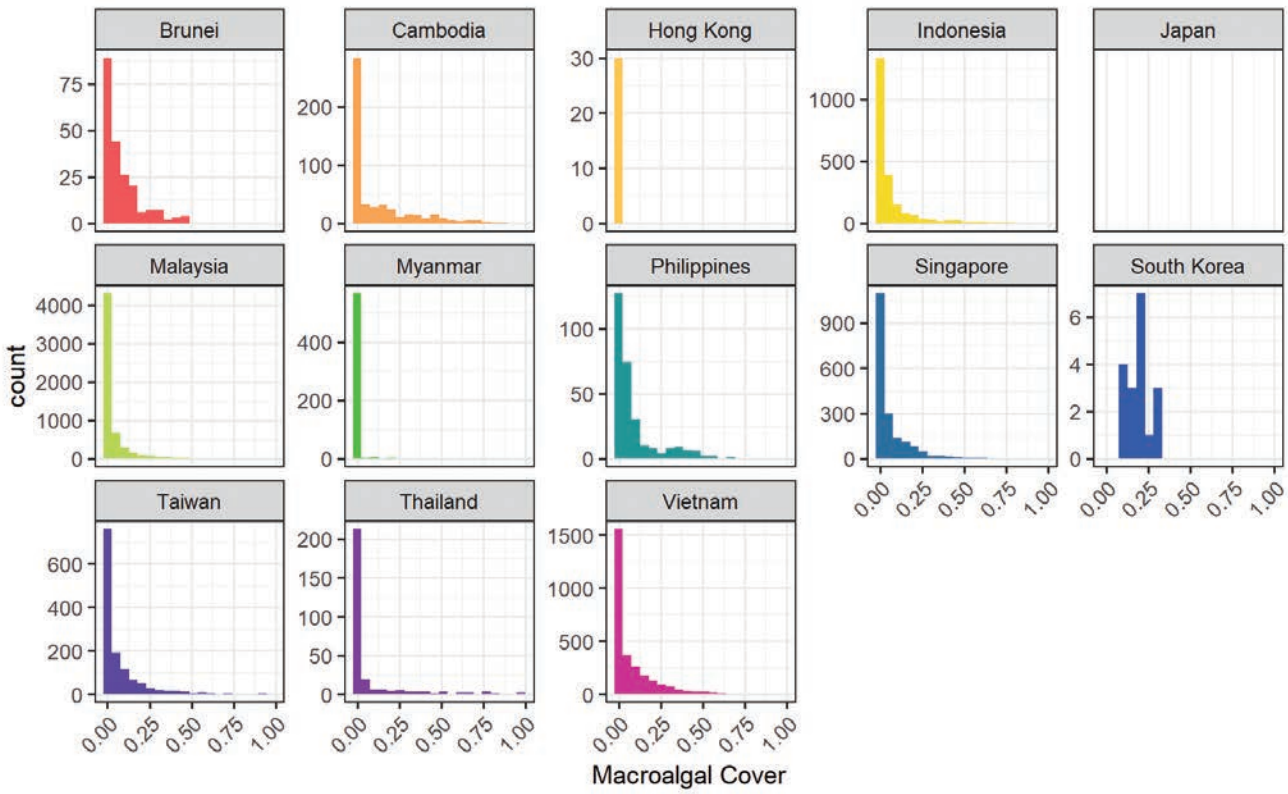

FIGURE 2: *Histogram of macroalgal cover across countries, showing strong positively skewed data, and a high proportion of low macroalgal cover (Note the varying y-axis values).*

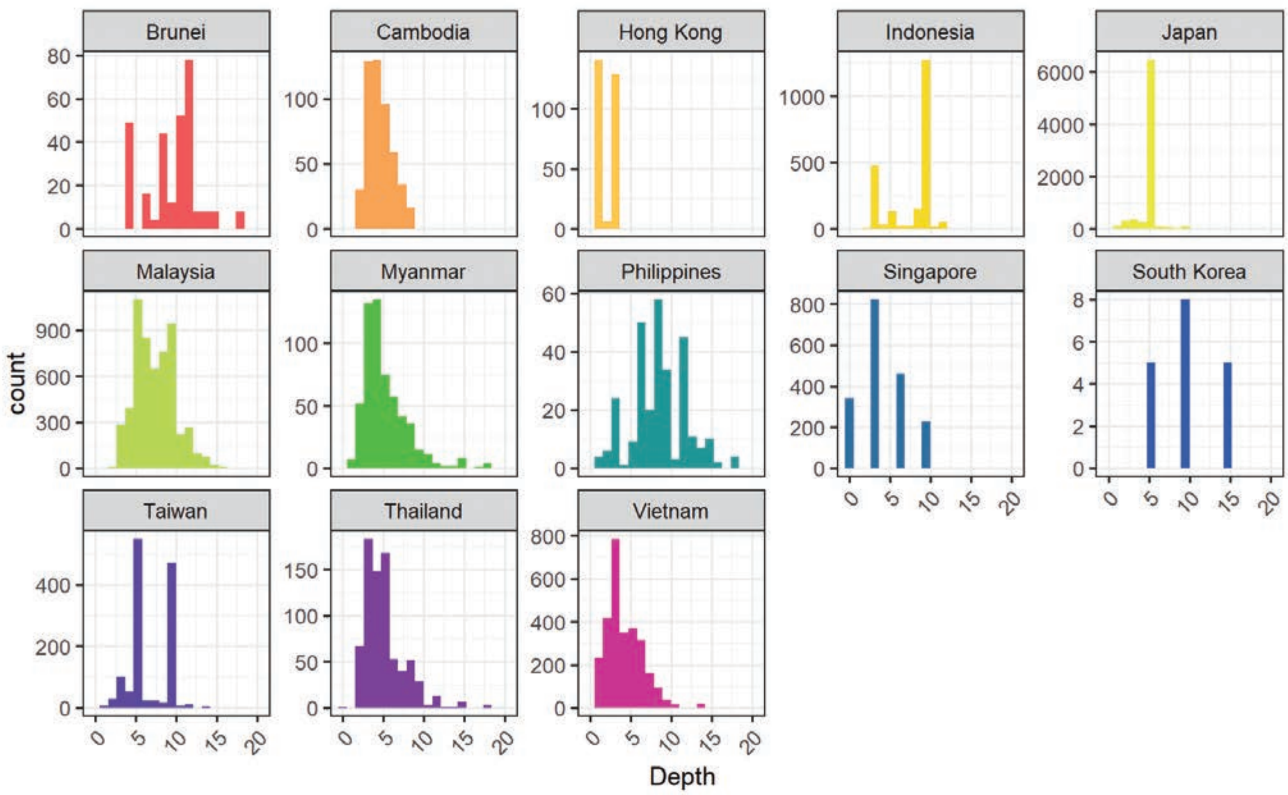

FIGURE 3: *Histogram of survey depths across countries, showing a spread of surveys across depths and peaks at different selected depths for multiple countries (Note the varying y-axis values).*

There were considerable increases in surveys and transects over time, especially after the 2000s and then again after the 2010s, in many of the countries represented. The increase in survey data is also non-linear (Figure 4), with more countries being represented across the years and the greatest numbers in 2017 coinciding with the data collection in 2018–2019. The survey numbers are also disproportionate (Figure 4), as countries with the highest reef cover are not necessarily those with the most numerous surveys. Additionally, the data also demonstrate that not all increases are constant; countries may have more surveys in certain years due to specific research projects, increased monitoring due to bleaching events, or greater research funding.

Mean coral and macroalgal cover patterns through time show similar trends. Coral cover varied between 0.25 and 0.37 through time (Figure 5). Macroalgal cover trended around 0.12 to 0.05 through time, with a much smaller change over time (Figure 5). Some of the peaks in coral cover between 1994–1996 are potentially the result of less available data. Country-level LOESS smoothers differed from the overall smoother, with many countries showing trends with wide standard deviations. Most of these show a trend towards a coral cover of around 0.25, with many countries showing declines at various stages followed by some recovery near 2018 (Figure 6). For the macroalgal cover, LOESS estimations were unavailable, and thus, GAM smoothers were utilized instead. These generally showed slight changes in macroalgal cover on much smaller scales compared to coral cover, highlighting the low macroalgal cover in most countries (Figure 6).

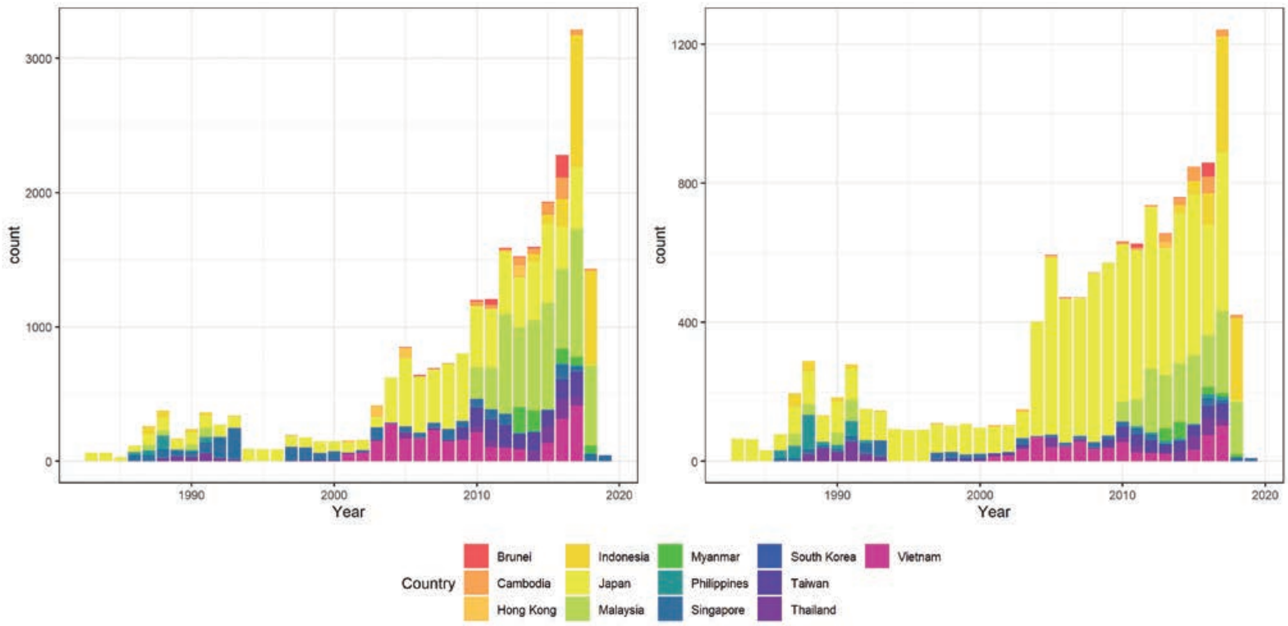

FIGURE 4: *Stacked bar plots of unique surveys (left) and transects (right) by year and colored by countries, showing large increases of both surveys and transects after the year 2000 across the different countries and regions.*

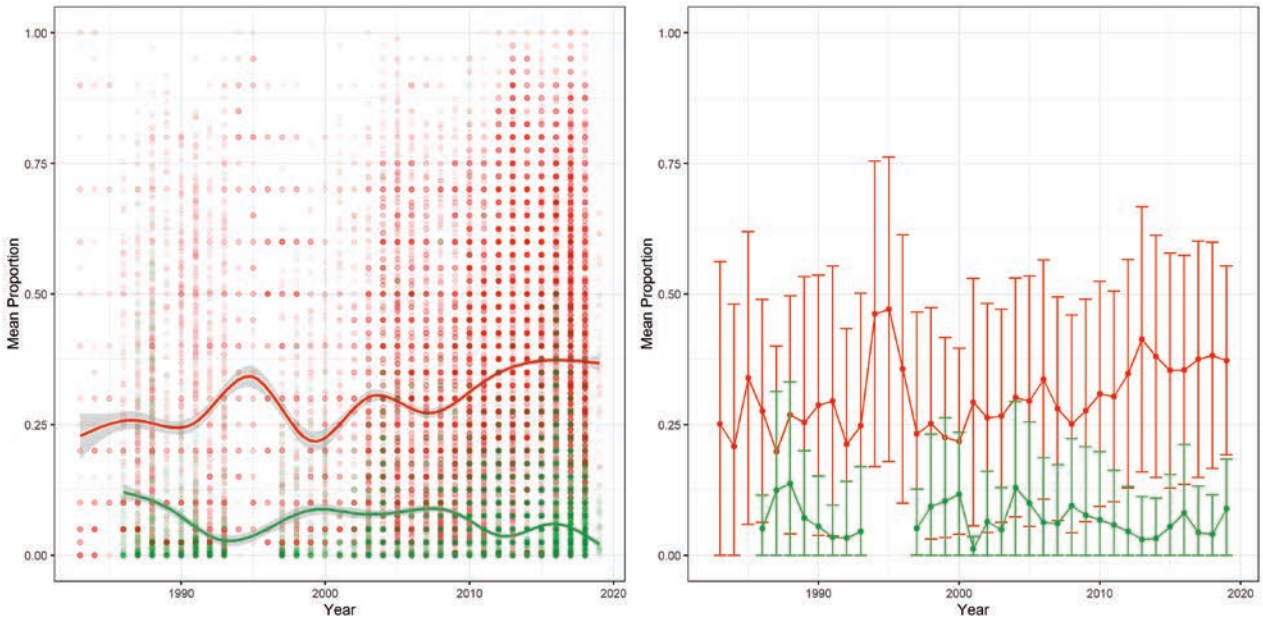

FIGURE 5: *Coral cover (red) and macroalgal cover (green) across years, with individual surveys as points and LOESS smoothers (left) and mean annual proportions and standard deviations (right) generally stable cover across time.*

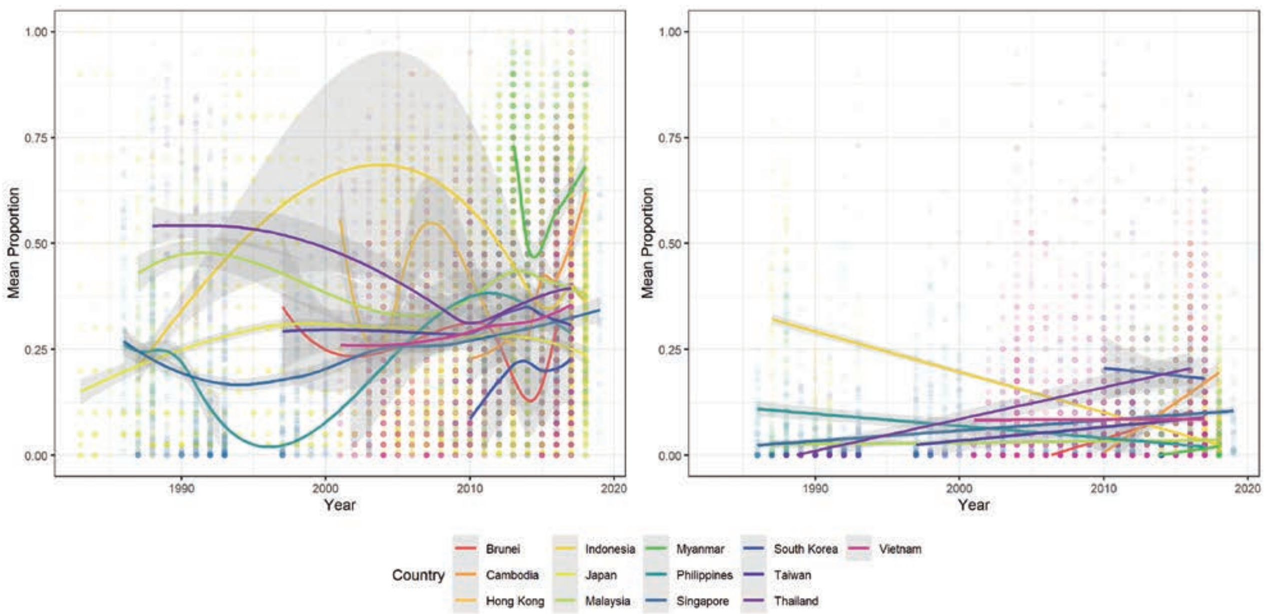

FIGURE 6: *Coral cover (left) and macroalgal cover (right) across years, with individual transect survey points and country-based LOESS smoothers for coral cover, and GAM smoothers for macroalgal cover.*

MODEL RESULTS – CORAL COVER TRENDS

A few separate Bayesian hierarchical generalized additive models (HGAMs) were generated for the final analysis, with the results interpreted together for a general trend. The first or country model contains a general global model smoother and a country-level smoother to generate a representative global trend and individual country trends. The second or regional model uses region instead of country as an explanatory variable. The best model determined by the LOOIC and WAIC is the first model, with both global smoothers and country-level smoothers (Table 3). However, both of these models' explanatory powers are relatively close, with the country model having a marginal R^2 around 0.16 and the regional model having a slightly lower R^2 of 0.14 (Table 3). Thus, these are considered in tandem.

TABLE 3: *Trend analyses models and model comparison variables for coral cover.*

MODEL	CODE	LOOIC ± SE	WAIC	R^2 / MARGINAL R^2
Country	Cover ~ s(Year, by=Country) + s(Depth) + Country + (1|Location) + (1|Site)	-28267.16 ± 422.82	-28315.25	0.53/0.16
Regional	Cover ~ s(Year, by=Region) + s(Depth) + Region + (1|Location) + (1|Site)	-28084.35 ± 420.46	-28143.26	0.53/0.14

Data before the 1990s are limited, resulting in large confidence intervals and great uncertainty around the actual trends in the country model. Variation decreases with time as more data are available for the trend analysis, resulting in uncertainty reduction. While there may be some decline based on the country model, the reduction before the 2000s is affected by large uncertainties (Figure 7). There is no decline detected with a relatively stable increasing coral cover trend towards 0.22. The regional model shows a lack of decline, though coral cover varies through time with both increases and decreases. These variations look to be in line with some of the major coral reef bleaching events in 1998 and 2006, decreasing following the bleaching events and then increasing following presumed recovery. In these models, the coral cover also tends towards around 0.22, with greater variation when split by regions.

Removing the country-level smoothers increases the annual variation, though the overall trend does still increase throughout the survey period.

In general, the coral cover also decreases with depth in all models, with the same trend reported. The cover declines from 0 m to 5 m before reaching a stable level from 5 m to 12 m, and then greatly decreasing past 12 m and abruptly increasing from 18 m, potentially due to a lack of data at depth (Figure 8). Most coral surveys across the various countries were conducted around shallow to mid-depths, though depth ranges varied considerably across countries and some sites. Thus, differing turbidity restricts the depth ranges of the various countries to different levels. The model results should also be considered a general pattern that does not apply to all countries. For example, countries like Hong Kong reported reefs surveyed only to 3 m, while Singapore's reefs were surveyed to a maximum depth of 10 m.

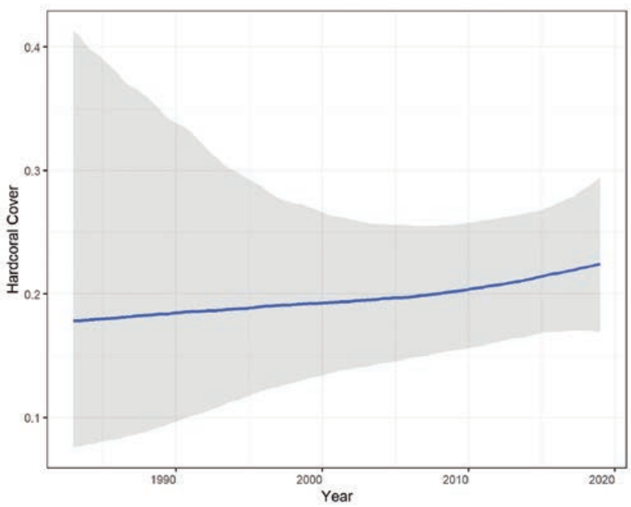

FIGURE 7: *Coral cover trends based on the Bayesian HGAMs across years, with the general smoothers from the country model showing high uncertainty before the 2000s. General coral cover remains relatively stable and increasing at around 0.2.*

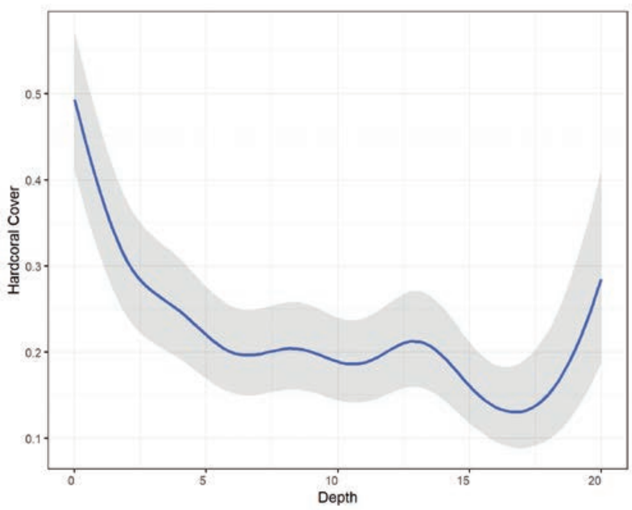

FIGURE 8: *Coral cover trends based on the Bayesian HGAMs across depths, with very similar smoothers for depth produced between models. Both models show highly similar patterns with an initial decline, followed by relatively stable cover across most of the mid-depths and then a decrease and increase past 12 m.*

The coral cover also shows large differences between countries with very different means and confidence intervals (Figure 9). Japan and Singapore have the lowest coral cover amongst surveyed countries, at around 0.18 with smaller intervals. Countries with higher coral cover from 0.2 to 0.3, like Brunei, Cambodia, Hong Kong, Indonesia, and the Philippines, also have larger coral cover intervals. Other countries with higher coral cover from 0.2 to 0.3 have smaller intervals, such as Malaysia, Taiwan, Thailand, and Vietnam. Myanmar shows up as a clear outlier with much higher coral cover and larger intervals as well. The regional model highlights a more distinct difference between the two regions than the country model (Figure 10). Southeast Asia (SEA) shows a much higher coral cover than Northeast Asia (NEA), with 0.34 compared to 0.19, though the confidence interval appears relatively similar across regions.

Country-level smoothers for coral cover show similar patterns to the global smoother (Figure 11). In general, most countries show some decline from the 1980s to 2000s from between 0.2 to 0.4, though the confidence interval for most countries is large before 2000, followed by some variation before an uptick near 2019. Some countries such as Singapore and Taiwan, also show little variation. The larger

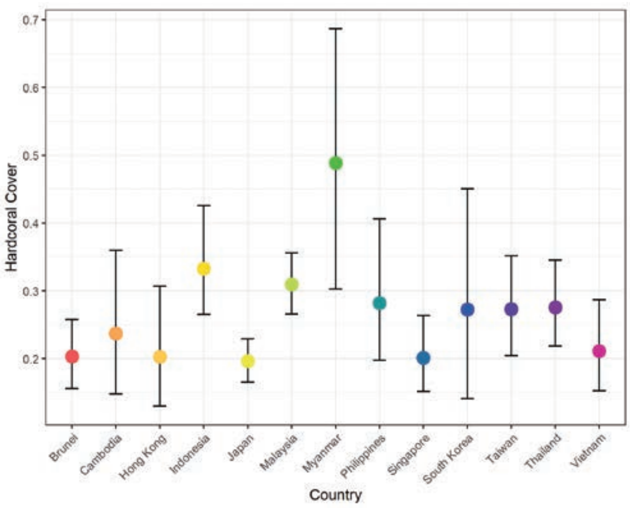

FIGURE 9: *Country-level variation in coral cover based on the Bayesian HGAMs, showing different means and confidence intervals among different countries.*

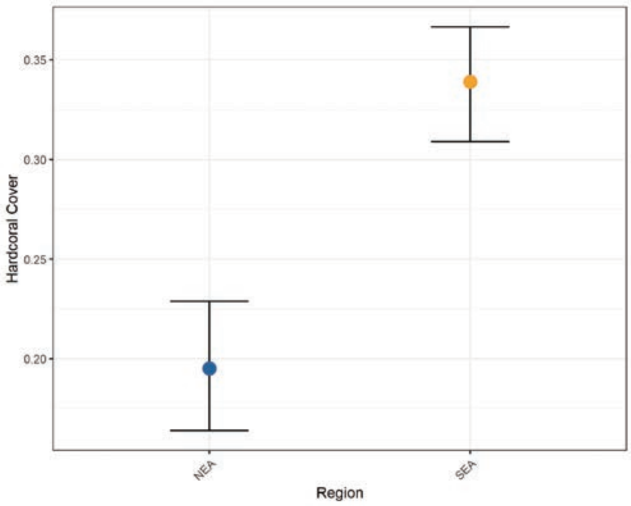

FIGURE 10: *Regional level variation in coral cover based on the Bayesian HGAMs, showing much lower mean coral cover in Northeast Asia (NEA) compared to Southeast Asia (SEA) though with similar levels of variation.*

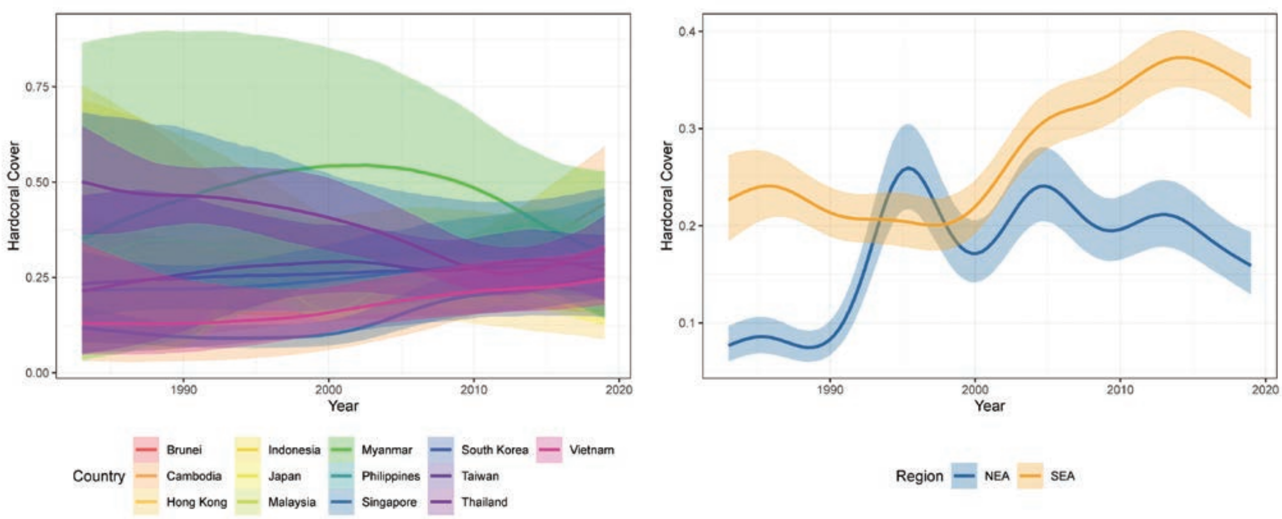

FIGURE 11: *Country-level smoothers (left) and region-level smoothers (right) in coral cover based on the Bayesian HGAMs in the country and regional models, respectively, showing different trends and cover variations between countries and regions across different time points.*

confidence intervals also highlight the likely lack of data in the years before the 2000s and the late 2010s, with some countries showing even longer periods with uncertainties. Region-level smoothers also show differences, with SEA having much higher cover than NEA. Increases in region-level cover are also evident for both regions at different time frames—2000 to 2015 in SEA and across multiple periods in NEA such as 1990 to 1995 and from 2000 to 2005.

MODEL RESULTS – MACROALGAL COVER TRENDS

Macroalgal cover patterns are less clear. In general, most surveys do not find much macroalgae, with many surveys recording no macroalgae at all. Similarly, multiple models were used in tandem to examine macroalgal trends, with the same model specifications – a country model with a global model smoother and country-level smoother, and a regional model with region instead of country. Compared to the coral cover models, the macroalgal models have much lower marginal R^2, with the highest R^2 of 0.07 in the country model and only 0.02 in the regional one (Table 4).

In general, macroalgal cover does not show much temporal variation, especially compared to coral cover change. Similarly, the country model highlights the large uncertainties before 1995 and followed by an increase towards the present (Figure 12). The macroalgal cover shows some general increase from the 2000s, though it stays below 0.1 through most of the survey times. Similarly, macroalgal decline across depths does not show much variation. Across depths, macroalgal cover decreases from 0.15 at 0 m, becoming stable past 5 m at around 0.06 (Figure 13).

TABLE 4: *Trend analyses models and model comparison variables for macroalgal cover.*

MODEL	CODE	LOOIC ± SE	WAIC	R^2 / MARGINAL R^2
Country	Cover ~ s(Year, by=Country) + s(Depth) + Country + (1|Location) + (1|Site)	-130270.16 ± 846.88	-130285.37	0.29/0.07
Regional	Cover ~ s(Year, by=Region) + s(Depth) + Region + (1|Location) + (1|Site)	-129853.69 ± 849.39	-129869.34	0.26/0.02

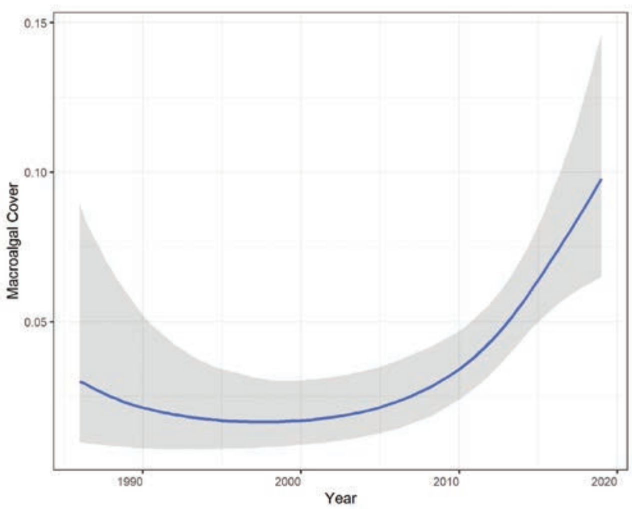

FIGURE 12: *Macroalgal cover trends based on the Bayesian HGAMs across years, with the general smoother showing higher uncertainty before the 2000s. Macroalgal cover appears to remain low through time below 0.1.*

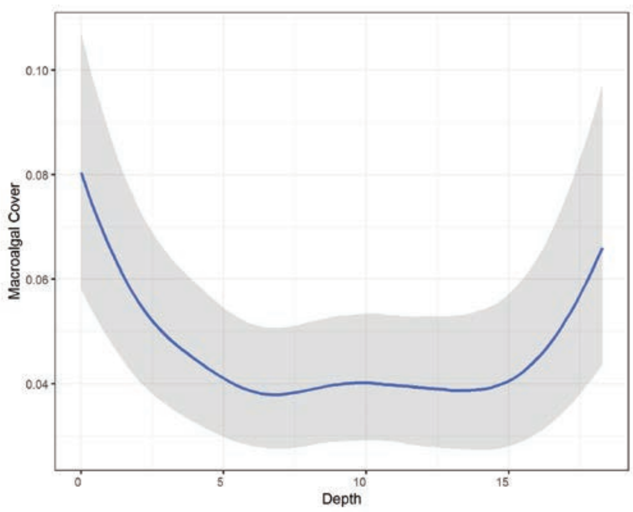

FIGURE 13: *Macroalgal cover trends based on the Bayesian HGAMs across depths, with very similar smoothers for depth produced between models. Both models show similar patterns with an initial decline followed by relatively stable macroalgal cover past 5 m.*

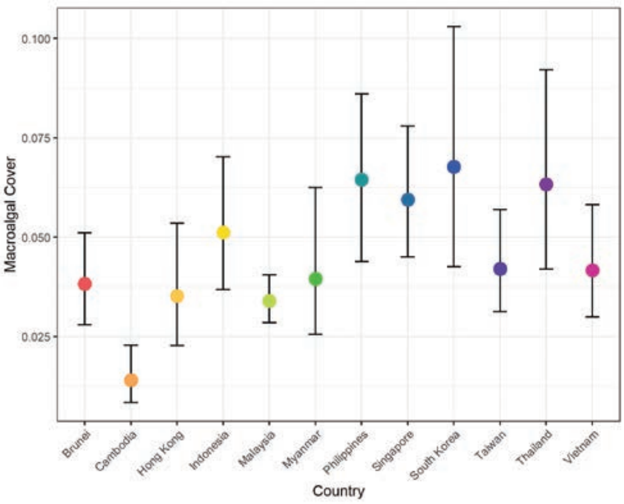

FIGURE 14: *Country-level variation in macroalgal cover based on the Bayesian HGAMs, with similar patterns between models, showing different means and confidence intervals among different countries, though all macroalgal cover remains low.*

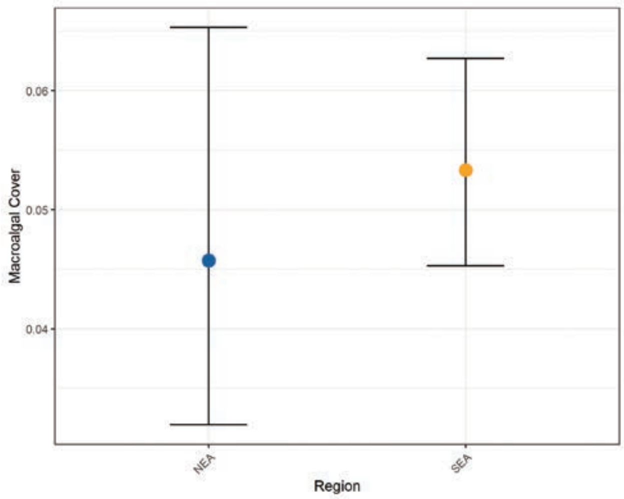

FIGURE 15: *Regional level variation in macroalgal cover based on the Bayesian HGAMs, showing similar means but much higher variation in macroalgal cover in Northeast Asia (NEA) compared to Southeast Asia (SEA).*

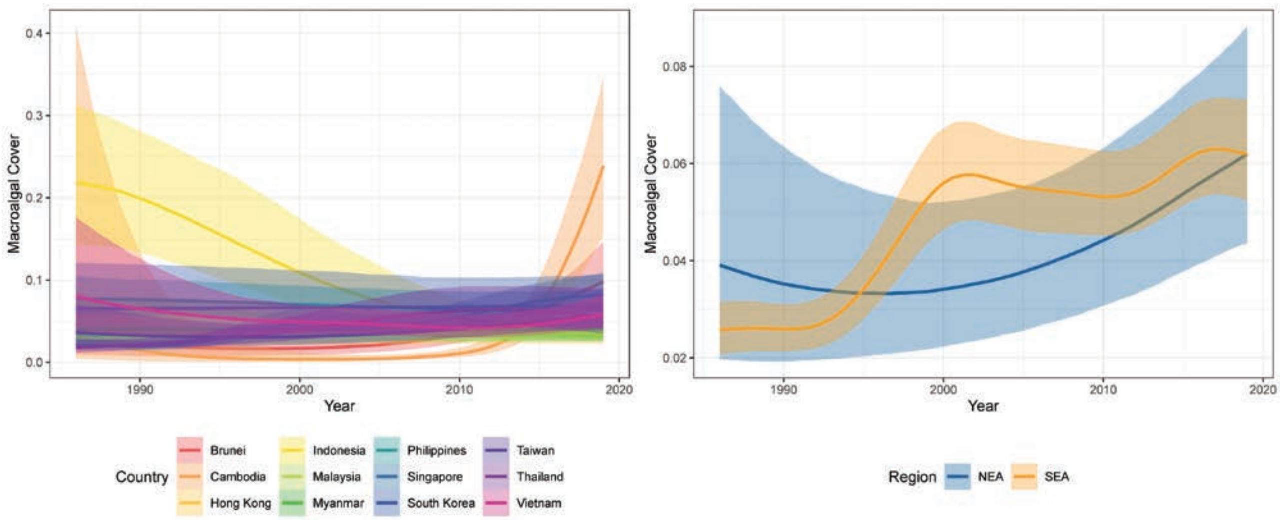

FIGURE 16: *Country-level smoothers (left) and region-level smoothers (right) in macroalgal cover based on the Bayesian HGAMs, showing different trends and cover variations between countries and regions across different time points.*

In general, most countries had low macroalgal cover with limited variation between them (Figure 14), ranging between 0.03 and 0.07. The regional model shows a slightly lower macroalgal cover in Northeast Asia (0.045) compared to Southeast Asia (0.055), though the variation in NEA is much larger than in SEA (Figure 15).

Country-level macroalgal smoothers from model 1 show generally similar patterns to the global smoother (Figure 16). Cambodia shows a very similar pattern to the smoother, while other countries show very stable macroalgal cover over time, with values tending towards 0.05. These trends are generally limited by the data available in more recent years. Regional trends show convergence over time towards 0.06, with NEA declining then increasing, while SEA showing increases till the 2000s before generally having more stable cover.

MODEL COMPARISONS & VERIFICATION

Various random effects were tested with the model before the final selection of both Site and Location. Sites represent smaller areas with likely very similar conditions, while Locations generally represent larger management areas within a country. While sites and locations differ in size across countries, smaller countries had smaller locations than other countries' representative sites; nonetheless, it provided *a priori* zonation by the data contributors. The addition of a random method effect does not improve model fit, which is likely due to the methods differing mostly by countries, as each country generally has one primary survey method used for all their surveys. Similarly, the addition of transect lengths into the random effects did not alter the model expectations.

SOCIO-ECONOMIC & ENVIRONMENTAL DATA

The averaged model for hard coral cover comprised two different models with country, depth, npp_max, npp_sd, and pop_est_100km being present in all models, while the other two variables, baa_max and reef_area_100km, each appeared in one of the component models (Table 5). All the variables except country were significant in the conditional averaged model, with all the variables predicting lower coral cover except for npp_max (Table 6). Maximum net primary productivity predicted increases in coral cover, while the standard deviation of NPP and reef area and population within 100 km predicted stronger declines in coral cover than bleaching alert levels (Table 6; Figure 17). Lastly, all country variables were not significant, except for Hong Kong.

TABLE 5: *Component models within the averaged model for centered coral cover and their respective ΔAICc and weights.*

COMPONENTS	DF	AICC	ΔAICC	WEIGHT
baa_max, country, depth, npp_max, npp_sd, pop_est_100km	21	-14124.17	0	0.87
country, depth, npp_max, npp_sd, pop_est_100km, reef_area_100km	21	-14120.35	3.81	0.13

TABLE 6: *Effect sizes and significance of variables within a conditional averaged model for centered coral cover.*

VARIABLE	EFFECT	ESTIMATE	ADJ SE	2.5% CI	97.5% CI	P-VALUE
Intercept		0.3654	0.0588	0.2501	0.4808	< 2e-16
baa_max	-	-0.0059	0.0014	-0.0086	-0.0032	1.87e-05
depth	-	-0.0085	0.0007	-0.0099	-0.0071	< 2e-16
npp_max	+	0.0852	0.0142	0.0574	0.1129	< 2e-16
npp_sd	-	-0.0799	0.0142	-0.1078	-0.0521	< 2e-16
pop_est_100km	-	-0.0392	0.0063	-0.0517	-0.0268	< 2e-16
reef_area_100km	-	-0.0316	0.0082	-0.0477	-0.0155	0.0001
Cambodia		0.0415	0.0876	-0.1302	0.2131	0.6358
Hong Kong		0.2670	0.0958	0.0793	0.4547	0.0053
Indonesia		0.0762	0.0613	-0.0439	0.1963	0.2135
Japan		-0.0551	0.0624	-0.1775	0.0673	0.3773
Malaysia		0.0667	0.0612	-0.0532	0.1866	0.2755
Myanmar		0.1939	0.1046	-0.0111	0.3990	0.0637
Philippines		-0.0831	0.0709	-0.2220	0.0559	0.2414
Singapore		0.1486	0.1317	-0.1095	0.4067	0.2592
South Korea		-0.0793	0.1357	-0.3453	0.1868	0.5593
Taiwan		0.0395	0.0702	-0.0981	0.1771	0.5735
Thailand		0.0935	0.0622	-0.0284	0.2154	0.1328
Vietnam		-0.0278	0.0724	-0.1698	0.1142	0.7013

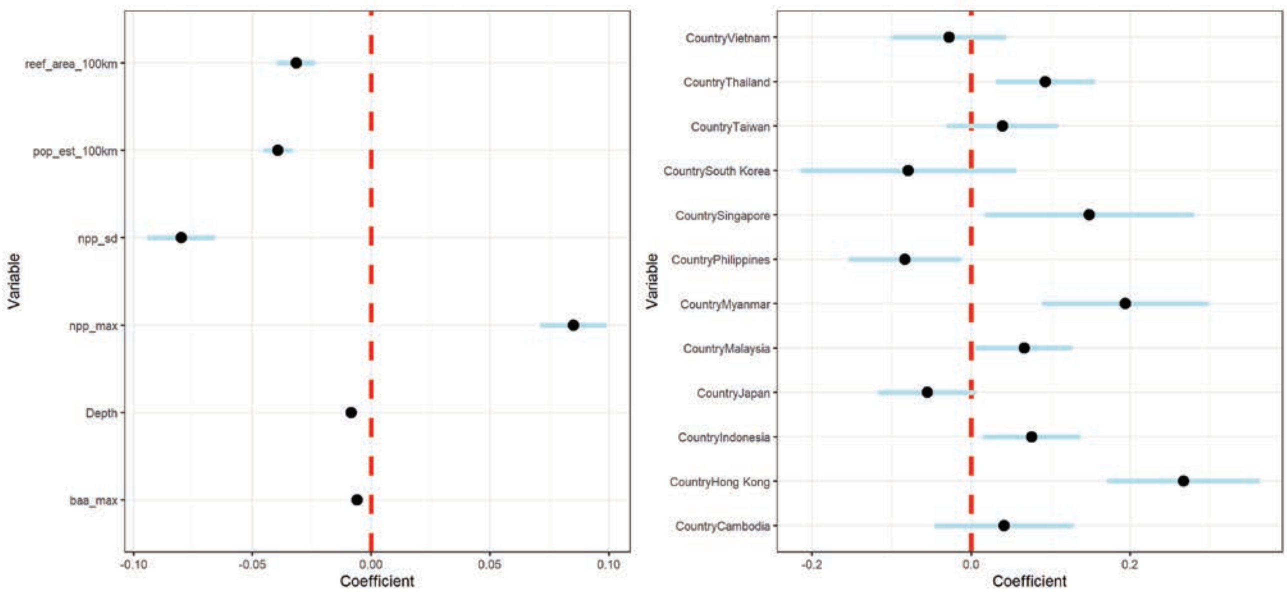

FIGURE 17: *Variable effects and coefficients (left) and country effects (right) in the averaged model for coral cover.*

For macroalgae, the averaged model comprised six different models with country, depth, and pop_est_100km present in all models. sst_max was present in five models, while sst_mean and dhw_max were present in four of the six models. The other variables, dhw_max, npp_max, npp_sd, sst_min, ssta_max, and year were present between one and three times (Table 7). Similarly, with the conditional averaged coral cover model, all the variables were significant except country, with all the variables predicting higher macroalgal cover except for depth, sst_max, and ssta_max (Table 8). Most predictors of macroalgal increase had small positive values aside from estimated population within 100 km and mean sea surface temperatures, while maximum SST predicted the strongest decline (Table 8; Figure 18). Finally, more country variables were significant here, including Hong Kong, Malaysia, and Singapore, compared to the coral model, but most country variables remained insignificant.

TABLE 7: *Component models within an averaged model for centered macroalgal cover and their respective ΔAICc and weights.*

COMPONENTS	DF	AICC	ΔAICC	WEIGHT
country, depth, dhw_max, pop_est_100km, sst_max, sst_mean	20	-26778.89	0	0.66
country, depth, npp_max, pop_est_100km, sst_max, sst_mean	20	-26775.79	3.10	0.14
country, depth, pop_est_100km, sst_max, sst_mean, year	20	-26774.05	4.84	0.06
country, depth, dhw_max , pop_est_100km, sst_max, sst_min	20	-26773.92	4.97	0.06
country, depth, dhw_max, npp_max, pop_est_100km, ssta_max	20	-26773.60	5.29	0.05
country, depth, npp_sd, pop_est_100km, sst_max, sst_mean	20	-26773.18	5.71	0.04

TABLE 8: *Effect sizes and significance of variables within a conditional averaged model for the centered macroalgal cover.*

VARIABLE	EFFECT	ESTIMATE	ADJ SE	2.5% CI	97.5% CI	P-VALUE
Intercept		0.0747	0.2631	-0.4409	0.5903	0.7765
depth	-	-0.0025	0.0004	-0.0033	-0.0018	< 2e-16
dhw_max	+	0.0028	0.0009	0.0010	0.0047	0.0028
pop_est_100km	+	0.0269	0.0048	0.0175	0.0363	< 2e-16
sst_max	-	-0.0158	0.0036	-0.0230	-0.0087	1.39e-05
sst_mean	+	0.0129	0.0035	0.0060	0.0198	0.0002
npp_max	+	0.0087	0.0036	0.0017	0.0157	0.0147
year	+	0.0005	0.0002	1.07e-05	0.0010	0.0451
sst_min	+	0.0065	0.0022	0.0021	0.0108	0.0036
ssta_max	-	-0.0042	0.0010	-0.0060	-0.0023	1.19e-05
npp_sd	+	0.0066	0.0037	-0.0007	0.0138	0.0763
Cambodia		-0.0220	0.0578	-0.1352	0.0912	0.7038
Hong Kong		-0.4380	0.0821	-0.5988	-0.2772	1.00e-07
Indonesia		-0.0450	0.0394	-0.1223	0.0322	0.2531
Malaysia		-0.0804	0.0388	-0.1565	-0.0043	0.0384
Myanmar		-0.1049	0.0716	-0.2452	0.0354	0.1428
Philippines		-0.0161	0.0428	-0.0999	0.0677	0.7058
Singapore		-0.1787	0.0701	-0.3162	-0.0412	0.0108
South Korea		0.0466	0.0721	-0.0947	0.1878	0.5181
Taiwan		-0.0655	0.0445	-0.1528	0.0217	0.1410
Thailand		-0.0623	0.0405	-0.1418	0.0172	0.1244
Vietnam		-0.0394	0.0467	-0.1309	0.0520	0.3981

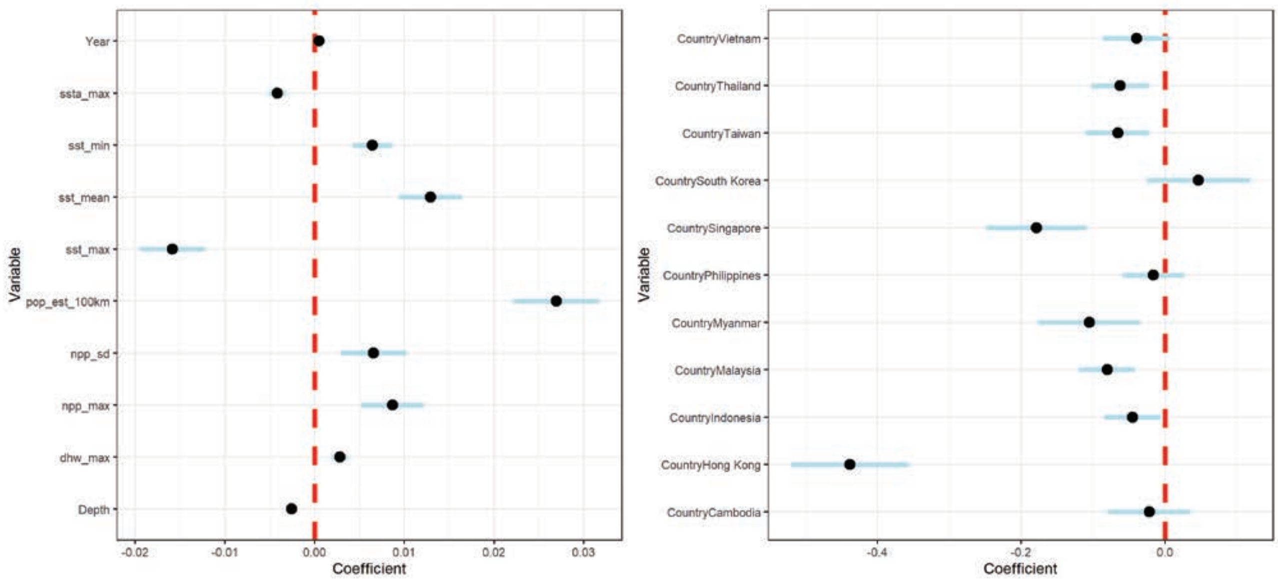

FIGURE 18: *Variable effects and coefficients (left) and country effects (right) in the averaged model for macroalgal cover.*

Discussion

This study represents the most comprehensive analysis of the reef trends in the region, improving upon previous studies which were based on the best available data at previous times (Bruno and Selig 2007, Heery et al. 2018). It also improves upon older, more qualitative summaries of the region's reefs, such as Kimura et al. (2014) and the culmination of work from as far back as Chou et al. (1994). While the use of multiple separate datasets is not ideal for long-term monitoring (Lindenmayer and Likens 2009), most of these datasets were still collected as parts of long-term monitoring programs in their individual regions. Considering the limited past regional monitoring data, this represents the best possible approach thus far for long-term analysis of reef trends till the present day. It is also clear that data availability remains an important issue to resolve so that long-term monitoring capabilities are maintained. There is a need for sufficient long-term datasets across multiple regions and countries to accurately ascertain the region's overall coral reef health. Therefore, it is insufficient to have data from just a few countries, especially if they are not proportionate to the region's reef areas.

The initial trend derived from the model highlights a distinct lack of reef decline compared to many other regions, such as the Caribbean and Australia (Gardner et al. 2005, De'ath et al. 2012), contrary to what other studies have found. This trend is likely the present case for the Northeast and Southeast Asian regions, owing to very different historical contexts and with a few possible rationales to explain this trend. First, while the coral cover trend did not decrease, they remained relatively low at around 0.2–0.3, compared to historical and anecdotal accounts of reef cover in the region. This trend may result from historical declines before recent surveys through an already shifted regional baseline (Knowlton and Jackson 2008), with the late 1980s representing an already altered state, which is the case for Caribbean reefs (Jackson et al. 2014). Similarly, the declines reported by Bruno and Selig (2007) are also limited by the lack of data before the 1980s, with the available early data showing much higher coral cover representing a more pristine state than today. This general average coral cover also highlights that specific reef areas have much higher coral cover, e.g., Indonesia, Malaysia, and the Philippines. Essentially, much more surveys need to be conducted around the reefs in the region, and more historical data need to be procured, even through images or descriptive notes (e.g., Fine et al. 2019), to provide a better understanding of the region's reefs in the past and ascertain the declines prior to the 1980s.

The trends also represent the change in the average reef through time and not the absolute change of all reefs through time. Our findings suggest that while the average monitored reef might not have declined drastically in cover, the overall health and state of the reefs in this region may have declined, especially so in regions like the many archipelagic countries and islands of both Northeast and Southeast Asia, where the reef areas are massive (Spalding and Grenfell 1997), potentially remote, and many unmonitored reefs may be destroyed or degraded before being noticed. Additionally, monitoring efforts are also concentrated on easily accessible or well-known localities, presenting problems in island archipelagos across the region with many hidden reefs inaccessible to researchers and NGOs.

The trends obtained here are also interpreted based on pre-defined reefs delimited by the regions, and they do not account for the total area or depth extents of the different reefs or the limitations of transect replicates in precisely evaluating the entire reef (Facon et al. 2016). Potentially, if larger reefs decline more than the other reefs, such trends may not encapsulate how the region's reefs are performing. Nevertheless, analyses will continue to improve through the Allen Coral Atlas Initiative and other reef mapping services (Lyons et al. 2020). Through individual countries and regional models, some differences between the overall model and the regional models can be seen, with certain countries showing a much greater decline in reef cover while others have maintained a higher coral cover.

Coral cover remains but one measure of coral reef health. While low cover is synonymous with declining reef habitat, high cover does not necessarily mean a healthy coral reef (Richards 2013, Cannon et al. 2019). The East Asian region encompasses much of the greater Coral Triangle region, a marine domain with the greatest diversity of reef species globally (Veron et al. 2009). Potentially, the diversity of coral and other reef organisms may also help mitigate some of the potential losses from environmental and anthropogenic effects by acting as redundancies or promoting resilience of reefs to perturbations (Nyström 2006, Hughes 2010). Maintenance of the different ecological functions provided by these reefs necessitates the integrity of a certain level of this diversity (Bellwood et al. 2019, Brandl et al. 2019). Therefore, reefs with large stands of single species would not sustain the same level of ecosystem functions provided by a more diverse reef (McWilliam et al. 2018).

Changes in community assemblage are already being observed in many reefs across the world, with loss of functional redundancies and species in certain areas (Adjeroud et al. 2009, Graham et al. 2014, Guest et al. 2016, Darling et al. 2019, Keshavmurthy et al. 2019). Shifts towards weedy or stress-tolerant species that thrive under greater stressors may present a reef with lower ecosystem functioning despite the lack of change in coral cover (Darling et al. 2013). The resilience of reefs is also built up through different components of diversity (Lam et al. 2017, 2020), not only in terms of coral species diversity but also supplemented by population genetic diversity (Selkoe et al. 2016) and richness of other reef organisms (Bellwood and Hughes 2001, Bellwood et al. 2003, Darling et al. 2017). In addition, documented shifts toward macroalgal dominance are less common in the region, with potential shifts towards other major reef benthos like zoantharians, anemones, and soft corals (Bruno et al. 2009, Norström et al. 2009). Together, these issues highlight that changes in coral community assemblages on individual reefs remain important in reef monitoring and require more studies.

Comparing the change in coral and macroalgal cover across time with the various socio-economic and environmental variables also highlights which potential variables can predict changes to the habitats. These variables are easier and quicker to measure on larger scales, potentially warning of changes in the ecosystem. The strongest predictor of coral cover, both positively and negatively, was net primary productivity maximum values and standard deviation, respectively. These were modeled through photosynthetically active radiation, sea surface temperature, and chlorophyll-a concentration, which generally increases with coral cover and fluctuates with other non-coral photosynthetic organisms such as algae (Yeager et al. 2017). The decrease in coral cover with greater population sizes within 100 km highlights the anthropogenic impacts of coastal use and development on coral reefs (Hughes et al. 2003, Cleary et al. 2006, Darling et al. 2019). Maximum bleaching alert levels were the only temperature-related variable to appear in the model with a small effect, likely due to the use of monthly temperature data, representing the shorter-term temperature effects on coral reef cover as well (McClanahan et al. 2019). Likewise, with the previous model trends, depth and year respectively also weakly predict a slight decline and coral cover increase, while country variables were only significant for a few countries. The lack of country significance also points towards coral reefs crossing geopolitical boundaries, which do not usually coincide with ecologically meaningful limits. Surprisingly, coral cover was also negatively predicted by reef area within 100 km, though the rationale for this remains unknown.

For macroalgal cover, more environmental correlates were inferred, though many of these included various temperature terms. Mean and minimum sea surface temperature and maximum degree heating weeks predict an increase in macroalgal cover, while maximum SST and maximum SST anomalies predict a corresponding decrease. These factors highlight that an increase in temperature promotes macroalgae growth, though larger fluctuations in temperature (with maximum SSTA) can negatively affect macroalgal growth. While also negatively correlating with macroalgal growth, maximum SST had a large standard error, potentially due to the different effects across regions. It is also possible that these temperature variables are indicative of coral degradation, leading to corresponding algal growth and colonization. Both maximum and standard deviations of primary productivity also correlated with increases in macroalgal cover, with better conditions for growth and greater variability potentially allowing for more macroalgal growth. Similarly, human populations within 100 km were predictive of greater algal growth, likely representative of anthropogenic impacts increasing with macroalgal growth. Again, in tandem with the coral model, year, depth, and country were all in the model, predicting an increase, decline, and varying significance between countries, respectively.

The multi-model averaging approach of studying environmental and socio-economic covariates helps to elucidate variables that correlate with cover trends. Through these correlations, several important similarities and differences were highlighted. For one, while declines in coral reef health are not evident in the current analyses, the presence of anthropogenic effects such as population within 100 km imply that development and human pressures have deleterious effects on coral and macroalgal cover (Todd et al. 2010, Heery et al. 2018, Bang et al. 2021). Furthermore, the effects observed here highlight that development may already have had impacts on coral cover prior to the 1980s when surveys were conducted, with other paleo-ecological studies finding impacts from as far back as when agricultural development occurred (Pandolfi et al. 2003). The impacts of productivity and temperature are more varied and complex, with differing effects depending on the variations and scale of their impacts on both coral and macroalgal cover change, as they influence both the individual growth of coral and macroalgal cover and their interaction (Darling et al. 2019, McClanahan et al. 2019). Lastly, while countries did show up as an effect in the ensemble models, most country comparisons were not significant, especially for coral cover. This effect means that despite reefs being managed within individual countries, these geopolitical boundaries are likely not ecologically meaningful for reefs and other natural habitats in general. Management thus needs to contend with any related mismatches in priorities to be effective, and cross-country collaborations are necessary since coordinated approaches and actions tend to have much larger positive effects (Chou 2013, Borja et al. 2020).

RECOMMENDATIONS

During the data acquisition phase for the analysis, it was recognized there were likely more data available from unidentified and non-traditional or non-English sources, especially for governmental reports, considering the vast diversity of languages used in the region (Nuñez and Amano 2021). Translation of these documents and extracting this information to be analyzed and for contributing towards a regional database would improve the ease of conducting such analyses in the future. This database would provide a repository for the long-term storage of the data beyond projects and details the various contact personnel for data sharing agreements. Collaborative science remains necessary for conservation progress, especially in the present day, and local input is increasingly paramount for putting these findings into context (Costello et al. 2012, Stefanoudis et al. 2021). With the clear ecological differences between coral reef regions and proportionately fewer studies in the Indo-Pacific, it is increasingly important that the differences are studied for better management decisions to be made for the region (Roff and Mumby 2012, Chou 2013, Aswani et al. 2015, Williams et al. 2016).

It is also evident that traditional reef monitoring through field visual surveys remains important, especially if these monitoring efforts are continued through long time periods (Lindenmayer and Likens 2009, Flower et al. 2017, Hughes et al. 2017). This remains the best approach for ground-truthing and cross-validation of the reef data for comparisons across time or against other methodologies such as remote sensing, even with improvements in satellite imagery and analysis (Obura et al. 2019). With the difficulties in identifying taxonomic details (e.g., species identification) through visual surveys, and more limited resources available for surveying the higher diversity in this region, monitoring of sites needs to be prioritized based on available resources and expertise (Madin et al. 2019, Borja et al. 2020). More specific locality-based monitoring studies examining changes in benthic cover and compositions in the region has been on the increase, with recent studies done in Indonesia (Cleary et al. 2014), Philippines (Licuanan et al. 2019), Singapore (Guest et al. 2016), Thailand (Phongsuwan et al. 2013, Yeemin et al. 2013), Vietnam (Long and Vo 2013), Hong Kong (Wong et al. 2018b), Japan (Hongo and Yamano 2013), and Taiwan (Keshavmurthy et al. 2019). As reefs continue to change in composition with each stressor, particularly bleaching events and typhoons, monitoring must accommodate and note these changes in the long term (Baird et al. 2005, Harii et al. 2014, White et al. 2017, Ng et al. 2020). With considerable monitoring requirements needed to cover the vast area in this region, greater participation, especially with citizen science and new local capacity, can expand the scope of studies and help provide more detailed insights into how reefs change over time (Lau et al. 2019, Obura et al. 2019, Gurney et al. 2019).

Here we suggest that long-term study sites are prioritized, and funding made available for the annual monitoring of a select number of representative reef sites. Other sites can be surveyed with additional funding or when projects arise, while training in the survey methodology and species identification continues to be scaled up through time. Long-term monitoring is imperative as longer datasets only garner more value over time, especially if the program is effectively planned (Lindenmayer and Likens 2009, Hughes et al. 2017). However, it remains difficult to track coral reef community changes for highly diverse regions. From the results of the current analysis, change in coral cover appears to be less important in the shorter term, potentially hiding compositional changes between functional groups or evolutionary diversity that may be more representative of reef health (Huang et al. 2016). These metrics need to be incorporated in future analyses to provide more holistic reef assessments to inform policy changes and establish mitigative measures (Lam et al. 2017).

The greater availability of remotely sensed data can also improve monitoring outcomes. Socio-economic and environmental correlations provide insight into how the different factors may affect reef trends, though these are merely correlations and not definitive causations. As a starting point, these variables provide some measures which can be used in tandem to track how reef health is changing across larger scales and may provide early warning signs of change (Darling et al. 2019, McClanahan et al. 2019). Certain anthropocentric variables highlight the environmental impacts that are evident in the region as well, despite the lack of coral cover decline observed through this analysis. It remains necessary to identify if these impacts are caused by the current state of coastal development to better understand the baselines here, and predict how reefs will change with future development, warming trajectories, and management (Knowlton and Jackson 2008, Bang et al. 2021).

Conservation and management of reefs are important endeavors to ensure that coral reefs thrive into the future. With the data and analyses presented here, it is evident that research into reefs needs to continue and guide management decisions. We hope that this report drives a greater appreciation for the differences amongst reef regions and garners greater resources to monitor reef habitats in this region. Reef conservation efforts can only benefit from more resources and actors, with greater participation of citizen scientists, local researchers, and the younger generation of scientists and managers, guided by the experience and work of those more senior.

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PART II:
REPORTS FOR INDIVIDUAL COUNTRIES

The second part of this report (Part II) were submitted by GCRMN national coordinators from Northeast Asia and Southeast Asia and provides a more detailed update on the coral reefs for each individual country with references to all the compiled sources of survey data from 1983 to 2019, and 2020 in some instances. Some national country reports presented in Part II may also analyze and discuss data dissimilar from the overall regional synthesis presented in Part I.

With minor omissions, each national report contains eight main sections, which have been formatted as follows:

1) **Introduction:** General introduction of the coral reef distribution, MPAs, and monitoring programs, including maps, number and location of sites, survey methods, target areas, etc.

2) **Status and Trends:** Describes the status and trend of coral reef monitoring data through time and across various depths compiled by each country for this regional analysis. This section includes graphs of average changes in coral cover based on quantitative surveys and may also consist of summary tables for the specific country.

3) **Coral Bleaching 2016:** Details the timeline, extent, and severity of the third global coral bleaching event observed by each country during the peak of the last strong El Niño from 2014 to 2017.

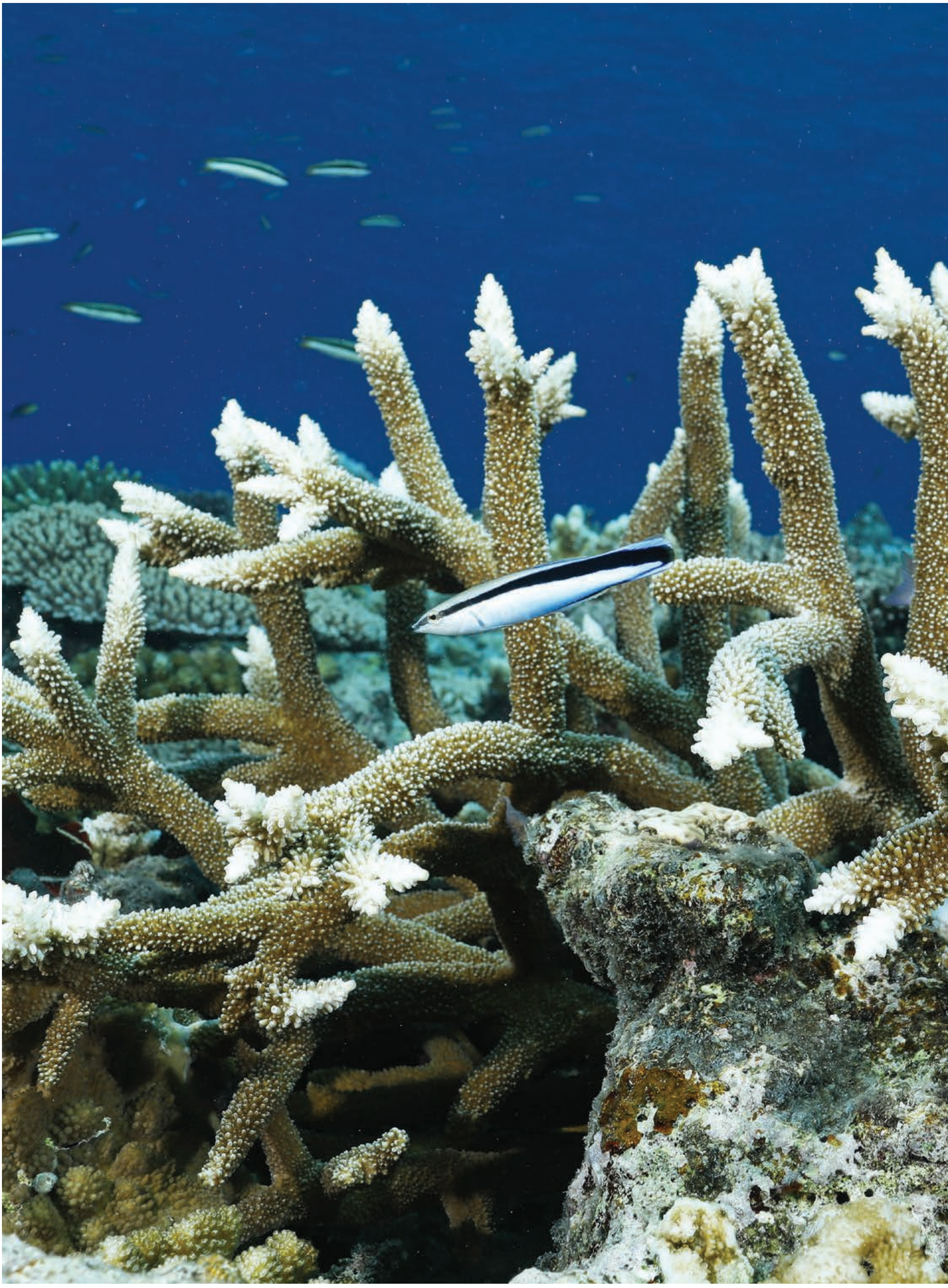
4) **Drivers and Pressures:** Selective list of major natural and anthropogenic disturbances and events (e.g., bleaching, COTS outbreak, typhoon) that affect and drive the status and trend of local coral reefs gathered by the co-authors in each country.

5) **Recommendation:** Provides recommendation(s) on the local management of coral reefs and other conservation efforts.

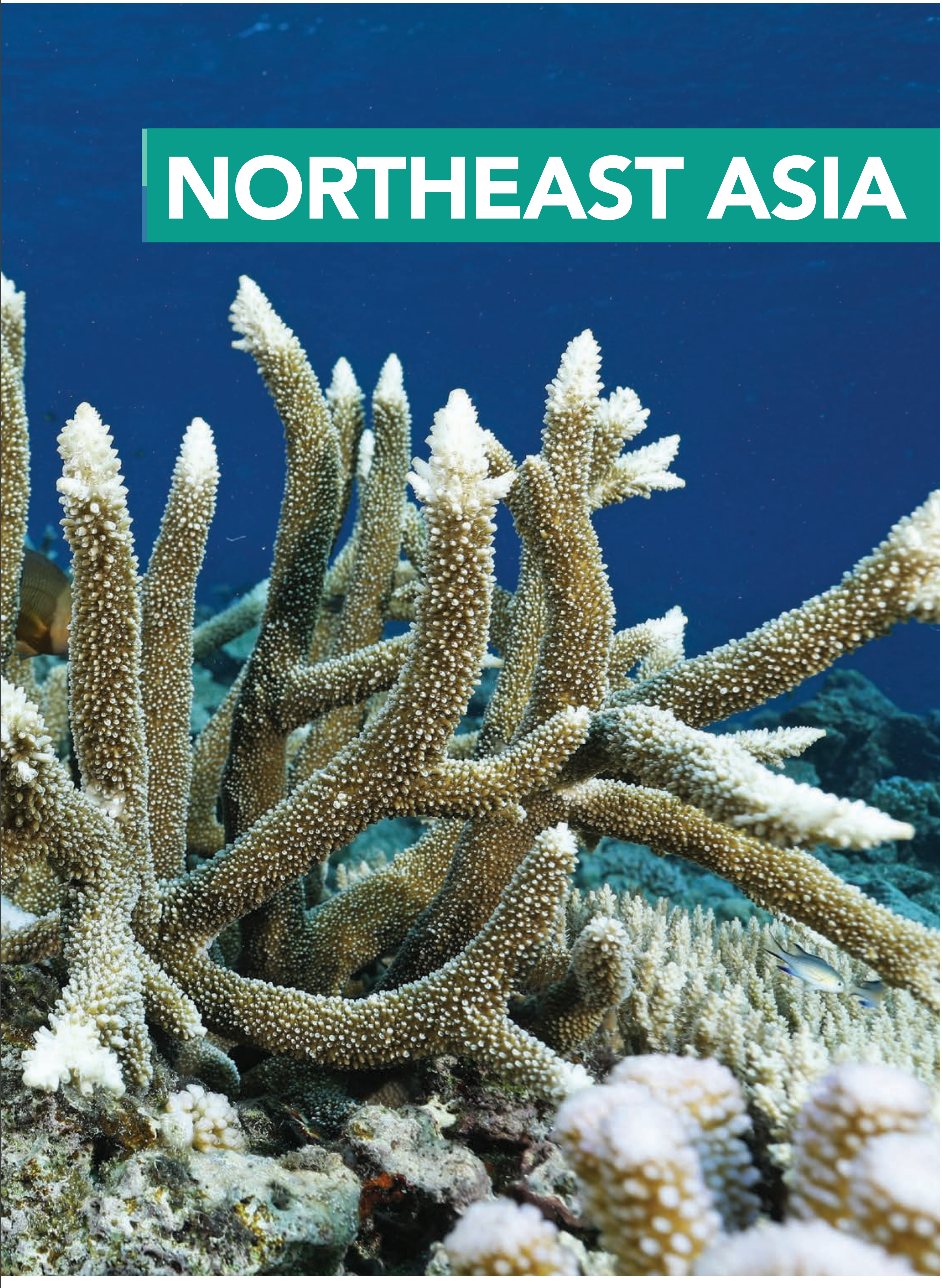
6) **Data Contributors:** List of the national contributors who provided monitoring data for this regional analysis and individual national reports.

7) **Acknowledgment:** Special thanks or acknowledgment by each national monitoring program.

8) **References:** List of primary references, published scientific articles, or general literature on local coral reefs gathered from co-authors in each country and cited in the national report.



NORTHEAST ASIA



NORTHEAST ASIA

China

Introduction

This country report is mainly based on the book *The Status Report of Coral Reefs in China: 2019*, which compiled data from coral reef ecological monitoring conducted by many institutions between 2011 and 2018. The compilation of *The Status Report of Coral Reefs in China: 2019* is organized by the Coral Reef Branch, Pacific Society of China (CRBPSC), together with ten units or teams engaged in coral reef monitoring. This consortium launched the China Coral Reef Monitoring Network (CCRMN) in July 2018 to formalize the regional and nation-wide networks for cooperation, as well as to establish mechanisms and norms for data exchange and sharing.

The data presented in this report were collected by various coral reef survey and monitoring teams using standard protocols published in 2005: the technical standard for coral reef ecological monitoring methods, i.e., "Technical Specifications on Coral Reef Ecological Monitoring (HY/T 082-2005)". Coral reef ecological survey and monitoring consist of physical, chemical, and biological indicators. Physical and chemical indicators include water depth, water temperature, salinity, transparency, substrate type, and pH. Biological indicators include the health status of hermatypic corals (i.e., species, coverage, mortality, disease, and recruitment), coral reef fishes, large coral reef benthic animals, and algae. The Reef Check method was also used for some of the monitoring surveys.

The coral reefs and coral communities of China are mainly distributed around the tropical islands and atolls in the South China Sea (SCS) and as fringing reefs along mainland China's southern coast (Figure 1). China's coral reefs occur from Xuwen (20°15′N), Guangdong Province to the South China Sea's Dongsha (Pratas) Islands (20°40′N), Xisha (Paracel) Islands (17°08′~15°46′N), Zhongsha Islands (Macclesfield Bank) (19°33′~13°57′N), and Nansha Islands (Spratly Archipelago) (11°55′~3°35′N). Coral communities also occur in Daya Bay (22°40′N), Guangdong Province, and neighboring Hong Kong waters. Based on the path of the mainstream Kuroshio warm water currents, the northernmost coral communities occur in Dongshan county (23°45′N), Fujian Province.

Status & Trends

SPECIES DIVERSITY & COVERAGE OF HERMATYPIC CORALS

China has a high diversity of hermatypic corals, with 443 species from 16 families and 77 genera. The Nansha islands have the highest species richness with 384 species, while Fujian has the lowest richness with only seven species recorded to date.

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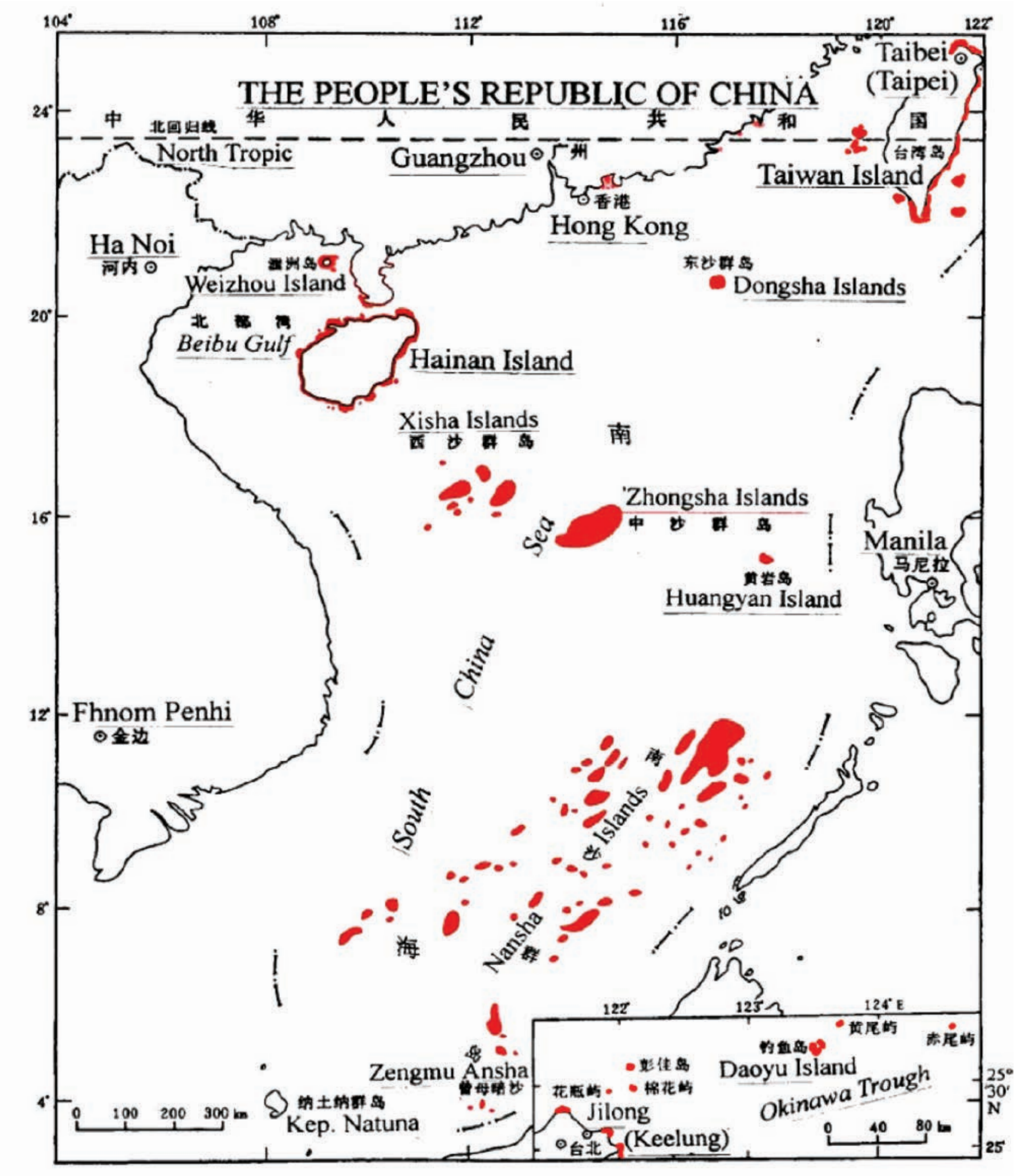


FIGURE 1: *Distribution of hermatypic corals in China.*

The status of live hermatypic corals in China is shown in Table 1 and Figure 2. We reviewed the historical changes of live hermatypic coral cover in various regions of China and analyzed their trend. Generally, the coverage of live hermatypic corals in China has declined in recent years (Figure 3a). Moreover, the offshore coral reefs have declined more rapidly than their coastal counterparts (Figure 3b).

TABLE 1: *Status of live hermatypic coral coverage in China.*

PROVINCE / REGION	REGION	YEAR	LIVE HERMATYPIC CORAL COVERAGE
Fujian	Dongshan	2013	5.90%
Guangdong	Daya Bay	2015	21.97%
Guangdong	Dapeng Bay	2016	35.21%
Guangdong	Wailingding Island	2018	36.00%
Guangdong	Jiapeng Islands	2018	8.11%
Guangdong	Xuwen	2018	6.86%
Guangxi	Weizhou Island	2018	17.60%
Guangxi	Xieyang island	2018	4.67%
Guangxi	Bailongwei	2018	0.90%
Hainan	East Hainan	2018	6.67%
Hainan	West Hainan	2018	9.00%
Hainan	South Hainan	2018	11.00%
Hainan	Xisha Islands	2016	5.44%

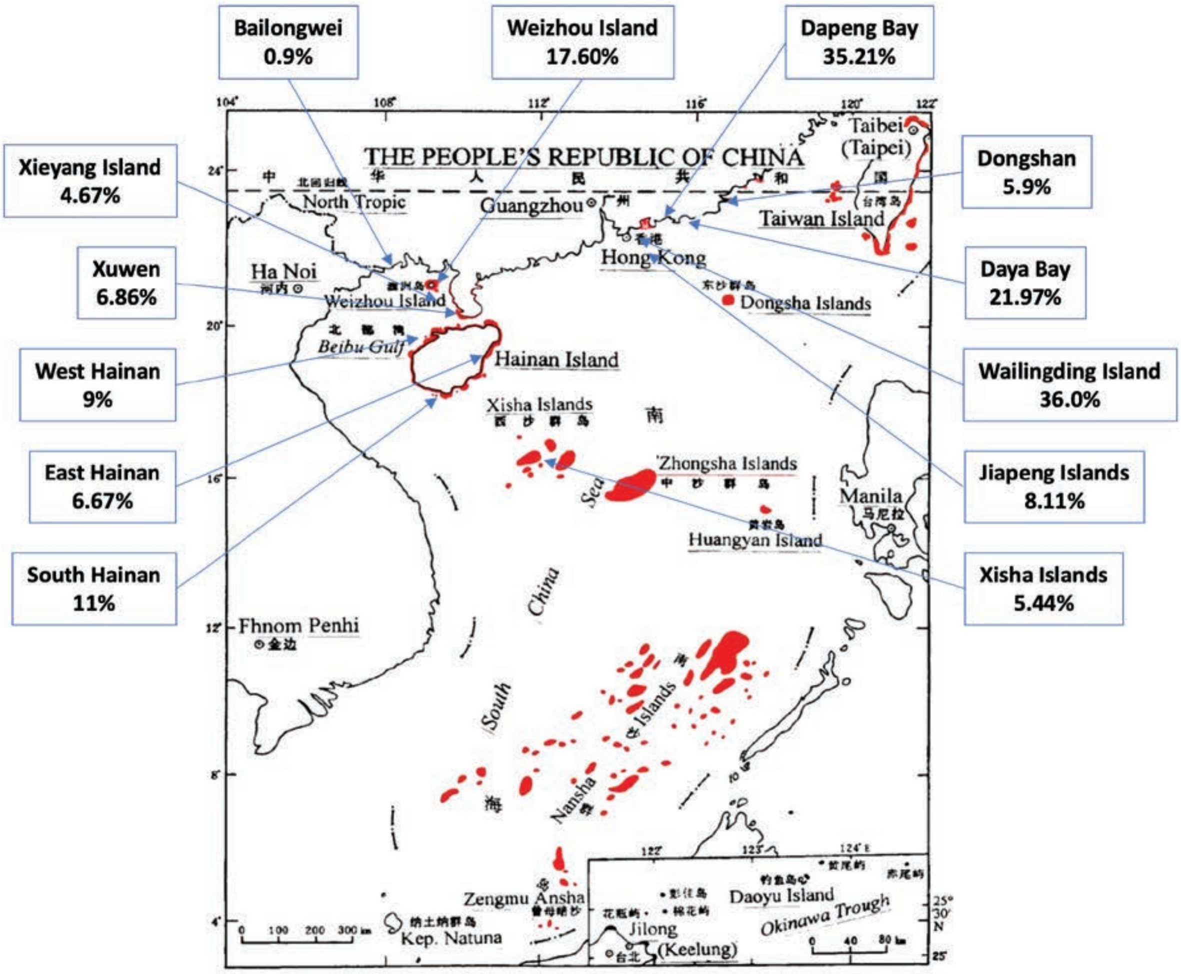


FIGURE 2: *The live hermatypic coral coverage in China.*

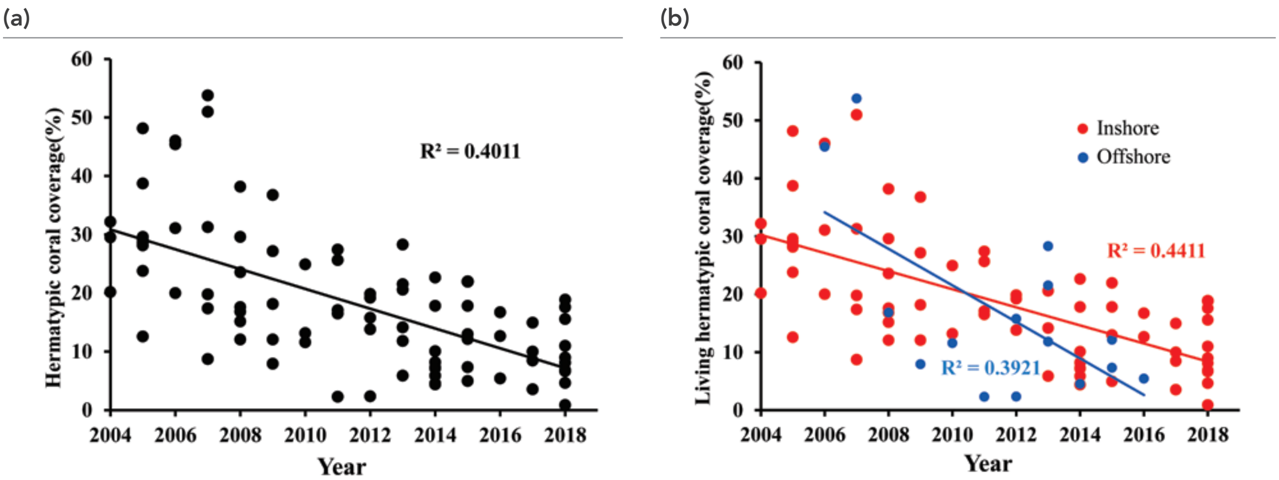


FIGURE 3: *(a) Changes in coverage of live hermatypic corals in China; (b) Changes in coverage of live hermatypic corals between inshore and offshore locations in China.*

HEALTH ASSESSMENT OF CORAL REEFS & HERMATYPIC CORAL COMMUNITIES

This report assesses the coral reefs or hermatypic coral communities in Dongshan of Fujian, Daya Bay, Dapeng Bay, Miaowan, Xuwen of Guangdong, Weizhou Island of Guangxi, Sanya, Wenchang, and the Xisha Islands of Hainan. See Table 2 and Figure 4 for the comprehensive assessment results of various coral reefs or hermatypic coral communities.

The methodology of coral reef health assessment here is as follows: the health of a coral reef is defined as "good" (75≤CHI≤100), "general" (35≤CHI<75), or "poor" (0<CHI<35) (Sun, 2018). This method used seven indicators: live hermatypic coral cover, sand substrate cover, *Acropora* cover proportion, *Porites* cover proportion, *Goniopora* cover proportion, *Galaxea* cover proportion, and the number of hermatypic coral species.

According to the assessment results, the coral reefs and hermatypic coral communities in China are mostly graded as "general" and "poor." The hermatypic coral communities in Dongshan, Miaowan, Xuwen, and Wenchang were graded as "poor," which means the coral reefs or hermatypic coral communities of these areas have changed fundamentally. The status of the coral reefs in Daya Bay, Dapeng Bay, Weizhou Island, Sanya, and the Xisha Islands were graded as "general."

TABLE 2: *Status assessment of coral reefs or hermatypic coral communities in China (coral reef ecosystem health assessment comprehensive index, CHI).*

LOCATION	YEAR OF DATA	CHI	GRADING
Dongshan, Fujian	2013	28	Poor
Daya Bay, Guangdong	2015	53	General
Dapeng Bay, Guangdong	2014	47	General
Miaowan, Guangdong	2018	25	Poor
Xuwen, Guangdong	2014	27.5	Poor
Weizhou Island, Guangxi	2018	42.5	General
Wenchang, Hainan	2014	32.5	Poor
Sanya, Hainan	2018	55	General

TABLE 2 *(Cont'd).*

LOCATION	YEAR OF DATA	CHI	GRADING
Xisha Islands (Yongle Islands)	2013	72.5	General
Xisha Islands (Xuande Islands)	2015	40.5	General
Daya Bay, Guangdong	2006*	65	General
Xuwen, Guangdong	2006*	42.5	General
Weizhou Island, Guangxi	2006*	61	General
Wenchang, Hainan	2006*	80.5	Good
Sanya, Hainan	2006*	85.5	Good
Xisha Islands	2006*	97.5	Good

* The data for 2006 are from the National 908 Survey, but as sand substrate cover was not surveyed, it was substituted by the same area's data in recent years.

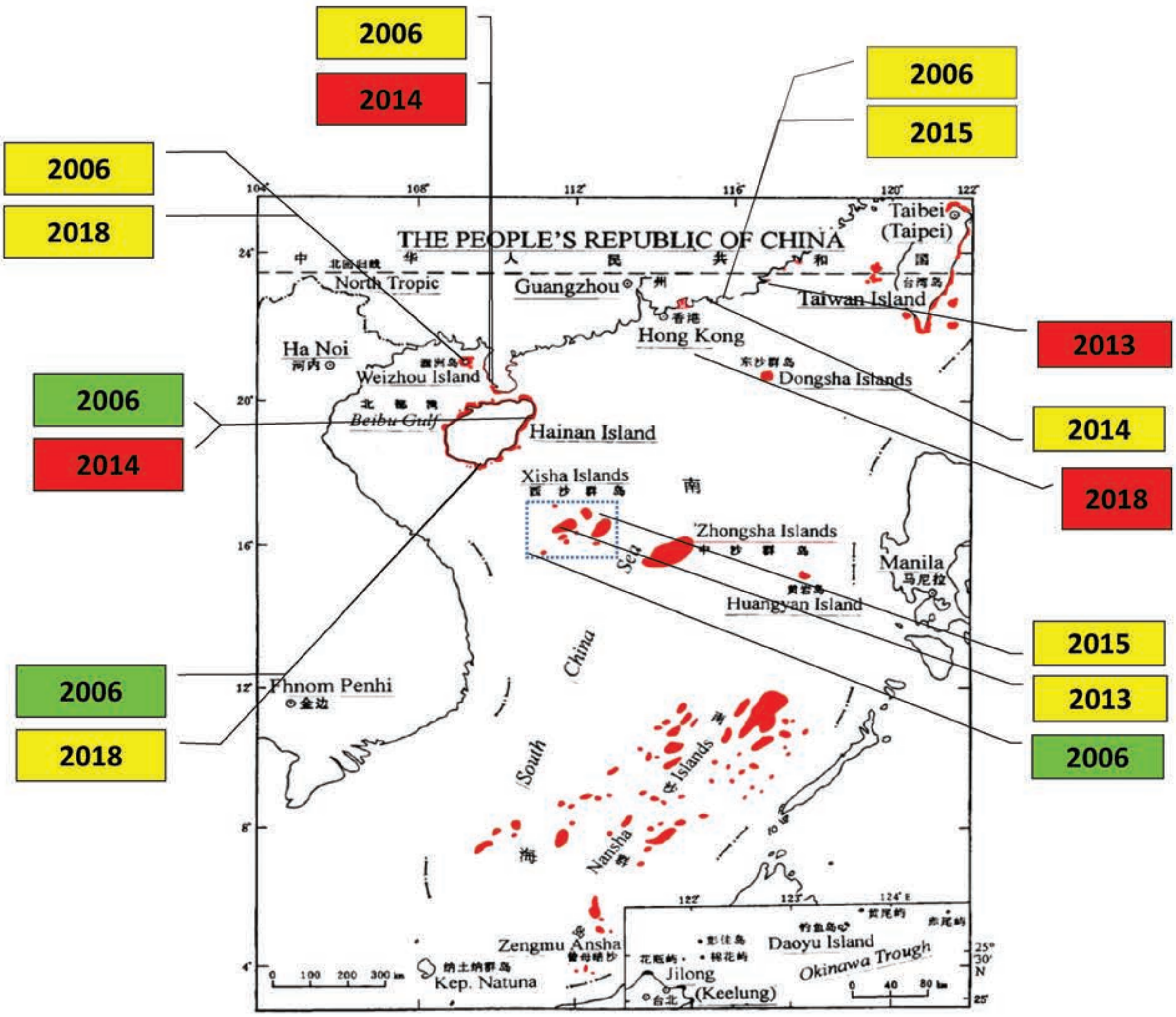


FIGURE 4: *The health status of coral reefs and coral communities in China graded as "good" (green), "general" (yellow), and "poor" (red), and the year surveyed.*

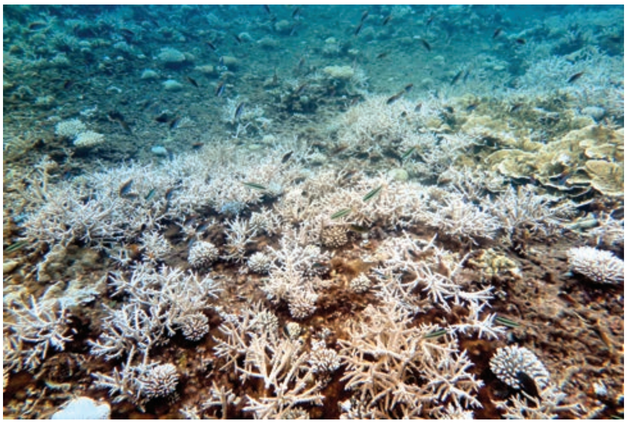
Coral Bleaching 2016

Being on the northern edge of the world's coral reef distribution and far from the equator, China's coral reefs are relatively unaffected by El Niño. At present, the impact of temperature anomalies caused by global climate change on China's coral reefs appears limited. However, localized coral bleaching events caused by abnormally high seawater temperatures have been recorded in Sanya, the Xisha Islands, and Nansha Islands with increasing annual frequency since 2016. For example, coral bleaching was observed in Luhuitou of Sanya in 2010, 2015, and 2017 (Figure 5) (Li et al. 2012, Huang Hui *unpublished data*). Coral bleaching was also recorded in the Xisha Islands in 2014 and 2019 (Figure 5) (Zuo et al. 2015, Li et al. 2016b), and in the Nansha Islands in 1998 and 2007 (Li et al. 2011) and each year from 2016 to 2020 (*unpublished data*). According to historical seawater temperature data and climate change models, it is speculated that the frequency of heatwaves would increase in the future and lead to a higher risk of coral bleaching for reefs in China.

(a)

(b)

(c)



(d)

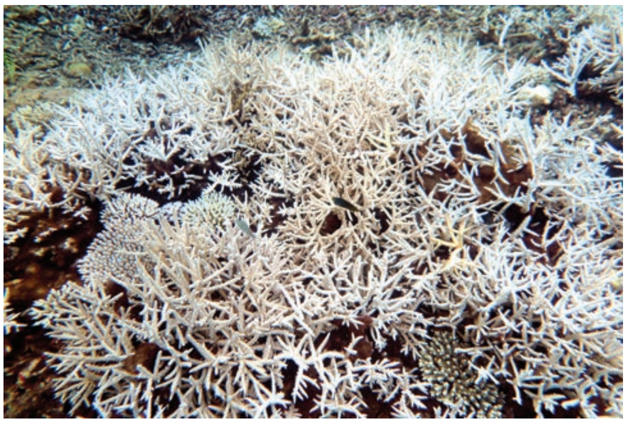


FIGURE 5: *(a),(b) Bleached massive corals* ***Porites lutea*** *and* ***Goniastrea pectinata*** *and healthy branching corals* ***Acropora digitifera*** *and* ***Pocillopora damicornis*** *in Luhuitou of Sanya in June 2015. (c), (d) Mass bleaching of staghorn corals in Yagong Island of the Xisha Islands in July 2019.*

Drivers & Pressures

With the fast growth of China's economy, heavy population and intensive land-use have brought severe negative impacts on the marine ecosystems, especially on coral communities and coral reefs. Coral reefs in the SCS have faced severe degradation in the past three decades, especially in Hainan and along the coasts of the mainland (Hughes et al. 2013). Without exception, the offshore live hermatypic coral coverage has declined dramatically in recent years. The major threats to Chinese coral reefs are climate change, human activities, coral diseases, and predators. Climate change is complicated with many associated outcomes, including abnormal seawater temperature, ocean acidification, sea-level rise, increasing frequency and intensity of extreme weather events such as typhoons and floods. Human activities also threaten China's coral reefs, including overfishing, illegal fisheries, eutrophication, coastal construction, and tourism. In addition, outbreaks of coral diseases and predators, such as crown-of-thorns starfish and the coral-killing sponge, *Terpios hoshinota*, also impact coral reefs in China.

The most severe threat to China's coral reefs in recent years is the crown-of-thorns starfish (COTS, *Acanthaster planci*). During the previous COTS outbreak from 2006 to 2009 in the Xisha Islands, almost all its coral reefs were destroyed and have yet to recover (Li et al. 2019). A COTS outbreak was once again recorded in Xisha and Nansha Islands

TABLE 3: *Impact factor assessment table of coral reefs in China.*

PROVINCE	REGION	MINING	ILLEGAL FISHERY	AQUACULTURE	OVERFISHING
Fujian	Dongshan	0	2	3↓1	2
Guangdong	Daya Bay	1	1	1	2
	Pearl Estuary	1	2	0	2
	Xuwen	3	1	2	2
	Dongsha Islands	1	1	0	1
Guangxi	Weizhou Island	2↓1	1	1	1
	Xieyang Island	0	1	0	1
	Bailongwei	0	1	1	1
Hainan	East Hainan	1	2	3↓1	2
	West Hainan	1	2	3↓1	2
	South Hainan	1	1	1	1
	Xisha Islands	2	2	0	1

NOTE: The numbers in the table represent the degree of influence of the factor on the coral reefs in the area; 0: no impact; 1: slight impact; 2: moderate impact; 3: severe impact. All values are historical conditions (2004–2018), and red values are current status and changes.

from 2018 to 2020, and a recent survey in 2020 also found a COTS outbreak in some parts of the Zhongsha Islands.

These stressors often threaten Chinese coral reefs collectively (Table 3), such as pollution, eutrophication, destructive fishing, etc. In the example of the Xisha Islands, coral reefs suffered from overfishing and the outbreak of crown-of-thorns starfish, which have caused high mortality and degradation of the reefs. In addition, the overfishing of herbivorous fishes could have hindered the natural grazing of benthic macroalgae and thus limited the recovery of these degraded coral reefs.

Together, these stressors have resulted in long-term low coral cover and low larval recruitment in the Xisha Islands (Wu 2011). Although it is widely accepted that climate change has less impact than human activities on coral reefs in China, we expect that it may become more prominent in the near future as climate change intensifies. It is particularly worth noting that climate change and human activities are likely to interact to produce more complex and severe impacts on our coral reefs. The coupling of these two major threats will affect and thereby determine the health status and ecological services of coral reefs in the SCS.

CONSTRUCTION	TERRESTRIAL POLLUTION	NATURAL DISASTER	CLIMATE CHANGE	PREDATOR	CORAL DISEASE	TOURISM
1	2	0	0	0	0	1
1	1	1	0	0	0	1
1	2	1	0	0	0	1
1	3	1	0	0	0	1
0	0	1	2	1	1	0
1	1	1	0	0	0	2
0	0	1	0	0	0	0
0	1	1	0	0	0	1
2↓1	2	1	0	0	0	1
1	2	0	0	0	0	0
3↓0	2	1	1	1	1	3
1	0	1	1	3	1	1

RECOMMENDATION

The most urgent need at present is the control of the COTS outbreaks. It seems there is still no effective technology available to control the COTS. We recommend that local government, NGOs, and all the institutions and societies of concern, take actions to address this problem. We have spoken out many times to the broader audience and made reports to the local government to mobilize human resources to remove the COTS. The local Sansha municipal government of Hainan province had sent fishing vessels to remove the COTS in 2019 and 2020. The South China Sea Bureau of the State Natural Resources Administration also sent ships to remove the COTS in 2019. Thus, there is an urgent need to research and develop COTS control technology, such as underwater drones to kill COTS.

The following recommendations may be suitable for the long-term:

1. Improve the management system, which includes further developing and implementing coral reef projects, improving laws and regulations, conducting conservation plans, strengthening team building, giving full play to the functions of the marine conservation area in protection, management, and science popularization.
2. Conduct systemic coral reef monitoring and develop scientific technology, which includes improving the coral reef monitoring system, consistently investing in ecological monitoring and fundamental research, deepening our understanding of coral reefs, establishing a scientific evaluation system, and forming a closed-loop operation mode that involves "monitoring-research-evaluation-demonstration".
3. Effectively carry out restoration and maintenance, which includes consistently promoting coral reef restoration and replenishment, developing a new model and benchmark for marine ecological restoration based coral reefs.
4. Actively carry out marine awareness education, which includes connecting government, research institute, social organization, and the public, establishing a government-guided innovative alliance on coral reef conservation, increasing the publicity of ocean ecological civilization construction by integrating various resources, fully enhancing the marine awareness of the nation, and promoting the demonstration mode of ocean ecological civilization construction based on coral reefs.

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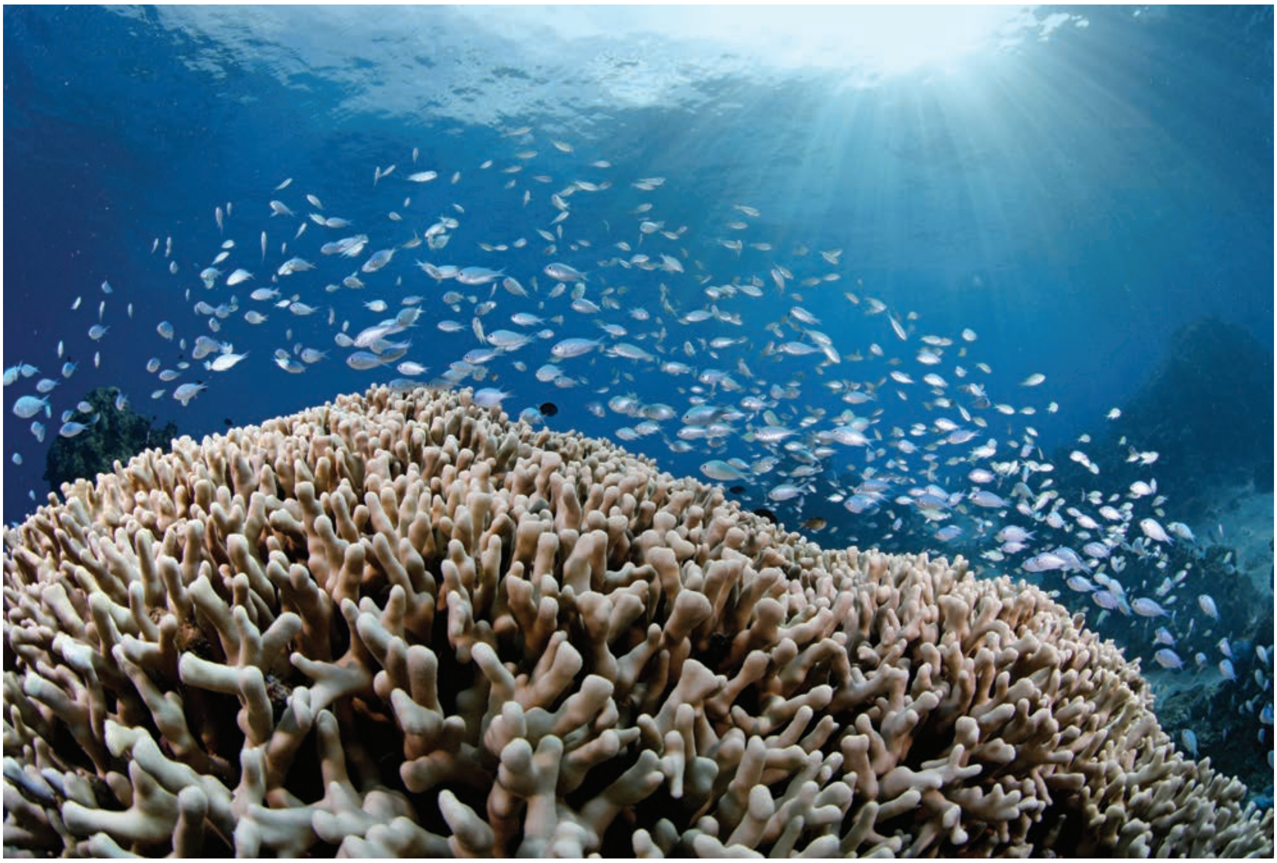

NORTHEAST ASIA

Hong Kong

Introduction

Hong Kong is located in southern China at the northern boundary of the South China Sea and has a marginal environment for coral growth (Ang et al. 2005) due to its fluctuating seasonal water temperature. In winter, the water temperature could drop to as low as 14°C, while in summer, the water temperature could reach 30°C. Thus, no coral reefs could be formed in Hong Kong waters. Nonetheless, at least 84 species of hermatypic corals could still be found in Hong Kong (Ang et al. 2003). These corals form patchy communities, dotting mainly the northeastern and eastern waters of Hong Kong, where the water condition is more oceanic. The large freshwater outflow from the Pearl River, the third largest river in China, makes the western Hong Kong waters brackish. In addition, high turbidity is brought about by a heavy sedimentation load from this river. All these make the western waters unfavorable for hard coral growth. Thus far, the gorgonian *Guaiagorgia* sp. is the only abundant octocoral in Hong Kong's western waters. Many other octocorals and black corals are found in the northeast to southern Hong Kong waters (Ang et al. 2010, 2011). However, most of these are found in deeper waters (>5 m) below the distribution range of most hermatypic hard corals.

Since the 1980s, there has been an increased awareness by the Hong Kong public on the importance of coral communities as a marine habitat. Efforts have therefore been made to protect these coral communities. Hoi Ha Wan Marine Park and Tung Ping Chau Marine Park (Figure 1) were established in 1996 and 2001, respectively, to protect coral communities located inside the parks. In the last 20 years, there was also increased effort by both the scientific community and the government to carry out studies related to different aspects of corals and coral communities. There are now coral-related research studies carried out in all major universities in Hong Kong. Research topics include coral reproductive biology, recruitment, phylogeography, genetic connectivity, physio-ecology, food nutrition, larval biology, microbiology, symbiosis, microbiology, ecosystem functions, and pollutant impacts, including microplastics. Longer-term monitoring programs have also been put in place, with participation from the scientific community as well as from citizen scientists and volunteers. This current update on the coral monitoring works in Hong Kong will not cover studies initiated by the different university researchers but will focus on works commissioned by the Agriculture, Fisheries, and Conservation Department (AFCD) of the Hong Kong SAR Government. Some of these works, however, also involved research teams from the university.

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Status & Trends

REEF CHECK

Hong Kong is one of the first places around the world to have participated in Reef Check. The first Reef Check was carried out in 1997 with 40 divers. Since then, Reef Check has become the main territory-wide activity engaged in the annual survey of coral community health supported by the Hong Kong Reef Check Foundation and the AFCD. In 2019, there were 88 volunteer teams with more than 900 volunteers involved, covering 33 sites (Figure 1). In 2020, despite COVID-19, 107 dive teams with more than 800 volunteers participated and surveyed the same 33 sites covered in the previous years.

Reef Check results from the last six years (2015-2020) indicate Hong Kong coral communities to be generally healthy (Table 1). A few sites, however, appear to have suffered some degradation, including some sites within the marine parks, e.g., Wong Ye Kwok in Tung Ping Chau Marine Park (TPCMP), Lai Chi Wo in Yau Chau Tong Marine Park (YCTMP); Moon Island, Pier and Gruff Head in Hoi Ha Wan Marine Park (HHWMP) (Figure 2). Nonetheless, some of these sites appear to be recovering. Moon Island experienced massive coral mortality in 2015, resulting in a big drop in its coral cover recorded in 2016. This mortality was probably related to red tide, but the actual cause has yet to be confirmed (Wong 2017, Wong et al. 2016). There was a continuing decline in coral cover in the Pier from 2015 to 2018, with an apparent rapid recovery thereafter. Some of the sites also suffered from bioerosion due to sea urchin grazing. The AFCD has since launched a restoration program in this marine park with some progress.

Most sites outside the marine park in the northeastern waters experienced relatively stable conditions with no significant change in coral cover. However, four sites are noted to have suffered >10% decline in coral cover from 2015 to 2020, including Wu Pai, Crescent Island West, Wong Chuk Kok Hoi, and Chek Chau (Port Island) (Figure 3A, Table 1). Bioerosion is very serious in Chek Chau and is the main cause of the coral decline at this site (see Section 3 below).

Sites in the eastern waters appeared to do better. Most sites experienced an increase in coral cover, but four sites, Long Ke Wan, Town Island, Pak Ma Tsui, and Shelter Island, suffered >10% decline (Figure 3B, Table 1). Of these, Shelter Island suffered the most. It is not clear if the coral decline in Shelter Island could be related to increased recreational diving. This site has become very popular for recreational activities in recent years, especially during summer. Further study is now underway to verify the potential cause of the coral decline in this site.

It should be noted that large fluctuations in coral cover are sometimes recorded in the same site over subsequent years. However, as coral growth is slow in Hong Kong (Ang et al. 2005), a large increase in coral cover of over 5% in a subsequent year should be taken with some caution. This increase may simply be a result of sampling artifacts. Nonetheless, data from Reef Check could still be valuable in providing some understanding of the long-term trend in coral cover change.

In Reef Check 2020, the health condition of corals was also assessed in 21 sites using the Coral Health Monitoring Chart.[1] The average health index was 4.31 (ranging from 3.15 to 5.45) and was comparable to that in 2019 (3.96) and 2018 (4.04). The average health index is above the general average value of 3, indicating the corals were generally healthy. Other more detailed results of Hong Kong Reef Check can be found on the AFCD website.[2]

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[1] *https://coralwatch.org/index.php/monitoring/*

[2] *https://www.afcd.gov.hk/english/conservation/con_mar/con_mar_cor/con_mar_cor_hkrc/con_mar_cor_hkrc.html*

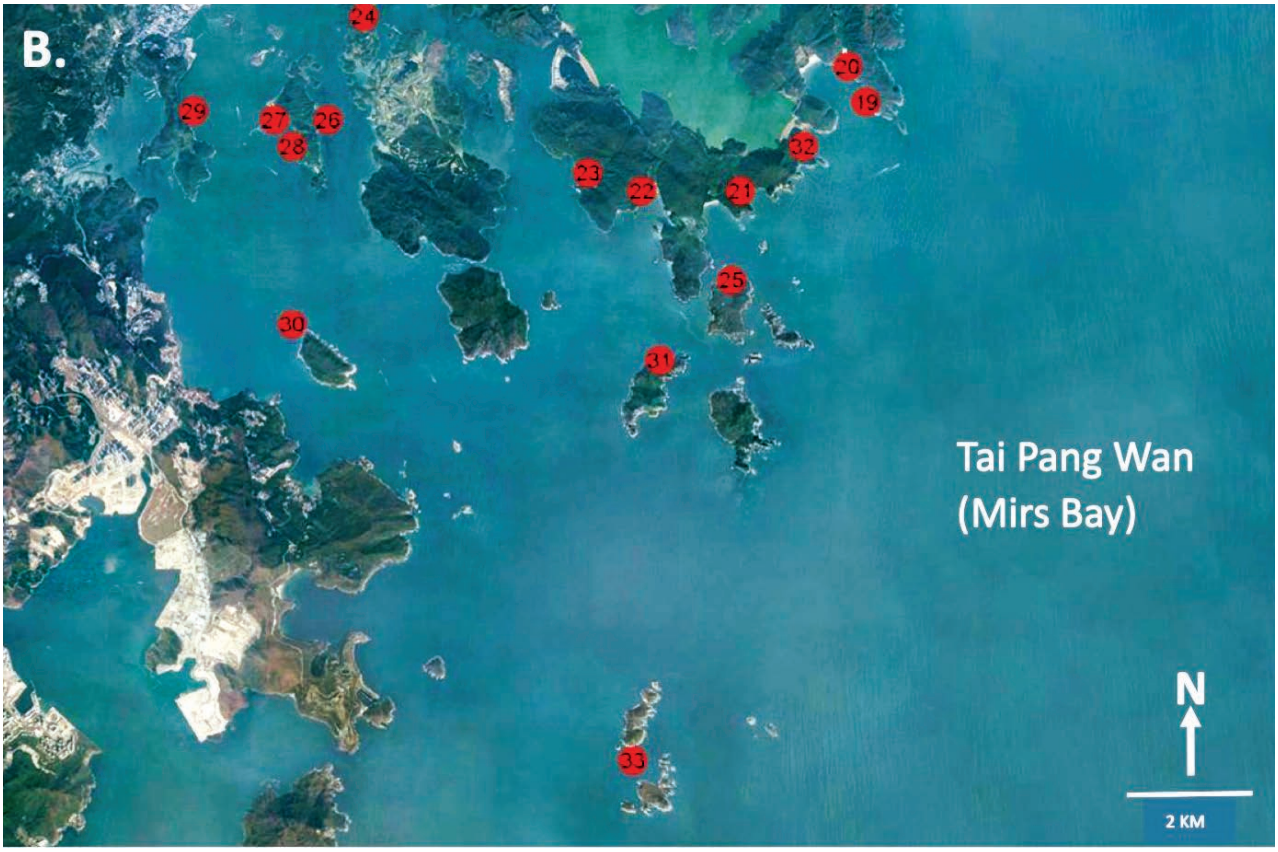


FIGURE 1: *Location of 33 Reef Check sites in Hong Kong. (A) These sites are mainly located in the northeastern Hong Kong waters (17 sites), including Reef Check sites in the three marine parks designated in blue; and (B) eastern Hong Kong waters (16 sites). Refer to Table 1 for the full names of the survey sites as indicated by the site number. Modified from Source: Agriculture, Fisheries and Conservation Department.*[3]

[3] *https://www.afcd.gov.hk/english/conservation/con_mar/con_mar_cor/con_mar_cor_hkrc/con_mar_cor_hkrc5.html*

TABLE 1: *Changes in coral cover from 2015 to 2020 as recorded by Reef Check at 33 sites in Hong Kong. Nine of these 33 sites are located in marine parks and are designated in bold. Refer to Figure 1 for the location of these sites in Hong Kong waters. Source: Agriculture, Fisheries and Conservation Department.*[4]

SURVEY SITES	CORAL COVER (%)					
	2015	2016	2017	2018	2019	2020
1. A Ma Wan, Tung Ping Chau	67.9	63.8	57.5	60.7	60.0	56.9
2. A Ye Wan, Tung Ping Chau	45.6	42.9	41.9	42.4	42.0	41.6
3.Wong Ye Kwok, Tung Ping Chau	63.4	64.6	58.5	53.4	43.0	39.9
4. Kai Kung Tau, Kat O	19.5	17.5	18.5	15.9	19.4	22.5
5. Tau Tun, Kat O	25.3	21.4	20.2	21.3	20.6	29.0
6. Lai Chi Wo, Yau Chau Tong	54.7	56.2	54.4	44.7	44.0	51.3
7. Au Yue Tsui, Yau Chau Tong	49.1	53.1	52.8	51.7	57.5	59.4
8. Ngau Shi Wu Wan	35.1	32.6	42.1	30.4	28.3	30.3
9. Wu Pai, Crescent Island	67.2	71.3	63.9	53.5	43.1	52.1
10. Crescent Island West	74.5	75.8	76.7	75.7	60.3	51.7
11. Crescent Island South	68.1	69.2	72.3	69.9	68.1	58.1
12. Tung Wan, Double Island	22.2	19.9	18.8	21.9	18.1	27.1
13. Wong Chuk Kok Hoi	23.4	18.4	16.4	13.3	10.0	15.6
14. Chek Chau (Port Island)	52.6	44.7	29.8	21.4	14.6	13.8
15. Moon Island, Hoi Ha Wan	32.7	9.3	11.4	13.0	24.4	25.4
16. Coral Beach, Hoi Ha Wan	72.3	65.9	66.3	61.9	55.0	66.2
17. Pier, Hoi Ha Wan	73.2	62.5	50.3	42.5	68.8	63.5
18. Gruff Head, Hoi Ha Wan	68.4	65.3	62.3	62.3	56.2	54.9
19. Long Ke Wan	46.8	47.9	55.0	49.9	39.4	35.0
20. Siu Long Ke	39.2	40.6	42.5	48.1	51.9	43.1
21. Pak Lap Tsai	54.2	55.2	54.7	54.7	53.1	55.0
22. Pak A	48.4	45.9	48.9	47.7	58.8	62.9
23. Tai She Wan	55.9	57.1	59.1	57.4	59.4	61.9
24. Tai Mong Tsai	65.9	64.2	75.0	65.1	76.3	70.7
25. Town Island	55.7	57.9	58.8	57.5	45.5	50.9
26. Sharp Island East	69.4	69.8	68.0	69.2	73.8	83.8
27. Sharp Island North	79.5	82.8	83.5	78.1	82.5	76.4
28. Sharp Island South	33.6	29.4	31.5	31.5	50.0	50.3
29. Pak Ma Tsui	46.8	38.3	42.8	34.2	38.1	36.3
30. Shelter Island	61.1	63.5	60.6	50.6	31.2	38.1
31. Bluff Island	75.9	78.6	79.5	64.4	81.3	72.5
32. East Dam	52.2	54.6	53.4	53.4	44.8	55.6
33. South Ninepin	24.2	26.4	25.3	23.8	24.4	23.1

[4] *https://www.afcd.gov.hk/english/conservation/con_mar/con_mar_cor/con_mar_cor_hkrc/con_mar_cor_hkrc5.html*

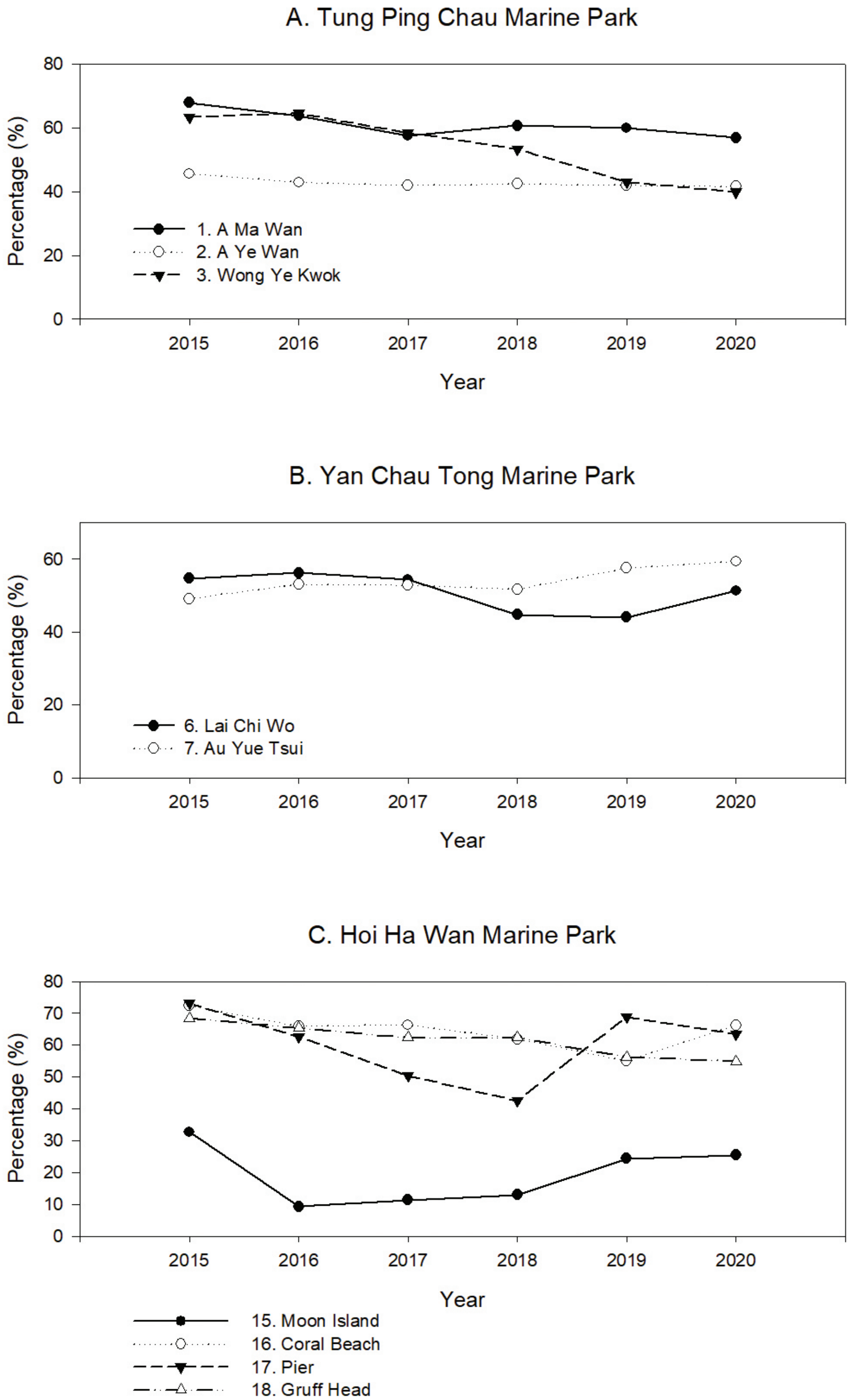
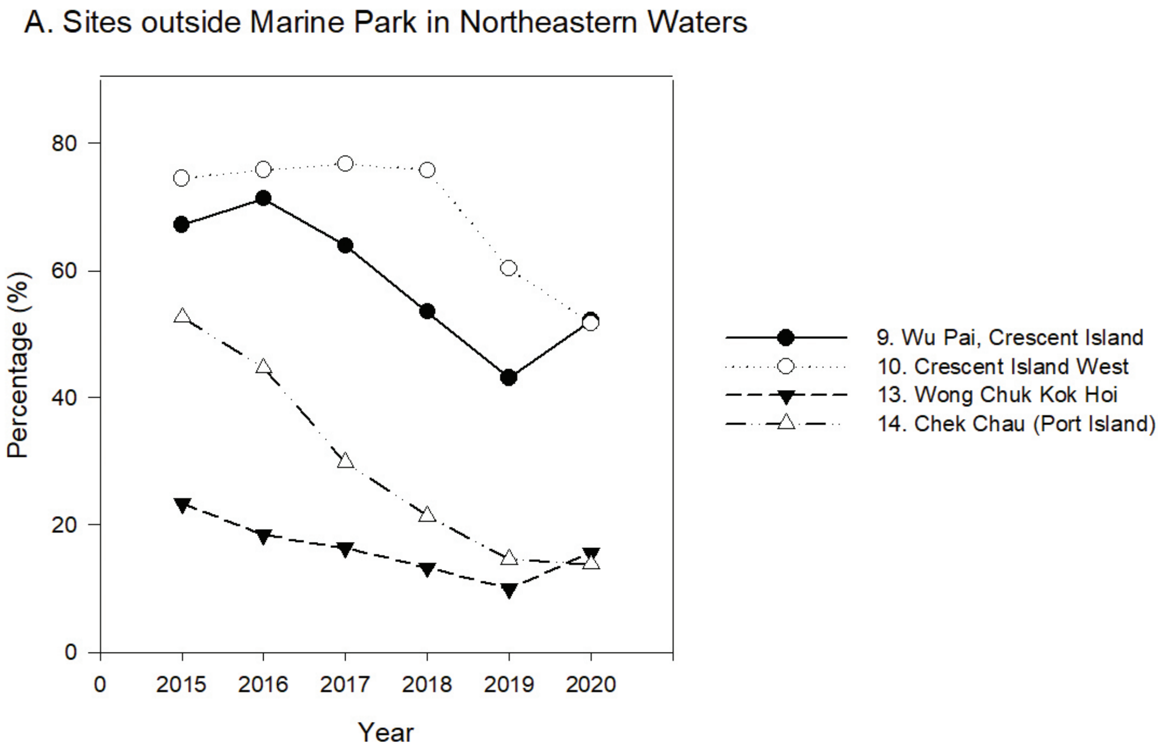


FIGURE 2: *Recorded changes in percentage coral cover in sites within the three marine parks, with some sites recording decline in coral cover. Some recovery, however, was also recorded in some other sites. Note different scales in the Y-axes.*



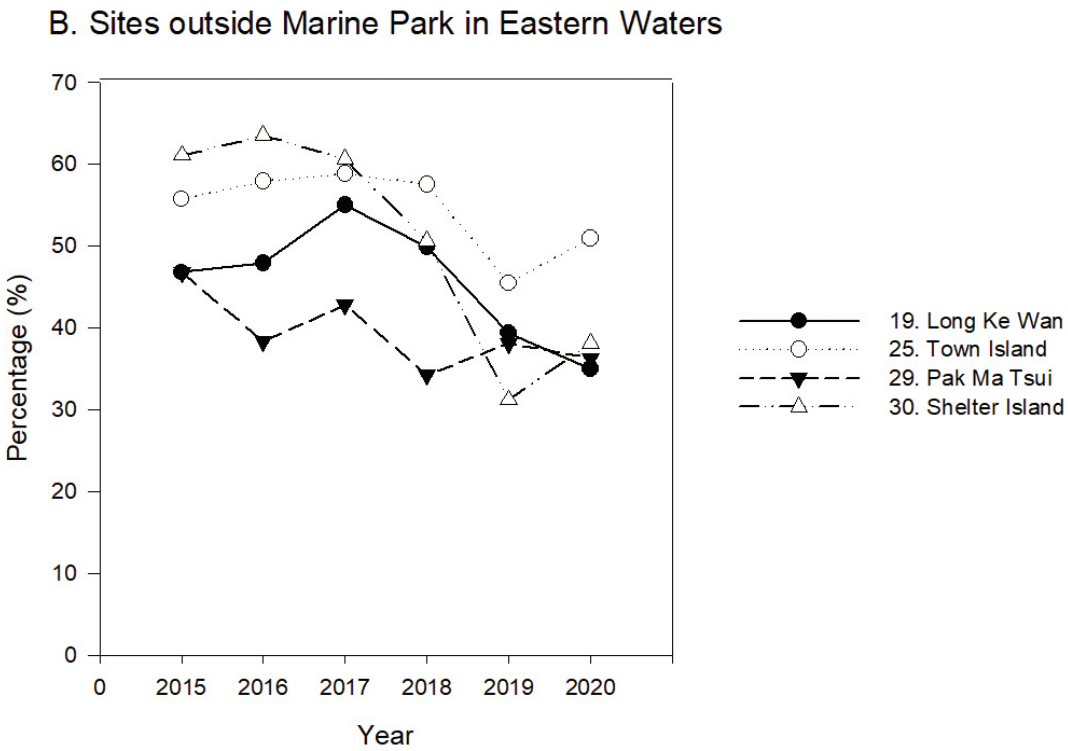


FIGURE 3: *Sites in northeastern (A) and eastern (B) Hong Kong waters that experienced more than 10% decline in coral cover from 2015 to 2019. Note different scales in the Y-axes.*

MONITORING OF CORAL DIVERSITY & PERCENTAGE COVER IN MARINE PARKS

Brief Historical Background

Historically, the first detailed survey on coral diversity in Hong Kong was carried out by Thompson and Cope in 1979–80 in Hoi Ha Wan located in the northeastern waters of Hong Kong, bordered by Tolo Channel and Tai Pang Wan (Mirs Bay) (Figure 1A) (Thompson and Cope 1982). They laid three transects and listed 26 species of corals. They reported *Leptastrea purpurea* to have dominated the shallow water area (-0.8 to -1.3 m below Chart Datum [CD]) with a living coral cover of around 70% to 100%. *Lithophyllon* cf. *edwardsi* (now *Lithophyllon undulatum*) and *Stylocoeniella guentheri* were the main species between -1.3 to -2.3 m CD. A more detailed assessment of Hoi Ha Wan coral diversity was later carried out in 1980 by Cope and Morton (1988) using line transects. Twenty-six species of corals were encountered in the transects, although 36 species were identified from the whole site. The percentage of live coral cover ranged from 51% in Sam Po Rock to 75% in Coral Beach. Dominant species included *Platygyra sinensis* (now recognized as *P. acuta/ P. carnosa*), *L. purpurea, Favites pentagona, Cyphastrea* spp., *Porites lobata,* and *Pavona decussata*.

In a re-survey carried out by Scott and Cope (1989) in 1986, they found a significant drop in the coral cover from 61% to 34–36% along the two original transects laid by Thompson and Cope in 1979–80 (Thompson and Cope 1982). Zou et al. (1992) re-surveyed Cope and Morton's (1988) original transects nine years later in 1989 and found a general decline in the number of species and species cover along most of these transects. The coral cover now ranged from 8% to 75%, with 24 species of corals recorded. There was a shift in dominant species from *Platygyra rustica* (identified as *P. sinensis* earlier) to *P. decussata,* and from *Hydnophora exesa* to *L. purpurea*. Pollution from Tolo Harbour / Tolo Channel outside Hoi Ha Wan, and the increase in run-off from the excavation of Tai Leng Tun borrow for filling materials in the reclamation of Ma On Shan in Tolo Harbour, were attributed as the most likely causes for the deterioration of the water quality in Hoi Ha Wai that led to decline in coral cover. Hoi Ha Wan continued to experience environmental disturbances in the early 1990s. In 1994, heavy rainfall resulted in a layer of water with low temperature and low dissolved oxygen staying at -2 m CD for nearly two months in Tai Pang Wan, including Hoi Ha Wan (Binnie 1995). This serious hypoxia event affected the corals of Hoi Ha Wan and caused up to 83% coral mortality (Wilson 1994).

Hoi Ha Wan and Yan Chau Tong were subsequently designated as the first two Hong Kong Marine Parks in 1996. With the help of other works initiated by different universities and NGOs, the Marine Parks Division of the AFCD recognized the need to re-survey the coral communities in Hoi Ha Wan and other marine parks in order to monitor the effectiveness of a marine park designation on the conservation and protection of corals.

Yan Chau Tong, located in the northeastern waters of Hong Kong (Figure 1A), was designated as a marine park to protect its mangrove and seagrasses. Hence, there were only minimal studies on the coral communities within this marine park. Only one site within the marine park, Au Ye Tsui, supported a significant abundance of corals. Another site near the northern edge of the marine park, Lai Chi Wo, was also found to support good coral cover.

Tung Ping Chau (Figure 1A) is an island located northeast of Hoi Ha Wan and Yan Chau Tong. It supports one of the largest coral communities in Hong Kong. The AFCD commissioned a feasibility study for its designation as a marine park that was carried out by the Marine Science Laboratory of the Chinese University of Hong Kong (MSL CUHK) in 1998–1999 (Ang et al. 2000). The area around the island was subsequently established as the fourth Hong Kong marine park in 2001. In this feasibility study, coral surveys were carried out at two sites on the northeast side of the island, A Ma Wan and A Ye Wan. These two sites already had permanent transects set up in 1998 by the MSL CUHK to monitor their coral health (Tam and Ang 2008).

Systematic Survey of Coral Cover & Diversity Within Marine Parks

The AFCD first funded a more extensive survey of corals in Hoi Ha Wan carried out by Oceanway Corporation Ltd. in 2001–2002. In their report (Oceanway 2002), 59 coral species were recorded, and high live coral covers >50% were recorded in many of the shallow-water sites surveyed. Subsequent standardization of the taxonomic identification of corals found in Hong Kong

(Ang et al. 2003) narrowed down the number of coral species recorded in Hoi Ha Wan to 52. From 2003 to 2005, at least three surveys were commissioned by the AFCD to monitor changes in coral biodiversity and percentage cover within Hoi Ha Wan, Yan Chau Tong, and Tung Ping Chau Marine Parks (Ang et al. 2004a, 2004b, 2006a, 2006b). Data collected from these surveys formed part of the baseline information. Unfortunately, similarly detailed monitoring was not followed up in the subsequent 10 years. A tender invitation was sent out in 2009 by the AFCD to start another round of coral surveys in the marine parks, but this tender was later withdrawn. It was not until 2015 when another survey was commissioned by the AFCD and carried out by AECOM, a consultant company, to conduct coral monitoring in the three marine parks. Another round of coral monitoring carried out by the City University of Hong Kong in the three marine parks started in 2019. This latest round is expected to be completed soon. Apart from coral surveys, the AFCD has also commissioned other studies covering algae, fish, and octocoral diversity and distribution in Hong Kong waters both within and outside the marine parks. Other more topic-oriented projects are also being carried out, including coral bleaching and bioerosion, coral restoration, and most recently, public education, in 2020.

Detailed coral monitoring data are currently available from the following studies as listed in chronological order:

- **1998–1999:** feasibility study to establish Tung Ping Chan as a marine park (Ang et al. 2000)
- **2003–2004:** coral monitoring in Hoi Ha Wan and Yan Chau Tong Marine Parks (Ang et al. 2004a)
- **2003–2004:** biological and coral monitoring in Tung Ping Chau Marine Park (Ang et al. 2004b, 2006a)
- **2004–2005:** ecological monitoring in Hoi Ha Wan and Tung Ping Chau Marine Parks (Ang et al. 2006b)
- **2015:** hard coral surveys at Hoi Ha Wan, Tung Ping Chau, and Yan Chau Tong Marine Parks (AECOM 2016)

While these studies covered different sites within the marine parks, a comparison of changes in coral cover can be made between the following sites covered in most of these studies:

- Two sites: A Ye Wan and A Ma Wan in Tung Ping Chau Marine Park (TPCMP) (Figure 4)
- Two sites: Au Yue Tsui and Lai Chi Wo in Yan Chau Tong Marine Park (YCTMP) (Figure 5)
- Four sites: Gruff Head, Moon Island, Coral Beach, and Pier in Hoi Ha Wan Marine Park (HHWMP) (Figure 6)

The feasibility study for TPCMP (Ang et al. 2000) and the subsequent monitoring study (Ang et al. 2004b) in the two study sites made use of permanent transects laid perpendicular to the shore and employed line intercept method to assess coral diversity and cover. All other monitoring studies employed both random point transect method and line intercept method on ten 10-m transects laid parallel to the shore, with the coral zones divided into shallow (-1 to -3 m CD) and deep (-4 to -7 m CD) areas. However, to make the data comparable, only line intercept transect data from shallow water sites are used in the current update.

For the line intercept method, a video was taken up-close along the transect, and any corals intercepted by the transect were recorded. The length of the transect intercepted by the corals was analyzed in the laboratory using an image analyzer. A videotape of the line transect was played back, and the length of transects intercepted by each species of coral was recorded. The percentage cover of coral species was calculated as the transect length intercepted by each species over the total transect length.

The monitoring period was divided into two seasons, wet season (late spring to summer months) and dry season (fall to winter months). Data from studies carried out earlier showed that the coral communities tended to be covered with lots of floating seaweeds during winter, mainly *Hypnea*, *Ulva*, and *Sargassum* spp. Therefore, winter data tended to show a lower cover of corals, as these floating seaweeds mostly covered the base of many massive corals. This abundant floating seaweed phenomenon was no longer observed in recent years, with the mean winter water temperature becoming warmer from 14°C to 16–17°C.[5] Thus, only wet season or summer data are used in the present comparative study to avoid biased assessment of coral cover change due to the absence of these floating seaweeds.

[5] Hong Kong Environmental Protection Department *https://www.epd.gov.hk/epd/english/environmentinhk/water/hkwqrc/waterquality/marine.html*

FIGURE 4: *Map of Tung Ping Chau Marine Park (TPCMP) showing the two sites, A Ye Wan and A Ma Wan, that were surveyed for coral cover in more detail over the years. See Figure 1A for the location of TPCMP relative to other Reef Check sites in northeastern Hong Kong. Map modified from Ang et al. 2006b.*

FIGURE 5: *Map of Yan Chau Tong Marine Park (YCTMP) showing the two sites, Au Yue Tsui and Lai Chi Wo, that were surveyed for coral cover in more detail over the years. See Figure 1A for the location of YCTMP relative to other Reef Check sites in northeastern Hong Kong. Map modified from Ang et al. 2004a.*

FIGURE 6: *Map of Hoi Ha Wan Marine Park (HHWMP) showing the four sites, Gruff Head, Moon Island, Coral Beach, and the Pier, that were surveyed for coral cover in more detail over the years. See Figure 1A for the location of HHWMP relative to other Reef Check sites in northeastern Hong Kong. Map modified from Ang et al. 2004a.*

Changes in Coral Diversity & Mean Percentage Coral Cover in Hong Kong Marine Parks

Tung Ping Chau Marine Park (TPCMP) (Figure 4)

In A Ye Wan, the dominant coral species, *Platygyra acuta*, remained stable before and after the establishment of TPCMP. It was consistently the most dominant species from 1999 to 2015, and its percentage cover increased from 4% to 5-6% (Figure 7A). However, this increase appeared to have slowed after 2005, with no apparent difference observed between 2005 and 2015. More variable patterns of change can be said of the other dominant species, including *Leptastrea purpurea*, *Favites flexuosa*, and *Porites lutea*, and their ranking order of dominance also changed over time. *Dipsastrea speciosa* was one of the top 10 dominant species in 1999 and 2003, but it was replaced by *Dipsastrea rotumana* in 2005 and 2015. As these two species look very similar in appearance, some misidentification by different surveyors cannot be ruled out.

In A Ma Wan, the situation was comparable; *P. acuta* and *Pavona decussata* were consistently the most dominant species from 1999–2015 (Figure 7B). The coral cover of *P. acuta* clearly increased after the marine park was established and remained stable through 2015. Other dominant species included *P. lutea*, *Goniopora lobata / G. columna*, *L. purpurea*, *Cyphastrea serailia*, *Hydnophora exesa*, and a few other *Platygyra* species. Although their ranking order may change over time, they remained one of the top 10 dominant species. Again, the replacement of *G. lobata* by *G. columna* in later years may simply be another case of misidentification.

For both sites in TPCMP, massive and submassive corals appeared to be the main structural and foundation species. The barge accident that crushed many large colonies of *Platygyra* spp. in A Ye Wan in the late 2000s did not appear to have changed the dominance of *Platygyra* in this site.

Yan Chau Tong Marine Park (YCTMP) (Figure 5)

Au Yue Tsui is the main site within YCTMP with a reasonable abundance of corals. On the other hand, Lai Chi Wo is located closer to the edge of the marine park. There were no pre-marine park data to be compared with, so assessing the pattern of change in these sites could only be based on two data points collected in 2003 and 2015.

Porites lutea was the most dominant coral in Au Yue Tsui in 2003, but this was replaced by *L. purpurea* in 2015 (Figure 8A). A very notable increase in the cover of *L. purpurea* was recorded, from 5.1% in 2003 to about 15% in 2015. There was also a large increase in the cover of *C. serailia* over this same period, whereas the cover of *P. lutea* dropped from 11% to about 4%. The other top 10 dominant corals generally remained the same, including *Porites lobata*, *Cyphastrea chalcidicum*, *L. undulatum*, *P. decussata*, and *G. columna*, with a slight increase in their overall cover.

In Lai Chi Wo, seven of the top 10 dominant species recorded in 2003 remained in the top 10 in 2015 (Figure 8B). *Platygyra acuta* remained the most dominant species, albeit with a drop of about 5% in its cover. The same was true for the other dominant species, where a lower cover in 2015 was generally observed.

Among all the study sites located within the marine parks, Au Yue Tsui and Lai Chi Wo are closest to Yim Tin Port of Shenzhen to the north. This port saw massive reclamation and development in the last 20 years, resulting in heavy sedimentation in the surrounding waters. It is perhaps not surprising to detect a general decline in the cover of most of the dominant coral species within these two sites.

Hoi Ha Wan Marine Park (HHWMP) (Figure 6)

As described earlier in the brief historical background, corals in Hoi Ha Wan have been monitored for close to 40 years, and the coral cover decline was already detected back in the 1980s, and this trend appeared to continue.

In Gruff Head, there was a general decline in the cover of the dominant species (Figure 9A). *Platygyra acuta* was most dominant from 2003-2005 with a mean percentage cover of 7% to 10%; it was replaced by *L. purpurea* in 2015. This latter species increased in cover from around 4% in 2003 to about 6% in 2015, whereas *P. acuta* dropped to <3% in 2015. There was also a noticeable change in the composition of the other less dominant species, with *Favites flexuosa, P. decussata, Montipora turgescens, C. serailia*, and *Acropora digitifera* becoming more dominant, replacing *L. undulatum, P. lutea, G. columna,* and other *Platygyra* species.

In Moon Island, a similar decline in the dominance of *Platygyra* spp. was also observed as early as 2005, with its dominance replaced by *P. decussata* (Figure 9B). Interestingly, one other species, *Echinophyllia aspera,* appeared to assume dominance over the years too. However, the other less dominant species, including *L. purpurea, P. lutea, G. columna, G. lobata,* and *L. undulatum,* persisted and maintained their presence in 2015.

A big patch of *P. decussata* dominated the shallow water area of Coral Beach, making this species the most abundant in this site (Figure 9C). In the 1980s, large colonies of *Platygyra* spp. were also reported to be characteristics of this site. Serious bioerosion by sea urchins decimated many of these massive coral colonies, but *P. decussata* remained unaffected. Although changes of the other less dominant species were recorded between 2005 and 2015, the extreme dominance of *P. decussata* makes these changes hardly detectable.

Lithophyllon undulatum was dominant in the Pier area during the 1980s, but a rapid decline in cover was detected between 2003 and 2005 (Figure 9D). Although any further decline appeared to have slowed down from 2005 to 2015, increased growth of *P. decussata* eventually led to this latter species becoming more dominant. A similar pattern could be observed for *L. purpurea*. This species was already the second most dominant coral in 2003. Its cover also dropped in 2005, but apparently, it recovered in 2015 to rank higher than *L. undulatum*. The cover of some of the less dominant species remained unchanged over the years, albeit with changes in their ranking order. These included *Leptastrea pruinosa, G. lobata, Plesiastrea versipora, P. lutea, D. rotumana, Coelastrea aspera,* and *S. guentheri.*

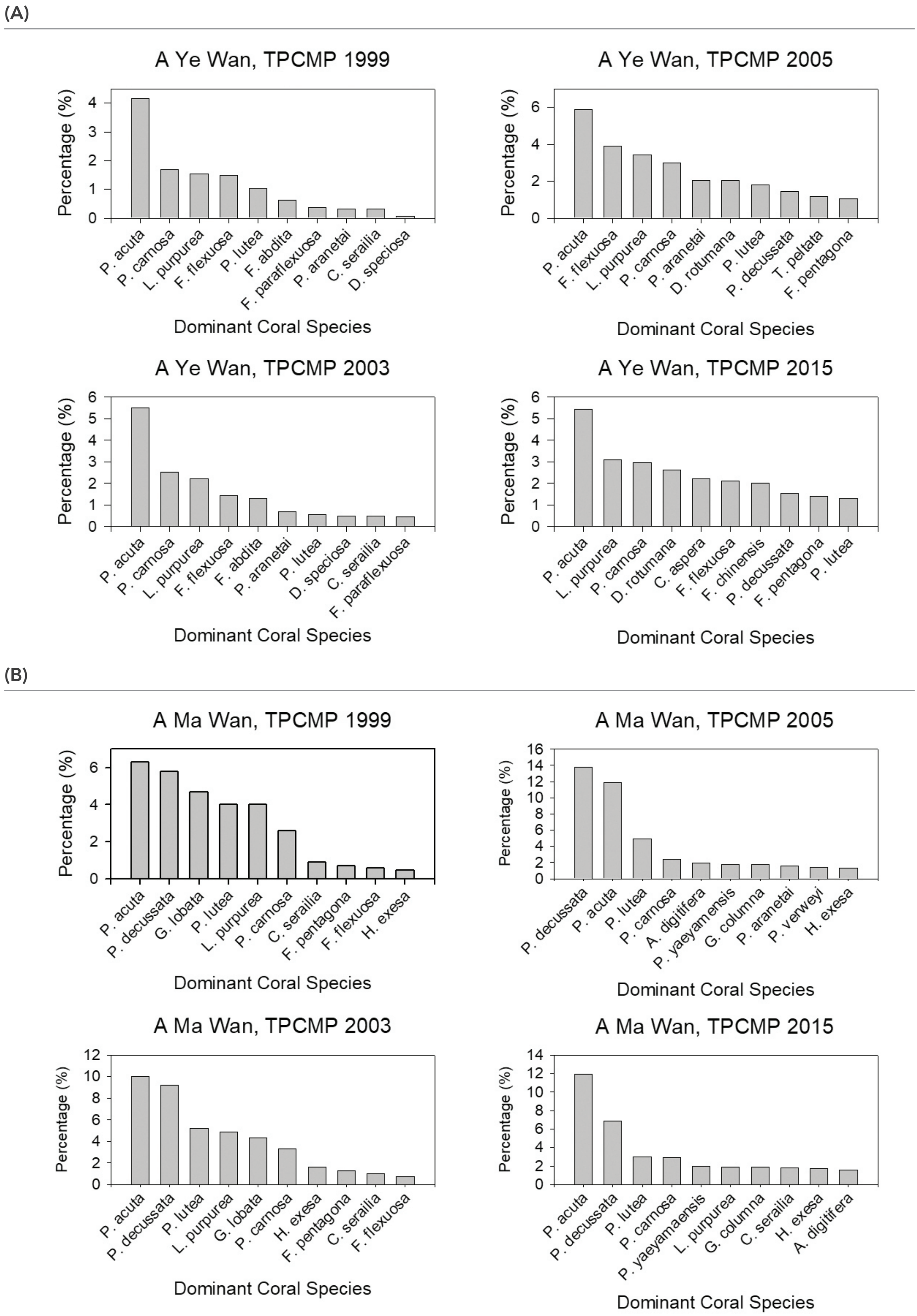


FIGURE 7: *Changes in the mean percentage cover of the top 10 dominant coral species in Tung Ping Chau Marine Park from 1999 to 2015; A. A Ye Wan and B. A Ma Wan. SD not shown. Note different scales in the Y-axes. Replotted based on data from Ang et al. (2000, 2004b, 2006b), AECOM (2016).*

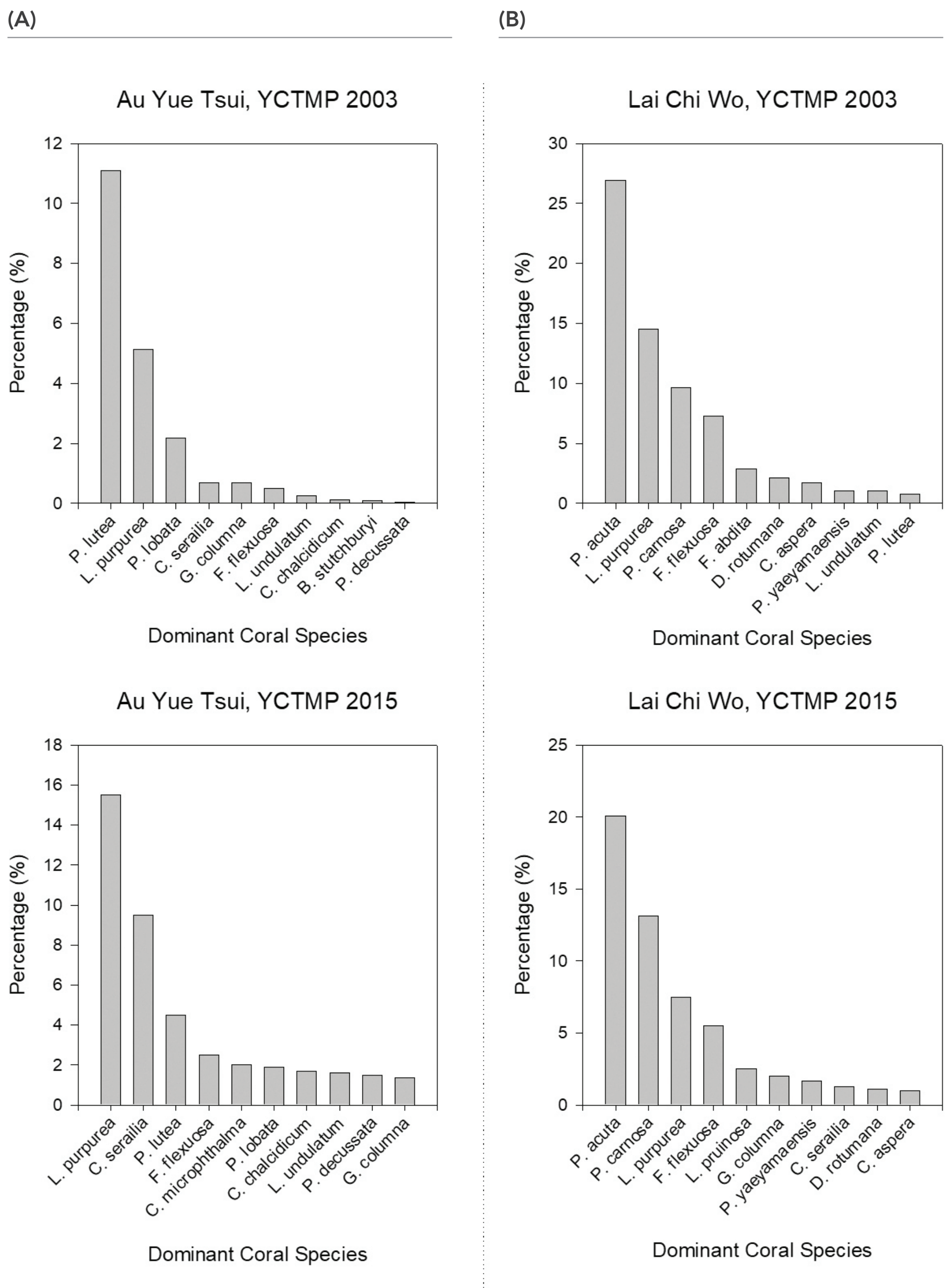


FIGURE 8: *Changes in the mean percentage cover of the top 10 dominant coral species in Yan Chau Tong Marine Park from 2003 to 2015; A. Au Yue Tsui and B. Lai Chi Wo. SD not shown. Note different scales in the Y-axes. Replotted based on data from Ang et al. (2004a), AECOM (2016).*

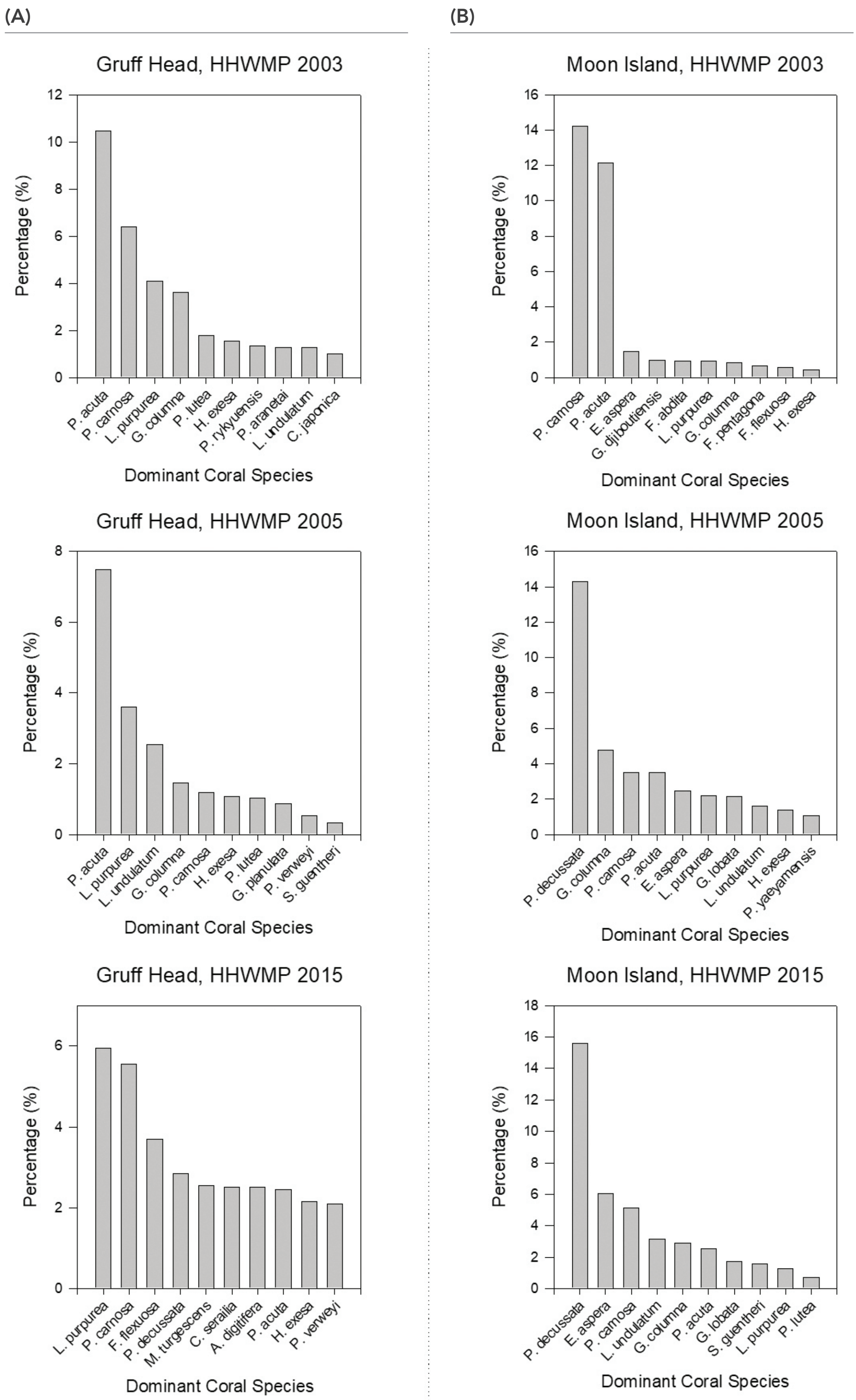


FIGURE 9: *Changes in the mean percentage cover of the top 10 dominant coral species in Hoi Ha Wan Marine Park from 2003 to 2015; A. Gruff Head and B. Moon Island. SD not shown. Note different scales in the Y-axes. Replotted based on data from Ang et al. (2004a, 2006b), AECOM (2016).*

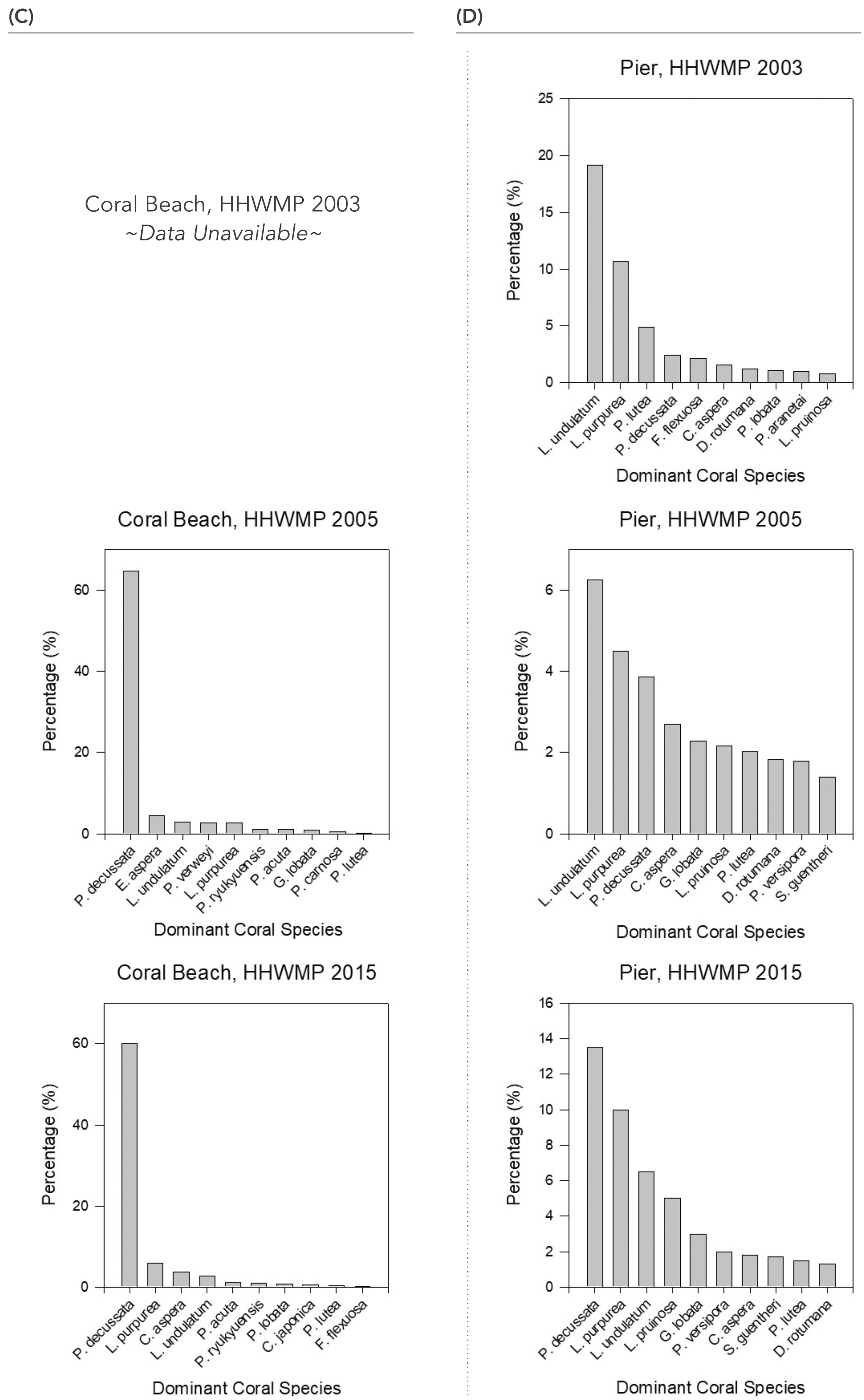


FIGURE 9 *(Cont'd). Changes in the mean percentage cover of the top 10 dominant coral species in Hoi Ha Wan Marine Park from 2003 to 2015; C. Coral Beach and D. Pier. SD not shown. Note different scales in the Y-axes. Replotted based on data from Ang et al. (2004a, 2006b), AECOM (2016).*

Summary of Comparative Study

Among the three marine parks, the overall mean coral cover in TPCMP appeared to have increased from 1999 to 2015 (Figure 10A). The dominant species remained unchanged but with changes observed in their ranking order of dominance (Figure 7). As data before the marine park establishment in 2001 were available, a comparison of pre- and post-marine park data suggests some positive marine protection effects on the coral cover. However, this effect was only apparent in the first few years after protection, with no obvious difference observed from 2005 to 2015. Some notes should be taken when looking at the data collected in 1999 and 2003 for A Ye Wan and A Ma Wan. The big increase in coral cover, especially in A Ye Wan between 2003 and 2005, is likely due to differences in the sampling efforts. These earlier data collected in 1999 and 2003 were based on transect lines laid perpendicular to the shore and were extended to deeper water covered with a large patch of sandy bottom. Thus, this significantly lowered the mean total coral cover in this site. In contrast, data from 2005 and 2015 were based on transects laid parallel to the shore in shallow areas (-1 to -3m CD) and hence did not extend to the sand patch. This discrepancy in the sampling effort is less of a problem in A Ma Wan as coral distribution is more uniform. Nonetheless, in either approach, an increase in the percentage cover of coral was detected.

Albeit limited, it is good to observe the positive effect of protection on the coral cover in this marine park. Nonetheless, serious caution should be taken as extensive sea urchin grazing on corals, particularly on the dominant *Platygyra* spp., has been observed recently (2020), especially in A Ma Wan. A similar event occurred in HHWMP during the 1990s that contributed to the severe decline in the cover and dominance of massive corals. On the other hand, coral recruitment rates in Hong Kong are very low, with <1 ind. m^{-2} (Chui and Ang 2017), so a section of A Ye Wan destroyed by the barge accident mentioned above in the section on Tung Ping Chau Marine Park, remained bare with no new coral colonization to date. This site has been observed to be covered with the brown algae Sargassum spp. during winter, and this micro-phase shift in its benthic community structure could have prevented further recruitment of corals.

YCTMP experienced a general decline in the coral cover of its dominant species, especially in Au Yue Tsui, resulting in a switch of the dominant coral species from *P. lutea* to *L. purpurea*. This decline, however, was not fully reflected in changes in the mean total coral cover (Figure 10B). On the contrary, this site even recorded an increase of 30% in mean total coral cover from 2003 to 2015. This coral cover increase could partly be from most other species other than *P. lutea*, the original dominant species. These increases were indeed very noticeable in the cases of *L. purpurea* and *C. serailia*.

Changes in mean total coral cover in Lai Chi Wo were less detectable (Figure 10B). Although a general decline in the cover of the top few dominant species was also detected, only a slight (<4%) decline in the mean total coral cover was recorded between 2005 and 2015. Perhaps, like the case in Au Yue Tsui, there were increases in the coral cover of the less dominant species, which compensated for the loss from the dominant species.

As mentioned earlier, both Au Yue Tsui and Lai Chi Wo in YCTMP face the Yim Tin Port in Shenzhen to the north, where increasing sedimentation from reclamation work during port development and resuspension of sediments due to the increasing traffic of huge container ships exert significant pressure on the corals at these sites. A more detailed study should be carried out to examine these impacts further.

Among the three marine parks, corals in HHWMP appeared to be impacted the most, with a decline in cover observed in the most dominant species and subsequent replacement by other more tolerant species. However, *P. decussata* remained dominant in Coral Beach since 2005, with a generally increasing trend in total coral cover detected from 2001 to 2015 (Figure 10C). The decline in the massive coral cover had occurred earlier in this site before any of these more detailed survey studies were put in place. Hence, only the dominance of *P. decussata* was detected because of its persistence, with a less discouraging picture of no apparent large decline in total coral cover. In comparison, the mean total coral cover in Moon Island and Gruff Head did not change much, with a <3% increase in Gruff Head and a <2% increase in Moon Island between 2005-2015. Total coral cover declined in the Pier from 55.5% in 2003 to 46.6% in 2005 (Figure 10C) but increased back to 51.9% in 2015.

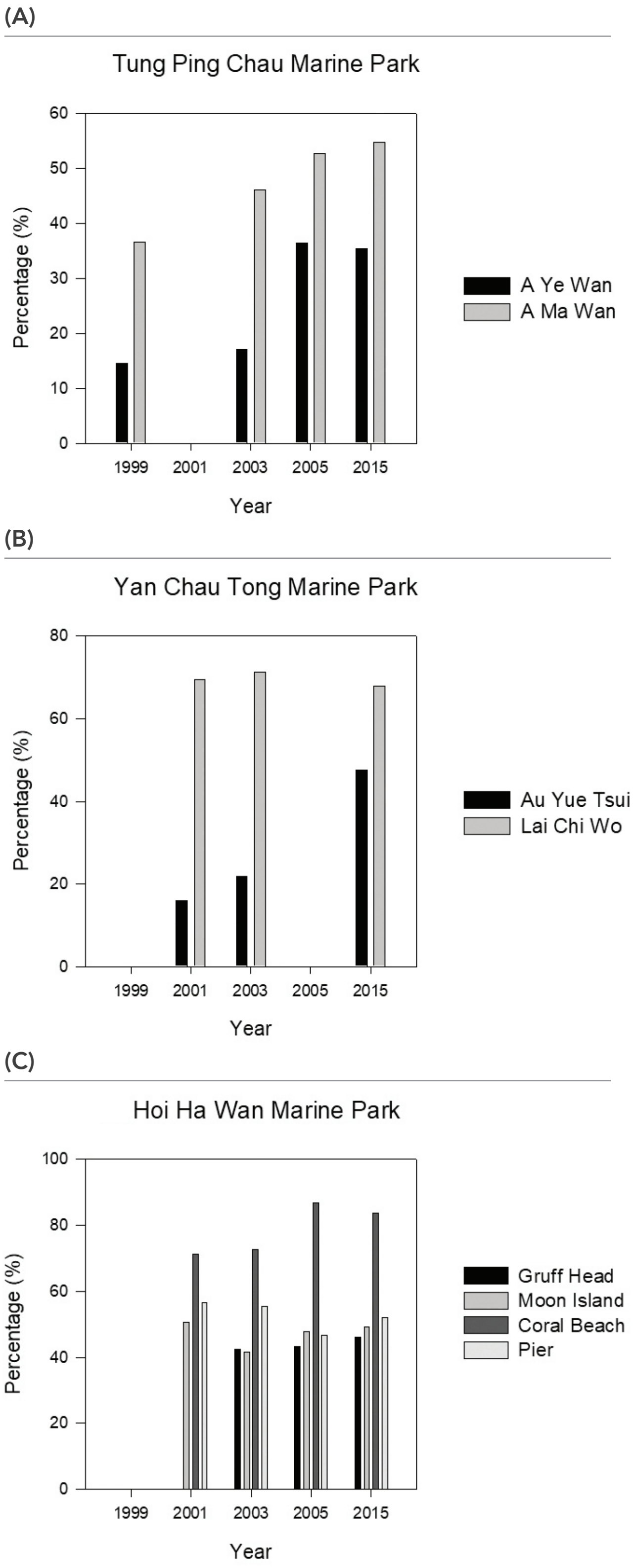


FIGURE 10: *Changes in the mean total percentage cover of live corals in the different sites in the three Hong Kong marine parks over time. SD not shown. Note different scales in the Y-axes. Replotted based on data from Ang et al. (2000, 2004a, 2004b, 2006b); Oceanway (2002), AECOM (2016).*

Overall, species composition in TPCMP appears to be stable, but a switch in dominant species is detected at YCTMP and HHWMP. This switch has compensated for the loss of cover suffered by the original dominant species. *Pavona decussata* and *L. purpurea* have become the most dominant, replacing the massive species like *P. lutea* and *P. acuta* and plate-like species like *L. undulatum*. While *P. decussata* is tolerant to temperature and salinity changes (Tsang 2017), no information is available for *L. purpurea*. This latter species has often been observed to be densely colonized by tubeworms. Yet, it became the dominant species in some of these sites, which indicates its survival and adaptive capability, and warrants further investigation of its physiological tolerances. It is likely to become the most dominant species in Hong Kong, together with *P. decussata,* if the marine environmental conditions in Hong Kong continue to deteriorate.

A major coral mortality event was reported in the early months of 2015 in Hoi Ha Wan, especially at Moon Island and at other sites nearby in Tolo Channel (Wong 2017). Impacts from this event were not reflected in the coral survey study reported by AECOM (2016). However, such decline was recorded by Reef Check (See Figure 2), which suggests the importance of carrying out coral monitoring at closer time intervals, especially in important sites like those within the marine park. A survey to monitor the coral bleaching event in 2017 (Qiu et al. 2019) also reported the total coral cover in Moon Island to be only 11%, far less than reported by AECOM (2016). More updates are needed to reflect a more recent condition of corals in this and other sites.

Coral Bleaching 2017–2018

Mass bleaching of corals has been reported worldwide several times over the last 40 years. While an increase in sea surface temperature has often been suggested as the major cause, there could be other causes as well.

In Hong Kong, bleaching of corals has been observed from time to time. Isolated cases have also been reported in Reef Check surveys. The extent of bleaching is usually small, and no significant coral mortality has been associated with the bleaching. However, a more serious event was documented in 1997 by McCorry (2002). A whole community was reported to have died-off in Sham Wan in the southern waters of Hong Kong and with 15 coral genera in several sites around eastern and northeastern waters being bleached or recently dead. These sites included Coral Beach in HHWMP, Sharp Island West, and TPCMP, where a quantitative line intercept transect survey revealed eight coral genera showing >15% of colonies bleached or recently dead. *Platygyra*, however, was shown to be very resistant, with only 3% of the colonies bleached, compared to *Montipora* (82%), *Goniopora* (64%), *Porites* (33%), *Pavona* (27%), *Stylocoeniella* (33%), *Acropora* (17%), *Hydnophora* (17%), and *Lithophyllon* (17%).

The year 1997 saw heavy annual precipitation of 3,343 mm – the highest in 131 years of weather records by the Hong Kong Observatory.[6] 2,359 mm (or 71%) fell between June and August, lowering the surface water salinity to 19 psu in Hoi Ha Wan and 20 psu in Sharp Island (McCorry 2002). Within these few months, the surface water temperature was lower (26.5°C) than normal because of a prolonged period of heavy cloud cover. However, the other environmental parameters were not exceptionally different from the years before and after 1997.[7] High temperatures were not the cause of bleaching; instead, this massive bleaching was likely caused by reduced salinity due to heavy rainfall.

A comprehensive survey of coral bleaching events was carried out by a joint team of researchers from four universities in Hong Kong, led by Baptist University (Qiu et al. 2019) and commissioned by the AFCD. The presence of any sign of coral bleaching was followed in the 33 Reef Check sites (see Table 1, Figure 1) in summer-early autumn (wet season, June–October 2017) as well as in winter (dry season, December 2017 – March 2018). The survey found bleached corals only in six of these sites, with three inside the marine parks (Coral Beach in HHWMP, A Ye Wan, and A Ma Wan in TPCMP) and three outside the marine parks (Pak Lap Tsai, Sharp Island East, Sharp Island North), mainly between June and August 2017. The major coral genera that exhibited bleaching included *Acropora, Goniopora, Leptastrea, Pavona, Platygyra*, and *Turbinaria*. Except for *Acropora*, at least 95% of the bleached corals recovered within the following three months. *Acropora*, especially those from Pak Lap Tsai (site 21 in Figure 1B), suffered a 30% mortality. No winter bleaching was recorded, and reduced salinity due to higher rainfall was likely to be the major cause of this bleaching event in 2017.

Drivers & Pressures

Although recreational diving activities are increasing in Hong Kong, evaluation of divers' pressure on Hong Kong coral communities has not been recently conducted. On the other hand, the impact of bioerosion and grazing on corals has attracted increasing attention.

In the same study (Qiu et al. 2019), the bioerosion of Hong Kong corals was also monitored. Bioerosion on corals, including grazing / predation on corals by the sea urchin *Diadema setosum* and the snails *Drupella* spp., has been well documented in Hong Kong (e.g., Lam et al. 2007; Tsang and Ang 2015, 2019). Bioerosion by sea urchins has caused the collapse of many massive *Platygyra* colonies in Coral Beach, HHWMP, and is currently threatening the massive corals in A Ma Wan, TPCMP. Results of the 2017 survey, however, did not find a very high density of *D. setosum* in most of the 33 sites, except for two: Wong Chuk Kok Hoi (site 13 in Figure 1A, 8.3 ± 1.35 ind. m^{-2} in December 2017–January 2018) and Chek Chau (site 14 in Figure 1A, 8.0 ± 0.42 ind. m^{-2} in August–September 2017), both in northeastern Hong Kong waters. The density recorded was much higher than "normal" of around 0.1-0.2 ind. m^{-2}, and the mean high density ranging from 1.8 to 6.2 ind. m^{-2} persisted throughout the year. Bioerosion by sea urchins has decimated the massive corals *Platygyra* spp. in Chek Chau and Coral Beach, HHWMP. Wok Chuk Kok Hoi also suffered a decline in coral cover of around 50% from 2015 to 2019 (see Figure 3A). This decline may have been contributed by bioerosion. However, other than these two sites, the other sites that also recorded a large (>10%) decline in coral cover (see Figure 3 and Table 1) did not show a high density of sea urchins. The coral decline in these other sites is likely to have been caused by other stressors.

High mean *Drupella* density was recorded only in Pak Lap Tsai in July–August 2017 at 17.5 ind. m^{-2}, which corresponded to the time with an increased number of bleached *Acropora* colonies in this site. These snails are likely to have been attracted to the highly disturbed and weakened corals, as has been shown previously that healthy corals do not attract *Drupella* (Tsang and Ang 2015, 2019).

[6] *http://www.hko.gov.hk/cis/climat_e.htm*

[7] Hong Kong Environmental Protection Department *https://www.epd.gov.hk/epd/english/environmentinhk/water/hkwqrc/waterquality/marine-2.html*

RECOMMENDATION

There is currently another round of coral monitoring carried out by the City University of Hong Kong, commissioned by AFCD. The World Wildlife Fund Hong Kong (WWF HK) has an Education Centre located inside HHWMP. It is now initiating a coral monitoring study that could hopefully become an annual activity supported by citizen scientists. The Institute of Space and Earth Information Science and the Marine Science Laboratory of the Chinese University of Hong Kong will likely resume their close monitoring of corals in TPCMP initiated since 1998. With all these renewed activities, there should be more detailed information on the general conditions of Hong Kong corals, especially in those sites located inside the marine parks. Reef Check will continue to provide a mechanism for collecting general information on the health status of Hong Kong corals. However, it is important that all these monitoring works should also be backed up by more detailed scientific studies to evaluate the underlying factors or mechanisms responsible for the patterns observed.

DATA CONTRIBUTORS

Data used in this study are obtained from various final reports submitted to the Agriculture, Fisheries, and Conservation Department (AFCD) of the Hong Kong SAR Government as well as from the Reef Check website of the AFCD.

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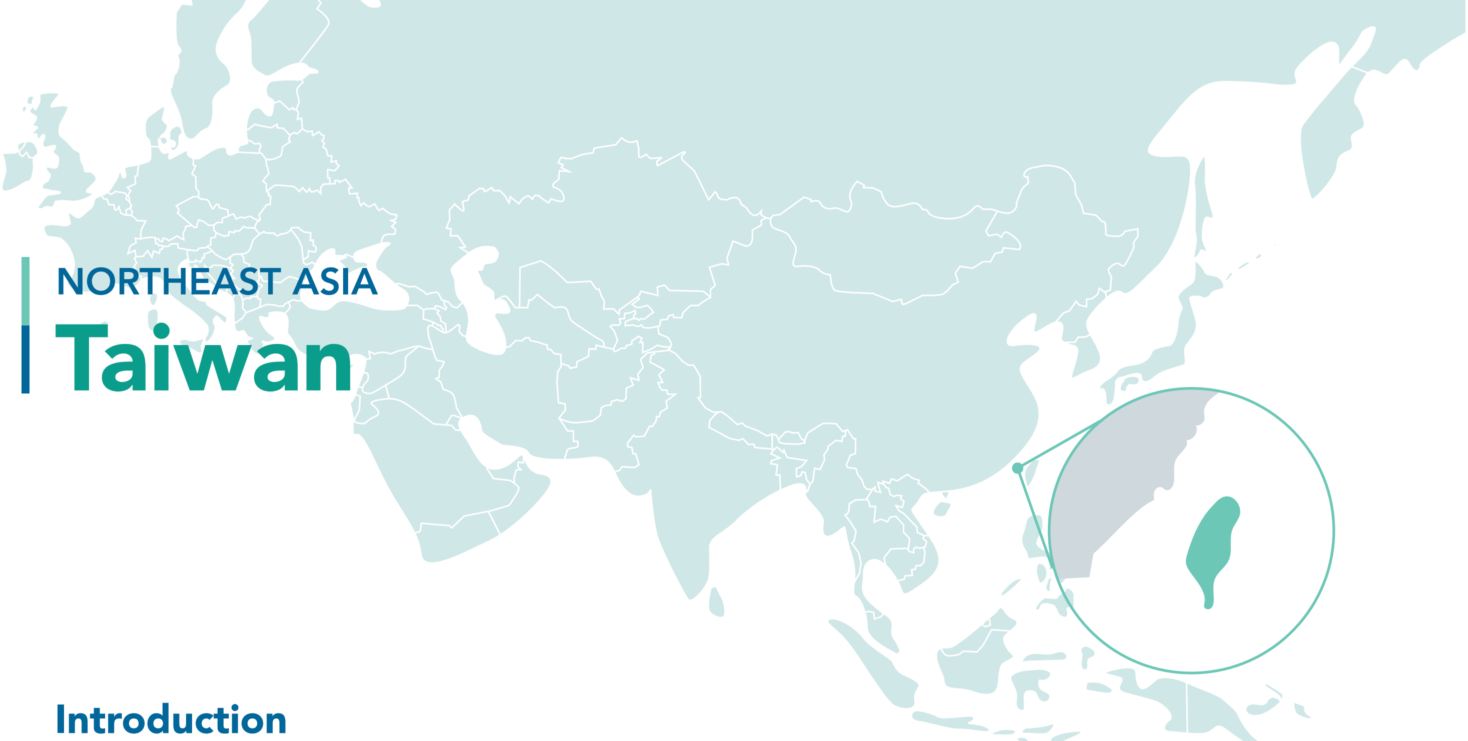
NORTHEAST ASIA

Taiwan



Introduction

Corals and coral communities are widely distributed in the jurisdictional waters of Taiwan, including the main island of Taiwan and offshore islets, such as Pinnacle Islet (Huaping yu, 花瓶嶼), Crag Islet (Miahua yu, 棉花嶼), Agincourt Islet (Pengjia yu, 彭佳嶼), Keelung Islet (基隆嶼), and Turtle Island (Guieshan dao.,龜山島) in the north, Green Island (Lyu dao, 綠島) and Orchid Island (Lan yu/Posunotao,蘭嶼) in the east, the Penghu (Pescadores) Archipelago (澎湖群島) and Liuqiu Islet (Xiaoliuqiu,小琉球) in the Taiwan Strait, and Dongsha Atoll (Pratas) and Taiping Island (Itu Aba) of the Spratly Islands in the South China Sea (Figure 1) (Dai 1997, 2011a, 2011b). Located at the junction between the tropics and subtropics along the Asian Continent surrounded by the Indo-Malaya and Philippine-Ryukyus Archipelagos, the marine environmental condition and distribution of coral species in Taiwan's waters are mainly influenced by climate and ocean currents (Chen and Keshavmurthy 2009, Keshavmurthy et al. 2019). The ocean currents include the Kuroshio Current that flows year-round from the Philippines towards southern Japan, and the South China Sea Surface Current (SCSSC), mainly driven by the southwest monsoon flowing into the Taiwan Strait during the summer. The Kuroshio Current splits into two branches in the Luzon Strait; the main branch consistently flows through the East of Taiwan. The other branch flows into the Taiwan Strait with seasonal variations driven by the southwestern or northeastern monsoon. In the summer, the southwestern monsoon drives the SCSSC through the Taiwan Strait towards the north of Taiwan. In the winter, the warm water of the Kuroshio Current is pushed south by cold, fresh China Coastal Water (CCW) driven by the northeastern monsoon and strapped around southern Penghu in the southern part of the Taiwan Strait (Wang and Chern 1988, 1992, Keshavmurthy et al. 2019).

The distribution and composition of coral species in Taiwan can be broadly divided into two main categories, non-reefal coral communities and tropical coral reefs (Chen 1999, Dai 2018a), due to seasonal variation of sea surface temperature (SST) and currents (Dai 1989, Chen 1999). Non-reefal coral communities include rocky shores in northern Penghu and northern Taiwan, where corals grow on

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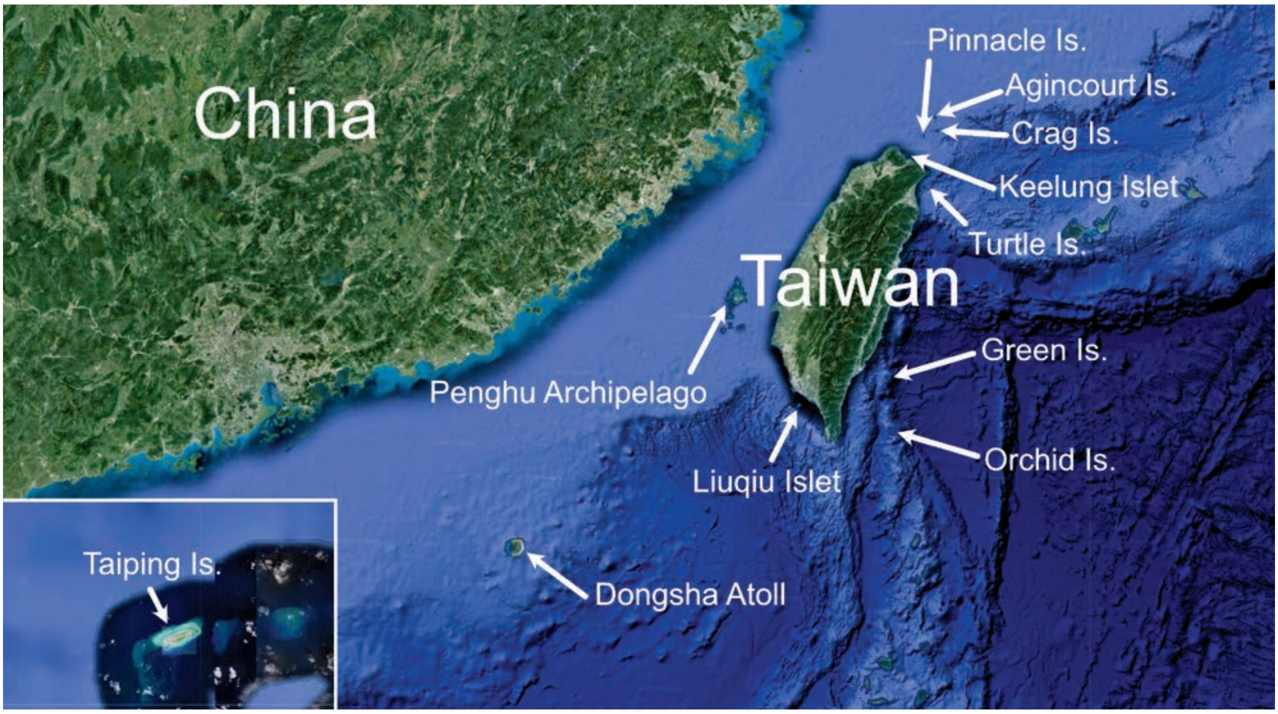

FIGURE 1: *Google map showing the islands of Taiwan.*

the top of volcanic rocks without forming aragonite reef structure due to SST below 18°C (Veron 1995). Tropical coral reefs can be found in southeastern Taiwan and its offshore islands, including southern Penghu, Green Island, Orchid Island, Liuqiu Islet, Dongsha Atoll, and Taiping Island (Chen 1999, 2014, Chen and Keshavmurthy 2009, Dai 2018a).

Seventy-one sites have been surveyed from 1997 to 2003 and 2008 to 2017 (Table 1, Figure 2a). Thirty-five sites are located on the main island of Taiwan, including fourteen along the northeastern coast (Figure 2b), fourteen along the eastern coast (Figure 2c), and seven along the southern coast (Figure 2d). Meanwhile, thirty-six sites are located on offshore islands, including eight at Green Island (Figure 2e), thirteen at Orchid Island (Figure 2f), six at Liuqiu Islet (Figure 2g), and nine at Penghu Archipelago (Figure 2h). However, not all these sites were surveyed annually. In general, around 18 sites were surveyed annually (range: 2–31 sites, median: 19 sites) (Figure 2). The number of surveys conducted at each site ranged from one to seventeen, and the majority of sites (40 out of the 71 sites) were surveyed less than five times. Among the remaining 31 sites, twenty-four sites were surveyed between 5 to 7 times, four sites between 8 to 10 times, two sites between 11 to 13 times, and only one site was surveyed more than 14 times (Figure 2).

All surveys were conducted by three organizations using the point intercept transect (PIT) method at two depth zones around 3–5 m and 9–11 m following the Reef Check principle. In general, the two survey periods, 1997 to 2003 and after 2008, can be divided by the method used as well as the main organization involved. Surveys from 1997 to 2003 were undertaken by the Taiwan Coral Reef Society (TCRS) using one or several 50 m long transects with 20 or 25 cm intervals between points at each depth (Dai 1997, 2005, 2006a, Lee 1998, Dai et al. 2001, 2002, 2003). For the surveys performed by the Coral Reef Evolution Ecology and Genetics Lab, Biodiversity Research Center, Academia Sinica in 2008 (Yang 2008), the Taiwan Association for Marine Environmental Education (TAMEE) in 2009, and Taiwan Environmental Information Association (TEIA) since 2009 (TAMEE and TEIA 2009, TEIA 2010, 2011, 2012, 2013, 2014, 2015, 2016, 2017), four 20-m long transects with 50-cm intervals were used. For maintaining data quality, all benthic community composition data were collected by postgraduate students who were trained and supervised by professors from several universities in Taiwan, as well as by experienced volunteers trained and supervised by graduate students and professors.

TABLE 1: *Sites surveyed from 1997 to 2003 and 2008 to 2017. Each site was numbered along the coastline, and "Site number" refers to the number used in Figure 2. "Times surveyed" refers to the number of repeated surveys conducted from 1997 to 2003 and 2008 to 2017. The site names are translated from Mandarin using the Pinyin System and the names in brackets are indigenous.*

SITE NO.	REGION	SITE	LATITUDE	LONGITUDE	TIMES SURVEYED
1	Northeastern	Yeliu *	25.20884	121.69518	8
2	Northeastern	Jilongyu	25.18931	121.78180	1
3	Northeastern	Chaojing protected area	25.14553	121.80628	1
4	Northeastern	Shenao 1 *	25.13553	121.82023	6
5	Northeastern	Shenao 2	25.13556	121.82194	4
6	Northeastern	Bitoujiao park *	25.12599	121.91481	9
7	Northeastern	Longdong 1	25.11891	121.91948	1
8	Northeastern	Longdong 1.5	25.11581	121.91700	1
9	Northeastern	Longdong 2	25.11394	121.91311	1
10	Northeastern	Longdong	25.11312	121.91923	5
11	Northeastern	Longdong 4 *	25.11345	121.91985	9
12	Northeastern	Hemei	25.07176	121.92268	5
13	Northeastern	Guian	25.02224	121.96249	5
14	Northeastern	Maoao	25.01674	121.99018	5
15	Northeastern	Yinggeshi	24.98539	121.96898	4
16	East	Fanshuliao	24.95121	121.91412	4
17	East	Dofujia	24.58501	121.87286	4
18	East	Neipi	24.57750	121.87310	4
19	East	Xinshe	23.65457	121.54412	1
20	East	Shitiping (Tidaan) harbor north	23.50043	121.50537	6
21	East	Shitiping (Tidaan) *	23.48408	121.51378	3
22	East	Sanxiantai (Cidifangan)	23.12165	121.41159	2
23	East	Jihui (Kihaw) *	23.11636	121.39708	6
24	East	Jiamuziwan (Kamod)	22.86328	121.20712	1
25	East	Xinlan (Alipengan)	22.85886	121.20143	5
26	East	Shanyuan (Fudafudak)	22.83095	121.18858	2
27	East	Shanyuan (Fudafudak) middle reef	22.82972	121.18889	5
28	East	Shanyuan (Fudafudak) southern reef	22.82708	121.19127	4
29	Southern	Jialeshui	21.99610	120.87040	7
30	Southern	Xiangjiaowan	21.92294	120.83222	7
31	Southern	Tiaoshi	21.95320	120.76884	7
32	Southern	Houbihu	21.93806	120.74628	6
33	Southern	Outlet of Nuclear Power Plant	21.93091	120.74442	7
34	Southern	Hejie *	21.95558	120.71189	2
35	Southern	Wanlitong	21.99558	120.70076	4
36	Green Island	Chaikou *	22.67926	121.48277	15

SITE NO.	REGION	SITE	LATITUDE	LONGITUDE	TIMES SURVEYED
37	Green Island	Gongguanbi	22.67971	121.49057	3
38	Green Island	Gongguan	22.67722	121.49258	6
39	Green Island	Jiangjunyan	22.67750	121.49750	5
40	Green Island	Shuimeiren	22.65274	121.50630	1
41	Green Island	Dabaisha *	22.63921	121.49196	2
42	Green Island	Guiwan	22.64289	121.48371	2
43	Green Island	Shilang *	22.65483	121.47351	11
44	Orchid Island	Tankeyan (Ji-Vaheynimanok)	22.08782	121.50535	1
45	Orchid Island	Yunyuyan (Ji-mavonot) *	22.08222	121.51778	4
46	Orchid Island	Langdao (Iraraley) harbour	22.08111	121.52972	1
47	Orchid Island	Langdao (Iraraley) waterworks	22.08288	121.54078	1
48	Orchid Island	Mujiyan (Jyakmi-manok)	22.08306	121.55778	3
49	Orchid Island	Shuangshiyan (Ji-panatoson)	22.08497	121.56876	3
50	Orchid Island	Junjianyan (Do jyaawod)	22.07776	121.57813	1
51	Orchid Island	Rutoushan (do-zako)	22.06621	121.57087	1
52	Orchid Island	Yeyin (Ivalino)	22.04180	121.56826	1
53	Orchid Island	Lanyu Airport (Do sinanap)	22.02373	121.53752	6
54	Orchid Island	Hutoupo (Ji zakazang)	22.03984	121.51983	1
55	Orchid Island	Yeyou (Yayo) *	22.05027	121.50937	4
56	Orchid Island	Tudigongmiao Temple *	22.07051	121.50824	4
57	Liuqiu Islet	Yuchengwei *	22.34928	120.38989	7
58	Liuqiu Islet	Daliaowan	22.33049	120.37123	2
59	Liuqiu Islet	Houshi *	22.32306	120.36556	8
60	Liuqiu Islet	Gebanwan	22.33537	120.35728	2
61	Liuqiu Islet	Duzaiping	22.35022	120.36486	2
62	Liuqiu Islet	Meirendong *	22.35508	120.37432	12
63	Penghu	Qitou	23.64710	119.60365	5
64	Penghu	Shetoushan	23.55148	119.54897	5
65	Penghu	Qingwan outer bay	23.53194	119.55249	5
66	Penghu	Qingwan inner bay	23.52963	119.55988	5
67	Penghu	Tiezhen east	23.27573	119.50494	1
68	Penghu	Xiyuping north	23.27346	119.50741	3
69	Penghu	Dongyuping west *	23.25822	119.51218	7
70	Penghu	Dongyuping east *	23.25614	119.51751	4
71	Penghu	Dongyuping south *	23.25460	119.51241	6

* Identifies sites surveyed by Reef Check Taiwan in 2016.

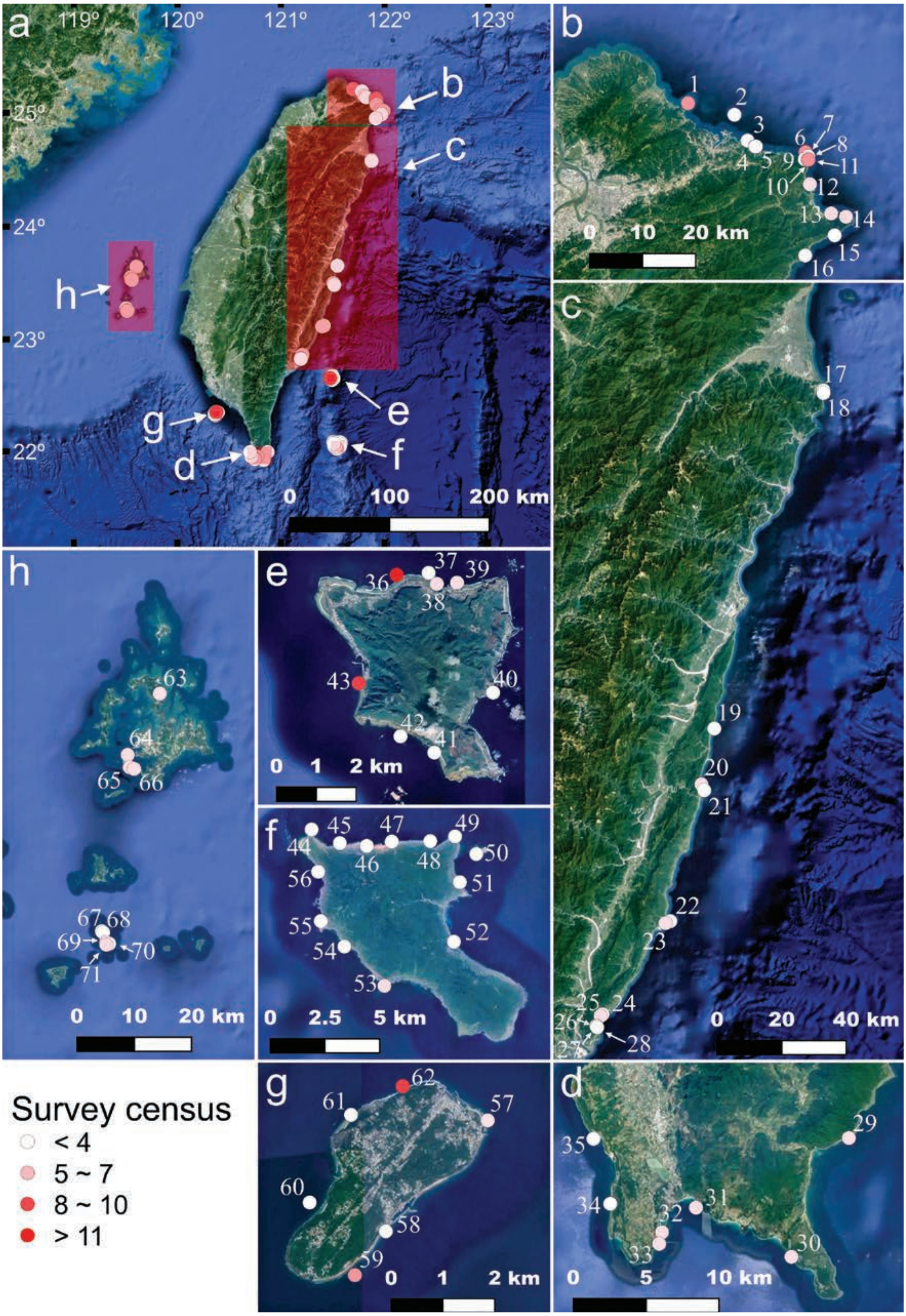

FIGURE 2: *Google maps showing the Reef Check monitoring sites in (a) Taiwan jurisdiction waters, (b) including (b) the northeastern coast, (c) the eastern coast, and (d) the southern coast (Kenting National Park) of the main island of Taiwan, and several offshore islands, such as (e) Green Island, (f) Orchid Island, (g) Liuqiu Islet, and (h) Penghu Archipelago. Each dot represents a survey site listed in Table 1, and the color of each dot indicates the number of repeated surveys conducted from 1997 to 2003 and 2008 to 2017.*

Status & Trends

The mean coral cover in Taiwan decreased from 33% in 1997 to 25.5% in 2017, a loss of about 23%, over approximately 20 years (Figure 3a). However, this was due to damages from multiple major disturbances in 2016. Nonetheless, it was generally stable at around 33%, with the lowest at 25% in 2010 and the highest coral cover at 47.5% in 2004 (Figure 3a). Changes in mean coral cover fluctuated with several periods of decline after disturbances. Two periods of recovery were observed (Figure 3a), while the mean coral cover was stable along the depth gradient from the surface to a depth of 10 m with a slight decline at about 7–8 m (Figure 3b). In all, five significant periods of change could be identified.

1. Between 1997 and 1999, the 1998 mass bleaching caused the decline of coral cover from 33% in 1997 to 29% in 1999 (Figure 3a), synergized by the impacts from human activities in the 90s (Dai 1997, Dai et al. 1998, 1999, Keshavmurthy et al. 2019).
2. Between 1999 and 2004, a period without major disturbances, coral cover recovered from the 1998 bleaching event and reached 47.5% in 2004, the highest mean coral cover recorded in this study (Figure 3a). In some sites, such as Wanlitong in the Kenting National Park, coral cover returned to levels around 46%, which was last observed in 1985 (Kuo et al. 2012).
3. After 2004, the mean coral cover dropped to 25% in 2010 (Figure 3a), the lowest recorded in this study, and was believed to be due to multiple disturbances.
4. Between 2010 and 2016, another period with only a few minor small-scale disturbances, mean coral cover increased to 31.5% in 2012, followed by a period of stasis where coral cover remained above 30% (Figure 3a).
5. In the last period from 2016 onwards, multiple disturbances such as typhoons and small-scale bleaching events resulted in a 16% loss of the coral cover relative to 2015 (Figure 3a).

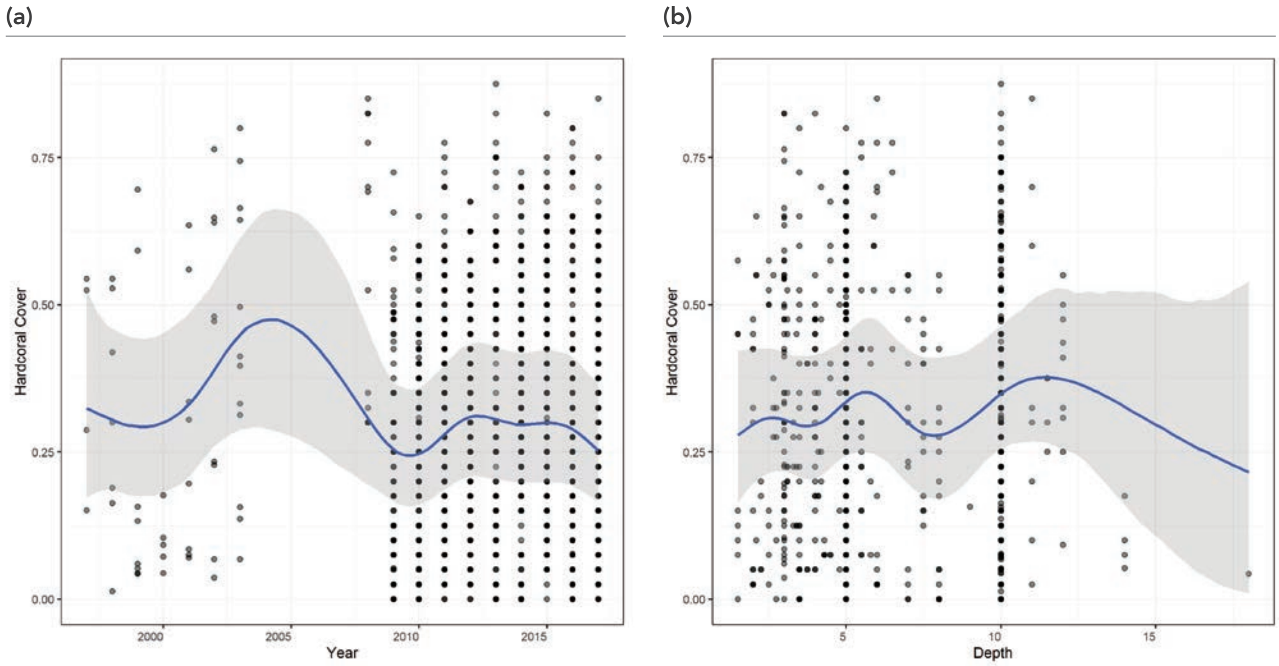

FIGURE 3: *Taiwan coral cover across (a) years and (b) depths.*

Coral Bleaching 2016

Small-scale bleaching was observed in Green Island, Orchid Island, Kenting National Park (south Taiwan), Penghu Archipelago, and Dongsha Atoll in 2016 (Table 2) (Chen 2016, TEIA 2016, Kuo et al. 2018). Bleaching was recorded in 9 of the 19 sites surveyed by Reef Check Taiwan (Table 1), operated by TEIA. The most severe bleaching was observed at a depth of 5 m at Yunuyan-Orchid Island (site 54 in Table 1), with about 35% of the corals bleached and the bleaching severity of each colony recorded to be around 10% (Table 2) (TEIA 2016). The remaining eight sites, two in Orchid Island, three in Green Island, and three in Dongyuping Islet-Penghu Archipelago (site 69–71 in Table 1), showed minor bleaching at the time of the surveys in April, June, and August (TEIA 2016). However, the 19 sites were only surveyed once due to limited capacity. As a result, the bleaching severity for the sites surveyed prior to July might be higher in summer (from summer to September in Taiwan).

Other bleaching observations at the regional scale helped to fill in the knowledge gap. For instance, the bleaching severity in Dabaisha, Green Island (site 41 in Table 1) increased from minor in April (TEIA 2016) to mild in August. Sixty percent of the corals in shallow water were bleached, mainly Acroporidae, Pocilloporidae, and soft corals, and the bleaching expanded to around 16 m in depth (Table 2) (Kuo et al. 2018). Another research team, led by Penghu Marine Biology Research Center, confirmed that minor bleaching and partially bleached coral colonies were restricted to the southern part of Penghu Archipelago. Bleaching was only observed on reefs less than 1 m in depth on two islets east of Dongyuping, Dongji Islet and Xiji Islet (Figure 2h), out of the 15 islands/islets visited across the Penghu Archipelago between July and September in 2016 (Table 2) (Kuo et al. 2018). In Kenting National Park, around 80% of the bleached corals were restricted within the 2.5 km coastline on the western side of Nanwan near sites 32 and 33 in Figure 2d. More precisely, Tan and Fan (C.-J. Tan and T.-Y. Fan, *unpublished data*) recorded bleaching in 61.3% and 61.2% of the coral colonies between a depth of 3 to 5 m at Houbihu and the outlet of the third Nuclear Power Plant, respectively (site 32 and 33 in Figure 2d), in July 2016 (Table 2).

Minor to mild bleaching was also observed in Dongsha Atoll in the South China Sea (Chen et al. 2017). Around 74.8% of the 1127 colonies at reef flat and reef slope on 15 patch reefs surveyed in July 2016 were recorded in bleaching Category 1 (No bleaching) and Category 2 (Pale), 19.5% belonged to Category 3 (1–50% bleached) and Category 4 (51–99% bleached), 5.3% belonged to Category 5 (100% bleached or fluorescent), and 0.4 belonged to Category 6 (already dead) (Table 2). Furthermore, there was no spatial pattern of bleaching severity among the patch reefs surveyed in Dongsha Atoll (Chen et al. 2017).

TABLE 2: *Summary of the 2016 mass bleaching survey results in Taiwan.*

REGION	SITE	BLEACHING SEVERITY	REFERENCE
Orchid Island	Yunuyan	Bleaching (% of population): 35% Bleaching (% of colony): 10%	TEIA 2016
Green Island	Dabaisha	In shallow water (<10 m in depth), from minor-mild in April to about 60% of the corals bleached in August	Kuo et al. 2018, TEIA 2016
Penghu Archipelago	Dongji and Xiji Islet	Minor bleaching was reported below 1 m in depth between July and September	Kuo et al. 2018
	The west, east and south side of Dongyuoing Islet	Minor bleaching recorded below a depth of 10 m in August	TEIA 2016
Kenting National Park	The west side of Nanwan	In general, 80% of the colonies were bleached	Kuo et al. 2018
	The outlet of the third Nuclear Power Plant	About 61.2% and 61.3% of the coral colonies between a depth of 3–5 m were bleached	Tan and Fan, *unpublished data*
	Houbihu		
Dongsha Atoll	Fifteen patch reefs crossing the atoll	Not bleached/Pale: 74.8% 1–99% bleached: 19.5% 100% bleached: 5.3% Already dead: 0.4%	Chen et al. 2017

Drivers & Pressures

Between 1997 and 2017, the main drivers for the status and trend of the coral reefs across Taiwan's waters were acute disturbances such as bleaching events, typhoons, as well as chronic disturbances from human activities, such as overfishing, habitat destruction, and the reduction of water quality.

Prior to the bleaching event in 2016, the 1998 bleaching event was the first and the only major impact on the tropical coral reefs in Taiwan since the 1980s (Lee 1998, Wilkinson 1998). Bleaching was first observed in the southern part of Penghu Archipelago in June 1998; about 30–40% of the corals between 1 m and 5 m in depth were bleached, and some corals died as a result of the anomalously high mean summer seawater temperature around 30°C. In August, the bleaching became extensive, with over 80% of the corals affected from the surface down to 20 m in Orchid Island with mean seawater temperatures of 34°C and 31°C at depths of 1 m and 20 m respectively (Lee 1998, Wilkinson 1998). Bleached corals were also observed down to 20 m in Kenting National Park and 30 m in Green Island and Dongsha Atoll (Li et al. 2000, Dai 2006b, 2018b).

The recovery from the 1998 bleaching event varied among reefs. Nearly all the corals within the 500 km^2 semi-enclosed Dongsha Atoll (25 km in diameter) were dead, and the recovery rate was prolonged due to the low exchange rate of water between the atoll and the open sea (Dai 2006b, 2019). In contrast, the corals in Kenting National Park, Green Island, and Orchid Island recovered comparatively faster from the bleaching event due to their exposure to the open ocean and better water exchange rate (Kuo et al. 2012, Dai 2018b, Keshavmurthy et al. 2019).

After the 1998 bleaching event until 2017, there has not been a Taiwan-wide bleaching event yet. All the other bleaching events; in 2002, 2007, 2010, 2014, and 2016 were at the local scale (summary in Chen 2019).

Typhoons are the other main driver shaping the coral reefs in Taiwan. Taiwan Island is situated in the Northwest Pacific Ocean, the most active basin of typhoons on the planet. During the study period between 1997 and 2017, 74 typhoons hit Taiwan (Typhoon Database, Central Weather Bureau, Taiwan), eight of which caused significant damage to the reefs, mostly in southern Taiwan Island and its neighboring offshore islands (summary in Chen 2019). Furthermore, the impacts from typhoons on reefs across Taiwan Island vary depending on the track, intensity, and duration. First, most of the typhoons causing damage traveled north-westward, either crossing through:

The central region – Typhoon Morakot (cat 1)[1] in 2009 and Typhoon Soudelor (cat 5) in 2015;

1. The southern part of the island – Typhoon Nanmadol (cat 5) in 2011 and Nepartak (cat 5) in 2016); or

2. The Luzon Strait between Taiwan and the Philippines, without landing on Taiwan Island – Typhoon Usagi (cat 4) in 2013.

Typhoons moving along other tracks include north-northwest from Luzon Channel into Taiwan Strait (Typhoon Chebei (cat 3) in 2001, and Typhoon Meranti (cat 5) in 2016) or north/north-northeast from the South China Sea into the Taiwan Strait (Typhoon Chanchu (cat 4) in 2006). Minor damage can result from typhoons that travel either along the east coast of Taiwan towards the higher latitude or by typhoons that travel westwards, crossing through the waters north of Taiwan. Six out of the eight typhoons that caused significant damage in Taiwan were category 4 or 5, such as Typhoon Nepartak (cat 5) and Meramti (cat 5). Most of the impacts from these storms were on the reefs in Kenting National Park except for the reefs on the east coast of Hengchun Peninsula (Chen 2016, Keshavmurthy et al. 2019), the west coast of Green Island, and the east coast of Liuqiu Islet in 2016 (TEIA 2016). For instance, along the west coast of Green Island, the 17 m high stormwater surge caused by Typhoon Meramti in 2016 not only overturned the concrete blocks filled with cement used for coral recruitment study in the upper mesophotic (40 m in depth) in Guiwan but also resulted in the toppling of the 1,000-year-old *Porites lobata* colony ("Big Mushroom") in Shilang, measuring 12 m in height and 31 m in circumference (Soong et al. 1999). This *Porites* coral colony was broken at around 2 m below the seabed and is now lying on its side.

[1] Saffir-Simpson Hurricane Scale (SSHS) Category

In general, the majority of typhoons causing significant damage in Taiwan have been strong typhoons (cat 4 or cat 5). However, weak typhoons can also impact coral reefs, given sufficient duration, such as Typhoon Morakot in 2009 (cat 1); the deadliest typhoon to impact Taiwan in recorded history (Kuo et al. 2011, Chen 2019). Because of its extended duration over Taiwan (48 hours), Typhoon Morakot significantly damaged most of the reefs, in particular the westward-facing coral reefs in Taiwan and its offshore islands, except the reefs in northern Taiwan (Dai 2009, TAMEE and TEIA 2009, Kuo et al., 2011, 2012, Keshavmurthy et al. 2019). For instance, the coral assemblage in Xiangjiaowan, southern Taiwan, had resisted multiple disturbances, including another major Typhoon Herb (cat 5), which damaged the reefs south of Taiwan in 1996 (Dai et al. 1998, 1999), and maintained the coral cover (>45%) for two decades since the first record in the 1980s (Dai 1993). However, two-thirds of the coral, from 58.8% to 18.5%, were removed by Typhoon Morakot (Kuo et al. 2011). Furthermore, the coral cover had not yet fully recovered before being damaged again by two typhoons in 2016. The coral cover increase was mainly contributed by *Acropora* instead of laminar and massive *Montipora*, which used to be dominant and caused the shift of coral assemblage composition significantly (Keshavmurthy et al. 2019).

Other smaller-scale disturbances to the reefs across Taiwan included the outbreaks of sea sponge *Terpios hoshinota* in Green Island and Orchid Island between 2007 and 2009 (Liao et al. 2007, Chen et al. 2009), and crown-of-thorns starfish (COTS) outbreak on islands in the southern part of Penghu Archipelago between 2009 and 2013 (summary in Table 3 in Chen 2019). A cold-shock event in Penghu Archipelago in 2008 also resulted in a significant drop of coral cover from >50% to <20% in shallow coral assemblages (Hsieh 2008, Hsieh et al. 2008).

The major chronic anthropogenic threats to the coral reefs in Taiwan are overfishing, habitat destruction, and pollution (Chen 2014). As one of the main protein sources that support the 23.5 million people in Taiwan, reef fishes have been heavily harvesting, resulting in the decline of the fish population for decades. This overfishing was evidenced by the low abundance of fish and invertebrate indices during the Reef Check surveys in Taiwan in the past two decades, all the way back to the beginning of the Reef Check in 1997 (Dai 1997). In addition, the entanglement from abandoned fishing gears, particularly fishing lines, and nets, causes long-lasting threats to reef organisms (Dai 2010). Coastal areas are heavily used and resulted in habitat destruction as a result of only up to one-third of the total land area in Taiwan (36.2 thousand km^2) are habitable. For example, fishing harbor construction, seafront hotel, housing development projects, and installation of concrete tetrapods for coastal storm protection have destroyed Taiwan's natural habitats (Chen 2014). The replacement of natural habitat with anthropogenically modified habitats might alter the species composition from reef-associated species to more generalist fauna (Wen et al. 2010). The sediments and nutrients carried by sewage runoff due to developments throughout the watershed, rivers to river mounts, as well as the direct human activities in coastal areas have reduced the water quality and caused eutrophication in most reefs (Chen 2014, Dai 2010). For instance, land-based human activities from tourist and local population resulted in the reduced water quality, such as the increased concertation of suspended solids and ammonia, in the intertidal zone and shallow water after heavy rainfall and further led to not only the abnormal growth of algae but also the decline of coral cover in Nanwan Bay, Kenting National Park, in the 90s and early 2000 (Dai et al. 1998, 1999, Meng et al. 2008, Liu et al. 2012). The water quality improved gradually after the operation of the sewage treatment facility in the early 2000s, which also result in the recovery of coral cover in Nanwan Bay (Keshavmurthy et al. 2019).

RECOMMENDATION

Results from Reef Check Taiwan and long-term ecological research projects in Taiwan have indicated that large-scale natural disturbances, including typhoons and bleaching events, are the primary drivers shaping the status of coral reefs. At the same time, chronic stress from human activities controls the resilience of coral reefs. However, there remain insufficient data on the reefs for interpreting coral reefs' changing patterns.

We are facing increasing threat from growing anthropogenic activities and climate change, estimating more MPAs while coordinating with existing marine protected areas (MPAs) to form a crossing-authority, ecologically coherent MPA network under proper law enforcement and a formal national-wide long-term ecological research program on coral reefs and coral assemblages crossing Taiwan are urgently needed.

At first, establishing more MPAs at sites to form an ecologically coherent MPA network is necessary for protecting the marine ecosystem in Taiwan's territorial sea. Several MPAs with different levels of restriction has been established to protect the marine ecosystems in Taiwan, such as national park, fishery conservation zone, and refuge or natural reserve, under the National Park Law, Fisheries Act, Wildlife Conservation Act (summary in OCA [Ocean Conservation Administration] 2021). However, current MPAs not only cover 8.16% of Taiwan's territorial sea area (OCA 2021), lower than the Aichi Targets 11 goal of 10% MPA coverage by 2020 but also were designed on an individual ad hoc basis that the several ecological processes, such as the connectivity between habitats along the dispersal track of creatures, were not taken into account. As a result, it is recommended to establish a series of MPAs at different habitats along with the main current in the two main geographical regions, the Kuroshio in the tropic and the China Coastal Water (CCW) in the subtropical region, to form the MPA network. Also, it is necessary to apply the concept of Morisatoumi Linkage; a concept developed from Satoyama-Satoumi Ecosystems focusing on establishing a total management system for the overall area ranging from forest to sea (Saito and Shibata 2012), for manage the protected areas along the watershed. Most important, it is vital to develop tools and strategies to ensure proper MPA enforcement.

Second, a formal national-wide long-term ecological research program is required to provide sufficient data for reef management and governance, which required regular and timely updates of the status of coral reefs. This program should include cross-generation and cross-disciplinary members to develop novel technology and innovation for monitoring the coral reefs ecosystem and act as a platform for building up the network through collaboration.

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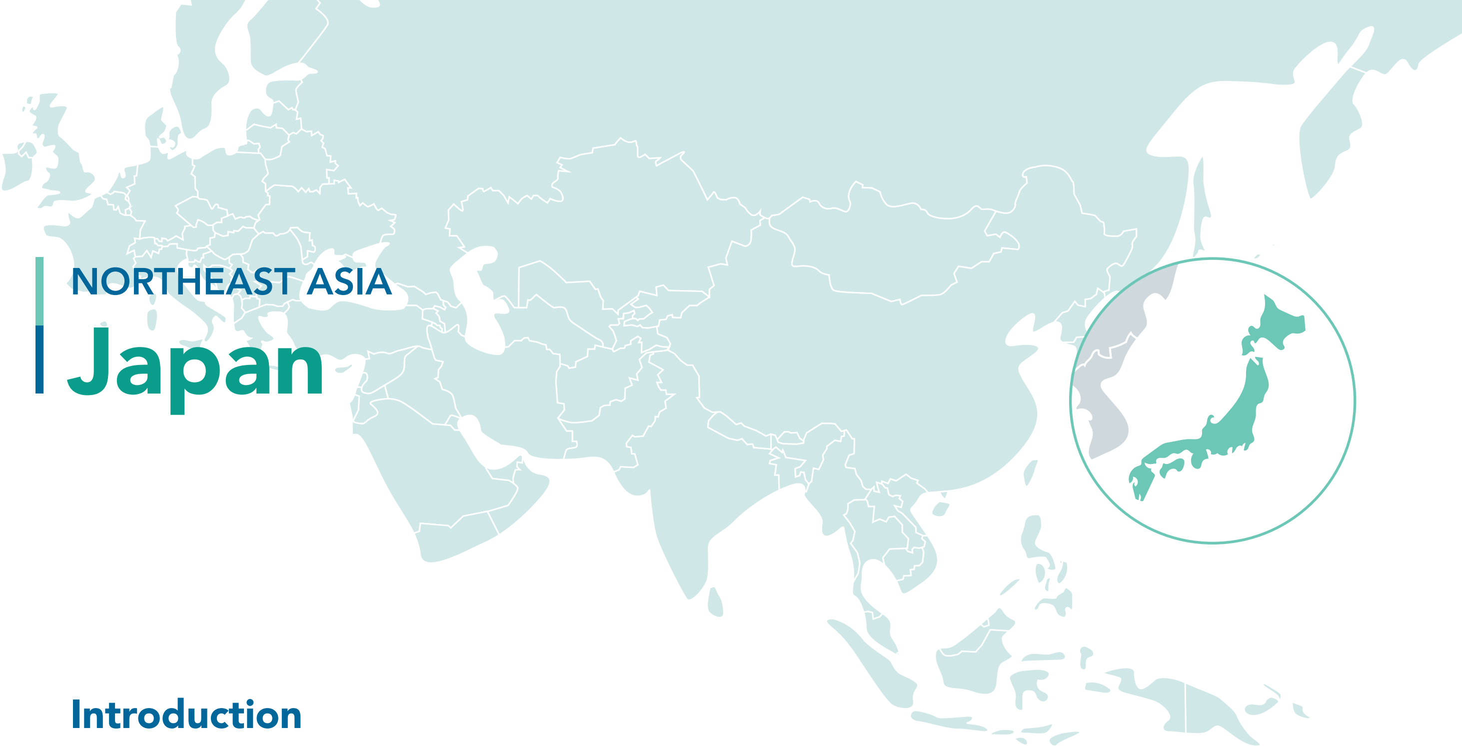

Introduction

Coral reefs are distributed in the southern area of Japan from Okinawa prefecture up to Iki Island in Nagasaki Prefecture within the Sea of Japan and Kagoshima Prefecture along the Pacific coast, while coral communities are observed along the rocky shores from Kyushu Island to Honshu Island. Since 2003, the Ministry of the Environment has conducted a national coral monitoring program at 24 sites to cover most of Japan's coral reef habitats and communities. The program developed a timed swim observations survey method named "spot check method," and most of the data for this country analysis was recorded using this method. Each site has 15 to 30 survey stations to estimate the percent cover of live coral and bleached coral, number of COTS, and other notable disturbances during a 15-minute swim around each station. Six sites are located in non-reef areas from Honshu to Kyushu islands, and 18 sites are located in the coral reef areas from Kagoshima to Okinawa prefectures, including the remote islands of Ogasawara of Tokyo Metropolis (Figure 1).

Before this national monitoring program, a series of coral reef monitoring surveys using the spot check method were conducted around the Marine Parks in the Sekisei Lagoon and Iriomote Island in Okinawa prefecture from 1983 to 2002, and they were incorporated into the national monitoring program in 2003 using a similar method. Therefore, the longest monitoring dataset in Japan is from Sekisei Lagoon and Iriomote Island. The second-longest data collection has been performed around Ishigaki Island in Okinawa prefecture since 1998 following the monitoring in Sekisei Lagoon and Iriomote Island. The survey in Ishigaki has also been incorporated into the national monitoring program since 2003. In addition, Reef Check data were also provided by voluntary groups from Coral Network, Okinawa Reef Check Society, and Marine Restoration Network Yoron to support this national analysis.

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Status & Trends

Before the national monitoring program on coral reefs started, coral cover data from Sekisei Lagoon and Iriomote Island collected by a local monitoring program from 1983 to 2003 was used for the National Marine Protected Area of Iriomote. From 1998 to 2003, data around Ishigaki Island was also compiled with data from Sekisei Lagoon and Iriomote Islands. Data were mainly collected from 2004 to 2018 at 24 sites of the national monitoring program on coral reefs by the Ministry of the Environment. In addition to these data, Reef Check data collected since 1997 were also analyzed for this study. Most of the national monitoring program data were collected at shallow reefs less than 10 m in depth, while Reef Check data were collected around 3 m (2–6 m) and 10 m (6–12 m) in depth. Trends in coral cover in Japan across the years and depths are presented below in Figure 2 and discussed in the following sub-sections.

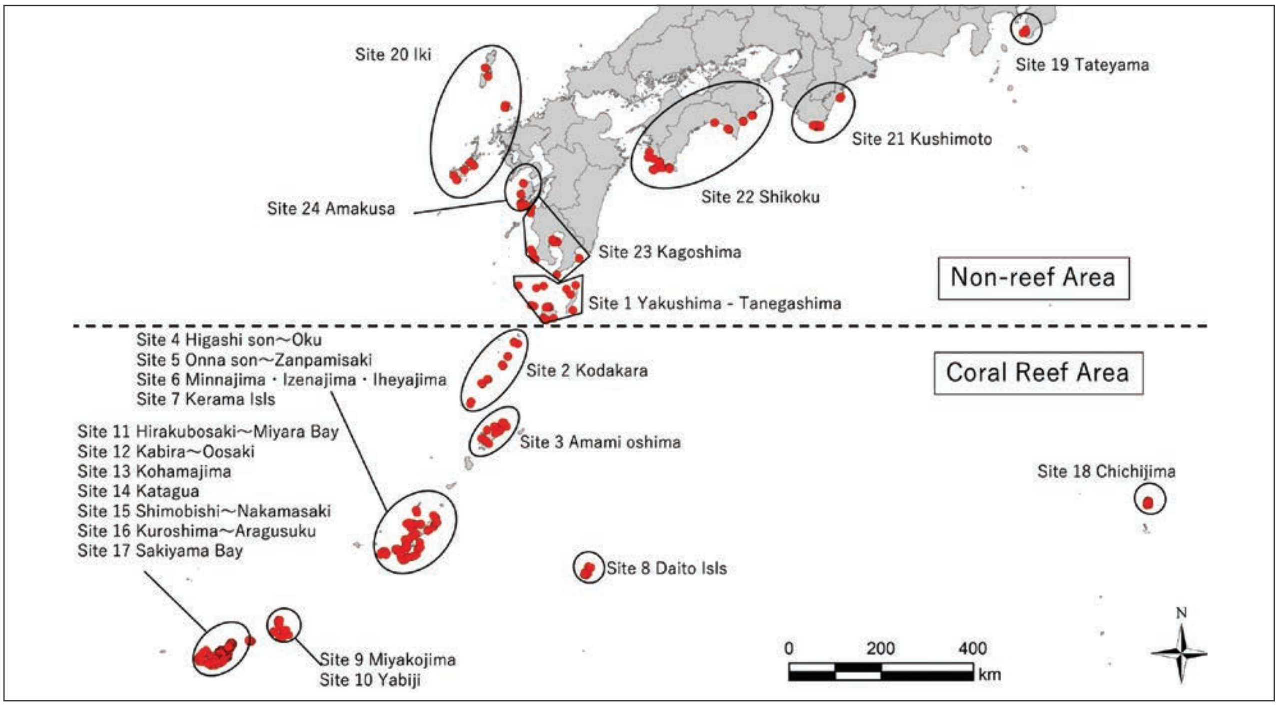

FIGURE 1: *Location of the monitoring sites of the national monitoring program on corals by the Ministry of the Environment (red circles).*

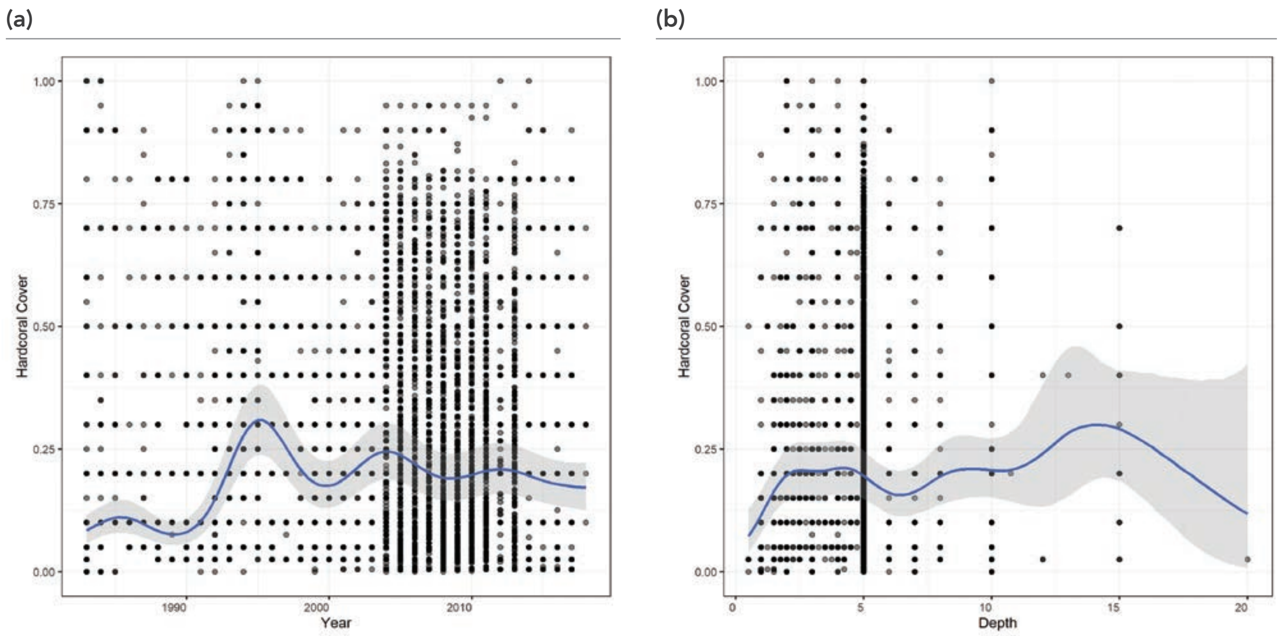

FIGURE 2: *Trends in coral cover in Japan across (a) years and (b) depths.*

CORAL COVER ACROSS YEARS

1983–1989: Low coral cover due to crown-of-thorns starfish (COTS) outbreak

Coral cover from 1983 to 1989 was relatively low, around 10%, because of the major disturbance by the outbreak of crown-of-thorns starfish (COTS) during this period. The outbreak occurred all around Okinawa Prefecture from Okinawa Island to Iriomote Island, including the outlying islands. However, all the obtained data during this period were collected specifically from a monitoring program at the Sekisei Lagoon of Iriomote National Park, which contained the largest semi-barrier reef system in Japan. Therefore, the trend of coral cover represented during this period is highly area specific.

1990–1995: Rapid recovery from COTS damages

After the COTS outbreak diminished in the early 1990s, coral cover rapidly recovered up to 30% by 1995, representing the highest coral cover in this study. This period's data were also collected from the monitoring program of Sekisei Lagoon of the Iriomote National Park.

1996–2000: Rapid degradation by chronic disturbance of typhoons & mass coral bleaching in 1998

From 1995 to 1997, coral cover gradually decreased because of frequent typhoon damage, and the mass coral bleaching in 1998 caused the coral cover to drop to less than 20% in 2000. As a new monitoring program around Ishigaki Island was started in 1998, additional data have been combined to analyze both programs at Sekisei Lagoon and Ishigaki Island since 1998.

2001–2004: Slow recovery from bleaching damage

From 2001 to 2004, coral cover gradually recovered from the mass coral bleaching damage in 1998, but the recovery rate was slower than from 1990 to 1995 after the COTS outbreak.

2004–2007: Slight degradation by chronic disturbances of typhoons, COTS, & local-scale bleaching in 2007

From 2004 to 2007, coral cover decreased gradually because of typhoon damage and local COTS outbreaks in Ishigaki and Iriomote Islands. In 2007, a local coral bleaching event also occurred in Sekisei Lagoon and Iriomote Island that caused coral cover to decline to less than 20%. The monitoring program in Sekisei Lagoon and Ishigaki Island has been synthesized into a newly started national monitoring program in 2004 with the other 17 sites in Japan, including coral reef and non-reef areas. Most of the data have been collected from this national program, with additional data from the Reef Check program.

2008–2012: Slower recovery from bleaching damages

After the local-scale bleaching event in Sekisei Lagoon and Iriomote Island in 2007, the coral cover had been recovering until 2012. However, the recovery rate again became slower than that of the previous recovery period from the 1998 mass coral bleaching event.

2013–2018: Gradual degradation by local typhoon disturbances, COTS, coral diseases, & extensive coral bleaching in 2016

Although no large-scale disturbances were recorded between 2012 and 2015, there were a few local-scale disturbances such as typhoons, COTS, and coral diseases. Then, extensive coral bleaching occurred at the southern Ryukyu archipelago, including Miyako, Ishigaki, Iriomote Islands, and Sekisei Lagoon in 2016 that lead to the degradation trend until 2018.

CORAL COVER ACROSS DEPTHS

The lowest coral cover has been recorded at the shallowest depth range of less than 2 m. Moderate coral cover has been recorded from 2 to 5 m, followed by a slight drop around 6 to 7 m. Coral cover gradually increases and is highest around the depth of 14 m. Coral cover deeper than 15 m showed a declining trend. However, only a few datasets were available for depths greater than 10 m (with most of the national monitoring data collected at less than 5 m depth). Coral populations with a high coral cover were potentially distributed at depths between 10 and 15 m. This depth-related variation may correspond to the more severe environmental conditions (e.g., exposure to higher seawater temperature, excessive light, etc.) at shallower depth during the mass bleaching events.

Coral Bleaching 2016

Consecutive coral bleaching events from 2015 to 2017 were the most severe disturbance recorded on coral communities in the coral reef area, especially Miyako Island, Ishigaki Island, Iriomote Island, Sekisei Lagoon, and the southernmost islands groups.

The highest mortality was 67.9% recorded at Sekisei Lagoon east, and the second-highest mortality was 67.5% at the Miyako outer reefs (Table 1). On the other hand, coral bleaching, and post bleaching mortality in the northernmost part of the coral reef area (Amami Islands) as well as the non-reef area was not significant in 2016 and 2017.

TABLE 1: *Overall coral bleaching results from 2015 to 2017 at all the national monitoring program sites by the Ministry of the Environment.*

AREA	NO.	SITE NAME	AVERAGE CORAL COVER (%)			AVERAGE CORAL BLEACHING (%)			AVERAGE MORTALITY (%)		
			2015	2016	2017	2015	2016	2017	2015	2016	2017
Coral Reef Area	3	Amami Islands	34.2	38.3	33.9	0	8.5	1.8	0	2.1	0
	4	Okinawa Island, East coast	31.2	34.5	35.5	0.7	21.0	31.5	0	0.7	0.7
	5	Okinawa Island, West coast	26.8	29.1	36.2	0.03	13.1	31.3	0	4.3	5.2
	6	Okinawa Outer Island	26.2	58.0	54.3	0.01	48.4	4.2	0	13.5	1.8
	7	Kerama Islands	17.4	15.8	22.5	0	7.3	0	0	5.4	0
	9	Miyako Island	25.5	18.0	17.0	0	68.8	0.5	0	31.0	0.5
	10	Miyako Outer Reefs	32.5	8.8	6.3	12.5	70.1	0	0	67.5	0
	11	Ishigaki Island, East coast	27.4	27.5	19.6	0	47.9	0.3	0	8.8	0.3
	12	Ishigaki Island, West coast	13.8	13.9	12.1	0	63.2	0.4	0	14.8	0.4
	13	Sekisei Lagoon, North	37.0	23.0	20.1	52.6	91.5	85.0	2.8	46.9	6.7
	14	Sekisei Lagoon, East	31.0	9.3	5.2	62.3	99.5	94.6	2.5	67.9	11.3
	15	Sekisei Lagoon, Center	34.3	18.8	17.2	65.4	94.9	92.0	3.1	49.7	8.5
	16	Sekisei Lagoon, South	31.2	17.9	13.2	66.3	98.2	94.1	2.5	50.0	10.2
	17	Iriomote Islands	48.9	32.4	27.3	39.5	94.3	84.7	1.6	34.8	6.7
	18	Ogasawara Islands	45.0	41.7	45.0	<1	2.9	1.3	0	1.9	0.2
Non-Reef Area	19	Tateyama	2.9	2.9	2.9	0	0	0	0	0	0
	20	Iki and Tsushima Islands	47.0	37.3	32.0	1.9	2.1	0.4	0.3	1.1	0.3
	21	Kushimoto (Wakayama pref.)	32.3	33.1	27.7	0	0.8	13.3	0	0.1	0.5
	22	Shikoku Southwestern coast	25.8	30.9	25.4	0	3.0	7.1	0	0.5	1.3
	23	Kagoshima Southern coast	19.0	18.4	16.4	0	20.7	0	0	0	0
	24	Amakusa (Kumamoto pref.)	26.6	27.6	31.1	0	0.2	0.5	0	0	0
	1	Tanegashima and Yakushima	34.2	38.3	33.9	0	8.5	1.8	0	2.1	0

Drivers & Pressures

The major disturbances driving coral cover decline on Japanese reefs were the outbreaks of *Acanthaster planci* (COTS) and mass coral bleaching events. The lowest coral cover in the 1980s until the early 1990s was caused by the severe COTS outbreaks in Okinawa prefecture. Following the recovery from the COTS damage in the late 1990s, all the negative trends of coral cover coincided with major coral bleaching events in 1998, 2007, and 2016. Between these disturbances, recovery potential seems to be suppressed by local scale pressures such as typhoon damage, coral diseases, sedimentation, localized COTS predation, and coral bleaching. Although eutrophication, marine pollution, and tourism-related activities are possible pressures on corals, data were not readily available by current monitoring programs to substantiate their impacts.

ACANTHASTER PLANCI (COTS: CROWN-OF-THORNS STARFISH)

A series of COTS outbreaks occurred in the 1980s, and the coral cover dropped to the lowest (10% or less) at the Sekisei Lagoon of Iriomote National Park during the entire study period. COTS outbreak widely occurred again in the early 2000s from Coral Reef Area to Non-Reef Area. The major outbreak diminished by the middle of the early 2000s, but some of the aggregations remained in specific areas among the non-reef coral populations during the 2010s.

According to the analysis, COTS predation has been one of the major disturbances in Coral Reef Areas since 2000, when the outbreak started in the Amami Islands and ended in 2007. The outbreak also started in Ishigaki, Iriomote Islands, and Sekisei Lagoon in 2001. The number of individuals increased rapidly from 2009, around Ishigaki, Iriomote Islands, and Sekisei Lagoon until 2012. COTS aggregations were also found around Miyako Island and Yabiji reefs in 2002, where the outbreak continued until 2012. An isolated outbreak also appeared in Kerama Islands in 2004 and ended in 2007.

Predation by COTS can also be a serious threat to coral communities in the Non-Reef Areas. Large numbers of COTS were observed around Kushimoto in early 2000, and the first aggregation was found in 2004. The maximum number of individuals and major degradation of coral cover was recorded in 2005. COTS numbers also started increasing along the Shikoku southwestern coast in 2004. Coral cover at this site had been affected by COTS predation since 2008, and the maximum number of individuals was observed in 2010. Although the peak in the COTS number has passed, some of the stations within the Shikoku site showed high aggregations in the following years. Along the Kagoshima southern coast, significant numbers of COTS were recorded in 2007, and predation impacts expanded around the site and remained until 2011. The numbers declined in 2012, but aggregations were still observed in 2017. In Amakusa, many aggregations of COTS appeared since 2002 before the monitoring program started. The number of individuals observed was low until 2007 when it increased rapidly from 2008 to reach a maximum in 2009 before decreasing.

CORAL BLEACHING

Coral bleaching by high water temperature was another major disturbance driving the decline of coral cover. The first mass coral bleaching event occurred in 1998 from Okinawa to the Non-Reef Areas and severely damaged corals. The second severe coral bleaching occurred in 2007 around the Sekisei Lagoon between Ishigaki and Iriomote islands. More localized bleaching events occurred mainly in Ishigaki Island, Sekisei Lagoon, and Iriomote Island in 2007, and 30 to 60% of the corals in the Yaeyama Islands were bleached. The third and most severe bleaching event occurred in 2016 from Miyako to Iriomote Islands, with more than 50% coral mortality. The average coral bleaching rate of all the sites from Non-Reef Areas to Coral Reef Areas was 54.9% in 2016, and this was the highest rate recorded during the entire study period. The second-highest rate of bleaching was in 2007 with 29.3%. The average mortality at all the sites was 24.9% in 2016 and 19.4% in 2007. The highest mortality of 67.9% was around the eastern part of Sekisei Lagoon, and the second-highest mortality of 67.5% was around the Miyako outer reefs in 2016. In addition, some coral bleaching events were caused by low water temperatures. Yabiji reefs in 2008 and 2009 and some coral communities in the Non-Reef Area in 2010 were slightly affected by low temperature bleaching events, although the damage of both events was not severe with less than 1% mortality. Bleaching by low water temperature in winter was also observed around Kushimoto, Shikoku southwestern coast, and Amakusa. It killed 20% of the coral community, including most of the *Acropora* species in Kushimoto, in 2012.

TYPHOON

Typhoons also damaged reefs and impeded coral recovery during summer seasons in most of the years in Okinawan reefs. Corals were also physically damaged by typhoons hitting this area right after the bleaching event leading to a steep decline of coral cover in 2007. Typhoon damage can also be a major driver of coral degradation in the Non-Reef Area. Reduction of coral cover was observed in 2012 in Iki and Tsushima islands due to typhoon impact. Coral Reef Areas and Non-Reef Areas in Japan are located along the major path of strong typhoons, where serious damage was recorded. In Kushimoto, severe damage was recorded in 2004, 2005, and 2009 with minor damage in 2006, 2007, 2011, and 2012. At Shikoku, typhoon damage was reported in 2007 and 2011. Kagoshima's southern coast was also affected by a typhoon in 2012. In Amakusa, the average coral cover did not show much degradation caused by the typhoons in 2006 and 2012.

CORAL DISEASES

Together with typhoon and COTS, coral diseases were one of the major disturbances that damage coral communities in Kushimoto and Shikoku Islands in the Non-Reef Area as well as Sekisei Lagoon and Iriomote Island in the Coral Reef Areas.

OTHER ANTHROPOGENIC DISTURBANCES

Sedimentation is one of the disturbances on corals in Japan. Soil runoff from agricultural fields and coral mining for marine transportation channels are common sources of sedimentation. In addition, pollution, including eutrophication from wastewater, has become a major concern. However, effective monitoring schemes for assessing the impacts of those disturbances were not well developed, with clear evidence for the national monitoring program. Also, the precise and cost-effective measurements of each pollutant and its direct/indirect impact are not readily available yet. Tourist impacts at these sites with intensive use were also a potential driver for the deterioration of coral health.

RECOMMENDATION

Integrated Approach for Tackling Climate Change

The major threat to Japanese reefs was coral bleaching, which occurred three times during the last two decades to reduce coral cover. Coral bleaching is considered a serious threat caused by global climate change. Integrated and simultaneous strategies should be taken urgently at the national level to reduce climate change to save coral reefs and other ecosystems.

Local Mechanism for COTS Control

At the local scale, COTS (*Acanthaster planci*) outbreaks have been a major disturbance on coral reefs in Japan. Efforts to manage COTS should be developed and organized as a part of local-level management process to conserve key coral communities to maintain their ecosystem functions. The mechanism of COTS management could also be used for the control of *Drupella* spp.

Resilience Based Management

In addition to global climate change impacts, corals have been under local and chronic stresses that hinder their recovery from the damage of major disturbances. Thus, resilience-based management is vital to reduce those pressures for maintaining the ability of the coral community to recover in the local environment. There are several different pressures on corals that should be reduced, such as sedimentation, coral diseases, water pollution, eutrophication, and tourism impact. Each site should develop a local management framework with clear and concrete short and long-term action to reduce all the pressures.

DATA CONTRIBUTORS

The national monitoring program on corals was conducted by the local scientists listed below who collected data and contributed to this national analysis.

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Coral cover data were also provided from the voluntary program of Reef Check teams as listed below.

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NORTHEAST ASIA

South Korea

Introduction

Coral assemblages in South Korea are restricted to the southern parts of the country, mostly in the South Sea and around Jeju Island (Ministry of Environment 2009). Soft and azooxanthellate stony corals comprise the majority of coral diversity in South Korea (Song 1991, Hwang and Song 2009); however, the diversity and abundance of zooxanthellate corals are rapidly increasing in response to climate change (Denis et al. 2013, Sugihara et al. 2014, Hong et al. 2015).

Several governmental and regional institutions, including the Ministry of Oceans and Fisheries (MOF) and the Jeju provincial government, conduct annual marine ecosystem survey programs. These programs include marine biodiversity and coastal habitat mapping surveys, aiming also to investigate the effect of progressive climate change on coastal marine ecosystems. Coral assemblages in South Korea were patchy and scarce in the past; however, recent increases in the diversity and abundance of corals portend critical ecosystem changes, potentially leading to the topicalization of Korean marine ecosystems. In this national chapter, we report an 8-year (2010–2018) long-term coral monitoring dataset collected as part of the National Ecosystem Survey led by the MOF covering three focal areas: Busan Metropolitan City region, Northern Jeju Island, and Southern Jeju Island (Figure 1). At each focal area, three 10-meter line transects were horizontally laid at 5, 10, and 15 meters in-depth, and benthic community composition within 50 cm of either side of the transect was video-recorded in situ, and snapshots of the video footage was merged to render a comprehensive record of the benthic community. Benthic coverage of coral colonies and the presence of biotic and abiotic disruptions, such as the abundance of crown-of-thorns starfish and *Drupella* snails and signs of bleaching, were recorded. In addition, four 0.25 m^2 quadrats (a total of 1 m^2) were placed at each depth to verify the accuracy and precision of data from the video transects.

Status & Trends

Hermatypic coral cover exhibited a minor increase over time, and the cover decreased towards deeper depths across the survey sites; however, the intrinsic lack of coral assemblages in South Korea and resultant small sample size renders these results tentative (Figure 2). Overall, coral cover was lower at the Busan Metropolitan City region (10–18%) compared to the survey site around Jeju Island (30–38% at the northern site; 15–22% at the southern site). *Alveopora japonica* was the dominant coral species at all sites and across depths. Other coral species, including *Montipora millepora*,

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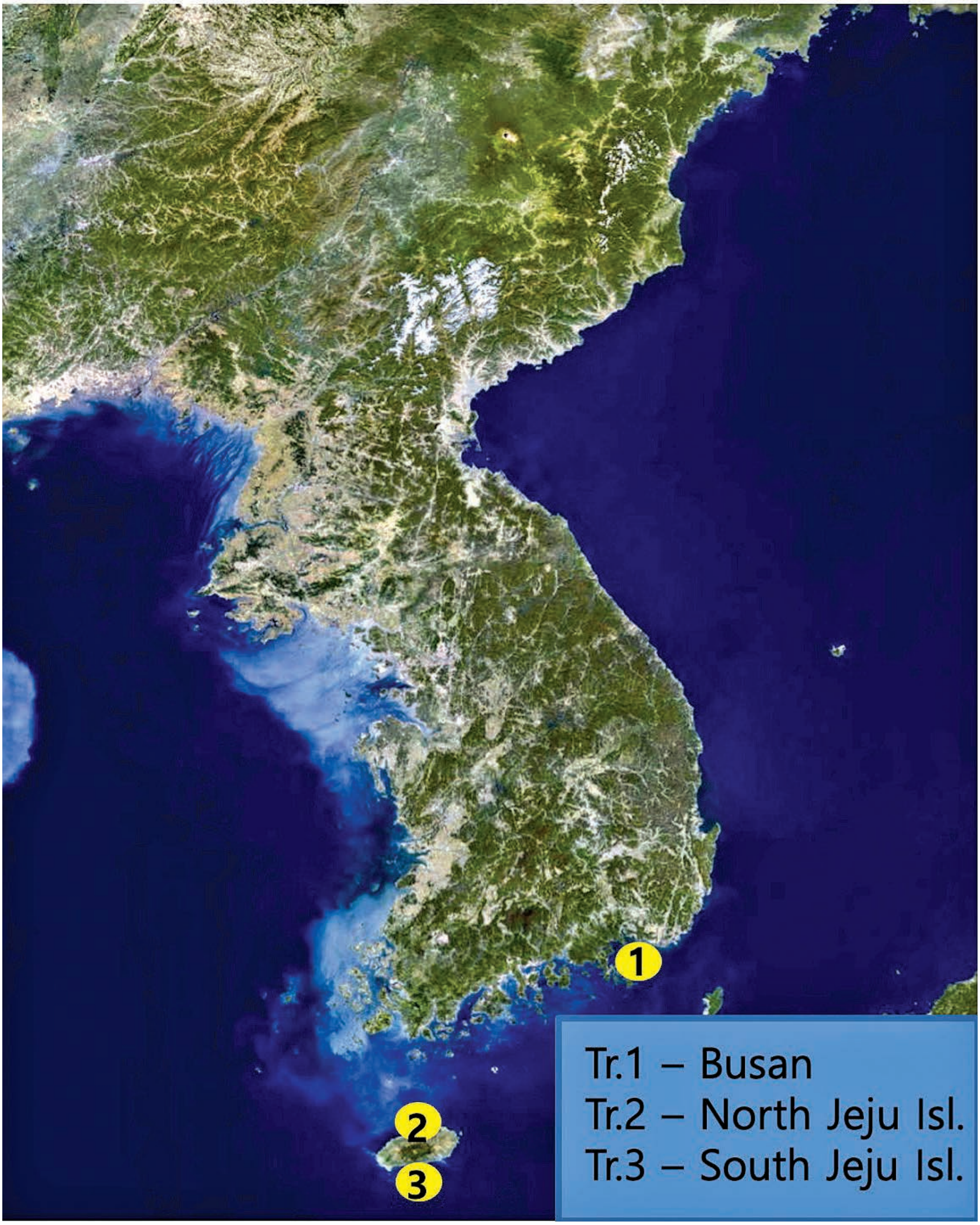

FIGURE 1: *Map of National Ecosystem Survey (MOF) focal areas that harbor coral species.*

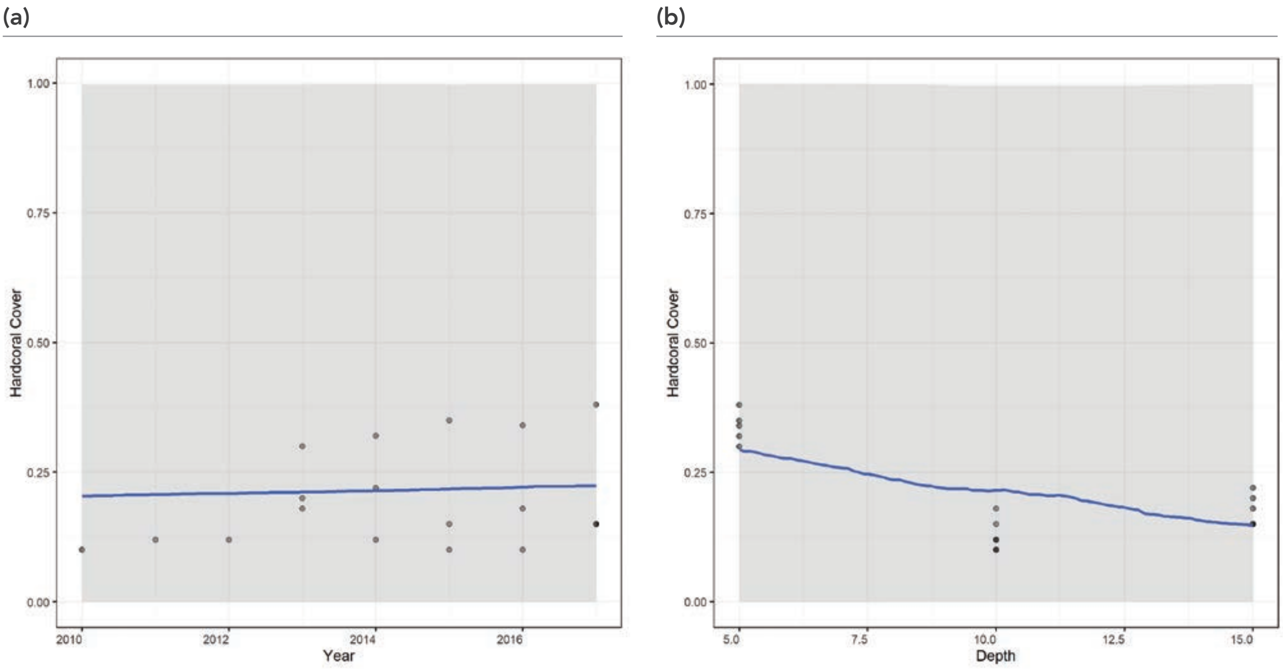

FIGURE 2: *South Korea coral cover across (a) years and (b) depths.*

Psammocora albopicta, Psammocora profundacella, Oulastrea crispata, Micromussa cf. *lordhowensis, Favites* sp., were previously recorded at the vicinity of survey sites (Sugihara et al. 2014). Yet only a small number of *M. millepora* were observed at the southern Jeju Island site during this survey program.

Coral Bleaching 2016

Coral bleaching has not been recorded in South Korea and coral colonies in South Korea were not affected by the 2016 mass coral bleaching event.

Drivers & Pressures

Abnormal thermal conditions do not appear to be a stressor for coral colonies yet in South Korea, even though the area is experiencing one of the fastest warming rates of sea surface water temperature globally (Takatsuki et al. 2007). Heat stress may become a major threat to Korean coral assemblages in the near future. In addition, physical damage caused by seasonal typhoons led to several coral mortality events. Macroalgae tend to occupy the space vacated by physical damage on corals, yet *Alveopora japonica* quickly gains its dominance over macroalgae. *Alveopora japonica* found in Jeju are genetically distinctive to its proximate conspecifics in Japan and Taiwan, and *A. japonica* in Jeju have undergone recent and rapid population increase (Kang et al. 2020). The rampant increase in the number of *A. japonica* colonies is more pronounced in the northern part of Jeju Island near aquaculture facilities, where the ambient nutrient level is much higher than in other areas (Hong et al. 2015).

RECOMMENDATION

Overall, changes in coral coverage and their ecological and economic impacts in South Korea are not as apparent as in countries with profound reef development. While coral colonies in South Korea form non-accreting reefs, replacement of the conventional habitat-forming temperate seaweed with tropical and subtropical scleractinian corals is occurring at an astounding rate (Figure 3). Subsequent changes in key ecological functions and their impacts on local and regional economies are particularly concerning (Vergés et al. 2019). Unfortunately, the number of survey programs focusing on subtropical and tropical organisms is still limited in South Korea. As tropicalization of temperate marine ecosystems accelerates (Yamano et al. 2011, Vergés et al. 2019), we recommend increasing the number of long-term survey programs with a specific focus on subtropical and tropical species range expansions and their impacts on local and regional marine community compositions.

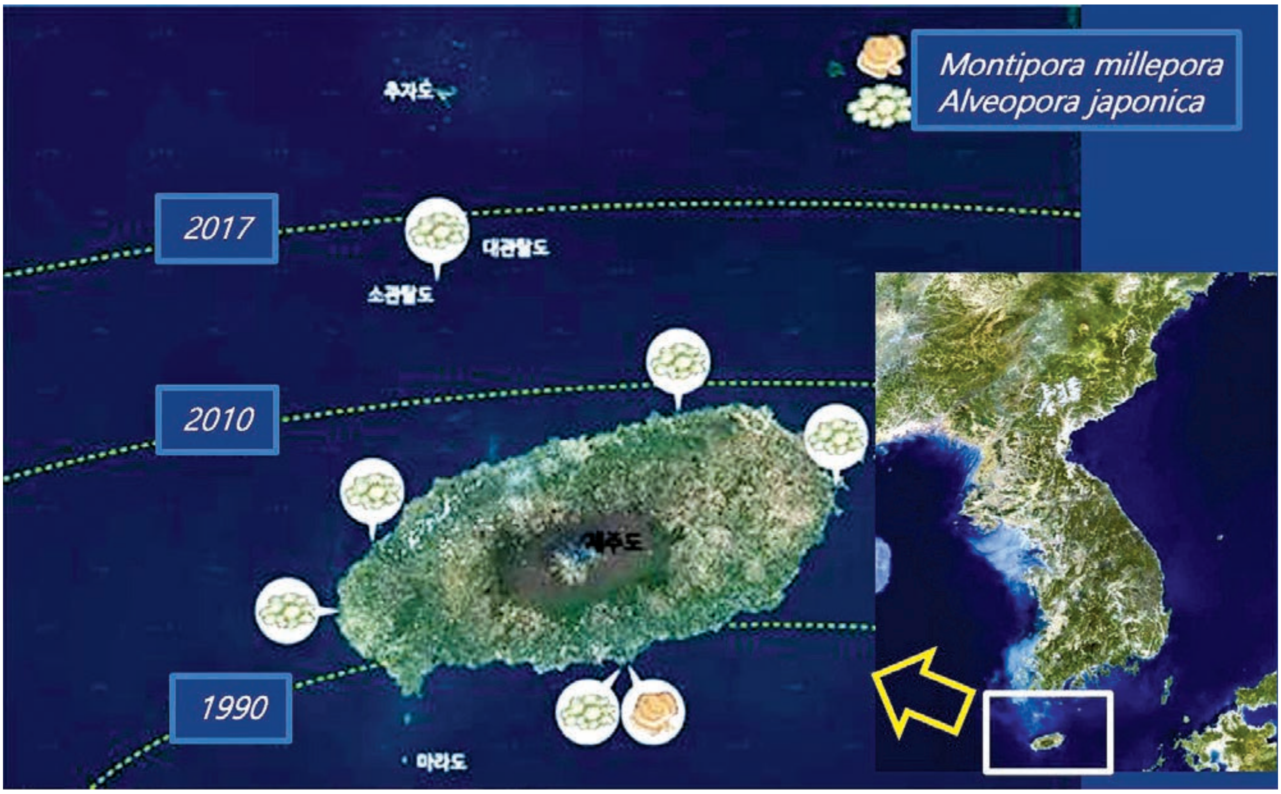

FIGURE 3: *Geographical extent of scleractinian coral range expansion over the past 20-years in the southern part of Korea. Range expansion of scleractinian corals within Jeju has been limited to* ***Alveopora japonica***. ***Montipora millepora,*** *since its initial new record in the 1990s, has not shown palpable poleward range expansion.*

DATA CONTRIBUTORS

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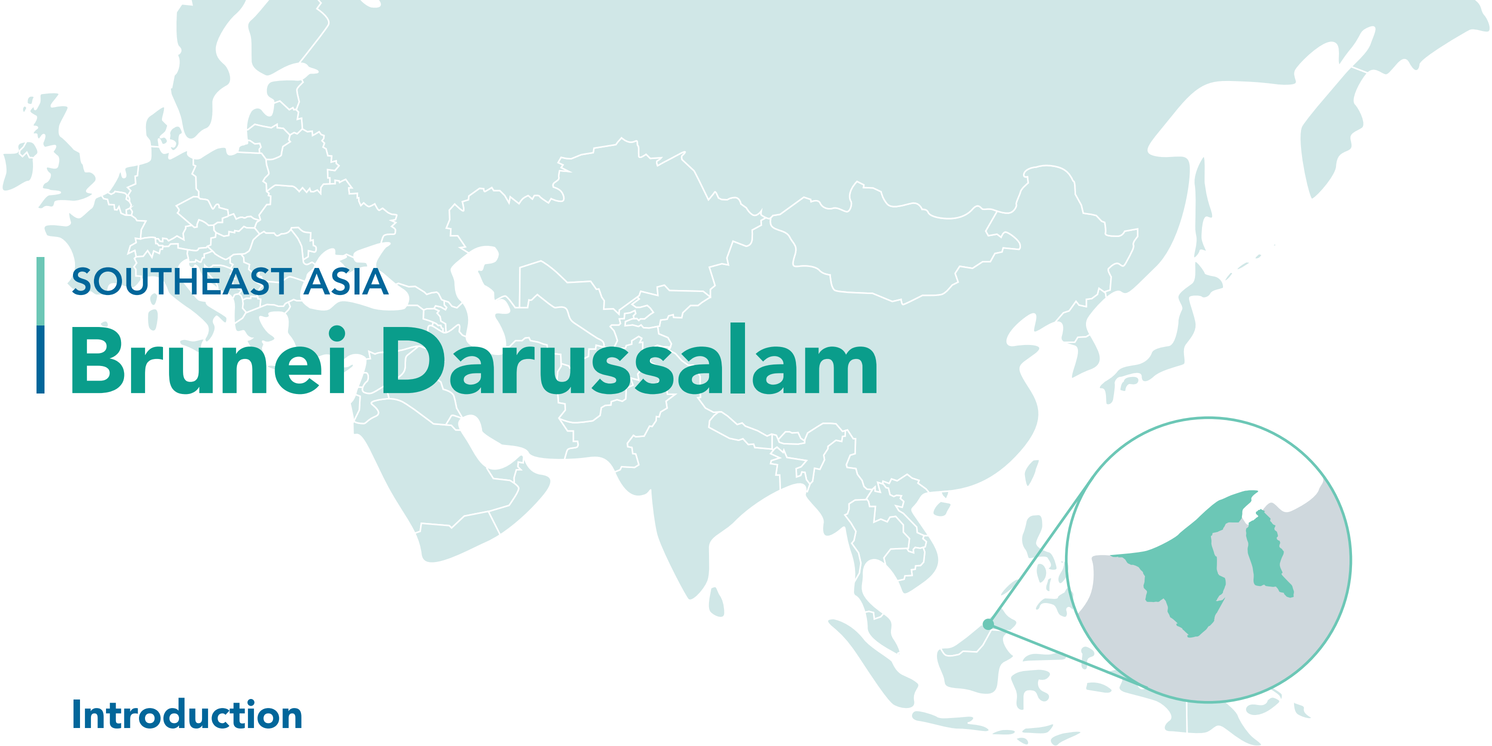
SOUTHEAST ASIA



SOUTHEAST ASIA

Brunei Darussalam

Introduction

Brunei Darussalam, located on the northwest coast of Borneo Island, has fringing reefs and numerous submerged coral reef patches, with a combined area of approximately 100 km^2. The majority of these reef patches are generally located at depths of 4 m to about 12 m with reef slopes and mesophotic reefs extending deeper. All of them are at least 4 km to 30 km offshore (Figure 1). This is mainly due to influences such as lower salinity from river runoff with associated high turbidity and deposition of suspended sediments inshore resulting in a sediment dominated mainland coastline with little coral growth, even on the few rocky outcrops.

The offshore submerged reef of Brunei, little known internationally until recent decades, are now known to be highly diverse. Species richness of corals is comparable to the peak diversity of reef forming Scleractinia in the neighboring 'Coral Triangle' zone (Lane 2011, Lane and Hoeksema, 2016). Studies on reef diversity in Brunei Darussalam have confirmed over 400 species of scleractinian corals recorded from 14 families, with the family Acroporidae accounting for at least 30% of the species inventory (DeVantier and Turak 2009, Turak and DeVantier 2011, Tanaka 2016).

A network of Marine Protected Areas (MPA) was established in January 2012, encompassing most of Brunei's coral reef patches. In addition to the MPAs, exclusion zones at Champion Oil Field and around other oil and gas platforms atop patch reefs provide additional protection from living resource extraction by the public and fishermen.

The first coral reef surveys, carried out in the late 1980s, (Chua et al. 1987, Silvestre et al. 1992) for the assessment of the country's coastal resources and as a baseline for establishment of long-term management efforts, involved only nearshore reef sites (Pelong Rocks, Two Fathom Rock, Pulau Punyit). Yet 88 species in 52 hard coral genera were recorded, an early indication of Brunei's rich coral diversity. Regular coral reef monitoring was started in 2005 by the second author for the Global Coral Reef Monitoring Network (GCRMN). Due to limited resources, only two sites, Pelong Rocks (nearshore) and Littledale Shoal (offshore), were regularly surveyed for ten years (for 10 years annually) using the Line Intersect Transect Method and photo-monitoring. Reef Check surveys were requested by the Department of Fisheries and were initiated in 2012 to monitor all known reef patches (44 sites) that were to be included within the newly established MPA zone. A second survey was conducted in 2016 at the same locations using Reef Check survey methods carried out by local Ecodivers. Following this, all survey results were submitted to the Global Coral Reef Monitoring Network for further analysis.

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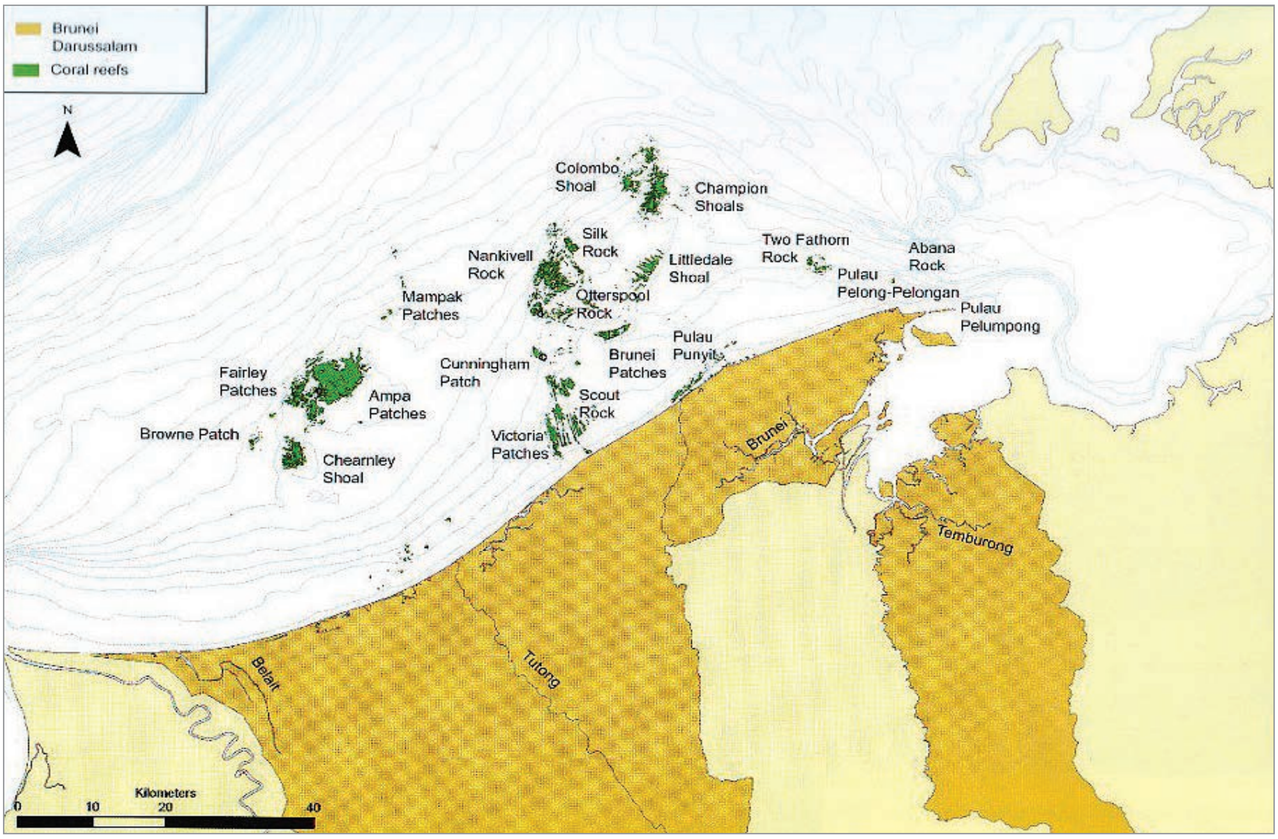

FIGURE 1: *Map showing coral reef patches along the coast of Brunei Darussalam (map sourced and adapted from Brunei Shell Petroleum and Environmental Resources Management [ERM] Hong Kong).*

Status & Trends

CORAL COVER ACROSS YEARS

Average hard coral cover across the years shows a slightly declining profile but maintained between 30% and 45% throughout the years. Figure 2a shows a slowly decreasing trend in live coral cover from the 1990s to 2011, but this increased gradually from 2014. The initial decline may also be driven by the sparse data points in the 1990s, and there was a distinct increase in the number of data points for the years 2011 and 2016 as Reef Check surveys were carried out over larger areas during both periods. This averaging of data does, however, mask a rapid and marked decline (54% to 26.8%) in 2010 that occurred at permanent offshore transects due to a crown-of-thorns starfish outbreak as well as a coral bleaching event. This decline was in contrast to inshore reefs, where live coral cover remained high at 46–48%. Although the coral reefs in Brunei are considerably smaller in size than neighboring countries, the percentage of hard coral cover tends to be relatively high (>30%). Coral cover has remained relatively consistent across time apart from predation and bleaching losses a decade ago.

CORAL COVER ACROSS DEPTH

Coral reefs in Brunei Darussalam are at least 4 m in depth, with most reef tops occurring at depths greater than 8 m. Hard coral cover peaks at about 9 m and decreases sharply at depths greater than 12 m. Figure 2b suggests that hard coral cover in Brunei Darussalam is generally found between depths of 4 m to 12 m. However, deeper reef profiles, including mesophotic reefs (>30 m), have to be fully surveyed to document changes in hard coral cover across depth.

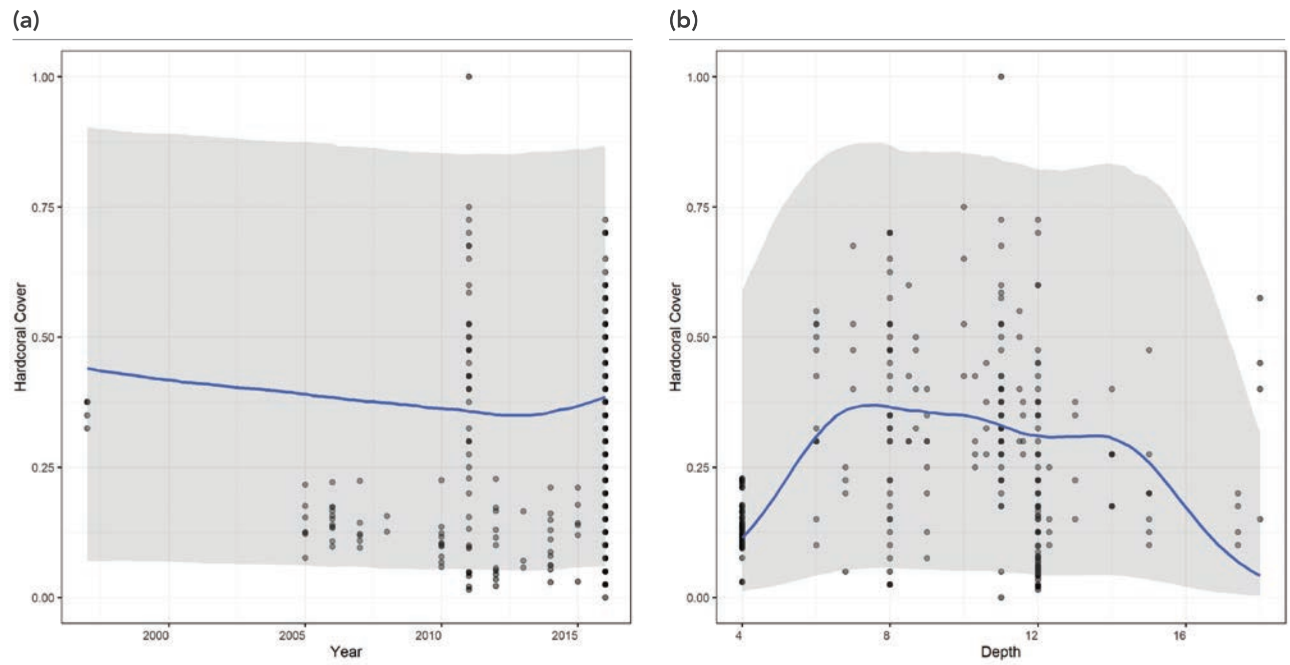

FIGURE 2: *Brunei coral cover across (a) years and (b) depths.*

Coral Bleaching 2016

Bleached corals were observed at ten (10) sites during the Reef Check surveys in May 2016. The highest percentages of bleached coral colonies recorded were found at Champion Middle Reef and Two Fathom Rock (30–60%). During the 2016 survey, the recorded maximum monthly mean sea surface temperature was 31°C and was uniform throughout the entire water column to a depth of 12 m. This phenomenon in Brunei coincided with the third global coral bleaching event from 2015 to 2017 (Skirving et al. 2019). Also noted during the 2016 survey was a partly bleached giant clam at Hornet 1, a small patch reef located east of Brunei Patches (Figure 1). However, the responsible stressor for the giant clam remains unknown as coral bleaching was insignificant at this site (Koehler and Abbas 2016).

Drivers & Pressures

The gradual decrease in hard coral cover before 2013/2014 has been attributed to factors such as sea surface temperature rise, lower bottom salinity, and predation. Mass coral bleaching was not recorded in Brunei before 2010, when a significant bleaching event impacted the country's coral reefs in July of that year. The depth of Brunei's patch reefs, several meters below the surface, provides a "thermal-refuge" for corals, away from warm sea surface layers and UV-induced bleaching (Lane 2011). However, in 2010, high temperatures (30°C+) throughout the water column, coupled with an anomalously low bottom salinity (29.0 ppt), resulted in a significant coral bleaching episode in Brunei (Tun et al. 2010, Lane, 2011), concurrent with a major regional bleaching episode in the Indian Ocean and Southeast Asia (Tun et al. 2010). Although some coral colonies soon recovered, further bleaching occurred in May 2016, again coinciding with a global coral bleaching event (Skirving et al. 2019). It is unknown whether the corals have recovered after this recent bleaching event. Brunei's reefs, being submerged, have a potential thermal refuge that improves their resilience, but this may change if warming events become more extreme and frequent; and if warmer surface layers mix throughout the water column.

Predation by *Acanthaster planci* crown-of-thorns starfish (COTS) and *Drupella* snails has been recorded several times between 2009 and 2016 (DeVantier and Turak 2009, Lane, 2011, 2012, Hodgson et al. 2012, Tanaka 2016, Koehler and Abbas 2016). Monitoring on Littledale Shoal between the year 2008 and 2010 revealed the build-up of an *A. planci* outbreak which reduced live coral cover significantly from 47.1% in 2008 to 26.8% in 2010 (Lane 2012), while visits in 2011 found aggregations of the predatory snail, *Drupella*, on *Acropora* colonies (Lane 2011). The extent and frequencies of predation by both *A. planci* and *Drupella* in Brunei had not been well documented in the past and only became noticeable with the increased regularity of reef surveys from 2005 (Lane 2011, 2012, Koehler and Abbas 2016). More recent data from Reef Check surveys have revealed that while both *A. planci* and *Drupella* snails are still present, their abundances currently remain very low (Hodgson et al. 2012, Koehler and Abbas 2016).

Pollution and sedimentation are two of the most common threats impacting reefs globally. With the majority of Brunei's reefs located offshore, there is limited impact from river discharge and sedimentation except for a few fringing reefs at Pelong Rocks and, more noticeably, at nearshore or coastal sites within the turbid coastal zone, namely Pulau Punyit, Pelumpong Island, and the breakwaters at Empire Resort (Hodgson et al. 2012, Tanaka 2016). Direct impacts from recent or ongoing industrial and infrastructure development projects in Brunei Bay, such as the refinery plant at Pulau Muara Besar, large-scale public housing projects near the coast, and the construction of the Brunei Temburong Bridge have yet to be quantified. These developments may ultimately raise the sediment load and turbidity of offshore waters. Additionally, a shift in Brunei's economic diversification from the oil and gas industry is leading to rapid development in the fisheries sector, particularly aquaculture. While the economic benefits of the fisheries sector look promising for the country, there needs to be careful monitoring and regulation of these developments to ensure that fish farms are operated in a sustainable manner that does not adversely affect coastal water quality. More recently, observational surveys by Reef Check Brunei and local dive operators have recorded a large number of ghost nets on several reef patches (*unpublished data*), and similar impacts had been noted in the past (Hodgson et al. 2012, Koehler and Abbas 2016). Although hard coral cover for Brunei remains consistently high over the past ten years, this status may be at risk of change for the worse due to increasing pressure on coastal resources.

RECOMMENDATION

Recommendations for the management of Brunei Darussalam's coral reefs by governmental agencies, NGOs, reef managers, and other stakeholders include:

1. Ensure the continuity and increase the frequency and regularity of long-term coral reef monitoring in Brunei Darussalam. These monitoring programs need to be consistent in tracking and managing changes such as percentage (%) live coral cover, bleaching events, and predation by *A. planci* and *Drupella*;
2. Enforce regulations on wastewater or sediment discharge, especially from ongoing industrial projects and aquaculture activities in Brunei Bay; and
3. Increase patrolling at Marine Protected Areas to reduce the impact of illegal fishing and abandoned nets in this area.

DATA CONTRIBUTORS

List of contributors who have provided baseline and monitoring data include ICLARM / ASEAN Coastal Resources Management Team for data years 1987–1992, Dr. David J.W. Lane for data years 2005–2015, Reef Check International & Reef Check Malaysia for 2012 & 2016 data, and Reef Check Brunei for 2016 data.

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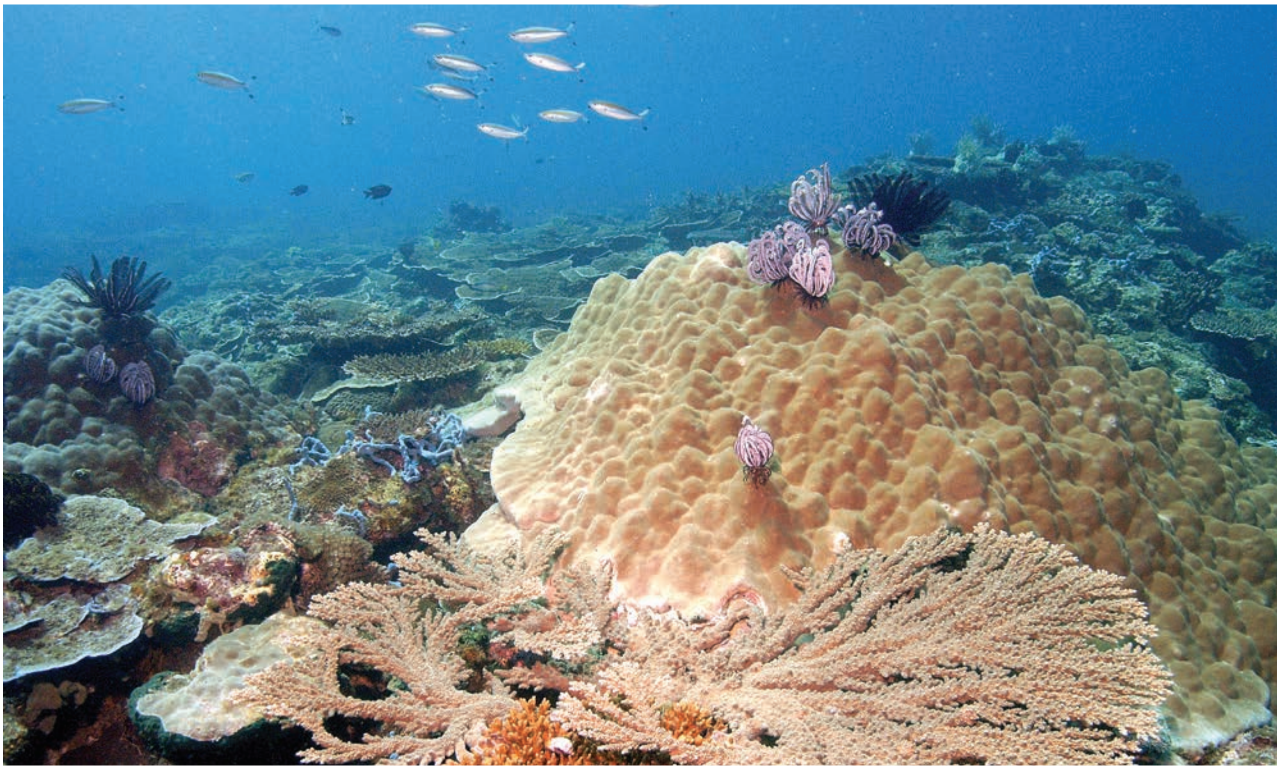

SOUTHEAST ASIA

Cambodia

Introduction

Cambodia has one of the shortest coastlines among the nations that border the Gulf of Thailand, with only 440 km of mainland coastline and 69 offshore islands (USAID 2001, Ya 2011) (Figure 1). Despite this limited coastline, Cambodia contains extensive mangrove, seagrass, and coral reef habitats (Boon et al. 2014). Fringing coral reef habitat is estimated to total 28 km^2, primarily situated around the nation's offshore islands (Savage et al. 2013). As one of the poorest countries in the Southeast Asia region, a significant proportion of the population still resides in rural areas. Consequently, direct resource extraction remains a primary livelihood mechanism for many households in coastal communities (Culas and Tek, 2016). Moreover, alongside resource extractive livelihoods (i.e., fishing and aquaculture), an ever-increasing coastal tourism sector is driving land-use change, income generation, and facilitating livelihood diversification among coastal communities (Mulligan and Longhurst 2014, Leng et al. 2015, Prak et al. 2018). Therefore, ensuring marine habitats are healthy and productive is vital for biodiversity conservation, coastal economies, and securing the livelihood of Cambodian coastal communities in the face of increasing anthropogenic stressors.

To date, there remains a paucity of peer-reviewed publications investigating the status and composition of Cambodian coral reef systems. This dearth of research has left the current status and resilience of Cambodian coral reefs relatively unknown throughout the wider scientific community. The few available publications unanimously define Cambodian coral reefs as characterized by low diversity among hard coral, fish, and invertebrate assemblages, compared to other nations in the Southeast Asia region (Chou et al. 2003, Savage et al. 2014, Thorne et al. 2015). Massive and encrusting growth forms primarily belonging to the genera *Porites* and *Diploastrea* dominate hard coral communities, which offer minimal structural complexity and refugia for reef dwelling species, thereby limiting biodiversity potential (Savage et al. 2014, Thorne et al. 2015, Darling et al. 2017). A myriad of factors can drive low morphological diversity of hard coral communities. However, a major contributor in the context of Cambodian reef systems can be partially attributed to the high turbidity of the water column due to sediment load, ubiquitous at all study sites (Thorne et al. 2015, Yim, 2015). High turbidity and sedimentation promote the recruitment and survivorship of stress-tolerant genera, which are primarily massive and encrusting morphological growth forms frequently observed during surveys (Darling et al. 2012, Padilla-Gamiño et al. 2012, Darling et al. 2013, Ng et al. 2016).

Furthermore, within fish communities, key predators such as Serranidae spp. and herbivorous Scaridae spp. are present in low abundance, with those

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observed frequently categorized within small size cohorts, an indicator of a reef system subjected to high fishing pressure (Muallil et al. 2014, Savage et al. 2014). High levels of nutrient indicator algae were observed at many survey sites, complemented by a high abundance of *Diadema* spp., which are potentially occupying an ecological niche left void due to a low abundance of herbivorous fish competitors (Sangmanee et al. 2012, Francis et al. 2019).

Conservation efforts aimed at mitigating the decline of coral reef habitats have shown that management strategies currently employed across Cambodia are having a positive influence. Small-scale localized conservation efforts have shown significant recovery of fish assemblages within a short space of time (Reid et al. 2019), while collaborative community-led approaches to marine management have been effective at mitigating and stabilizing habitat degradation (Glue and Teoh 2020). Marine Protected Areas (MPAs) are a relatively new concept in Cambodia, with the first MPA proclaimed in 2016 (Ministry of Agriculture Forestry and Fisheries 2016). However, progress is moving forward quickly, with a second MPA proclaimed by 2018 (Fisheries Administration 2019) and two more currently under development. Additionally, there has been a successful initiative to standardize coral reef monitoring protocols utilized by all Non-Governmental Organizations (NGOs) and government ministries throughout Cambodia, resulting in the establishment of the Cambodian Coral Reef Monitoring Network (CCRMN). Whereby all signatories within the network agreed to use a refined version of the methodology first recognized by Thorne and Longhurst (2013) and submit all coral reef monitoring data to a national database. The CCRMN will convene annually, and all members

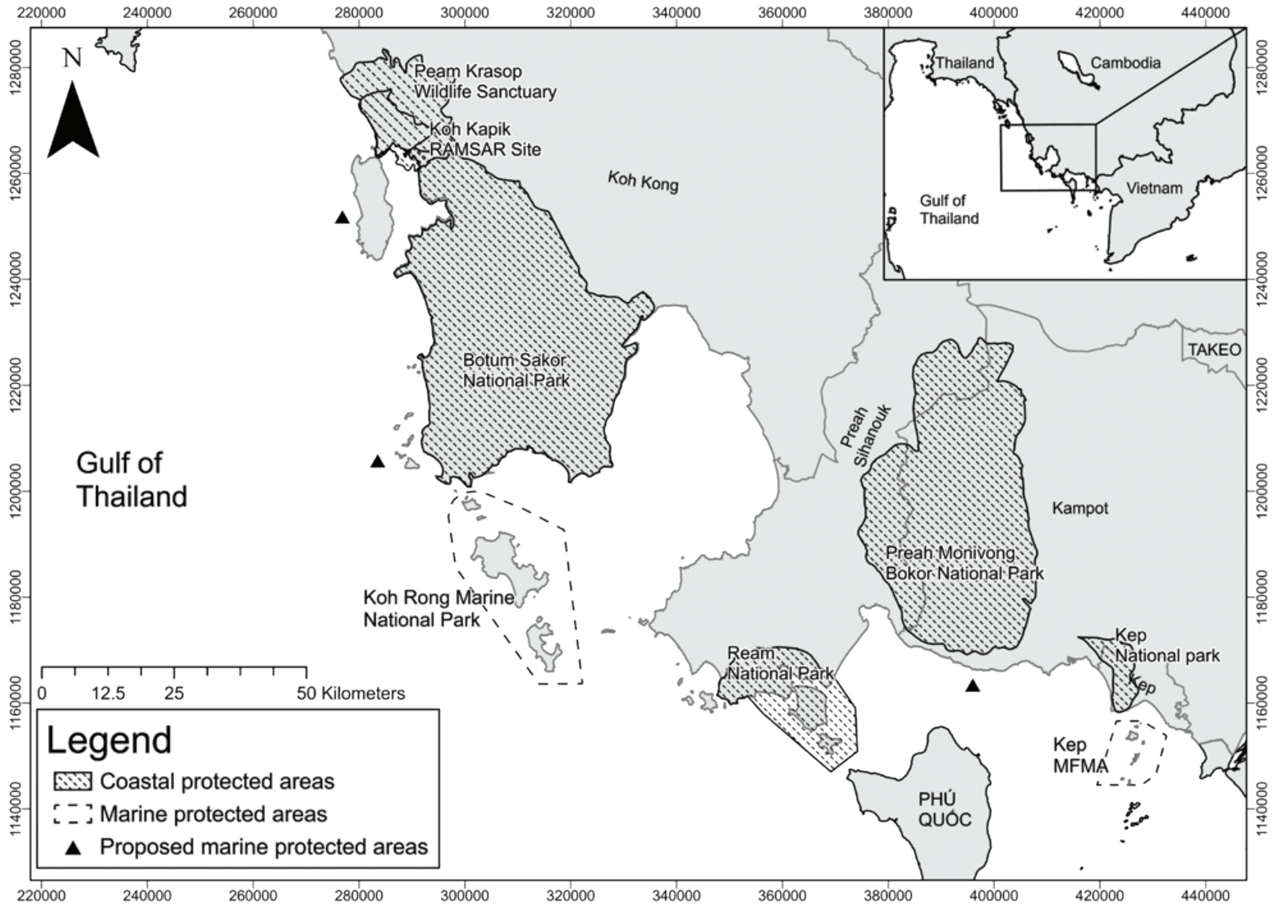

FIGURE 1: *The Cambodian coastline and offshore islands. Marine protected areas are outlined with a hashed border.*

actively involved in coral reef monitoring will submit data to the national database bi-annually. Lastly, analysis of key indicators (i.e., hard coral cover, parrotfish, and grouper abundance, etc.) will be conducted annually by the organization stewarding the database until handover to the Fisheries Administration, Royal Government of Cambodia. Results from this analysis will be used to track changes in Cambodian coral reef status and will be presented during the CCRMN annual meetings.

Monitoring programs that assess coral reef habitat with regularity are a comparatively new concept in Cambodia. Unlike other nations in the Indo-Pacific region, there are no multi-decadal datasets from which to draw temporal coral reef data to assess long-term trends. Data submitted from Cambodia for the GCRMN East Asia regional analysis incorporated datasets from the nation's two longest-running monitoring programs encompassing two study sites: The Koh Rong and Koh Sdach archipelagos. Developed to support the design and establishment of Marine Protected Areas, these monitoring programs utilize similar survey protocols with only marginal differences. Both programs initially surveyed many sites to characterize the marine habitats of their respective archipelagos before selecting permanent monitoring sites for regular and long-term monitoring.

The first dedicated coral reef monitoring program in the Koh Rong archipelago was initiated by Coral Cay Conservation in 2010, following an invitation to establish operations at the location from the Fisheries Administration. A large-scale habitat assessment program was established, and in total, 156 sites were surveyed over multiple years. Following the withdrawal of Coral Cay Conservation in 2015, this monitoring program was then continued by the Song Saa Foundation in 2015 with support from Fauna & Flora International, with monitoring effort refined and 18 permanent monitoring sites selected. These partner organizations utilized the methodology outlined in the agreement by Thorne and Longhurst (2013), which is a modified version of the Reef Check method whereby four 20 m pseudo-replicate transects are deployed at each survey site along a standardized depth of 6 m. Surveyors recorded information on benthic assemblage composition, fish abundance to species level and conducted length estimation for Scaridae spp. and Serranidae spp. Substratum composition was assessed using point intercept transects. The substrate composition was determined using the point-intercept transect methodology. Substratum directly under the transect tape at 50 cm intervals was identified to a pre-divined substrate category. The proportion of each substrate category (i.e., hard coral, soft coral, rock, etc.) was calculated as a percentage for each pseudo-replicate, with the mean percentage per site subsequently calculated between the four pseudo-replicates from every survey site. All target indices were modified to ensure they were appropriate for Cambodian coral reefs, with target fish species and invertebrate families altered to reflect this. In 2016, the Koh Rong archipelago was declared a multiple-use MPA, known locally as a Marine Fisheries Management Area, and further upgraded to Marine National Park in 2018. Monitoring programs in the Koh Rong archipelago aim to track the effectiveness of MPA management through biophysical indicators.

The Koh Sdach archipelago consists of nine islands situated in Koh Kong Province. The NGO Shallow Waters established and operated the first coral reef monitoring program in the area from 2013 to 2014, in partnership with the University of Southampton. No coral reef surveys had occurred at this location since a field expedition led by the National University of Singapore and the Fisheries Administration in 2002 (Chou et al. 2003). Surveys utilized a modified version of the Reef Check methodology (Savage et al. 2014). However, to avoid the risks of pseudo-replication, results from each 20 m transect were combined to create a single 80 m transect. Key fish families and invertebrates were identified to species level, and those with economic or ecological importance were further categorized by size. Information was collected on the substrate type with hard coral classified according to morphological growth form and genera. Hard coral cover was assessed utilizing the aforementioned point-intercept methodology. However, substratum was identified every 20 cm rather than 50 cm for data collection at a higher resolution. This monitoring program was inherited by the NGO Projects Abroad in 2014. Surveys sites were narrowed down to fifteen permanent monitoring sites to ensure all sites could be feasibly surveyed annually, and the methodology was modified to accommodate less-experienced volunteer surveyors. Transect length was reduced to one 50 m transect per site with fish data recorded to family level and hard coral data no longer incorporating genera or morphological growth form. All data from the Koh Sdach archipelago submitted to the regional analysis only incorporated the 15 sites utilized by Projects Abroad.

Status & Trends

Hard coral cover in Cambodia has gradually increased annually in all datasets submitted for this regional analysis (Figure 2a). Thermal stress-induced bleaching observed in 2016 had no perceivable influence on the increasing trend of hard coral cover. Non-uniform survey effort may partially explain the increase in hard coral reflected in the data, with certain years surveyed heavily in comparison to others. In particular, the survey-intensive years of 2015 and 2016 (Figure 2a) were not replicated in effort the subsequent years. If the survey effort was equal in 2017 and 2018, an increase in hard coral cover might have still been observed; however, this could have been less pronounced and more gradual, as observed during follow-up surveys in 2019 (Fauna & Flora International 2020, Glue and Teoh 2020). Additionally, data analyzed here for 2017–2018 represented surveys from Koh Sdach only, which overall has higher coral cover than Koh Rong; therefore, the national trend may be disingenuously inflated due to the absence of data from one site (Fauna & Flora International 2020, Glue and Teoh 2020). On a national scale, it would appear that anthropogenic impacts are not having a perceivable effect on hard coral cover.

A measured decline in hard coral cover was exhibited along the depth gradient of the reef (Figure 2b). However, the decline in hard coral cover was not dramatic, potentially due to the narrow range of depths which surveys have been conducted. Light availability plays a vital role in determining hard coral cover (Chow et al. 2019); the high turbidity of sites surveyed in Cambodia will regulate the diversity of coral genera present at depth and result in declining hard coral cover along the depth gradient. Due to the shallow depth profile of Cambodian reefs and a selective survey depth for monitoring programs, a large majority of surveys are conducted at a pre-determined depth of 5 m (Figure 2b). The small number of surveys conducted at deeper sections of reefs may have resulted in a misrepresentative trend. Therefore, additional surveys must be conducted at deeper reef sites where possible, before any trends in coral cover can be ascertained. However, the shallow topographic nature of Cambodian reefs limits the number of sites where surveys at deeper areas of the depth gradient can be conducted. Limiting the sample size of deeper reef surveys for further research.

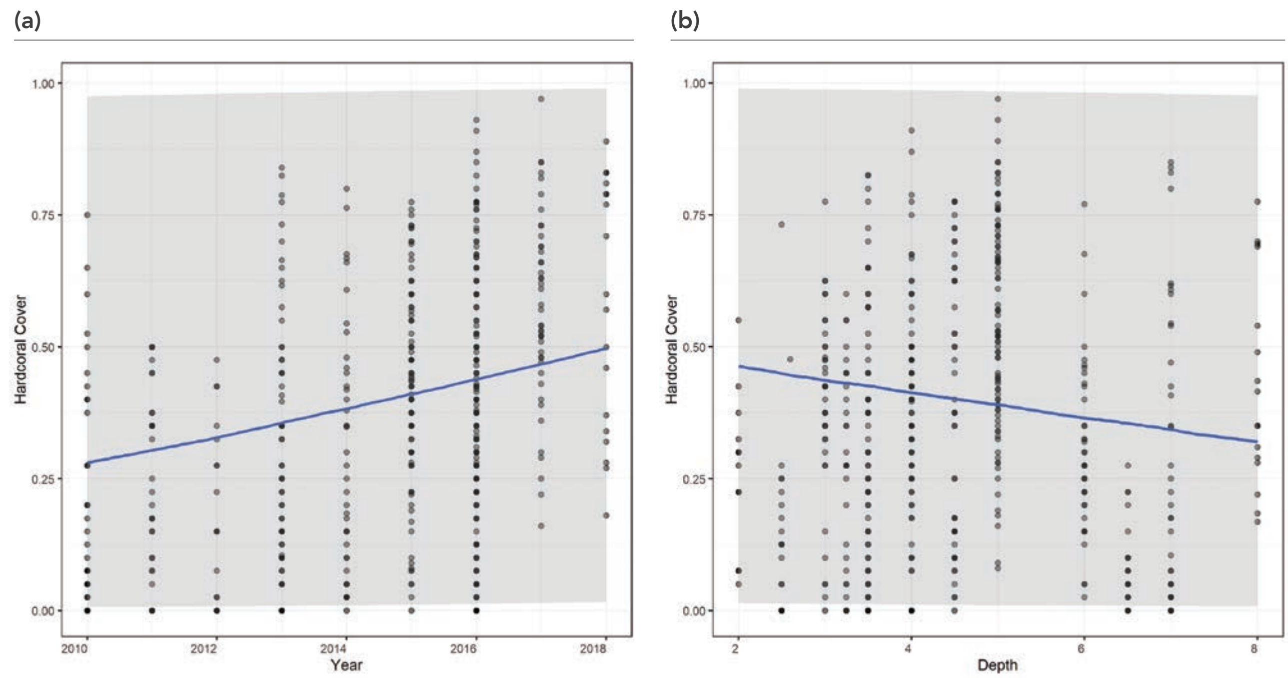

FIGURE 2: *Cambodia coral cover across (a) years and (b) depths.*

Coral Bleaching 2016

The third global-scale coral reef bleaching event, which occurred from 2015 to 2017, impacted all coral reef bioregions (Hughes et al. 2017). Historically, Cambodian coral reefs have not been immune to the influence of thermal stress events. Due to the absence of regular coral reef monitoring programs prior to 2010, the mass bleaching event observed from May to October 2010 was the most severe recorded in the country to date (Van Bochove et al. 2011). During the global bleaching event from 2015 to 2017, coral colonies on Cambodian reefs exhibited bleaching in response to prolonged thermal stress (Teoh 2018). Nominal bleaching was observed during 2015, with bleaching categorized as minimal[1] occurring at 57% of sites (Teoh 2018). Of the 17 sites surveyed in 2016, considerable levels of coral bleaching were observed. Surveyors encountered 47% of sites with moderate[2] bleaching and 24% of sites severely[3] bleached (Teoh 2018). The extent of mass mortality caused by this bleaching event remains unknown, as no follow-up surveys were conducted until 2019. Due to this temporal gap between surveys, it is difficult to ascertain whether any substratum categorized as recently killed coral recorded during 2019 surveys resulted from the 2015 to 2017 global bleaching event. However, there was no overall decline in coral cover observed between 2016 and 2019; therefore, any impact on the hard coral assemblage may have been minimal (Glue and Teoh 2020).

Drivers & Pressures

Coral reefs in Cambodia are rarely subjected to large-scale physical disturbance events such as severe storms and typhoons that impact other nations in the region. Outbreaks of crown-of-thorns starfish (COTS) have never been observed, despite their occurrence on Cambodian reef systems. However, the bleaching of coral colonies is often observed annually due to a build-up of thermal stress, which transpires towards the end of the dry season, with the shallow Gulf of Thailand acting as a thermal sink.

The primary threats to coral reefs and other marine habitats in Cambodia are localized stressors, including illegal unregulated and unreported fishing, unregulated development, sedimentation, nutrient enrichment, and marine litter. A rapidly growing population is increasing strain on marine resources, and aquatic-sourced protein comprises 76% of the nation's daily protein intake (IFREDI 2013). Presently, aquatic-sourced protein is primarily obtained from the Tonle Sap Lake, one of the largest freshwater fisheries in the world, comprising 80% of Cambodia's fisheries sector (Daly et al. 2019). However, as the nation becomes wealthier, domestic demand for seafood products has been on the rise.

Illegal, unregulated, and unreported fishing and destructive fishing practices are two of the primary threats to marine ecosystems in Cambodia. Highly destructive fishing techniques such as bottom trawling are commonplace (Reid et al. 2019), and dynamite fishing still occurs at remote outer islands where enforcement is lacking. Despite a ban on trawling within MPAs and in depths shallower than 20 m, incursions of these areas by trawlers remain a threat due to inadequate and low-capacity enforcement (Fisheries Administration 2018). The aforementioned fishing practices cause degradation of reef fish nursery grounds, increase turbidity due to substrate disturbance and directly destroy coral reef habitat. Although bottom-trawling is not practiced directly on coral reefs and incidences of dynamite fishing are infrequent, signs of overfishing are ubiquitous on Cambodian reefs, with low biomass of key indicator fish families for functional ecosystems (Fauna & Flora International 2020, Glue and Teoh 2020). Additionally, the true scale of the Cambodian fisheries sector remains unaccounted, due to an absence of fisheries catch monitoring and regulation, leaving marine resources open to continued overexploitation (Gillett 2004, Teh et al. 2014).

The international and domestic tourism sector in Cambodia is undergoing a rapid transition, with an increasing focus on coastal tourism, which

[1] Minimal bleaching = At least one bleached coral colony observed

[2] Moderate bleaching = >5% to 30% population level bleaching

[3] Severe bleaching = >30% population level bleaching

is rapidly driving land-use change in coastal provinces (Prak, Nay and Belyn, 2018). Vast tracts of mainland and island coastline have been leased for development, which has often progressed unchecked and unregulated. Insufficient measures have permitted increasing quantities of surface run-off and untreated wastewater to enter the marine environment unimpeded, creating the potential for harmful algal blooms to smother reefs and other marine habitats through nutrient enrichment.

Despite this, efforts to mitigate local threats have been increasing over the past five years. Awareness raising, fisheries management, and community-level monitoring, control, and surveillance (MCS) initiatives have begun, and momentum towards the development and effective management of MPAs is building. However, similar to many other low-income nations, lack of governmental capacity hinders the efficient enforcement of MPAs. Additionally, attempts are being made to diminish stressors generated from rapid unregulated coastal development through changes to the provincial government resulting in stricter enforcement against polluters.

RECOMMENDATION

Cambodian coral reefs would greatly benefit from management efforts that target local and national-level stressors. Current methods of community-led enforcement have been successful in preventing further degradation. However, community patrol teams cannot deter medium to large-sized fishing vessels due to a lack of enforcement ability, small-sized boats, and limited human capacity. Raising adherence to MPA restrictions by medium and large-sized fishing vessels would require closer collaboration and input from governmental ministries. Specifically, trained law enforcement officers with better-equipped boats and closer ties to the Cambodian navy would be needed to mitigate illegal fishers.

As the establishment of MPAs in Cambodia moves forward, it is imperative that these are developed with collaborative partnerships in mind and management plans stringently enforced. Development cannot go unchecked within MPAs, and governmental ministries such as the Ministry of Environment need to monitor the influx of wastewater into the marine environment. Especially for island environments where sewage systems are not present, and development continues to increase.

For robust assessments on the condition of coral reefs and the efficiency of the MPAs designed to protect them, ensuring there are no significant gaps in monitoring is vital. However, this cannot always be avoided, as in 2020, with the canceling of the annual coral reef monitoring field trips due to the COVID-19 pandemic. Furthermore, routine monitoring of remote outer islands should be established through collaborative partnerships between NGOs, governmental ministries, and the military forces that administer these remote sites. Through routine monitoring of these locations, it can be ascertained if fish and hard coral assemblages differ from those in closer proximity to the mainland and the efficiency of areas under marine management, as these locations remain unmanaged. Additionally, measures should be implemented to ensure marine habitat monitoring programs are sustainable through improved collaborations between NGOs, academic institutions, and the private sector and long-term financing mechanisms for MPAs that can support marine habitat monitoring.

DATA CONTRIBUTORS

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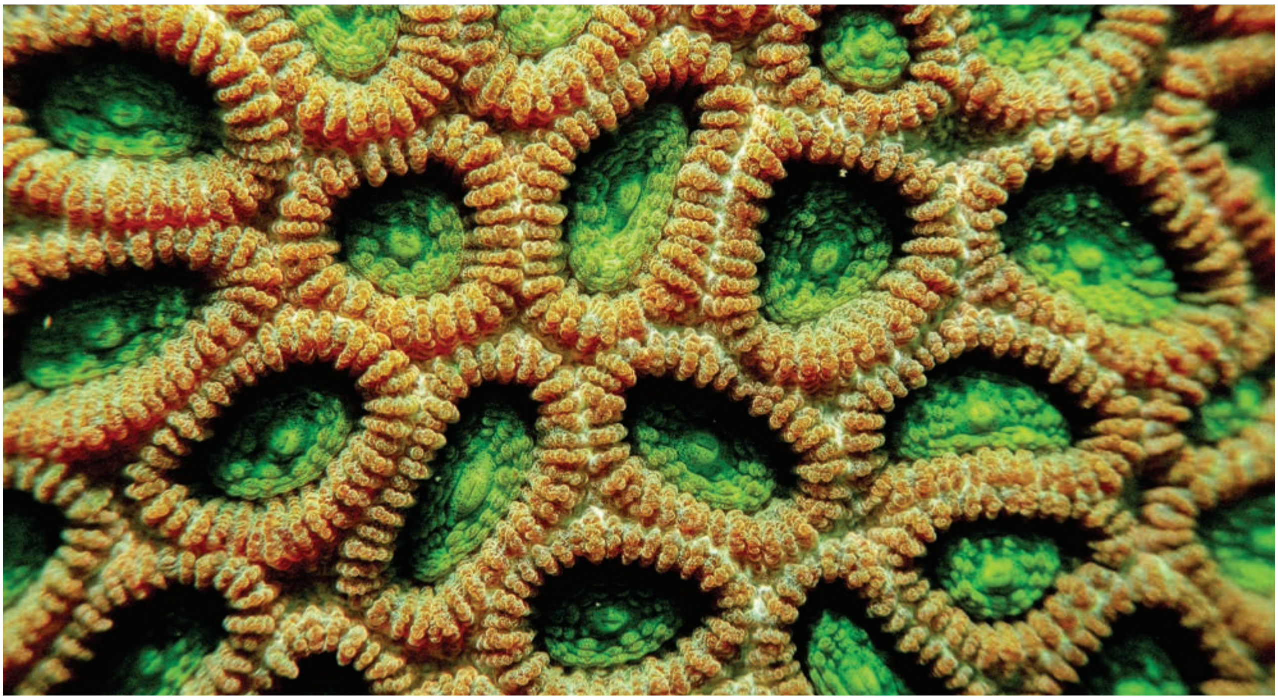

SOUTHEAST ASIA

Indonesia

Introduction

Indonesia is the largest archipelagic country consisting of 17,508 large and small islands (United Nations 2020), with a total coral reef area of 25,000 km^2 comprising various reef types, namely fringing reefs, barrier reefs, patch reefs, and atolls (Asian Development Bank 2014, Badan Informasi Geospasial 2013). Geographically, Indonesia is intersected by the equator, which results in a tropical climate throughout the archipelago, and makes Indonesian waters ideal for coral reefs to grow and develop.

Coral reef typologies and characteristics vary across the entire Indonesian archipelago from west to east and south to north. Coral reefs in the western and southern parts of Indonesia face the Indian Ocean. They are characterized by shallow fringing reefs, wide flat reefs, and mostly sloping patch reefs. In contrast, the central and eastern coral reefs are affected by the Pacific Ocean, traversed, and separated by the Indonesian Through Flow, and are characterized by deep-water fringing reefs with narrow reef flats and steep slopes, as well as several types of barrier reefs and atolls. This central and eastern region, together with several areas in the neighboring countries, including Malaysia, Papua New Guinea, the Philippines, Solomon Islands, and Timor-Leste, is known as the Coral Triangle, the center of marine biodiversity, (Hughes et al. 2002). Previous estimates of scleractinian coral richness in the Coral Triangle have ranged from 537 to 600 species (Turak and Souhoka 2003, Veron et al. 2015).

Spalding et al. (2007) has highlighted Indonesian waters as the center of coral biodiversity in the Indo-Pacific. Overall, Indonesia has the highest diversity of coral species in the world. In total, there are 569 species of corals (about 69% of the world's species) from 80 genera (about 76% of the world's genera) recorded in Indonesia (Suharsono 2017). These include up to 91 *Acropora* species out of 113 species in the world (Suharsono 2017). There are also endemic coral species, such as *Acropora suharsonoi*, *Indophyllia macassarensis*, *Isopora togianensis*, and *Euphyllia baliensis* (Giyanto et al. 2017) (Figure 1).

Coral reef monitoring in Indonesia has been carried out for several decades, but only more intensively in the last two decades since the establishment of the national Coral Reef Rehabilitation and Management Program (COREMAP), specifically COREMAP Phase 2 in 2004. This program's ultimate goal is to manage coral reef ecosystems in Indonesian waters to be sustainable and continue to benefit local

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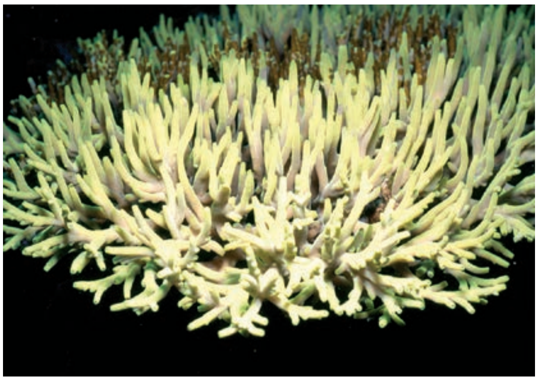

Acropora suharsonoi Photo: A. Budiyanto

Indophyllia macassarensis Photo: A. Budiyanto

Isopora togeanensis Photo: M. Abrar

Euphyllia baliensis Source: Turak et al. (2012)

FIGURE 1: *Endemic coral species in Indonesia.*

communities. This national program regularizes coral reef monitoring activities in Indonesia, with many observations covering a wider area. During the COREMAP, a total of 37 monitoring locations were surveyed from 2014 to 2019; 13 of which belong to marine conservation areas (MCA) (see Table 1), where human activity is minimal, while the rest are non-MCA. However, several considerations did not allow for all of the survey activities to be carried out in all of the locations within the same year, including (i) the large number of sites observed, (ii) limited transportation and access to the locations, and (iii) limited trained personnel for monitoring coral reefs.

At the beginning of coral reef monitoring activities in Indonesia, qualitative field data were collected through visual observations, which then developed into quantitative methods using the Line Intercept Transect Method. Starting in 2015, coral reef monitoring activities at COREMAP began using the Underwater Photo Transect (UPT) method. Data have been collected by taking photos perpendicular to the substrate along a 50-meter transect line, with 1 m intervals between images. Thus, a total of 50 photos will be obtained for each observation station. The area for each photo is approximately 2,500 cm^2. The photographs are then analyzed and processed using the Coral Point Count with Excel extensions (CPCe) program (Kohler and Gill 2006) to calculate the percentage cover from each biota category and substrate in the coral reef ecosystem, including live coral cover.

Status & Trends

Overall, the status and trends of coral reefs in Indonesia show a slight general increase in live coral cover from the late 1980s and early 1990s compared to post-2010 (Figure 2a). In addition, two stages can be identified. In the first stage, live coral cover increased in two folds from ~20% in the late 1980s and reached the highest point (>50%) in the 1990s. Since then, live coral cover has fallen and reached its lowest point after 2010. Currently (2019), the percentage of live coral cover is more stable, averaging around 30%.

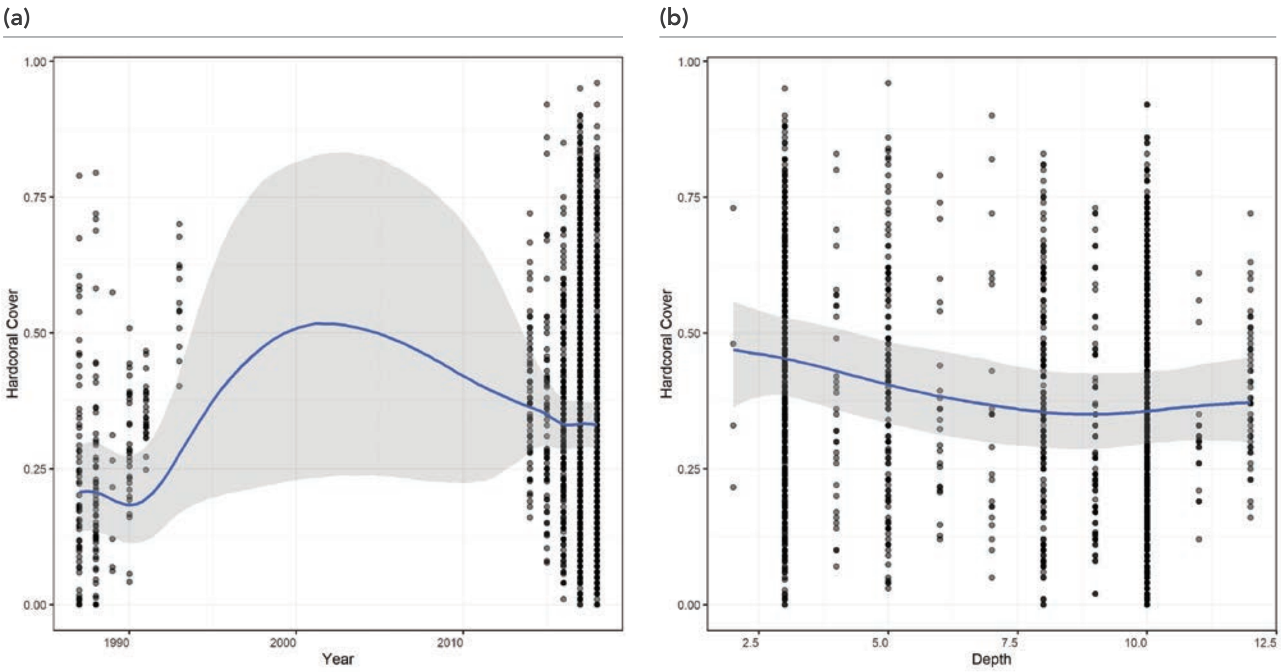

FIGURE 2: *Indonesia coral cover across (a) years and (b) depths.*

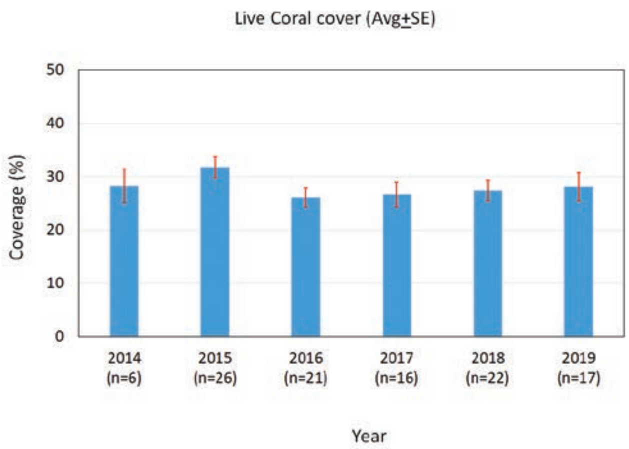

FIGURE 3: *The average live coral cover and its standard error value for each year of observation.*

Live coral cover was shown to decline with increasing depth (Figure 2b). Optimal coral cover is in shallow waters, around a depth of 2 m. At a depth of about 7 m, it appears that coral cover is relatively stable. Coral reefs in the western parts of Indonesia are characterized by large reef flats with gentle and shallow reef slopes. In contrast, the Eastern parts of Indonesia, where reef flats are narrower with steep and deep reef slopes.

A summary of the coral reef conditions at the monitoring locations is shown in Table 1. The condition of coral reefs is determined by the percentage of live coral cover (HC), and it can be divided into four categories: Poor (0%<HC≤25%), Fair (25%<HC≤50%), Good (50%<HC≤ 75%), and Excellent (75%<HC≤100%). The results of the COREMAP monitoring activities from 2014 to 2019 show that the percentage of coral cover in Indonesia is relatively stable, with live coral cover ranging from 26–32% (Figure 3).

Coral cover increased from 2014 to 2015 and decreased from 2015 to 2016. After 2016, there has been an upward trend in coral cover. The increase in live coral cover from 2014 to 2015 shows that natural environmental conditions support coral development, and there was less pressure on coral reefs. The decrease in coral cover from 2015 to 2016 was due to increased seawater temperatures, which resulted in coral bleaching throughout much of Indonesian waters. In the western Indonesia waters, the coral bleaching event in 2016 occurred in West Sumatra, in areas such as North Nias, Central Tapanuli, Mentawai Islands, TWP Pieh Island, and the surrounding areas. In general, after the bleaching event, recovery in this region has been slow because of the Indian Ocean's relatively large waves. Such local natural conditions may not support rapid coral recovery from impacts. Other locations, such as the Mentawai Islands and Natuna Islands, also show a decreasing trend of live coral cover before the bleaching event up to 2018 (Table 1). Thus, monitoring of coral reefs after coral bleaching events in western Sumatra shows that the condition of its coral reefs has not improved from the "poor" criteria and has gotten worse in some locations. In contrast, Sumatra's eastern regions, such as Bintan, Batam, Belitung, and Lingga (Table 1), located in the Natuna Sea, are shallow-water coral communities tolerant of warmer surface temperatures and the more turbulent waters around small islands. These natural conditions make bleaching less impactful, where there was a slight increase in live coral cover and stable reef conditions before and after the coral bleaching event.

The coral bleaching event in 2016 also impacted several locations in the central part of Indonesia, such as Central Buton, Wakatobi, and the Selayar Islands. For the last two sites, although there was no change in the category of coral reef conditions, coral bleaching has caused a decrease in coral cover (Tuti et al. 2017, Yusuf et al. 2017). The decline was more visible in the following year (2017), where the condition of the coral reef, which was initially in "moderate" condition, turned "poor." However, coral recovery in both locations was observed in 2018 with the change of coral reef conditions back to "moderate." The coral reef recovery process in central Indonesia was faster than in the western parts of Indonesia. This recovery was demonstrated by the improving coral reef condition in the central parts of Indonesia from "poor" to "moderate" within one or two years. It may be related to better water quality, hydrodynamic conditions, and relatively deeper waters in the central parts of Indonesia compared to the western parts of Indonesia.

TABLE 1: *A summary of the coral reef condition* at the monitoring location of COREMAP.*

NO.	LOCATION	2014	2015	2016	2017	2018	2019
1	Batam	--	Medium	Medium	Medium	Medium	--
2	Belitung	--	Medium	Medium	Medium	Medium	--
3	Biak Numfor	--	Poor	Poor	Poor	Poor	--
4	Bintan	Medium	Medium	Medium	Medium	Medium	Medium
5	Buton	--	--	Medium	Medium	Medium	--
6	Kendari	--	Medium	Medium	Medium	Medium	--
7	Kepulauan Derawan	--	--	--	--	--	Medium
8	Kepulauan Selayar	--	Medium	Medium	Poor	Medium	Medium
9	KKPN SAP Kepulauan Aru Tenggara (MCA)	--	--	Medium	--	--	--
10	KKPN SAP Raja Ampat (MCA)	--	Medium	--	--	--	Medium
11	KKPN SAP Waigeo Barat (MCA)	--	Medium	--	--	--	--
12	KKPN TNP Laut Sawu (MCA)	--	Poor	--	--	--	--
13	KKPN TWP Gili Matra (MCA)	Poor		--	--	--	--
14	KKPN TWP Kapoposang (MCA)	--	Medium	--	--	--	--
15	KKPN TWP Kepulauan Anambas (MCA)	--	Medium	--	--	--	Medium
16	KKPN TWP Kepulauan Pieh (MCA)	Medium	--	--	--	--	Medium
17	KKPN TWP Laut Banda (MCA)	--	Medium	--	--	--	
18	KKPN TWP Padaido (MCA)	--	Medium	--	--	--	Medium
19	TN Komodo (MCA)	--	--	--	--	--	Medium
20	Lampung	--	Medium	Medium	Medium	Medium	Medium
21	Lingga	--	Medium	Medium	--	Medium	--
22	Makassar	--	Poor	Poor	Poor	Medium	--
23	Mentawai	Medium	Poor	Poor	--	Poor	Poor
24	Merauke	--	--	--	--	--	Poor
25	Natuna	Poor	Poor	Poor	--	Poor	Poor
26	Nias Utara	Poor	Medium	Poor	Poor	Poor	--
27	Pangkep	--	Medium	Medium	Medium	Medium	--
28	Raja Ampat	--	Medium	Medium	Poor	Poor	Medium
29	Sabang	--	--	--	--	Medium	--
30	Sekotong, Lombok	--	Medium	Poor	Poor	Poor	Poor
31	Sikka	--	Poor	Poor	Poor	Poor	--
32	Sumba	--	--	--	--	Poor	--
33	TN Takabonerate (MCA)	--	--	--	--	--	Poor
34	Tapanuli Tengah	--	Medium	Poor	--	Poor	--
35	Ternate, Tidore & Halmahera barat	--	Medium	Medium	Medium	Medium	--
36	Tual	--	--	--	--	--	Medium
37	TN Wakatobi (MCA)	--	Medium	Medium	Poor	Medium	Medium
	Average	**Medium**	**Medium**	**Medium**	**Medium**	**Medium**	**Medium**

***NOTE: Live coral cover (HC) Category** Poor: 0%<HC≤25% Fair: 25%<HC≤50% Good: 50%<HC≤75% Excellent: 75%<HC≤100%

Coral Bleaching 2016

Coral bleaching in Indonesia has occurred several times, but the most significant impact occurred from March to June 2016. Coral mortality rate between 30% and 90% was recorded in the waters of East Nusa Tenggara, West Nusa Tenggara, South Java, West Sumatra, North Bali, Lombok, Karimun Java, and Selayar. Since the first recorded massive coral bleaching event in 1983, the intervals between periods of successive bleaching events in Indonesia have shortened, i.e., 14–15 years, 12 years, and only 6 years, between the four severe bleaching events. Coral bleaching also tends to be more widespread and intense, resulting in dramatic changes to the reefs and leading to coral extinction (Hoegh-Guldberg 1999). For example, Pandolfi (1999) reported the rapid extinction of two species of widespread Caribbean corals (*Pocillopora* cf. *palmata* and *Montastraea annularis*), which coincide with the temperature rise impacted in the reduction of the sea level at the last glacial maximum (18,000 years BP). In Indonesia, Wouthuyzen et al. (2017) reported 20 out of 34 provinces had severe impact due to increased surface water temperature. The highest degree heating week was >8°C, resulting in up to 90% of corals bleaching in Bali and Lombok waters.

FIGURE 4: *(Left): Bleaching event at KKPN TWP Kepulauan Pieh (MCA) in April 2016 (Photo: KKPN TWP Kepulauan Pieh Islands); (Right): The Mentawai Islands in May 2016 (Photo: M. Abrar).*

FIGURE 5: *After bleaching in North Nias in September 2016 (Photos: Rikoh M.S.).*

Drivers & Pressures

Disturbance to coral reefs will adversely affect the surrounding natural ecosystems. Impacts on coral reefs in Indonesia are caused by various factors due to natural and anthropogenic activities. Some of the coral disturbances reported, among others, include strong storms, land base pollution, coral mining, destructive fishing practices, and global climate change, such as rising seawater temperatures. Coral reef ecosystems can experience a variety of disturbances simultaneously. Cleary et al. (2014) reported that coral reefs in the Thousand Islands, Jakarta, had severe effects caused by various disturbances, including river discharge, urban development, tourism, destructive fisheries, and coral mining.

Among the threats to coral reefs, the impact of climate change is now the most dominant because it will trigger coral bleaching and can cause coral death. Coral bleaching is the most severe threat because it cannot be directly prevented, and its impact can last for several years. In unhealthy waters, coral bleaching can occur more quickly and disrupt the coral recovery process, for example, in the western waters of the island of Sumatra. There, bleaching occurred very quickly, but the recovery process was prolonged. This recovery process is different from the existing coral reefs in Eastern Indonesian waters, which generally have healthier water quality. The bleaching impact is still small and is thus faster than in Western Indonesia.

Crown-of-thorns starfish (COTS) outbreaks also cause declining coral health and threaten coral reefs (Mathiew et al. 2012). Nutrient enrichment and increasing temperatures trigger COTS outbreaks (Lane 2012). Several monitoring locations were affected by COTS outbreaks, including Tapanuli Tengah, Pieh, Kendari, and Sikka regency (Siringoringo et al. 2015, Pramudji 2017). Such COTS outbreaks hinder the ability of corals to recover from bleaching events. Figure 6 shows the situation after a coral bleaching event and COTS predation on corals in Kendari.

Although rare, intense storms can cause coral damage. A strong storm followed by strong waves can break off coral branches and overturn massive corals. With their growth disrupted, they may die after a few weeks, and algae may overgrow their skeleton. The occurrence of a strong storm in Biak Numfor (Papua) in 2009 caused damage to coral reefs in the waters of Owi Island, Biak Numfor (Giyanto 2017).

Land-based pollution mainly occurs in waters close to big cities, seaports, large river mouths, and residential areas and farming grounds. The coral reefs around the waters of Jakarta, Makassar, Manado, and several other big cities in Indonesia have generally decreased due to human activities occurring on the mainland.

Damage to coral reefs caused by coral mining generally occurs in coastal areas where people use coral as a building material. Coastal communities, especially those far from cities, prefer to use corals from their surrounding areas, which are cheaper than other commercial alternatives.

Destructive fishing practices such as bombing, cyanide poisoning, and trawling are considered illegal in Indonesia. Even so, harmful fishing practices are often reported in Indonesian waters, especially in areas far from the city center where surveillance is sub-optimal (Figure 7). On the other hand, public and community awareness campaigns for coral reef sustainability, carried out during COREMAP Phase 2, can reduce destructive fishing; nonetheless, destructive fishing practices still occur in all the COREMAP sites (Widayatun and Hidayati 2012).


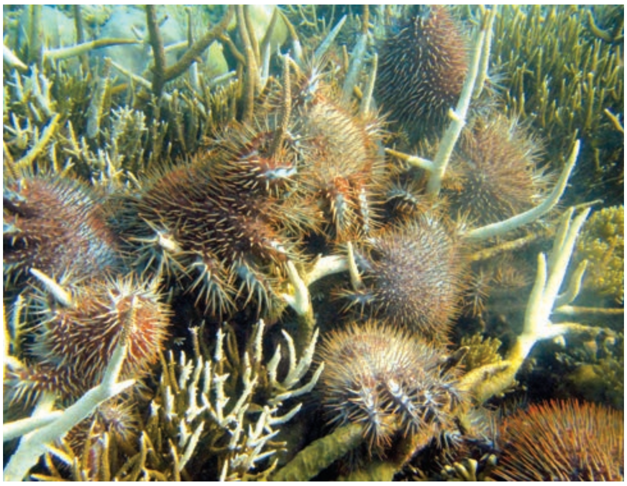

FIGURE 6: *COTS attack at Kendari in 2017 (Photo: M. Abrar)*


FIGURE 7: *Destructive reef fishing (bombing) at Makassar Reefs in 2018 (Photo: N.W.P Sari)*

RECOMMENDATION

Information on the status and trends of coral reef monitoring results is a useful tool for assessing or evaluating the achievement of coral reef management in Indonesia. As the frequency of bleaching events increases and human impacts become more intense, governments must accelerate the creation of MCAs with appropriate zoning systems, especially in eastern Indonesia, where destructive fishing practices are still prevalent. Although bleaching events are inevitable, minimizing the human factors to allow the reefs to recover naturally is a feasible option to save the reefs. With intense and continuous education and socialization about the importance of coral reefs through public awareness campaigns, the community can be a valuable asset that can contribute actively to ecosystem conservation. Law enforcement should also be applied fairly for people who violate regulations that cause damage to coral reef ecosystems. Reef restoration, such as those carried out in Bali, Karimun Jawa, should also be carried out in other locations when natural recovery is unsuccessful. Lastly, national support from all relevant stakeholders is vital to integrate the efforts by government authorities and communities to manage the coral reef ecosystem.

DATA CONTRIBUTORS

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SOUTHEAST ASIA

Malaysia

Introduction

Malaysia is composed of two land areas separated by the South China Sea: Peninsular Malaysia on the Asian continent and East Malaysia (the states of Sabah and Sarawak, and the Federal Territory of Labuan) on the island of Borneo (Figure 1). The Strait of Malacca (SOM) lies on the west coast of Peninsular Malaysia, while the east of Sabah is bounded by the Sulu and Sulawesi seas.

Approximately 90% of the reefs in Malaysia are in East Malaysia (Choo, P. L. et al. *unpublished manuscript*) and some of the most diverse reefs are on the northern and eastern coast of Sabah (Burke et al. 2011, 2012). The east coast of Sabah is within the scientific boundary of the Coral Triangle (Veron et al. 2009, 2011), the area with the highest marine biodiversity in the world (Hoeksema, 2007). The Sunda Shelf ecoregion of the South China Sea, east of Peninsular Malaysia and west of Sarawak, is considered to be part of the Coral Triangle (Veron et al. 2015) as it has been documented to have 571 species of scleractinian (stony) corals (Huang et al. 2015). Large concentrations of reefs are found at the Lahad Datu-Semporna area on the east coast of Sabah and the Kudat-Pitas area in the north. Other important reef areas can be found off Sandakan and several locations on the west coast of Sabah. In Sarawak, reefs are located off the coast of Miri (Miri-Sibuti), Bintulu (Luconia), and Kuching (Talang-Satang and Tanjung Datu) (Shabdin 2014). For Peninsular Malaysia, the reefs in the SOM are few and far between. They are generally located in turbid waters except for Pulau Payar (pulau=island), which is the only gazetted Marine Park in the SOM. Most of Peninsular Malaysia's clear water reefs are fringing the offshore islands in the South China Sea.

The coral reefs of Malaysia are primarily fringing and patch reefs. In Sabah, the Semporna area has the most diverse reef habitats. Within the Tun Sakaran Marine Park, north of Semporna, there is part of an atoll with three islands forming the northern rim of a volcano, while the southern rim is submerged underwater (Wood 1987). South of Semporna, a chain of reefs and small islands form a barrier reef system (Wood 1978) at the edge of the Borneo continental shelf (Wood 1987). Just south of the Borneo continental shelf lies Pulau Sipadan, one of two oceanic islands in Malaysia; the other oceanic island, Pulau Perak, lies in the Strait of Malacca.

In Peninsular Malaysia, Reef Check Malaysia (RCM) began coral reef monitoring in 2000, and in East Malaysia, it was started in 2008. Before that, surveys were carried out on a short-term project basis or during one-off marine expeditions. This report's analysis is mainly based on the data collected using the Reef Check methodology from 2010 to 2018, at up to 220 reef sites with 6,841 surveys on Malaysian reefs with most of the sites being permanent monitoring sites. Other small datasets were from independent groups on marine park islands.

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FIGURE 1: *Map of Peninsular Malaysia and East Malaysia (Sabah and Sarawak) showing the coral reef areas in yellow (Coral Triangle Atlas, 2021).*

Status & Trends

In general, Malaysian reefs had nearly 50% hard coral cover before 1990 and declined to almost half of that by 2010 (Figure 2). Surprisingly after the mass bleaching event of 2010, the hard coral cover recovered to nearly 40% in 2016. Another mass coral bleaching event occurred in 2016, and the coral cover decreased to about 30% and has not shown any significant recovery as of 2018. However, based solely on RCM data, Malaysian reefs in 2018 had a mean coral cover of 42.2% (Reef Check Malaysia 2018). This discrepancy in results might be due to the fact that Reef Check surveys are usually done on the best reefs and only a subset to the dataset used in this report. Expectedly, Malaysian coral reefs show lower mean hard coral cover with depth, ranging from over 40% on reefs shallower than 4 m to about 25% down to a depth of 15 m due to lower light quality (Figure 2).

The mean live coral cover of Peninsular Malaysia's reefs has fluctuated between 30% to 45% from 2010 to 2018 and has maintained slightly over 40% (Figure 3). The recovery and slight increase after the 2010 mass coral bleaching event was negated by the 2016 bleaching event, and the coral cover has remained relatively constant since then (Reef Check Malaysia 2018).

In East Malaysia, hard coral cover was stable between 2010 and 2015 following a decline between 2016 and 2018 (Figure 4). Between 2010 and 2015, the average hard coral cover was 38% and ranged from 34% to 41%. The percentage cover declined to 29% in 2016, which coincided with the bleaching event on several reefs (Reef Check Malaysia 2017) and the crown-of-thorns starfish (COTS) outbreak on Lankayan Island, Sabah, where more than 10,000

COTS were removed from the infested reefs (Reef Guardian 2017). Coral cover improved to 34% in the following two years. Nutrient indicator algae cover from 2010 to 2018 was generally low, with an average of 3% cover and was highest in 2016 (6%).

The reefs of Sabah had the highest rubble cover (19%), followed by Labuan (7%) and Sarawak (2%), which reflect the damage caused by blast fishing in the former. Blast fishing has also been reported in Labuan (Awang and Chan 2013).

(a) (b)

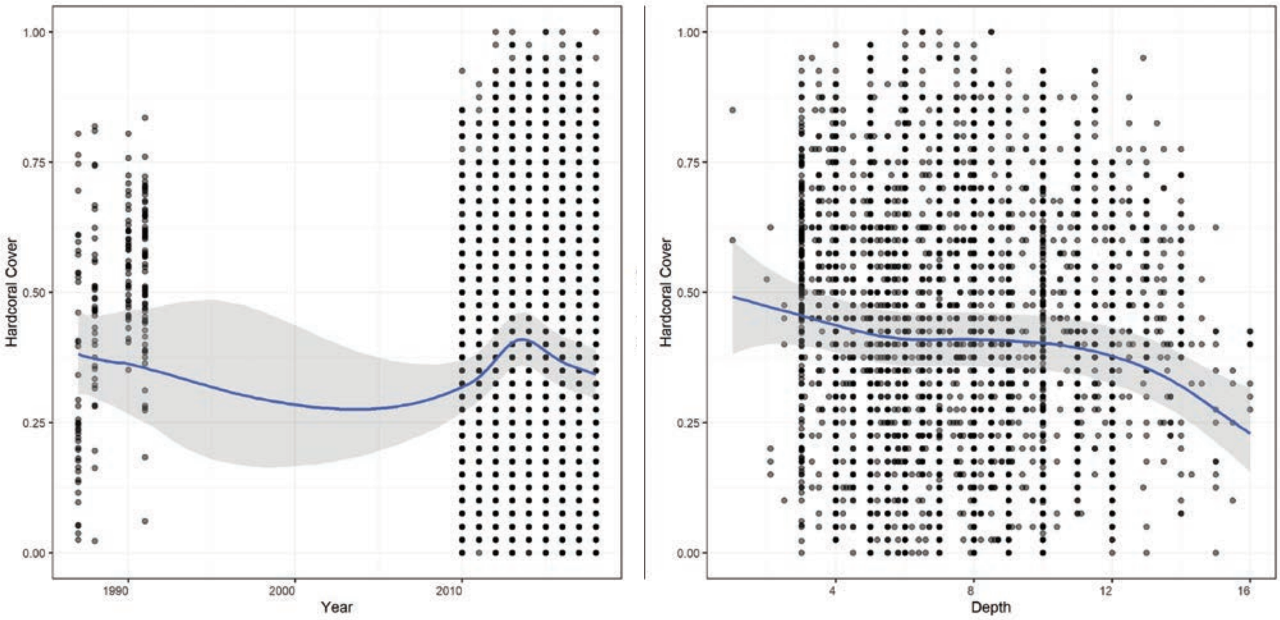

FIGURE 2: *Malaysia coral cover across (a) years and (b) depths.*

(a) (b)

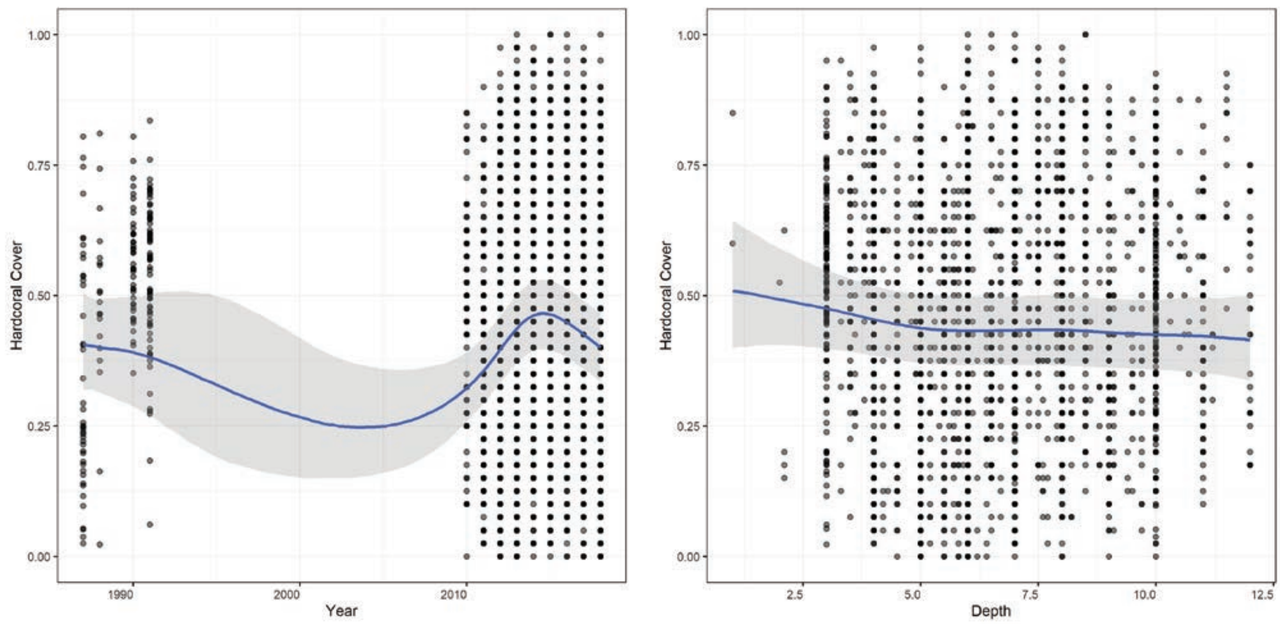

FIGURE 3: *Peninsular Malaysia coral cover across (a) years and (b) depths.*

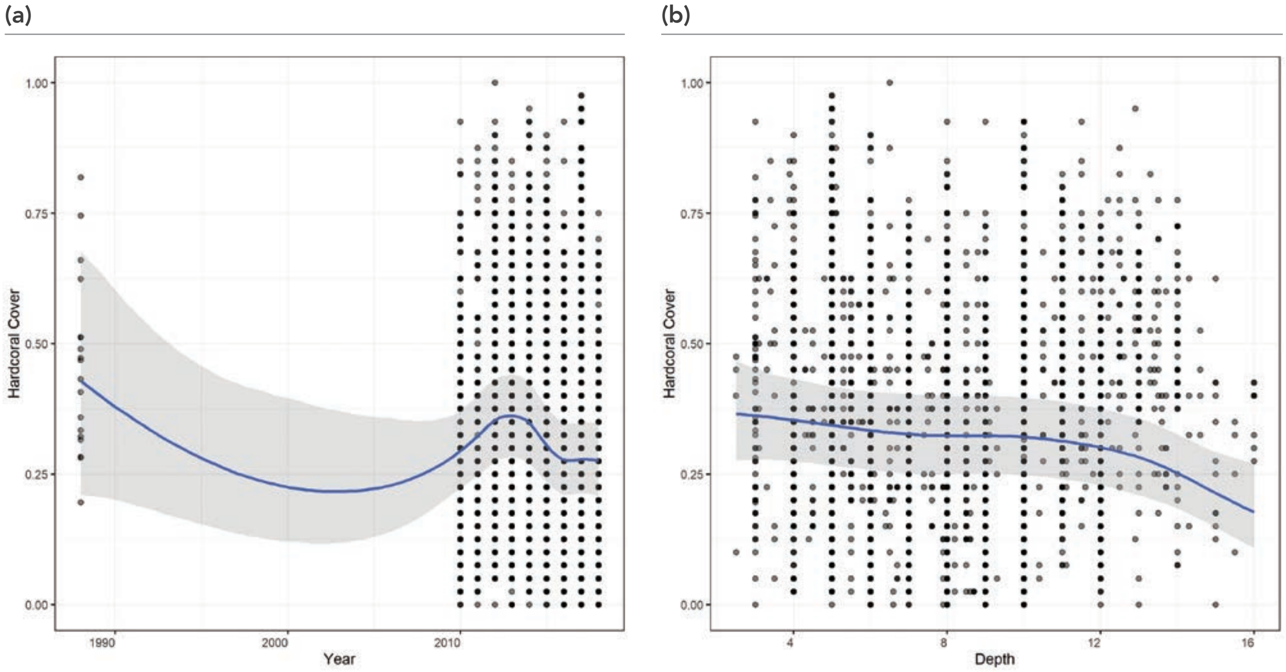

FIGURE 4: *East Malaysia coral cover across (a) years and (b) depths.*

Coral Bleaching 2016

Among the localities surveyed in the federal marine parks of Peninsular Malaysia during the 2016 coral bleaching event, bleaching was the most severe in the Strait of Malacca (Table 1). At Pulau Payar Marine Park in the SOM, 60% of its coral community was bleached, while individual colonies were up to 90% bleached. On the other hand, the highest reported bleaching was 50% for Pulau Sibu, Johor in the South China Sea, and individual colonies were up to 70% bleached. Only one marine park area was not reported to be bleached in 2016—Pulau Perhentian, located to the northeast of Peninsular Malaysia. This observation was rather surprising as Pulau Redang, which is approximately 30 km southeast of Pulau Perhentian, had reported a small degree of bleaching. The other federal marine park with no reports of bleaching was Pulau Rusukan Besar, Labuan, on the west coast of Sabah.

Moderate bleaching was observed at two localities in Sabah. In Lankayan Island, Sandakan, on the east coast of Sabah, signs of bleaching were observed in April 2016 when seawater temperatures increased by up to 2°C above average from mid-April until November with a constant average of 30.4°C, which peaked in June 2016 at 31.92°C on reefs of 5 m depth. A similar trend was observed on reefs at 10 m depth. Bleaching was observed at 34 out of 38 reef sites ranging from 2–50% coral bleaching down to 15 m. It was estimated that an average of 26% of the coral community at the deep reefs (≥10 m depth) and 6% of the coral community at the shallow reefs (≤5 m) had bleached, with branching corals most affected (Reef Guardian 2017). On Sepangar Island, Kota Kinabalu, on the west coast of Sabah, based on estimates from manta tow in early July 2016, approximately 40% of coral colonies from a 400 m stretch of the reef showed signs of bleaching at between 4–6 m depths (*personal observations*).

Milder bleaching events were observed in Miri, Sarawak (<3% coral community), Labuan (<1%), Mantanani and Usukan, Kota Belud, Sabah (<6%), and Kapalai and Pom Pom, Semporna (<1%) (Reef Check Malaysia 2017).

TABLE 1: *Coral bleaching report for islands within Marine Parks (MPA) from the Department of Marine Park Malaysia on 2 June 2016 (see Figure 5).*

AREA	SITE	BLEACHING OF THE CORAL POPULATION (MEAN, %)	BLEACHING OF CORAL COLONIES (MAX, %)
Straits of Malacca	Pulau Payar, Kedah	60	90
South China Sea	Pulau Perhentian, Terengganu	0	0
	Pulau Redang, Terengganu	10	30
	Pulau Tioman, Pahang	20	30
	Pulau Sibu, Johor	50	70
	Pulau Tinggi, Johor	30	50
	Pulau Rusukan Besar, Wilayah Persekutuan Labuan	0	0

FIGURE 5: *Coral bleaching reported by the Department of Marine Park Malaysia for islands within Marine Parks (MPA) (Coral Triangle Atlas, 2021).*

Drivers & Pressures

REEF FISHERIES

Malaysia is one of the largest consumers of seafood globally, with per capita consumption of fish at about 59 kg in 2016 (FAO 2019). Driven by this demand for seafood, Malaysia's fishery resources have been overexploited to the extent that almost all reefs are threatened by unsustainable fishing (Burke et al. 2012). Published data for Sabah reef fisheries is more extensive compared to Peninsular Malaysia and Sarawak. The demand for fishery products, including reef fisheries, has caused adverse environmental impacts due to unsustainable and destructive fishing practices (listed below).

ARTISANAL FISHERIES

Small-scale traditional fishers mainly target reef and reef-associated species and small pelagic species (Biusing 2001, Teh et al. 2005) using hook and line, gill nets, traps, spears, and speargunѕ (Teh et al. 2011). It has been estimated that there are 83,720 reef fishers in Malaysia (Teh et al. 2013), which are highly dependent on reef resources. In Sabah, small-scale fisheries have been estimated to potentially support 170,197 dependents, approximately 5% of the state population, and valued at RM 664 million in 2009 (Teh et al. 2011). The number of dependents from 1989 to 2009 has increased by 48.6%. Such dependence on the resources is putting much (yet undetermined) pressure on the reefs. Between 2010 and 2018, about 50% of the monitored reef sites in Sarawak and 16% of the sites in Sabah had abandoned fishnets or fishing lines from artisanal fisheries. In addition, reefs can also be damaged when corals are overturned to be gleaned for invertebrates such as sea cucumber, lobster, abalone, etc. Most of the targeted species are over-exploited, such as the sea cucumber in Semporna, where fishers have expressed that the landings have decreased in past years (Choo 2002). Other unsustainable and destructive fishing practices include blast fishing (detailed in the next section) and cyanide fishing (Pilcher and Cabanban 2000), which is commonly used to collect fish for the live reef fish trade (see Export of Fisheries Resource). Compressor diving (hookah) is used to remain underwater longer while accessing deeper reef areas to collect invertebrates or fish when used alongside blast and cyanide fishing.

BLAST FISHING

Blast fishing is a longstanding threat to the reefs in Sabah (Lulofs 1973, Lulofs et al. 1974, Wood 1977, 1978), however in Peninsular Malaysia, this method of destructive fishing is very rare and anecdotal. Between 2010 and 2018, almost 25% of monitored reef sites across Sabah had noticeable damage caused by dynamite blasts, with 40% of sites on the west coast and 17% of sites on the east coast impacted by this practice. As the east coast contains most of Sabah's coral reefs, 17% of sites on the east coast represent a larger proportion of reefs than 40% of sites on the west coast. The percentage of rubble is 19% and 18% on the east and west coasts, respectively. Semporna on the east coast of Sabah has the highest percentage of rubble fields (23% of the reef substrate). In some sites, the rubble percentage exceeds 70% of the substrate, such as in Pom Pom Island. On the west coast of Sabah, the reefs of Kota Kinabalu have the highest rubble percentage covering 20% of the reef substrate. Although the values presented here appear to be lower than previously estimated, such as 85% of Malaysia reefs affected by blast and poison fishing (Burke et al. 2012), it must be noted that only a fraction of the reefs is monitored annually.

COMMERCIAL FISHERIES

Between 2014 and 2017, commercial fisheries in Malaysia consisted of catches largely from trawlers (44–48.6%) followed by purse seines (20.6–25.7%) (Ahmad Faisal 2017). In Peninsular Malaysia, trawlers mainly have to operate outside of reef areas since most of the areas are within Marine Protected Areas and they are more active on the east coast of the Peninsular since the west coast (Malacca Straits) has intensive shipping activities. Discarded fishing nets or 'ghost nets' are also frequently found on reefs on the east coast of Peninsular Malaysia especially in Pulau Tioman (https://www.reefcheck.org.my/blog/cintai-tioman-april-june-2021-news). In Sabah, trawling grounds were reported to be more extensive on the west coast, stretching from Brunei Bay in the south up to Marudu Bay in the north (Biusing 2001). However, fish landings from trawling on the east coast of Sabah accounted for 40% in 2016, with the highest landings from Sandakan (Department of Fisheries Sabah 2016). Trawlers have been found to encroach into no trawling zones which are usually within Marine Protected Areas, causing conflict with artisanal fishers as well as inflicting damage to reefs.

Large, discarded trawl nets were observed by divers over reefs in Marudu Bay in 2016 and Semporna (south of the Tun Sakaran Marine Park) in 2019. In Lankayan, an abandoned gill net measuring 400 m in length was found entangled on a reef; in 2011, it was removed by volunteer divers and crew from Lankayan Island Dive Resort (Reef Guardian 2012).

EXPORT OF FISHERIES RESOURCE

In 2014, Malaysia exported fish and fishery products worth USD 0.87 billion, including live fish (USD 37.04 million) and ornamental fish (USD 19.82 million) (Ahmad Faizal 2017). The live reef fish trade (LRFT) in Sabah was established in the mid-1980s and has since been a major concern for Sabah's coral reefs (Biusing et al. 1999). Fish are caught by hook and line, fish traps, and cyanide fishing (Daw et al. 2002). The fish traps are camouflaged with corals and rocks, usually removed from nearby natural reefs, thereby causing impacts. The use of cyanide would affect the coral and its zooxanthellae and trigger coral bleaching (Cervino et al. 2003). The two main hubs for the LRFT are Kudat and Semporna (Bentley 1999). Stocks of LRFT species have reduced significantly between 1994 and 2000, and fish being caught were also believed to have decreased in size over time (Daw et al. 2002), indicating impacts to the fish population due to the trade (Scales et al. 2007). This reduction is evident by the absence or very low numbers of LRFT species such as the humphead wrasse (*Cheilinus undulatus*) and mouse grouper (*Cromileptes altivelis*) on the monitored reefs. The trade has resorted to unsustainable harvesting of juvenile fish to be kept in captivity until market size while employing trash fish as feed (Kassem and Wong 2011).

In Sabah, experimentation of crossbreeding groupers emerged in the aquaculture industry in the 2000s. The hybridization of groupers has resulted in over ten grouper species (see Ching et al. 2018). The Sabah grouper (tiger grouper, *Epinephelus fuscoguttatus* x giant grouper, *E. lanceolatus*, also referred to as hybrid grouper TGGG) has led the way since 2006 and became popular in the LRFT (USAID 2013). These hybrids, such as the Sabah grouper, are grown-out in cages at sea and could have potential environmental implications from accidental release or escape to the wild (Sadovy et al. 2017). As of 2016, the Sabah grouper has been reported to spawn naturally in captivity (Ching et al. 2018), suggesting that they could be spawning in the cages at sea (particularly matured broodstock), thus establishing a population in the wild. There have been anecdotal reports of the Sabah grouper in the coastal waters of Tuaran, on the west coast of Sabah, and two individuals have been caught in trawl nets (USAID 2013). It is not evident yet how this hybrid in the wild will affect the natural fish stock or the reefs in Sabah. In the Pulau Payar Marine Park of Peninsular Malaysia, no tiger groupers have been reported for a few years, where they were abundant previously and instead, the hybrid groupers are found in abundance especially at Pulau Segantang (Yusuf, Y. *personal communication*).

COASTAL DEVELOPMENT

Approximately 5 million people in Malaysia live on the coast within 30 km of a coral reef. For Peninsular Malaysia, in the SOM, intense coastal development is ongoing due to the rapid expansion of the big cities such as Pulau Pinang, Kuala Lumpur, Melaka, and ports of Kuantan and Kelang. Coastal development on the east coast is slower except for the port of Kuantan and the southeast coastline of Johor.

One-third of the Malaysian coastal population are in East Malaysia, where coastal development threatens 35% and 45% of reefs in Sabah and Sarawak, respectively (Burke et al. 2012). Here, Kuching and Kota Kinabalu, the largest two cities and the state capital of Sarawak and Sabah respectively, are near the coast. In fact, most of Kota Kinabalu city center is built on reclaimed land. Many other large towns have also been established and are expanding along the coast, causing impacts from urbanization such as land clearing, land reclamation, sedimentation, sewage discharge, and domestic litter (Heery et al. 2018). Urbanization often entails the construction of seawalls. In Sabah, land and seafront conversion for agriculture and aquaculture are ongoing in Lahad Datu and Kudat. Reefs near centers of development have seen reduced water visibility. Towns such as Semporna that have thrived on tourism and the diving industry have seen structures built over or close to reefs for accessibility to tourism infrastructure.

TOURISM

Tourism is one of Malaysia's fastest-growing sectors, contributing 5.9% to the total GDP in 2018, in third place after manufacturing and commodities (Hirschmann 2020).

Much of the tourism in Peninsular Malaysia, which involves coral reefs, are on the islands within marine parks due to their beautiful reefs. As a result, two nautical miles of the surrounding waters around each island are protected from human exploitation. However, the land on these islands is not within the marine parks' jurisdiction, and there are instances where unsustainable development for tourism has taken place.

In Sarawak, the tourism industry contributed RM 11 billion to the state's economy, 7.5% of the total GDP (Department of Statistics Malaysia 2019), while in Sabah, the tourism industry grew by almost 120% between 2004 and 2018 (Sabah Tourism Board 2019), the second-largest income contributor to the state after agriculture. Sarawak and Sabah are marketed as ecotourism destinations in taking advantage of the states' wealth in biodiversity and natural resources. Sarawak is a newcomer to the marine tourism business, but Sabah has been offering marine-based tourism since the 1980s. Some of the pressures that stem from this industry include development for tourism facilities, which have similar impacts as detailed under coastal development, but with varying degrees of stress depending on the scale of tourism. In Kota Kinabalu and Semporna, boat traffic from tourism has increased tremendously over the past ten years (*personal observation*). Boat/anchor damage has been noted at 15% of the monitored sites in Semporna. Although reports suggest that damage is caused by tourist boats, artisanal fishers may also be involved. Impacts may also be caused by direct contact with the reef during snorkeling, scuba diving, and sea-walking. Tourism is seen as a means of alternative livelihood for the local communities to relieve the pressure on reefs. However, this mechanism is counterintuitive as Sabah prides itself to be a seafood haven, and perhaps one of the reasons that appeal to tourists, but this adds pressure to the existing high demand for seafood.

TRASH

While it was not unusual to see errant trash underwater in the 1980s and 1990s, it has become a common feature on beaches adjacent to coral reefs in Malaysia, particularly near urban areas such as Port Dickson, Peninsular Malaysia and Tanjung Aru, Sabah (Fauziah et al. 2015). On most of the islands within Peninsular Malaysia's marine parks, trash and recycling management have remained a challenge and are similar to some inhabited islands across the country.

Many beach clean-ups have been conducted along the coasts of Sarawak and Sabah, removing tons of marine debris and trash, of which plastic has been quantified as a major component (Mobilik et al. 2014, Adnan et al. 2015). Underwater clean-up efforts have also reduced the impact of trash, such as plastics from smothering corals or consumed by marine mammals, turtles, and fish (in the form of microplastics). Several marine stranding cases have been linked to this reason (Daily Express 2015, Borneo Today 2018) and fish sold in the market with microplastics in their gut (Karbalaei et al. 2019).

COTS OUTBREAKS

Crown-of-thorns starfish (COTS) have been observed on Sabah's reefs since the late 1960s (Yonge 1968, Lulofs 1974, Morris 1977, Wood 1978), but outbreaks were first reported in the 1990s on the reefs of the Tunku Abdul Rahman Park, Kota Kinabalu. These outbreaks resulted in significant losses of branching corals such as *Acropora* (Pilcher and Cabanban 2000). Extensive outbreaks of COTS were recorded from Lankayan Island in 2005–2007, 2013, and 2016–2018 (Reef Guardian 2007, 2008, 2014, 2017, 2018, 2019). More than 10,000 COTS were removed from the reefs of Lankayan each year in 2005, 2016, and 2017. It appeared that by 2017, the COTS outbreak had moved southwards from Lankayan towards Semporna, whereby thousands of COTS were removed in 2018 by local dive operators (*personal communication*) and Sabah Parks (Taman-Taman Sabah 2019). Reduced live coral cover was observed on the east coast of Sabah in 2013 and 2016, which could have been caused by the COTS outbreak and coral bleaching in 2016. In other areas of Sabah, outbreaks occurred in the southeast of Banggi Island, Kudat in 2009 (Waheed et al. 2019) and the Tunku Abdul Rahman Park, Kota Kinabalu in 2015 (Mohd Firdaus, N. et al. *unpublished manuscript*). In Miri, Sarawak, COTS were noted on two reef sites in 2017.

CORAL BLEACHING

For Peninsular Malaysia, mass coral bleaching was first reported at Pulau Redang in 1998. In 2010, coral bleaching was reported at Pulau Payar in the northern part of the SOM; on the east coast, bleaching was reported for Pulau Redang, Pulau Perhentian, Pulau Tioman, Pulau Tinggi and Pulau Sibu (Guest et al. 2012, Kushairi 1999, Tan and Heron 2011, Tun et al. 2010). In 2016, bleaching was reported for Pulau Payar, Pulau Redang, Pulau Tioman, Pulau Sibu, and Pulau Tinggi (Department of Marine Parks 2016).

In Miri, Sarawak, approximately 20% of coral colonies bleached at the monitored reef sites in 2010 (Reef Check Malaysia 2011).

In Sabah, the widespread coral bleaching event of 1997–1998 had affected the reefs of Kota Kinabalu. About 30–40% of the coral cover at Gaya Island was bleached when the water temperature peaked at 32°C, while minor bleaching occurred at Mamutik Island (Wilkinson 1998, Pilcher and Cabanban 2000). Since then, other reported bleaching events due to increased seawater temperatures (1–2°C) were as follows:

2010 Bleaching on the reefs of Sabah rated as moderate (25–50% of the coral cover) and extended down to 20–25 m depth (Tun et al. 2010). Reefs of Kota Kinabalu had high bleaching rates (46-90% of the coral cover) (Aw and Muhammad Ali 2012).

2012 Minor bleaching was observed in Mabul (~3%), and Mataking (8%), Semporna.

2014 Bleaching in 14% of the coral community in shallow and deep reefs in Lankayan Islands, Sandakan (Reef Guardian 2015).

2016 Bleaching at 34 reef sites ranging from 2-50% coral cover in Lankayan Island, Sandakan (Reef Guardian 2017), and bleaching in 40% of the coral colonies at Sepangar Island, Kota Kinabalu.

2017 Bleaching at 33 reef sites affected 16% and 6% of the coral population in the deep and shallow reefs, respectively in Lankayan Islands, Sandakan (Reef Guardian 2018).

STORM & HEAVY RAINFALL

Damage to reefs caused by tropical storms is infrequent in Malaysia. However, in 1996, tropical storm Greg caused damage to most shallow coral reefs within the marine park in Kota Kinabalu, Sabah (Pilcher and Cabanban 2000). Severe damage was observed in four reef sites, but coral recruitment was noticeable by 1999, showing early signs of recovery. Some reefs did not recover, which prompted the park authorities to deploy Reef Balls at these damaged sites in 2005. In 2011, coral damage caused by a storm was noted in the shallow reefs of Talang Besar East, Kuching, Sarawak. In 2018, coral damage was reported in two shallow reefs of Boheydulang Island, Semporna, three reefs in Mantanani Island, Kota Belud, and one site in Miri, Sarawak. In January 2019, tropical storm Pabuk struck and there were reports of reef damage and coastal erosion on the east coast of Peninsular Malaysia from Pulau Perhentian in the northeast to Pulau Rawa in the southeast.

In 2007, continuous rainfall on the mainland along the east coast of Sabah caused freshwater runoff that reached Lankayan Island. The mean seawater salinity around the island was 25.4 ppt, and the waters were greenish brown, possibly containing a high sediment load. After 12 days, dead corals, sea cucumbers, and giant clams were observed on the shallow reef flats (Reef Guardian 2008).

SEDIMENTATION & RIVER RUNOFF

The reefs in the Strait of Malacca on the west coast of Peninsular Malaysia is constantly under turbid water conditions especially areas near river mouths such as Kepulauan Sembilan, Perak where it is impacted by sedimentation and runoff from the adjacent Perak River during the southwest monsoon seasons.

Reefs are sparse along the coast and shallow sea shelf of Sarawak. There is a lack of rocky substrate for a long way out to sea as many rivers discharge terrigenous sediment runoff from the mainland. Sand mining and dredging had increased sediment loads from upstream areas in the past (Pilcher and Cabanban 2000, Shabdin 2014). Due to these reasons, the reefs are not well developed (Burke et al. 2012).

In Sabah, an increase in sediment and nutrient loading from surrounding watersheds into the reefs of Kota Kinabalu have caused an increase in turbidity, which has affected the coral condition (Pilcher and Cabanban 2000). Sedimentation and reduced water clarity have also been observed in the lagoon of the Tun Sakaran Marine Park surrounded by Boheydulang and Bodgaya islands in Semporna and several reefs in Lahad Datu and Mantanani. Hard coral cover ranged between 6% and 61% on these turbid reefs, suggesting that some coral colonies can adapt and survive in less-than-ideal environmental conditions.

RECOMMENDATION

Recommendations to improve coral reef sustainability, conservation, and management:

1. Promote community-based management of coral reefs in partnership with local institutions (government or non-governmental agencies) or stakeholders. The local communities must understand how management can help sustain their livelihood - and they should also have a say in the decision-making process, for example, providing input in setting boundaries of conservation zones, fishing grounds, etc. There are very few community-managed areas especially in Sabah.
2. Explore alternative livelihoods and diversify income sources of local fishing communities. Tourism and mariculture have been promoted as alternative livelihood measures, but these sectors do not necessarily relieve pressure from reefs in Sabah. Several programs have been introduced, such as cultivating seaweed, but success is ambiguous due to competition with neighboring countries and limited interest of the local community. Income that does not rely on marine resources would be ideal, such as having the people promote arts, crafts, and culture of their communities. One example is the Persatuan Wanita Pulau Omadal (Women Society of Omadal Island) in Semporna, whereby the womenfolk have shifted from collecting turtle eggs to creating handicrafts to be sold.
3. Investigate options for reef restoration. Many reefs that have not recovered from blast fishing especially in Sabah, should be assessed for potential reseeding. For the past 20 years, artificial reefs appear to be the go-to method in attempting to rehabilitate damaged reefs, so much so, in certain situations, it has caused more harm than good. In the past five years, coral planting programs have gained popularity. They are currently the favored method for reef restoration. At times, they are carried out together with the deployment of artificial reefs, but this comes with a different set of problems, such as poor practice in acquiring corals for restoration and the introduction of unsuitable artificial substrates.
4. Build awareness of local pressures and conservation issues through environmental education and capacity building programs for coastal populations, especially those whose livelihoods are intimately associated with reefs. It is important to target all age groups, especially the youth, to empower the community to become active stewards of their reefs and environment. Commendable examples of community groups are Cintai Tioman, Cintai Mantanani, Banggi Corals Conservation Society (formerly known as the Banggi Youth Club), Kudat, and Green Semporna.
5. Increase public awareness on the impact of the LRFT industry. Based on a questionnaire carried out at seafood restaurants in Kota Kinabalu, consumers agreed that if they knew that some fish species were threatened or decreasing in population, they would stop or reduce eating them. Almost half of the consumers were not aware that cyanide was used in LRFT capture, and they were not aware of the harmful effects of cyanide on human health or the reef environment (Komilus et al. 2012). Several recommendations have been provided by Kassem and Wong (2011), Komilus et al. (2012), and Poh and Fanning (2012) on the LRFT and should be considered seriously.
6. Impose strict fines for blast fishing, illegal trawling, and commercial fishers that break the law. There are state and federal legislations that address these issues, but enforcement by the authorities needs to be intensified.

7. Uphold laws and policies in regulating coastal development. There must be a political will to ensure that development is done with minimal impact on the environment. In some areas, stopping development when there are threatened, or endemic marine species or habitats present.
8. Impose a standard for tourism best practices. Non-complying tourist operators should be penalized or even stand to lose their license to operate, thereby giving new operators opportunities to enter the tourism market. There are existing good practices in the dive (and snorkeling) industry, such as Green Fins. These or similar programs for good tourism practices should be used as a standard for all dive operators and resort owners in tourism management. At present, several dive operators in Malaysia have Green Fins certification, and a few operators or resorts in Peninsular Malaysia and Sabah have set up eco-friendly establishments, with commendable practices such as serving sustainably produced foods and promoting responsible diving practices. When appropriately managed, tourism can be a powerful tool for reef conservation. Radical changes need to be imposed to ensure the well-being of coral reefs, which could make or break the economy of an area such as Semporna, Sabah. For example, in 2004, the Sabah government ordered all resorts on Sipadan Island, Semporna, to vacate from the island to reduce pressure on the reefs.
9. Advocate banning single-use plastics, and work towards mitigating the use of plastics in everyday life. Most of the trash that is observed on the reefs is of plastic components. In 2018, the government initiated a plan towards reducing plastic use and waste outlined in Malaysia's Roadmap Towards Zero Single-use Plastics 2018–2030. However, this plan will only work if Malaysians are willing to embrace lifestyle changes.
10. Local authorities need to engage with the local islanders to establish a workable trash management system.
11. Increase capacity building among academia and agencies to conduct continuous and impactful research on coral reefs.
12. Coordinate reef monitoring and establish database/repository of information at the national and state levels. Information should be well managed to accommodate changes in leadership and management systems.
13. The federal coral bleaching taskforce and its bleaching management plan must be updated and strengthened. Coordination and help by respective local and state authorities, local people, and businesses must also be strengthened.
14. Increase the capacity of locals and businesses near coral reefs and help them lower the negative impacts on reefs. In the long run, this will make their reefs more resilient and, therefore, always provide services and income to them.
15. The Coral Triangle Initiative (CTI) areas within Malaysian waters must be managed and protected with regular monitoring and enforcement.
16. Introduce marine sciences and conservation topics early in the school education syllabus nation-wide. It is important to stress that no matter how far removed a community is from coastal areas and coral reefs, its people would be affected by or are unknowingly impacting the reefs and marine environment by the choices they make in their everyday lives, e.g., seafood consumed, marine ornaments bought, etc. Ultimately, population growth and urbanization are inevitable, so we need to shift our consumption patterns to a sustainable manner to achieve the well-being of the people, biodiversity, and environment.

DATA CONTRIBUTORS

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SOUTHEAST ASIA

Myanmar

Introduction

Myanmar is the largest country in mainland Southeast Asia, with a coastline length of 2,280 km, and is geographically located between 9° 32′ and 28° 31′ N latitude and 92° 10′ and 101° 11′ E longitude, bordered by the Bay of Bengal in the west and the Andaman Sea in the south (Figure 1) (Psomadakis et al. 2020, Lunn 2014). Several riverine systems influence Myanmar's coastal zones, including May-yu and Kladan rivers in the Rakhine Coastal Region; Ayeyarwady, Sittaung, and Thanlwin rivers in the Deltaic coastal region; and Ye, Dawei, Tanintharyi, and Lenyar rivers in the Tanintharyi coastal region. These riverine discharges influence the continental shelf formation along Myanmar's coast. The continental shelf is wide along the Deltaic and Tanintharyi coasts situated in the Andaman Sea and narrows along the Rakhine coast located in the Bay of Bengal (Psomadakis et al. 2020). Because of these riverine discharges, Myanmar's inshore coastal areas are silty and turbid, especially near the Deltaic Coast.

The Myeik Archipelago is located in the southernmost part of Myanmar, in the north-eastern Andaman Sea. The archipelago, which stretches along the coast for over 600 km and is up to 125 km from the mainland, comprises approximately 800 limestone and granite islands (Figure 1; Obura et al. 2014). The Myeik Archipelago is situated on the shallow continental shelf and is strongly influenced by riverine discharge, predominantly from the Tanintharyi and Lenyar rivers. The Rakhine coast, facing the Bay of Bengal, has a narrow continental shelf and is influenced by several riverine discharges. From the north, the major influence comes from the Brahmaputra River, as well as the Kaladan and May-yu rivers, amongst several other tributaries along this area of the coast. The Coco Islands are situated in the outermost area of the Deltaic Region in Myanmar, bordering the Indian Andaman Islands chain, with the Bay of Bengal to the west and the Andaman Sea to the east.

In Myanmar, corals are found on fringing coral reefs (Psomadakis et al. 2020, Howard 2018, Obura et al. 2014, Murray-Jones et al. 2017), distributed along the Rakhine and Tanintharyi coastal regions (e.g., Myeik Archipelago) and around the Coco Islands (Figure 1). Corals can be found in coastal and inner island habitats, even in very turbid waters. Still, they mostly inhabit the outer islands where water quality and clarity levels are higher due to lower impact by riverine discharge, higher wave energy, and stronger currents.

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FIGURE 1: *Coral distribution map in Myanmar.*

The Myanmar Marine Conservation Program of Fauna & Flora International (FFI) has signed several memoranda of understanding (MOUs) with the Forest Department, Department of Fisheries, Myeik University, and Pathein University, and is implementing marine and coastal conservation projects in the Myeik Archipelago in the Tanintharyi Coastal Region, as well as in the Mawdin coast in the Rakhine Coastal Region in Myanmar. FFI-Myanmar has conducted regular coral reef surveys since 2013 in the Myeik Archipelago, Tanintharyi Coastal Region, and since 2014 in the Mawdin coast, Rakhine Coastal Region, in collaboration with the local universities and government departments, and also supported by international scientists. Coral survey sites from 2014 to 2019 are shown in Figure 2.

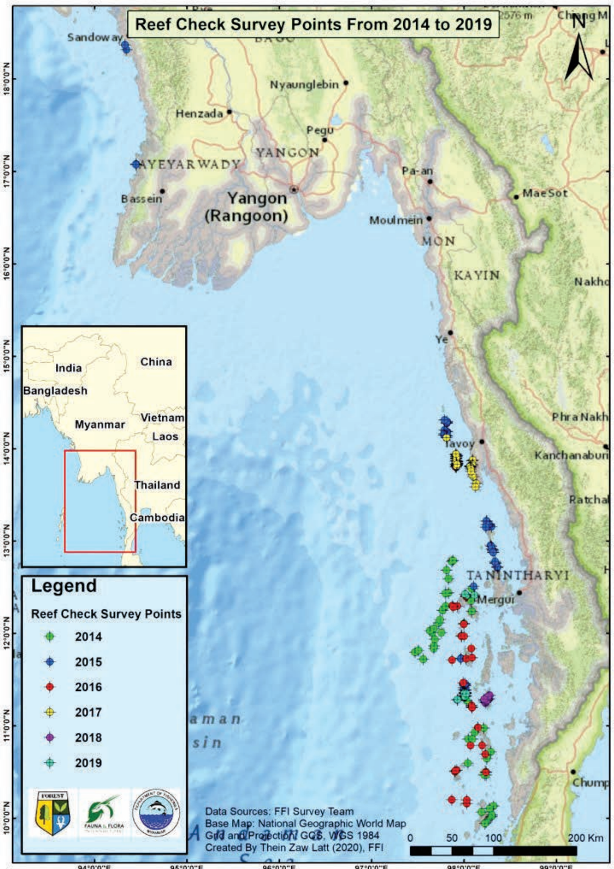

FIGURE 2: *Coral survey sites from 2014 to 2019.*

A modified Reef Check method is used to monitor Myanmar's coral reefs (Hodgson et al. 2006), whereby five 20-meter replicate transects with 5-meter intervals are surveyed at each site. During these surveys, the abundance of a set of readily identifiable indicator species is recorded to monitor change over time and understand the health of the coral ecosystem. To date, FFI-Myanmar has conducted approximately 400 survey dives at 35 sites around the Myeik Archipelago islands in the Tanintharyi Region and conducted surveys at 14 sites along the Mawdin Coast in the Rakhine Region (Marry-Jone et al. 2017). In 2019, Wildlife Conservation Society (WCS) Myanmar also initiated coral monitoring surveys in the Rakhine Coastal Region (Mizrahi, M. *unpublished manuscript*).

In 2019, FFI-Myanmar started to install and initiate surveys using fixed transects in selected coral sites within and adjacent to Locally Managed Marine Areas as a part of its long-term coral monitoring program in the Myeik Archipelago. Although FFI has so far planned to install permanent transects at three monitoring locations in the Myeik Archipelago, the installation of permanent transects could only be completed at one site in 2019 due to the COVID-19 pandemic (Figure 3). Permanent transects will, however, be installed at the two other locations in the upcoming dry season. Drone surveys were also conducted in Myanmar to compare with in-situ coral survey data (Figure 4).

Coral species diversity is the highest on inner reefs due to the dominance and diversity of the genus *Acropora.* However, coral communities are dominated by *Porites*, particularly on outer fringing reefs in the Myeik Archipelago. Coral communities in the Myeik Archipelago can be categorized into three main types: a) fringing reefs on relatively exposed boulder slopes of outer islands, from the surface to about 15 m depth where the boulders transition into sandy slopes; b) fringing reefs on relatively sheltered slopes of the inner islands with high turbidity and strong currents; and c) steeply sloping/vertical rock walls on small isolated rocks or outer island cliff faces, extending into deeper waters over 20 m (Howard 2018, Obura et al. 2014).

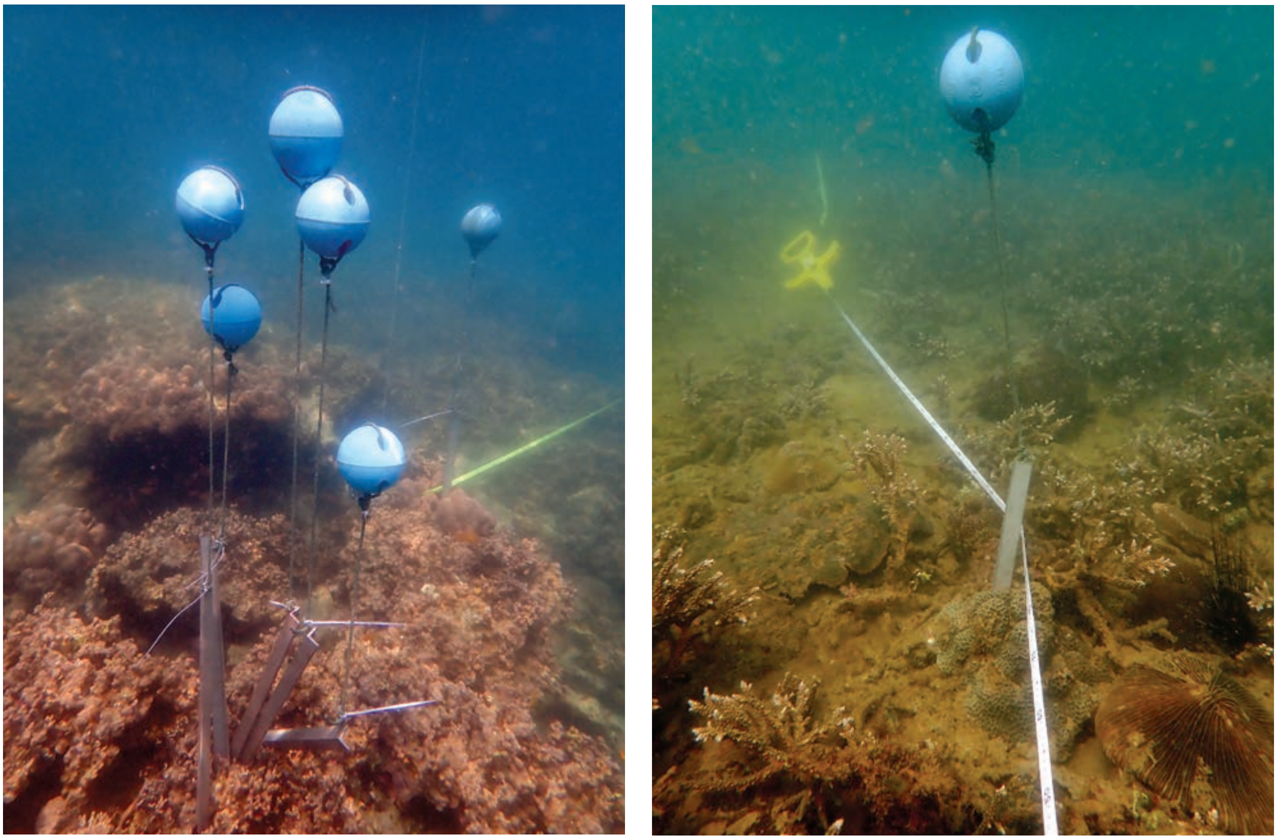

FIGURE 3: *Fixed transects for long-term coral monitoring (Photos: FFI-Myanmar).*

FIGURE 4: *Drone surveys (Photos: FFI-Myanmar).*

Although there are impacts from riverine discharge to the inner reefs in the Myeik Archipelago, coral diversity is higher in the inner reefs compared with outer reefs due to possibly better existing inshore conditions such as protection from sea warming and natural disaster impact from the open sea. The dominant hard coral species in the Myeik Archipelago are *Acropora* and *Porites* species (Howard 2018, Obura et al. 2014). Obura et al. (2014) reported that hard corals occurred at an overall cover of 33% and was highest on inner reefs, then rock walls, then outer reefs in the Myeik Archipelago. A total of 287 hard coral species (68 genera, 17 families) were observed, including two coral species listed as Endangered (*Acropora roseni* and *A. rudis*), and 36 as Vulnerable (Obura et al. 2014). According to updated surveys by FFI-Myanmar, hard coral cover varies significantly across the Myeik Archipelago from 0% to 92%, with an average of 48.9% (Howard 2018). Obura et al. (2014) pointed out that the coral community composition was similar to the Coral Triangle/Indonesian Region. However, some coral species characteristic of the west and the north Indian Ocean were also present (including *Acropora roseni, A. rudis, Plesiastrea devantieri,* and *Anomastrea irregularis*), emphasizing the Andaman Sea as a transition zone between the western and eastern parts of the Indo-Pacific.

Murray-Jones et al. (2017) reported that most corals in the Rakhine coast are found on fringing reefs, are less developed, and often consist of small coral patches growing directly on the rocky substrate. The reefs of the Mawdin Coast, situated in the southern part of the Rakhine Coastal Region of Myanmar, are not true coral reefs but rather rocky reefs with some corals (Murray-Jones et al. 2017). Some fringing corals can be found around the islands in the northern part, such as Munaung (Cheduba Island) and Ramree Islands. Coral cover in the Rakhine coast was found to vary from 9% to 69% with relatively low diversity, mainly consisting of *Acropora* and *Porites* species (Murray-Jones et al. 2017) and about 34% mean coral coverage in the Gwa Kyun area (Meiraza 2020). Murray-Jones et al. (2017) pointed out that the result is likely due to a combination of factors, such as high turbidity, which confines corals to shallow water and limits growth. In the Rakhine Coastal Region, the pressures of seasonal storms and strong waves during the rainy season are among the causes of the low coral community profile (Murray-Jones et al. 2017). Corals inhabiting offshore locations are generally in better condition than the other sites because its water quality is better than nearshore locations (Murray-Jones et al. 2017).

The corals on the Andaman Islands shelf where the Coco Islands are situated, have fringing reefs in the east and barrier reefs in the west. Corals here cover an area of 11,000 km^2 while the Nicobars have 2,700 km^2 of coral reefs. So far, 39 genera with 179 species have been recorded in the Andaman Region, including the Myeik Archipelago, 117 species in the Gulf of Myanmar and Palk Bay, India, and 134 species (65 genera) in Sri Lanka (Vineeta Hoon 1997). The data show that corals in the Andaman Islands group are more diverse than the Myeik Archipelago.

Status & Trends

According to data from the Myeik Archipelago (Figure 5 & Figure 6) collected from 2014 to 2019, coral cover has increased slightly in this region during this period despite facing several negative impacts. The distinct feature of the Tanintharyi Coastal Region is the high hard coral coverage in waters less than 10 meters in depth (Figure 5a); within this depth range, coral cover is greater in slightly deeper waters (Figure 5b & Figure 5c). The coral cover also appears to have increased compared to 2014 surveys (Figure 5d). From data collected in 2014, the linear trend of the mortality rate was more than 10% and was greater at shallow depths than in deeper waters (Figure 6a). In 2019, the mortality rate dropped below 10% (Figure 6c). Mortality was again more significant at shallower depths. Some coral recruitment was recorded during every reef survey conducted, showing the resilience of Myanmar's coral communities.

The outer islands may be facing several impacts, including ocean warming. However, the effects of ocean warming on coral ecosystems in Myanmar have yet to be surveyed appropriately, which is a big challenge for comparing the consequence of climate change impacts with that of other reefs in the region.

The overall reef resilience of the Myeik Archipelago was scored at average to below-average levels (Obura et al. 2014). Survey results have shown that reef health in the Myeik Archipelago appeared compromised (Howard 2018). While some sites have relatively healthy coral communities, others showed distinct evidence of past mortality events, such as the presence of dead coral skeletons, eroding areas of reef and rubble frameworks, and a high cover of algal turf (Obura et al. 2014). Outer reefs in the Myeik Archipelago have greater evidence of past mortality impacts compared to the inner reefs, which are generally covered with fast-growing *Acropora* species. However, even in areas that show past damage and impact, corals have been found to recover relatively quickly, and there appears to be an adequate supply of recruits throughout all sites in the Myeik Archipelago.

As reported, the benthos in the Myeik Archipelago was dominated by hard corals, with an average cover of approximately 33% (Obura et al. 2014), followed by turf algae and other benthic invertebrates (excluding soft corals) at 17% and 16.5%, respectively. During the 2013 surveys, inner reefs had the highest average hard coral cover at 55%, with the coral cover between 22% and 29% on fringing and rocky reef sites (Obura et al. 2014). Howard (2018) pointed out that the cover of soft corals, other invertebrates, and coralline algae was highest on rocky reef sites (52%) compared to 15–20% on fringing and inner reefs. This coverage may reflect the impact of strong waves or current conditions on the rocky reef sites. Degradation and mortality due to previous impacts from trawling and blast fishing on some coral reef sites covered with algal turf indicate that resilience may be lower in some locations. These areas were dominated by large populations of sea urchins, dead corals with algae, and very low fish abundance.

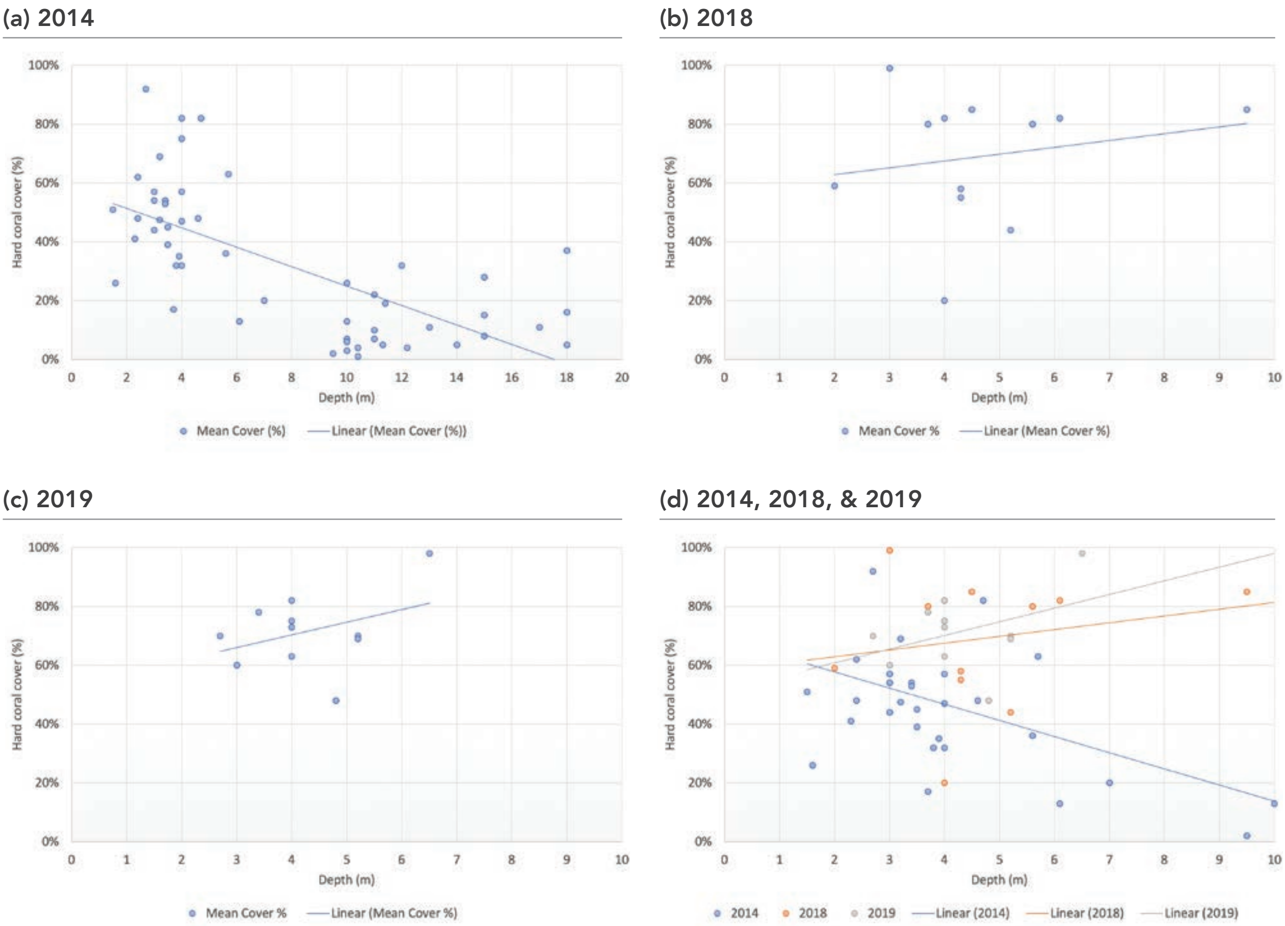

FIGURE 5: *Coral cover percentage; (a), (b) and (c) for yearly coverage in percentage; (d) for coral cover percentage comparison between years.*

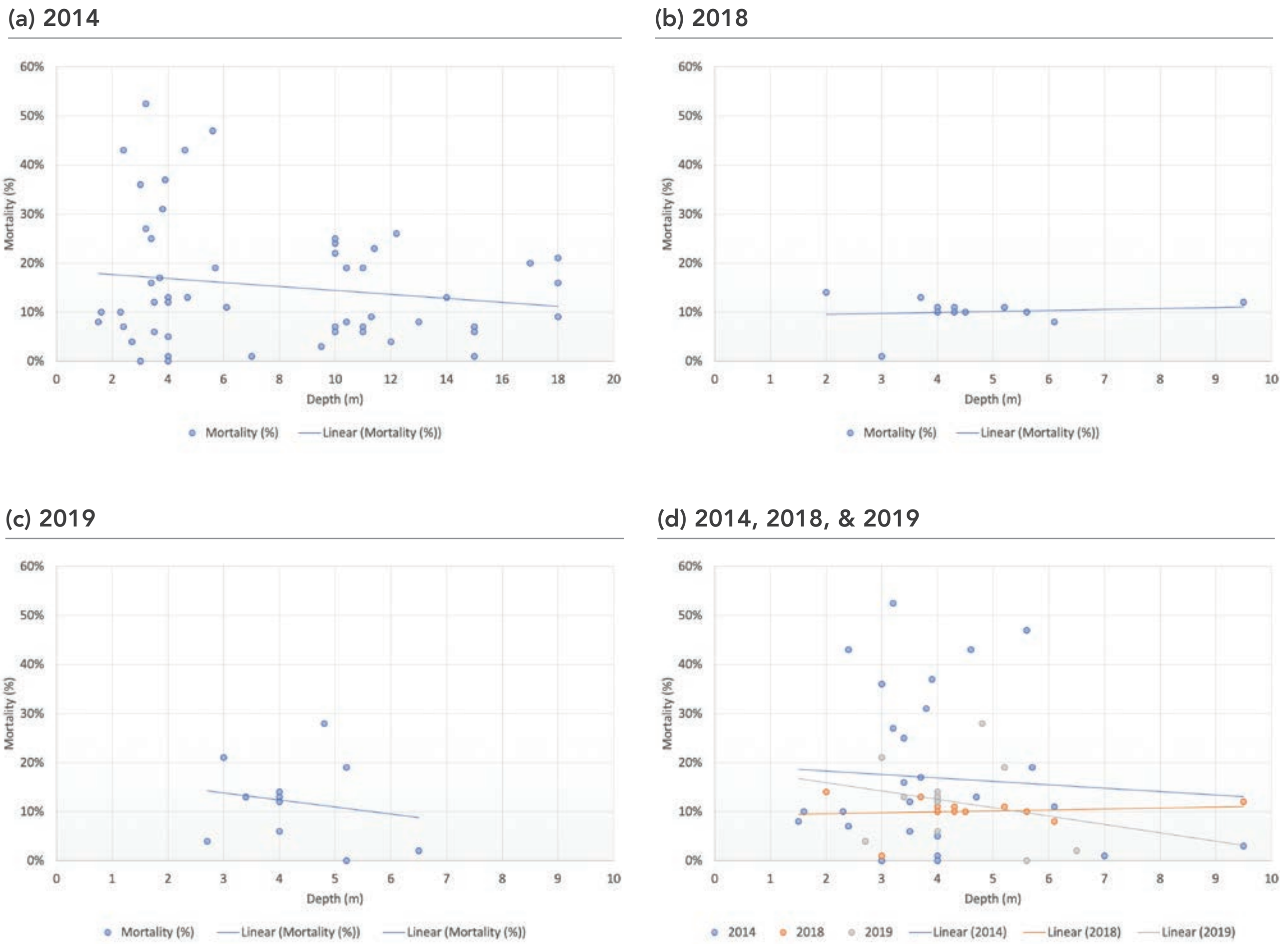

FIGURE 6: *Coral mortality percentage; (a), (b) and (c) for yearly mortality in percentage; (d) for coral mortality percentage comparison between years.*

Coral Bleaching

Coral Bleaching caused by increased water temperatures has not yet been appropriately surveyed in Myanmar. However, some instances of bleaching were noticed during the coral surveys. The photos of suspected bleaching corals are shown in Figure 7.

Howard (2018) commented that corals in the Myeik Archipelago might not have experienced bleaching as severely as the other countries. Corals may have experienced bleaching, but most of the inner reefs have recovered in the surveyed sites with fast-growing *Acropora* species. Some coral sites dominated by staghorn corals showed signs of having experienced thermal stress in 2005 and 2010, due to a combination of El Niño and a negative Indian Ocean Dipole (IOD) but have since recovered with fast-growing corals (Obura et al. 2014). However, because sea surface temperatures (SST) increased at similar rates throughout the archipelago during these events, differences in past mortality between inner and outer reefs may not be attributed to differential thermal stress. Turbidity, as indicated by chlorophyll, was markedly higher at some inshore reefs and may have provided some protection to these reefs during thermal stress events (Obura et al. 2014).

Bleaching has also been observed during surveys of the Mawdin coast but may be related to seasonal storms and wave surge exerting intense pressures on the corals (Murray-Jones et al. 2017).

The corals in the Myeik Archipelago are situated in a relatively shallow area of the Andaman Sea, and sea surface temperatures in this area are usually higher than the open ocean (Figure 8). Corals in this area are also influenced by riverine discharge from the mainland coast almost continuously. Therefore, these reefs are generally subjected to a relatively higher stress load than what might be considered normal, and hence may be more robust than coral reefs in other countries. On the other hand, the coral reefs in the Rakhine Coastal Region and outer Cocos Islands may be more susceptible to increases in sea surface temperatures and large-scale bleaching events.

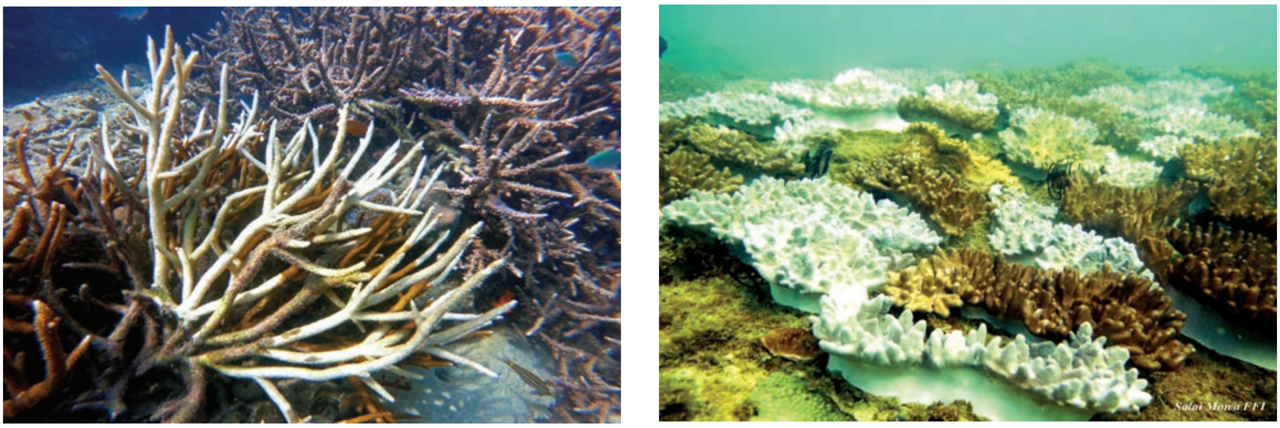

FIGURE 7: *Coral bleaching (Photos: FFI-Myanmar).*

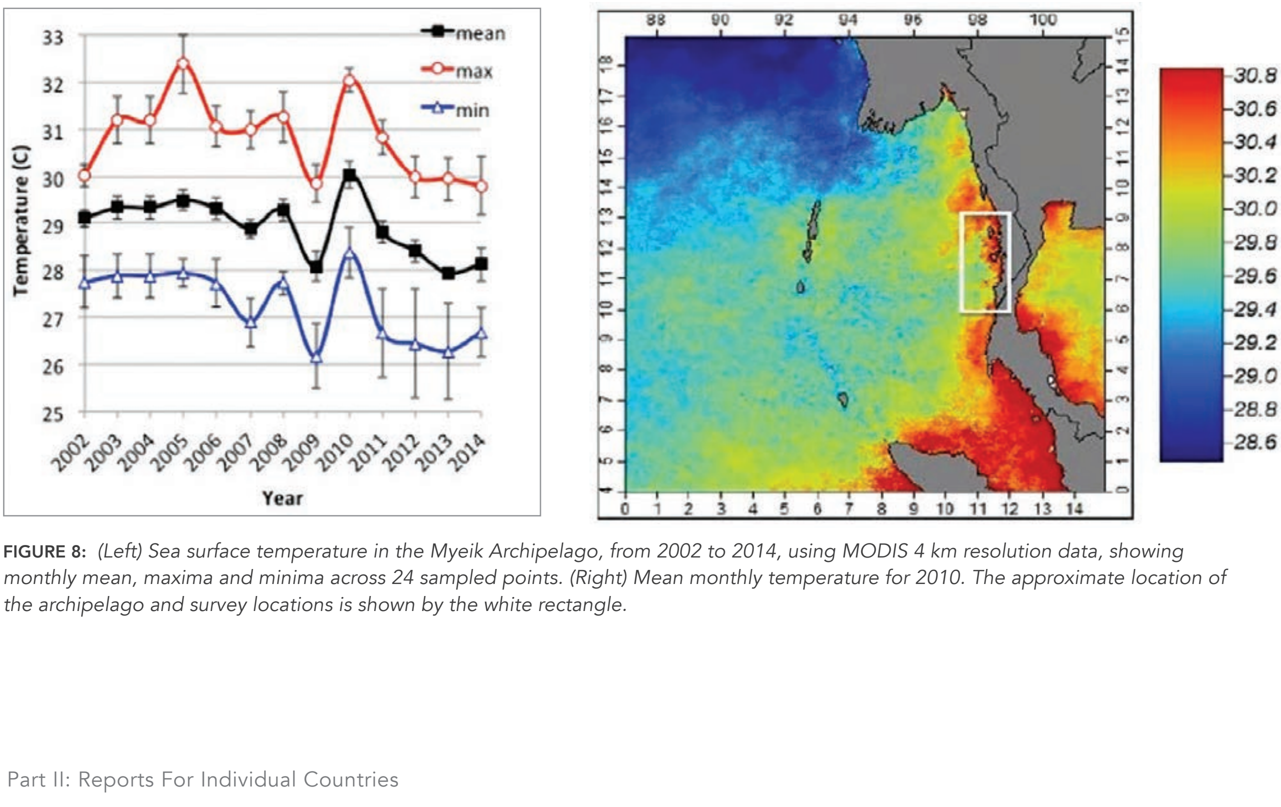

FIGURE 8: *(Left) Sea surface temperature in the Myeik Archipelago, from 2002 to 2014, using MODIS 4 km resolution data, showing monthly mean, maxima and minima across 24 sampled points. (Right) Mean monthly temperature for 2010. The approximate location of the archipelago and survey locations is shown by the white rectangle.*

Drivers & Pressures

The main driver of coral ecosystem degradation in Myanmar is the over-dependence on marine fisheries resources predominantly due to commercial and local fisheries in recent decades. Krakstad et al. (2014) reported from the 2013 Dr. Fridtjof Nansen survey results in Myanmar that there seems to be evidence that fish stocks may be overharvested. However, it is noted that it would be important to carry out another survey during a contrasting season, considering that productivity and fish abundance may be subject to seasonal cycles and or migrations.

Increasing pressure is being placed on the marine resources of Myanmar due to increased investment in the fishing industry by local communities and commercial fisheries, a weak understanding of coral ecosystem ecology by the general public, forest clearing in the riparian areas of the river systems and coastal regions increasing sedimentation into the marine environment, increased trawling, low awareness and disregard for the standing rules and regulations of the Department of Fisheries by the fishing communities, and constraint in the law enforcement capacity of the relevant government departments.

Fishing pressure has increased in recent decades due in part to the limited livelihood options available, leading many rural communities to invest in the inshore fishing business. Hookah fishers on coral reefs are also a problem in the Rakhine Coastal Region (Mizrahi, M. *unpublished manuscript*). Commercial offshore fishing vessels operate in inshore fishing grounds, especially near the coral reefs, to reduce operating costs. Commercial trawling may also be increasing, as evidenced by the numbers of ghost nets found fouled on the reefs in the region (Figure 9).

One of the critical pressures on nearshore coral reefs, including those exposed during low tides, is a lack of education and awareness among the public regarding coral reef ecosystem function and ecology. During low tides, corals close to shore are exposed, and local communities and visitors walk on the corals for leisure and collect corals as souvenirs (Figure 10). Thus, public education and awareness is an important future consideration as pressure from tourism will undoubtedly increase as island tourism is developed in Myanmar. The sale of corals on public beaches provides further evidence of low awareness and weak law enforcement at the local level (Figure 11). Blast or dynamite fishing has unfortunately occurred extensively in the past in Myanmar waters. In recent years, however, local authorities and communities have managed to control the use of this destructive practice more effectively, and blast fishing has only been rarely recorded during recent surveys of the Myeik Archipelago (Figure 12).

Other pressures include litter, anchor damage as well as crown-of-thorns starfish (COTS) have also

FIGURE 9: *Ghost net & impacts (Photos: FFI-Myanmar).*

FIGURE 10: *Fringing coral reef exposed in the extreme low tide, and local visitors walking on the corals and damaging corals (Photos: FFI-Myanmar and Facebook).*

FIGURE 11: *Corals are sold along the public beach as souvenirs (Photos: FFI-Myanmar and Facebook).*

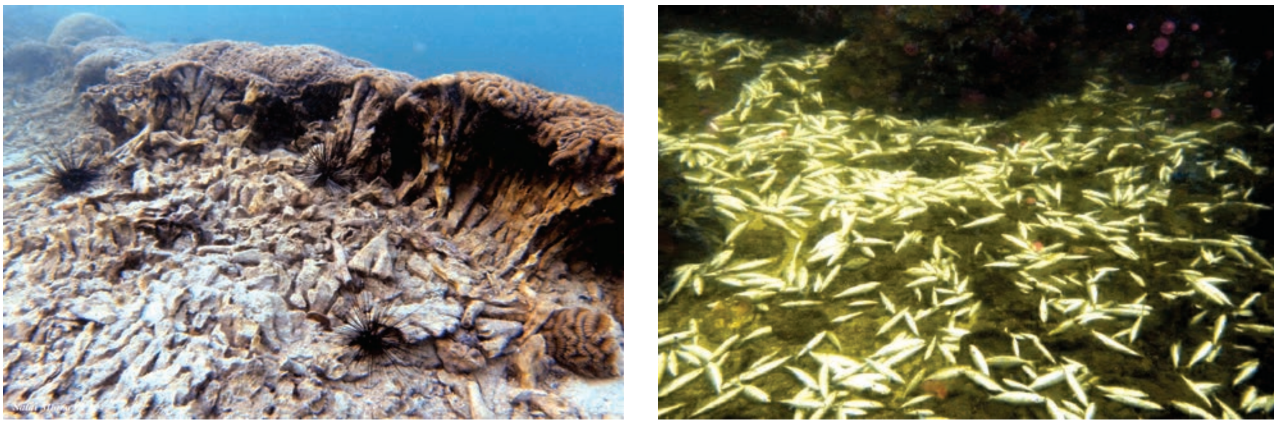

FIGURE 12: *Blast fishing evidence and impact to corals and fish (Photos: FFI-Myanmar).*

been recorded at some survey sites in the Myeik Archipelago (Figure 13); however, COTS are currently of minor concern as they are relatively uncommon and have not been found in outbreak proportions. The expansion of oil and gas exploration in Myanmar's offshore waters has not been a problem until recently. As it increases, it poses a more significant threat to marine ecosystems through possible habitat destruction, waste, and accidental oil spills into the marine environment.

A systematic coral disease survey was conducted between 2014 and 2016 in the Myeik Archipelago (Howard 2018). Some disease phenomena were

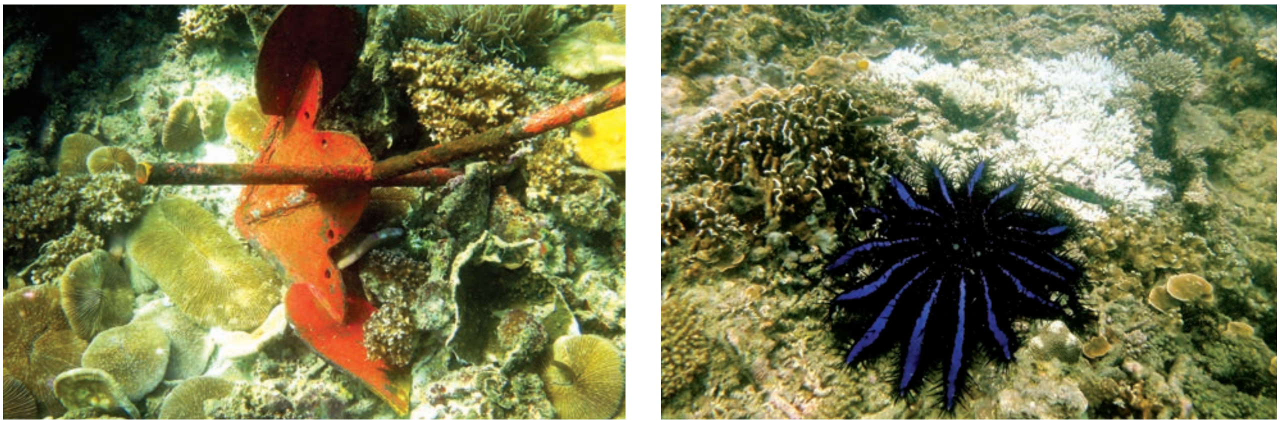

FIGURE 13: *Left: Anchor impact; Right: Crown-of-thorns starfish impact (Photo: FFI-Myanmar).*

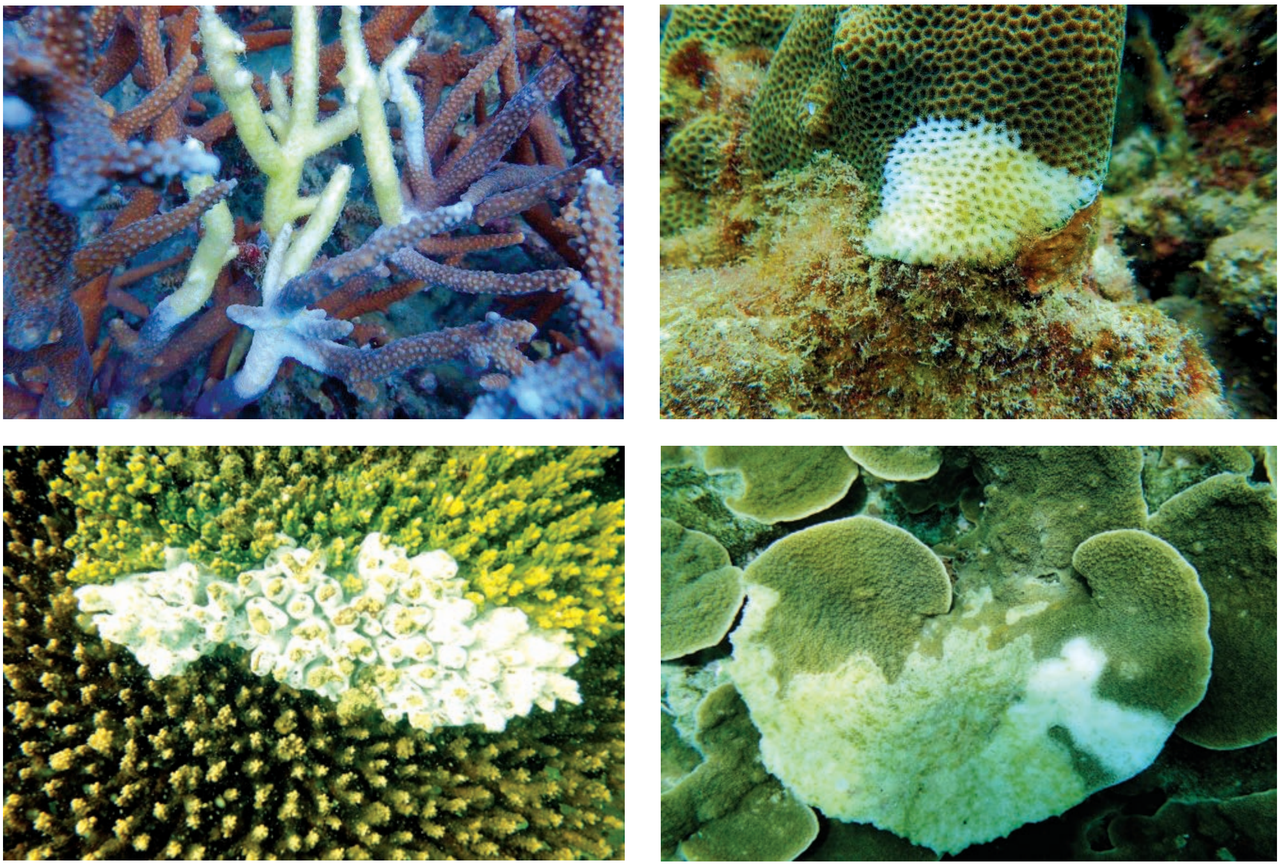

FIGURE 14: *Coral disease (Photos: FFI-Myanmar).*

observed during these surveys (Figure 14). Coral disease prevalence in this area ranged from 0% to 15% across all sites surveyed, with a mean disease level of 4.9%, which is similar to other Asia-Pacific locations (Howard 2018). However, disease prevalence was significantly greater at coral reefs adjacent to villages, possibly due to past dynamite fishing and untreated sewage from the settlements. The increased discharge of organic matter and nutrients from the Tanintharyi River watershed, notably during the wet season from terrestrial runoff, explains the higher prevalence of disease on those reefs close to the catchment (Howard, 2018). During the Mawdin coast surveys, there was also evidence of coral disease, and many reefs also appeared to be recovering from a past mortality event (Murray-Jones et al. 2017).

RECOMMENDATION

For effective conservation and management of coral reef ecosystems in Myanmar, the following recommendations are deemed essential:

1. Develop an education program for the high school curriculum regarding marine ecosystems, including coral reefs.
2. Use media tools to strengthen public awareness on marine and coral ecosystems in general, consequences for human well-being, as well as relevant laws and regulations.
3. Further develop effective and operational government marine conservation policies to protect the marine ecosystems, including coral reefs.
4. Continue to establish and strengthen a network of LMMAs in Myanmar to protect marine resources effectively, supported by relevant government departments and the empowerment of local communities.
5. MPA establishment based on the ecosystem-based management approach and strengthen collaboration between different government departments and local communities to conserve terrestrial and marine ecosystems effectively.
6. Develop collaborative mechanisms between relevant government departments, commercial fisheries companies, and local communities to conserve the coral ecosystem (e.g., market chain development).
7. Develop sustainable livelihood opportunities for communities near important coral reef sites (e.g., community-based ecotourism).
8. Develop standard monitoring and evaluation tools among countries to understand reef health effectively and efficiently.
9. Develop community monitoring tools to assist with local resource-use monitoring (e.g., fish catch monitoring).
10. Initiate collaborations among international coral conservation communities, organizations, and governments to fulfill the United Nations Sustainable Development Goals 2030.
11. Develop regional initiatives to address climate change problems in the region.
12. Increase capacity and funding for the effective conservation and management of coral ecosystems in Myanmar.

Response & Conservation Measures

On 30 November 2016, the Myanmar government founded the "National Coastal Resources Management Committee" chaired by the vice president and composed of representatives from concerned ministries to strengthen marine conservation in Myanmar. The coral reef ecosystem is included under the "Marine National Park" category for effective conservation in the new "Biodiversity and Conservation of Protected Areas Rules" adopted by the Assembly of the Union on 21 May 2018 and signed by the President of the Union of Myanmar. Corals are also included in the Myanmar Marine Fisheries Law (1990).

The Department of Fisheries has encouraged the establishment of Locally Managed Marine Areas (LMMAs) to effectively manage and conserve marine resources in collaboration with local communities and concerned government departments, as well as conservation organizations such as FFI and WCS. LMMAs help to conserve marine habitats, including coral reef ecosystems, for the sustainability of fisheries resources and livelihoods of local communities. Three LMMAs were established in the Myeik Archipelago in 2017 to support the sustainability of fisheries resources in the Myeik Archipelago.

The Forest Department is designating Marine National Parks (MNPs) for the inclusive conservation of marine and terrestrial ecosystems, protecting natural biodiversity along with their underlying ecological structures, supporting environmental processes, and promoting education and recreation. Currently, only one MNP exists in Myanmar, Lampi MNP, situated in the Myeik Archipelago and was designated in 1996. More MPAs are being established in consultation with various concerned stakeholders, supported by the regional government. All coral species are protected from harvest as they are listed under the 'Myanmar National Red List' issued by the Forest Department on 3 March 2020. FFI-Myanmar is assisting the Forest Department in developing a Marine Protected Area policy for Myanmar to aid in the effective conservation of marine ecosystems.

Universities are undertaking coral research supported by international organizations in the Myeik Archipelago and Rakhine coastal regions. FFI supports small grant programs for M.Sc. students and university staff, allowing further marine ecosystems research, including corals.

Several international organizations are also undertaking coral conservation activities in Myanmar. The Wildlife Conservation Society (WCS) started coral surveys in the Rakhine Coastal Region in 2018 to support MPA establishment for the sustainability of marine resources and the livelihood of local communities. The Myanmar Ocean Project (MOP) has started ghost net cleaning activities to support reef health from 2018, focusing on the Myeik Archipelago (Figure 15). The project has been working together with local authorities and communities in 87 sites across the Myeik Archipelago, including Lampi, Langann, Thayawthadangyi, and surrounding areas. Approximately 1,800 kg of nets have been removed from these locations since 2018. The majority of these marine entanglements were from gill nets lost or discarded from fishing vessels. In the future, the MOP plans to expand their activities throughout the coastal waters of Myanmar, supported by the Myanmar government and international organizations to support reef health for the sustainability of marine fisheries resources in Myanmar (Thanda, K. G. *personal communication*).

FIGURE 15: *Cleaning ghost nets (Photos: Myanmar Ocean Project).*

DATA CONTRIBUTORS

Data presented here were collected and compiled by Antt Maung (WCS-Myanmar) and Thanda Ko Gyi (Myanmar Ocean Project).

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SOUTHEAST ASIA

Philippines

Introduction

This status report covers the period from 2014 to 2020. Trends for both benthic and fish biomass were presented through time. Furthermore, data presentations were based on biogeographic regions: West Philippine Sea, Sulu Sea, North Philippine Sea, South Philippine Sea, Visayas Region (inland seas), and the Celebes Sea. All benthic data were obtained from line intercept transect (LIT), photo transect, and video transect methods. While reef fish data were obtained using the underwater fish visual census technique. There is a declining trend in the benthic cover for the South Philippine Sea and the Sulu Sea. In contrast, an increasing trend is observed for the West Philippine Sea. The other biogeographic regions showed a slight decline. Trends in fish biomass levels are site-specific and only pronounced when values are greater than 30 mt/km^2, while lower biomass values do not describe any trend.

Philippines is considered the center of marine biodiversity for habitat-forming species, marine shore fishes, and zooxanthellate corals (Carpenter and Springer 2005, Veron et al. 2009, Sanciangco et al. 2013). The extent of Philippine reefs is assessed to be approximately 25,000 km^2 (Gomez et al. 1994). The total annual economic value derived from fisheries and tourism is estimated to 4 billion US$/year or 140,000 US$/km^2/year (Tamayo et al. 2018).

The most recent coral reef status report at the country level was reported by Aliño et al. (2014). In that report, the highlight was the exploration conducted in the Philippine Rise (formerly known as Benham Rise) (Dizon 2014). The report also provides the historical dimension of the country's effort in reporting the status of coral reefs initiated by Gomez and Alcala (1979) and Gomez et al. (1981), which was followed by the production of the "Atlas of the Philippine Corals" headed by Aliño et al. (2002). From here on, the regular reporting of the status of coral reefs in the country commenced through the series of publication, the "State of the Coasts Report" by the Coral Reef Information Network of the Philippines (PhilReefs)" published in 2004, 2008, 2010, and 2014 (Aliño et al. 2014).

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This report covers the coral and reef fish status from 2014 to 2020, and it emphasizes the time series monitoring inside and outside the Marine Protected Areas (MPAs) conducted by various academic, government, and private institutions. However, due to the issue of data ownership, the results presented here do not cover all the data generated by various researchers. To further protect data ownership, the reporting was grouped by biogeographic regions and political provinces, namely: West Philippine Sea, Sulu Sea, North Philippine Sea, South Philippine Sea, Visayas Region (inland seas), and Celebes Sea (Aliño and Gomez 1994). This report also highlights other ways of interpreting coral cover introduced by Licuanan et al. (2019).

Status & Trends

STATUS OF CORAL COVER

The Philippine reefs have been categorized into six biogeographic regions. These are the North Philippine Sea, South Philippine Sea, West Philippine Sea, Visayan Sea, Sulu Sea, and Celebes Sea (Figure 1). There is a general decline in coral cover across the biogeographic regions except for the West Philippine Sea (WPS) (Figure 2). A decade of data shows that WPS increased from 10 to 32%, and the present cover is 6% higher than reported by Licuanan et al. (2019). This difference in coral covers relative to Licuanan's report, which are lower than the values in this report, is consistent with other biogeographic regions such as the North Philippine

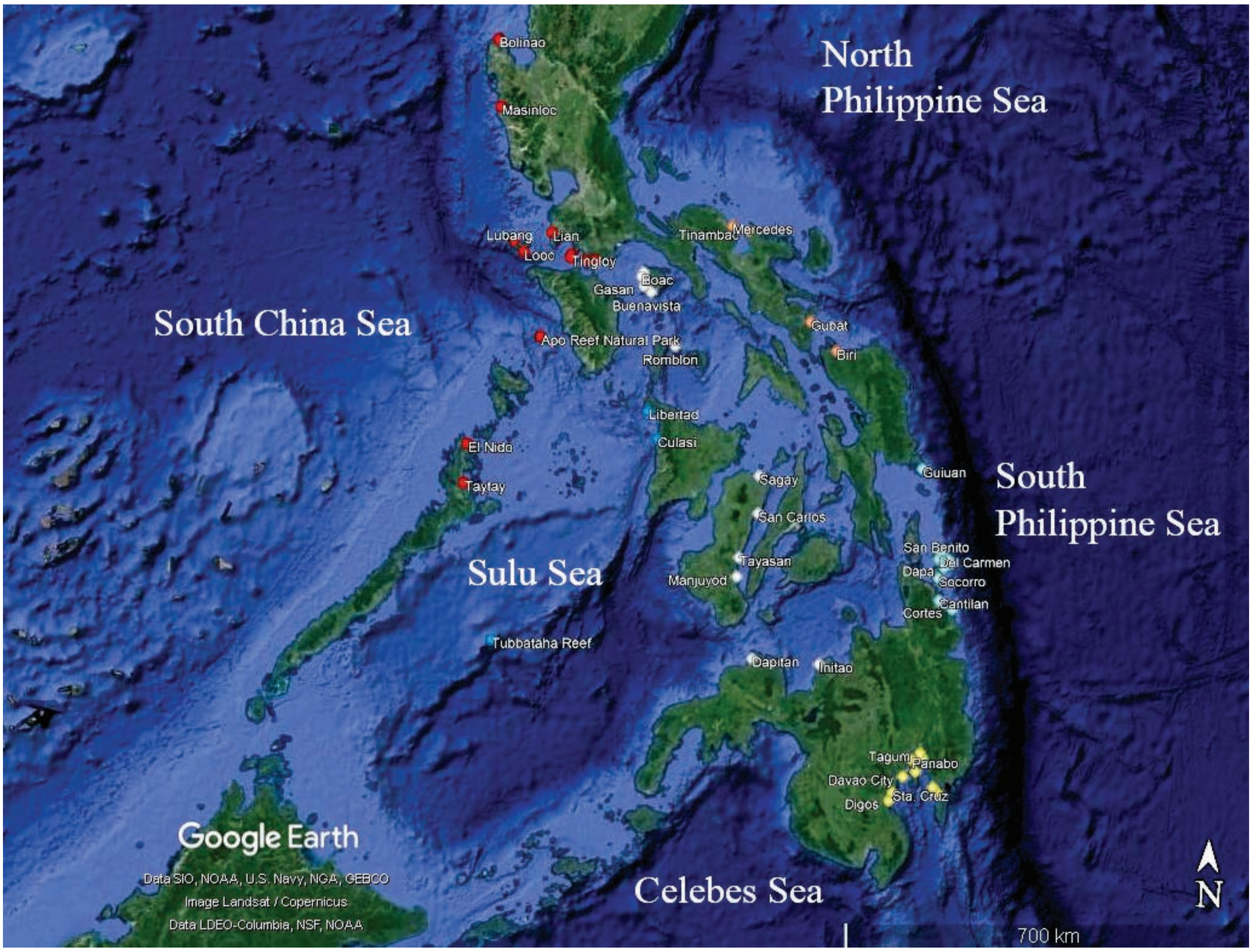

FIGURE 1: *Locations of coral reefs monitored for coral cover in 6 biogeographic regions in the Philippines.*

Sea and the Visayan Sea. On the contrary, the Sulu Sea (24%) is 4% lower than Licuanan et al. (2019). A similar result of 24% was generated for the Celebes Sea.

In this report, most of the sites are located inside the Marine Protected Areas, and this is probably one reason why it generated higher coral cover than Licuanan's report, which used consistent method across sites, while data used in this report were collected using various protocols such as photo quadrats and line intercept transect (LIT) (Licuanan et al. 2019, English et al. 1997). For this report, the standard error in many sites cannot be computed because of the limited information available.

STATUS OF REEF FISHES

The method employed in gathering fish data originated from the basic technique of "underwater fish visual census" by English et al. (1997). Although various modifications have been employed, the original concept is still followed, and the measurements are standardized by establishing a fixed area covered, which is 500 m^2. However, a new tool has been introduced called "Fishdrop" (Naval and David 2016), now known as "Fish-I." This technique uses stereo video recording, but it is not yet universally employed by various research groups in the Philippines due to data analyses requirement

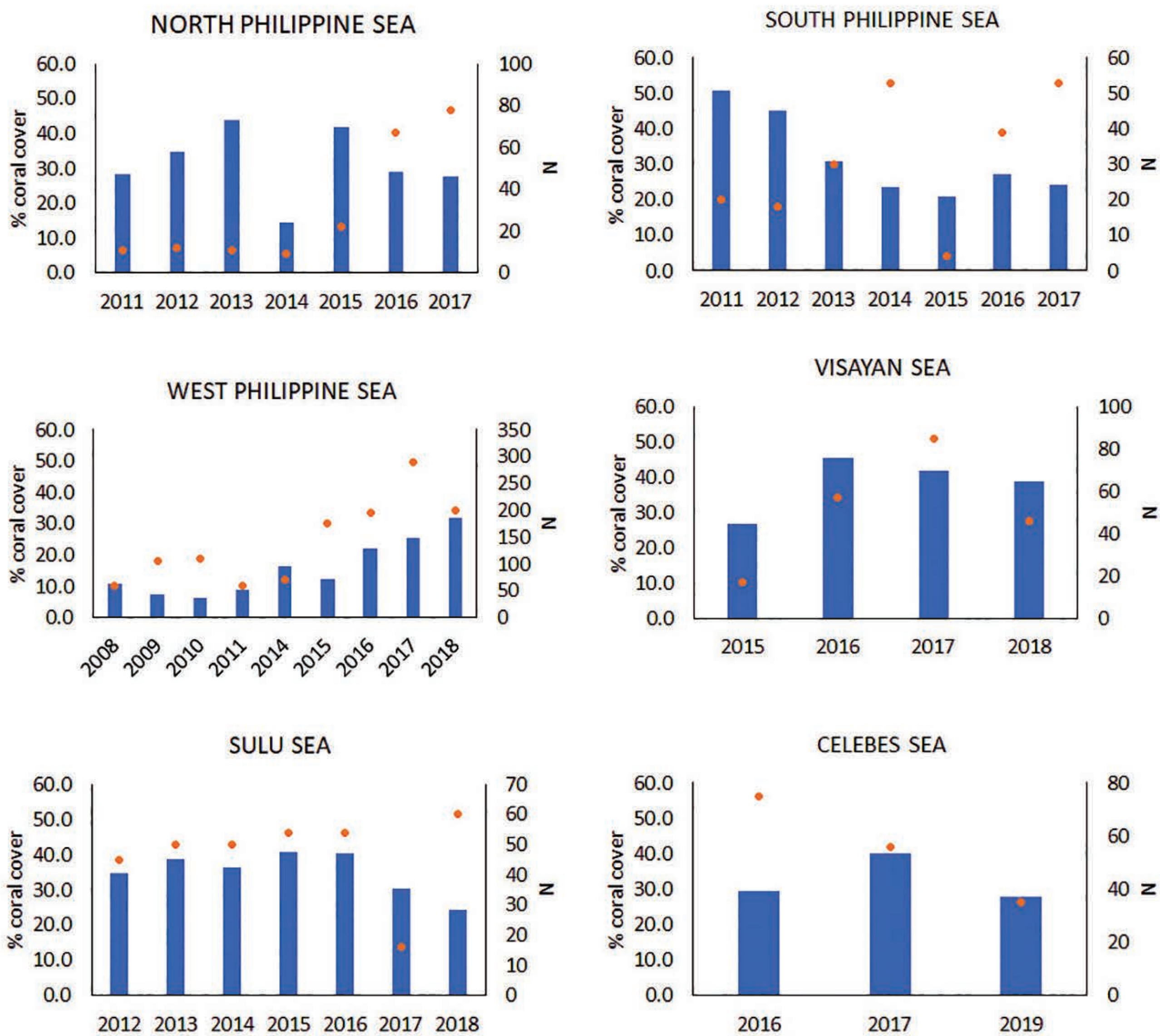

FIGURE 2: *Percent coral cover (hard and soft corals) of Philippine reefs across biogeographic regions through time. N represents the total number of transects used in LIT and photo quadrats inside and outside the Marine Protected Areas (MPAs).*

(e.g., computer software). But this tool has many advantages; for one, it provides a digital recording, which can be reviewed to verify or check particular species groups and serve as a data repository that can be revisited time after time.

It can be noticed that fish biomass data varies a lot between the biogeographic regions, averaging from 10 to 90 mt/km^2 (Figure 3A–F). The region with the highest fish biomass was in the Sulu Sea, specifically in the area of Tubbataha Reef Natural Park (Figure 3A). Management and monitoring activities on this site have been reported by Dygico et al. (2013). The rest of the regions had the highest values, averaging 40 to 50 mt/km^2 (Figure 3). All regions also had areas with fish biomass below 20 mt/km^2. These values fell within the upper limit of moderately fished sites based on the fishing category level by Nañola et al. (2006). However, there were many areas in all regions with less than 10 mt/km^2. This value is considered highly exploited, but other details cannot be presented in this report, as mentioned earlier. Further, within the Sulu Sea in the Province of Antique with 3 MPAs examined, it also showed very high fish biomass averaging 56 mt/km^2. This high biomass is not surprising considering that the Sulu Sea is one of the most productive sites with high reef fish diversity and density across regions (Muallil et al. 2020, Nañola et al. 2011).

For the West Philippine Sea biogeographic region (Figure 3B), data obtained was limited only for the year 2017. The areas examined were in Bacuit Bay with 8 MPAs, Provinces of Zambales (5 MPAs), and Occidental Mindoro (2 MPAs) (see Figure 1). The trend is showing higher fish biomass inside than outside the MPAs except for Bacuit Bay. The probable reason for higher fish biomass outside the MPAs in Bacuit Bay could be due to the impact of tourism activities in the area. There are minimal illegal fishing activities because there are other sources of income brought about by tourism. Furthermore, reefs covered in the area are within Bacuit Bay and are easy to protect and manage. It has been shown that illegal fishing and other destructive fishing practices are the main reasons that damage the reefs, and eventually, there is a decline of fish biomass due to loss of habitat (Russ et al. 2020).

The North Philippine Sea and South Philippine Sea (NPS & SPS) showed the same trend in fish biomass levels. Both regions had sites with high fish biomass values both inside and outside the MPAs averaging around 40 mt/km^2 (Figure 3C and Figure 3D). For the NPS, this was observed in the Province of Camarines Sur with 2 MPAs examined (Figure 3C). For SPS, this was observed in the Province of Surigao del Sur, specifically from Siargao Island, with 2 MPAs examined (Figure 3D). The rest of the sites had low fish biomass values between 10 and 20 mt/km^2 for both regions. These were in the Provinces of Northern Samar with 4 MPAs examined and in Camarines Norte and Sorsogon with one MPA each (Figure 3C). The same trend was observed for 6 MPAs in Southern Samar (Figure 3D) and is an indication that there could have been other factors affecting the slow recovery in these areas. As can be noticed in terms of benthic cover in Figure 2, the coral cover is declining slowly through the years. Though the benthic and fish data cannot be directly compared, it shows that it could have been brought about by the declining trend of benthic cover reported by Russ et al. (2020). The only exception is the data from the 6 MPA sites in Siargao, where fish biomass increased between 2014 and 2017.

In the Celebes Sea biogeographic region, based on 8 MPAs examined around the coast of Davao Gulf and 5 MPAs from Samal Island, there were no clear trends between inside and outside the MPA sites. Fish biomass values were below 20 mt/km^2 for most of the sites in Davao Gulf, with a few sites reaching 30 to 40 mt/km^2 (Figure 3E). The data variability may be attributed to the low biomass level (<20 mt/km^2). Patterns are only pronounced if biomass levels range from 30 mt/km^2 and higher, as seen in other biogeographic regions such as the Sulu Sea, West Philippine Sea, and the Philippine Sea.

In contrast, the Visayas Region had fish biomass values greater than 20 mt/km^2, where patterns are discernable (Figure 3F). Sites within the MPAs have higher fish biomass levels compared to sites outside the MPAs, as seen in Verde Island Passage (VIP) with 6 MPAs, Romblon (3 MPAs), Batangas (3 MPAs), and Negros Oriental (5 MPAs).

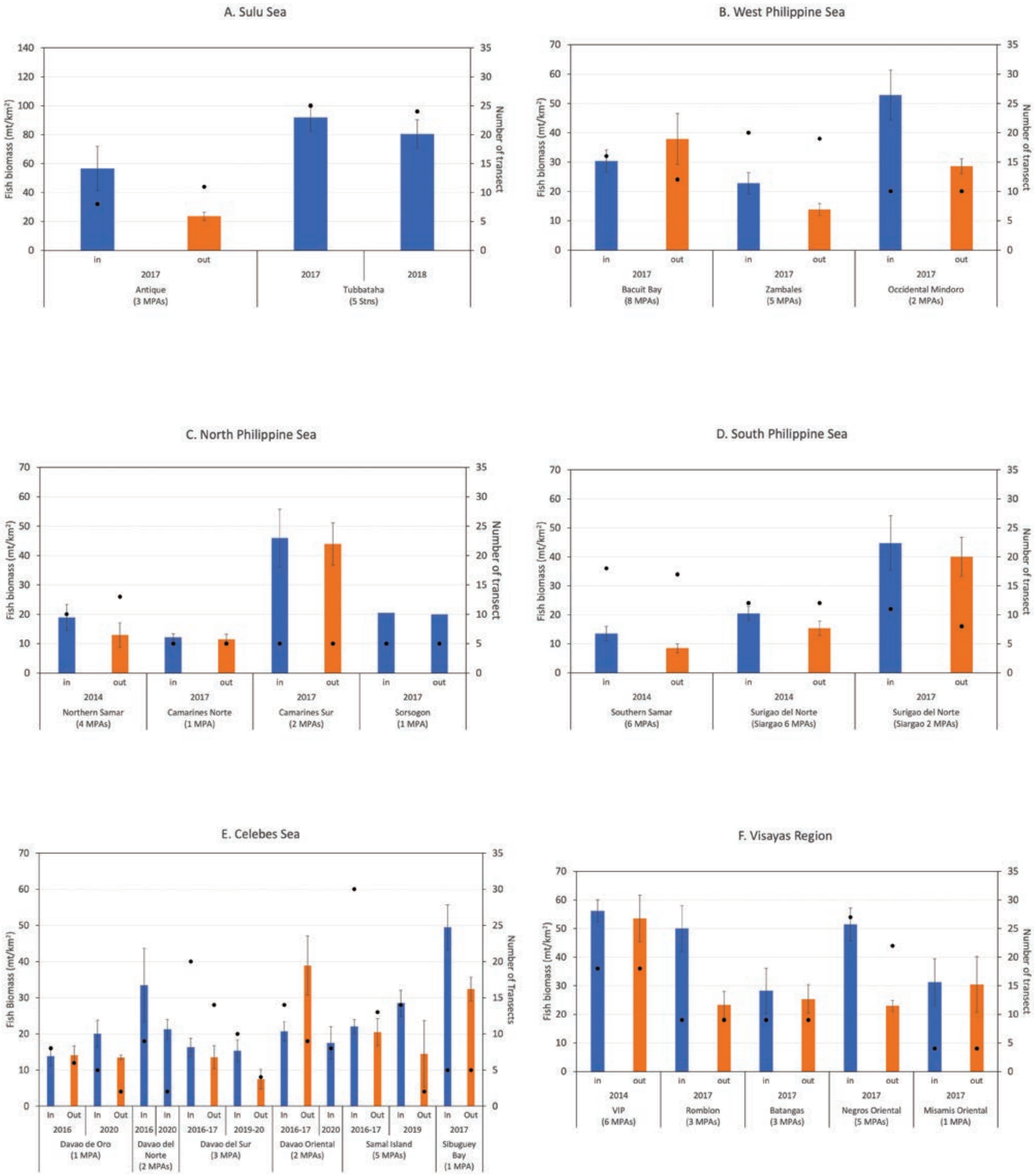

FIGURE 3: *Fish biomass levels in mt/km2 in various biogeographic regions in the Philippines, namely: (A) Sulu Sea, (B) West Philippine Sea, (C) North Philippine Sea, (D) South Philippine Sea, (E) Celebes Sea, and (F) Visayas Region.*

Coral Bleaching 2015–2017

OVERVIEW

Historically, anomalously high sea surface temperatures due to global warming and the El Niño Southern Oscillation (ENSO) events have caused mass coral bleaching events in the Philippines in 1997–1998 (Arceo et al. 2001), 2010 (Tun et al. 2010), and 2015–2017 (Philippine Coral Bleaching Watch unpublished data).

In 2010, the Philippine Coral Bleaching Watch (PCBW) — a citizen science initiative of the University of the Philippines Marine Science Institute (UP-MSI), was established to document the extent and severity of coral bleaching in the country. In 2015, the PCBW officially launched its online reporting platform via its social media platform to enjoin both scientists and non-scientists to document the impacts of the third Global Coral Bleaching Event (GBCE) that was unfolding at that time (Eakin et al. 2019). In 2016, PCBW forged a partnership with the Biodiversity Management Bureau of the Department of Environment and Natural Resources (BMB-DENR). This partnership has been instrumental in improving the reporting of coral bleaching incidence during major mass coral bleaching events in the Philippines. It is also creating awareness and educating the public on the importance and urgency of protecting and managing Philippine coral reefs, which are deteriorating rapidly in recent years due to multiple local stressors and global threats such as thermal stress and coral bleaching (Burke et al. 2012).

DATA COLLECTION

Through social media channels, PCBW regularly disseminates the US NOAA Coral Reef Watch (CRW) bleaching alert area (BAA) maps generated using their 5-km daily products (Liu et al. 2014). PCBW encourages the public to monitor and assess coral bleaching severity using a five-point scale in areas where bleaching is likely due to elevated thermal stress.

The PCBW online report form (http://tinyurl.com/phbleachingreport) is open to the public and can be accessed through any web browser or through an Android application available in Google® Play Store for smartphones and tablets (https://bit.ly/PHBleachingApp).

The form's offline editable version (pdf) can also be submitted through the official PCBW email address (phbleaching@msi.upd.edu.ph). All PCBW report submissions were carefully checked for completeness and validated by the PCBW technical team to confirm if the reports submitted coincided with the BAA maps and if the reported bleaching may have been caused by other stress factors (e.g., COTS infestation, disease, etc.).

RESULTS

In terms of spatial extent, coral bleaching incidences were observed in 36 (54%) out of the 66 coastal and island provinces (Figure 4A). Of the 201 confirmed coral bleaching reports, 79% (159) were recorded in 2016 and 21% (42) in 2017. In 2016, coral bleaching incidences were reported almost year-round (Figure 4B). Around 81% (129 of 159) reported low to mild bleaching (severity 1 and 2), while 19% (30 of 159) reported between moderate to severe/extreme bleaching (severity 3, 4, and 5). Moderate to severe coral bleaching was reported in April until October (Figure 4B), with the most severe bleaching reported for Honda Bay, Palawan (Figure 5), and Suhoton Bay, Surigao del Norte reported in May and June 2016, respectively. PCBW also received 125 out of 284 (44%) reports of no bleaching from February to June 2016.

In 2017, there were only 42 reports of coral bleaching. Similar to 2016, 71% (30 of 42) reported low to mild bleaching, while 29% (12 of 42) reported moderate to high bleaching (severity 3 and 4). Moderate to high bleaching was observed in April and from June to November 2017 (Figure 4B).

In 2015, only four (4) coral bleaching reports were submitted to PCBW (Table 1). Coral reef areas in the provinces Polillo, Quezon, and Maripipi, Biliran were reported to have suffered high bleaching in April and November, respectively. Moderate bleaching was recorded in Bauan, Batangas in October, while low-level bleaching was also reported in November in another reef site in Polillo, Quezon.

(A) (B)

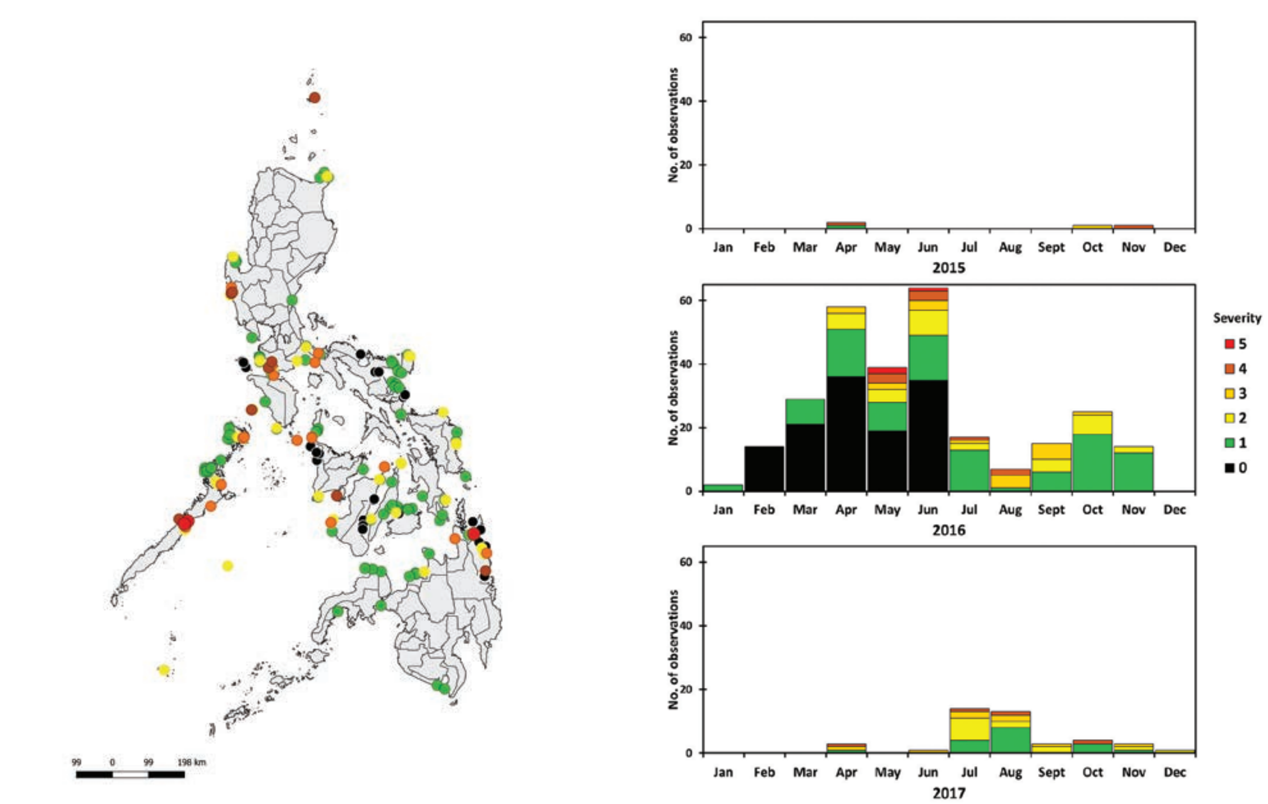

FIGURE 4: *(A) Extent and severity of coral bleaching incidence (or the absence of) from 2015 to 2017 in the Philippines; (B) Summary of coral bleaching and no bleaching reports submitted to PCBW from 2015 to 2017.*

FIGURE 5: *Extreme coral bleaching in Honda Bay, Palawan in 2016.*

A total of 326 reports were submitted from 2016 and 2017 (Table 1), i.e., coral bleaching (201) and no bleaching (125). A majority of the submitted reports (67%) came from two national-level assessment programs on coral reefs and monitoring of MPAs in the country at that time, namely: the National Assessment of Philippine Coral Reefs (NACRE) Program funded by the Department of Science and Technology (DOST) and the Fish Forever Project of Rare Philippines. The rest (33%) were reports submitted by citizens.

DISCUSSION

Presented here are the most updated coral bleaching observations by coral reef researchers/scientists and non-scientists during the global coral bleaching event (GCBE) from 2015 to 2017. Overall, low to mild bleaching was reported in several locations in the Philippines for the said period. There were significantly more reports received in 2016 compared to 2015 and 2017. One possible reason for the low to mild bleaching incidence is the

TABLE 1: *Summary of coral bleaching severity from 2015 to 2017 in the Philippines by the Philippine Coral Bleaching Watch (PCBW).*

SEVERITY*	2015	2016	2017	NO. OF OBSERVATIONS (2015–2017)
1 (Low; 1–15%)	1 (25%)	98 (61.64%)	17 (40.48%)	115
2 (Mild; 16–33%)	0	31 (19.5%)	13 (30.95%)	44
3 (Moderate; 34–45%)	1 (25%)	18 (11.32%)	8 (19.05%)	26
4 (High; 46–75%)	2 (50%)	9 (5.66%)	4 (9.52%)	13
5 (Severe; 76–100%)	0	3 (1.89%)	0	3
Total Bleaching Reports	**4**	**159**	**42**	**201**
Total Reports of No Bleaching	**0**	**125**	**0**	**125**
Total Observations	**4**	**284**	**42**	**326**

*Severity scale modified from Marshall and Schuttenberg (2006) to track coral bleaching progression.

fact that the average hard coral cover in shallow reef areas sampled (i.e., less than 5 meters) is 22.8%, as well as the absence of reefs in 'excellent' condition compared to previous national assessments (Licuanan et al. 2019). In addition, coral reefs in the Philippines have been previously impacted by the 1998 and 2010 mass bleaching events, which could have significantly reduced coral cover and possibly the cover of the more bleaching susceptible coral taxa. These are some of the plausible reasons for the degree of severity in succeeding major bleaching events.

Moreover, accurate estimates of coral bleaching severity are highly dependent on the observers' expertise, skills, and experience in assessing bleaching severity. The timing of these observations within a known bleaching year is also important to note; it is possible that severity scores reported can change through time (from 1 to 5), depending on the coral reef monitoring frequency for specific locations within a known bleaching year/s.

Despite PCBW's best efforts to solicit coral bleaching reports from the general public through social media platforms and improve the ease of reporting through the development of the PCBW app, there is still a significant level of underreporting at the national scale. No coral bleaching reports were submitted to PCBW for reef areas in Southern Mindanao, Southern Palawan, Tawi-Tawi Archipelago, northeast, and northwest Luzon. The Philippines has one of the longest coastlines in the world, and it would be challenging to assess the impact of mass coral bleaching unless there is a nationwide coordinated response effort to make sure that representative coral reef regions within the country are adequately sampled. Underreporting could also be because coral bleaching events in the Philippines coincides with the southwest monsoon (habagat) from May to October. The SW monsoon is characterized by heavy rains and the passage of tropical cyclones, which are not ideal conditions for conducting scientific field surveys and recreational activities like snorkeling and diving. Given these realities, ordinary citizens, especially those who live near coastal areas and those involved in the tourism industry - like dive operators, resorts, etc. need to be appropriately trained on assessing coral bleaching and reporting.

It is interesting to note that there were also coral reef areas that had no reported bleaching in 2016, which could mean two things: 1. that there are still coral reef areas that are possibly more resistant or resilient to thermal stress or 2. the thermal stress has not set in at the time of observation. In the absence of a national coral reef monitoring system using standard protocols and methodologies, there is no way to accurately determine the contribution of bleaching-induced coral mortality vis-a-vis baseline coral cover conditions and given impacts from other local stressors such as sedimentation.

There is also a need to identify and monitor key environmental factors and biological indicators such as coral recruitment rates that confer resilience to specific coral reef locations.

Coral reef areas that survive well despite exposure to repeated bleaching events are candidate areas for protection since these could serve as source reefs for rehabilitating degraded areas. Enhancing coral recovery rates of bleached coral reefs should also be done at local scales, which can be achieved by implementing appropriate coastal and fisheries management interventions to ensure the long-term resilience of Philippine coral reefs given local stressors and more frequent local and global coral bleaching events in the future.

Drivers & Pressures

The main driver of coral bleaching is the increased sea surface temperatures higher than ambient conditions due to climate change (Hoegh-Guldberg et al. 2007), which is exacerbated by local disturbances such as UV radiation, eutrophication, excessive sedimentation, and reduced salinity (Gleason and Wellington 1993, Carilli et al. 2009; Alemu and Clement 2014, Fisher et al. 2019).

RECOMMENDATION

By law, the management of Philippine reefs, particularly along the coastal areas, is under the local government units (LGUs), and these are mainly the small MPAs. The much larger MPAs cut across several LGUs, and they are managed by the National Government, particularly the Department of Environment and Natural Resources - Biodiversity Management Bureau (DENR-BMB). The main issue on the management aspect, particularly the small MPAs, is budget limitation and availability of expertise. Regular budget allocation at the LGU level for the monitoring of reefs is not yet in place. It is always outsourced somewhere else. It is only in some areas that the local government has collaborated with Higher Educational Institutions (HEIs) in doing the reef monitoring. However, the direct management (e.g., patrolling, strengthening of the management body) is in the hands of the LGUs.

DATA CONTRIBUTORS

The list of contributors served as project leaders of various programs supported by various agencies. Project staff who were also involved in collecting data are not included.

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SOUTHEAST ASIA

Singapore

Introduction

The Republic of Singapore is a metropolitan island state with a population of 5.7 million people (DoS 2020) and measures 50 km east to west and 27 km north to south, and it is fringed by more than 50 smaller islands (Chou et al. 2019). Singapore's territorial sea is estimated to be 600 km^2 (Sin et al. 2016) and is heavily utilized for shipping activities. To the north, the narrow estuarine Johor Strait separates Singapore from Peninsular Malaysia, with depths averaging 10 m. To the south, the wider Singapore Strait has an average depth of 30 m and supports marine habitats that experience a more saline environment (Chou et al. 2019).

The differences in environmental conditions have restricted coral reefs to the southern islands of Singapore. Extensive coastal developments that commenced in the 1960s have reduced the present reef area to 13.3 km^2, compared to an estimated 39.9 km^2 in 1953 (Tun 2012). Despite losing more than 60% of its original coral reef area, there remains considerable diversity, with over 30% coral cover along the shallow reef crests that extends to the lower depth limit of 5 m. Studies have recorded more than 250 hard coral species (Huang et al. 2009), nearly 200 fish species (Low and Chou 1994), and numerous invertebrate taxa (Wee et al. 2019).

While over 60% of Singapore's shorelines are now modified by the construction of seawalls (Lai et al. 2015), these coastal defense structures have also inadvertently created new habitats and enabled distinct coral and fish communities to recruit and thrive (Ng et al. 2012, Taira et al. 2018, Lim et al. 2020). Due in part to such habitat modifications, high sediment load, especially at sites closer to the mainland, and natural environmental variation, there is considerable spatial structuring of coral and fish communities despite the small total reef area (Dikou and van Woesik 2006, Taira et al. 2018, Wong et al. 2018, Chow et al. 2019).

While chronic sedimentation, coastal development, and high shipping volume (Tan et al. 2016) are the biggest threats to Singapore's coral reefs, acute disturbance posed by globally elevated sea surface temperatures such as in 1998, 2010, and 2016, have significantly impacted Singapore's coral reefs (Guest et al. 2012, 2016a, Ng et al. 2020). In response to pressing conservation needs, Singapore established its first marine park in 2014, the Sisters' Islands Marine Park, to promote outreach, education, research, and conservation of its coastal and marine environment (Figure 1).

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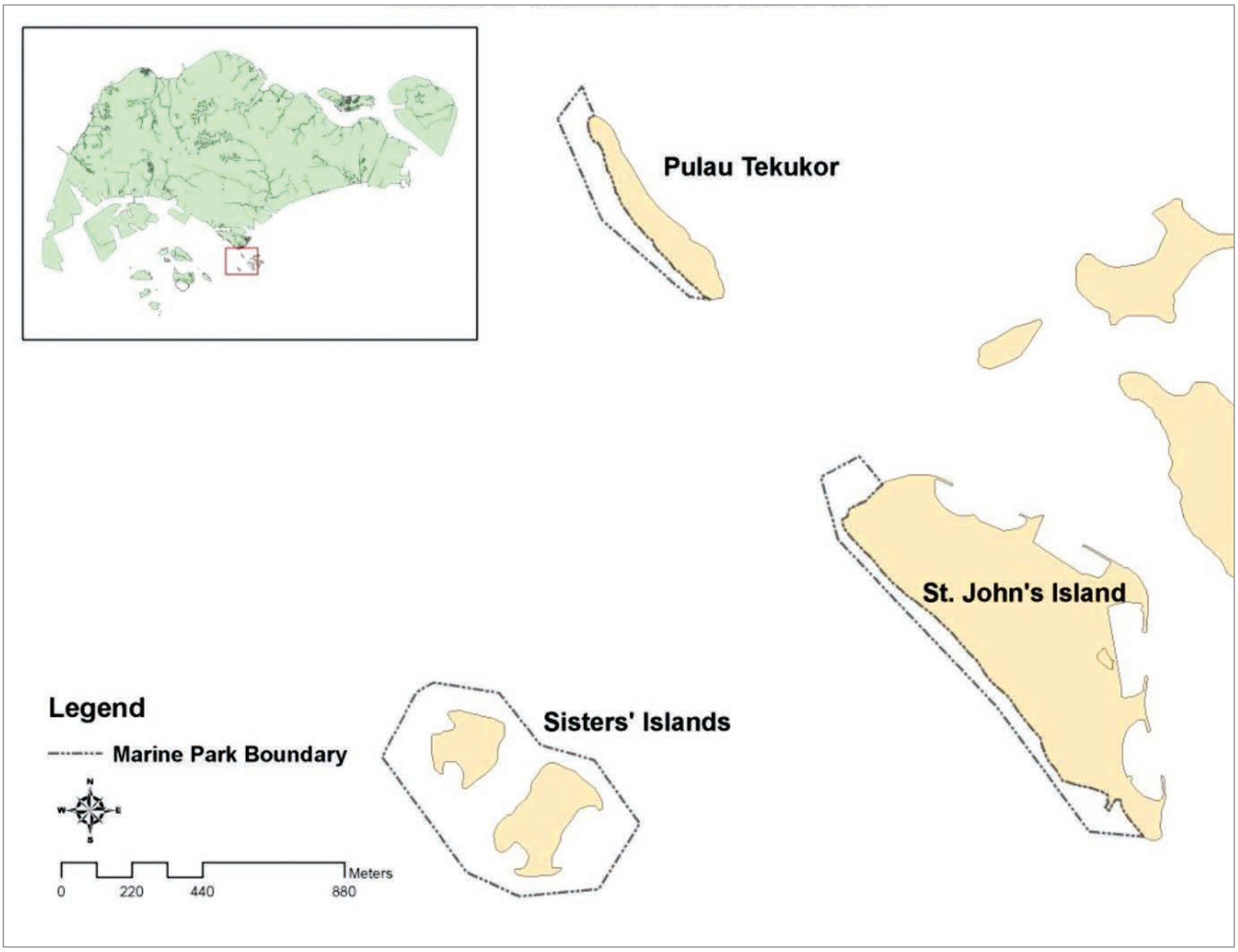

FIGURE 1: *Sisters' Island Marine Park established in 2014.*

Coral reef monitoring in Singapore is undertaken by different research groups. The monitoring frequency and sites chosen depends on the scope of each project, with sites such as Pulau Satumu, Pulau Hantu and Kusu Island more frequently surveyed than others (Guest et al. 2016b). The data presented in this chapter were collated from published literature and empirical data from various research teams (National University of Singapore, Nanyang Technological University, National Parks Board and Blue Water Volunteers, Loo and Chou 1995, Guest et al. 2016b, Wong et al. 2018) and are focused on the subtidal coral reefs south of mainland Singapore.

Status & Trends

Between 1986 and 2019, mean coral cover on Singapore's reefs had fluctuated in response to major disturbances. The largest decline in mean coral cover was from 30% in 1986 to 10% in 1990, driven at least in part by reclamation-related degradation of the reef environment during this period. There were signs of recovery from 1990, but mean coral cover decreased by 4% in the late 1990s, coinciding with the 1998 mass bleaching episode. Following this decline, the mean coral cover had increased to 25% in 2009. Interestingly, the 2010 mass bleaching resulted in a decline of 2% mean

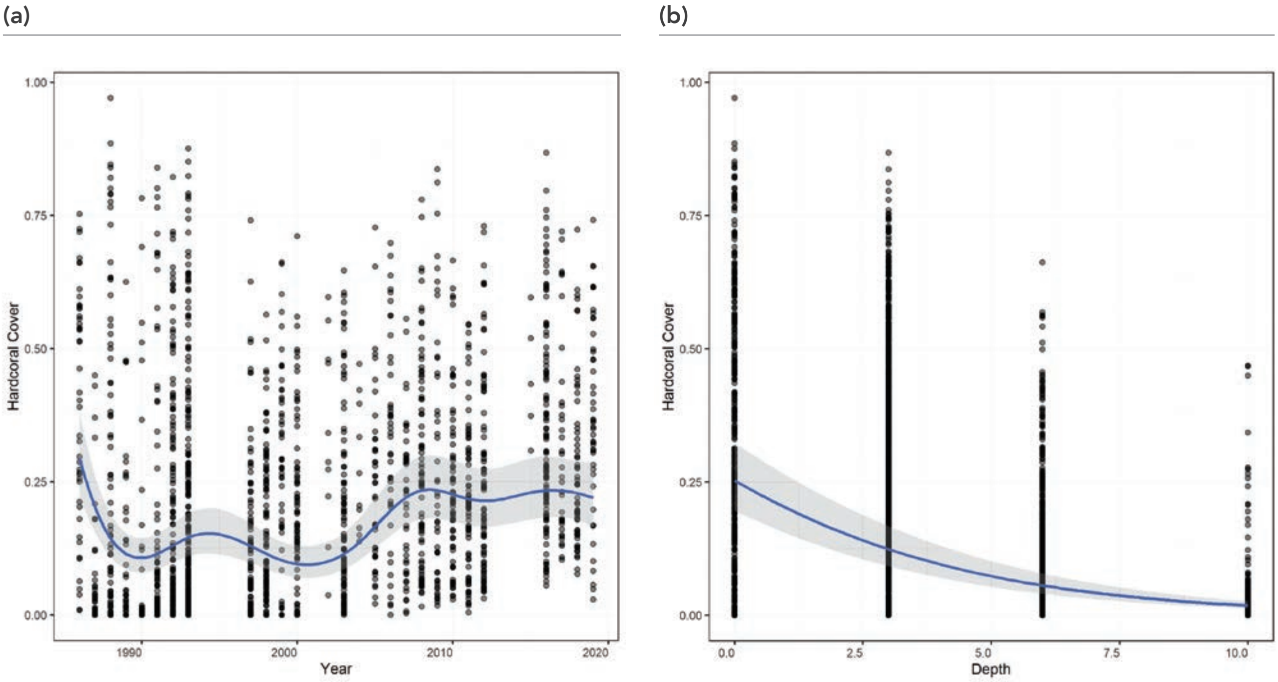

FIGURE 2: *Singapore coral cover across (a) years and (b) depths.*

coral cover, and the impact was less pronounced than in 1998 (Guest et al. 2016a). Singapore's reefs showed signs of recovery following that, but the ensuing 2016 mass bleaching episode caused a dip in mean coral cover. At present, the live coral cover remains high and at levels last seen in the 1980s; this can be as high as between 50% and 60%, such as at Pulau Satumu and Kusu Island (Guest et al. 2016b, Wong et al. 2018). Coral communities have changed considerably over the past three decades—genera such as *Montipora*, *Merulina*, and *Platygyra* have decreased in abundance, accompanied by an increase in *Favites, Mycedium, Pachyseris, Dipsastraea*, and *Echinophyllia* (Guest et al. 2016b). Mean coral cover decreases considerably with increasing depth, to less than 10% beyond the first 3 m below chart datum, indicating that reduced light availability is driving losses at the deeper reef areas (Guest et al. 2016b, Chow et al. 2019).

Coral Bleaching 2016

In early April 2016, sea surface temperature (SST) exceeded the maximum monthly mean (MMM, 29.8°C) (NOAA 2000) for three and a half months and peaked at 31.4°C in May. During this period, sea surface temperatures were higher than the bleaching threshold (30.8°C) on ten days. Consequently, subtidal reefs exhibited moderate to severe bleaching (25.5% to 76.3%), and species such as *Dipsastraea matthaii, Goniastrea favulus*, and *Montipora stellata* were among the most affected (Ng et al. 2020). This bleaching episode reduced the mean coral cover in the four sites surveyed from 24.4% (pre bleaching) to 15.7% (post bleaching), with coral community shifts observed at some sites (Ng et al. 2020). While most reefs recovered by March 2017, some did not revert to pre-bleaching levels of coral cover, indicative of differing resilience and resistance of the reefs at different sites to heat stressors (Ng et al. 2020).

Drivers & Pressures

In addition to the 2016 mass bleaching event, there were two major bleaching events in 1998 and 2010, as well as two mild to moderate bleaching episodes in 2013 and 2014. From March to June 1998, sea surface temperature (SST) exceeded the maximum monthly mean for four months and peaked at 32°C in May. Sea temperatures around two sites surveyed (Pulau Hantu and St John's Island) were elevated by 1–2°C, leading to the bleaching of 50–90% of zooxanthellate reef organisms, particularly the hard corals, soft corals, and anemones (Reef Ecology Lab 2020). The bleaching effect extended to 6 m, the lower depth limit for coral growth locally, and eventually resulted in the deaths of nearly 30% of coral colonies. In April 2010, sea surface temperature (SST) exceeded the maximum monthly mean for three months and peaked at around 31°C in May. This episode resulted in 66% of the hard corals bleached, with the coral cover declining by 21% at Pulau Satumu (Guest et al. 2012, Guest et al. 2016a).

From June to July 2013, sea surface temperature (SST) exceeded the maximum monthly mean and peaked at 31°C in June. Surveys conducted during this period recorded a bleaching prevalence of 2.3% to 12.4% (Chou et al. 2016). In the following year, sea surface temperature (SST) exceeded the maximum monthly from early May 2014. The daily mean temperature peaked at 31.1°C in June, and bleaching prevalence ranged from 0% to 11.8% (Bleach Watch Singapore 2016, Taira et al. 2017).

In the early 1900s, historical development and settlement have already led to some impacts on the reefs, with smaller losses in reef area (Hilton and Manning 1995). Since the 1960s however, extensive land reclamation and port development work have reduced reef area, while spoil dumping as well as constant dredging to maintain shipping lanes and anchorages have also led to waters around Singapore carrying increasingly high sediment loads (Dikou and van Woesik 2006, Sin et al. 2016, Chou et al. 2019). These have led to chronic impacts on corals due to reduced light penetration and direct deposition on corals, as more energy is expended by the coral holobiont to carry out photosynthesis and clear sediment. The negative effect on reef communities is manifested in the reduced coral cover between the 1980s and 2000s (Guest et al. 2016b).

RECOMMENDATION

With the growing intensity and frequency of coral bleaching events, it is recommended that a formal long-term coral reef monitoring program be instituted to provide regular and timely updates of the status of coral reefs. The monitoring of environmental parameters is also important, both through satellite imagery and in-situ loggers, as these enable early warning of stressors and allow for correlation of parameters to coral community changes. As development and land reclamation works are expected to continue around the southern islands, current environmental management practices should be maintained; including conducting environmental impact assessments before any planned development with recommended mitigation measures such as coral translocation and sediment control; followed by the establishment and implementation of an Environmental Monitoring and Management Plan (EMMP) during and after the development to monitor development impacts. Habitat restoration and eco-engineering efforts should continue to be implemented and developed to recover reefs impacted by past development works (Ng et al. 2016, 2017, Toh et al. 2017). Integrative research efforts among disciplines, public and private institutions are also needed to manage the increasingly complex challenges facing Singapore (Jaafar et al. 2018).

DATA CONTRIBUTORS

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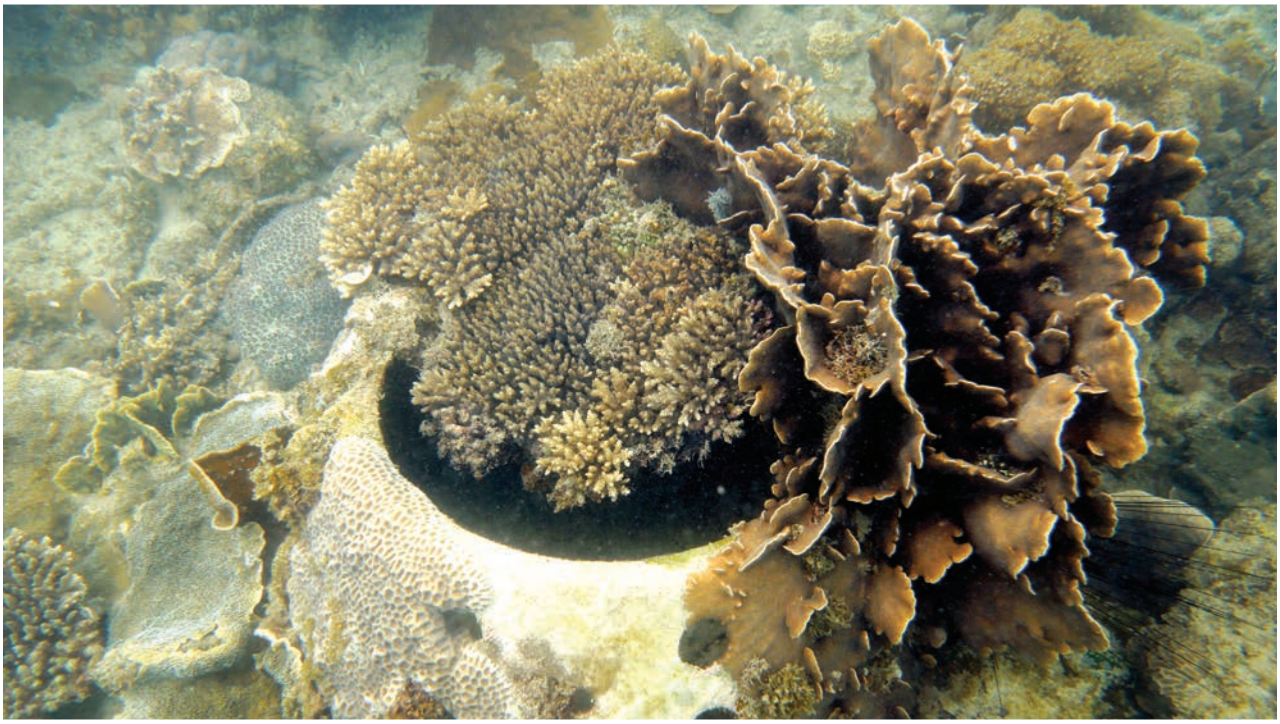

SOUTHEAST ASIA

Thailand

Introduction

Thailand is located in the middle of mainland Southeast Asia. The coastal areas of Thailand bordering the Gulf of Thailand and the Andaman Sea have suitable environmental conditions for coral reef growth and development. More than 296 scleractinian coral species have been recorded in Thai waters. There are over 300 relatively small islands along Thailand's coastline that span over 3,000 km, hosting approximately 240 km^2 of coral reef habitat. The coral reef areas may also be underestimated as there are several unreported coral communities on underwater pinnacles. Coral reef ecosystem services play a significant role in Thailand's economy, particularly in the tourism and fishery sectors; small-scale fisheries are dependent on healthy coral reefs and associated ecosystems, and many coral reefs are vital for marine ecotourism. Moreover, several coral reefs act as natural barriers, protecting coastal areas, communities, and beaches from strong waves.

Three types of coral reefs in Thai waters are recognized, i.e., coral communities with no true reef structure, developing fringing reefs, and early formation of fringing reefs. Coral reefs cover approximately 121 km^2 in the Gulf of Thailand, and there are about 119 km^2 of coral reefs in the Andaman Sea. Between these two regions, coral reefs in Thailand are generally classified into four geographical areas with different oceanographic conditions:

1. The Inner Gulf of Thailand: Chonburi Province.
2. The Eastern Gulf of Thailand: Rayong, Chanthaburi and Trat Provinces.
3. The Western Gulf of Thailand: Prachuab Kirikhan, Chumphon, Suratthani, Nakhon Si Thammarat, Songkhla, Pattani, and Narathiwat Provinces.
4. The Andaman Sea: i.e., Ranong, Phuket, Phangnga, Krabi, Trang, and Satun Provinces.

The coral reef monitoring data presented here were compiled from major monitoring programs in Thailand, implemented by the Department of Marine and Coastal Resources, Department of National Parks, Wildlife and Plant Conservation, and Marine Biodiversity Research Group of Ramkhamhaeng University. Most coral reef surveys were conducted using the belt-transect method (three replicates of 0.5 x 30 m) in the shallow (0–6 m in depth) and deep zones (over 6 m in depth). The live coral cover was observed and evaluated as colony area/unit area in each belt-transect. Ninety-six study sites in the Gulf of Thailand and the Andaman Sea were analyzed (Figure 1), including:

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- **11 study sites in the Inner Gulf of Thailand**
  - » Mu Ko Sichang, Chonburi Province (9 study sites)
  - » Ko Lan and Ko Khrok, Chonburi Province (2 study sites)

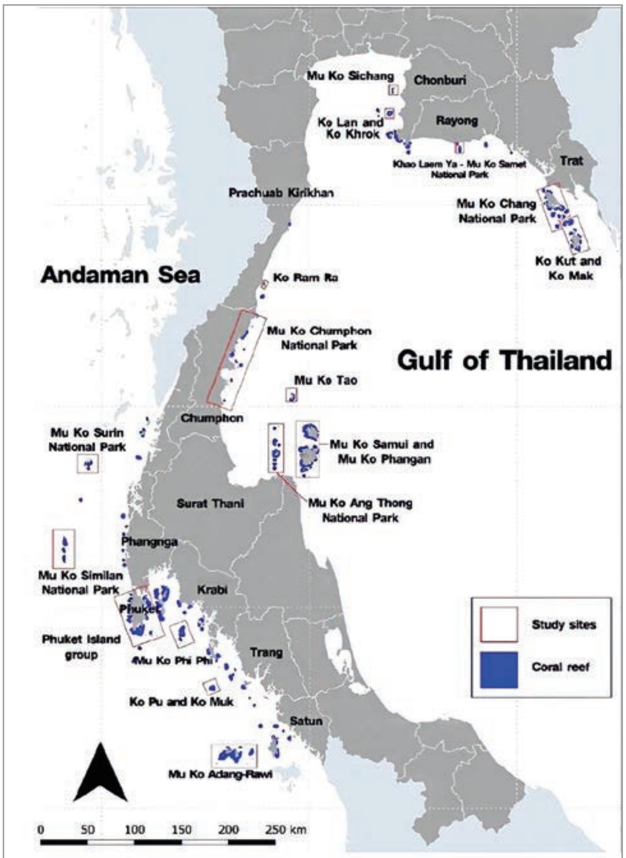

FIGURE 81 *Map showing the reef sites for monitoring programs in the Gulf of Thailand and the Andaman Sea.*

FIGURE 1: *Map showing the reef sites for monitoring programs in the Gulf of Thailand and the Andaman Sea.*

- **17 study sites in the Eastern Gulf Thailand**
  - » Khao Laem Ya - Mu Ko Samet National Park, Rayong Province (8 study sites)
  - » Mu Ko Chang National Park, Trat Province (7 study sites)
  - » Ko Kut and Ko Mak, Trat Province (2 study sites)

- **34 study sites in the Western Gulf Thailand**
  - » Ko Ram Ra, Prachuab Kirikhan Province (4 study sites)
  - » Mu Ko Chumphon National Park, Chumphon Province (7 study sites)
  - » Mu Ko Ang Thong National Park, Surat Thani Province (13 study sites)
  - » Mu Ko Tao, Surat Thani Province (3 study sites)
  - » Mu Ko Samui and Mu Ko Phangan, Surat Thani Province (7 study sites)

- **34 study sites in the Andaman Sea**
  - » Mu Ko Surin National Park, Phangnga Province (9 study sites)
  - » Mu Ko Similan National Park, Phangnga Province (7 study sites)
  - » Phuket Island group, Phuket Province (3 study sites)
  - » Mu Ko Phi Phi, Krabi Province (4 study sites)
  - » Ko Pu and Ko Muk, Trang Province (2 study sites)
  - » Mu Ko Adang-Rawi, Satun Province (9 study sites)

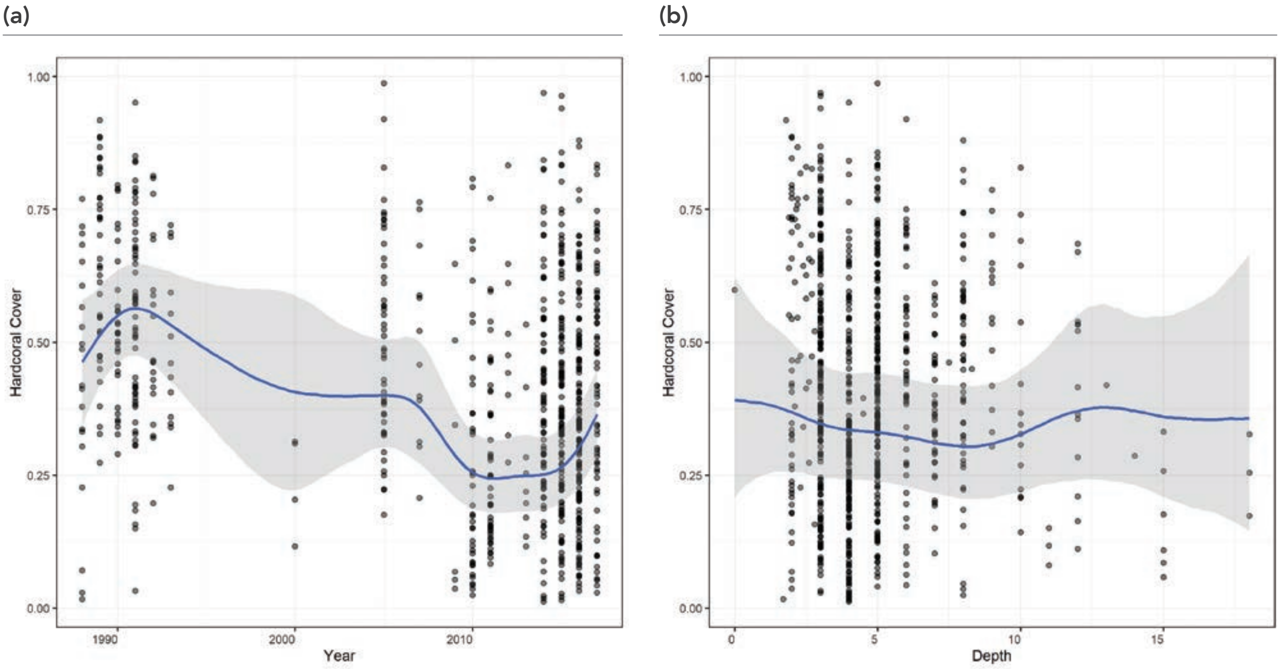

FIGURE 2: *Thailand coral cover across (a) years and (b) depths.*

Status & Trends

Total scleractinian coral cover at the reef monitoring sites in Thai waters has decreased considerably, from about 50% before 1990 to <40% in the last decade (Figure 2a). The moderate coral bleaching events in the Andaman Sea in 1991, 1993, as well as the severe coral bleaching event in the Gulf of Thailand in 1998, caused a reduction of coral cover by about 10% from 1991 to 2000 (Brown et al. 1996, Yeemin et al. 1998, Phongsuwan and Chansang 2000, Yeemin et al. 2009, Phongsuwan and Chansang 2012). Coral cover remained low after the year 2000 and dropped to 25% in 2010 because of severe coral bleaching in both the Andaman Sea and the Gulf of Thailand (Yeemin et al. 2010, Dunne 2012, Hoeksema et al. 2012, Sutthacheep et al. 2012, Phongsuwan et al. 2013, Sutthacheep et al. 2013a, 2013b). After the 2010 bleaching event, coral recovery has occurred slowly in the initial five-year period, followed by a higher recovery rate, particularly at Mu Ko Surin National Park in the Andaman Sea, and rising to about 38% coral cover in 2017 (Yeemin et al. 2012a, Yucharoen et al. 2015).

Coral cover on the shallow reef sites (0–6 m in depth) is about 5% higher than those on the deep sites (over 6 m in depth) (Figure 2b). *Acropora* species dominate at several reef sites; however, their percentage cover exhibits large spatial and temporal variations because of their sensitivity to coral bleaching events. Some reef sites in the Gulf of Thailand show a long-term decline in *Acropora* species due to coral bleaching events (Yeemin et al. 2013a).

Coral recruitment has varied between years and reef sites in the Gulf of Thailand (Klinthong et al. 2013; Pengsakun et al. 2012, Yeemin et al. 2013b, Klinthong et al. 2015) and the Andaman Sea (Klinthong et al. 2014, Putthayakool et al. 2016;). After the 2010 coral bleaching event, coral recruitment in the Eastern Gulf of Thailand showed that *Porites* was the most dominant juvenile coral genus at several reef sites. Other common juvenile colonies were the genera *Fungia*, *Lithophyllon*, *Hydnophora*, *Turbinaria*, *Dipsastraea*, and *Oulastrea* (Sutthacheep et al. 2011). However, the common juvenile corals in Mu Ko Phi Phi in the Andaman Sea were *Acropora*, *Dipsastraea*, *Favites*, *Goniastrea*, *Lithophyllon*, *Pocillopora*, and *Porites* (Chamchoy et al. 2015). In contrast, the density of coral recruits in early 2016 at Ao Suthep in Mu Ko Surin National Park in the Andaman Sea was 26.3 recruits.m^{-2}, where *Acropora* spp. were the most common coral recruits (Pengsakun et al. 2016). According to their community structure, coral reef resilience in Thai waters has varied greatly among the study sites and major reef groups, mostly because of the differing bleaching resistance and tolerance of the dominant coral species. The study sites in the Gulf of Thailand often had much lower coral recruitment rates than other reef sites in the Andaman Sea (Sutthacheep et al. 2018, 2019).

Coral Bleaching 2016

A mass coral bleaching event in the Gulf of Thailand and the Andaman Sea was reported from April to May 2016. There were significant differences in the severity of bleaching among reef sites. The coral bleaching surveys were conducted at 102 stations in 14 provinces: 62 stations in the Gulf of Thailand and 40 stations in the Andaman Sea. Severe coral bleaching, where bleached corals comprised over 50% of live coral cover, was observed at 37 stations (36.27%). Among them, 20 stations are located in the Andaman Sea (50%), and the remaining 17 stations are in the Gulf of Thailand (27.42%). However, mortality following the bleaching in 2016 was much lower than the 2010 coral bleaching event because the southwest monsoon started from the end of April 2016, and the seawater temperature dropped rapidly (DMCR 2016, 2019).

A detailed study of coral bleaching in 2016 was conducted at 11 sites in Prachuap Khiri Khan, Chumphon, and Surat Thani Provinces, in the Western Gulf of Thailand. Peak seawater temperatures recorded were over 33°C in mid-May (Figure 3). The levels of coral bleaching varied significantly among the study sites. The most severe coral bleaching was observed at Ko Ngam Noi, Mu Ko Chumphon National Park, Chumphon Province (Figure 4), while the least severe was recorded at Ko Sang, Prachuap Khiri Khan Province. The levels of bleaching susceptibility also varied significantly among coral taxa. *Acropora florida, Fungia fungites, Montipora tuberculosa, Plerogyra sinuosa, Pocillopora* spp., and *Porites rus* showed high bleaching susceptibility. In contrast, *Favites abdita, Galaxea fascicularis, Goniopora planulata, Pavona cactus, P. decussata*, and *P. frondifera* were highly resistant to bleaching (Sutthacheep et al. 2016).

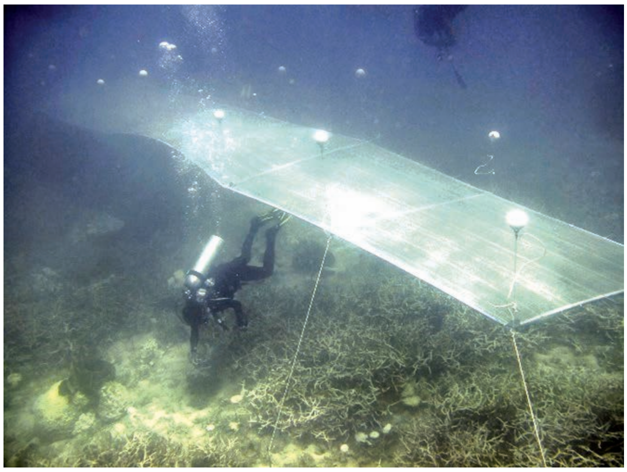

FIGURE 4: *A field shading experiment at Mu Ko Chumphon National Park, the Western Gulf of Thailand.*

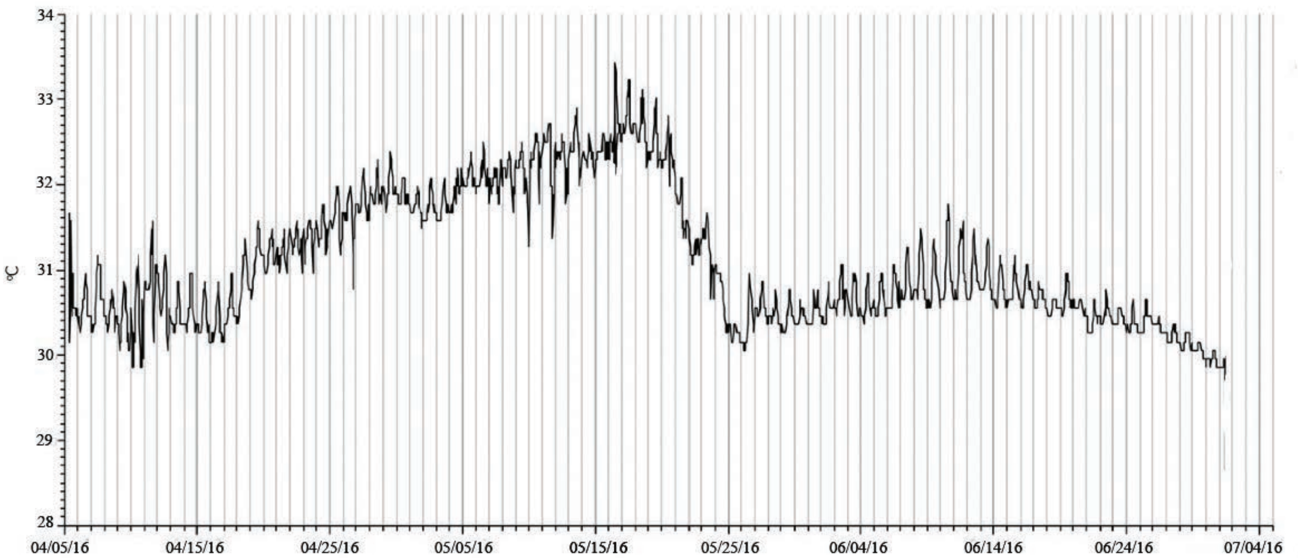

FIGURE 3: *Seawater temperatures recorded by a data logger at Ko Ngam Noi, Mu Ko Chumphon National Park, the Western Gulf of Thailand.*

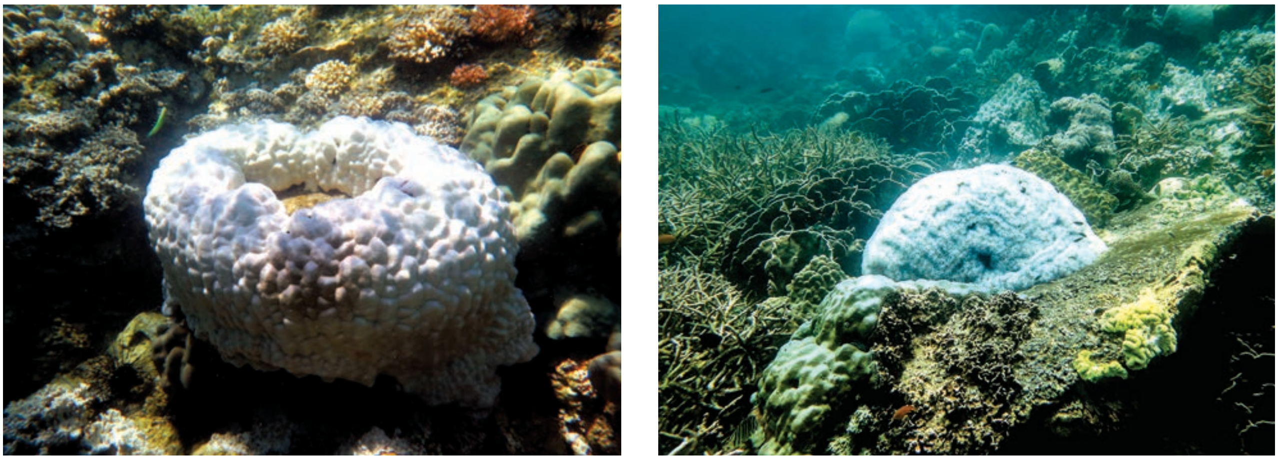

FIGURE 5: *The impacts of 2016 coral bleaching event at Ko Ngam Noi, Mu Ko Chumphon National Park, the Western Gulf of Thailand.*

The 2016 coral bleaching event was a good opportunity to conduct the first field experiment studying the impacts of shading on coral bleaching recovery and survival in Thai waters. The field shading experiments were carried out on coral communities of Ko Ngam Noi, Chumphon Province, in the Western Gulf of Thailand during the coral bleaching event (Figure 5). The experimental corals were shaded under mid-water floating shading nets. The light intensity levels and sedimentation rates under the shaded areas were lower than those in the unshaded areas. The recovery rates of some corals, such as *A. muricata, Fungia fungites, Lobophyllia hemprichii, P. decussata, Pocillopora* spp. and *P. lutea*, under the shaded areas, were higher. Mortality rates of *A. muricata* under the shaded areas after the bleaching event were also lower than unshaded colonies (Yeemin et al. 2016).

Drivers & Pressures

The key drivers of coral reef degradation in Thailand include national coastal development plans and policies, global climate change, and natural disturbances. In particular, the national economic and social development plans push for coastal development, fisheries, maritime activities, tourism, and industry development, resulting in pressures on coral reefs such as sediment, sewage, etc. The coral reefs in the Gulf of Thailand and the Andaman Sea have also been affected by natural and anthropogenic disturbances at different levels (Yeemin et al. 2012b, Phongsuwan et al. 2013).

The Indian Ocean tsunami hit the coastal provinces in the Andaman Sea in 2004. Subsequent surveys at 174 reef sites revealed that 13% of the monitored sites were severely damaged, 47% suffered low to moderate impacts, and 40% had no obvious effect from the tsunami (Phongsuwan et al. 2006, Satapoomin et al. 2006). The coral damage included overturned massive corals, broken branching corals, and corals smothered by sediments and coral rubble. The corals in shallow waters were more severely impacted. In general, the coral damage was mostly localized and affected only some reef zones in the direct path of the tsunami. Most damaged reef sites showed natural recovery a few years after the tsunami. Severe disturbances from natural events also include a storm in 1986 in the Andaman Sea (Phongsuwan 1991) and typhoon storms in 1989 and 1997 in the Gulf of Thailand (Sudara et al. 1992, Chansang et al. 1999). A large-scale population outbreak of crown-of-thorns starfish, *Acanthaster planci,* has occurred in the Andaman Sea during the last four decades (Chansang et al. 1986) and a sporadic outbreak at some other reef sites during the previous two decades (Phongsuwan et al. 2013). A high density of gastropods *Drupella* spp. was observed only on some reef sites in the Gulf of Thailand. However, this coral predator's impacts were localized and only in small sectors of the reef sites (Hoeksema et al. 2013).

The severity of coral bleaching events differed among reef sites and years. However, the overall impacts on coral reefs were very high. In the Andaman Sea, coral bleaching impacts may be categorized as mild in 1998, moderate in 1991, 1995, 2003, and severe in 2010 (Phongsuwan and Chansang 2000, Yeemin et al. 2010, Phongsuwan and Chansang 2012). Coral bleaching events in the Gulf of Thailand were severe in 1998 and 2010

(Sutthacheep et al. 2013a). The 2010 coral bleaching event was the most extensive event ever observed in the Andaman Sea. Sea surface temperature anomalies of between 30°C and 34°C were recorded during March–June 2010 in the Andaman Sea and the Gulf of Thailand. The collaboration between the Department of Marine and Coastal Resources and a network of Thai universities and NGOs conducted reef surveys that showed widespread and severe coral bleaching of over 80% at several reef sites. The coral bleaching in the Andaman Sea was more severe and extensive than in the Gulf of Thailand. The coral bleaching in the Inner Gulf of Thailand had the least impact. The most susceptible corals were *Acropora* spp. and *Pocillopora* spp. Coral mortality following the bleaching event was about 50–60% within the Andaman Sea and 30–40% within the Gulf of Thailand. Some coral species, especially *Porites,* showed good recovery. It is estimated that the 2010 bleaching event is similar in extent but with greater severity than the 1998 bleaching event within the Gulf of Thailand but was greater in extent and severity within the Andaman Sea. Coral reefs on the eastern coast of the islands in the Andaman Sea are subjected to internal waves and usually stronger wave action, and thus had a much higher impact than coral reefs on the western coast. Some corals, including *Galaxea* sp., *Diploastrea heliopora, Heliopora coerulea,* and free-living fungiids, showed high resistance to bleaching. Some soft corals, zoanthids, and giant clams were also bleached (Yeemin et al. 2012a, Phongsuwan et al. 2013, Sutthacheep et al. 2013a, 2013b).

The coral communities around Ko Khang Khao in the Inner Gulf of Thailand were affected by the low salinity and associated environmental factors during the heavy flooding event in 2011, which caused mass coral bleaching around the island. Over 90% of *Pocillopora damicornis* group colonies that survived the 2010 bleaching event were bleached in 2011. Other impacted corals were *Acropora* spp. and *Porites* spp. The flooding in 2011 also caused mass mortality of the sea urchin *Diadema setosum*, while some bivalves and sea cucumbers were also impacted (Pengsakun et al. 2012, Pengsakun et al. 2019).

Direct anthropogenic disturbances are clearly observed at some reef sites where tourism infrastructure has been highly developed, leading to high sedimentation and nutrient loads along the shoreline (Reopanichkul et al. 2010). Dredging and jetty constructions near coral reefs are also a major cause of coral reef degradation at several sites. The densities of juvenile corals were relatively low at Mu Ko Samui in the Western Gulf of Thailand. They suffered high mortality rates because of high sediment deposition and macroalgal overgrowth. Coral recovery is a difficult task requiring an ecosystem approach involving several management strategies and measures, including appropriate coastal development, enhancing coastal wetlands, and effective fishery management (Yeemin et al. 2013c).

Tourism and fishery activities can cause coral reef degradation at some locations. The impacts of tourism on coral reef ecosystems include anchor damage and pollution from tourist boats, trampling on coral colonies and inexperienced divers, and collecting reef animals, particularly herbivorous fish (Yeemin et al. 2006, Worachananant et al. 2008). Coral disease prevalence increased 3-fold at dive sites with high use compared to sites with fewer divers. Coral white syndromes are strongly related to sediment accumulation on coral tissues (Lamb et al. 2014, Samsuvan et al. 2019). The ecological impacts of common marine fishing gears used in Thailand have previously been reported. Abandoned fishing gear is a major component of marine debris on coral reefs in Thai waters. Such derelict and lost fishing gear is mostly composed of fishing nets, ropes, and lines made of plastics, and their impacts on coral reefs include tissue loss and fragmentation of reef corals (Suebpala et al. 2017, Ballesteros et al. 2018).

RECOMMENDATION

Based on the previous national report on coastal and marine resources and coastal erosion in Thailand (DMCR 2019), there are several recommendations on the management and conservation of Thailand's reefs, and they include:

1. Marine spatial planning and MPAs
    - Establish marine protected areas with effective management.
    - Support marine spatial planning and coral reef zoning for regulated activities.
    - Improve laws and regulations concerning coral reef management.
    - Promote the effectiveness of law enforcement for coral reefs and related ecosystems.
2. Tourism
    - Install and maintain mooring buoys at several reef sites.
    - Improve wastewater treatments and reduce marine debris from the tourism and fishery sectors.
    - Prevent boat anchoring and coral damages caused by tourist activities, such as trampling, touching, kicking, unexperienced diving, etc.
    - Strengthen regulations for coral reef tourism.
    - Adopt best practices for tourism business operators, boat owners, tour guides, and tourists.
3. Coral reef restoration and research
    - Enhance coral reef restoration and artificial reefs for diving.
    - Enhance stakeholders' participation in wastewater and solid waste management, particularly from households, hotels, restaurants, and local communities.
    - Encourage coral reef research and monitoring programs, particularly environmental economics, coral reef status, biodiversity, ecological processes, assessment of ecological restoration, and artificial reefs.
    - Establish new science, technology, and innovation for coral reef conservation and management.
4. Outreach, training, and capacity building
    - Promote coral reef outreach and education programs.
    - Encourage the use of social media for public awareness and education on coral reef ecosystems.
    - Promote training, meetings, seminars, and workshops for coral reef conservation and management.
    - Encourage capacity building for coral reef conservation and management.
    - Encourage integrated coral reef management with public participation and communication.

DATA CONTRIBUTORS

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SOUTHEAST ASIA

Vietnam

Introduction

Vietnam's coastline spans 14 degrees of latitude, extending for some 3,260 km with more than 3,000 inshore islands and islets. These habitats have provided favorable conditions for coral reef formation and development (Vo and Hodgson 1997, Vo et al. 2005). Coral reefs are important habitats in Vietnam's coastal waters, serving as reservoirs of biodiversity, providing goods and services for fisheries and tourist development in the coastal zone. Recent publications have reported 462 coral species (Vo 2014, Huang et al. 2015).

Recent economic development has resulted in several environmental concerns in coral reef management, including habitat loss and degradation, overfishing, and pollution (Vo et al. 2005, Nguyen and Vo 2013). In addition, coral bleaching has become more prevalent in recent years, considering the severe events recorded in 1998 at Con Dao islands and in 2010 at Phu Quoc islands. Coral bleaching was also observed in other areas such as Nha Trang Bay and Ninh Hai waters in 2010, and at Phu Quoc islands in 2013 and 2014, albeit with less impact (Vo 2000, Nguyen and Vo 2014).

As part of the management process, reef monitoring is used as the primary tool for understanding changes in coral reef communities under natural and anthropogenic impacts, and providing appropriate recommendations for coral reef management, especially in several marine protected areas (MPAs). The data presented in this report were collated from published and empirical data within the framework of different projects. In the period between 2001 and 2017, a total of 651 transects were surveyed, spreading across 335 sites at six major reef areas in the coastal waters of Vietnam, including Cu Lao Cham MPA (Quang Nam Province), Van Phong Bay and Nha Trang Bay MPA (Khanh Hoa Province), Nui Chua MPA (Ninh Thuan Province), Con Dao MPA (Ba Ria-Vung Tau Province) and Phu Quoc MPA (Kien Giang Province) (Figure 1). The numbers of transects, sites, and monitoring period at each location are presented in Table 1.

The method and parameters used in the monitoring program followed the Reef Check method (Hodgson et al. 2006) with some additional indicators suitable for local coral reef conditions. Monitored parameters included the benthic cover of hard coral (HC) and soft coral (SC) genera, recently killed corals (RKC), dead coral with algae (DCA), turf algae (TA), coralline algae (CA), macroalgae (SW), nutrient indicator algae (NIA), sponges (SP), rock (RC), rubble (RB), sand (SD), silts/clay (SI), others (OT) and bleaching corals (BL). Also recorded were target fish families and Reef Check's fish indicators (Chaetodontidae, Haemulidae, Lutjanidae, Serranidae, Labridae,

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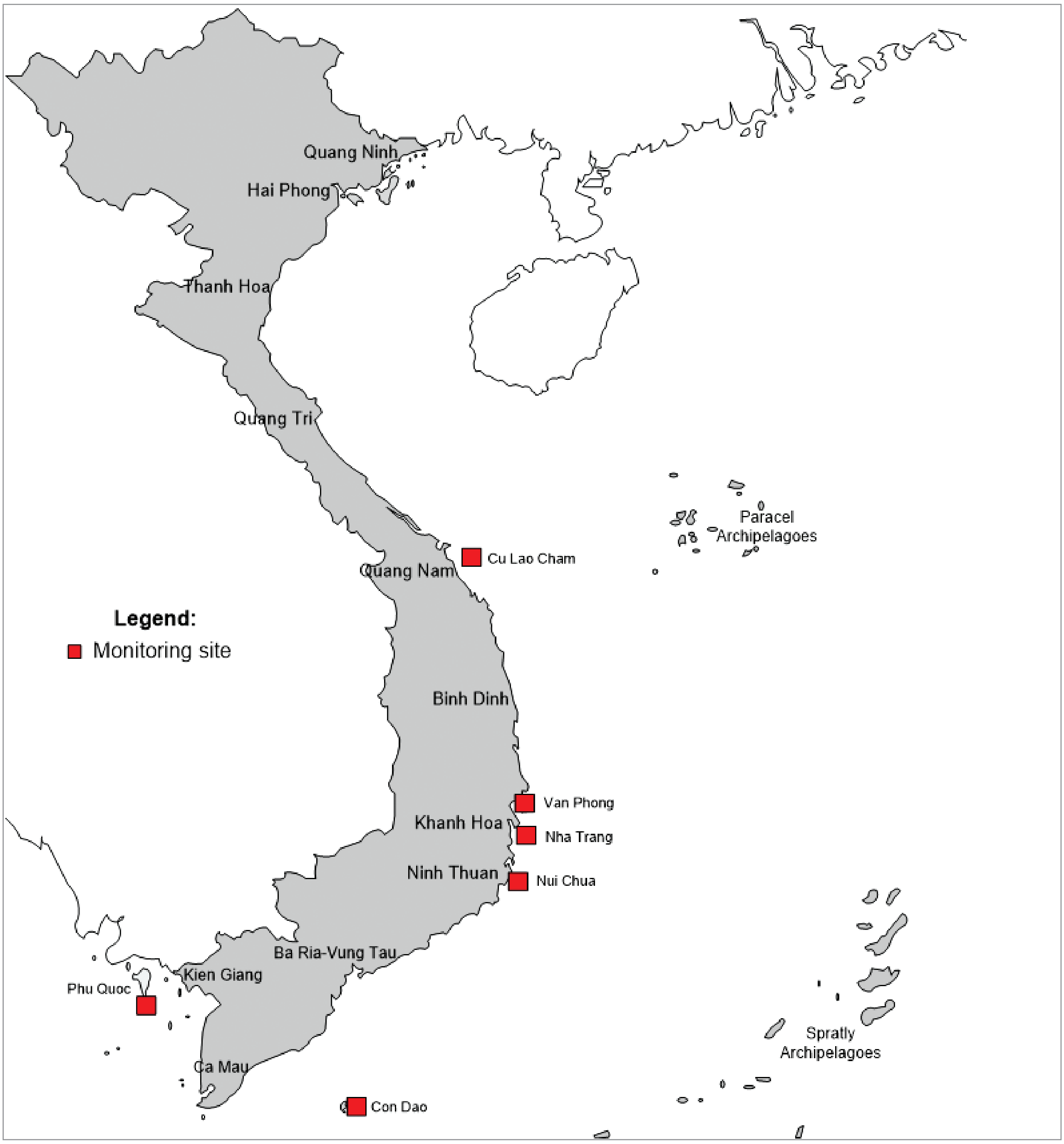

FIGURE 1: *Map showing monitoring areas in the coastal waters of Vietnam from 2001 to 2017.*

Scaridae, Muraenidae), and Reef Check's benthic macro-invertebrates (triton shell *Charonia tritonis*, giant clams *Tridacna* spp., Lobsters *Panulirus* spp., banded coral shrimp *Stenopus hispidus*, sea cucumbers, long-spined black sea urchin *Diadema* spp., pencil urchin *Heterocentrotus mammilatus*, and crown-of-thorns starfish *Acanthaster planci*). Ambient variables, such as physical parameters of impacts, were also monitored. However, this report only describes the status and trends of changes in coral cover.

TABLE 1: *Numbers of transects, sites, and monitoring period for each monitoring area.*

NO.	NAME OF AREA	NO. OF TRANSECTS	NO. OF SITES	TIME
1	Cu Lao Cham MPA	54	30	2004–2016
2	Van Phong Bay	62	31	2003–2017
3	Nha Trang Bay MPA	141	72	2003–2017
4	Nui Chua MPA	191	99	2005–2017
5	Con Dao MPA	72	37	2001–2017
6	Phu Quoc MPA	131	68	2003–2017
	Total	**651**	**335**	

Status & Trends

Analysis of data collected from 105 transects at 54 sites in the six major reef monitoring areas between 2015 and 2017 shows that coral reefs in the coastal waters of Vietnam had a mean live coral cover of 33.3 ± 18.7% (range 3.1–86.9%), in which hard coral cover was 29.2 ± 20.3% and soft coral cover was only 4.1 ± 8.6%. Among them, the reefs in fair condition (26–50% cover) occupied 50% of the total, 35.2% were in poor condition (<25% cover), 14.8% in good condition (51–75 % cover), while no reefs were in excellent condition (>75% cover).

Between 2001 and 2017, mean live coral cover at the six major reef monitoring areas in Vietnam's coastal waters had fluctuated annually (Figure 2a). Mean hard coral cover was stable from 2001 to 2005 (23.1–28.8%), increased to 33.3% in 2010, notably declined to 27.5% in 2011 after mass bleaching occurred in 2010, and remained at 27.6% in 2015. Following this decline, the mean cover of hard corals increased to 32% in 2016 and 34.2% in 2017. Although the mean cover of live corals had fluctuated due to several bleaching events in recent years, there are signs of recovery in the coastal waters of Vietnam. Comparison between depth shows that the mean cover of hard corals decreased considerably with increasing depth, from 42.4% at a depth of ≤2 m to 37.5% at 4 m, 26.7% at 5 m, and 19.2% at ≥8 m (Figure 2b).

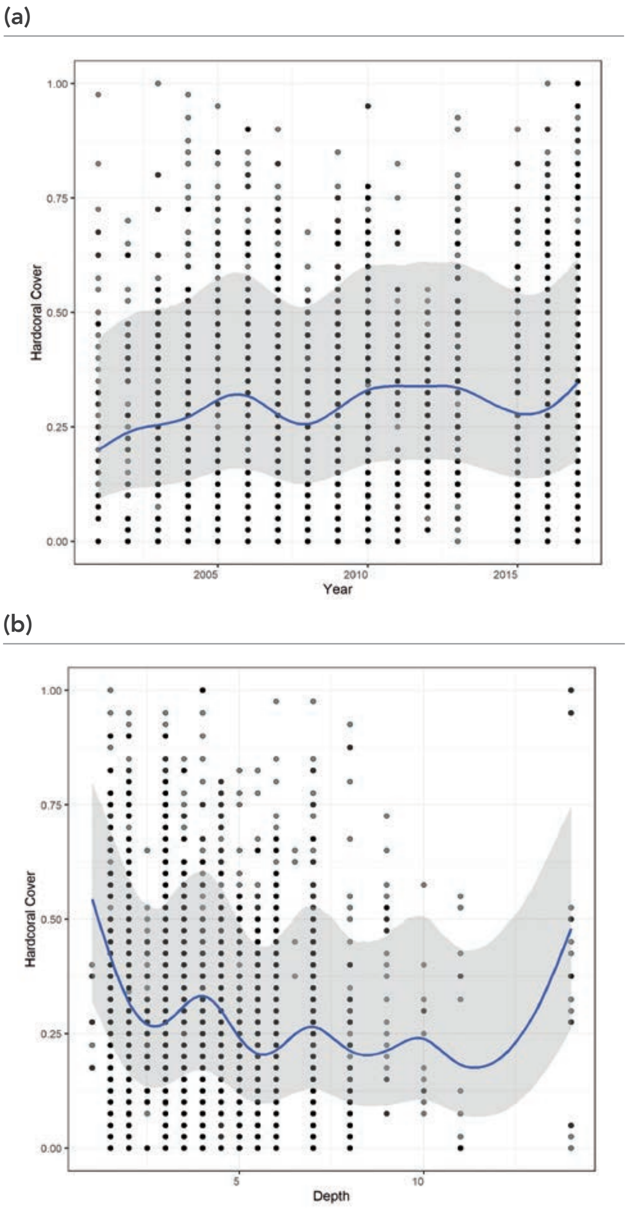

FIGURE 2: *Vietnam coral cover across (a) years and (b) depths.*

Coral Bleaching 2015–2016

Based on unpublished data surveyed in 2015–2016, bleaching events occurred from May to July in 2015 and 2016, with different impacts on the coral reefs in Vietnam's coastal waters.

a) In 2015, coral bleaching occurred at a few shallow sites (2–7 m in depth) in some areas with an extremely low mean cover and ratio of slightly bleached corals recorded in South-Central Vietnam, e.g., Van Phong Bay (1.8 ± 7.1% and 3.7 ± 11.3%, respectively), Nha Trang Bay (0.2 ± 1.0% and 1.0 ± 4.9%), and Cam Ranh Bay (0.6 ± 1.2% and 2.5 ± 4.6%).

b) In 2016, the bleaching caused more severe impacts to reef corals in Phu Quoc up to a depth of 10 m, where the mean cover and ratio of bleached corals was 20.2 ± 16.6% and 28.2 ± 12.2%, respectively, and this was much higher than in other areas located in South-Central Vietnam including Van Phong Bay and Nha Trang Bay (no bleaching), Nui Chua (4.8 ± 12.5% and 5.6 ± 12.6%), and in Con Dao, South-East Vietnam (3.8 ± 5.4% and 8.8 ± 8.1%).

Drivers & Pressures

Overfishing is considered a significant pressure on Vietnam's coral reefs. Monitoring data from most of the areas indicate that the densities of larger fishes belonging to the target groups (groupers, snappers, emperors, sweetlips, jacks and trevallies, triggerfishes, rabbitfishes, etc.) have become very rare and have not recovered after more than ten years of establishment and management as marine protected areas such as at Cu Lao Cham between 2004 and 2016 (Nguyen 2017), Nha Trang Bay between 2002 and 2015 (Nguyen 2015) and Phu Quoc between 2006 and 2019 (Nguyen 2019). In addition, a notable decline in the mean density of sea urchins (*Diadema* spp.) in Phu Quoc MPA from 2006 to 2019 indicates that this resource was over-harvested due to a rapid increase in the number of tourists and their consumption demand (Figure 3) (Nguyen et al. 2019).

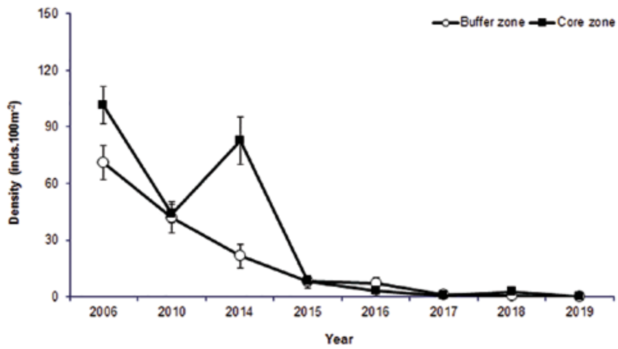

FIGURE 3: *Decline in density of sea urchins in Phu Quoc MPA from 2006 to 2019.*

Bleaching events have become more frequent and intensive in recent years, severely impacting coral reefs in Vietnam. The bleaching events were mainly recorded during May–June in 1998 (Vo 2000), 2010 (Nguyen et al. 2011), 2015, and 2016 (Thai et al. 2016, Vo, S. T. *unpublished data*) (Figure 4a).

Outbreaks of crown-of-thorns (COTS) were commonly recorded in several areas such as Nha Trang Bay in 2002 (Vo et al. 2004) and 2015 (Nguyen 2015), Tho Chu islands in 2013 (Vo 2013), and Van Phong Bay and Cam Ranh Bay in 2020 (Nguyen, V. L. *unpublished data*) (Figure 4).

Marine tourism has played an essential role in the economic development of many coastal cities and provinces in Vietnam during the last decade. However, a rapid increase in visitor numbers to some MPAs has adversely impacted Vietnam's coral reefs, especially at some attractive sites in the MPAs' core zone. From 2011 to 2015, the number of scuba divers on coral reefs ranged between 1,425 and 2,864 divers/year (mean: 2,463 divers/year) in Cu Lao Cham MPA (Nguyen 2017) and 28,802–23,100 divers/year (mean: 25,359 divers/year) in Nha Trang Bay (Nguyen, 2015). The number of divers calculated in Phu Quoc MPA in 2018 was about 25,304 divers/year (Nguyen, 2019). The mean numbers of divers in Nha Trang Bay and Phu Quoc MPAs were about five times higher compared to the sustainable numbers recommended by UNEP (4,000–6,000 divers/year for a destination diving site) or the average number of divers visiting the Sipadan MPA, Malaysia (about 3,650 divers/year) (Musa 2002).

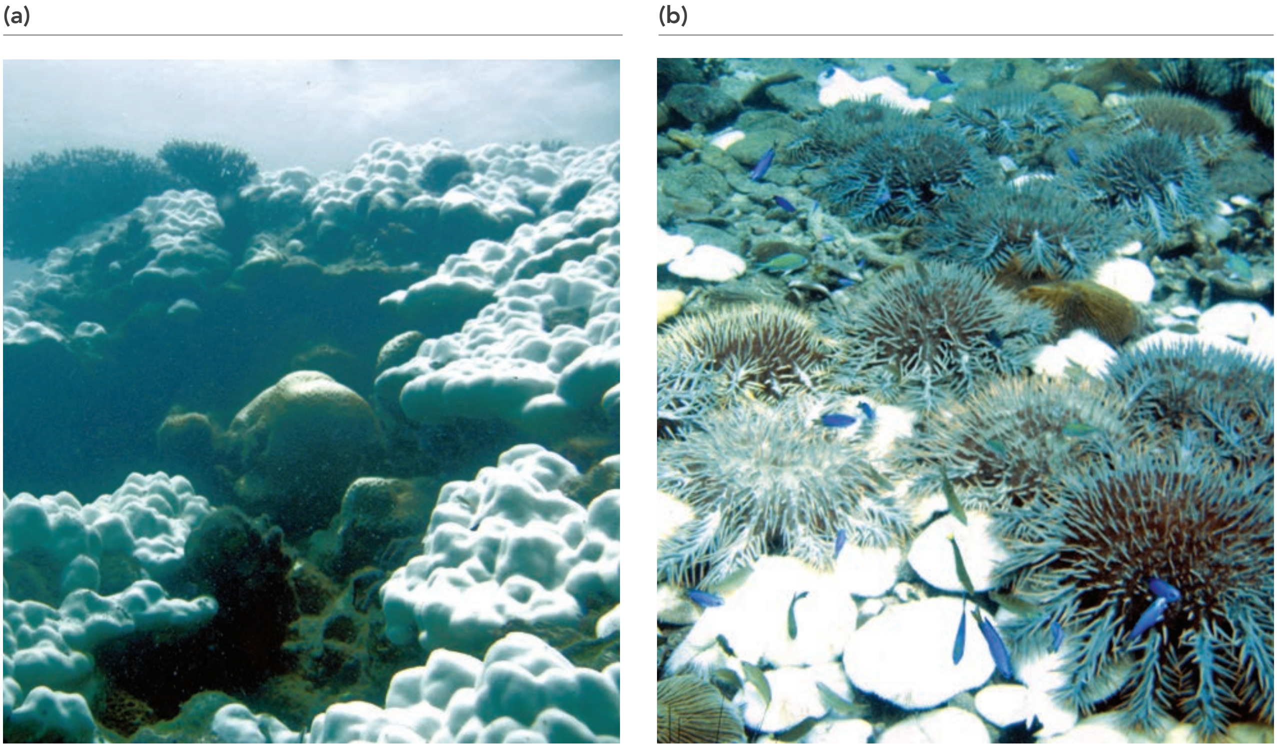

FIGURE 4: *(a) Mass bleaching in Con Dao in 2016 (Photo by Phan Kim Hoang); and (b) an outbreak of COTS in the Tho Chu islands in 2013 (Photo by Nguyen Van Long).*

In recent years, coastal development, including infrastructure development for marine tourism and transportation, has reduced the area of coral reefs in several locations. In Nha Trang Bay MPA, its coral reef area declined from 871.5 ha in 2002 to 754.1 ha in 2015 (Nguyen and Tong 2017) (Figure 5). However, a decline of less than 1 ha was recorded in Cu Lao Cham MPA between 2004 and 2016 (Nguyen 2017) and was stable in Phu Quoc MPA between 2005 and 2018 (Nguyen 2019).

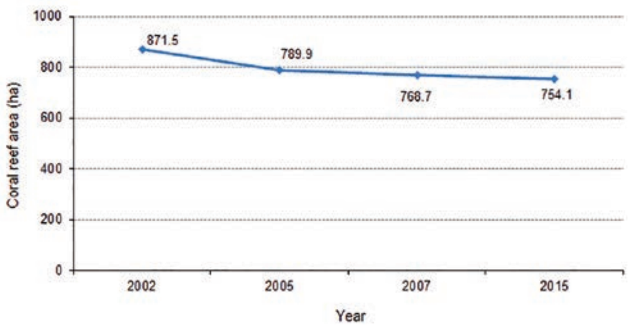

FIGURE 5: *A notable decline in the area of coral reefs in Nha Trang Bay between 2002 and 2015.*

Coral damage from natural events has not been studied during the last decades. However, low salinity integrated with high temperature severely impacted coral reefs at Con Dao in October 2005 (Hoang et al. 2008), and typhoons caused reef damage at Cu Lao Cham in 2006 (Nguyen et al. 2008) as well as at Nha Trang Bay in 2017 (*unpublished data*).

Land-based sediments have caused significant impacts on coral reefs in several areas in Vietnam's coastal waters due to upland silt run-off, sediment load of terrestrial mining, and other domestic and industrial discharges (Nguyen and Vo 2014). In 2015, analysis of semi-quantitative data from sediment deposits on the coral reefs in Khanh Hoa province, South-Central Vietnam, indicated that several reefs in the inner Van Phong Bay (Hon O, Bai Tre, W Ran Trao) and the nearshore waters of the Nha Trang Bay (Bai Tien, NE Hon Mieu, Mia Resort) had suffered sedimentation (Vo et al. 2019).

RECOMMENDATION

Coral reefs are critical habitats in most of the 16 existing and proposed MPA networks in Vietnam approved by the Prime Minister under Decision No. 742/QĐ-TTg dated on 26 May 2010, with some MPAs located in the upwelling area in South-Central Vietnam, including Nha Trang Bay, Nui Chua, Hon Cau, and Phu Quy. These sites may be more resilient against the increase of sea surface temperature in the summer, especially Nui Chua, compared to the other MPAs (Nguyen et al. 2013). Thus, these MPAs in South-Central Vietnam should be prioritized as the most important areas for developing and managing Vietnam's MPA networks. Up to the present, there were two MPAs established in this area, including Nui Chua and Hon Cau, and there will be one more established in 2021 (Phu Quy islands). However, scientific-based management for MPA networks and management capacity, including manpower and financial support invested for these MPAs, have been lacking.

Maintaining long-term monitoring activities should be a key priority and requirement, both now and in the future, especially at the MPA sites including Co To-Dao Tran, Cat Ba, Con Co, Cu Lao Cham, Ly Son, Van Phong Bay, Nha Trang Bay, Nui Chua, Hon Cau, Con Dao, and Phu Quoc. In addition, training MPA staff to monitor coral reefs should be continued for some recently established MPAs (Co To-Dao Tran, Con Co, Hon Cau) to enhance monitoring capabilities for the future.

DATA CONTRIBUTORS

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