GCRMN
Global Coral Reef Monitoring Network

Australian Government

Australian Institute of Marine Science

ICRI
International Coral Reef Initiative

STATUS OF CORAL REEFS OF THE WORLD

2025



EDITED BY

González-Rivero · Dallison
Wicquart · Brigdale · Logan
Fobert · Staub · Planes

Gouvernement Princier
Principauté de Monaco

Australian Aid

BLOOMBERG PHILANTHROPIES OCEAN INITIATIVE

MSCFOUNDATION

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Front cover. Victor Huertas (John Brewer Reef, Great Barrier Reef) | **Page 5** and **Page 273**. Hans Sjoeholm (*Naso hexacanthus* on a Jeddah Reef, Red Sea, Saudi Arabia) | **Page 8** and **Page 232**. Joshua Rambahiniarison (Mitsamiouli, Comoros) | **Page 9** and **Page 30**. Irene Pancrazi (Ari Atoll, Maldives) | **Page 11** and **Page 260**. Rebekka Pentti (Sir Bu Nair Island, UAE) | **Page 12**. Muhammad Ramadhany, SHAMS (Prince Mohammed bin Salman Royal Reserve, Red Sea, Saudi Arabia) | **Page 14** and **Page 257**. Julien Wickel (*Acanthurus triostegus* over a coral reef, Mayotte) | **Page 20** and **Page 221**. Naseef Villan, SHAMS (Gulf of Aqaba, Saudi Arabia) | **Page 46**. Nick Thake (Rowley Shoals, Australia).

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Jérémy Wicquart

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2021 2030 United Nations Decade of Ocean Science for Sustainable Development



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BLOOMBERG
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OCEAN INITIATIVE

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The **MSC Foundation** is the philanthropic arm of the MSC Group, advancing environmental, humanitarian and development initiatives worldwide by harnessing the Group's global reach, unique knowledge of the sea and logistical expertise. Through science-based partnerships in conservation, marine science education and public engagement, the Foundation helps build the knowledge, collaboration and action needed to restore marine ecosystems and strengthen the resilience of coastal communities.

Australian Aid

Australia proudly supports global partnerships to protect coral reefs and the ocean, reflecting Australia's commitment to safeguarding the marine environments that underpin communities, livelihoods, biodiversity and global climate stability for future generations.



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ACRONYMS

INSTITUTIONS

AIMS	Australian Institute of Marine Science
GCRMN	Global Coral Reef Monitoring Network
ICRI	International Coral Reef Initiative
ICRS	International Coral Reef Society
NOAA	National Oceanic and Atmospheric Administration
ROPME	Regional Organization for the Protection of the Marine Environment

TERMINOLOGY

CCA	Crustose coralline algae
CoTS	Crown-of-thorns starfish
DHW	Degree heating week
ENSO	El Niño Southern Oscillation
EEZ	Exclusive economic zone
FAIR	Findable, accessible, interoperable, reusable
IBTrACS	International best track archive for climate stewardship
MPA	Marine protected area
SST	Sea surface temperature



FOREWORD

For millions of people, especially in Small Island Developing States and coastal communities, coral reefs are more than ecosystems. They are home. They sustain an extraordinary diversity of life while feeding families, supporting livelihoods, protecting coastlines, and carrying the knowledge and traditions of generations. Yet they are changing at a pace and scale unprecedented in recorded history. Their decline is not a distant environmental concern — it is one of the defining challenges of our time.

This report from the Global Coral Reef Monitoring Network provides a rigorous assessment of the condition of the world's coral reefs at a pivotal moment in their history. It reflects decades of scientific collaboration and the dedication of researchers, Indigenous Peoples, practitioners, and local communities whose stewardship and knowledge have shaped our understanding of coral reefs. More than a scientific assessment, this report is a call to action.

The findings are clear. Coral reefs are under unprecedented pressure. Rising ocean temperatures, ocean acidification, pollution, and unsustainable human activities are driving increasingly frequent mass bleaching events, while the window for recovery continues to narrow. The greatest threat to coral reefs is no longer a lack of knowledge. It is our failure to act with the urgency the science demands.

This is not only an environmental crisis. It is a question of justice.

Those who depend most on healthy coral reefs have contributed least to the climate crisis that now threatens them. Yet Small Island Developing States, Indigenous Peoples, and coastal communities are not waiting to be rescued. They have long served as stewards of these ecosystems, drawing on generations of knowledge, experience, and innovation to protect them. Their leadership reminds us that recovery is possible when pressures are reduced and communities are empowered to lead.

We know what works. Local conservation efforts, effective marine management, and the integration of Traditional Knowledge with scientific innovation have already shown positive results in many parts of the world and offer powerful solutions. The challenge before us is no longer identifying what needs to be done, but finding the collective will to do it.

We must also be honest about the limits of local action. No community, no matter how committed, can protect coral reefs from unchecked global warming on its own. The ocean has never recognized the borders we draw on maps. Its currents connect our futures whether we acknowledge them or not. The survival of coral reefs ultimately depends on our collective ability to address the fundamental drivers of climate change. Ambitious and sustained reductions in greenhouse gas emissions remain essential. Without them, gains made at the local level will be increasingly difficult to sustain.

That is why protecting coral reefs is not simply a matter of conservation. It is a matter of ocean justice. It requires the international community to match the leadership already being shown on the frontlines — with ambitious climate action, sustained invest-

ment, and genuine partnership that empowers communities to shape their own future.

This report therefore speaks far beyond the scientific community. It calls on governments, international institutions, the private sector, philanthropy, and civil society to turn commitments into measurable action. Supporting those on the frontlines is not charity. It is an investment in our shared future.

Coral reefs are living expressions of the ocean's abundance. Their vibrant colors, extraordinary diversity, and quiet resilience have inspired generations while sustaining millions of people. They remind us that our greatest strength lies not in standing apart, but in recognizing how deeply connected we are — to one another and to the ocean that sustains us all.

We stand at a pivotal moment. Let this report be more than a record of decline. Let it be the turning point that transforms knowledge into action and promises into progress. The choices we make today will determine not only the future of coral reefs, but also the future of the people, cultures, and communities whose lives are intertwined with them. Let us ensure that future generations inherit reefs alive with color, abundance, and possibility — not by chance, but through our stewardship.



H.E. Ilana V. Seid

Palau Ambassador to the United Nations



EXECUTIVE SUMMARY

KEY FINDINGS

- **25.8%** global average hard coral cover in 2024.
- Overall, a **9.5% relative decline** in hard coral cover has been observed when comparing modern averages (2020–2024) with the reference period (1980–2009).
- **Four global coral bleaching events** recorded since 1998, each associated with a **6.5–9.9% relative loss** of global hard coral cover.
- **8.9% decline between 2023 and 2024** — the largest single-year loss on record, with the fourth global bleaching event not yet fully captured.
- **Recovery intervals are shrinking** from approximately 10 years to 5–6 years in many regions, with **no return to reference-period cover at any point since 2010**.
- **6% relative recovery** in global hard coral cover following the third global bleaching event, demonstrating coral reefs can recover when disturbance frequency eases.
- Projected frequency of global bleaching events suggests global hard coral cover will continue to decline.
- While the fourth global bleaching event was the largest and most intense event on record, the full ecological impacts of this event are not yet reflected in this assessment.
- **44.1% relative increase** in average global macroalgal cover relative to the 1980–2009 reference period, which nonetheless remains low at 6.8% globally.
- **21.1 million observations** analysed from **87,299 surveys**, across **36,886 reef sites**, representing the largest coral reef assessment to date.

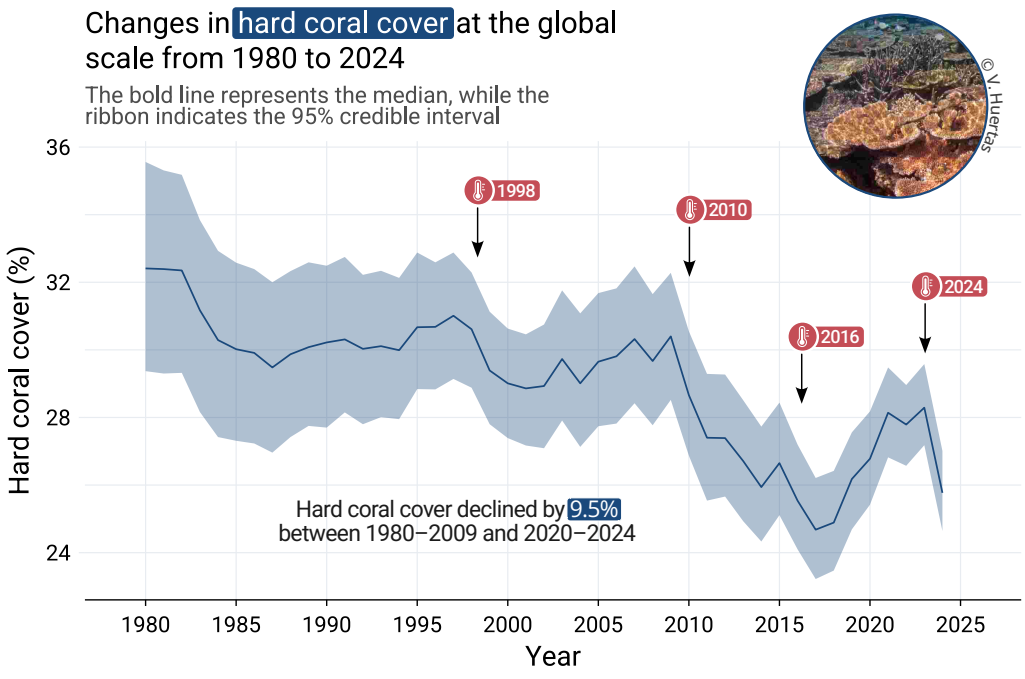
OVERVIEW

Coral reefs worldwide are in decline, and the interval in which they recover between disturbances is shortening. Based on 21.1 million observations collected through 87,299 surveys across 36,886 reef sites between 1980 and 2024, the *Status of Coral Reefs of the World: 2025* describes multi-decadal trends in hard coral and macroalgal cover across the world's shallow-water, warm-water coral reefs, with detailed analyses for each of the ten regions of the Global Coral Reef Monitoring Network (GCRMN). The *Status of Coral Reefs of the World: 2025* — the seventh edition of the GCRMN's flagship assessment — is the most comprehensive assessment of coral reef condition and trends ever produced by the Network. Synthesising global and regional monitoring data, the report provides an unparalleled evidence base to inform coral reef conservation and policy worldwide.

The report draws on data from 600 contributing partners across more than 120 countries, territories, and economies; roughly three times the data available in 2020. Regional chapters were written by data contributors and regional experts, and the global report was externally peer-reviewed by the International Coral Reef Society.

Trends in hard coral and macroalgal cover were estimated using Bayesian hierarchical models, with regional estimates weighted by coral reef area to derive global averages. Change is expressed by comparing 2020–2024 with the reference period of relative stability (1980–2009). Estimates are medians with 95% credible intervals ($CrI_{95\%}$), and confidence that change occurred is expressed as evidence strength derived from Bayesian exceedance probabilities.

Part 1 presents the global synthesis. Part 2 assesses each of the 10 GCRMN regions. The Conclusion considers future trajectories and implications, including an invited section on the fourth global bleaching event.

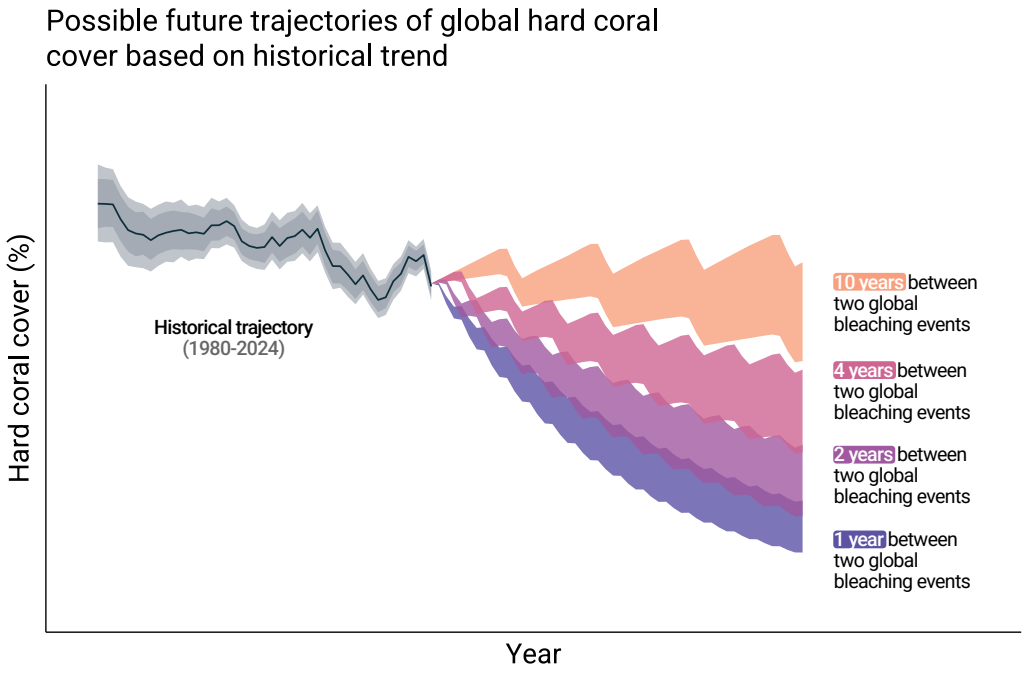


GLOBAL HARD CORAL COVER HAS DECLINED BY APPROXIMATELY 9.5%

Hard coral cover averaged 30.2% across the 1980–2009 reference period and 27.3% across 2020–2024, a relative decline of 9.5% (CrI $_{95\%}$: 4.0–14.5%). Between 2023 and 2024 alone, cover fell by a relative 8.9% (CrI $_{95\%}$: 6.4–11.3%), the largest single-year decline on record, resulting in a single-year estimate for coral cover of 25.8% (CrI $_{95\%}$: 24.6–27.0%) for 2024. At this level of coral cover, many reef systems would still be expected to deliver key ecological services, including carbonate production, biodiversity, fisheries, coastal protection, and cultural value, though this is not the case across all regions. At local scales, a loss of this size is often recoverable; detected globally, it indicates a systemic shift across large spatial extents that is far harder to reverse.

CORAL DECLINE IS STEPWISE, ACCELERATING AFTER 2010 AND DRIVEN BY INTENSIFYING MARINE HEATWAVES

Four periods of steep decline coincide with the four global bleaching events between 1998 and 2024, each associated with a relative loss of 6.5–9.9%. Since 2010, these events have recurred every four to six years. Satellite records show sea surface temperatures over coral reefs rose 0.81 °C between 1985 and 2025, amplified periodically by the El Niño Southern Oscillation (ENSO). Coupling this temperature data with the coral reef data in this report, declines in hard coral cover clearly coincide with peaks in the proportion of reefs exposed to heat stress during each bleaching event, indicating that the primary driver of coral decline is marine heatwaves, the frequency and intensity of which are increasing under climate change.



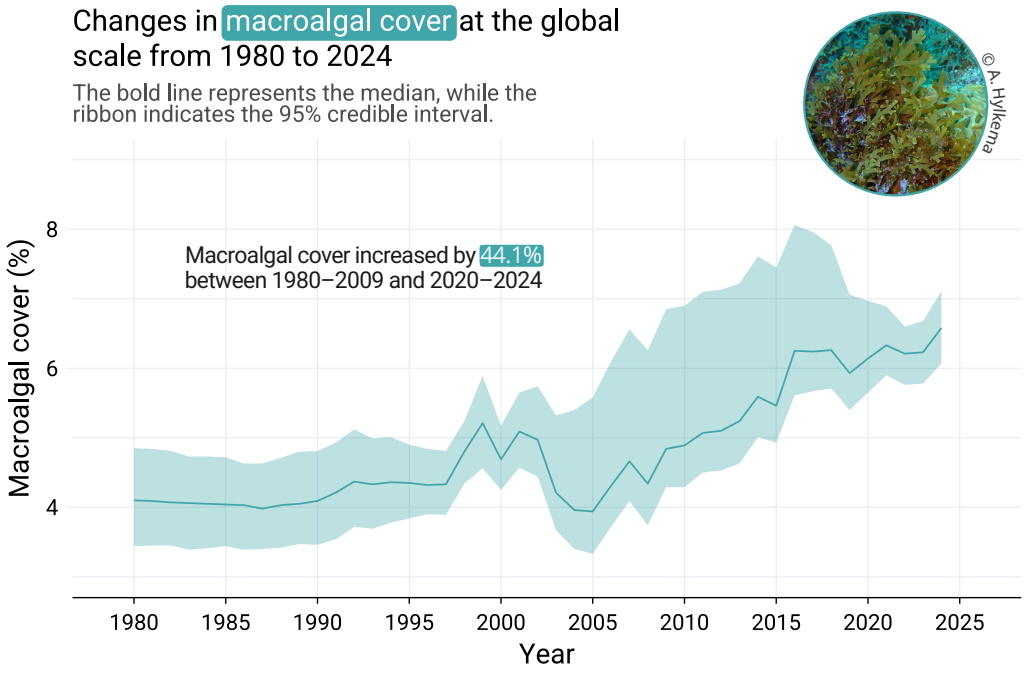
RECOVERY IS POSSIBLE BUT CURTAILED BY INCREASING FREQUENCY OF LARGE-SCALE DISTURBANCES

Coral reefs continue to demonstrate the capacity to recover following major disturbances. This pattern is particularly evident after the third global bleaching event, when global average hard coral cover increased by 6% between 2017 and 2019 (CrI $_{95\%}$: 2.1–10.0%), relative to post-disturbance levels. While this pattern of increase did not fully restore hard coral cover to historical reference levels, it demonstrates that coral reefs can regain hard coral cover following disturbances given sufficient time. However, recovery intervals have shortened dramatically, from approximately one major disturbance per decade to major disturbances every four to six years in many regions after 2010. Marine heatwaves, compounded by local human and environmental pressures, are pushing coral reefs towards a sustained global decline. As recovery windows shrink, continued reef degradation is likely unless urgent climate mitigation and local management action occur.

Simulations projecting future trends with varying recovery times suggest an increase in the frequency of major disturbances — exceeding decadal events (*i.e.* more than one event every 10 years) — will lead to a continual decline in global coral cover. When the frequency of major disturbances exceeds historical patterns, the decline in coral cover is expected to be more severe than currently documented in this report.

MACROALGAL COVER HAS RISEN STEADILY BUT REMAINS LOW GLOBALLY

Globally, average macroalgal cover on coral reefs has increased steadily by 44.1% since 1980 (CrI $_{95\%}$: 28.0–59.0%), with much of that increase occurring after 2010. At 6.8% (CrI $_{95\%}$: 6.1–7.1%) in 2024, this trend does not indicate a global shift toward algal dominance.

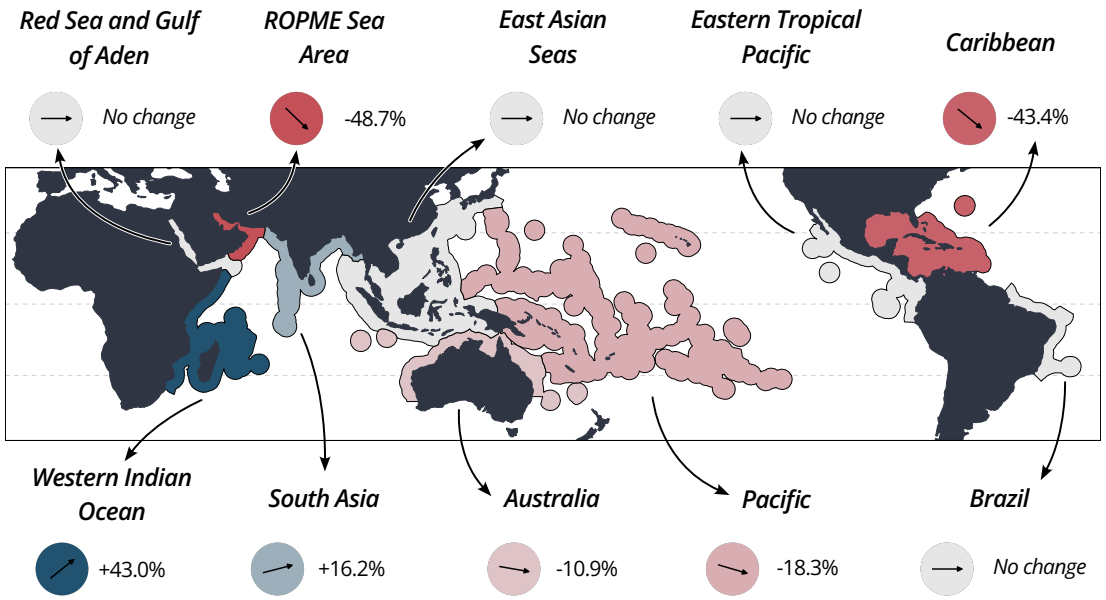


However, it remains concerning as increases in macroalgal cover at regional and local scales can hinder reef recovery. Macroalgal dominance is well documented in the Caribbean, and nearshore reefs that are subject to nutrient enrichment and herbivore loss remain at risk.

CORAL REEFS ARE NOT CHANGING UNIFORMLY

Global hard coral cover trends mask regional differences. Regional trajectories demonstrate that coral reef decline is not uniform and that recovery remains possible under favourable conditions. Four regions have shown strong evidence of decline (Caribbean, ROPME Sea Area, Australia, and the Pacific). Four regions demonstrated fluctuations in hard coral cover over time and have not shown strong evidence of long-term changes relative to the reference period (Brazil, East Asian Seas, Eastern Tropical Pacific, and Red Sea and Gulf of Aden). In contrast, two regions demonstrated a strong recovery from already-degraded reference states (South Asia and the Western Indian Ocean). This divergence of regional trajectories is related to the compounding effects of local and regional-scale environmental and anthropogenic stressors — including acute events such as increased levels of coral predation (*e.g.* by crown-of-thorns starfish), tropical cyclones, and the often-chronic pressures from pollution, coastal development, and unsustainable marine resource use — particularly outside of global bleaching years. These diverging trajectories demonstrate that while multi-decadal decline is observed at global scales, coral reef trajectories are highly region-specific, and highlight the need to prioritise threat reduction at every scale.

Comparison of relative changes in hard coral cover between 1980-2009 to 2020-2024 across the ten GCRMN regions



CONFIDENCE AND DATA LIMITATIONS

The decline in hard coral cover reported here is statistically robust but conservative. Data are scarce in the early years, limiting reference-period accuracy, and the reference period almost certainly already includes degraded reefs. The true long-term loss for regions with limited data available during the reference period is likely steeper than the data suggest. Throughout the report, the confidence in the changes is presented by assessing the strength of the evidence within a Bayesian inference framework.

The fourth and most intense global bleaching event concluded in 2025. This bleaching event began in 2023 and reached unprecedented levels of heat stress in 2024 (see Box 2 for details). However, the full impact of this event has not been captured here as data in this report only extend to 2024.

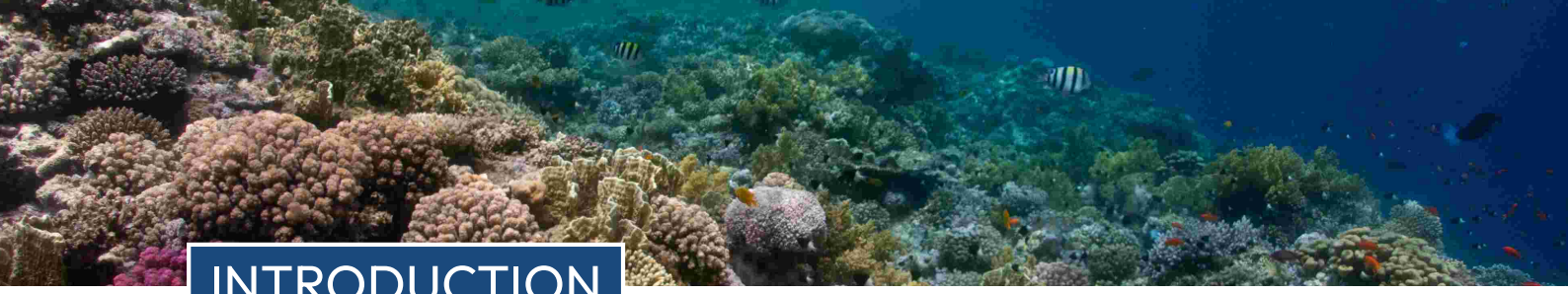
IMPLICATIONS

Coral reefs have not crossed a point of no return. They retain the potential to recover where disturbance intervals allow, and whether that potential is realised depends principally on climate change trajectories and their impact on ocean warming trends. Reducing local pressures cannot substitute for climate mitigation, but reefs under lower local pressure resist and recover from heat stress more effectively.

Standardised coral reef monitoring and reporting efforts have significantly improved the collective understanding of coral reef trends, but further expansion — especially in high-priority, vulnerable regions and using a broader set of indicators — is essential to guide more effective conservation. Six priorities emerge for strengthening coral reef monitoring on which all of this depends:

1. **Resourcing**. Increase funding for monitoring, particularly for developing nations;
2. **Coverage**. Extend monitoring in time and space, prioritising regions that hold a large share of the world's reefs but remain understudied;
3. **Standardisation**. Converge on common methods so data are comparable across programmes;
4. **Data access**. Improve data discoverability and interoperability while recognising data ownership under FAIR principles;
5. **Indicators**. Raise taxonomic resolution and broaden indicators beyond hard coral cover;
6. **Integration**. Combine ecological with socio-economic indicators to support whole-system interpretation.

The evidence presented in this report shows that coral reefs remain under unprecedented pressure, but also that recovery is still possible. The future condition of the world's coral reefs will depend on the actions taken during the coming decade to limit climate change, reduce local pressures, and sustain long-term monitoring and management.



INTRODUCTION

WHY CORAL REEFS MATTER

Shallow water coral reefs are amongst the most biologically diverse and socio-economically valuable ecosystems on Earth. They support high biodiversity, underpin the well-being of millions of people, and provide essential services to national economies, coastal communities and global biodiversity [1]. Yet these ecosystems are experiencing profound and accelerating change. The combination of global disturbances, chronic local pressures, and increasing human dependency [2] has made coral reefs one of the most threatened ecosystems of the Anthropocene [3, 4]. Tracking their status and trends over the long term is therefore critical for guiding effective management and protection [5].

Although coral reefs occupy less than 0.2% of the ocean floor, they support over 30% of all known marine species [6, 7]. Their complex physical structures create habitats, feeding grounds, and nursery areas for a vast array of organisms and connect with other coastal systems, such as seagrass meadows and mangrove forests. Together, these systems sustain the ecological processes that underpin productivity, connectivity, and resilience across tropical seascapes.

ECOSYSTEM SERVICES AND HUMAN DEPENDENCE

Coral reefs also provide substantial benefits to people. Close to one billion people are now estimated to live within 100 km of coral reefs, equivalent to 13% of the global population [2]. Coral reefs are present in over 100 countries and territories, of which over 54 are considered Small Island Developing States (SIDS) [2], with millions of livelihoods worldwide depending on healthy and productive coral reefs [5, 8, 9].

The value of goods and services provided by coral reefs has been estimated at up to USD 9.9 trillion per year [10], and the contributions of coral reefs span a wide range of critical services, including:

- **Human health and well-being**. 50–90% of the animal protein in the diets of Pacific Islanders comes from reef-associated fisheries [11].
- **Shoreline protection**. The complex three-dimensional coral reef architecture can dissipate wave energy by up to 97% [12], reducing the risk of storm surge, coastal flooding and erosion, and protecting an estimated 5.3 million people and USD 109 billion in GDP globally from storm impacts over decadal periods [13].
- **Food security and livelihoods**. Coral reef fisheries support as many as six million people [14] and are worth over USD 6 billion per year, providing an average annual seafood yield of 1.42 million tonnes [15, 16].
- **Tourism**. Coral reef tourism contributes an estimated USD 36 billion to the global tourism industry annually [17].

Coral reefs also hold value that cannot be captured in goods and services, including significant ecological, cultural, and histori-

cal value. This intrinsic worth is, in some respects, more valuable than an economic measure:

- As biodiversity hotspots, coral reefs have substantial ecological value and are critical for supporting ocean life [18]. Coral reef ecosystems contain over 800 species of reef-building corals [19], creating habitats that harbour an estimated 32% of all named marine species, excluding microbes and fungi [20]. For some marine taxa, the proportion is even higher: over 37% of the ~16,800 named marine fish species [21] are associated with tropical coral reef environments [22]. Furthermore, recent estimates suggest more than 90% of coral reef species remain unnamed, and the total number of coral reef-supported species likely exceeds 800,000 [20].
- Coral reefs hold deep cultural, spiritual, and historical significance, and value for many Indigenous Peoples and local communities [23, 24]. Coral reef ecosystems are integral to their identity, social cohesion, and customary practices, whilst supporting education and scientific discovery and elevating global heritage [25].

The economic value of the Great Barrier Reef

In 2025, Deloitte Access Economics and the Great Barrier Reef Foundation assessed the Great Barrier Reef's contribution to the Australian economy, *At what cost? Safeguarding the Great Barrier Reef's role in Australia's economy* [26], updating its 2017 valuation. The assessment reported:

- **AUD 9 billion** contributed to the Australian economy in 2023–2024, up from AUD 6.4 billion reported in 2017, generated directly through Reef-related businesses and indirectly through their supply chains.
- **AUD 7.9 billion annually from tourism**, the Reef's largest economic contributor at nearly 90% of its annual economic value. Over 30% of all leisure tourism spending in Queensland is linked to the Great Barrier Reef region.
- **77,000 full-time-equivalent jobs** supported by the Great Barrier Reef, up from 64,000 reported in 2017, which would make the Reef equivalent to Australia's fifth-largest employer if it were a single company.
- **AUD 95 billion total asset value**, spanning the Reef's economic, social and "icon" (cultural and identity) value, including both use and non-use value — up 69% from AUD 56 billion in 2017.
- **AUD 124 billion** in future economic opportunity for the region over the next 50 years (to 2075) — but only if action is taken. This value comprises AUD 110 billion from global climate action to limit warming below 2 °C and AUD 14 billion from investment in the Reef's adaptation and resilience.

However, heat stress and the increasing frequency of extreme events remain the greatest threat to the Great Barrier Reef's health, compounded by poor water quality from land-based run-off and crown-of-thorns starfish outbreaks, placing its value to the national economy at risk.

THREATS TO CORAL REEFS

Coral reefs face a suite of interacting pressures that vary in extent, intensity, and duration, threatening the ecosystem services they provide. Among these, anomalous summer marine heatwaves — characterised by sea surface temperatures (SST) exceeding the 90th percentile of the average for at least five consecutive days [27] — are well-documented as a dominant global driver of coral decline [28–31] across local, national, regional, and global scales [32, 33].

The potential risk to corals during these events is a function of both the intensity and duration of heat stress. When thermal thresholds are exceeded, corals typically undergo bleaching — a stress response in which they expel their colourful symbiotic algae (zooxanthellae) living in their tissues, causing the living tissue to appear white. Following this stress, coral decline generally occurs via two distinct pathways. First, acute, extreme heat stress can trigger immediate coral mortality, causing the animal to die without ever visibly bleaching. Alternatively, severely bleached corals may die from starvation due to the prolonged loss of their photosynthetic symbiotic algae. If thermal conditions return to normal before starvation occurs, surviving corals can slowly reacquire their zooxanthellae and recover their normal colouration [32, 34]. However, this is physiologically demanding, and thus even surviving corals may suffer reduced growth and fecundity, and increased susceptibility to disease [35–37].

Mass coral bleaching events are relatively recent phenomena; the first mass bleaching event was documented in the 1980s, with widespread bleaching across the eastern Pacific [38–40]. In 1998, the first recorded pan-tropical, or "global", bleaching event was documented, affecting each of the three tropical ocean basins: Atlantic, Indian, and Pacific [41–43], and caused loss of hard coral cover worldwide [30]. Subsequent global bleaching events occurred in 2010 and during 2014–2017 [29, 31]. These global bleaching events have caused declines in hard coral cover at both global and regional scales [30, 44, 45].

From April 2023 to mid-2025, coral reefs were subjected to the fourth global coral bleaching event (GCBE-4; [46]), declared by the National Oceanic and Atmospheric Administration (NOAA) Coral Reef Watch (CRW) in April 2024, following a year of heat-stress accumulation in all three tropical ocean basins. As of December 2025, over 87% of the world's coral reefs had experienced bleaching-level heat stress [47].

Heat stress is not the only pressure facing coral reefs; local stressors compound these globally driven impacts, reducing reef resilience and impairing reef capacity to recover following disturbance. Nutrient run-off and sedimentation — driven by poor land and coastal management and coastal development — degrade water quality and promote algal growth, including both turf and macroalgae, with varying impacts across major coral reef regions [48–50]. Overfishing of herbivorous fishes and invertebrates reduces grazing levels on coral reefs, enabling macroalgae and turf algae to outcompete corals [51, 52]. Additionally, overexploitation of reef fishes, including the removal of large predators, alters the reef fish assemblage, reducing biodiversity and overall community resilience, and destructive fishing practices fragment the physical reef structure [53, 54]. However, where local stressors are effectively managed, reef-builders are conserved, resilience is increased, and coral reefs can recover [30].

CONSERVING CORAL REEFS FOR THE FUTURE

Despite their immense value and the services they provide, the future of coral reefs remains uncertain [55, 56]. Their continued

ability to support biodiversity, economies, and cultures depends on urgent global action to address changing ocean conditions, reduce local pressures, and strengthen long-term management — supported through continued spatial and temporal monitoring of reef status and trends [30, 44, 45].

Addressing ocean warming is recognised as the foremost priority [57]. Rising ocean temperatures and intensifying marine heatwaves are the primary global driver of coral decline [30, 58], a reality underscored by the fourth global bleaching event of 2023–2025, the most extensive and intense bleaching event on record [47]. While managing local stressors is critical for conserving healthy coral reefs, it is not a substitute for limiting ocean warming. The difference between warming scenarios is stark. Statistical downscaling modelling of thermal projections suggests that only 2–5% of reefs remain below severe-bleaching thresholds at 1.5 °C, and none at 2 °C [59], and that coral reefs are among the ecosystems most vulnerable to loss beyond warming of around 1.5 °C [60]. Therefore, holding warming as close as possible to the 1.5 °C target is the single most important action to enable the persistence of coral reefs through this century.

Reducing local pressures on coral reefs is a critical second-stage intervention to improve reef health and resilience. Where coral reefs face lower pressures from overfishing, pollution, nutrient run-off, sedimentation, and coastal development, they can withstand and recover from disturbance more effectively [61]. Therefore, local management practices, such as effective fisheries management, water-quality improvement, and well-enforced, well-resourced protected areas, can increase the resilience of coral reefs and their likelihood of surviving and recovering from extreme events.

Active intervention can complement these measures at local scales. For example, coral restoration can accelerate habitat recovery between disturbances [62], coupled with emerging approaches such as the use of coral probiotics [63, 64] and assisted evolution, which aim to enhance the thermal tolerance of corals themselves [65, 66]. However, restoration projects are limited in scale; the value of restoration and active intervention therefore lies in supporting coral recovery at priority sites, rather than as a substitute for action on the underlying drivers of coral decline [1, 62].

Sustained, standardised monitoring of coral reefs underpins all of the above management and conservation actions. Long-term spatial and temporal data on coral reef status and trends are critical for managers to detect change, target interventions, and evaluate the impact of actions [30], thereby facilitating prepared, anticipatory, and adaptive management rather than reactive management. Large-scale, coordinated assessments of long-term data, such as the *Status of Coral Reefs of the World: 2025* report, are therefore critical for detecting changes and adaptive management at the global scale. Sustained and increased investment in monitoring capacity — particularly in data-sparse regions — is essential for tracking progress towards conservation targets, detecting early warning signs of reef decline, and evaluating the effectiveness of conservation and restoration interventions.

The scientific evidence presented in this report can help inform international policy commitments and conservation action. Nationally Determined Contributions (NDCs), National Adaptation Plans (NAPs), and National Biodiversity Strategies and Action Plans (NBSAPs) are identified as mechanisms to support local biodiversity protection and contribute to climate mitigation [18]. Area-based management, through effective marine protected areas (MPAs), Other Effective Area-Based Conservation Measures (OECMs) and Locally Managed Marine Areas (LMMAs), is central to doubling the area of coral reefs under ef-

fective protection by 2030, a Coral Reef Breakthrough target that contributes to Target 3 (30×30) of the Convention on Biological Diversity's (CBD) Kunming-Montreal Global Biodiversity Framework (KM-GBF). Well-designed protected areas are a recognised priority for meeting global biodiversity goals, from preventing extinctions to retaining the most intact ecosystems [67].

This report, alongside a wider body of research, underscores that coral reefs remain gravely threatened by climate change compounded by local stressors. Yet the data also demonstrate that coral reefs, given appropriate time and conditions, have the potential to recover, and their collapse is not inevitable. The status of coral reef tipping points is uncertain, and reefs do not respond in a binary way, offering local, national, regional, and global opportunities for action. The evidence presented in this report demonstrates both the scale of current challenges and the opportunities that remain for conservation and management action. The coming decades will be critical in determining the future condition of coral reef ecosystems worldwide.

THE GLOBAL CORAL REEF MONITORING NETWORK (GCRMN)

Long-term monitoring is essential for detecting changes, interpreting drivers, and understanding the effectiveness of management interventions, policy decisions, and conservation measures [68, 69]. Delivering monitoring of this kind — sustained, standardised, and coordinated across regions and decades — is the purpose of the Global Coral Reef Monitoring Network (GCRMN).

For three decades, the GCRMN has provided the global evidence base needed to understand long-term trends in coral reef condition, cementing its status as the world's leading scientific network on the status and trends of coral reef ecosystems. Established in 1995 as an operational network of the International Coral Reef Initiative (ICRI), the GCRMN celebrated its 30th anniversary in 2025 — marking three decades of building the global scientific knowledge base needed to understand, protect, and manage coral reefs. The GCRMN operates as a global network of scientists, managers, and organisations committed to monitoring the condition of coral reefs worldwide. The network is hosted by the Australian Institute of Marine Science (AIMS). It operates through 10 regional nodes spanning all major coral reef regions, from the Caribbean and Brazil to the Pacific, the Indian Ocean, East Asia, and the Red Sea.

The mandate of the GCRMN is to 1) improve understanding of coral reef status and trends, globally and regionally; 2) analyse and communicate coral reef biophysical, social and economic trends, providing science-based recommendations in support of raising awareness, management, and policy development; 3) enable and facilitate greater utilisation of coral reef data, including in research; and 4) build human and technical capacity to collect, analyse and report biophysical and socio-economic data on coral reefs. In December 2018, an Implementation and Governance Plan was adopted to strengthen the GCRMN's ability to track and report on coral reef status and trends in a systematic, transparent and scientifically rigorous manner. The plan defines the network's governance structure, the roles of regional nodes and the GCRMN Steering Committee, and the principles that guide data collection, sharing, and reporting. It provides the operational framework within which this report has been produced.

The flagship products of the GCRMN are its global *Status of Coral Reefs of the World* reports, first published in 1998. These reports draw on raw monitoring data contributed by GCRMN members from all coral reef re-

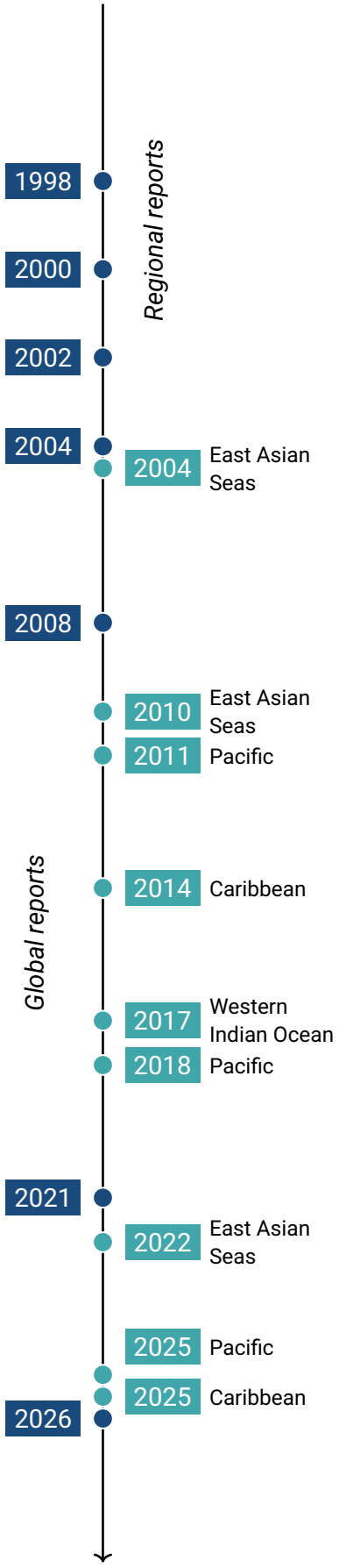
gions, synthesised into quantitative assessments of coral reef health over time. The role of regional nodes in coordinating data contribution, analysis, and reporting is central to this process, and in many regions the GCRMN works in close collaboration with the UNEP Regional Seas Programme.

Beyond informing management at local and regional scales, coral reef monitoring also plays an increasingly important role in international environmental policy and reporting frameworks. As pressures continue to mount on coral reefs, the GCRMN plays a central role in ensuring that monitoring data are comparable, accessible, and fit for purpose to inform policy, support management decisions, and guide practitioners.

THE INTERNATIONAL POLICY CONTEXT FOR CORAL REEFS

As international ambition to address the interconnected climate, biodiversity, and sustainable development crises has grown, so too has the demand for reliable information on the condition and trends of coral reefs, which respond directly and measurably to environmental changes. Trends in coral reefs can serve as critical ecological indicators for understanding the effectiveness of international environmental agreements and regional conventions, in which governments are increasingly expected to demonstrate progress towards global commitments and ensure that management actions are evidence-based and effective.

The GCRMN provides this essential scientific foundation through long-term, standardised observational evidence that these frameworks need to track progress, establish baselines, and inform adaptive management at multiple scales.



Timeline of the publication of global (in blue) and regional (in green) GCRMN reports. ▶

The GCRMN has a clear role: "to analyse and communicate coral reef biophysical, social, and economic trends, providing science-based recommendations in support of raising awareness, management, and policy development (GCRMN Implementation and Governance Plan [70]; Goal 2)". Specifically, the GCRMN's collated data and analysed trends of the world's reefs directly contribute to the monitoring framework of the CBD Kunming-Montreal Global Biodiversity Framework (KM-GBF), particularly for Goal A and Headline Indicator A.1 (Red List of Ecosystems), namely through hard coral and macroalgal cover across the Network's 10 regions. Moreover, ICRI has identified 16 of the 23 framework targets that are highly relevant to coral reef ecosystems [71].

Within the UN Framework Convention on Climate Change (UNFCCC), coral reefs are recognised as one of the most temperature-sensitive ecosystems on Earth, serving as a "frontline early warning system" for climate impacts. The *Status of Coral Reefs of the World: 2025*, alongside previous global and regional reports, provides long-term observational and modelled evidence of the impacts of climate-induced heat stress and ocean warming. It documents coral reef responses to increasingly frequent and intense marine heatwaves, whose frequency, duration and magnitude are amplified by climate change, informing the impact of Article 2 under the Convention. Specifically, these observational trends support the ecosystem and food security dimensions of the UAE Framework for Global Climate Resilience (Article 7) and serve as a natural-science input to the Ocean and Climate Change Dialogue. By documenting these changes, the GCRMN enables Parties to effectively integrate reef-based adaptation and nature-based mitigation into their Nationally Determined Contributions and National Adaptation Plans. The Ocean Breakthroughs, launched under the Sharm el-Sheikh Adaptation Agenda in 2023, further establish global targets to accelerate action for ocean resilience, including the Coral Reef Breakthrough.

In 2025, the seventh United Nations Environment Assembly (UNEA) adopted Resolution 7/1: accelerating global actions to promote the climate resilience of coral reefs. This resolution specifically encourages Member States and stakeholders to contribute data and expertise to the GCRMN to facilitate evidence-based and informed decision-making towards global targets. Similarly, the IUCN Resolution WCC-2025-Res-037, adopted in response to the fourth global bleaching event, underscores the urgency of mitigating climate impacts and preventing catastrophic ecosystem collapse. The resolution calls for investment in long-term monitoring and data sharing through the GCRMN to ensure that the best available science underpins regular assessments of coral reef status.

The 2025 International Court of Justice (ICJ) Advisory Opinion on climate change further reinforced the importance of robust scientific evidence to demonstrate climate impacts and support accountability. For climate-sensitive ecosystems such as coral reefs, long-term observations provided through GCRMN are essential for documenting ecological change, establishing evidence of climate impacts, and informing effective adaptation and conservation responses.

To ensure the findings of the *Status of Coral Reefs of the World: 2025* provide timely evidence to guide management and policy decisions, without prescribing specific policy positions, the GCRMN established a Policy Task Force for the first time. The Task Force frames scientific findings for use across multilateral and regional policy contexts, such as the Convention on Biological Diversity and the UN Framework Convention on Climate Change. Through its global and regional summaries for policymakers, the GCRMN provides the best available science to underpin policy and conservation actions.

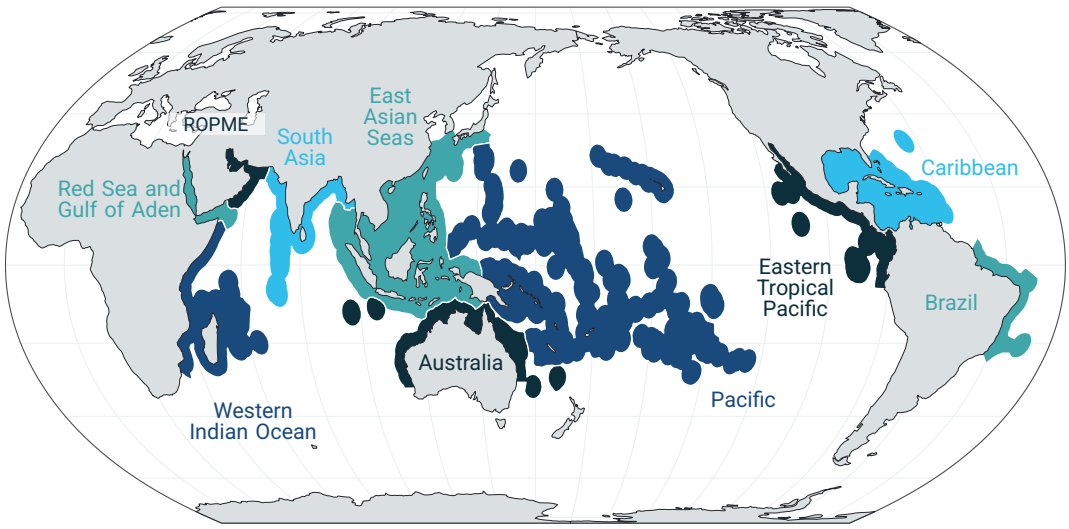
Beyond biodiversity and climate policy, the GCRMN supports a range of global commitments related to coral reef conservation, biodiversity, sustainable development, and ocean resilience. Coordinating global coral reef monitoring since 1995, GCRMN continues to provide standardised, long-term observations of coral reef conditions and trends, enabling progress to be assessed, priorities to be identified, and actions to be guided by robust scientific evidence. These data remain critical not only for the future of coral reef ecosystems, but also for the economies, food security, livelihoods, and well-being of the communities that depend on them. Science can identify the challenges, track progress, and guide solutions, but safeguarding coral reefs ultimately depends on the decisions, commitments, and actions taken at the global, regional, and national policy levels.

ABOUT THIS REPORT

This report is released at a pivotal moment. Coral reefs have just experienced the most severe and extensive bleaching event on record. The year 2025 also marked the halfway point towards the 2030 deadlines set by the Kunming-Montreal Global Biodiversity Framework and the Sustainable Development Goals (SDGs), a critical juncture at which the world must assess whether commitments made to nature are being translated into action fast enough to matter.

The *Status of Coral Reefs of the World: 2025* is the seventh edition in the GCRMN's global flagship series, and provides new results on the status and trends of the world's shallow-water (warm-water) coral reefs, synthesising the latest global and regional data from 1980 through 2024. It draws upon the networked capacity of the GCRMN's 10 regional nodes to deliver updated metrics, interpret drivers of change, and support evidence-based conservation.

The 10 GCRMN regional nodes. ROPME = ROPME Sea Area. ▼



This report provides the GCRMN's most complete scientific assessment to date of the status and trends of coral reefs worldwide. It presents new evidence on where coral reefs stand today and how they have changed over time, offering a more representative assessment than the GCRMN's previous global analysis. It provides crucial evidence for assessing the efficacy of the ambitions and actions to which governments have committed. The evidence outlined in this report is clear: ocean temperatures continue to break records, bleaching events occur with increasing frequency and severity, and coral reefs have less time to recover between successive disturbances.

The *Status of Coral Reefs of the World: 2025* draws on a global database compiled from data contributed by scientists and managers across the GCRMN, including more than 21 million benthic observations from over 87,000 surveys at 36,000 sites in more than 120 countries, territories, and economies.

Data are represented across the Network's 10 regional nodes: Australia, Brazil, Caribbean, East Asian Seas, Eastern Tropical Pacific, Red Sea and Gulf of Aden, Pacific, ROPME Sea Area, South Asia, and Western Indian Ocean. Data in this report extend to 2024 and include new observations, monitoring programmes, and coral reefs not represented in previous GCRMN reports. This report examines the global patterns in hard coral and macroalgal cover, explores regional variations, and assesses the key drivers of change at both global and regional scales.

The *Status of Coral Reefs of the World: 2025* provides the scientific evidence base to underpin policy, management, and conservation decisions, equipping governments, policymakers, and practitioners with the knowledge needed to act. This report specifically seeks to:

1. Provide an updated global assessment of the status of the world's coral reefs following the previous report (*Status of Coral Reefs of the World: 2020*).

2. Use regionally consistent methodologies to present temporal trends and spatial differences in the status of coral reefs, and identify drivers of change to enable governments to benchmark status and prioritise conservation measures.

3. Interpret observed patterns in relation to heat stress, cyclones, and other environmental and anthropogenic disturbances at regional and global scales.

4. Provide greater granularity in reporting on algal cover trends, including macroalgal and turf algal cover.

5. Promote an inclusive and collaborative reporting process for the world's coral reefs that appropriately recognises and acknowledges contributors.

6. Support reporting and benchmarking against international biodiversity and climate frameworks, including the Kunming-Montreal Global Biodiversity Framework, the Paris Agreement, and Sustainable Development Goal 14 — Life Below Water.

Consistent with past GCRMN reports, the 2025 report is divided into two main parts:

- Part 1 — Global status of coral reefs delivers the global synthesis, corresponding to the synthesis of coral reef temporal trends at the global scale. This part is further divided into three sections: 1) Status and trends of coral reefs of the world, 2) Major drivers of change in global coral reef condition, and 3) Diverse trends in hard coral cover across regions.

- Part 2 — Regional status of coral reefs offers the regional syntheses, including summaries for each of the GCRMN's 10 regional nodes, developed by their respective coordinators, regional co-authors, and data contributors.

A conclusion summarises the report's critical findings, with an invited section focusing on the fourth global bleaching event, as reported by NOAA Coral Reef Watch, and reflecting on its implications for this report.

To make this report more transparent, citable, and usable, the GCRMN has taken three further steps. Regional chapters have been written by experts in each region, including those who collected and contributed the underlying data, resulting in an expansive co-authorship. Each regional chapter has been assigned a digital object identifier (DOI). Lastly, the global report has been externally peer reviewed by the International Coral Reef Society (ICRS).

What is new in this report?

Compared with the *Status of Coral Reefs of the World: 2020*, this report incorporates substantial advances in data coverage, analytical capacity, transparency, and regional participation. Key developments include:

- Approximately three times more monitoring observations than were available for the 2020 global assessment.
- Over 21 million benthic observations, from more than 87,000 surveys and 36,000 monitoring sites.
- Data extending to 2024 and an improved representation of historical monitoring data, enabling more robust assessment of long-term trends (Box 1).
- Expanded regional representation.
- Greater reporting on macroalgal and turf algal trends.
- Regional chapters with assigned DOIs to improve citation.
- Assessment contextualised by the fourth global bleaching event.
- Expanded authorship and external peer review by the International Coral Reef Society.

PART 1

Global status of coral reefs



STATUS AND TRENDS OF CORAL REEFS OF THE WORLD

Stepwise global decline in hard coral cover, 1980–2024

Global hard coral cover has declined by 9.5% between 1980 and 2024, driven by the increasing frequency and severity of global bleaching events — signalling a systemic, difficult-to-reverse degradation of coral reef ecosystems with major implications for the billions of people who depend on them.

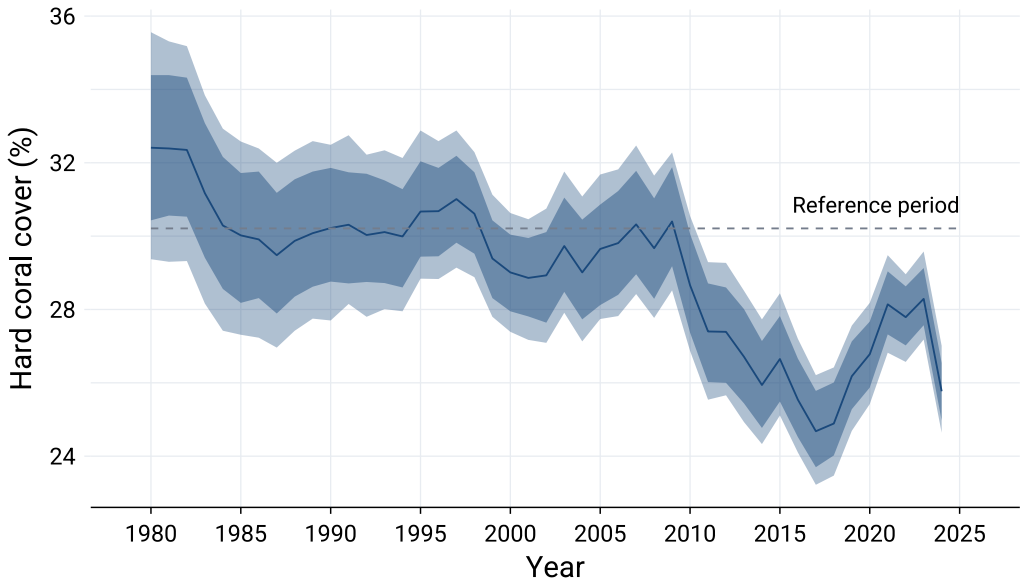
Globally, coral reefs have lost an average of 9.5% of their hard coral cover over the 1980–2024 reporting period (95% credible intervals [$CrI_{95\%}$]: 4.0–14.5%; Figure 1.1). This change is estimated by comparing the most recent hard coral cover estimates (2020–2024) with estimates from a reference period (1980–2009)[1] when hard coral cover was relatively stable.

This global decline in hard coral cover is both statistically robust and concerning as broad-scale losses outweigh local gains, resulting in a net decline at the global scale. At local scales, a 10% decline is often recoverable given sufficient time between disturbances (*e.g.* the Maldives [72], Great Barrier Reef [73], and Moorea [74]). However, when such changes are detected on a global scale, they indicate systemic shifts in the overall condition of coral reefs across large spatial scales, which are more difficult to reverse. Such global shifts in coral reef condition threaten the food security, livelihoods, and coastal protection that coral reefs provide to the one billion people living within 100 km of these critical ecosystems [2, 58].

The hard coral cover trajectory reveals a stepwise pattern of change, with a ratcheting-down pattern over the last four decades of monitoring. This pattern accelerates after 2010 (Figure 1.2), when coral reefs at regional and local scales were severely impacted by both more frequent and intense global bleaching events and local environmental and anthropogenic pressures, leading to a steeper decline in hard coral cover (see Part 2 - Regional status of coral reefs). At global scales, four distinct periods of steep declines in hard coral cover coincide with the last four global bleaching events, in the years when the largest proportion of coral reefs experienced severe heat stress (1998–1999, 2010–2011, 2016–2017, 2023–2024 [31, 47]; see the Marine heatwaves section for details). Each global bleaching event was associated with an estimated 6.5–9.9% relative loss of the average hard coral cover worldwide (Figure 1.2).

At the level of hard coral cover recorded in 2024 — 25.8% global average (median; $CrI_{95\%}$: 24.6–27.0%) — many reef systems would still be expected to provide key ecological services, such as carbonate production, biodiversity, fisheries, coastal protection, and cultural and spiritual value [47, 72]. However, this may not hold across all GCRMN regions. Global averages capture the overall status of coral reef ecosystems, but status and trend estimates at regional and local scales provide the finer resolution needed for conservation and management (see Part 2 - Regional status of coral reefs for detailed assessments across regions).

[1] 1980–2009 was chosen as a reference period to acknowledge the scarcity of data in the early years of the temporal series, disparity of data availability across regions, and the natural variability of hard coral cover over expected periods of decline and recovery. Therefore, this period is considered to cover a well-represented proportion of reefs monitored across the globe during a period of relative stability.

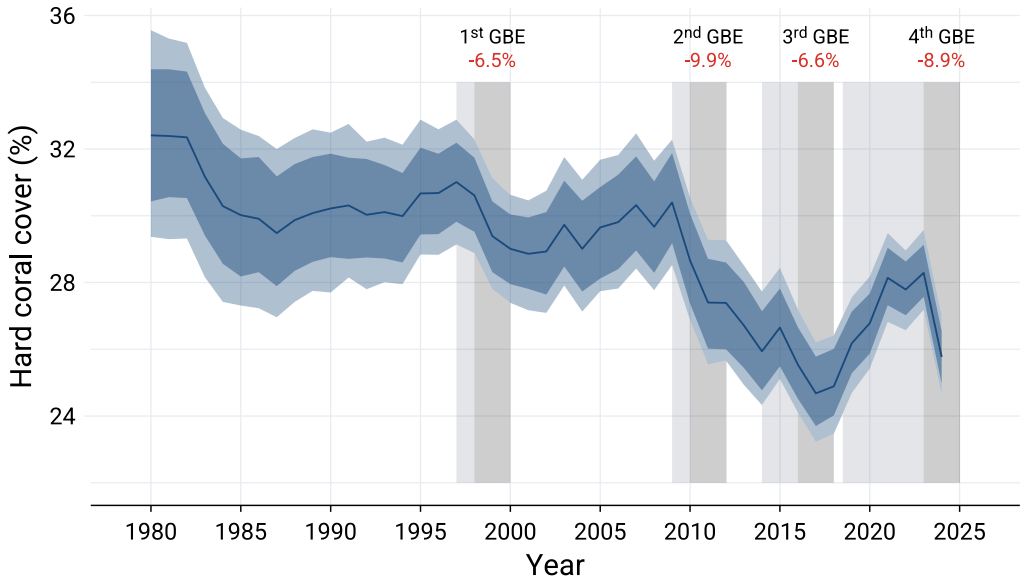


▲ **Figure 1.1.** Modelled temporal trends of hard coral cover at the global scale from 1980 to 2024. The solid line represents the estimated median hard coral cover, while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. The grey dashed line represents the median estimated hard coral cover for the reference period (1980–2009). Note: the y-axis is scaled to start at 22% rather than zero, to provide a more detailed visualization of the modelled trend line.

At global scales, major declines in hard coral cover coincide with the years when the highest proportion of coral reefs were exposed to thermal anomalies during the global bleaching events (Figure 1.2). However, local and regional disturbances also contributed to the observed long-term loss of hard coral cover, particularly in the periods between global bleaching events (see the Marine heatwaves section for details). To account for the fact that global bleaching events occur across austral and boreal summers and have a legacy impact on hard coral cover, the impact of each event was assessed by comparing hard coral cover immediately before the event with cover two years afterwards, as mortality and associated losses continue to accumulate in years following the initial heat stress (see Materials and Methods).

In 1998, the first global bleaching event exposed up to 2.5% of the world's coral reefs to heat stress severe enough to risk reef-wide bleaching and mortality (Degree Heating Weeks (DHW) $\geq$ 8 °C-weeks), leading to a 6.5% decline in hard coral cover relative to pre-bleaching levels (median, CrI $_{95\%}$: 1.5–11.2%; Figure 1.2). The second global bleaching event had the greatest impact on hard coral cover between 2010 and 2012, with 10.1% of reef area exposed to bleaching-level heat stress (DHW $\geq$ 8 °C-weeks). This event led to a 9.9% reduction in hard coral cover relative to pre-bleaching levels (median, CrI $_{95\%}$: 6.1–13.7%). Since 2010, global bleaching events have become more frequent, occurring every four to six years, narrowing the window for coral recovery between events. At the peak of the third global bleaching event, in 2016, 21.1% of coral reefs were exposed to bleaching and mortality-associated heat stress (DHW $\geq$ 8 °C-weeks), leading to a cumulative 6.6% decline in hard coral cover relative to pre-bleaching coral cover (median, CrI $_{95\%}$: 2.3–10.5%; Figure 1.2).

The fourth and most intense global bleaching event concluded in 2025 [47]. This bleaching event began in 2023 and reached unprecedented levels of heat stress in 2024 [47] (see Box 2 for details).



▲ **Figure 1.2.** Modelled temporal trend in hard coral cover at the global scale from 1980 to 2024 highlighting the global bleaching events. The solid line represents the estimated median hard coral cover, while darker and lighter blue ribbons represent 80% and 95% credible intervals, respectively. The y-axis is scaled to start at 22% rather than zero to provide a more detailed visualization of the modelled trend line. Grey shading depicts periods of global heat stress on coral reefs during the global bleaching events (GBE, [47]), with light grey shading indicating the beginning of global heat-stress periods and dark grey shading indicating the years during which the largest proportion of coral reefs experienced mortality-associated heat stress (see Figure 1.4). The relative changes in hard coral cover following these mortality-associated heat stress events are indicated above the grey shading.

However, the full impact of this event has not been captured here as data in this report only extend to 2024. Even so, a relative decline in hard coral cover of 8.9% was estimated between 2023 and 2024 (median, $CrI_{95\%}$: 6.4–11.3%; Figure 1.2) — the largest single-year decline in hard coral cover recorded to date. With 42.6% of the world's coral reefs exposed to heat stress severe enough to risk reef-wide bleaching and mortality (DHW $\geq$ 8 °C-weeks) in 2024, and 27.4% in 2025 [47], global average hard coral cover is likely to continue declining from 2025 onwards, suggesting the fourth global bleaching event is on track to be the most severe event recorded to date.

The stepwise decline in global hard coral cover since 2010 is a significant concern, as it may signal a progressive shift of coral reef ecosystems towards unstable and degraded states, rather than the natural cycles of decline and recovery seen in healthier systems [75].

While the documented decline in hard coral cover is robust, limitations in the underlying data mean the trend estimates reported in this assessment are likely conservative. Despite a three-fold increase in collated monitoring data compared with the previous assessment (GCRMN's *Status of Coral Reefs of the World: 2020* [30]), data remain scarce in the early years of the time series, as in previous global assessments. This data scarcity limits the accuracy of hard coral cover estimates for the reference period (1980–2009). Additionally, while the reference period (1980–2009) was intended to capture a long, relatively undisturbed baseline, it likely already includes reefs in a degraded state [58]. Taken together, these baseline limitations and data gaps mean the figures presented here should be understood as a floor, not a ceiling — the actual decline in hard coral cover is likely steeper than the current evidence can measure.

In the absence of major large-scale disturbances (*e.g.* global bleaching events) over periods of five to 10 years, hard coral cover has shown the capacity to partially recover on coral reefs globally. This is evident in the 6% relative increase in global average hard coral cover between 2017 and 2019 (median; CrI $_{95\%}$: 2.1–10.0%), following the third global bleaching event (Figure 1.2). However, since 2010, coral reefs globally have not fully recovered to historical values, when compared to the reference period (Figure 1.1). Furthermore, the fourth global bleaching event has since driven unprecedented heat stress on coral reefs, effectively reversing this partial recovery and returning global hard coral cover to a net loss.

Steady increase in macroalgal cover, 1980–2024

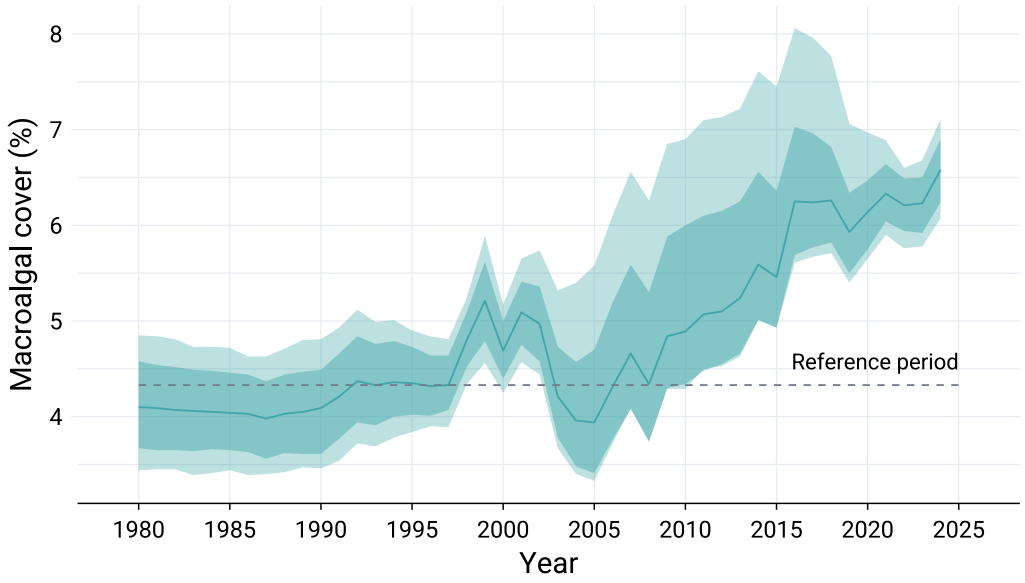
Globally, average macroalgal cover on coral reefs has increased steadily by ~44% since 1980, with much of that increase occurring after 2010. While absolute levels remain low, this trend is concerning because, at regional and local scales, increases in macroalgal cover can inhibit reef recovery.

Macroalgal cover has steadily increased over the past four decades (44.1% median increase between 1980–2009 and 2020–2024, CrI $_{95\%}$: 28.0–59.0%; Figure 1.3), with much of the rise occurring after 2010, when the largest global loss of hard coral cover was recorded (Figure 1.1). This change was estimated by comparing recent macroalgal cover estimates (2020–2024) with estimates from a reference period of relative stability (1980–2009).

Macroalgae are important components of coral reef ecosystems and provide critical ecological services (*e.g.* fish recruitment habitats, nutrient cycling, productivity) [76, 77]. At high cover, however, macroalgal cover can inhibit reef recovery following hard coral mortality by monopolising available space on a reef and inhibiting coral recruitment through a range of physical, chemical, and biological feedback mechanisms [78, 79].

At the global scale in 2024, macroalgal cover remained low at 6.8% (median, CrI $_{95\%}$: 6.1–7.1%) and does not signal irreversible changes to coral reef functioning [79–81]. At regional scales, however, the picture varies. Macroalgal dominance and its effects on coral reef resilience are well documented in some regions (*e.g.* the Caribbean) [76], whereas offshore reef systems elsewhere (*e.g.* the Pacific, Australia, and the East Asian Seas) show no comparable concerns. Within regions, nearshore coral reefs tend to support higher macroalgal cover [82, 83] as they are exposed to chronic drivers that favour macroalgal growth over hard corals, such as nutrient enrichment and overfishing of herbivores [82–84].

Thus, while global macroalgal cover remains low, regional and nearshore coral reefs, where chronic pressures persist, remain at risk of macroalgal dominance and its consequences for reef resilience at local and regional scales. Furthermore, the strong statistical evidence of a steadily increasing trend in macroalgal cover, detectable globally from 1980 to 2024 (Figure 1.3), is concerning, as it suggests potential future impacts should macroalgae continue to increase to levels that may compromise the resilience of coral reefs.



▲ **Figure 1.3.** Modelled temporal trends of macroalgal cover at global scales from 1980 to 2024. The solid line represents the estimated median macroalgal cover, while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. The grey dashed line represents the median estimated macroalgal cover for the reference period (1980–2009). Note: the y-axis is scaled to start at 3% rather than zero, to provide a more detailed visualization of the modelled trend line.

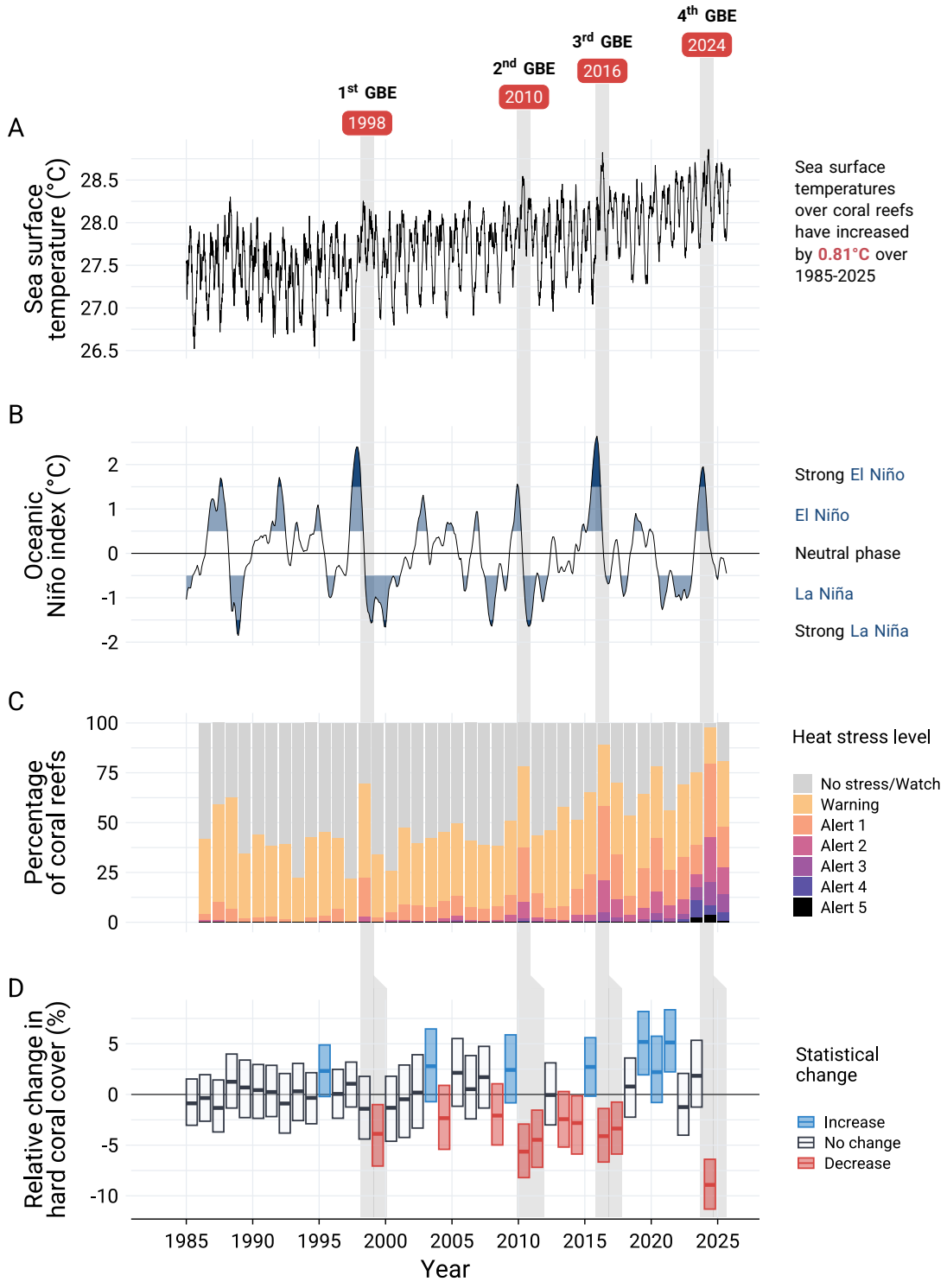
MAJOR DRIVERS OF CHANGE IN THE CONDITION OF CORAL REEFS OF THE WORLD

Marine heatwaves

Marine heatwaves are the primary driver of global hard coral cover loss, driven by rapid ocean warming and a sharp increase in the frequency and intensity of heat stress, which has triggered four global bleaching events in the past three decades.

Marine heatwaves remain the dominant driver of hard coral cover loss across reefs worldwide, with anthropogenic climate change leading to more frequent episodes of elevated ocean temperatures and a 20-fold increase in the frequency of marine heatwaves [85]. This has resulted in four global bleaching events in the past three decades, with a marked increase in their frequency, duration, and magnitude since 2010 [47].

Coral reefs are particularly vulnerable to heat stress, especially during summer marine heatwaves [86]. Sea surface temperature over coral reefs worldwide rose by 0.81 °C between 1985 and 2025 (Figure 1.4A). Marine heatwaves are now the single strongest driver of coral cover decline worldwide [30, 57], with their frequency and magnitude amplified by global climate change [85] (see Box 2 for details). At a global scale, marine heatwaves are exacerbated by natural climate variability, mainly the El Niño Southern Oscillation (ENSO, Figure 1.4B) [87].



▲ **Figure 1.4.** Occurrence of global bleaching events on coral reefs worldwide between 1985 and 2025 and their impact on relative change in hard coral cover. Daily sea surface temperature average over coral reefs of the world (A). Oceanic Niño Index representing the different phases of the El Niño Southern Oscillation (ENSO) (B). Percentage of coral reefs of the world experiencing heat stress levels per year (C). See Table 4.2 for a description of the different heat stress levels. Derived from NOAA Coral Reef Watch data (see Materials and Methods). Year-to-year relative change in hard coral cover and associated statistical change level, based on Bayesian exceedance probabilities and using the prior year to evaluate the change (D). The upper and lower bounds of each box indicate the 95% credible interval of hard coral cover change for a given year, while the central line denotes the median estimate.

As background temperatures rise, a larger proportion of coral reef area is exposed to more severe heat stress during each subsequent global bleaching event, with the highest stress level (Alert level 5) reported during the most recent — the fourth global bleaching event (Figure 1.4C). Across the integrated monitoring data in this report, declines in hard coral cover coincide with the peaks in the proportion of coral reefs exposed to heat stress during each bleaching event (Figure 1.4D).

Local and regional pressures

The observed stepwise decline in global hard coral cover, predominantly driven by episodic climate events, is compounded by local and regional-scale environmental and anthropogenic stressors, including acute events such as increased levels of coral predation (*e.g.* by crown-of-thorns starfish), tropical cyclones, and the often-chronic pressures from pollution, coastal development, and unsustainable marine resource use [88, 89].

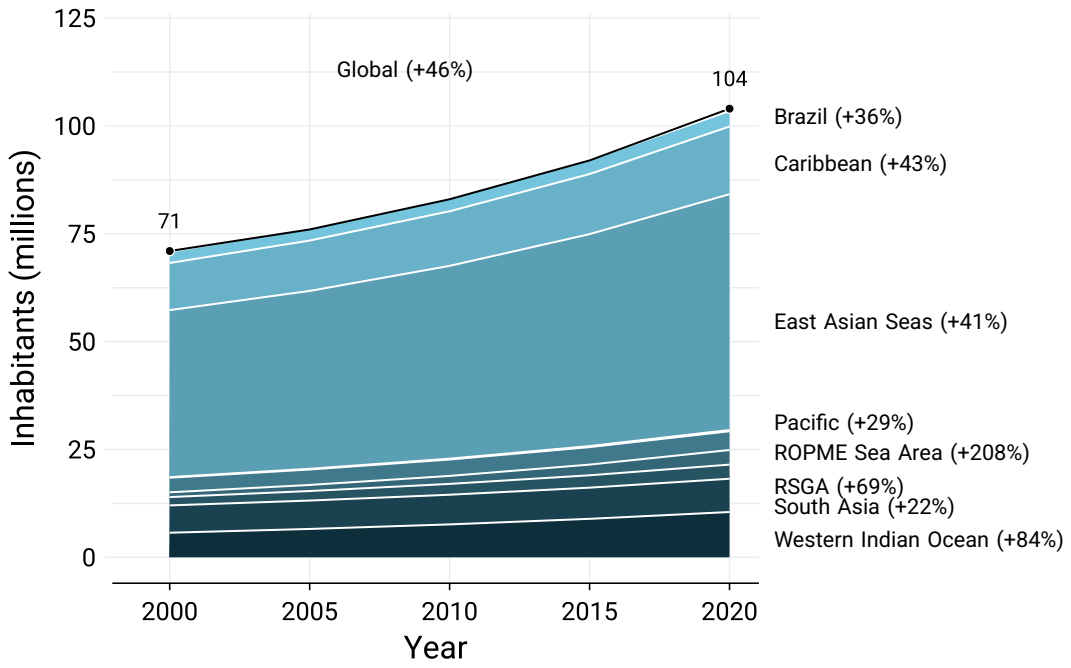
Global and regional aggregation of coral reef status and trends can reveal the overall influence of these impacts, but more detailed information is provided in the regional chapters (Part 2), which expand on localised pressures. Across coral reefs worldwide, the observed long-term decline in hard coral cover is strongly linked to global bleaching events. However, there are distinct periods when regional and acute pressures can help explain the global decline in hard coral cover during years not impacted by global bleaching events (Figure 1.4D). Specifically, there was strong evidence of declines in hard coral cover in 2003–2004, 2007–2008, and 2013–2015 (Figure 1.4D), which coincide with large-scale events affecting the GCRMN regions. For example, the physical damage and subsequent impacts on sedimentation and water quality from tropical cyclones, crown-of-thorns outbreaks, and regional marine heatwaves were documented as acute drivers of change that differentially affected the GCRMN regions during these periods (see Part 2 - Regional status of coral reefs for more detailed insights).

Regarding chronic pressures, overfishing and water pollution (nutrient and sediment inputs) affect about 30% of the global coral reef area [89]. Increased sedimentation, eutrophication, coastal acidification, and overfishing can have cumulative effects on key drivers of recovery, including reduced coral recruitment and coral diversity, and increased macroalgal abundance, coral disease, and coral predation [90, 91].

A useful high-level indicator for trends in local and regional anthropogenic pressures is human population statistics. As of 2020, the human population living within 5 km of coral reefs worldwide was 104 million inhabitants, representing a 46% increase over 20 years (Figure 1.5). Overall, the reported population increase strongly suggests pressures on coral reefs are increasing from human activities such as fishing, coastal and industrial development, tourism, and nutrient pollution [89]. East Asian Seas, the Caribbean, and the Western Indian Ocean are among the GCRMN regions with the largest human populations living near coral reefs, and these regions had high rates of population growth between 2000 and 2020 (41%, 43%, and 84%, respectively; Figure 1.5). While the highest human population growth rates were reported in the ROPME Sea Area (208%), population numbers near coral reefs are comparably low in this region.

Human population figures, on the other hand, may also indicate the growing importance of coral reefs to people through the ecosystem services they provide [92]. This is particularly important for Small Island Developing States, which have a population density in proximity to coral reefs (within 5 km) that is four times the global average [2].

Effectively managing coral reefs requires confronting a complex set of compounding pressures across local, regional, and global scales, while also safeguarding the communities and livelihoods that depend on healthy reefs [93, 94]. This poses a formidable challenge for coral reef conservation and management [1, 88]. Robust information on the status and trends of coral reef ecosystems provides a strong foundation for understanding and projecting the diverse and locally specific responses of reef systems to those pressures. As the impacts of global climate change on coral reefs worldwide intensify, adequate management of human pressures is an essential component of a suite of coordinated conservation actions to protect and better manage coral reef ecosystems, promote their ability to recover from acute disturbances, and ultimately halt their observed global decline [1, 93].



▲ **Figure 1.5.** Change in the number of people living within 5 km of coral reefs at the global and regional scales. The values in parentheses show the relative change in the number of inhabitants living within 5 km of the reefs between 2000 and 2020. Because the human population in Australia and the Eastern Tropical Pacific is small, these values are not shown in the figure (but see Table A.2). Derived from Gridded Population of the World v4.11 data (Center for International Earth Science Information Network (CIESIN) [95]; see Materials and Methods).

DIVERSE TRENDS IN HARD CORAL COVER ACROSS REGIONS

Global hard coral cover trends mask regional differences, with two GCRMN regions experiencing severe long-term declines to critically low levels of hard coral cover, while other regions show stability or limited recovery, underscoring the region-specific resilience and trajectories of coral reefs.

Beneath the global trends in hard coral and macroalgal cover, coral reef trajectories vary considerably across regions. Heterogeneity in reef condition at the regional level is shaped by distinct regional pressures and environmental conditions, resulting in different trajectories in hard coral cover across all 10 GCRMN regions (Table 1.1).

Four regions exhibit declining hard coral cover, with two concerning trajectories

Among the 10 GCRMN regions, four regions have demonstrated a multidecadal decline in hard coral cover: the ROPME Sea Area, the Caribbean, the Pacific, and Australia (Table 1.1). Importantly, two of these regions — the ROPME Sea Area and the Caribbean — are highlighted as being of significant concern due to their major regional declines and critically low levels of hard coral cover. In the ROPME Sea Area, coral reefs have lost about half of their hard coral cover (49.0% median; CrI $_{95\%}$: 38.7–57.8%) in recent years (2020–2024) compared to the reference period (1980–2009). In the Caribbean, hard coral cover began declining in the 1980s (median: 51.6%; CrI $_{95\%}$: 45.7–57.2%), and a persistently impaired functional state of coral reefs has been well documented in the scientific literature [96]. The significant decline and current low regional average hard coral cover (2024) of 15.0% in the ROPME Sea Area (median; CrI $_{95\%}$: 11.6–18.4%) and 7.1% in the Caribbean (median; CrI $_{95\%}$: 6.4–7.7%) indicate reef functioning may be compromised and future recovery may be constrained by reduced and spatially isolated local hard coral populations [92, 97, 98]. Australia and the Pacific regions have also shown moderate declines over the reporting period, particularly after the third and fourth global bleaching events, though not to the same extent as the Caribbean and ROPME Sea Area regions (Table 1.1).

Hard coral cover has not changed in four regions and increased in two

By contrast, the Red Sea and Gulf of Aden, Brazil, the East Asian Seas, and the Eastern Tropical Pacific regions show no or weak evidence of change relative to the reference period (Table 1.1). While hard coral cover has fluctuated over time, the comparison between the reference period (1980–2009) and the modern estimates in the 2020s (2020–2024) provides no or weak evidence of change. The East Asian Seas region has shown remarkable stability in hard coral cover over the last four decades. The Red Sea and Gulf of Aden region has been stable until the most recent impacts from the fourth global bleaching event. However, the recent impacts are insufficient to provide strong statistical evidence of a multidecadal decline in hard coral cover for that region. Brazil and the Eastern Tropical Pacific regions, in contrast, have fluctuated over the years, showing patterns of decline and recovery but with no evidence of an overall directional change.

Importantly, South Asia and the Western Indian Ocean are the only two regions showing a significant increase in hard coral cover when comparing recent years to the reference period (16% and 43% relative recovery, respectively). While the positive trends in hard coral cover are encouraging, the direction and magnitude of regional trends are interpreted relative to hard coral cover during the reference period (1980–2009) and are therefore contingent on the quality and availability of data from that period in each region. In South Asia, data were not available for the entire reference period and were mainly limited to years when severe disturbances affected coral reefs in the region (see the South Asia chapter for details). In the Western Indian Ocean, data sparsity prior to 1998 and uneven monitoring efforts across space and time during the reference period limit the accuracy of estimates of hard coral cover (see the Western Indian Ocean for details). If we consider published baseline estimates for the Western Indian Ocean (up to 40% [99]) and South Asia (~50% [58]), current hard coral cover estimates for the 2020s (31% and 35%, respectively; Table 1.1) indicate that both regions may have in fact experienced a decline in hard coral cover.

The diverse trends in hard coral cover across the 10 GCRMN regions are provided and discussed in more detail in Part 2 of this report. Environmental and anthropogenic drivers of the observed changes in hard coral cover are regionally specific, and the regional chapters in Part 2 provide a deeper interpretation of coral reef status and trends across geographies.

▼ **Table 1.1.** Comparison of changes in hard coral cover across regions between the reference period (1980–2009) and the 2020s (2020–2024). *Region* indicates the GCRMN region (RSGA = Red Sea and Gulf of Aden, ETP = Eastern Tropical Pacific, WIO = Western Indian Ocean). *Temporal trend* shows the trajectory of hard coral cover, with values on the left and right indicating average cover during 1980–2009 (black) and 2020–2024 (blue), respectively. *Change* shows the direction of change (arrow) and the relative change in hard coral cover between the two periods (NC = No Change). *Weight* shows each region's proportion of global reef area, used to weight regional estimates when calculating global-average hard coral cover. *Evidence* indicates the strength of evidence for a change between periods based on Bayesian exceedance probabilities: No evidence (---), Weak evidence (▮---), Evidence (▮▮-), and Strong evidence (▮▮▮) (see Table 4.5 for details).

Region	Temporal trend	Change	Weight	Evidence
ROPME Sea area	34.9% 17.8%	-48.7%	0.7%	▮▮▮
Caribbean	19.5% 11.0%	-43.4%	9.9%	▮▮▮
Pacific	34.5% 28.2%	-18.3%	26.2%	▮▮-
Australia	30.2% 26.8%	-10.9%	16.9%	▮▮▮
RSGA	32.9% 28.8%	NC	5.0%	▮---
Brazil	19.1% 18.8%	NC	0.5%	---
East Asian Seas	30.5% 30.7%	NC	30.3%	---
ETP	24.0% 25.1%	NC	0.2%	---
South Asia	30.1% 35.0%	+16.2%	4.3%	▮▮-
WIO	21.7% 31.0%	+43%*	5.9%	▮▮▮
Global	30.2% 27.3%	-9.5%		▮▮-

*This increase must be interpreted with caution since a very limited amount of data was available over the reference period (1980-2009, see Figure A.1).

BOX 1

Why do global hard coral cover trends differ between the 2020 and the 2025 GCRMN reports?

Authors

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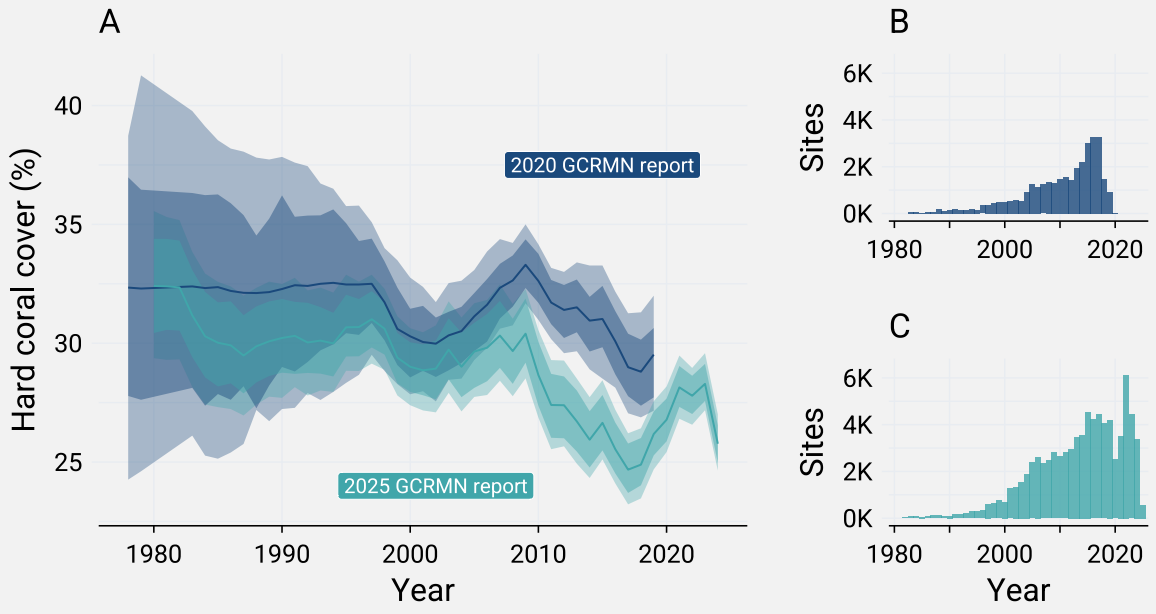
Science is a process, advancing through debate, methodological refinement, and correction, yet it is often perceived as a fixed body of established facts. As a result, when studies reach different conclusions, some may question the reliability of science altogether [100]. Explaining why results differ is therefore important, particularly for the GCRMN, which regularly publishes status reports where results may differ from those presented in earlier editions. This box explains why the global hard coral cover trends reported in the 2020 [30] and 2025 (this report) GCRMN reports differ, and, more broadly, why estimates can vary across scientific studies.

Differences between 2020 and 2025 global reports

Two differences stand out between the global trends in hard coral cover reported in the 2020 and 2025 GCRMN reports (Figure B1.1A). First, the credible intervals for 1980–1995 are roughly half as wide in the 2025 report, compared to those reported in the 2020 report for the same period (Figure B1.1A). These narrower credible intervals reflect a substantially larger dataset used in this report. Benthic cover trends were estimated from 12,160 monitoring sites in 2020, compared to 36,886 sites in 2025 — a threefold increase in data that sharpens the estimates and raises confidence in the trend. Changes in the priors used in the Bayesian models can also contribute to deviations in outcomes (see Materials and Methods).

Second, the decline in hard coral cover between 2009 and 2017 appears steeper in the 2025 report than in the 2020 report (Figure B1.1A). This is largely due to the addition of data for the years 2018–2019 for the 2025 report (Figure B1.1B). Surveys conducted one to two years before publication are typically underrepresented in GCRMN reports, because recent data take time to compile (Figure B1.1B and C). In the 2020 report, the limited availability of data for 2018–2019, immediately following the third global bleaching event, likely resulted in an underestimation of the decline in global hard coral cover. With a larger dataset across years, the 2025 report provides a more accurate assessment of post-bleaching changes during that same period.

Despite differences in data volume and model settings, the two reports agree closely in the magnitude and direction of the trend in hard coral cover. The updated 2025 analysis primarily refines the global hard coral cover trajectory and extends it through 2024. At the regional scale, trends show the same pattern — narrower 1980–1995 credible intervals and a steeper decline during the 2010s (results not shown) — with some region-specific changes. For example, the inclusion of historical data (1980–1990) for the Caribbean region in the 2025 report resulted in a steeper decline during the 1980s compared with the slope in the 2020 report. This steeper slope is consistent with patterns documented in the scientific literature (*e.g.* Gardner *et al.* [96] and Schutte *et al.* [101]).



▲ **Figure B1.1.** Comparison of the hard coral cover estimated trends at the global scale between the *Status of Coral Reefs of the World: 2020* [30] and the *Status of Coral Reefs of the World: 2025* (this report) GCRMN reports (A). The solid lines represent the estimated median hard coral cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Distribution of the number of monitoring sites per year for the *Status of Coral Reefs of the World: 2020* report [30] (B) and the *Status of Coral Reefs of the World: 2025* report (this report) (C). Note that the number of sites on the y-axes of subfigures B and C are expressed in thousands.

Why do estimates differ across scientific studies more broadly?

Differences like these are not unique to the global GCRMN reports. Estimates of trends can vary across studies for several reasons, including differences in the quantity and quality of the underlying data, and the modelling approaches used. For example, hierarchical Bayesian models were used in the 2020 [30] and 2025 global reports, whereas machine learning models were applied in the Pacific [44] and Caribbean [45] regional reports. Even within one approach, settings such as Bayesian priors or machine learning hyperparameters may vary. Studies also differ in the time periods and spatial boundaries they use. For example, several regional and subregional boundaries were revised between the 2020 and 2025 reports. Reef distribution layers (see Box 3), data quality-control procedures, and benthic category definitions can all contribute to differences in estimates as well.

The degree to which studies agree is itself informative. When analyses based on different data and methods reach the same or similar conclusions, they form a robust body of evidence. For example, to our knowledge, every study of Caribbean coral reefs reports a pronounced decline in hard coral cover since 1980 [45, 96, 101]. Conversely, divergent conclusions between studies highlight where further research is needed. For instance, earlier GCRMN Pacific regional reports [44, 102] concluded hard coral cover remained relatively stable, whereas the estimates presented in this 2025 report indicate a decline in hard coral cover between 1980 and 2024. Divergent results do not necessarily mean one study is wrong. More often, they reflect genuine differences in data, methods, and time frames, which is also why trends should be compared across studies with caution.

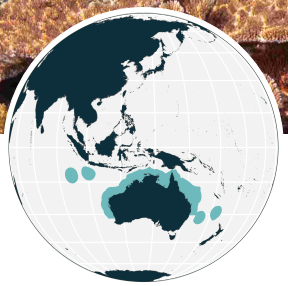
PART 2

Regional status of coral reefs





John Brewer Reef, Great Barrier Reef
Credit: Victor Huertas



AUSTRALIA

Authors

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Citation

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Traditional Owner acknowledgement

The authors recognise Aboriginal and Torres Strait Islanders as the Traditional Owners of the land and sea Country on which we work. We recognise their enduring cultural connections to land, water, and community, and their leadership, knowledge, and ongoing stewardship of the reef systems described in this report.

We pay our respects to their Elders, past and present, and extend that respect to all First Nations peoples.

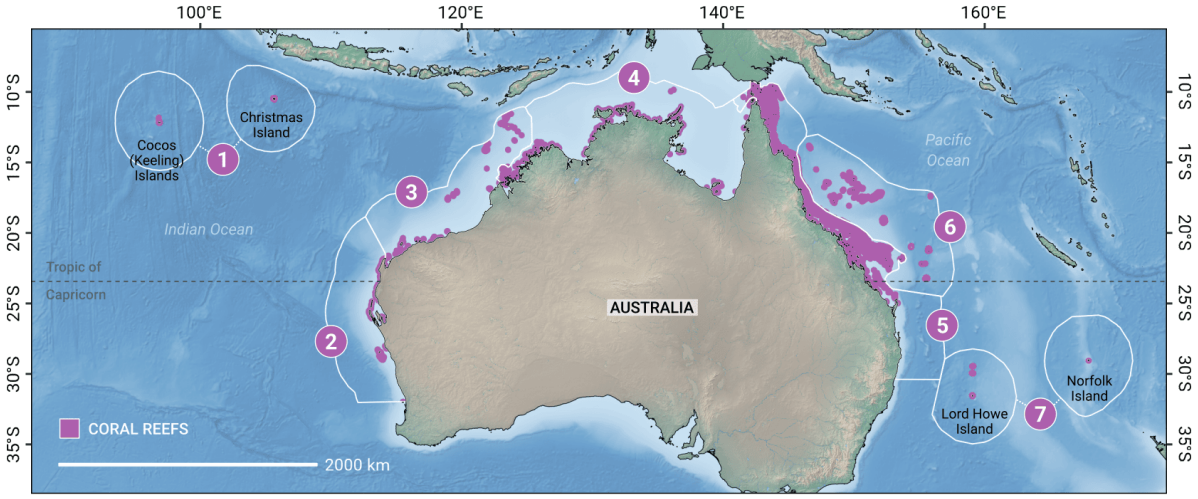
The data analysed in this study were collected from coral reef systems across many places that are part of the lands and Sea Country of Aboriginal and Torres Strait Islander peoples. The authors sought to acknowledge the Traditional Owners associated with these monitoring locations. However, it was not possible within the scope of this report to verify a complete and accurate list. To avoid the risk of omission, misidentification or misattribution, we have used a broader acknowledgement here. We recognise this as a limitation of the current chapter and an important area for improvement in future GCRMN Australia reporting.

INTRODUCTION

Australia hosts one of the world's most extensive and diverse coral reef assemblages. These coral reefs span more than 20° of latitude along the east and west coasts, from the tropical reefs of the Torres Strait and north-west shelf, such as the Rowley Shoals, to temperate reefs at Lord Howe, Norfolk, and the Houtman Abrolhos Islands [103]. This broad geographic range supports a continuum of reef assemblages, from species-rich tropical coral communities to low-diversity, high-latitude systems shaped by strong seasonal variability. Reef geomorphology is equally diverse, including oceanic atolls in the Indian Ocean, seamounts in the Coral Sea rising from abyssal depths, the extensive continental shelf barrier reef system of the Great Barrier Reef, and turbid-water fringing reefs along continental margins. Three of Australia's reef systems have been recognised as World Heritage Areas: Ningaloo Reef, the world's largest fringing reef; Lord Howe Island, which supports one of the southernmost coral reefs; and the Great Barrier Reef, the largest coral reef system on Earth. Together, Australia's reefs form interconnected ecological networks that support biodiversity, fisheries, coastal protection, and cultural values.

Recent national reef-mapping initiatives identified more than 21,000 reef structures within Australia's exclusive economic zone [104], highlighting the scale and complexity of the systems encompassed by the Australia GCRMN region. The region extends over a maritime area of approximately 5 million km^2, ranging across 20° of latitude and 70° of longitude (Figure 2.1.1) and includes at least 43,000 km^2 of coral reef, representing approximately 17% of the world's coral reef extent. This makes Australia the third-largest GCRMN region by reef extent, and the country with the largest national extent of coral reef globally [2]. Capturing long-term trends in reef condition across such varied and often remote systems is logistically and methodologically demanding. As a result, the regional and subregional summaries presented here do not include all reefs and sampling years, and may smooth or obscure important local-scale dynamics, including pulse disturbances and subsequent recoveries.

The GCRMN Australia region is divided into seven subregions (Figure 2.1.1), which vary markedly in coral reef extent (Table A.1) and mapping accuracy (see Box 3). As a result, their contributions to regional trends in hard coral and macroalgal cover are not equal. Subregion 5, which includes the Great Barrier Reef, Torres Strait and South East Queensland, contains 81.6% (34,503 km^2)



▲ **Figure 2.1.1.** The distribution of coral reefs within the Australia region and respective subregions. White polygons and numbers represent subregions. Subregion 1 = Cocos Keeling and Christmas Island, Subregion 2 = Central West Australian Shelf, Subregion 3 = Northwest Australian Shelf, Subregion 4 = Northern Australia, Subregion 5 = Eastern Australian shelf / Great Barrier Reef, Subregion 6 = Coral Sea, Subregion 7 = Lord Howe and Norfolk Islands. Coral reefs are delineated with a 20 km buffer to enhance interpretation, and only coral reefs within the region are represented. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region.

of Australia's mapped reefs and strongly influences regional model outputs. In contrast, subregion 7, which includes Lord Howe Island, Norfolk Island, and Elizabeth and Middleton Reefs, comprises just 0.3% (126 km^2) and has minimal influence on regional trends (Table A.1).

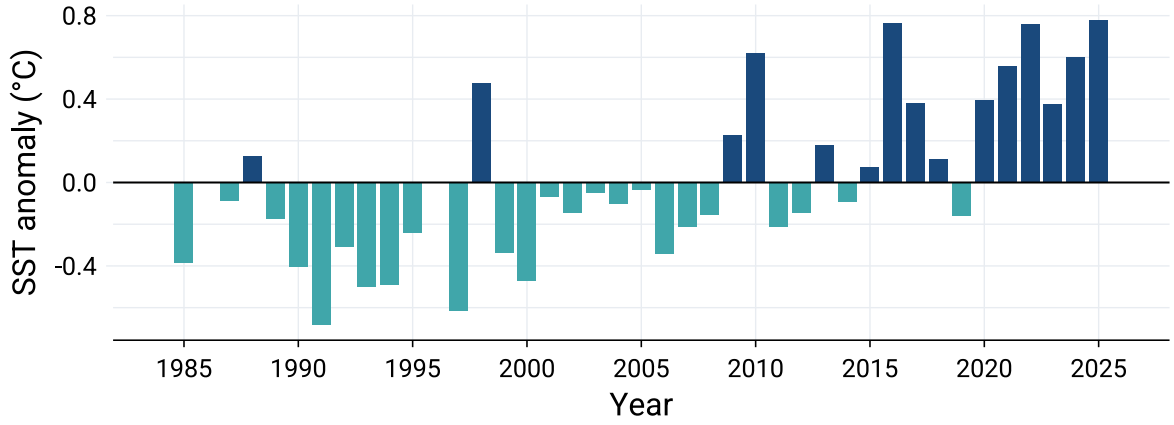
In Australia, over a million people live within the Great Barrier Reef catchment and an estimated two million benefit from Australian reefs through tourism, recreational and commercial fisheries, and reef science and management [16, 105]. Overall, 37% of Australia's population is located within 100 km of a coral reef, the highest proportion of all GCRMN regions [2], which may help explain why coral reefs are a defining element of Australia's national identity [106]. Equally important is their deep and enduring cultural significance for Australia's First Nations peoples, whose custodianship and knowledge of coral reefs extend back millennia. These long-standing relationships with Sea Country represent the oldest continuous reef monitoring system on Earth, with knowledge about reefs and their changes embedded in stories, practices, and cultural governance that continue to inform contemporary reef stewardship [24].

THREATS AND DISTURBANCES

Heat stress

On average, Australian reefs experience slightly cooler temperatures than most GCRMN regions because they span a broad latitudinal gradient, with a long-term mean satellite-derived sea surface temperature (SST) of 26.4 °C across reefs over the past 40 years. Between 1985 and 2025, average regional SST increased by 0.78 °C (0.19 °C per decade; Table A.3).

Over the 40-year record, the severity, spatial extent, and duration of annual heat stress events increased (Figure 2.1.2). Between 1986 and 2024, up to 80% of coral reefs in Australia accumulated Degree Heating



▲ **Figure 2.1.2.** Annual mean sea surface temperature (SST) anomalies (°C) over the coral reefs of the Australia region from 1985 to 2025. SST anomalies were calculated based on the long-term average over the period 1985–2025. Positive values (red bars) indicate SST warmer than the long-term average, whereas negative values (blue bars) indicate SST cooler than the long-term average. Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).

Weeks (DHW) over 4°C-weeks in a single year, with the highest value recorded in 2024 (Figure 2.1.3). The proportion of reefs experiencing severe heat stress (≥ Alert Level 2) has increased since 2016 (Figure 2.1.3).

Periods of elevated heat stress have led to several subregional or widespread mass coral bleaching events across Australian reefs over the reporting period, including in 1998, 2002, 2004 (largely restricted to the Coral Sea Marine Park), 2011–2013 (primarily central Western Australia and the Northern Territory), 2016–2017, 2020–2022 and most recently during 2024–2025; the impacts of the latest event are still being assessed (see Box 2).

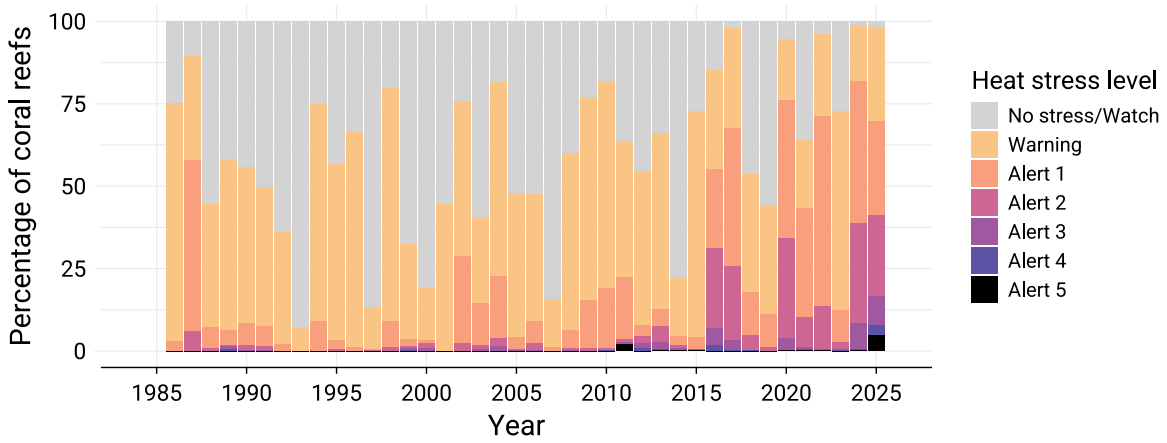
Cyclones

Across Australia, severe tropical cyclones have caused major structural damage to coral reefs, with the most frequent and severe impacts in mid-latitudes (≅12–22°S). From 1980 to 2025, a total of 191 cyclones passed within 100 km of coral reefs within the region (Figure 2.1.4). Swells generated by more distant cyclones, not captured in this metric, as well as seasonal and monsoonal storms, may have also affected coral reefs [108].

The Eastern Australian Shelf has experienced episodic but severe structural damage from cyclones, while the Central West Australian Shelf has experienced more frequent cyclonic events. Major cyclones on the Eastern Australian Shelf (cyclones Larry 2006, Hamish 2009, Yasi 2011) caused extensive coral mortality and declines in hard coral cover [109] (Figure 2.1.5). On the Northwest Australian Shelf several cyclones have caused moderate or severe damage. On the Central West Australian Shelf, cyclones Vance (1999), Seroja (2021) and Isla (2023) caused severe damage to Pilbara and Ningaloo reefs. Cyclone Seroja was notable for its unusually southerly track, damaging reefs at the Houtman Abrolhos Islands (28.5°S), which are rarely exposed to intense cyclones.

Coral predators

Australian reefs have experienced periodic outbreaks (also referred to as "irruptions") of the coral-eating crown-of-thorns starfish (*Acanthaster* cf. *solaris*, CoTS). Since the 1960s, four major outbreaks (beginning 1962, 1979, 1993 and 2010) have occurred on the Eastern Australian Shelf, with a fifth



▲ **Figure 2.1.3.** Percentage of the coral reefs within the Australia region experiencing heat stress levels per year, from 1986 to 2025. No stress/Watch (DHW = 0); Warning (possible bleaching, 0 < DHW < 4); Alert 1 (reef-wide bleaching, 4 ≤ DHW < 8); Alert 2 (reef-wide bleaching and mortality of heat-sensitive corals, 8 ≤ DHW < 12); Alert 3 (multi-species mortality, 12 ≤ DHW < 16); Alert 4 (severe multi-species mortality, 16 ≤ DHW < 20); and Alert 5 (near-complete mortality, DHW ≥ 20). Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).

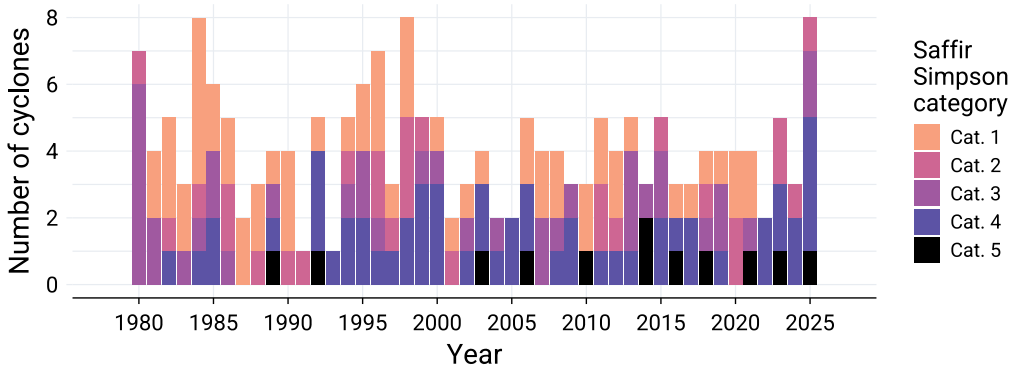
wave now emerging [110]. These outbreaks typically originate in the northern Great Barrier Reef and spread southwards, contributing substantially to regional coral loss [111]. In other subregions, only localised CoTS aggregations have been reported, although outbreak-level densities were recorded locally on Elizabeth and Middleton Reefs in the early 1980s and on the Central West Australian Shelf in 2014 [112]. Outbreaks of the corallivorous gastropod *Drupella* spp. occurred on the Central West Australian Shelf in the late 1980s and early 1990s [113]. Densities remained low thereafter, although observations following the 2025 bleaching suggest localised aggregations on surviving corals.

Water quality

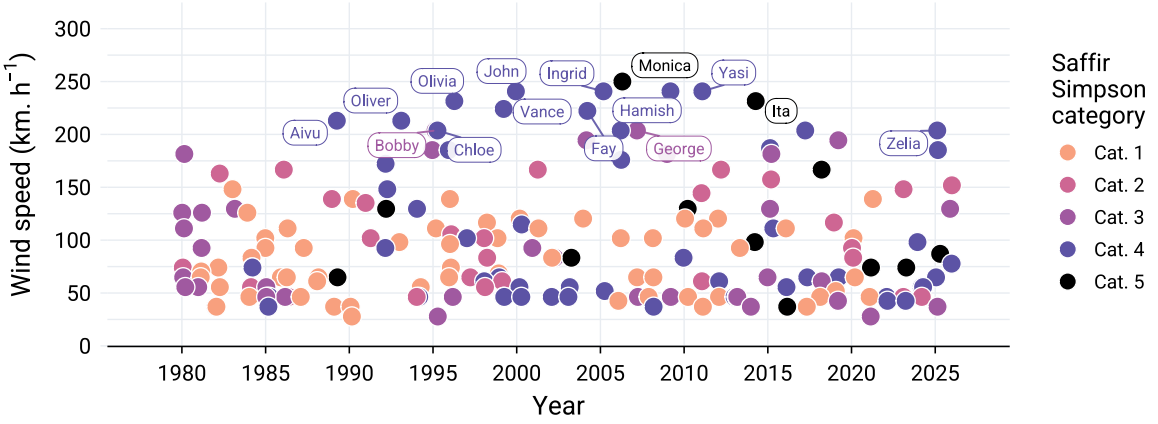
Water quality pressures vary among subregions. On the Eastern Australian Shelf, land-based runoff transports sediments, nutrients and pesticides to inshore reefs, degrading inshore ecosystem condition and limiting recovery after disturbances [114, 115]. Since European settlement, land use change has increased river loads of fine sediments and dissolved inorganic nitrogen by approximately 1.4–5 and 1.5–3 times, respectively [114]. Extreme rainfall and flooding may further intensify these pressures. Some coastal Western Australian reefs are occasionally affected by episodic flood events and localised dredging operations [116].

Other disturbances

Direct pressures to Australian coral reefs from coastal development, dredging, port maintenance and other local impacts (*e.g.* oil contamination, plastics, marine debris, ship groundings and overfishing) remain relatively low by global standards but are increasing in areas [117]. Management actions, including zoning, fishery controls, crown-of-thorns control and water-quality improvement programs can reduce local pressures and support recovery, but their benefits are increasingly constrained by recurrent climate-driven disturbances.



▲ **Figure 2.1.4.** Number of cyclones that passed within 100 km of the coral reefs of the Australia region from 1980 to 2025. Colors correspond to the Saffir–Simpson category of the cyclone along its entire track. Derived from IBTrACS data (Knapp *et al.* [118]; see Materials and Methods).



▲ **Figure 2.1.5.** Maximum sustained wind speed of cyclones that passed within 100 km of a coral reef in the Australia region between 1980 and 2025. Colors correspond to the Saffir–Simpson category of the cyclone along its entire track. However, the values of sustained wind speed are extracted from the nearest cyclone position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir–Simpson scale (*i.e.* 119 km.h^{-1}). Derived from IBTrACS data (Knapp *et al.* [118]; see Materials and Methods).

STATUS OF CORAL REEFS IN THE REGION

Across the last four decades hard coral cover in the Australia region has been impacted by numerous disturbances. However, the reefs have generally been resilient, with rapid increases in hard coral cover following disturbance. Macroalgae have rarely proliferated on coral reefs in the region, with hard coral cover largely replaced by turf algal cover. However, as the rates of coral loss and sometimes recovery have increased in more recent years, the volatility of benthic cover dynamics in this region appears to reflect an escalating disturbance regime [119].

Hard coral cover

Median hard coral cover across the Australia region ranged between 20% and 34% over the past four decades, with cycles of disturbance-driven decline and recovery (Figure 2.1.6A). Prior to the early 2000s, variability was largely driven by impacts from cyclones and by CoTS on the Eastern Aus-

tralian Shelf, whereas mass bleaching has become the dominant cause of coral decline over the past decade [73].

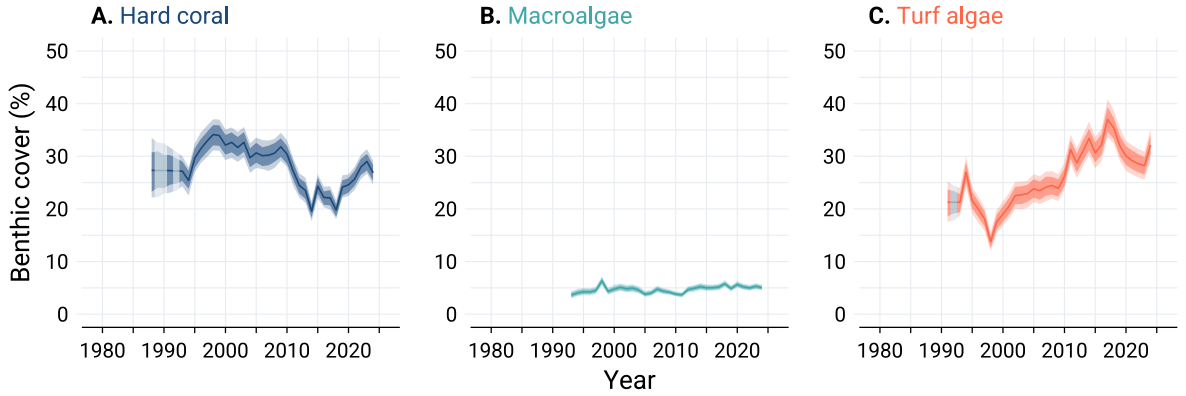
Prior to 1993, estimates of hard coral cover were sparse and unreliable. Regional hard coral cover increased to 34.1% (credible interval at 95% [CrI $_{95\%}$]: 31.1 – 37.0%) in 1998, the highest on record for the region, before declining to 29.7% (CrI $_{95\%}$: 26.8 – 32.5%) in 2004 following coral bleaching in the Eastern Australian Shelf and Northwest Australian Shelf, coupled with local impacts from multiple cyclones. Hard coral cover then remained relatively stable until 2010, after which it declined rapidly to 19.6% (CrI $_{95\%}$: 17.7 – 21.4%) by 2014, primarily due to CoTS and two cyclones on the Eastern Australian Shelf, and mass coral bleaching in the Central West Australian Shelf.

Hard coral cover returned to 24.3% (CrI $_{95\%}$: 22.0 – 26.6%) in 2015, before further mass bleaching events across most subregions and localised cyclone impacts reduced regional cover to 19.8% (CrI $_{95\%}$: 18.1 – 21.7%) in 2018. Cover then increased steadily to 29.0% (CrI $_{95\%}$: 26.7 – 31.4%) by 2023 during a period when disturbances causing extensive coral mortality were limited, although localised bleaching and cyclone impacts occurred in some subregions.

From 2024, severe heat stress and widespread coral bleaching and mortality were recorded in at least five of the seven subregions. The impacts of this most recent global bleaching event are not accounted for in this report, as the analysis includes data only to the end of 2024. However, a decline in hard coral cover was already evident in 2024. Summer temperature anomalies for this event were unprecedented in at least 400 years [120], and the 2024–2025 event is expected to represent the most severe mass bleaching event on record for the Australia region (see Box 2).

Macroalgal cover

Macroalgal cover is generally low across the Australia region (Figure 2.1.6B), with higher levels often limited to coastal reef habitats. For example, on the Great Barrier Reef [83] and in Western Australia, macroalgal cover is typically limited to inshore reefs, where more turbid, nutrient-rich waters favour macroalgal growth compared to other reefs in the region.



▲ **Figure 2.1.6.** Modeled temporal trends of benthic cover in the Australia region from 1980 to 2024 for hard coral (A), macroalgae (B), and turf algae (C). The solid line represents the estimated median benthic cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

Between 1988 and 2024, macroalgal cover ranged from 3.6% to 6.3%, and has remained relatively stable since 1998. Despite recurrent disturbances and substantial declines in hard coral cover, consistently low macroalgal cover provides no evidence of widespread coral-macroalgal phase shifts across the Australia region.

Turf algal cover

In the Australia region, turf algal cover has generally varied inversely with hard coral cover, ranging from 13.8% (CrI $_{95\%}$: 12.0 – 15.6%) in 1998 to 37.0% (CrI $_{95\%}$: 33.6 – 40.9%) in 2017, with the most recent value in 2024 being 32.1% (CrI $_{95\%}$: 29.3 – 35.1%) (Figure 2.1.6C). There was a 4.6 percentage point increase in turf algal cover from 2012 to 2014, coinciding with the absolute loss of 4.9 percentage points in hard coral cover. Conversely, turf algal cover declined by 3.3 percentage points between 2018 and 2024, while hard coral cover increased by 7.1 percentage points over the same period.

Since 2011, turf algae have been the dominant benthic group on coral reefs in the Australia region, with median turf algal cover exceeding that of hard corals by 1–16% in all but one year, although the evidence is limited for some years (Figure 2.1.6). By contrast, during the preceding 23 years, turf algal cover exceeded that of hard corals in only a single year. The occupation of space by turf algae following coral loss represents a natural transitional stage on healthy reefs, as short turf algae generally do not inhibit coral recruitment [121]. In many cases, increased turf cover reflects repeated disturbances rather than shifts in selective pressures [122]. However, alterations to key reef processes (*e.g.* herbivory or sediment deposition) can promote the development of longer turfs, which can trap sediment and hinder coral recruitment and recovery (see Case study 1).

STATUS OF CORAL REEFS IN THE SUBREGIONS

Australia 1 - Cocos Keeling and Christmas Island

The remote Cocos (Keeling) atoll and Christmas Island volcanic seamount (>970 km to its northeast) occur in Australia's external territories in the Indian Ocean, ~2100 km from mainland Australia. Reefs in this region are generally characterised by high hard coral cover and have shown an overall upward trajectory since monitoring began in 1997 (Figure 2.1.7A), despite periodic heat-stress events and chronic pollution at some sites on Christmas Island [123]. The only major decline occurred during a marine heatwave at Christmas Island in 2016, when hard coral cover declined from 47.5% (CrI $_{95\%}$: 36.3 – 59.4%) in 2015 to 28.4% (CrI $_{95\%}$: 19.4 – 38.2%) in 2017. Hard coral cover then increased to 55.8% (CrI $_{95\%}$: 43.7 – 65.7%) in 2023, among the highest recorded in the Australia region.

Australia 2 - Central West Australian Shelf

Coral reefs in this subregion span two World Heritage Areas across a tropical-temperate transition zone. Ningaloo Reef is the world's longest fringing reef, while the Houtman Abrolhos Islands have the southernmost coral reefs in the Indian Ocean. Coral monitoring began in 1988 with estimated cover being relatively stable at 21.0–24.5% until 2010. However, considerable variation around the estimated cover reflects non-continuous data collection in the Houtman Abrolhos and Shark Bay ecoregions, particularly prior to 2008. Hard coral cover decreased from 23.9% (CrI $_{95\%}$: 16.8 – 33.8%) in 2010 to 15.4% (CrI $_{95\%}$: 9.6 – 25.6%) in 2012 following a marine heatwave and mass bleaching that affected Ningaloo and Shark Bay [124, 125]. Hard coral cover has since

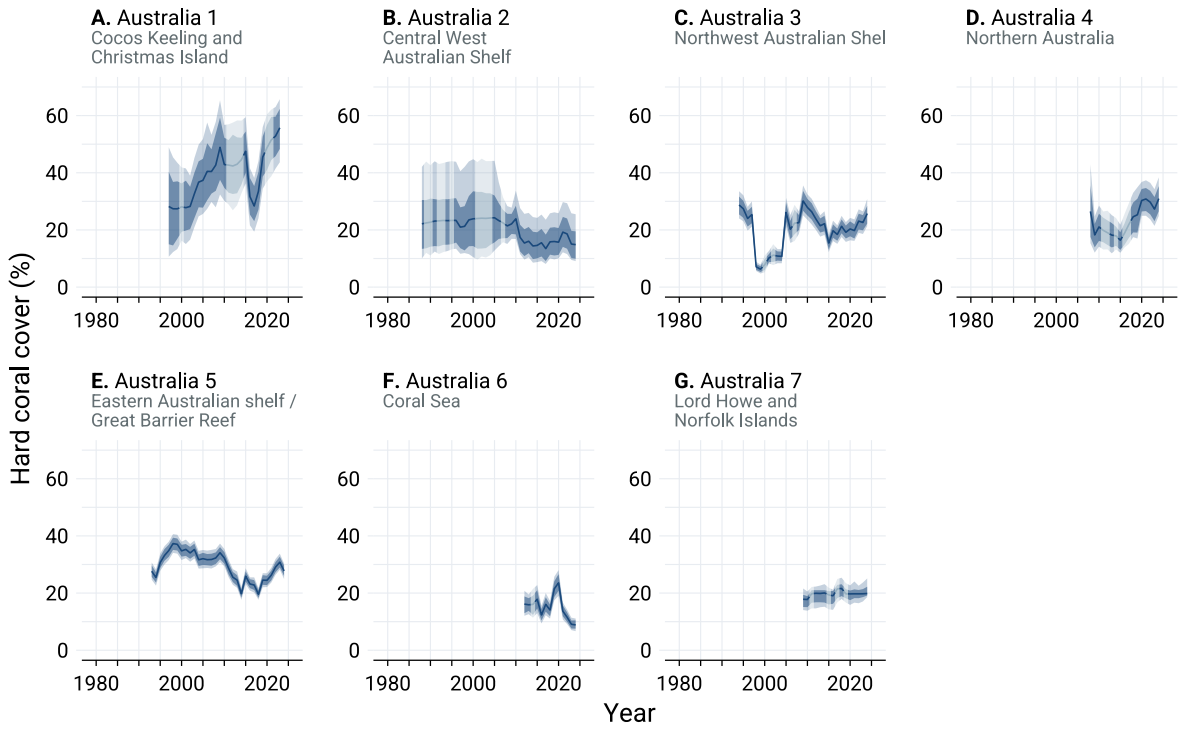
fluctuated between 13.4% and 19.2% (Figure 2.1.7B).

Australia 3 - Northwest Australian Shelf

Coral reefs in this subregion range from inshore turbid reefs to clear-water oceanic atolls near the continental shelf edge. Many reefs were affected by a marine heatwave in 1998, particularly two oceanic atoll systems, causing a subregional decline in hard coral cover from 25.4% (CrI $_{95\%}$: 21.1 – 30.0%) in 1997 to 6.3% (CrI $_{95\%}$: 4.7 – 7.9%) in 1999; the lowest on record [126]. Hard coral cover took more than a decade to recover, reaching a peak of 30.1% (CrI $_{95\%}$: 24.8 – 35.7%) by 2009. Cover then gradually declined to 15.4% (CrI $_{95\%}$: 12.4 – 18.9%) by 2015 following recurrent marine heatwaves and coral bleaching, particularly during 2011–2013 when some inshore reefs were affected. Subregional cover gradually increased to 25.7% (CrI $_{95\%}$: 20.8 – 30.8%) by 2024 (Figure 2.1.7C), as recovery across the subregion outpaced the impacts of local marine heatwaves and cyclones, including multiple bleaching events on the Northwest Australian Shelf from 2016.

Australia 4 - Northern Australia

Coral reefs in this subregion persist in extreme environmental conditions, including high turbidity and tidal ranges of up to 11 m that regularly expose intertidal reefs [127]. Prior to 2020, the percentage cover of hard corals ranged from 16.4% to 26.5%. However, data used in this analysis before 2020 were limited to sparse surveys during a single year across different locations, making early trend estimates less reliable.



▲ **Figure 2.1.7.** Modeled temporal trends of hard coral cover in Australia subregions from 1980 to 2024. The solid line represents the estimated median hard coral cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

From 2020 to 2024, repeated monitoring of sites was conducted along the Bonaparte Coast (north Kimberley), and subregional cover remained relatively stable at around 29.8% (CrI $_{95\%}$: 24.0 – 36.7%) (Figure 2.1.7D).

Australia 5 - Eastern Australian Shelf

Hard coral cover across eastern Australia's tropical and subtropical reefs — including the Torres Strait, Great Barrier Reef and South East Queensland — has fluctuated over the past four decades, reflecting an environment increasingly characterised by short recovery windows between acute disturbances and rising pressures associated with coral bleaching [128]. Hard coral cover increased from 1994 to a peak in 1998 (Figure 2.1.7E), before entering a prolonged period of decline caused by cyclones, marine heatwaves, CoTS outbreaks and, in some areas, flood-related water quality impacts. These pressures were particularly pronounced on the Great Barrier Reef during the early to mid-2010s. Although hard coral cover increased from 2018, this recovery was reversed due to marine heatwaves in 2024, the impacts of which are not included here.

Australia 6 - Coral Sea

The Coral Sea subregion comprises 32 widely dispersed reef systems within the Coral Sea Marine Park, an area spanning nearly one million square kilometres and extending over 1,000 km from the Australian mainland. These reefs occur on seamounts rising from deep-water plateaus and, due to their isolation, experience limited direct human pressures such as fishing and terrestrial runoff. They are, however, increasingly exposed to climate-driven disturbances.

Overall, hard coral cover remained relatively stable from 2012 to 2018, ranging from 12–18% (CrI $_{95\%}$: 9.4 – 22.8%) (Figure 2.1.7F), but with reductions in cover due to substantial heat stress and bleaching in the central Coral Sea in 2016 and 2017 [129]. Hard coral cover subsequently recovered, peaking at 23.6% (CrI $_{95\%}$: 19.0 – 28.0%) in 2020, before rapidly declining to 8.9% (CrI $_{95\%}$: 6.7 – 11.1%) in 2024 due to severe heat stress and widespread bleaching across much of the region [130].

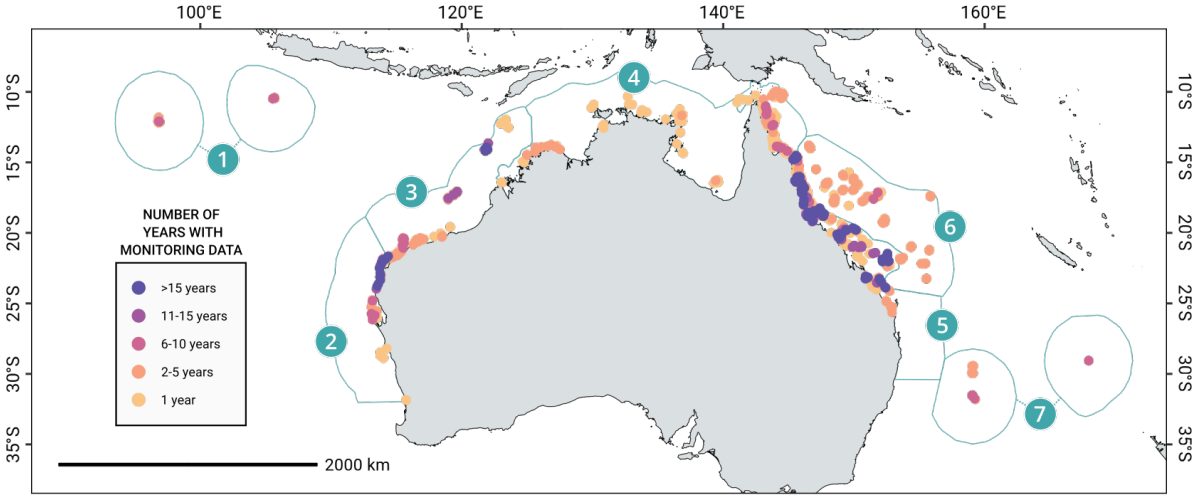
Australia 7 - Lord Howe and Norfolk Islands

The world's southernmost coral reefs, Norfolk and Lord Howe Island coral reef ecosystems occur within Australia's Temperate East Marine Park, where subtropical coral communities develop on shallow and submerged reef systems.

Overall, hard coral cover has remained stable at ~20% since monitoring began in 2009 and was estimated at 18.8% (CrI $_{95\%}$: 11.5 – 29.5%) in 2025 (Figure 2.1.7G). Despite severe bleaching (up to 53%) at some sites at Lord Howe in 2019 and 2024, and bleaching in 2020 and 2024 coupled with pressure from catchment runoff at Norfolk Island, no decline in hard coral cover was detected. This temporal stability is likely due to the proximity of reefs to cooler, deeper water and the cooling effect of cyclones, which reduced heat stress and aided recovery from bleaching.

DESCRIPTION OF MONITORING DATA

Temporal trends in benthic cover for the Australia region were estimated using 14 datasets encompassing 2,645 monitoring sites. These data include 13,211 in-water surveys conducted between 1988 and 2025, with 90% of surveys taking place after 2001 (Figure A.1).



▲ **Figure 2.1.8.** Spatio-temporal distribution of benthic cover monitoring sites across the Australia region and respective subregions. Light blue polygons and numbers represent subregions. Subregion 1 = Cocos Keeling and Christmas Island, Subregion 2 = Central West Australian Shelf, Subregion 3 = Northwest Australian Shelf, Subregion 4 = Northern Australia, Subregion 5 = Eastern Australian shelf / Great Barrier Reef, Subregion 6 = Coral Sea, Subregion 7 = Lord Howe and Norfolk Islands. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region.

Survey effort was unevenly distributed geographically (Figure 2.1.8). Of all the surveys conducted between 1988 and 2025, 71.4% (representing 58.5% of surveyed sites) occurred in the Eastern Australian Shelf (subregion 5), whereas only 1.2% (2% of sites) were conducted in Northern Australia (subregion 4) (Table A.4). This concentration of long-term monitoring in the Eastern Australian Shelf, particularly on offshore reefs, drives the regional patterns in hard coral cover and biases the contribution of other benthic groups. For example, lower macroalgal cover typical of offshore reefs may be overrepresented relative to inshore and turbid fringing reef habitats that have received less monitoring effort.

Monitoring efforts were also biased through time, with sparse and discontinuous data from the 1980s–2000s limiting our confidence in historical estimates in many reefs and some subregions (Figure 2.1.7). Much of the effort was again concentrated on the Great Barrier Reef, with 220 of the 241 long-term (>15 years) monitoring sites on the Eastern Australian Shelf (Figure 2.1.8).

The uncertainty in historic data highlights the need for sustained investment in long-term monitoring around Australia, which is increasingly critical in our rapidly changing environment. As data from the fourth global bleaching event continue to be analysed, forecasts indicate an extreme El Niño in 2026 that could elevate bleaching risk in 2026–2027. Assessing the impacts on Australian coral reefs, identifying potential refugia and tracking recovery all depend on regular, comparable and rigorous in-water surveys. The monitoring data presented here represent a substantial expansion from the 2020 report, increasing from 372 to 2,645 sites: a seven-fold increase in spatial coverage. As a result, Australia's proportional contribution to the global GCRMN dataset has more than doubled, from 3.1% to 6.7%. Nevertheless, this contribution remains low relative to Australia's share (~17%) of the world's coral reefs.

Data were contributed by a diverse range of organisations, including three citizen sci-

ence programs, four universities and five government agencies, highlighting the collaborative commitment to reef monitoring across sectors. However, significant geographic gaps persist, particularly in northwest and northern Australia. Monitoring in these regions is logistically challenging due to remoteness, high turbidity and safety constraints associated with in-water surveys. Addressing these gaps will require sustained investment, stronger partnerships with local ranger groups and Traditional Owners, and increased adoption of remote and autonomous monitoring technologies to support surveys in inaccessible areas. Recognising and incorporating Indigenous stewardship is essential to building a more representative, inclusive and regionally balanced understanding of coral reef conditions across the Australia region.

CONCLUSIONS

Australian coral reefs remain in relatively good condition in an international context, with many regions continuing to support high hard coral cover and ecological processes that facilitate recovery following disturbance. However, these ecosystems are facing mounting pressure from climate change, with marine heatwaves and mass coral bleaching events increasing in frequency, scale and severity, alongside ongoing impacts from crown-of-thorns starfish outbreaks, cyclones, flood plumes, disease and other local pressures. Many Australian reefs have demonstrated a capacity to recover when given sufficient time between major disturbances. However, the increasing frequency and overlap of disturbance events are reducing opportunities for recovery, and there is no guarantee that historical patterns of recovery will continue under future climate conditions. Importantly, the findings presented here do not fully capture the ecological consequences of the 2024–2025 global bleaching event, the most widespread and severe bleaching event recorded in Australia.

As climate-driven disturbances intensify, long-term monitoring is becoming increasingly important for detecting change, understanding recovery trajectories, identifying emerging risks and informing management responses. Continued investment in monitoring, including expansion of its spatial and ecological coverage, will be essential to support the stewardship of Australia's diverse reef ecosystems in a rapidly changing climate.

DATA CONTRIBUTORS

Tracy Ainsworth, Tahlia Bassett, David R. Bellwood, Graham J. Edgar, Michael Emslie, Richard Evans, Jan Freiwald, James Gilmour, Andrew Hoey, Thomas Holmes, Bill Leggat, Nick Leigh, Jennie Mallela, Jenny Mihaly, Molly Moustaka, Elizabeth Oh, Melanie Orr, William Robbins, Chris Roelfsema, Claire Ross, Nicole Ryan, Rick D. Stuart-Smith, Sterling B. Tebbett, Angus Thompson, Damian Thomson.

Access to online supplementary materials

The modelled benthic cover data, detailed outputs from the data analyses, and the CRediT author contribution statement are available on Zenodo.

CASE STUDY 1

Algal phase-shifts on coral reefs and the role of turfs

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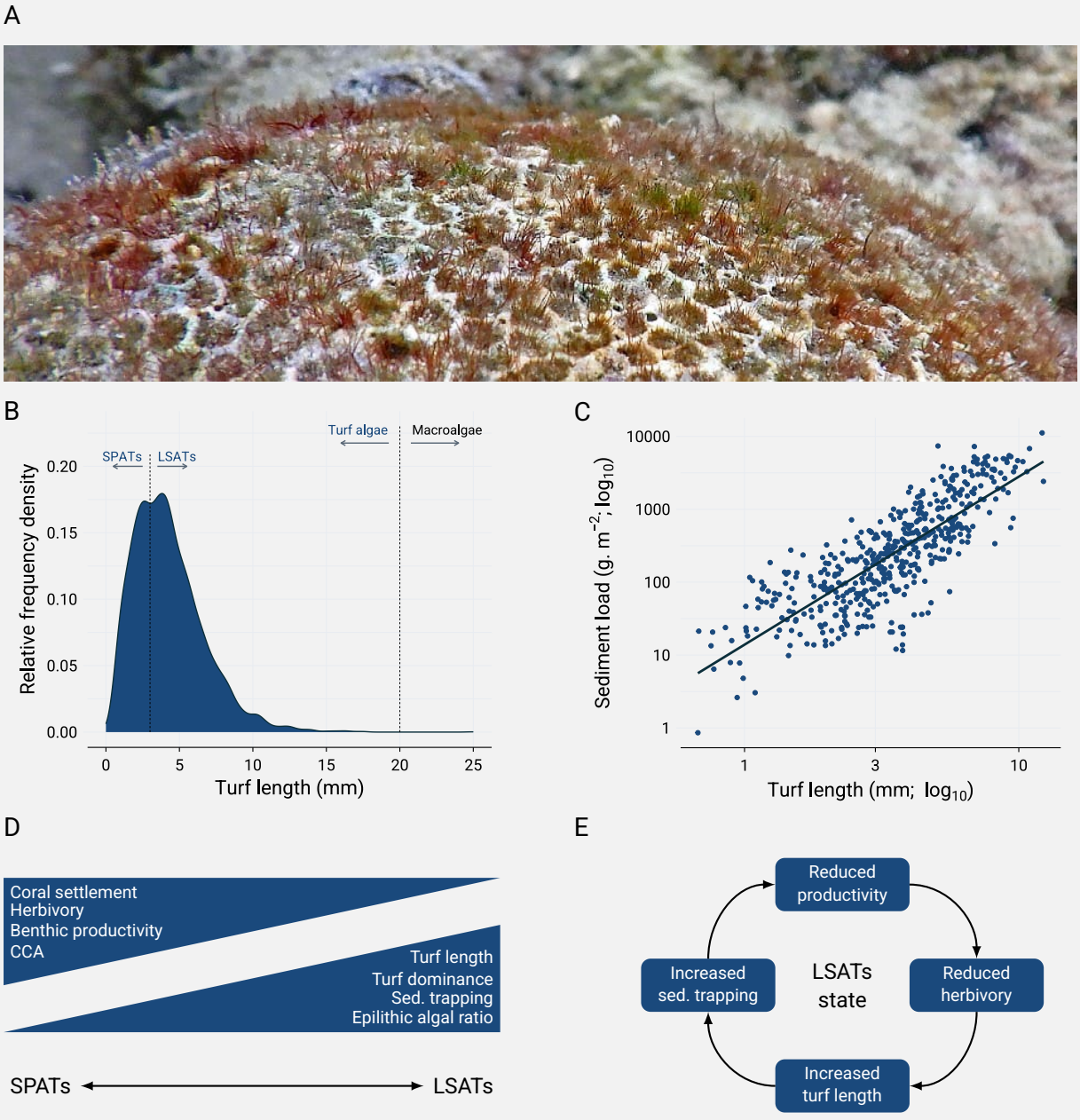
The decline of hard corals on the world's reefs is generally associated with an increase in various groups of algae, especially fleshy macroalgae and turfs. The inverse relationship between coral cover and different algae is underpinned by both indirect (*e.g.* a stressor such as increased sea surface temperature results in coral mortality, creating space and enabling an increase in turfs [131]) and direct (*e.g.* fleshy macroalgae outcompete coral via allelopathic effects [132]) mechanisms. In extreme cases, change in reef condition manifests as a "phase-shift" (*i.e.* a change in the equilibrial community in response to environmental change [133]). On coral reefs, the transition from a hard-coral-dominated state to one dominated by fleshy macroalgae exemplifies a phase-shift [134]. However, this form of phase-shift is relatively rare and often limited to specific contexts (*e.g.* high nutrient inputs and reduced herbivory [135]) or biogeographic locations (*e.g.* the Caribbean [136]). More commonly, a decline in hard corals is associated with a rise of turfs [122].

Turfs are highly abundant on reefs globally, however they are often poorly defined and widely overlooked [137]. Here, turfs are defined as short ($<$ 2 cm) multispecies assemblages of algae and cyanobacteria (Figure C1.1A). At such lengths, the algae and cyanobacteria that comprise turfs are not readily identifiable in situ; with the term "turf" facilitating rapid categorisation in the field. Importantly, this definition is also informed by 1000s of quantitative measurements, with 99.98% of these falling within this 2 cm threshold (Figure C1.1B). Despite this short stature, the structure and composition of turfs at the scale of millimetres can have fundamental implications for the functioning of coral reefs.

Turfs form an ecological interface that directly interacts with a range of processes that are central to coral reef functioning and resilience. Emerging evidence suggests many of these processes are directly mediated by the structure and composition of turfs at the scale of millimetres. For example, a 5 mm increase in turf length can mean a $>$ 1,000% increase in trapped sediments (Figure C1.1C). These sediments can re-shape coral reef functioning as they reduce herbivory rates by 70–90% [138, 139], benthic productivity yields to fishes by 95% [140], and coral settlement rates by 90–99% [141, 142] (Figure C1.1D).

The capacity of turfs to trap sediments is key to how they shape ecosystem functions. Terms such as "short productivity algal turfs" (SPATs) and "long sediment-laden algal turfs" (LSATs) have been coined to recognize this variation [139]. A turf length of ~3 mm and associated sediment load of ~100 g.m^{-2}, represents the nominal transition between SPATs and LSATs. However, strictly defining a threshold is difficult as the state of turfs spans a continuum from crustose coralline algae (CCA) with short sparse turfs through to LSATs, with this continuum recently captured via the development of the epilithic algal ratio (Figure C1.1D, [143]). Given the

high cover of turfs on coral reefs, a subtle, but potentially major, form of phase-shift appears to be a change in the state of the turf itself from SPATs to LSATs. The strong feedback processes involved in this shift (Figure C1.1E) underscore the need for better consideration of turfs in coral reef research, monitoring, and management. In this respect, the current *Status of Coral Reefs of the World* report has taken fundamental steps forward, specifically considering divisions between major algal groups at a global scale. This report, therefore, generates knowledge that is vital to understanding coral reef change at regional and global scales.



▲ **Figure C1.1.** Short productive turf growing on dead coral (A). The frequency distribution of turf length (B, *n* = 6,133 measurements), showing the approximate transition point between short productive algal turfs (SPATs) and long sediment-laden algal turfs (LSATs) at ~3 mm. The positive relationship between turf length and the load of accumulated turf sediments (C, *n* = 497). Key processes and benthic characteristics that can vary along a gradient of turf states (D). An example of a feedback loop which may stabilise an LSAT state (E). Data were based on those compiled in Tebbett *et al.* [122], with additional data from Tebbett *et al.* [143, 144].

CASE STUDY 2

Beyond coral cover: A new framework to assess the condition and resilience of coral reef habitats

Authors

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Introduction

Ecosystem management aims to maintain healthy and resilient ecosystems over time. These attributes can be summarised under a general term widely used in management, *ecosystem condition*, which defines the overall quality of an ecosystem relative to a reference state. However, measuring ecosystem condition remains a challenge.

Hard coral cover is a widely used indicator of coral reef condition because it is easily measured and understood by a non-technical audience and has been consistently collected worldwide over decades. Nonetheless, a comprehensive assessment of coral reef condition requires (1) indicators that describe the multiple processes driving the resistance and recovery capacity, (2) habitat-specific thresholds that define their reference condition, (3) accounting for their spatial and temporal dynamics, and (4) summarising information to classify habitat condition and assess management effectiveness strategies.

This case study outlines a newly developed framework that integrates five indicators derived from conventional monitoring metrics to assess and communicate the condition of coral reef habitats (Table C2.1). Collectively, these indicators assess the resilience capacity of coral reefs, assist in interpreting monitoring data, support the evaluation of management targets and provide a prognostic assessment of ecosystem condition using reference values (for more information see Gonzalez-Rivero *et al.* [145]).

Two examples from the Great Barrier Reef

Drawing on over three decades of monitoring data from the Australian Institute of Marine Science in the Great Barrier Reef (GBR), the following examples demonstrate the value of considering multiple indicators for assessing reef condition.

The fringing reefs of Snapper Island, northern GBR were severely impacted by a cyclone (2014) and bleaching (2017), leaving coral cover low by 2018 (Figure C2.1A). The framework analysis, conducted for 2018, shows underperformance relative to baselines, with high macroalgal

prevalence, low juvenile coral abundance, and taxonomic compositional changes, indicating a compromised reef condition in 2018 ("Warning II"; Figure C2.1B). Indeed, continued monitoring until 2023 showed only slight recovery (Figure C2.1A, grey lines). In contrast, while Lady Musgrave Reef, southern GBR, was also heavily impacted by disturbances in 2008–2010, it maintained expected values for most indicators after the impact, despite low coral cover, suggesting a less compromised condition ("Watch"). Six years post-impact, the reef had regained high coral cover, demonstrating strong resilience (Figure C2.1C, grey lines).

▼ **Table C2.1.** List of indicators included in the framework with a brief rationale for their inclusion.

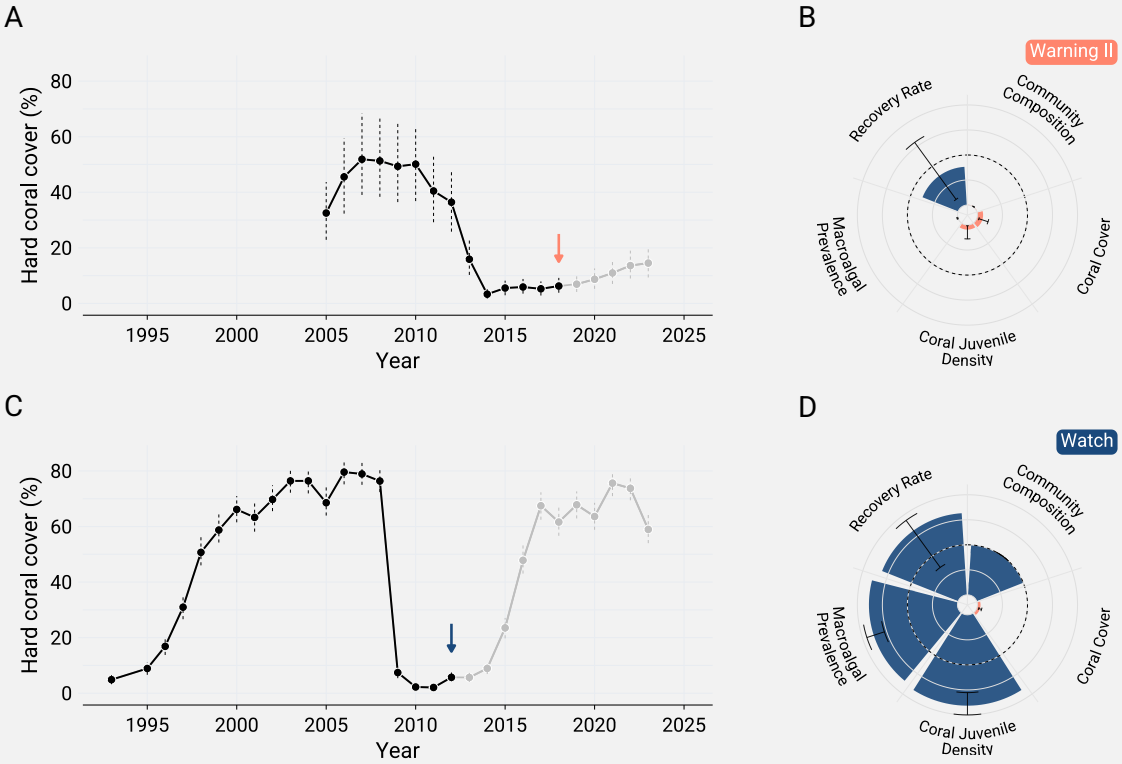
Indicator	Metric	Ecological interpretation	Reported by the GCRMN
Coral cover	Proportion of the reef substrate covered by hard corals	Represents key functional attributes, including reef accretion, structural complexity, abundance of coral-dependent species, and larval production	Yes
Macroalgal prevalence	Proportion of macroalgae relative to total algal cover	An increasing prevalence of macroalgae indicates increased competition for space, reduced herbivory intensity, and/or increased turbidity and nutrient availability	Yes*
Community composition	Relative abundance of benthic taxa within a reef	Indicates changes in the taxonomic representation of the benthic community that may impact ecological functions, successional dynamics, and vulnerability to further disturbances	No
Recovery rate	Change in hard coral cover over time from repeated surveys	Slowing recovery rates are early warnings of an impaired capacity of coral reef habitats to withstand current and future pressures	No
Coral juvenile density	Abundance of hard coral juveniles relative to the available area for recruitment	Represents the ability of coral larvae to settle and survive, thereby sustaining population recovery and long-term reef persistence	No

**Functional groups of algae, including macroalgae, turf algae and crustose coralline algae, have recently been incorporated in GCRMN reports.*

Conclusions and recommendations

Coral reefs are intricate and dynamic systems at the frontlines of climate change. Recognising their complexity while finding effective ways to communicate their condition and resilience is crucial to inform timely and effective conservation efforts. The framework offers a practical way to increase the relevance of monitoring to management, especially for local and regional decision-making. It applies monitoring data analysis for prognostic assessments of coral reef condition, facilitating the prioritisation of reefs for conservation and helping explain complex responses of these ecosystems to disturbances. Importantly, the proposed framework can utilise available data and expertise within a region (sensu Gudka *et al.* [146]), if monitoring remains consistent over time.

Currently, changes in key coral reef attributes besides live hard corals and macroalgae cannot be reliably assessed globally, such as in the GCRMN reports, due to limited indicator data [1]. Therefore, implementing this framework more widely would require strengthening coral reef monitoring (see Obura *et al.* [147]). Over the medium term, collectively advancing towards higher-quality, repeatable and more comprehensive monitoring will be essential to inform conservation in a rapidly changing world.



▲ **Figure C2.1.** Prognostic assessments of coral reef condition in the Great Barrier Reef, including temporal trends in coral cover and indicator values for habitat condition, were evaluated for specific years (represented by the arrows) at Snapper Island (A and B) and Lady Musgrave Reef (C and D). Indicator values below the reference threshold (dashed line) are shown in red, and those at or above are shown in blue. The grey lines in the temporal coral cover plots show the observed trajectories after the evaluated year.

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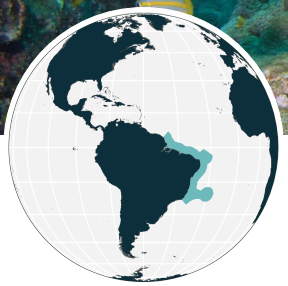
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Smallmouth grunt (*Brachygenys chrysargyrea*) over a coral reef in Fernando de Noronha, Brazil
Credit: Zaira Matheus



BRAZIL

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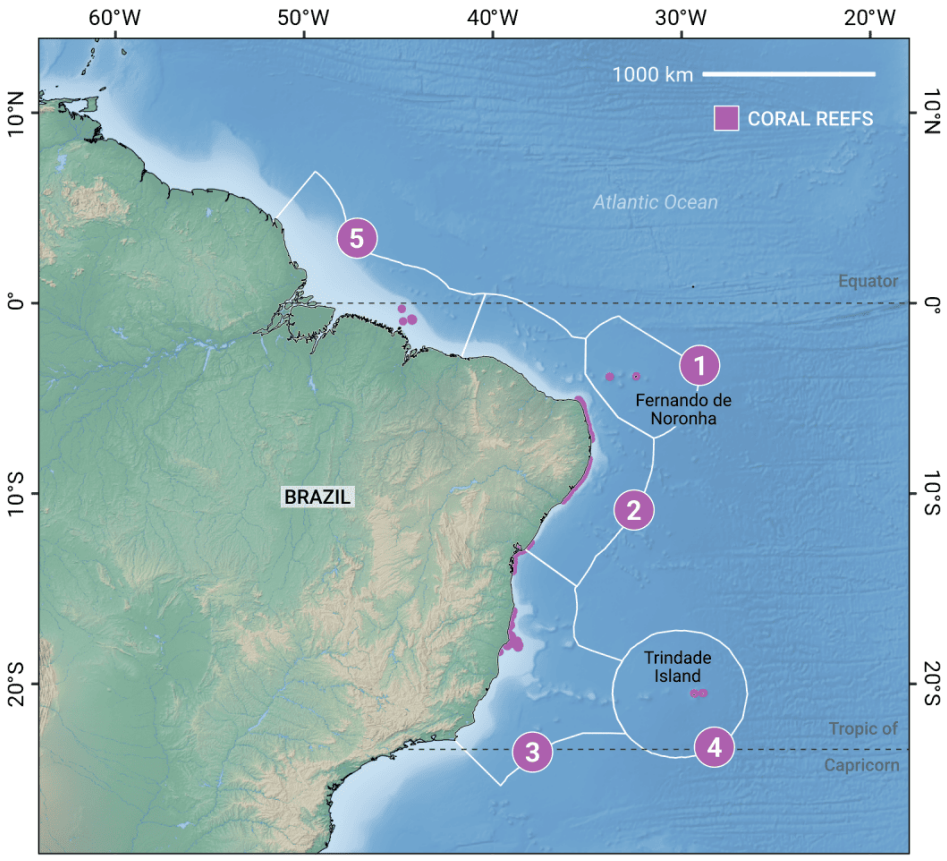
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INTRODUCTION

The GCRMN Brazil region extends over a maritime area of around 2.6 million km^2, ranging across 32° of latitude and 28° of longitude. The region hosts 1,178 km^2 of coral reefs, representing 0.5% of the world's coral reef extent, and is the 9th-largest GCRMN region in terms of coral reef extent. Coral reefs in this region lie entirely within Brazil's exclusive economic zone, with shallow tropical reef formations occurring on oceanic islands and along the coastline, from Parcel de Manuel Luís in the north to the Abrolhos region in the east (Figure 2.2.1). These reefs support a relatively low-diversity coral fauna with a high level of endemism, including many relic species dating back to the Tertiary (Paleogene-Neogene periods). Reef formations are diverse and include shallow bank reefs near the coast, fringing reefs, coral knolls, patch reefs, isolated bank reefs, and coral pinnacles known locally as "chapeirões" [148]. The region also encompasses extensive mesophotic reefs [149], as well as rocky reef systems, such as those found west of Trindade Island [150].

For the purpose of this report, the GCRMN Brazil region was divided into five subregions (Figure 2.2.1). However, subregion 4 (Trindade and Martin Vaz Islands), although monitored, does not host true shallow carbonate-accretive coral reefs and is instead characterised by rocky reefs with encrusting coral colonies [151, 152].



▲ **Figure 2.2.1.** The distribution of coral reefs within the Brazil region and respective subregions. White polygons and numbers represent subregions. Subregion 1 = Fernando de Noronha and Rocas Atoll, Subregion 2 = Northeastern Brazil, Subregion 3 = Eastern Brazil, Subregion 4 = Trindade and Martin Vaz Islands, Subregion 5 = Amazonia. Coral reefs are delineated with a 20 km buffer to enhance interpretation, and only coral reefs within the region are represented. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region.

This subregion was therefore excluded from the analyses. Similarly, subregion 5 (Amazonia), where shallow reefs are represented by the unique Parcel de Manuel Luís formation off Maranhão state [153, 154], lacked long-term monitoring data and was also not included in this report. Subregion 3 (Eastern Brazil) hosts 56.9% of coral reefs in the region (670 km^2), whereas subregion 1 (Fernando de Noronha and Rocas Atoll), hosts only 0.8% of coral reefs in the region (10 km^2, Table A.1).

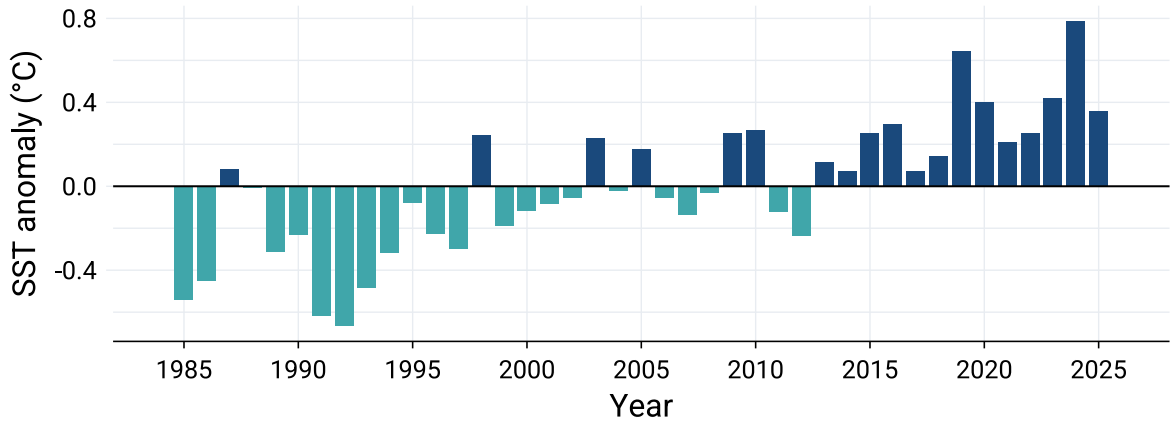
As of 2020, approximately 3.5 million inhabitants were living within 5 km of coral reefs. From 2000 to 2020, the region recorded a 36.3% increase in human populations residing within 5 km of coral reefs, representing an average annual increase of around 50,000 inhabitants. The greatest increase in populations over this period was recorded in subregion 2 (Northeastern Brazil), which also recorded the highest number of inhabitants in 2020 (Table A.2).

THREATS AND DISTURBANCES

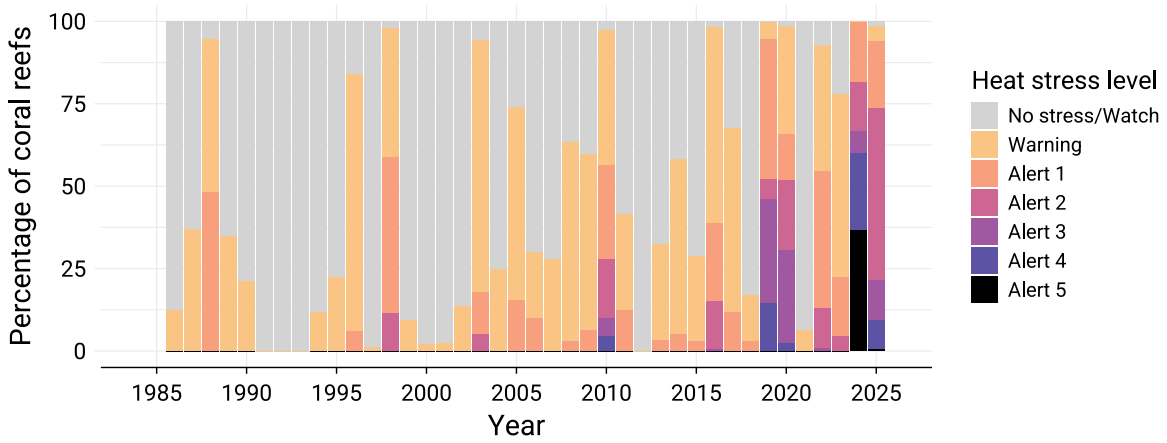
Heat stress

From 1985 to 2025, the long-term average of sea surface temperature (SST) recorded over coral reefs within the Brazil region was 26.65 °C. During this period, the SST over coral reefs in the region increased by 0.76 °C, corresponding to an average warming rate of 0.19 °C per decade (Table A.3).

Between 1986 and 2025, major episodes of heat stress affecting Brazil coral reefs occurred in 1998, 2010, 2016, 2019–2020, and 2024–2025. Among these, four correspond to global bleaching events (1998, 2010, 2016, and 2024) and one regional bleaching event (2019–2020). Although the intensity and impacts varied across subregions, the events of 2019, 2020, and 2024 rank among the most extreme heat-stress events ever recorded in the region (Figure 2.2.3).



▲ **Figure 2.2.2.** Annual mean sea surface temperature (SST) anomalies (°C) over the coral reefs of the Brazil region from 1985 to 2025. SST anomalies were calculated based on the long-term average over the period 1985–2025. Positive values (red bars) indicate SST warmer than the long-term average, whereas negative values (blue bars) indicate SST cooler than the long-term average. Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).



▲ **Figure 2.2.3.** Percentage of the coral reefs within the Brazil region experiencing heat stress levels per year, from 1986 to 2025. No stress/Watch (DHW = 0); Warning (possible bleaching, 0 $<$ DHW $<$ 4); Alert 1 (reef-wide bleaching, 4 $\leq$ DHW $<$ 8); Alert 2 (reef-wide bleaching and mortality of heat-sensitive corals, 8 $\leq$ DHW $<$ 12); Alert 3 (multi-species mortality, 12 $\leq$ DHW $<$ 16); Alert 4 (severe multi-species mortality, 16 $\leq$ DHW $<$ 20); and Alert 5 (near-complete mortality, DHW $\geq$ 20). Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).

Regional disturbances

Impacts on coral reefs across the entire Brazil region include mining [155] and the exploitation and transport of oil and gas in nearby areas, which may cause spills, such as the 2019 event [156], and which can act as vectors of invasive species such as the sun coral *Tubastrea* spp. [157]. The invasive lionfish (*Pterois* sp.) has been recorded along the Brazilian coast since 2020 [158], with occurrences in subregion 5 (Amazonia), subregion 3 (Eastern Brazil) and subregion 2 (Northeastern Brazil). To date, Fernando de Noronha Archipelago has recorded the highest lionfish densities, raising concerns about the potential impacts of this species on local biodiversity [159].

The continuous intensification of multiple local pressures, including increased overfishing and water quality deterioration, may reduce reef resilience and compromise post-disturbance recovery potential across the entire region, requiring broader national strategies to address the issues [160–162].

Subregion 1 comprises the Biological Reserve of Rocas Atoll and the Fernando de Noronha Archipelago, both located in oceanic waters that were intensely exploited by fisheries until the establishment of marine protected areas in the late 1970s and 1980s. Rocas Atoll is a no-take no-entry marine protected area that hosts a research station with a maximum occupation of six people. The Fernando de Noronha Archipelago hosts a permanent population of over 3,000 people [163] and receives over 100,000 visitors per year [164]. While tourism is the main economic activity, fishing is permitted in parts of the main island. Inadequate wastewater treatment is a challenge and associated negative effects on water quality have been reported [160].

Subregion 2 (Northeastern Brazil) holds coastal reef formations, located within 5 km from the coast. Chronic stressors include sedimentation resulting from deforestation for sugarcane cultivation, a practice that began during the colonial period, as well as coral extraction for lime production and construction [165]. In response to historical coral degradation and the need to manage growing pressures, marine protected areas were established in the late 1990s (*e.g.* Costa dos Corais environmental protection area) as well as ban of coral removal [166],

which may have contributed to subsequent reef recovery. Today, tourism growth represents one of the main threats to the subregion, with compounding stress from increasing fishing pressure, land-based runoff, and coastal development [161, 167, 168].

Subregion 3 (Eastern Brazil) harbors inshore and offshore reef formations distributed on the extended Abrolhos shelf and encompasses the largest and richest coral reefs in the region [169]. The first marine protected area in subregion 3 was established in the 1980s (*e.g.* Parque Nacional Marinho dos Abrolhos). Stressors in the subregion include dredging, coastal development, and overfishing, including illegal fishing [170]. In 2015, a dam collapse reached the southern portion of the region through the Doce river [171, 172] with associated metal contamination detected in corals [173].

STATUS OF CORAL REEFS IN THE REGION

Hard coral cover

Pre-monitoring scenario (prior to 2002)

While the modelling output provides estimates from 2002 onwards, coral bleaching events were also recorded in 1993 [174] and 1998 [175]. The first documented bleaching event occurred during the austral summer of 1993–1994 in Eastern Brazil (subregion 3), when 50–90% of coral colonies were affected [174]. Subsequent reports only emerged in 1998 along the Brazilian coast, during the first global bleaching event [153, 175]. Due to the limited number of established reef monitoring programs in Brazil at the time, the national-scale impact of this event was not quantitatively assessed. Nevertheless, significant coral bleaching and some mortality rates, for sensitive species, were reported in regions such as Fernando de Noronha and Rocas Atoll (subregion 1), Tamandaré (within subregion 2) (Ferreira, B.P., *pers. obs.*; [165]), Abrolhos (within subregion 3) [148, 175] and Parcel de Manuel Luís (within subregion 5) [153].

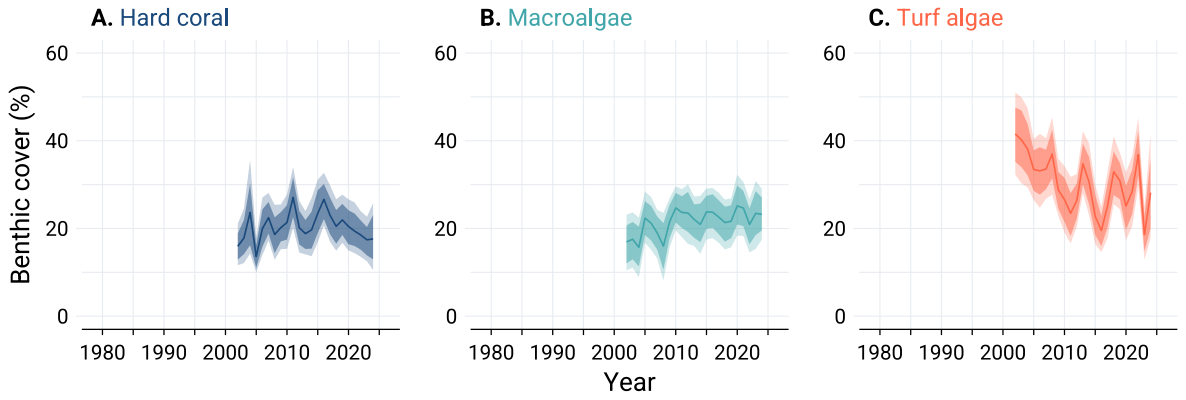
In coastal reefs (located in subregion 2 and subregion 3), chronic pressures such as sedimentation and the removal of corals contributed to a decline in hard coral cover until the 1990s [165, 175, 176]. In the 1970s, Laborel [169] reported estimated coral cover values of approximately 20–25% for subregion 2 (Northeastern Brazil) and up to 50% for subregion 3 (Eastern Brazil), indicating a marked decline compared with the hard coral cover recorded in 2002 (4.8% and 24.2%, respectively). Therefore, the beginning of the temporal trends presented in Figure 2.2.4 likely reflect reef conditions that were already experiencing some degree of degradation from multiple stressors.

Partial recovery and stabilization (2002–2016)

From the 1980s onward, five major marine protected areas were established across the studied subregions, with full implementation achieved by the 2000s. The positive effects of these and other management measures, combined with the absence of severe and widespread coral mortality associated with bleaching events after 1998, may have created favorable conditions for coral recovery. Indeed, hard coral cover in the region showed a relative increase of 66.9% between 2002 and 2016, rising from 15.9% (credible interval at 95% [$CrI_{95\%}$]: 11.6 – 21.1%) to 26.6% ($CrI_{95\%}$: 20.7 – 32.7%). This trend likely reflects the synergistic effects of successful management and relatively low heat stress during this period [177].

Sharp decline associated with repeated and intense bleaching events (2016–2024)

Although all three subregions experienced a notable increase in the intensity and



▲ **Figure 2.2.4.** Modeled temporal trends of benthic cover in the Brazil region from 1980 to 2024 for hard coral (A), macroalgae (B), and turf algae (C). The solid line represents the estimated median benthic cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

frequency of significant heating events during 2016–2024 [177], the timing and maximum intensity of these events varied among subregions, as did the responses of benthic communities (see the Status of coral reefs in the subregions section). Nevertheless, trends in the region reflected the impact of the combined strong regional heat-stress event of 2019–2020 and the fourth global bleaching event of 2024, leading to a relative decline in hard coral cover of 33.8% between 2016 and 2024, decreasing from 26.6% (CrI $_{95\%}$: 20.7 – 32.7%) to 17.6% (CrI $_{95\%}$: 10.5 – 25.8%).

Macroalgal and turf algal cover

Macroalgal and turf benthic cover varied among subregions in both magnitude and temporal trends. These differences likely reflect multiple interacting factors, including latitude, distance from shore, local stressors, depth and monitoring zonation, as well as seasonality and species composition [83, 178].

Increase in macroalgal cover and decline in turf algae (2002–2010)

Between 2002 and 2010, macroalgal cover showed a relative increase of 46.1%, rising from 16.9% (CrI $_{95\%}$: 10.5 – 23.1%) to 24.7% (CrI $_{95\%}$: 19.6 – 29.6%). In contrast, turf algal cover showed a declining trend over the same period, with a relative decline of approximately 36.2%, ranging from 41.5% in 2002 (CrI $_{95\%}$: 32.3 – 51.0%) to 26.6% in 2010 (CrI $_{95\%}$: 19.1 – 34.2%) (Figure 2.2.4C).

Relative stabilization in macroalgal cover (2010–2024) and increase in turf algae (2016–2024)

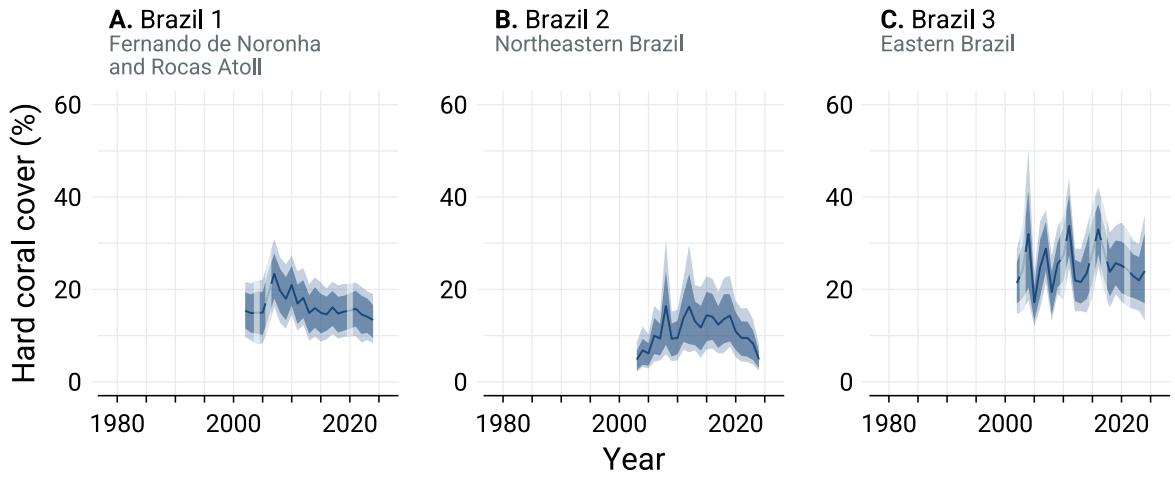
A relatively stable macroalgal cover across the region was observed from 2010 to 2024. This trend suggests this benthic category has not undergone major changes despite the intensification of thermal anomaly events at a regional scale [177]. For turf algal cover, the data show evidence of an increase between 2016 and 2024, rising from 19.5% (CrI $_{95\%}$: 14.6 – 24.9%) to 28.2% (CrI $_{95\%}$: 17.7 – 41.6%), corresponding to a 43.4% relative increase. This increase was likely associated with the rapid colonization of newly exposed substrates following coral mortality after successive bleaching events [131, 179].

STATUS OF CORAL REEFS IN THE SUBREGIONS

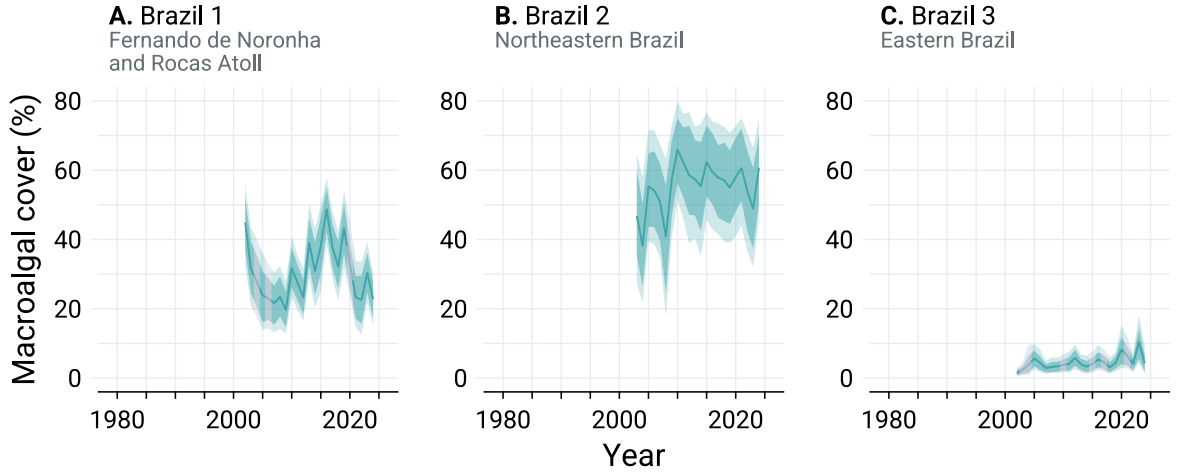
Brazil 1 - Fernando de Noronha and Rocas Atoll

Subregion 1 (Rocas Atoll and Fernando de Noronha) experienced changes in hard coral cover primarily driven by thermal anomaly events over the past decades. Although the impacts of the 1998 El Niño were not monitored, local accounts report widespread bleaching, which may help explain the recovery trajectory observed at the beginning of the study period in 2002. This is inferred from a relative increase of 36.4% between 2002 and 2010, with hard coral cover rising from 15.4% (CrI $_{95\%}$: 9.7 – 21.6%) to 21.0% (CrI $_{95\%}$: 14.5 – 27.4%). In 2010, a significant heat stress event exposed all coral reefs in the subregion to more than 4 DHW leading to mass bleaching and mortality [180]. Consecutive heat stress events occurred in 2019 and 2020, where all coral reefs were exposed to more than 4 DHW, and 8 DHW, respectively. In 2024, the most intense heat stress event on record in this subregion occurred, exposing all coral reefs to more than 16 DHW. Together, these events led to a relative decline of 35.7% in hard coral cover between 2010 and 2024, resulting in a hard coral cover of 13.4% (CrI $_{95\%}$: 8.4 – 19.0%) in 2024 (Figure 2.2.5A).

In contrast, macroalgal cover in subregion 1 (Rocas Atoll and Fernando de Noronha) showed no association with thermal anomaly events. A relative decline of 29.6% was observed between 2002 and 2010, with macroalgal cover decreasing from 44.9% (CrI $_{95\%}$: 32.9 – 55.9%) to 31.6% (CrI $_{95\%}$: 23.4 – 41.0%). This was followed by an increase in macroalgal cover until 2016, reaching levels similar to those recorded in 2002. After 2016, macroalgal cover declined again by 53.4% through 2024, reaching a macroalgal cover of 22.7% (CrI $_{95\%}$: 15.2 – 30.4%) by 2024 (Figure 2.2.6A). Turf algal cover fluctuated substantially over the period, averaging around 26.3% (CrI $_{95\%}$: 13.2 – 42.1%) (Figure 2.2.7). These observed patterns in macroalgae and turf algal cover may have been influenced by varying depths and pronounced seasonality in the subregion, coupled with the mismatch between monitoring efforts and seasonal cycles.



▲ **Figure 2.2.5.** Modeled temporal trends of hard coral cover in Brazil subregions from 1980 to 2024. The solid line represents the estimated median hard coral cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

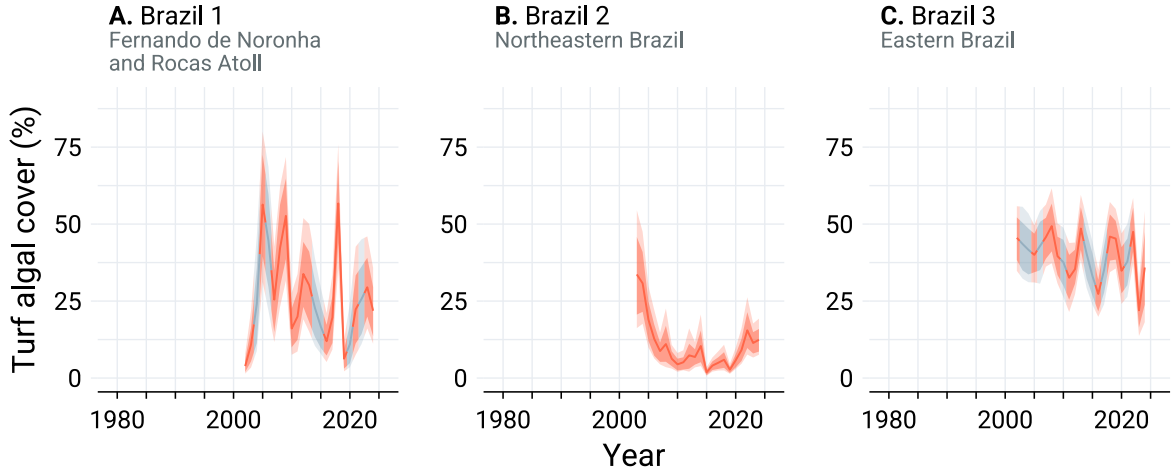


▲ **Figure 2.2.6.** Modeled temporal trends of macroalgal cover in Brazil subregions from 1980 to 2024. The solid line represents the estimated median macroalgal cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

Brazil 2 - Northeastern Brazil

Formed by coastal reefs, subregion 2 (Northeastern Brazil) experienced declines in hard coral cover before 2002 when monitoring began, due to chronic impacts including land run-off, coral collection, and overexploitation of fisheries resources, as well as impacts from the 1998 El Nino, which exposed 90% of coral reefs to more than 4 DHW. These impacts were particularly felt by reefs closer to the coast [176, 181]. Following the establishment of a large marine protected area in the 1990s, the Costa dos Corais environmental protection area, and in the absence of severe bleaching events, hard coral cover showed a recovery trend, with a relative increase of approximately 97.1% from the 2003 to 2010, rising from 4.8% (CrI $_{95\%}$: 2.1 – 8.7%) to 9.5% (CrI $_{95\%}$: 4.6 – 15.6%). Despite a global coral bleaching event recorded in 2010, which exposed around 70% of coral reefs to more than 4 DHW, hard coral cover did not show evidence of change between 2003–2009 and 2020–2024. In 2019 and 2020, consecutive heat stress events exposed around 80% of coral reefs to more than 8 DHW, causing mass bleaching and high mortality in sensitive coral species such as *Millepora alcicornis* [161]. These events were associated with a 35.6% relative decline in hard coral cover between the 2010s and 2020s. Without sufficient time for hard coral cover recovery, Northeastern Brazil experienced the most intense heat stress recorded among all monitored subregions in 2024, with all coral reefs exposed to more than 12 DHW. Consequently, between 2019 and 2024, hard coral cover declined by 66.2%, dropping from 14.3% (CrI $_{95\%}$: 7.6 – 22.9%) to 4.8% (CrI $_{95\%}$: 2.3 – 8.0%) (Figure 2.2.5B).

Macroalgae have historically dominated benthic cover in the Northeastern Brazil subregion [167]. Macroalgal cover showed a relative increase of 19.8% from 2000s to 2010s. From 2019 to 2024, a relative increase of 9.6% was detected, with cover rising from 55.0% (CrI $_{95\%}$: 39.0 – 70.8%) to 60.7% (CrI $_{95\%}$: 45.2 – 74.5%) (Figure 2.2.6B). By contrast, turf algal cover showed a relative decline of 68.7% in trends between the 2000s and 2010s, possibly associated with reduced grazing pressure by sea urchins [182]. Following recent bleaching events, turf algal cover exhibited a relative increase between 2016 and 2024, rising from 4.0% (CrI $_{95\%}$: 2.1 – 6.7%) to 12.4% (CrI $_{95\%}$: 7.6 – 19.3%) (Figure 2.2.7B). This increase may be associated with coral mor-



▲ **Figure 2.2.7.** Modeled temporal trends of turf algal cover in Brazil subregions from 1980 to 2024. The solid line represents the estimated median turf algal cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

tality, as turf algae are typically among the first colonizers of recently dead coral skeletons [179].

Brazil 3 - Eastern Brazil

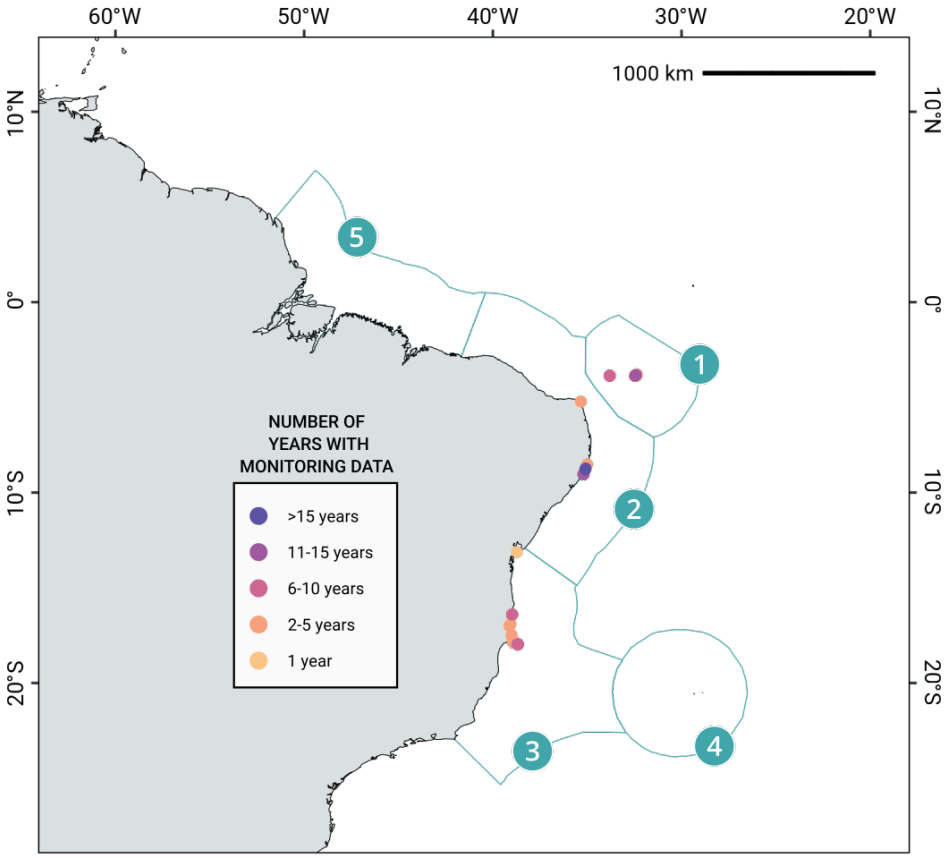
Subregion 3 (Eastern Brazil) encompasses several locations monitored at different periods, from inshore to mid shelf reefs, including reef banks, pinnacles ("chapeirões") and island fringing formations. Hard coral cover in this subregion exhibited high fluctuation between 2002 and 2016, with a relative increase of 54.2%, rising from 21.4% ($CrI_{95\%}$: 14.7 – 28.8%) to 33.0% ($CrI_{95\%}$: 25.0 – 42.0%). No mass bleaching events associated with high subsequent coral mortality were reported in the subregion during this period [183], despite around 40% of coral reefs being exposed to more than 4 DHW in 2003 and 2010. The first record of severe bleaching occurred in 2016, where almost 80% of coral reefs in the subregion were exposed to more than 4 DHW, marking the onset of a continuous decline in hard coral cover in subsequent years [162, 183]. The most intense thermal anomaly was observed in 2019 [131, 179], when 95% of coral reefs were exposed to more than 12 DHW, producing a relative decline in hard coral cover of 22.0% between 2016 and 2019, with hard coral cover decreasing from 33.0% ($CrI_{95\%}$: 25.1 – 42.0%) to 25.7% ($CrI_{95\%}$: 18.9 – 33.9%). A similar decline in hard coral cover of approximately 15% was also reported by Carlos-Júnior *et al.* [162] in the subregion, along with shifts in coral assemblages, including the collapse of branching corals and an increase in weedy species. In 2024, around 60% of coral reefs were exposed to more than 8 DHW, but with no reports of severe bleaching or mortality [184]. After 2019, there was no evidence of a change in hard coral cover, with median hard coral cover estimated at 24.0% ($CrI_{95\%}$: 13.4 – 36.2%) in 2024 (Figure 2.2.5C).

Unlike subregions 1 and 2, macroalgal cover does not dominate total benthic cover in Eastern Brazil, which is instead largely represented by turf algae. There was strong evidence that macroalgal cover increased between 2002 and 2016, rising from 1.5% ($CrI_{95\%}$: 0.6 – 2.8%) to 5.3% ($CrI_{95\%}$: 2.4 – 9.4%). There was no evidence of change in macroalgal cover between 2016 and 2024 (Figure 2.2.6C). Other studies in the region have reported high spatial heterogeneity in macroalgal cover, with con-

sistently low values (<5%) at offshore sites and pinnacle walls, as well as a region-wide decline between 2006 and 2017–2018 followed by relative stability until 2023 [162]. Turf algal cover, in turn, showed a relative decline of 40.0%, with cover decreasing from 45.5% ($CrI_{95\%}$: 34.8 – 55.8%) in 2002 to 27.3% ($CrI_{95\%}$: 19.9 – 35.2%) in 2016 (Figure 2.2.7C).

DESCRIPTION OF MONITORING DATA

Temporal trends in the percentage cover of benthic categories were estimated using data from four datasets encompassing 77 monitoring sites. These data were collected through 488 surveys conducted between 2002 and 2025, with 90% of surveys taking place after 2006 (Figure A.1).



▲ **Figure 2.2.8.** Spatio-temporal distribution of benthic cover monitoring sites across the Brazil region and respective subregions. Light blue polygons and numbers represent subregions. Subregion 1 = Fernando de Noronha and Rocas Atoll, Subregion 2 = Northeastern Brazil, Subregion 3 = Eastern Brazil, Subregion 4 = Trindade and Martin Vaz Islands, Subregion 5 = Amazonia. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region.

Of all surveys carried out in the region between 2002 and 2025, 37.1% (representing 40.3% of surveyed sites) were conducted in subregion 2 (Table A.4). Differences between the *Status of Coral Reefs of the World: 2020* GCRMN report [30] and this report are due to the inclusion of different datasets with different monitoring periods, which increased the number of sites in subregion 3 (Eastern Brazil).

CONCLUSIONS

The regional and subregional patterns of benthic cover trends in Brazilian coral reefs point to broader ecosystem transformations over the past three decades and provide insights into reef recovery potential as well as current and future risks. The absence of reported high coral mortality rates following global bleaching events [31], that nevertheless reached up to 16 DHWs in 2010, led to the belief that the region was a potential climate refugium and that local corals were less susceptible [185]. The subsequent regional thermal anomalies of 2019–2020 and the global bleaching event of 2024 however, leading to an overall reduction of approximately 34% in hard coral cover by 2024, highlighted the vulnerability of Brazilian reefs to recurrent and intense thermal anomalies [184].

Subregions showed distinct trends in hard coral and algal cover, reflecting varied responses to climate change and local pressures. Hard coral cover responses are linked to species composition, as taxa differ in thermal tolerance. The Eastern Brazil subregion, with higher hard coral cover and richer species composition [148], experienced smaller declines but reports indicated shifts in coral assemblages, including mortality of branching and massive endemic forms and an increase in "weedy" fast-growing species [162, 179].

Macroalgal cover generally increased across the region, while turf algal cover decreased, with temporary spikes likely related to post-bleaching coral mortality. Macroalgal cover is higher closer to the coast and in shallow areas, notably in subregion 2 (Northeastern Brazil). In subregion 1 (Fernando de Noronha and Rocas Atoll) and subregion 3 (Eastern Brazil), varied depths, distances from the coast, and exposure levels complicated the detection of response thresholds. Future monitoring should establish location-specific thresholds accounting for natural gradients (*e.g.* latitude, shelf position) and seasonality, and include more functional/morphological groups to enhance the resolution of algae as reef health indicators [83].

Despite the clear effects of climate change, marine protected areas are vital to maintaining key ecological processes that support coral recovery after extreme events. Enhancing reef resilience however will require reducing local stressors, improving coastal management, and limiting ocean warming, supported by long-term monitoring to guide adaptive management [186–188].

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DATA CONTRIBUTORS

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Access to online supplementary materials

The modelled benthic cover data, detailed outputs from the data analyses, and the CRediT author contribution statement are available on Zenodo.

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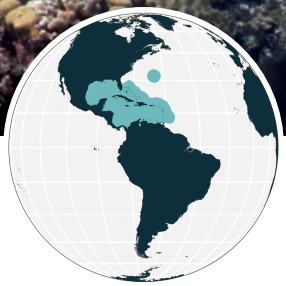
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Coral reef in Puerto Morelos, Mexico
Credit: Lorenzo Álvarez-Filip



CARIBBEAN

Authors

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Citation

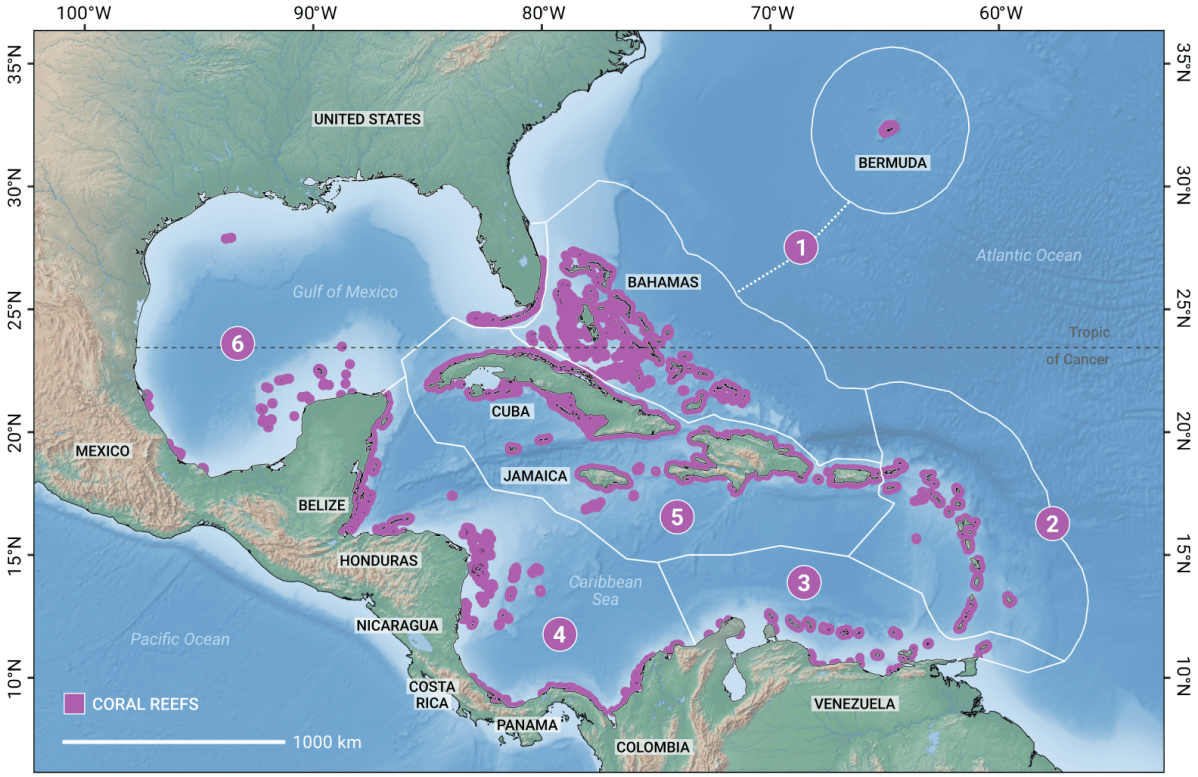
Mathon, L., Rivera-Sosa, A., McField, M., O'Brien, A. H., *et al.* (2026). Chapter 3 – Caribbean. *In González-Rivero, M., et al. Status of Coral Reefs of the World: 2025. Global Coral Reef Monitoring Network and International Coral Reef Initiative.* https://doi.org/10.59387/LFPR6347.CAR

INTRODUCTION

The GCRMN Caribbean region extends over a maritime area of approximately 6.4 million km^2. Ranging across 28° of latitude and 42° of longitude, it encompasses a wide range of oceanographic conditions, reef types, and disturbance regimes. With 24,671 km^2 of coral reefs, it is the fourth-largest GCRMN region by coral reef extent, representing about 10% of the world's coral reef extent. Coral reefs can be found in the territorial waters of 44 countries and territories, from Bermuda in the North to Colombia in the South, and from Barbados in the East to Mexico in the West (Figure 2.3.1).

For this report, the Caribbean region was divided into six subregions (Figure 2.3.1). The largest, subregion 5 (Greater Antilles), hosts 32.8% of Caribbean coral reefs (8,089 km^2) whereas the smallest, subregion 2 (Eastern Caribbean), hosts only 7% of Caribbean coral reefs (1,709 km^2, Table A.1).

As of 2020, a total of 15.7 million inhabitants lived within 5 km of coral reefs, representing a 43.5% increase since 2000 (approximately 238,000 additional inhabitants per year). The greatest rate of population increase over this period was recorded in subregion 5 (Greater Antilles), which also had the largest number of inhabitants in 2020 (Table A.2).



▲ **Figure 2.3.1.** The distribution of coral reefs within the Caribbean region and respective subregions. White polygons and numbers represent subregions. Subregion 1 = Bahamas and Bermuda, Subregion 2 = Eastern Caribbean, Subregion 3 = Southern Caribbean, Subregion 4 = Western Caribbean, Subregion 5 = Greater Antilles, Subregion 6 = Florida and Gulf of Mexico. Coral reefs are delineated with a 20 km buffer to enhance interpretation, and only coral reefs within the region are represented. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region.

THREATS AND DISTURBANCES

Heat stress

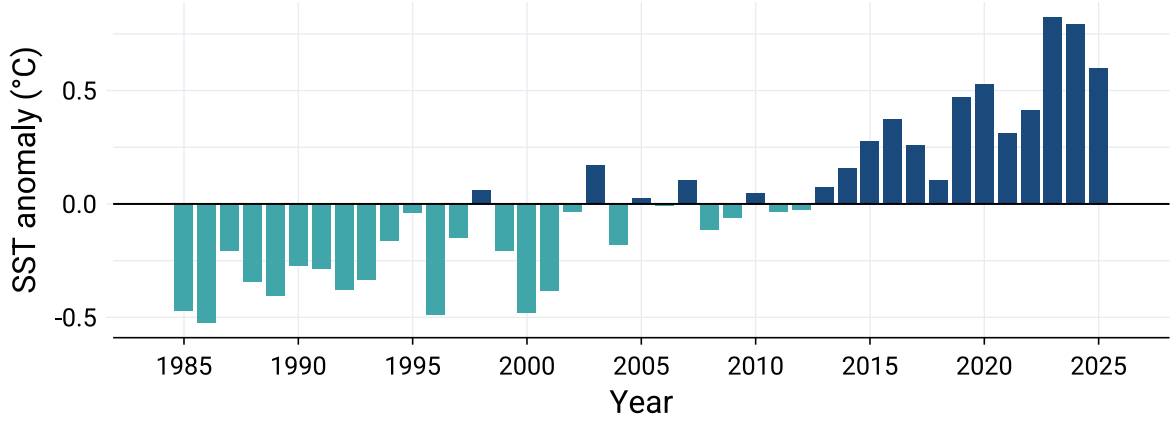
Annual mean sea surface temperature (SST) anomalies (°C) show an increasing trend from 1985 to 2025 (Figure 2.3.2). In addition, maximum degree heating weeks (DHW) show increasingly frequent and more severe bleaching alerts over time, culminating in extreme region wide stress during 2023–2024. From 1985 to 2025, the SST over coral reefs in the region increased by 1.03 °C, corresponding to an average warming rate of 0.26 °C per decade (Table A.3).

Major heatwaves in 1987, 1995, 1998, 2005, 2010, 2015–2017, 2019–2020, and 2023–2025 caused bleaching of varying extent. The 1998 event had regionwide impacts (especially severe in the Bahamas), while later events showed spatial variability: 2005 affected the Greater Antilles and Eastern Caribbean, 2010 the Southern Caribbean, and 2015–2017 the Gulf of Mexico and Western Caribbean [189]. The 2023–2024 phase of the fourth global bleaching event caused severe, widespread bleaching and mortality, with DHW exceeding 20 °C-weeks in some areas and exposure to bleaching-level stress for almost all Caribbean reefs in 2023 ([47], Figure 2.3.3). Impacts on *Acropora* spp. corals are especially alarming, with reports of catastrophic losses and local extirpations despite their previously lower bleaching susceptibility [190–192].

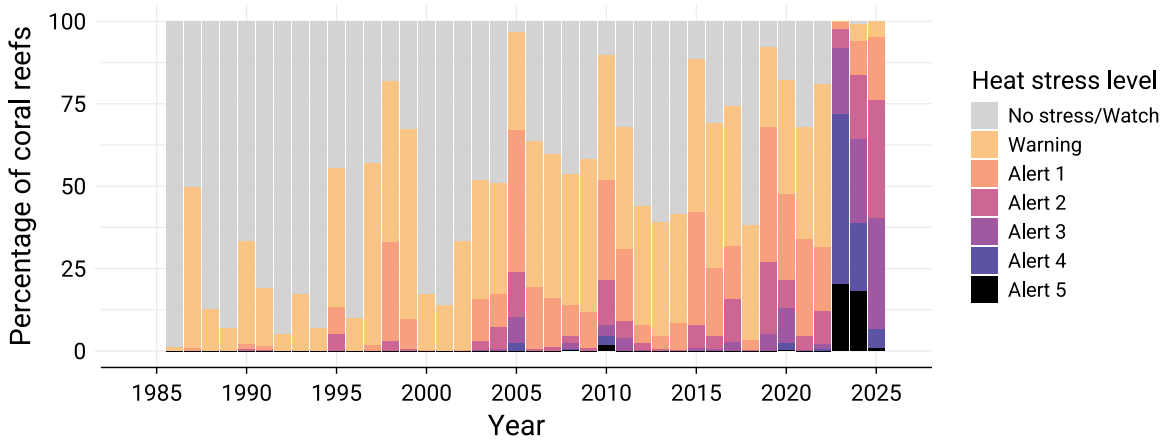
Hurricanes

In the Caribbean region, cyclones are known as hurricanes. Increasingly intense hurricanes have contributed to declines in Caribbean reef structural complexity and coral cover. Between 1980 and 2025, 180 hurricanes passed within 100 km of reefs, an average of 4 $\pm$ 2.5 per year ([108], Figure 2.3.4). Impacts of hurricanes are uneven across the region, with storms typically tracking through the Lesser Antilles and northern Caribbean. For instance, the Bahamas experienced 54 hurricanes (1980–2024), while Costa Rica and Panama recorded only one [193, 194].

Hurricane intensity has increased in recent decades, particularly after 2005, likely linked to climate change [195].



▲ **Figure 2.3.2.** Annual mean sea surface temperature (SST) anomalies (°C) over the coral reefs of the Caribbean region from 1985 to 2025. SST anomalies were calculated based on the long-term average over the period 1985–2025. Positive values (red bars) indicate SST warmer than the long-term average, whereas negative values (blue bars) indicate SST cooler than the long-term average. Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).



▲ **Figure 2.3.3.** Percentage of the coral reefs within the Caribbean region experiencing heat stress levels per year, from 1986 to 2025. No stress/Watch (DHW = 0); Warning (possible bleaching, 0 < DHW < 4); Alert 1 (reef-wide bleaching, 4 ≤ DHW < 8); Alert 2 (reef-wide bleaching and mortality of heat-sensitive corals, 8 ≤ DHW < 12); Alert 3 (multi-species mortality, 12 ≤ DHW < 16); Alert 4 (severe multi-species mortality, 16 ≤ DHW < 20); and Alert 5 (near-complete mortality, DHW ≥ 20). Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).

Notable high-intensity events include Hurricane Allen (1980; 306 km.h^{-1}), Hurricane Irma (2017; 287 km.h^{-1}), and Hurricane Maria (2017; 260 km.h^{-1}) (Figure 2.3.5). Even single storms can reduce reef structural complexity [196], while stronger hurricanes drive abrupt declines in live coral cover [197].

Diseases

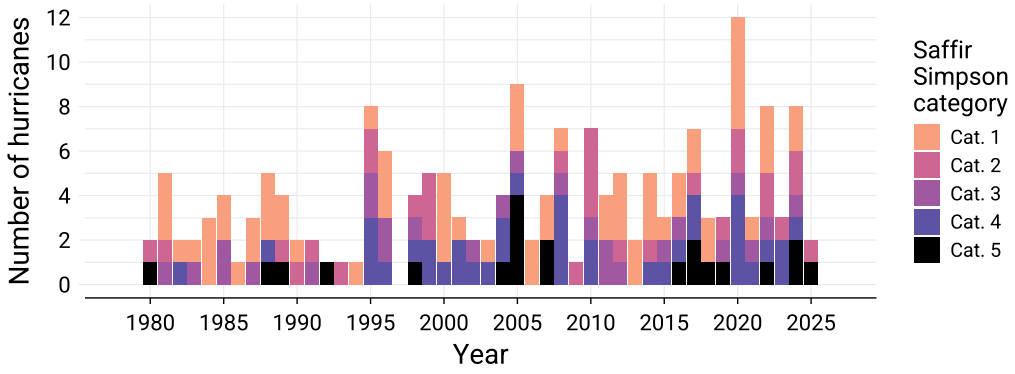
Coral diseases are a major driver of reef decline in the Caribbean, causing rapid and widespread mortality [197]. In the late 1970s, white-band disease devastated key *Acropora* spp. populations. Since then, outbreaks of white plague and yellow-band disease have further affected reef-building corals such as *Orbicella* spp. [198]. The emergence of Stony Coral Tissue Loss Disease (SCTLD) in 2014 triggered an unprecedented regional crisis, leading to severe coral losses across multiple subregions since 2019 [199, 200]. Additionally, mass die-offs of long-spined sea urchin (*Diadema antillarum*) in 1983–1984 and again in 2022 reduced herbivory, promoting algal overgrowth and weakening reef resilience [201].

Coastal development

Rapid coastal population growth, tourism, and infrastructure expansion since the 1980s have intensified pressure on Caribbean reefs. Resort and port construction have caused direct damage while increasing sedimentation and nutrient inputs [202]. Agriculture, deforestation, and inadequate wastewater treatment have elevated nitrogen and phosphorus runoff. Major rivers, including the Magdalena, Amazon, Orinoco, and Mesoamerican river systems, contribute to sedimentation and eutrophication [203, 204]. Pollution has also increased, with plastics dominating marine debris and chemical residues accumulating in reef environments, further degrading water quality and coral health [205].

Unsustainable fishing

Fishing pressure has intensified over time, depleting predatory fish such as groupers and snappers and collapsing spawning aggregations [206]. In response, over 70% of Caribbean fisheries nations now harvest parrotfish species, a key group of herbivores, disrupting food webs and enabling



▲ **Figure 2.3.4.** Number of hurricanes that passed within 100 km of the coral reefs of the Caribbean region from 1980 to 2025. Colors correspond to the Saffir–Simpson category of the hurricane along its entire track. Derived from IBTrACS data (Knapp *et al.* [118]; see Materials and Methods).

the growth of macro- and turf algae [207]. Invertebrate fisheries targeting spiny lobsters and queen conch are also heavily exploited, often including illegal harvest. Fully protected areas (no fishing), have proven effective in supporting fish population recovery [208, 209].

Sargassum

Since 2011, the Caribbean coral reefs are experiencing a new, potentially irreversible, regime shift triggered by the massive recurring influx of *Sargassum fluitans* and *S. natans* [210]. These massive influxes are characterized by reduced light availability, severe hypoxia, localized water acidification, and the release of toxic leachates and gases [211, 212]. In this new regime, mass mortalities of marine organisms are more frequent, resetting their ecological succession under lower recruitment conditions, and negatively affecting human health [213, 214].

Invasive species

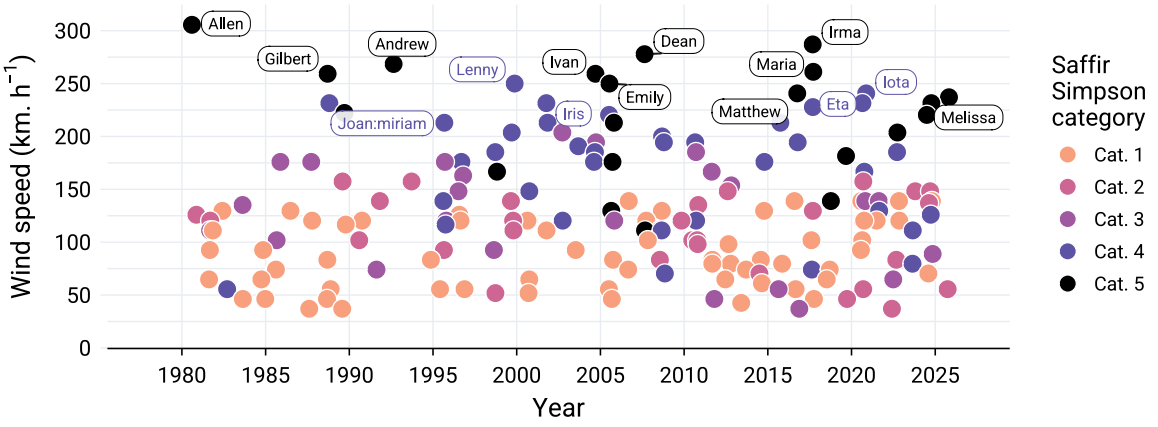
Invasive species such as lionfish (*Pterois* spp.) and invertebrates like *Unomia stolonifera* [215], *Xenia umbellata* [216] and *Latissimia ningalooensis* [217] have also spread widely across Venezuelan, Puerto Rican, and Cuban reefs, outcompeting native species and altering reef community structure and ecosystem function [218]. More recently, patches of the Indo-Pacific acroporid *Acropora tenuis* were documented in reefs of Morrocoy National Park, Venezuela [219], further indicating that invasions of soft and hard corals are becoming more frequent.

STATUS OF CORAL REEFS IN THE REGION

Hard coral cover

Across the Caribbean region, median hard coral cover decreased from 19.5% (credible interval at 95% [CrI $_{95\%}$]: 17.1 – 21.9%) in 1980–2009 to 11.0% (CrI $_{95\%}$: 10.3 – 11.8%) in 2020–2024, representing a 43.4% relative decline. There is strong evidence of a long-term decline from 1980 to 2024. Major decreases in hard coral cover are evident in 1982–1985, 1997–2000, 2004–2006, and 2023–2024, coinciding with periods of bleaching induced high heat stress and outbreaks of coral diseases across the region.

The marked declines observed during the 1980s reflect the convergence of multiple acute and chronic stressors. White-band disease, which spread widely across the

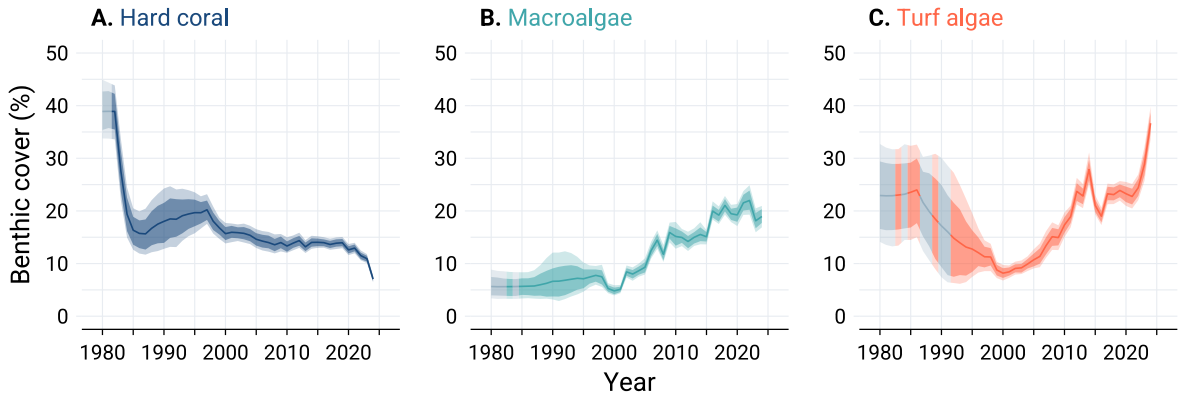


▲ **Figure 2.3.5.** Maximum sustained wind speed of hurricanes that passed within 100 km of a coral reef in the Caribbean region between 1980 and 2025. Colors correspond to the Saffir–Simpson category of the hurricane along its entire track. However, the values of sustained wind speed are extracted from the nearest hurricane position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir–Simpson scale (*i.e.* 119 km.h^{-1}). Derived from IBTrACS data (Knapp *et al.* [118]; see Materials and Methods).

Caribbean from the late 1970s through the early 1990s, caused catastrophic mortality of acroporid corals, previously among the most important reef-building taxa in the region. In parallel, the 1983–1984 mass mortality of the herbivorous sea urchin *D. antillarum* removed a major source of grazing pressure. Combined with continued, chronic unsustainable fishing of herbivorous fishes and increasing nutrient enrichment in coastal waters, this loss of herbivory facilitated widespread algal proliferation, likely impairing coral recruitment and constraining post-disturbance recovery [220].

In addition to these local and regional pressures, anomalously high sea surface temperatures in 1987 and 1995 triggered bleaching across roughly one-fifth of Caribbean reefs, further contributing to coral losses during the late 1980s and early 1990s. The first global bleaching event in 1998, associated with a strong El Niño, led to a clear regional decline in hard coral cover, with a further 2.3 percentage point decline (about 13.0%) in comparison to 2000. A second, particularly severe thermal disturbance in 2005 affected much of the basin, and an unprecedented hurricane season likely compounded physical damage to reefs while also providing some localized thermal relief through mixing. More recent marine heatwaves in 2010 and 2015–2017 led to a regional decline in hard coral cover, with a further 2.8 percentage point decline (about 20.4%) in comparison to the 2020s.

This trajectory shifted again after 2020. Between 2022 and 2024, modeled hard coral cover declined by 4.5 percentage points, equivalent to a further 38.7% relative loss, with the majority of this reduction concentrated in 2023 and 2024. This decline is consistent with the fourth global bleaching event, during which widespread bleaching and mortality were documented across multiple Caribbean locations. The magnitude of this decline is likely underestimated due to limited survey coverage in 2024 in the context of ongoing bleaching. These thermal impacts occurred alongside the continued spread of SCTLD, which reached nearly all Caribbean countries and territories by 2023. Collectively, these results indicate that Caribbean reefs have experienced nearly five decades of progressive coral loss, driven by the interplay of bleaching, disease, trophic



▲ **Figure 2.3.6.** Modeled temporal trends of benthic cover in the Caribbean region from 1980 to 2024 for hard coral (A), macroalgae (B), and turf algae (C). The solid line represents the estimated median benthic cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

imbalance, local anthropogenic pressures, and increasingly frequent and intense marine heatwaves, with the most recent years marking a new, and potentially accelerating phase of regional decline.

Despite differences in modelling approaches, the hard coral cover trends presented here are broadly consistent with previous regional syntheses, including the recent *Status and Trends of Caribbean Coral Reefs: 1970–2024* report [45]. Estimates of hard coral cover change during the 1980s fall within the ranges reported by previous Caribbean-wide assessments and values for the 2000s closely match those documented by independent analyses [221]. While some earlier reconstructions suggest larger absolute losses since the 1970s, differences in baselines, spatial coverage, and survey approaches limit direct comparability. Importantly, 1980 should not be interpreted as a pristine reference point, as substantial coral degradation had already occurred by that time, particularly through the collapse of *Acropora palmata* and *A. cervicornis* populations beginning in the 1960s due to direct anthropogenic impacts.

Macroalgal cover

Median macroalgal cover increased from 7.6% ($CrI_{95\%}$: 6.2 – 9.4%) in 1980–2009 to 20.0% ($CrI_{95\%}$: 18.3 – 21.7%) in 2020–2024 at the regional scale, representing a relative gain of 162.7%. The results show strong evidence of an increasing trend, which reflects the widespread shift from coral to algal-dominated reef states documented throughout much of the Caribbean [220, 221].

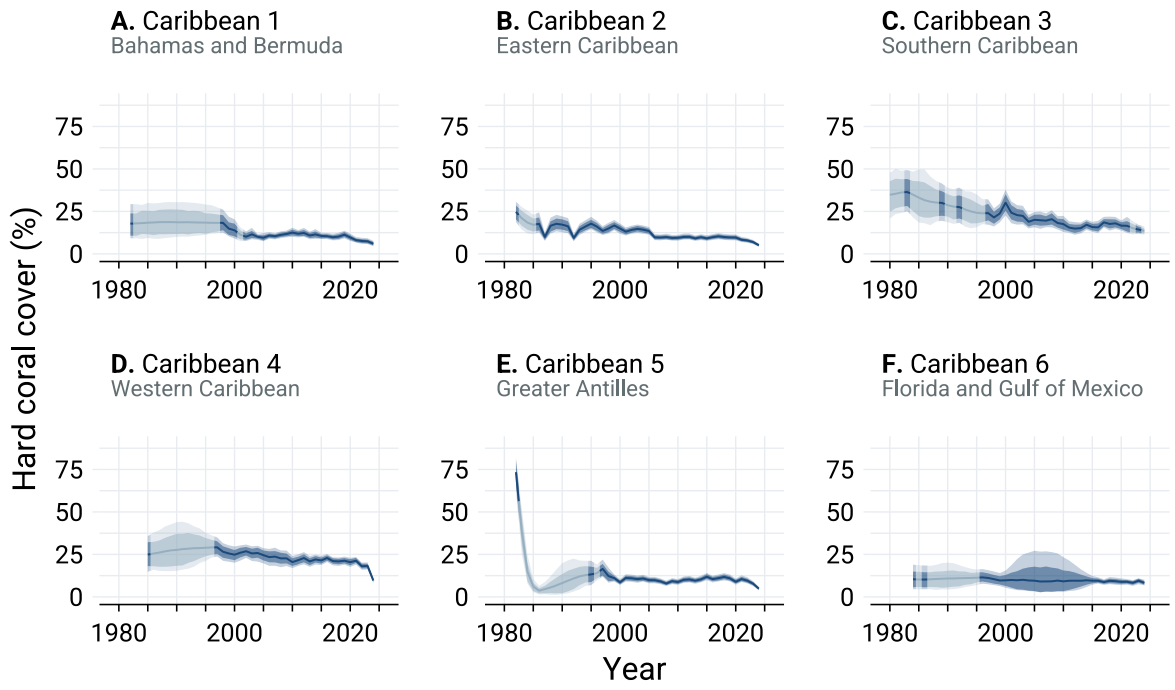
Overall, data show an increasing rate of macroalgal cover, particularly from the mid-2000s, which appears to have accelerated following major coral mortality events. For instance, macroalgal cover increased 5.2% on average (a relative increase of 55.4%) following the 2005 heatwave in comparison to 2007, a pattern consistent with the rapid colonization of newly available benthic space created by coral loss. Although data show an average decline in macroalgal cover from 1998–2000 (35.9% decline), we suspect this trend may be caused by a reduction in monitoring sites that occurred in 2000. More recently, macroalgal cover increased further in 2023 and 2024, coinciding with the sharp decline in hard coral associated with the fourth global bleaching event and the continued spread of SCTLD. These coupled trajectories underscore the tight ecological linkage be-

tween coral mortality and macroalgal expansion at the basin scale.

A critical early trigger of this long-term macroalgal expansion was the Caribbean-wide mass mortality of the herbivorous sea urchin *D. antillarum* in 1983–1984, during which *D. antillarum* population densities declined up to 100% on reefs throughout the Caribbean [222]. Although partial compensatory grazing by herbivorous fishes occurred at some locations, *D. antillarum* populations have remained well below historical densities across most of the region for more than four decades, limiting the recovery of this critical herbivory function. A second regional mass mortality of *D. antillarum* was documented in 2022 [201]. Although this event likely reduced grazing pressure in affected areas, absolute population declines appear smaller than during the 1983–1984 die-off, suggesting more limited basin-wide consequences for macroalgal proliferation.

Chronic human pressures have reinforced conditions favoring macroalgal dominance on Caribbean reefs. Coastal development, agriculture, and poor wastewater treatment have increased nutrient inputs, sedimentation, and turbidity, promoting macroalgal growth while inhibiting coral recruitment and survival. At the same time, unsustainable fishing of key herbivores, especially parrotfishes (Scaridae) and surgeonfishes (Acanthuridae), has reduced grazing pressure and allowed algal biomass to expand [220]. Some countries, including Bonaire, Bermuda, the Dominican Republic, Mexico, Guatemala, and Honduras, have implemented protections for herbivorous fish to counter these trends.

Overall, macroalgal cover has increased consistently over the past four decades, closely linked to declining hard coral cover. This shift reflects the combined effects of herbivore loss, chronic local stressors, and repeated climate disturbances.



▲ **Figure 2.3.7.** Modeled temporal trends of hard coral cover in Caribbean subregions from 1980 to 2024. The solid line represents the estimated median hard coral cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

The recent acceleration of macroalgal dominance suggests reinforcing feedbacks between coral loss, reduced structural complexity, and algal expansion, further weakening reef resilience under ongoing climate change.

Turf algal cover

The data indicate a long-term regional increase in turf algal cover from 15.6% ($CrI_{95\%}$: 11.5 – 19.9%) in 1980–2009 to 27.2% ($CrI_{95\%}$: 25.4 – 29.3%) in 2020–2024 (Figure 2.3.6C), a change supported by strong evidence. This corresponds to an absolute increase of 11.7 percentage points, or a relative increase of 74.7%. Unfortunately, major data gaps and potential inconsistencies in how turf were categorized between 1980 and 1990 make it challenging to discern clear temporal patterns in turf algal cover during this period.

Between the 2000s and 2010s, turf algal cover increased 11.0 percentage points on average, a relative increase of 100.1%. This increase is likely associated with similar factors as macroalgal cover, including the decline of hard coral cover after bleaching events, which creates available substrate for colonization [223]. Additionally, bottom-up processes such as nutrient enrichment [221, 224] may have promoted turf algae proliferation, while top-down controls may have weakened due to declines in herbivorous fish populations [225, 226] and *D. antillarum* sea urchins [227], resulting in reduced grazing pressure. Turf algal cover also increased 5.1% on average from 2010–2020, a 23.1% increase in total.

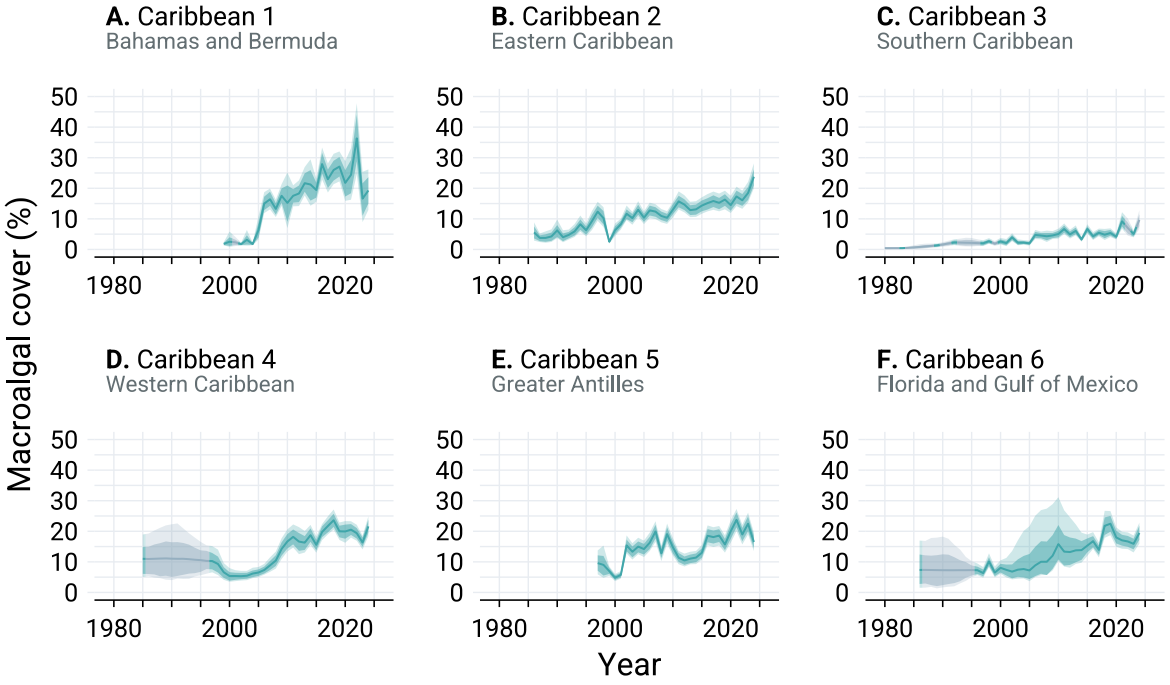
Due to limited data availability at the subregion scale, and inconsistencies in categorization among monitoring programs, trends in turf algal cover were not considered at the subregion scale.

STATUS OF CORAL REEFS IN THE SUBREGIONS

Caribbean 1 - Bahamas and Bermuda

The Bahamas and Bermuda subregion shows a distinct reef trajectory relative to the Caribbean as a whole. Hard coral cover increased modestly from the early 1980s to the late 1990s before entering a prolonged period of stagnation and gradual decline through the 2000s and 2010s (Figure 2.3.7A). Overall trends show hard coral cover decreased from 15.8% to 7.8% and macroalgal cover increased from 7.0% to 23.8% between 1982–2009 and 2020–2024, respectively (Figure 2.3.7A and Figure 2.3.8A).

Environmental conditions strongly shape reef trends in Bermuda and the Bahamas but these reef systems are inherently different. The modelled data are primarily from the Bahamas, with only one dataset from Bermuda. Although periodic cold stress from cooler winters slows coral growth and recovery, and marine heatwaves, storms, and high wave energy can cause damage, the differences between these locations today are due to differential impacts of coral diseases and fisheries management. SCTLD has not affected Bermuda but has reduced hard coral cover at some locations in the Bahamas [228]. Other diverse coral diseases occur in Bermuda, but these have low incidence rates [229], and coral bleaching is generally mild in Bermuda with very little mortality, in contrast to the Bahamas [31]. Additionally, Bermuda removed non-specific fish traps in 1990 [230] and protected all parrotfishes in 1993, resulting in Scarid populations showing a strong recovery across Bermuda's reefs [231], while Sherman *et al.* [232] pointed to the growing harvest of scarids in the Bahamas. Thus, through fisheries management practices and low prevalence of coral disease, Bermuda's



▲ **Figure 2.3.8.** Modeled temporal trends of macroalgal cover in Caribbean subregions from 1980 to 2024. The solid line represents the estimated median macroalgal cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

reefs remain healthy with moderate coral cover and low algal abundance [233–235].

Caribbean 2 - Eastern Caribbean

The Eastern Caribbean extends from the U.S. and British Virgin Islands to Saint Vincent and the Grenadines. Overall trends show hard coral cover decreased from 15.0% to 7.6% and macroalgal cover increased from 7.6% to 18.0% between 1982–2009 and 2020–2024, respectively (Figure 2.3.7B and Figure 2.3.8B).

This subregion lies within the Atlantic hurricane belt and is frequently impacted by severe storms, causing extensive damage to shallow reefs. Many islands are Small Island Developing States (SIDS) with limited capacity for monitoring, enforcement, and wastewater and fisheries management. High coastal population density and intensive tourism further increase pressures from development, nutrient enrichment, and sediment runoff. Together, these stressors have accelerated coral decline and reinforced persistent macroalgal dominance, highlighting the vulnerability of Eastern Caribbean reefs.

Caribbean 3 - Southern Caribbean

The southern Caribbean subregion includes the eastern mainlands of Colombia and Venezuela, as well as Aruba, Bonaire, Curaçao, and Trinidad and Tobago, and is reflective of wider region-wide trends. Overall trends show hard coral cover decreased from 19.5% to 11.0% and macroalgal cover increased from 2.1% to 7.0% between 1980–2009 and 2020–2024, respectively (Figure 2.3.7C and Figure 2.3.8C).

This subregion comprises extensive oceanic reefs, banks, and archipelagos influenced by large river plumes from the Amazon and Orinoco, seasonal upwelling, and exposure to storms. Economies depend heavily on tourism and reef-based fisheries. The region is also increasingly threatened by invasive species, particularly the

soft coral *Unomia stolonifera* and related xeniids, which have rapidly expanded along Venezuelan coasts and displaced native reef communities [216, 236]. However, even in highly anthropogenic areas, there is still hope. The Varadero reef, located just off of a highly polluted bay off Cartagena, hosts a diverse coral community, with hard coral cover peaking at 50–60% [237].

Caribbean 4 - Western Caribbean

The western Caribbean encompasses the Mesoamerican Reef system of Mexico, Belize, Guatemala, Honduras, and the reefs of Nicaragua, Costa Rica, Panama, and Caribbean Colombia. Overall trends show hard coral cover decreased from 26.4% to 17.6% and macroalgal cover increased from 9.4% to 19.6% between 1985–2009 and 2020–2024, respectively (Figure 2.3.7D and Figure 2.3.8D).

This subregion faces severe land-based and disease-driven pressures, distinguishing it from other Caribbean subregions. Influenced by sediment and nutrient inputs from deforested agricultural catchments, *Sargassum* blooms, and expanding coastal settlements, elevated turbidity has degraded nearshore reefs, suppressing coral recruitment and promoting persistent macroalgal growth. Coastal development, particularly along the Mexican Caribbean, Belizean cayes, and the Bay Islands of Honduras, exacerbates these challenges, compounded by significant impacts from SCTLD and the 2023–2024 marine heatwave that have decimated critical reef-building corals. Other countries, such as Nicaragua, Costa Rica, Panama, and Caribbean Colombia, are starting to face threats for unsustainable tourism development. However, some reefs still thrive, as is the case for the Cayman Crown reef between Belize and Guatemala. This reef's relatively high live hard coral cover (>50%) is a testament to the effectiveness of conservation efforts [238].

Caribbean 5 - Greater Antilles

The Greater Antilles include Cuba, Jamaica, Haiti, the Dominican Republic, and Puerto Rico. Overall trends show hard coral cover decreased from 13.5% to 8.3% and macroalgal cover increased from 9.7% to 20.3% between 1982–2009 and 2020–2024, respectively (Figure 2.3.7E and Figure 2.3.8E).

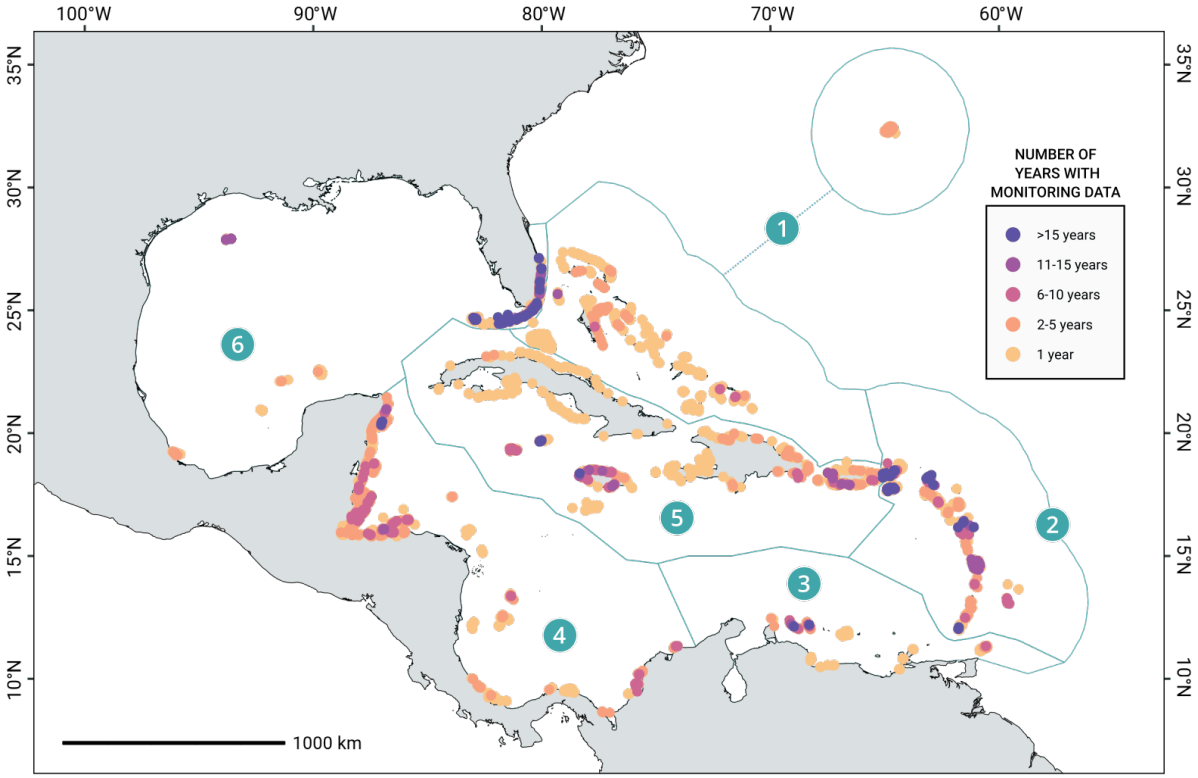
The Greater Antilles host extensive coral reefs, but face limited financial and technical resources, weak enforcement, and high dependence on marine fisheries for food security and livelihoods. Reef conditions are shaped by the combined effects of recurrent marine heatwaves, hurricanes, coral disease, and persistent water quality challenges, which have reduced structural complexity and coral recruitment while favoring stress-tolerant taxa such as macroalgae. Although localized management actions — such as improved wastewater controls, herbivore protection, and community-based fisheries management — have shown positive effects in some areas, these gains remain uneven and vulnerable to extreme events. Strengthening regulatory enforcement and integrated management across climate, water quality, and fisheries is essential to support coral reef recovery and regional resilience.

Caribbean 6 - Florida and Gulf of Mexico

This subregion is a significant marine ecosystem spanning over 1.5 million km^2, with a continental shelf of approximately 360,000 km^2, yet less than 1% of this area consists of coral reefs. Reefs in this subregion are connected but highly ecologically interdependent [239], and are mainly located in the Veracruz Reef System, Campeche Bank in Mexico and Florida Keys. Overall trends show that hard coral cover decreased from 10.8% to 8.8% and macroalgal cover increased from 8.4% to 17.4%, between 1984–2009 and 2020–2024, respectively

(Figure 2.3.7F and Figure 2.3.8F). However, it's important to acknowledge that this report does not contain information on the healthiest coral reefs, or on whether these reefs are outliers, and their cover may not significantly contribute to the regional trend.

Despite their limited extent, these coral reefs provide critical ecosystem services, including fisheries, tourism, and coastal protection. However, both global and local stressors have degraded reef health in recent decades. Rising sea temperatures have driven mass coral bleaching and mortality, while hurricanes and land-based runoff further damage coral reef systems. Additional pressures from unsustainable fishing, unregulated tourism, agricultural pollution, and oil activities continue to undermine ecosystem stability [240, 241]. Notably, some sites in the Flower Garden Banks and parts of the Campeche Bank still maintain relatively high hard coral cover in contrast with highly degraded reefs in the Florida Keys. Coral reefs in the East and West Flower Garden Banks for example, still harbor high live hard coral cover (~56%), dominated by *Orbicella* spp., and host healthy populations of herbivorous fishes [242], in contrast to highly degraded reefs in the Florida Keys.



▲ **Figure 2.3.9.** Spatio-temporal distribution of benthic cover monitoring sites across the Caribbean region and respective subregions. Light blue polygons and numbers represent subregions. Subregion 1 = Bahamas and Bermuda, Subregion 2 = Eastern Caribbean, Subregion 3 = Southern Caribbean, Subregion 4 = Western Caribbean, Subregion 5 = Greater Antilles, Subregion 6 = Florida and Gulf of Mexico. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region.

DESCRIPTION OF MONITORING DATA

Temporal trends in the percentage cover of benthic categories were estimated using data from 72 datasets, encompassing 14,076 monitoring sites. These data were collected from 24,011 surveys, with 90% conducted after 2002 (Figure A.1). Of all surveys carried out in the region between 1973 and 2025, 34.9% (representing 41.5% of surveyed sites) were conducted in subregion 2, whereas only 4.9% (3% of sites) were conducted in subregion 1 (Table A.4).

CONCLUSIONS

Caribbean coral reefs have experienced over forty years of decline, driven by anthropogenic climate impacts, coral disease, coastal development, unsustainable fishing practices, and the introduction of invasive species. Collectively, these stressors have resulted in a 51.8% decrease in hard coral cover between 1980–2023, with an additional decline from 2023–2024 associated with the fourth global bleaching event and continued spread of SCTLD. At the same time, macroalgal cover has increased, signaling possible shifts to new ecosystem structures. Though management successes demonstrate that targeted interventions can yield measurable benefits for Caribbean reefs, broadly consistent declines in health are evident across all subregions.

Altering this trajectory is essential to preserving coral reef ecosystem services that directly or indirectly benefit millions of people. The implementation of conservation measures such as fully protected areas, improved water quality and wastewater management, amelioration of coastal impacts, strengthened fisheries management, restoration, research on genetically resilient species and other mitigation approaches (alongside urgent global reductions in greenhouse gas emissions) are critical to bolstering the climate-resilience of Caribbean coral reefs. Additionally, standardizing reef monitoring approaches, establishing effective data sharing across global and regional initiatives — particularly through open access datasets, and building regional consistency in classifying key benthic indicators of reef health (*e.g.* turf algae) will facilitate more effective reef health tracking.

DATA CONTRIBUTORS

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Access to online supplementary materials

The modelled benthic cover data, detailed outputs from the data analyses, and the CRediT author contribution statement are available on Zenodo.

CASE STUDY 3

Effective strategies for rebuilding Caribbean reef fish populations

Authors

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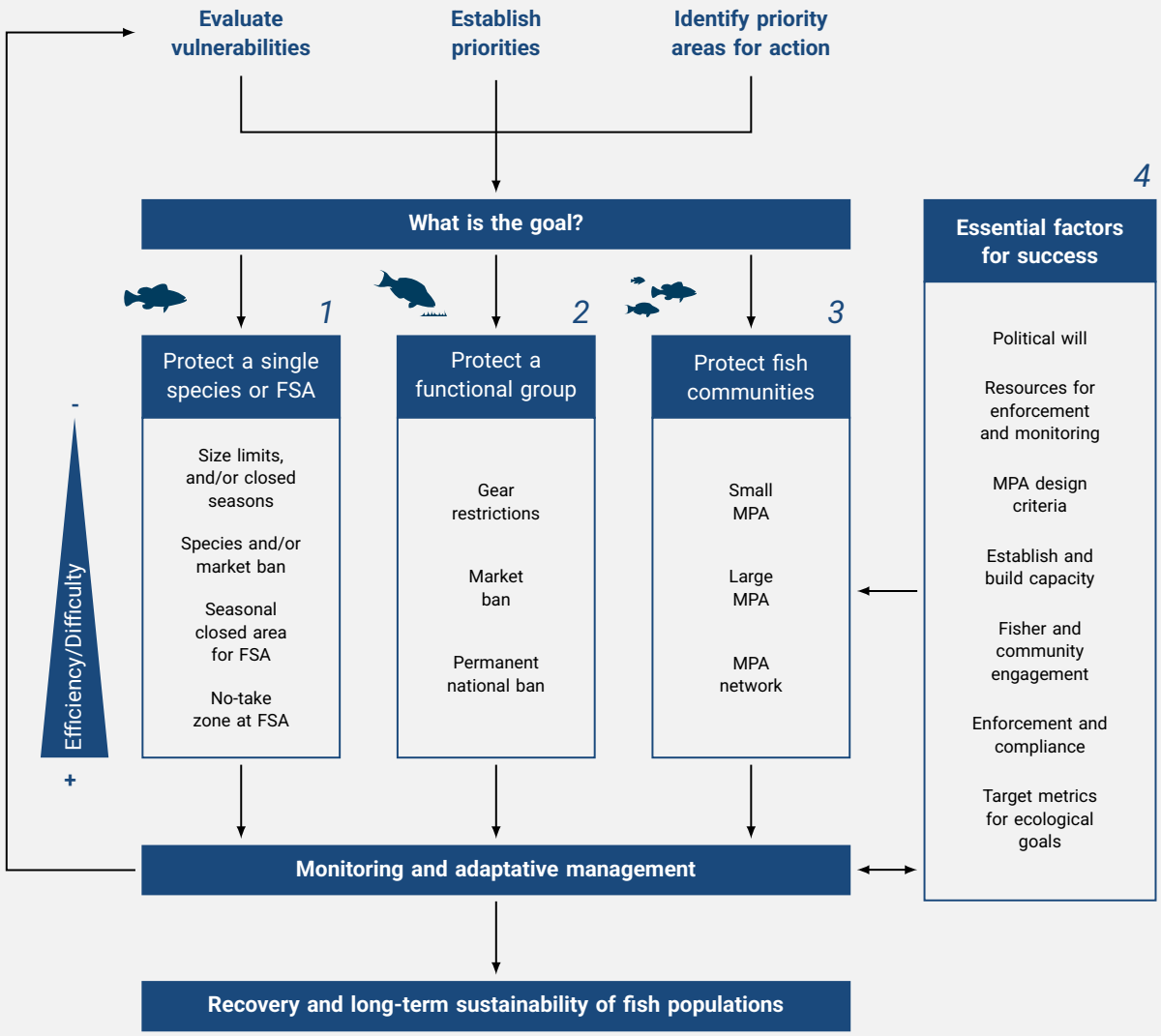
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Caribbean coral reefs support a rich diversity of reef fish that play essential roles in maintaining ecosystem health. However, these fish populations have experienced dramatic declines in recent decades, primarily due to overfishing and habitat degradation [221, 225]. The loss of key fish species not only threatens the ecological balance of the reefs, but also impacts the livelihoods of coastal communities that depend on healthy fisheries for food and income. Recognizing the urgency of this crisis, Caribbean nations and territories have adopted a suite of science-based management strategies aimed at rebuilding vulnerable fish populations and restoring reef resilience. The key aspects of these strategies include 1) evaluating vulnerabilities, 2) establishing priorities, 3) targeting areas needing improvement, and, most importantly, 4) identifying target metrics for achieving ecological goals and socio-economic needs that inhibit progress for sustainable fisheries (Figure C3.1). Case studies from the US Virgin Islands, Bonaire, the Dominican Republic, and the broader Mesoamerican Reef illustrate both the challenges and successes of strategies for rebuilding vulnerable fish populations, protecting ecologically important fish, harmonizing transboundary fish management, and investing in effective marine management.

Rebuilding vulnerable fish populations

Many commercially and ecologically important reef fishes form spawning aggregations, where hundreds to thousands of individuals gather for reproduction at predictable times and locations. Once discovered by fishers, these large gatherings are very vulnerable to overfishing and population collapse [243].

In the US Virgin Islands, the decline of Nassau grouper (*Epinephelus striatus*) and red hind (*Epinephelus guttatus*) populations prompted a comprehensive conservation response focused on fish spawning aggregations. Regulatory actions developed with community participation, such as harvest bans and possession prohibitions, were enacted to protect these species during critical spawning periods. Spatial closures, including the Red Hind Bank Marine Conservation District in 1990 and the Grammanik Bank in 2005, provided safe havens for spawning aggregations. Adaptive management, informed by acoustic telemetry and ongoing monitoring over several decades, allowed for the refinement of protected areas and closure periods to maximize conservation outcomes.



▲ **Figure C3.1.** The decision tree for rebuilding sustainable fish populations is designed to help natural resource managers evaluate vulnerable fish populations, establish ecological and socio-economic priorities, and target areas needing improvement. The ecological goals recognize the different management approaches needed to address the vulnerabilities of (1) individual species, especially those that form fish spawning aggregations (FSA), (2) ecologically important functional groups such as parrotfishes, and (3) biologically complex fish communities. The level of difficulty of implementing the various management alternatives does not necessarily translate to greater success of reaching ecological goals. The success of any one of these management alternatives in rebuilding sustainable fish populations is largely dependent upon seven essential social-economic components (4). These components range from strong leadership that provides thoughtful, integrative and inclusive planning and implementation strategies to achieve ecological goals, to sufficient resources to build capacity for science-based management, monitoring, and enforcement. Regular evaluation of target progress, via monitoring and fisher engagement, are required to adapt management actions to new information, and recover and stabilize vulnerable fish populations. As seen in the case studies, it can take decades for depleted fish populations to recover, and recovery timelines are dependent on individual species life history patterns.

The involvement of local fishers was crucial for compliance, and the results were striking. The Marine Conservation District (41 km^2) was changed to a permanent no-take zone in 1999 and, after 30 years of protection, red hind populations showed significant gains in size (35% increase) and density; and spawning potential increased to its highest levels [244–246]. The 1.5 km^2 Grammanik Bank, a three-month seasonal closed area, allowed Nassau grouper abundance to increase from 30 individuals to over 1,000 individuals in 20 years [247]. Due to their continued vulnerability, Nemeth *et al.* [248] recommended that the Grammanik Bank be expanded from 1.5 to 6.5 km^2 and the closed period extended to five months. Enacting these changes would increase spatial and temporal protection of the Nassau grouper spawning population to 95%.

The key to recovery of these vulnerable populations in the US Virgin Islands and elsewhere is long-term, dedicated collaborative and adaptive management based on research and monitoring data, and highlights the importance of protecting fish spawning aggregation sites as a critical management tool [249, 250]. Because fish spawning aggregation sites are used for reproduction by multiple species, like groupers and snappers, but at different times of year, protecting a single spawning aggregation site will have far-reaching benefits for rebuilding spawning populations of many important functional groups.

Protecting ecologically important fish

Herbivorous fish, especially parrotfish, are a cornerstone of reef health, as their grazing prevents algae from overgrowing corals and facilitates coral recruitment. Both Bonaire and the Dominican Republic have implemented regulations to protect parrotfish, including harvest bans and gear restrictions [251]. Bonaire's success in parrotfish protection is attributed to strong enforcement, sustainable funding through a reef usage tax, and systematic ecological monitoring; which, combined, have stabilized parrotfish populations and boosted coral recruitment [252]. In contrast, the Dominican Republic's efforts have been hampered by a higher demand on parrotfish as a staple food source, weak enforcement, insufficient funding, and inconsistent monitoring, leading to continued declines in parrotfish numbers and diminished conservation effectiveness [253].

Strengthening protection through enforcement and adaptive management based on consistent long-term monitoring data will provide the protection needed to help rebuild key ecological fish populations. In the Mesoamerican Reef, species-specific fisheries regulations have become important tools to protect key functional groups. Belize pioneered regional action in 2009 by banning the harvest of herbivorous fishes, setting a precedent later followed in Honduras, Guatemala, and Mexico through national bans, protected species listings, and restrictions on destructive fishing gear in reef areas [254]. Parrotfish in the wider Caribbean region are protected under the Specially Protected Areas and Wildlife (SPAW) Protocol, a regional environmental treaty among 19 countries. In 2023, all parrotfish species were added to Annex III, which requires countries to regulate their sustainable use.

Harmonizing fish management across borders

Harmonizing fish management across borders is central to rebuilding fish populations in the Mesoamerican Reef and relies on a complementary set of management strategies. These include the protection of fish spawning aggregations through layered spatial measures (*e.g.* no-take zones within Marine Protected Areas (MPAs)), national and regional fisheries bans

(*e.g.* parrotfish), and species and gear-specific regulations implemented inside and outside protected areas. Evidence from across the Mesoamerican Reef demonstrates that fully protected areas consistently sustain much higher fish biomass (~1,200 g.100 m^{-2}) compared to general-use or unprotected areas (500 g.100 m^{-2}). Cozumel (Mexico) exemplifies how these strategies can act synergistically: while much of the reef system is under some form of protection, approximately 35% of the surrounding marine area is fully protected, resulting in fish biomass levels that are roughly double those of less protected areas [254].

Regional cooperation has led to species-specific bans, gear restrictions, and seasonal closures, but enforcement and coverage of protected areas remain below international targets. The most promising path forward involves strategic expansion of fully protected reserves, science-based regulations, robust enforcement, and cross-border collaboration and advocacy, such as the Healthy Reefs for Healthy People Initiative.

Investing in essential components for success

Key factors contributing to MPA success in restoring Caribbean reef fish populations include ongoing commitment to science-based adaptive management, strong enforcement, fisher engagement and compliance, strategic MPA design, and regional collaboration [255].

Significant progress is being made through multifaceted strategies that integrate regulatory protections, spatial and temporal closures, long-term monitoring, and robust stakeholder engagement.

Looking ahead, priority actions should include protecting vulnerable and ecologically important fish species, especially those forming spawning aggregations, expanding fully protected marine reserves, and securing sustainable funding. Together, these are essential steps towards the long-term recovery and sustainability of fish populations and the sustained resilience of reef ecosystems (Figure C3.1).

CASE STUDY 4

Adaptive coral reef restoration in The Bahamas: the MSC Foundation's Ocean Cay Restoration Programme

Authors

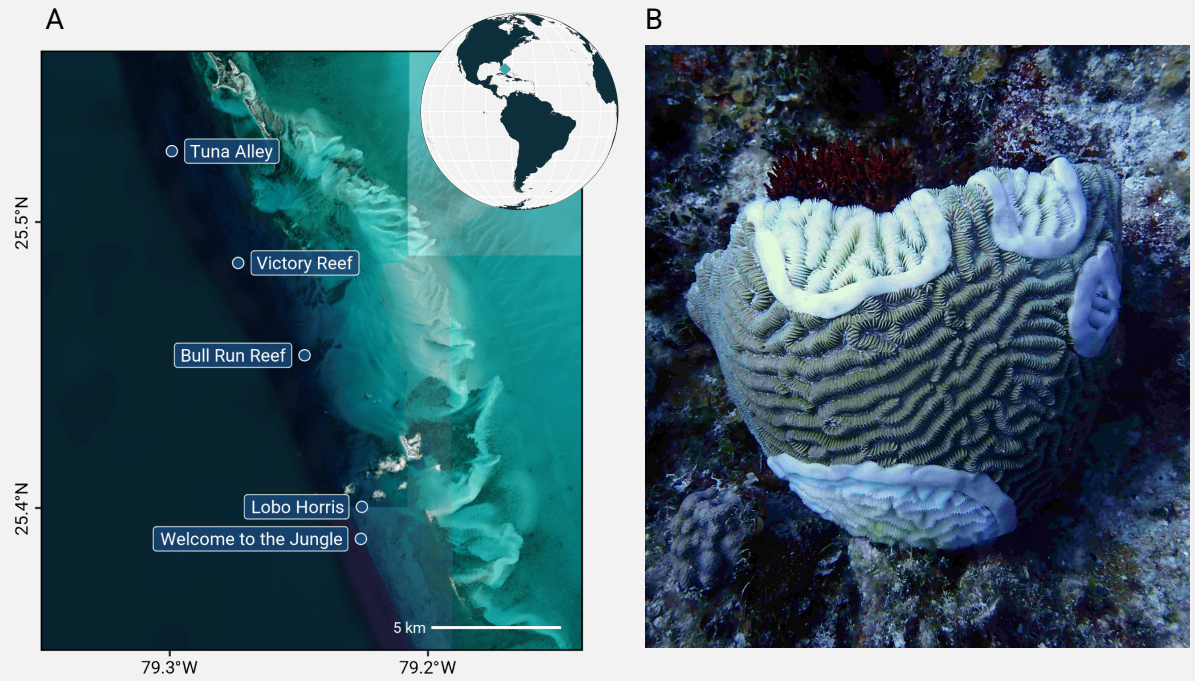
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The MSC (Mediterranean Shipping Company) Foundation's marine conservation programme at Ocean Cay (The Bahamas) constitutes a science-based restoration initiative designed to restore ecologically resilient coral reef ecosystems while helping to further develop national capacity in The Bahamas. Integrating coral restoration, applied research, education, and public engagement, the programme aims to enhance the resilience of coral reef ecosystems to environmental stressors whilst maintaining and, where possible, enhancing local biodiversity. Implementation of the programme is underpinned by strong collaboration with experts around the world, international institutes, including the Perry Institute for Marine Science, University of Miami, Nova Southeastern University, and national institutions such as the University of The Bahamas and The Bahamas Agriculture and Marine Science Institute. This global collaboration enables the MSC Foundation to align international technical expertise with national knowledge and priorities.

To establish robust ecological baselines and quantify reef condition trajectories, Perry Institute for Marine Science, supported by the MSC Foundation, conducted Rapid Ecological Assessments in 2019 and 2025 (Figure C4.1), applying standardised Atlantic and Gulf Rapid Reef Assessment (AGRRA) protocols alongside permanent photomosaic monitoring ($\geq$100 m^2 plots with georeferenced markers). These assessments captured reef dynamics in the context of acute and chronic stressors, including the 2023 marine heatwave and the regional expansion of Stony Coral Tissue Loss Disease (SCTLD).

Across surveyed sites, live hard coral cover remained persistently low (2–5.5%), with no statistically significant temporal change at most locations between 2019 and 2025, although one site exhibited a decline from ~5% to ~2%, potentially influenced by fine-scale habitat heterogeneity. Notwithstanding this apparent stability in overall hard coral cover, pronounced shifts in coral community composition were detected. Coral assemblages documented in 2019 exhibited higher species richness and more even relative abundances, whereas by 2025, declines in thermally sensitive coral taxa, including *Agaricia agaricites* and *A. tenuifolia*, and to a lesser extent *Porites furcata*, were evident, consistent with impacts from the fourth global bleaching event. In contrast, relatively thermally tolerant species such as *Porites astreoides* and *Siderastrea siderea* increased in proportional abundance. Importantly, the persistence of remnant populations of sensitive coral taxa suggests potential local acclimatory or adaptive capacity that may be leveraged in restoration planning.



▲ **Figure C4.1.** Location of the five surveyed sites, in Cat Cays, The Bahamas (A). A diseased *Colpophyllia natans* colony in Bull Run Reef, May 2026 (B) (credit: Emeline Bouchet).

Disease emergence represents a compounding pressure. SCTLD, first recorded nationally in 2019, was not observed at Ocean Cay until 2024 but expanded to multiple sites by 2025, affecting several susceptible coral species. Coral disease progression has been quantified using high-resolution photogrammetry and digital reef "twins", enabling colony-scale tracking of lesion dynamics. Active *in situ* mitigation, including antibiotic treatment trials, is being implemented to slow tissue loss and preserve reproductively viable colonies. These results, and active mitigation measures, highlight Ocean Cay as a site of particular conservation importance as a refuge for relatively intact breeding populations of SCTLD-susceptible species within The Bahamas.

Fish assemblages remained broadly stable over the assessment period, with spatial variability attributed primarily to recruitment dynamics rather than directional environmental change. Densities of key functional groups, including herbivorous parrotfish, snappers, and grunts, showed no consistent temporal decline, although a localised increase in juvenile *Scarus taeniopterus* was recorded in 2025.

Informed by these findings, the MSC Foundation's Marine Conservation Center, located at Ocean Cay, prioritises increasing coral biomass and structural complexity through integrated restoration strategies that combine asexual propagation (*in situ* and land-based nurseries) with sexual reproduction (larval rearing, settlement, and outplanting). These interventions are coupled with continuous environmental monitoring and targeted disease mitigation to enable adaptive management. Through coordinated implementation with national partners, the programme aligns with broader conservation objectives, contributing to the protection and long-term resilience of Bahamian coral reef ecosystems while advancing scalable restoration approaches relevant to the wider Caribbean.

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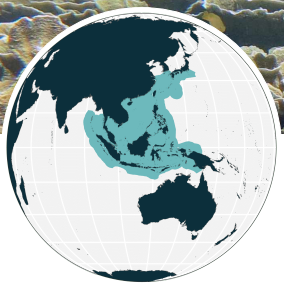
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Coral reef in Pulau Pemanggil, Malaysia
Credit: Affendi Yang Amri



EAST ASIAN SEAS

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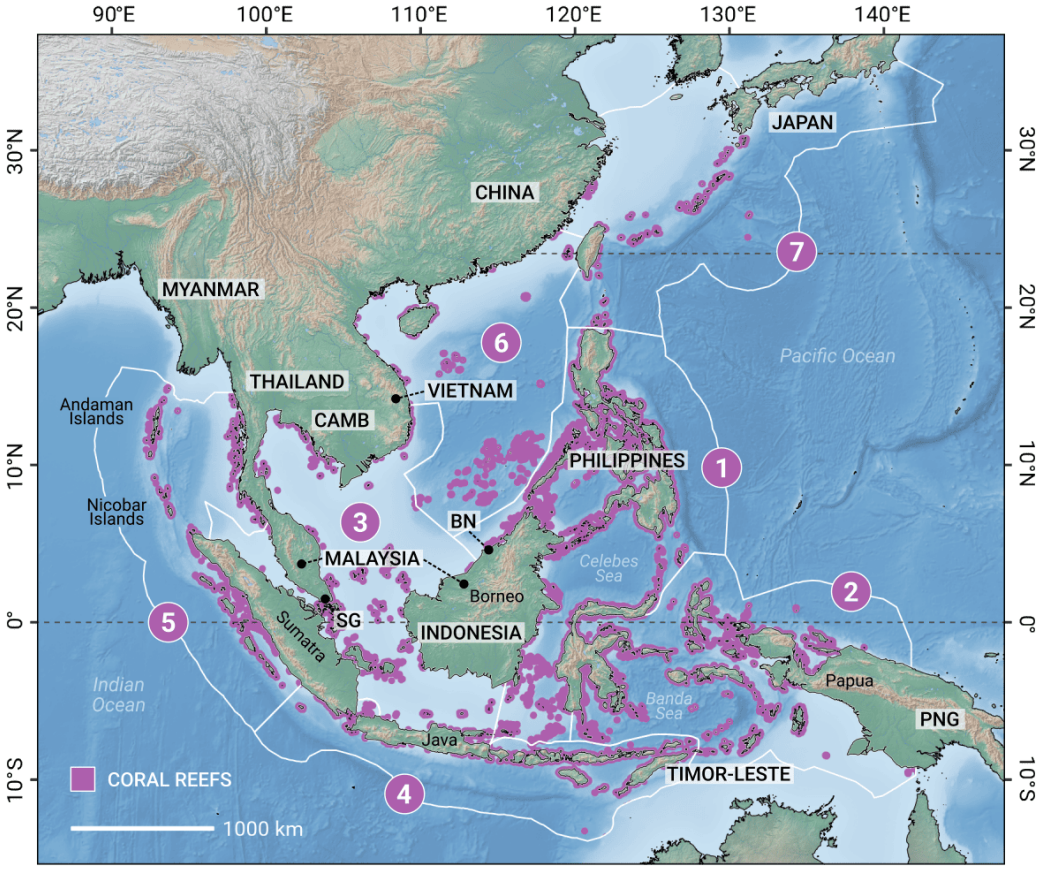
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INTRODUCTION

The GCRMN East Asian Seas region extends over a maritime area of around 13.5 million km^2, ranging across 50° of latitude and 55° of longitude. The region hosts 75,765 km^2 of coral reefs, which represent 30.3% of the world's coral reef extent, and is the largest GCRMN region in terms of reef extent. Coral reefs reside within the exclusive economic zones of 18 countries and territories within the region, ranging from Japan in the north to Indonesia in the south, and from the Andaman and Nicobar Islands in the west to the Indonesian province of Papua in the east (Figure 2.4.1).

For the purpose of this report, the GCRMN East Asian Seas region was divided into seven subregions (Figure 2.4.1). Subregion 1 (Northern Coral Triangle) hosts 42.3% of coral reef extent in the region (32,019 km^2) whereas subregion 7 (Kuroshio) hosts 3% of coral reef extent (2,286 km^2) (Table A.1).



▲ **Figure 2.4.1.** The distribution of coral reefs within the East Asian Seas region and respective subregions. White polygons and numbers represent subregions. Subregion 1 = Northern Coral Triangle, Subregion 2 = Central Coral Triangle, Subregion 3 = Sunda Shelf, Subregion 4 = Southern Java and Sunda, Subregion 5 = Andaman, Subregion 6 = South China Sea, Subregion 7 = Kuroshio. Coral reefs are delineated with a 20 km buffer to enhance interpretation, and only coral reefs within the region are represented. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region. BN = Brunei Darussalam, CAMB = Cambodia, SG = Singapore.

As of 2020, a total of 54.7 million inhabitants were living within 5 km of coral reefs of the East Asian Seas. From 2000 to 2020, the region recorded a 41.2% increase in populations residing within 5 km of coral reefs, representing an average annual increase of approximately 800,000 inhabitants. The greatest increase in populations over this period was recorded in subregion 1 (Northern Coral Triangle), which also recorded the highest number of inhabitants in 2020 (Table A.2).

THREATS AND DISTURBANCES

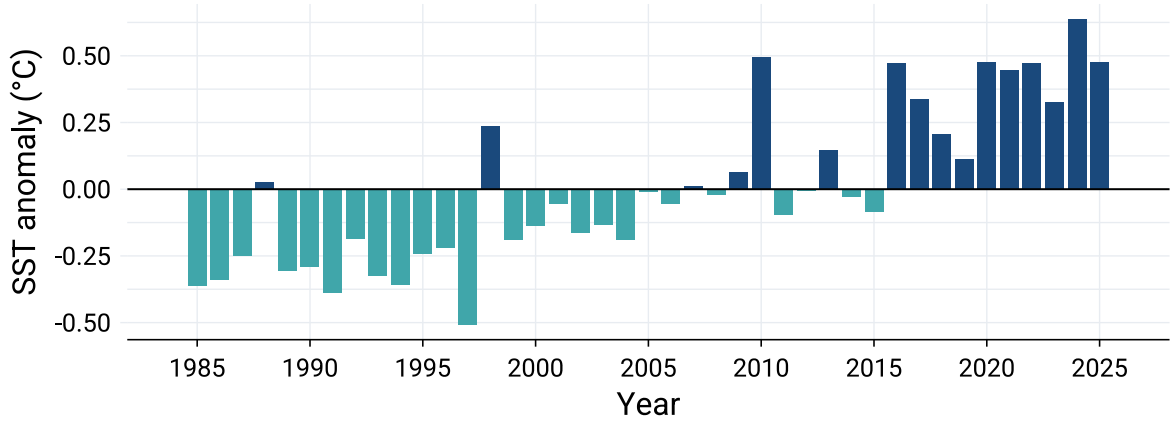
Heat stress

From 1985 to 2025, the long-term average sea surface temperature (SST) recorded over coral reefs within the East Asian Seas region was 28.64 °C. During this period, the SST over coral reefs in the region increased by 0.83 °C, corresponding to an average warming rate of 0.21 °C per decade (Table A.3).

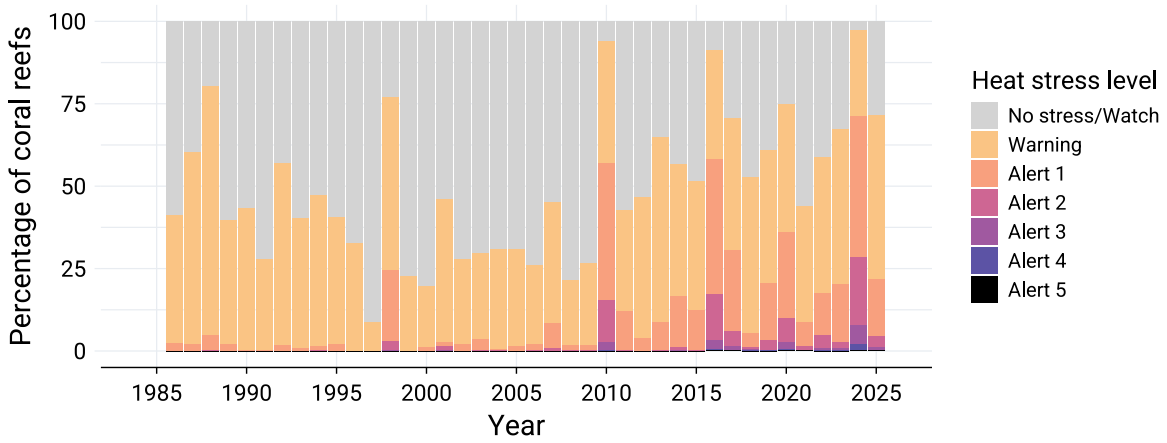
From 1986 to 2024, periods of major heat stress occurred in and around the years of 1998, 2010, 2016, and 2024, corresponding to the four global bleaching events. In addition, there was also significant coral bleaching observed in 2020 at various localities in the region [256, 257]. Each of the last three major heat stress events since 2010 placed over 15% of coral reefs in the region at minimum Alert 2 stress level (reef-wide bleaching and mortality of heat-sensitive corals, 8–12 DHW, Figure 2.4.3).

Typhoons

In the East Asian Seas region, cyclones are known as typhoons. From 1980 to 2025, a total of 558 typhoons passed within 100 km of coral reefs within the East Asian Seas region, representing an average of 12.1 $\pm$ 2.6 SD typhoons per year (Figure 2.4.4). The typhoon with the highest sustained wind speed over the studied period was the typhoon Meranti, in September 2016, which passed 25 km from a coral reef with sustained wind speed of 306 km.h^{-1} (Figure 2.4.5). Although the 100 km threshold was selected, typhoons passing at a greater distance from a coral reef, not represented within this figure, may also have had an impact on coral reefs [108].



▲ **Figure 2.4.2.** Annual mean sea surface temperature (SST) anomalies (°C) over the coral reefs of the East Asian Seas region from 1985 to 2025. SST anomalies were calculated based on the long-term average over the period 1985–2025. Positive values (red bars) indicate SST warmer than the long-term average, whereas negative values (blue bars) indicate SST cooler than the long-term average. Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).



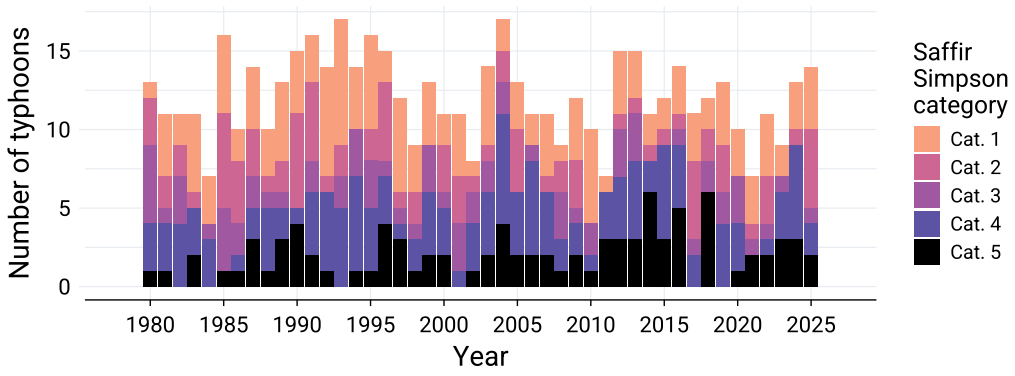
▲ **Figure 2.4.3.** Percentage of the coral reefs within the East Asian Seas region experiencing heat stress levels per year, from 1986 to 2025. No stress/Watch (DHW = 0); Warning (possible bleaching, 0 $<$ DHW < 4); Alert 1 (reef-wide bleaching, 4 $\leq$ DHW $<$ 8); Alert 2 (reef-wide bleaching and mortality of heat-sensitive corals, 8 $\leq$ DHW $<$ 12); Alert 3 (multi-species mortality, 12 $\leq$ DHW $<$ 16); Alert 4 (severe multi-species mortality, 16 $\leq$ DHW $<$ 20); and Alert 5 (near-complete mortality, DHW $\geq$ 20). Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).

Regional disturbances

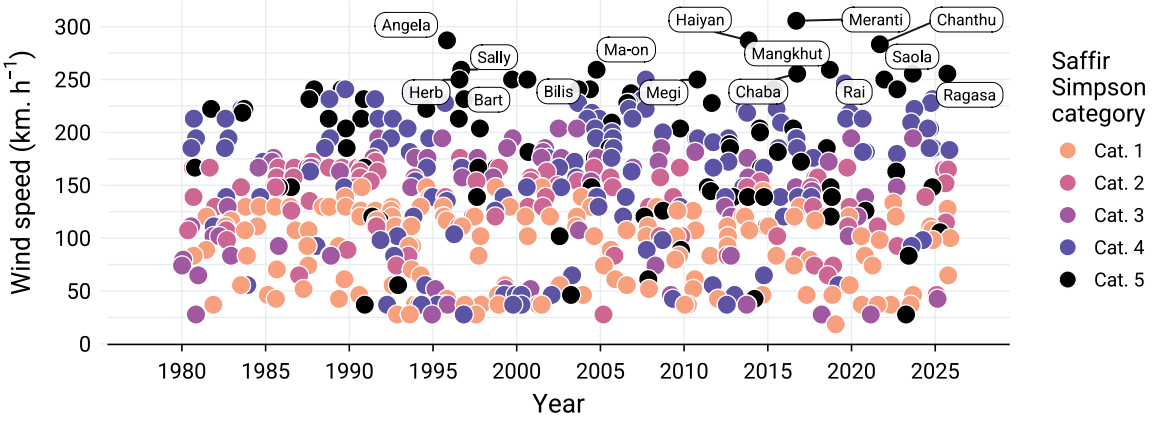
Region-wide analyses of benthic cover over nearly four decades have shown that large variations in nutrient influx from land-based activities adjacent to coastal coral reefs have a negative influence on coral cover, while high levels of nutrients have a positive effect on macroalgal cover [178, 258]. Elevated nutrient levels have been reported on the reefs of various reef environments in the East Asian Seas region [259, 260], where they can drive macroalgae to proliferate and outcompete corals [261], and also increase the severity of coral bleaching and disease [262, 263]. Marine litter in the form of marine plastics and microplastics is an emerging threat, with countries in the East Asian Seas region accounting for the majority of marine plastic pollution in the world [264, 265].

Increased sedimentation due to coastal development and land reclamation occurring throughout the region may also account for long-term changes in coral reef communities [266–268]. Sediment can physically abrade and smother corals by generating an anoxic microenvironment under layers of deposited material [269–271]. Suspended sediment can also lead to increased water turbidity, thereby curbing the growth and survival of corals [272–274].

Unabated fishing pressure continues to threaten key reef areas in the region. Overfished reef sites may exhibit uncontrolled macroalgal growth, curtailing coral recruitment and recovery [275]. A previous estimate showed that there are over 3 million coral reef fishers in Southeast Asia, accounting for half of the total number worldwide [14]. However, with a higher proportion of underfished stocks compared to the rest of the world, and as the human population in the region is expected to stabilise over the next few decades, fisheries demand could level off [276], providing opportunities for more effective reef management and restoration.



▲ **Figure 2.4.4.** Number of typhoons that passed within 100 km of the coral reefs of the East Asian Seas region from 1980 to 2025. Colors correspond to the Saffir–Simpson category of the typhoon along its entire track. Derived from IBTrACS data (Knapp *et al.* [118]; see Materials and Methods).



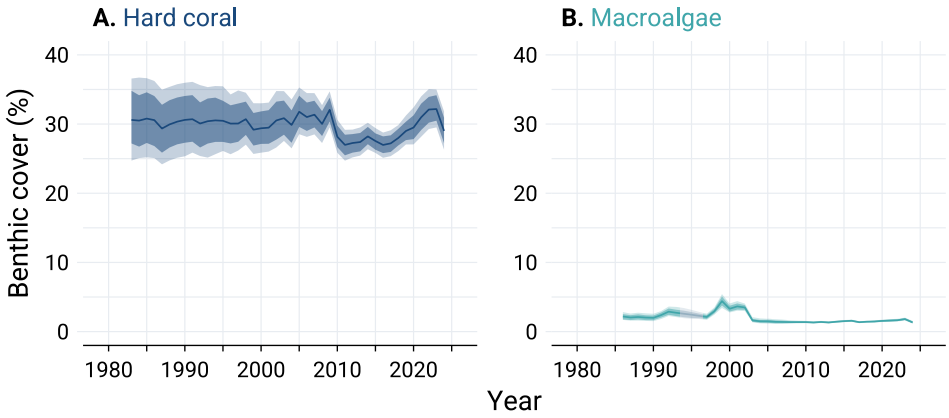
▲ **Figure 2.4.5.** Maximum sustained wind speed of typhoons that passed within 100 km of a coral reef in the East Asian Seas region between 1980 and 2025. Colors correspond to the Saffir–Simpson category of the typhoon along its entire track. However, the values of sustained wind speed are extracted from the nearest typhoon position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir–Simpson scale (*i.e.* 119 km.h^{-1}). Derived from IBTrACS data (Knapp *et al.* [118]; see Materials and Methods).

STATUS OF CORAL REEFS IN THE REGION

Based on the modelled results, regional hard coral and macroalgal cover trends (Figure 2.4.6) are described in periods from 1983–1997, 1998–2019 and 2020–2024. Turf algal cover trends were excluded from the East Asian Seas region due to limited and unevenly distributed data across subregions to support a meaningful analysis.

1983–1997: Relatively stable hard coral cover, variable macroalgal cover

Although data are available from 1983 to 1997, this period was generally characterised by limited availability and uneven spatial coverage of data across the East Asian Seas region.



▲ **Figure 2.4.6.** Modeled temporal trends of benthic cover in the East Asian Seas region from 1980 to 2024 for hard coral (A) and macroalgae (B). The solid line represents the estimated median benthic cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

Over the period, hard coral cover remained relatively stable at a median of 30.3% (credible interval at 95% [CrI $_{95\%}$]: 25.3 – 35.7%), with limited interannual variability.

From 1986 to 1997, median macroalgal cover averaged around 2.3% (CrI $_{95\%}$: 1.8 – 3.0%).

1998–2019: Persistence through repeated thermal disturbances

Ranging over two decades, this period covered the longest time period, during which the first three global bleaching events occurred, in 1998, 2010, and 2016 (Figure 2.4.3), representing distinct disturbance intervals and pronounced shifts in both hard coral and macroalgal cover in the regional time series.

Following the first global bleaching event in 1998, hard coral cover showed a slight decline of 1.5 percentage points, decreasing to 29.2% (CrI $_{95\%}$: 25.7 – 33.0%) in 1999. This was followed by a prolonged period of variable but gradually increasing hard coral cover through the 2000s, with median values generally increasing to a high of 32.0% (CrI $_{95\%}$: 29.5 – 34.7%) in 2009. After the second global bleaching event in 2010, the regional median hard coral cover experienced its steepest decline, falling to 27.0% (CrI $_{95\%}$: 24.7 – 29.2%) in 2011. It then remained broadly stable at that level through the third global bleaching event in 2016. Interestingly though, despite the repeated thermal anomalies that occurred after 2016 (Figure 2.4.3), regional hard coral cover registered a distinct upward trend to 29.0% (CrI $_{95\%}$: 27.1 – 31.2%) in 2019.

In contrast to hard coral cover, macroalgal cover increased abruptly at the beginning of the period, reaching a short-term regional peak of 4.4% (CrI $_{95\%}$: 3.4 – 5.4%) in 1999, before declining rapidly until 2003. From 2003 onward, macroalgal cover remained persistently low, at around 1.4% (CrI $_{95\%}$: 1.2 – 1.7%), with limited interannual variability and no sustained upward trend over the remainder of the period.

2020–2024: Increased variability signals

There is strong evidence that hard coral cover increased between the 2010s and 2020–2024, with a gain of 3.1 percentage points, corresponding to a 11.1% relative increase. This is reflected further by the continued rise until 2023, slightly surpassing its

2009 peak at 32.2% (CrI $_{95\%}$: 29.5 – 35.0%). However, the fourth global bleaching event, declared in 2023 resulted in a 3.2 percentage point drop in regional hard coral cover, reducing median hard coral cover to 29.0% in 2024 (CrI $_{95\%}$: 26.3 – 31.8%). As this report and analysis does not capture the impacts of the fourth global bleaching event on reef status, it is unclear if regional hard coral cover will rebound like it did over the last two decades. Additional monitoring is therefore required to determine whether recent declines represent short-term disturbance responses or a more persistent shift in regional hard coral cover [277].

Macroalgal cover remained persistently low from 2020 to 2024, with a modest uptick in 2023 (median cover of 1.8%), followed by a drop in 2024, to 1.3% (CrI $_{95\%}$: 1.2 – 1.5%). However, there is strong evidence that macroalgal cover increased between the 2010s and 2020–2024, although the relative change was modest (+11.9%).

STATUS OF CORAL REEFS IN THE SUBREGIONS

Temporal trends in hard coral cover varied considerably across the seven East Asian Seas subregions. Availability of data, including of long-term datasets, was also variable across subregions.

East Asian Seas 1 - Northern Coral Triangle

The Northern Coral Triangle subregion carries the largest relative coral reef extent (42.3%), making its trends disproportionately influential to the East Asian Seas region. Median hard coral cover in the Northern Coral Triangle remained relatively stable at 31.2% (CrI $_{95\%}$: 23.8 – 39.0%) between 1986 and 1998, followed by a modest decline and increased variability around the late-1990s, coinciding with the first global bleaching event in 1998. There is no evidence of a change in hard coral cover between 1986–2009 and 2020–2024.

Macroalgal cover exhibited a pronounced but short-lived increase centred around the late-1990s to early-2000s, peaking briefly at 8.0% in 1999 before declining rapidly and remaining persistently low at below 2% thereafter.

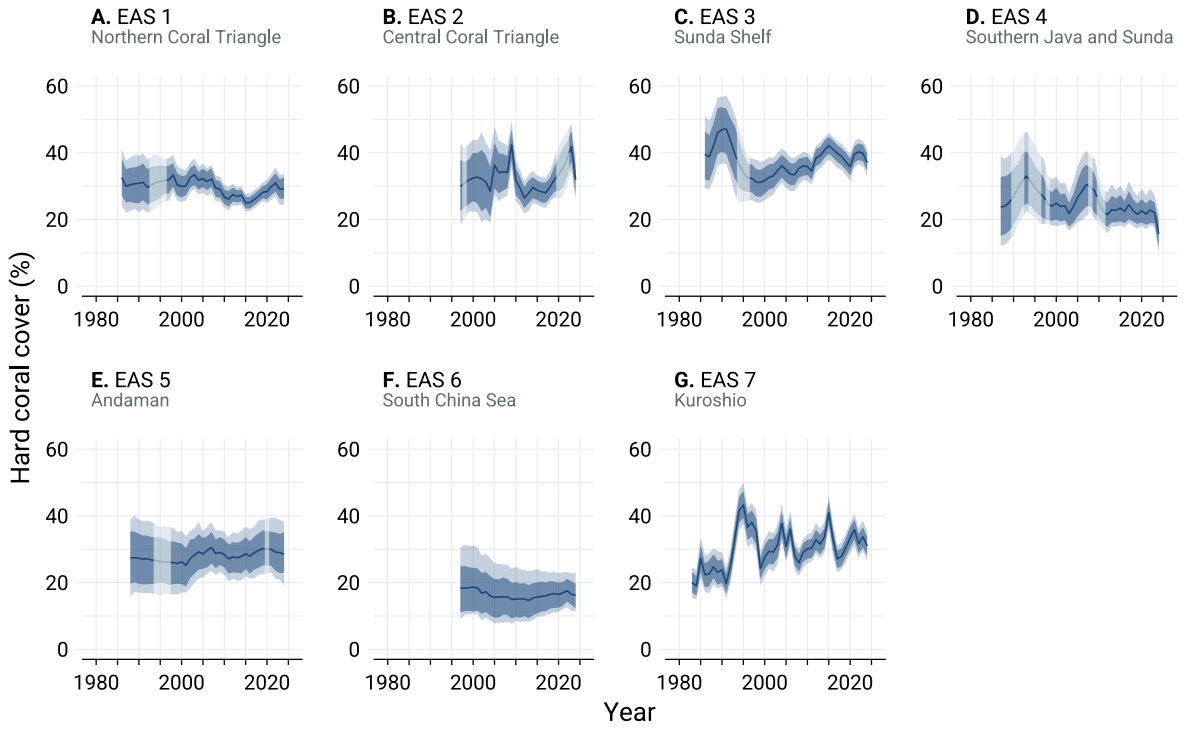
East Asian Seas 2 - Central Coral Triangle

In the Central Coral Triangle subregion, the results showed no evidence of a change in hard coral cover between 1997–2009 and 2020–2024. However, marked post-disturbance variability was observed, particularly following the second global bleaching event in 2010. Hard coral cover declined from 36.2% (CrI $_{95\%}$: 28.4 – 45.2%) in 2005–2009 to 29.0% (CrI $_{95\%}$: 23.8 – 34.4%) in 2010–2014, corresponding to an absolute loss of 7.2 percentage points. This decline was followed by an increase in hard coral cover into the early 2020s.

Macroalgal cover remained relatively low at a median of 1.1% (CrI $_{95\%}$: 0.8 – 1.9%) from 1997 to 2024, with no discernible increase following disturbance events, and no evidence of change in cover, despite a slight increase to 2.3% in 2023.

East Asian Seas 3 - Sunda Shelf

Hard coral cover within the Sunda Shelf subregion was high from 1986 to 1991, with cover peaking at 47.1% (CrI $_{95\%}$: 36.7 – 57.2%) in 1991, but declining thereafter to 31.1% (CrI $_{95\%}$: 25.0 – 37.3%) in 1999, and subsequently recovering to stable levels through the 2000s and early 2010s. There is strong evidence that hard coral cover increased between the 2000s and 2010s, with a gain of 4.6 percentage points over the period. While interannual



▲ **Figure 2.4.7.** Modeled temporal trends of hard coral cover in East Asian Seas (EAS) subregions from 1980 to 2024. The solid line represents the estimated median hard coral cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

variability was evident, hard coral cover remained consistently moderate to high, with a median of 37.5% (CrI $_{95\%}$: 31.3 – 43.7%) from 1986 to 2024. The Sunda Shelf subregion showed no evidence of a change in hard coral cover between 1986–2009 and 2020–2024.

Macroalgal cover remained generally low in the Sunda Shelf subregion, at 2.3% (CrI $_{95\%}$: 1.7 – 2.9%) from 1986 to 2024, showing only minor fluctuations and no evidence of a change over the period.

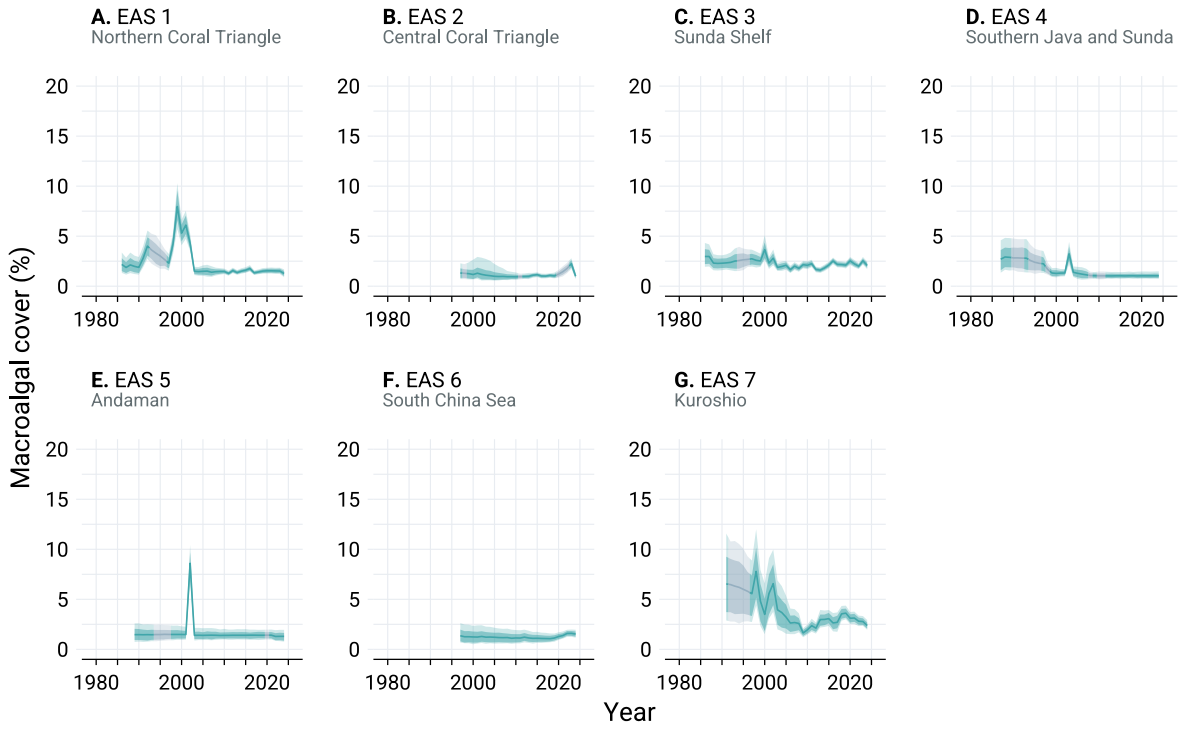
East Asian Seas 4 - Southern Java and Sunda

Despite having the largest data gaps among the subregions, there is strong evidence of a decline in hard coral cover in the Southern Java and Sunda subregion between 1987–2009 and 2020–2024, with a relative decrease in hard coral cover of 22.4%, corresponding to an absolute loss of 6.0 percentage points. This subregion experienced the greatest decline in hard coral cover over the study period across the East Asian Seas region.

Macroalgal cover, although generally low at 1.6% (CrI $_{95\%}$: 1.0 – 2.5%) from 1987 to 2024, decreased sharply around the late-1990s and remained low at around 1.2% thereafter. There is strong evidence that macroalgal cover declined between 1987–2009 and 2020–2024.

East Asian Seas 5 - Andaman

Hard coral cover in the Andaman subregion showed no evidence of a change between 1988–2009 and 2020–2024. Median hard coral cover from 1988 to 2024 was 27.9% (CrI $_{95\%}$: 19.7 – 37.0%) and rem-



▲ **Figure 2.4.8.** Modeled temporal trends of macroalgal cover in East Asian Seas (EAS) subregions from 1980 to 2024. The solid line represents the estimated median macroalgal cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

ained relatively stable throughout the entire study period, with moderate interannual variability.

Except for a peak in 2002 at 8.6% (CrI $_{95\%}$: 6.8 – 10.5%), macroalgal cover remained consistently low between 1988 and 2024. There is also weak evidence of a decline between 1988–2009 and 2020–2024, corresponding to a relative decrease of 25.4%.

East Asian Seas 6 - South China Sea

The South China Sea subregion is characterised by consistently low hard coral cover, compared to other subregions, with a median of 16.4% (CrI $_{95\%}$: 9.7 – 25.6%) from 1997 to 2024. There is no evidence of a change in hard coral cover between 1997–2009 and 2020–2024 (but see Xu *et al.* [278]). The reefs within the South China Sea subregion stand out for their low absolute hard coral cover and relative stability over time, compared to the other subregions.

Macroalgal cover remained low at 1.2% (CrI $_{95\%}$: 0.7 – 2.0%) from 1997 to 2024 and was largely stable throughout the entire period, with no evidence of a change between 1997–2009 and 2020–2024.

East Asian Seas 7 - Kuroshio

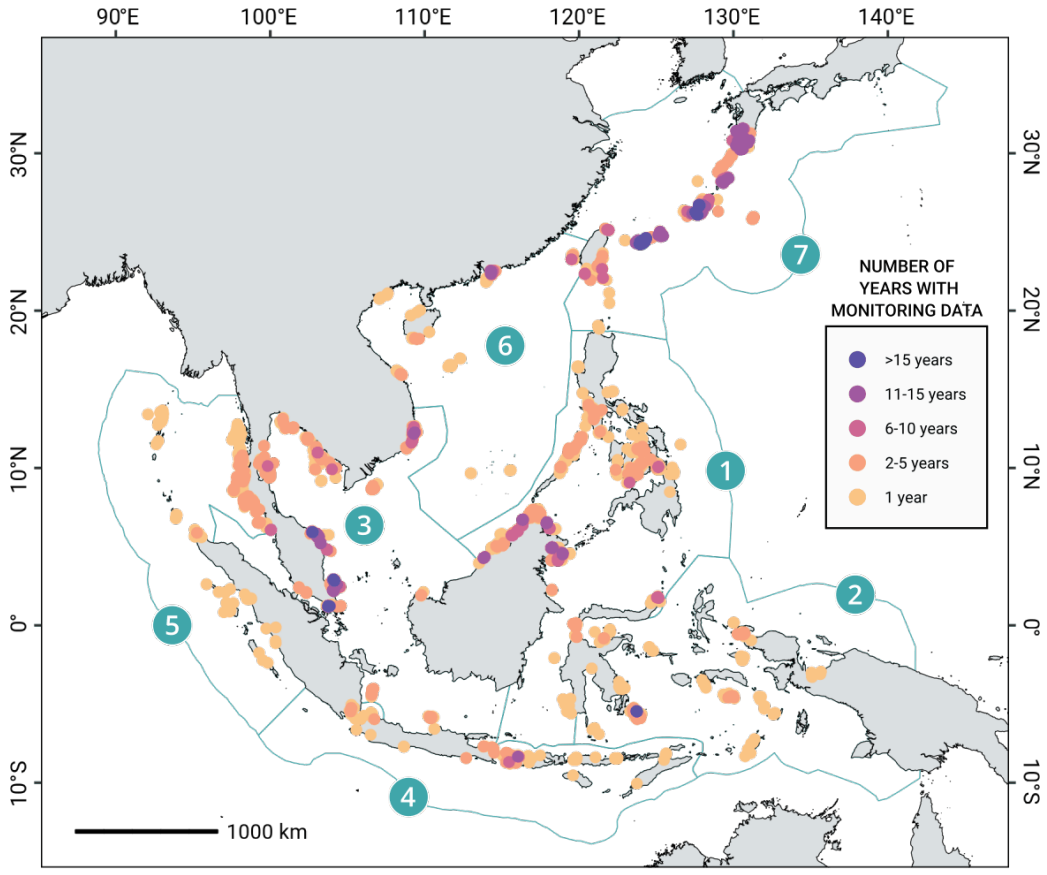
The Kuroshio subregion exhibited substantial interannual variability in hard coral cover, but also recorded one of the largest increases among the East Asian Seas subregions between 1983 and 2024. Indeed, there is strong evidence that hard coral cover increased between 1983–2009 and 2020–2024, by 4.0 percentage points, corresponding to a relative gain of 13.7%. However, substantial declines in hard coral cover

were observed in 1998–1999 (-11.2 percentage points) and 2015–2017 (-13.8 percentage points), potentially associated with the first and third global bleaching events.

Macroalgal cover was relatively high and variable from 1991 to 2004, with a median of 5.6% ($CrI_{95\%}$: 2.7 – 9.3%), but declined over the 2010s. There is strong evidence of a decline between 1991–2009 and 2020–2024, with a 42.1% relative decrease. Among the seven East Asian Seas subregions, the Kuroshio subregion showed the clearest inverse temporal relationship between hard coral cover and macroalgal cover.

DESCRIPTION OF MONITORING DATA

Temporal trends in the percentage cover of benthic categories were estimated using data from 29 datasets encompassing 5,360 monitoring sites. These data were collected through 19,994 surveys conducted between 1983 and 2025, with 90% of surveys taking place after 2001 (Figure A.1). Of all surveys carried out in the region between 1983 and 2025, 43.1% (representing 33.2% of surveyed sites) were conducted in subregion 7, whereas only 2.2% (4.3% of sites) were conducted in subregion 5.



▲ **Figure 2.4.9.** Spatio-temporal distribution of benthic cover monitoring sites across the East Asian Seas region and respective subregions. Light blue polygons and numbers represent subregions. Subregion 1 = Northern Coral Triangle, Subregion 2 = Central Coral Triangle, Subregion 3 = Sunda Shelf, Subregion 4 = Southern Java and Sunda, Subregion 5 = Andaman, Subregion 6 = South China Sea, Subregion 7 = Kuroshio. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region.

In addition, just over 50% of the surveys were surveyed only once, and another 27.4% surveyed between 2 to 5 years (Table A.4).

CONCLUSIONS

The results indicate that although hard coral cover trends were highly variable across subregions, they were generally moderated at the regional scale due to spatial averaging of aggregated data. For example, declines in Southern Java and Sunda and parts of the Northern Coral Triangle were offset by gains in the Central Coral Triangle and Kuroshio, resulting in relatively stable regional averages despite substantial subregional changes. This suggests that apparent regional stability may mask important local variability and vulnerability. However, resolving such patterns will require consistent, fine-resolution data, which are currently lacking at the regional scale.

Across all subregions, bleaching-associated change in 1998 appears as a pulse rather than a phase shift. Disturbance signals were evident around 1998–2000, but reefs across subregions largely re-equilibrated thereafter. Notably, the reefs of the East Asian Seas appear to have transitioned from poorly resolved baseline conditions (pre-1998) to a period of increased hard coral cover with relative persistence through repeated thermal disturbances (1998–2020). Since 2020, however, there are indications of declining hard coral cover, coinciding with chronic heat stress associated with sustained sea surface temperature elevation following the third global bleaching event in 2016, most notably in the Southern Java and Sunda subregion. Whether the post-2020 period reflects a transient disturbance response or the onset of a longer-term regional shift remains uncertain, and will depend on future climate trajectories and the availability of comprehensive monitoring data.

Despite considerable subregional differences, macroalgal cover remained consistently low (generally below 5%), particularly after 2000. There is no clear evidence of macroalgal regime shifts; instead, macroalgal increases were short-lived, region-specific, and largely confined to the late 1990s. Importantly, there was no evidence of a macroalgal increase corresponding to declines in hard coral cover. This pattern is especially evident in the Southern Java and Sunda subregion, where hard coral cover declined without a concomitant rise in macroalgal cover.

DATA CONTRIBUTORS

Muhammad Abrar, Yang Amri Affendi, Abigail Alling, Put Jr. Ang, Maria Vanessa Baria-Rodriguez, Victor Brun, Anna Celis, Chaolun Allen Chen, Loke Ming Chou, Apple Pui Yi Chui, Luigi Colin, Alistair Cole, Frederick Chong, Kitty Currier, Laure de Ville d'Avray, Orla Doherty, Graham J. Edgar, Jan Freiwald, Giyanto, Rickdane Gomez, Danwei Huang, Hui Huang, Lintao Huang, Ya-Yi Huang, Lei Jiang, Yuko Kitano, Taihun Kim, Tadashi Kimura, Lucas Yutaka Kimura, Lena Kraft, Chao-Yang Kuo, Jiansheng Lian, Zau Lunn, Tom Martin, Jenny Mihaly, Carol Milner, Tamal Mondal, Takashi Nakamura, Cleto L. Nañola, C.S. Lionel Ng, Van Long Nguyen, Alex H. O'Brien, Elizabeth Oh, Heung-sik Park, Chelladurai Raghunathan, Marlen Schlotheuber, Rikoh Manogar Siringoringo, Rick D. Stuart-Smith, Makamas Sutthacheep, Frédéric Tardieu, Minh Quang Thai, Ray Jing Tong, Tai Chong Toh, Karenne Tun, Ouk Vibol, Si Tuan Vo, Christian R. Voolstra, Zarinah Waheed, Thamasak Yeemin, Leonora Yeo.

Access to online supplementary materials

The modelled benthic cover data, detailed outputs from the data analyses, and the CRediT author contribution statement are available on Zenodo.

CASE STUDY 5

Indigenous stewardship of coral reefs: the Tagbanwa of Calawit Island, Palawan

Authors*

Chris **Poonian**[1], Roy **Dabuit**[2]

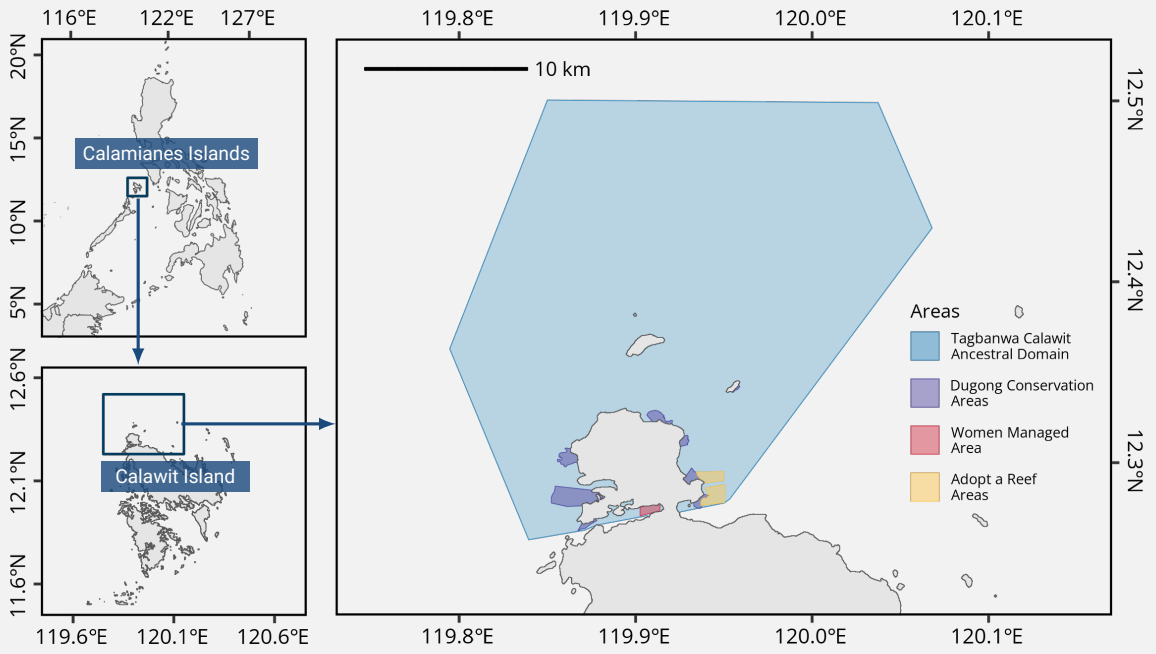
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Across the tropics, coral reefs have long been stewarded by Indigenous peoples and local communities, whose knowledge systems, cultural values, and customary governance have shaped the sustainable use and protection of marine ecosystems for generations. In many places, these systems predate modern conservation frameworks by centuries, embedding ecological understanding within spiritual beliefs, social norms, and daily livelihoods. Yet, despite growing recognition of their effectiveness, Indigenous stewardship has often been overlooked or undervalued in formal reef monitoring and management. As global efforts intensify to halt reef degradation and biodiversity loss, there is an urgent need to recognise, support, and integrate these knowledge systems alongside scientific approaches. Integration of knowledge systems not only strengthens conservation outcomes, but also upholds the rights, identities, and resilience of the communities who have long served as guardians of coral reef ecosystems.

The Municipality of Busuanga is part of the Calamianes Islands, in the province of Palawan, Philippines. It lies within the Coral Triangle, a region of globally significant marine biodiversity. Busuanga's coral reefs have an average hard coral cover of around 35% and high reef fish diversity, estimated at 53 species per 500 m^2 [279]. Calawit Island, situated northwest of Busuanga, supports extensive mangroves, seagrass beds, and coral reefs, and is a key habitat for the dugong (*Dugong dugon*) (Figure C5.1; [280]). The island is part of the ancestral homelands of the Tagbanwa people, whose history has been profoundly shaped by displacement and resilience.

In 1976, the Tagbanwa were forcibly relocated from the island by the Marcos government to create an African game reserve with the island becoming home to imported animals including giraffes, impala, gazelles and zebras. The Calawit Tagbanwa were resettled in nearby Culion. However, the land provided was unsuitable for habitation and agriculture, leading to many years of struggle to legally return to Calawit Island. They were eventually successful in reclaiming their settlement rights in 2008, when they were awarded a Certificate of Ancestral Domain Title from the Philippines National Council of Indigenous Peoples, which also included over 50,000 ha of ancestral waters (Figure C5.1). However, it was not until 2015 that all outstanding challenges to their claim were dismissed by the Supreme Court.

*This study was made possible with support from the United Nations Environment Programme (UNEP).



▲ **Figure C5.1.** Location of the different management areas within the Tagbanwa Calawit Ancestral Domain, Calamianes Islands, Philippines.

The Tagbanwa have long viewed care for their environment as an inherited responsibility. Today, they manage their coastal habitats through a formal Indigenous Peoples Organization and an Ancestral Domain Sustainable Development and Protection Plan (ADSDPP), guided by traditional systems of governance and led by the Indigenous Political Structure and general assembly. Key initiatives of the SDPP include the implementation of community-based dugong-watching guidelines in demarcated Dugong Conservation Areas with local supervision and a fee system; establishment of a 130-hectare women-managed area for farming window-pane oysters (*cachipay*, *Placuna* spp.); and designation of no-take zones where only snorkeling and diving are permitted (Figure C5.1). Through partnerships with local non-government organizations (NGOs) and government agencies, the Tagbanwa are integrating traditional ecological knowledge with formal monitoring protocols, ensuring data from their ancestral domain contributes to regional biodiversity and habitat assessments. For example, an "adopt a coral reef" scheme allocates approximately one hectare of reef within no-take zones to individual clans or families who take responsibility for its protection and regular observation, reinforcing accountability and intergenerational knowledge transfer (Figure C5.1). Endangered species such as dugongs, and critical habitat such as seagrass beds, are monitored by teams of Tagbanwa and supported through partnerships with NGOs and academic institutions.

Across the ancestral waters, harvesting or selling marine resources such as sand (*buhangin*), rock (*bato*), mangrove (*bakawan*), shells (*samong*), coral (*bahura*), sea urchin (*tirik*), and lobster (*banagan*) is prohibited, as are destructive practices such as dynamite, cyanide, or muro-ami fishing. Zoning of the ancestral waters is guided by traditional beliefs, such as certain reefs and rock outcrops that are recognized as *yungto* — sacred places guarded by powerful sea spirits like the *mamalyaw*, a vast creature said to reveal itself when its domain is disturbed. Elders recount that boats may sink or fishers lose their way if they enter without ritual consent, affirm-

ing deep spiritual reverence for coral reefs as living guardians of the sea. Corals themselves hold cultural significance; for instance, octocorals are traditionally used to ward off malevolent spirits, hung on the doorways of Tagbanwa homes, or burnt before important fishing trips to ensure safe passage and good fortune.

The Calamian Tagbanwa case demonstrates how Indigenous stewardship of coastal ecosystems can simultaneously sustain biodiversity and cultural survival. It underscores the value of legal recognition of ancestral rights, participatory management rooted in customary law, and the complementarity between Indigenous and scientific knowledge in achieving long-term conservation objectives.

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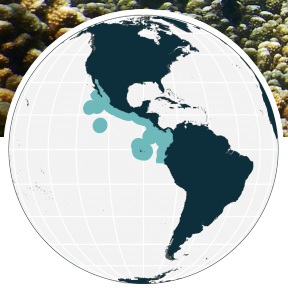
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Cortez rainbow wrasses (*Thalassoma lucasanum*) over a Pocilloporid reef, La Paz, Gulf of California
Credit: Manuel Olán-González



EASTERN TROPICAL PACIFIC

Authors

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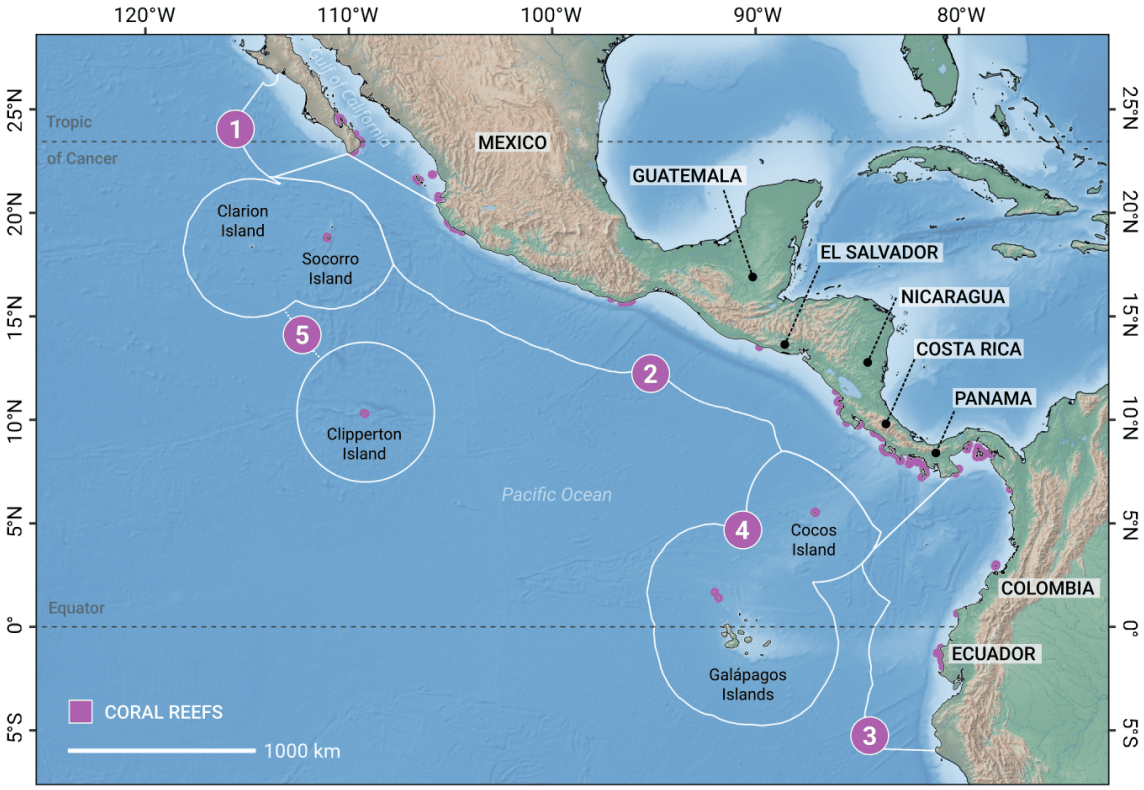
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INTRODUCTION

The Eastern Tropical Pacific GCRMN region extends over a maritime area of approximately 4.8 million km^2, ranging across 30° of latitude and 41° of longitude. The region hosts 593 km^2 of coral reefs, which represent 0.2% of the world's coral reef extent, and is the smallest GCRMN region by coral reef extent. In addition to Clipperton Island, a French overseas territory, coral reefs occur within the exclusive economic zones of six other countries within the region, ranging from Mexico in the north to mainland Ecuador in the south and from Colombia in the east to Mexico (Clarion Island) in the west (Figure 2.5.1). The Chilean oceanic islands of Rapa Nui (Easter Island) and Salas y Gómez have recently been considered as part of the Eastern Tropical Pacific, extending its boundary to ~27° S [281], however, these two locations are not included within this chapter.



▲ **Figure 2.5.1.** The distribution of coral reefs within the Eastern Tropical Pacific region and respective subregions. White polygons and numbers represent subregions. Subregion 1 = Southern Baja California, Subregion 2 = Mesoamerican Pacific, Subregion 3 = Panama Bight/Guayaquil, Subregion 4 = Galápagos and Cocos, Subregion 5 = Revillagigedo and Clipperton. Coral reefs are delineated with a 20 km buffer to enhance interpretation, and only coral reefs within the region are represented. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region.

THREATS AND DISTURBANCES

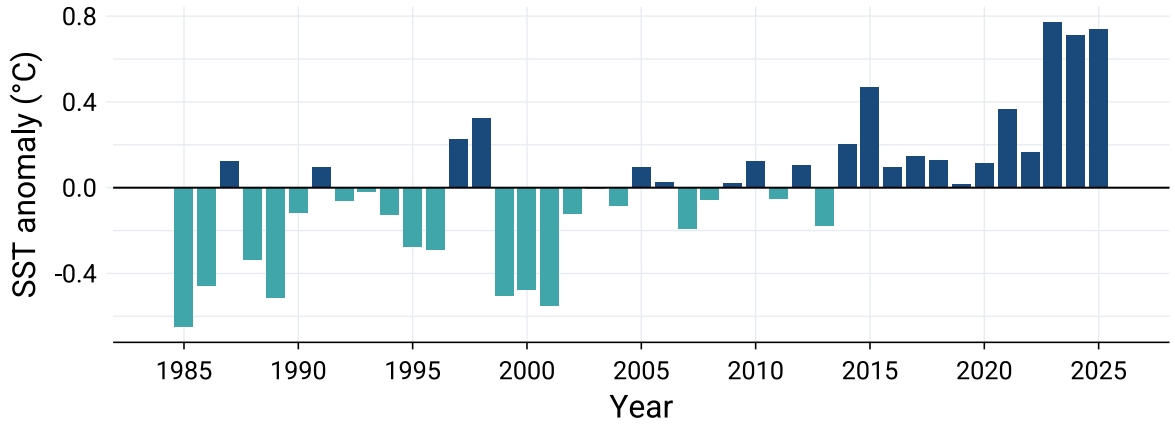
Heat stress

From 1985 to 2025, the long-term average sea surface temperature (SST) over Eastern Tropical Pacific coral reefs was 27.85 °C. During this period, regional SST increased by 0.79 °C, an average warming rate of 0.20 °C per decade (Table A.3). A significant warming trend since 2000 features increasingly frequent and larger positive anomalies, alongside diminishing negative anomalies (Figure 2.5.2). Notably, the period 2023–2025 exhibits the highest positive anomalies recorded.

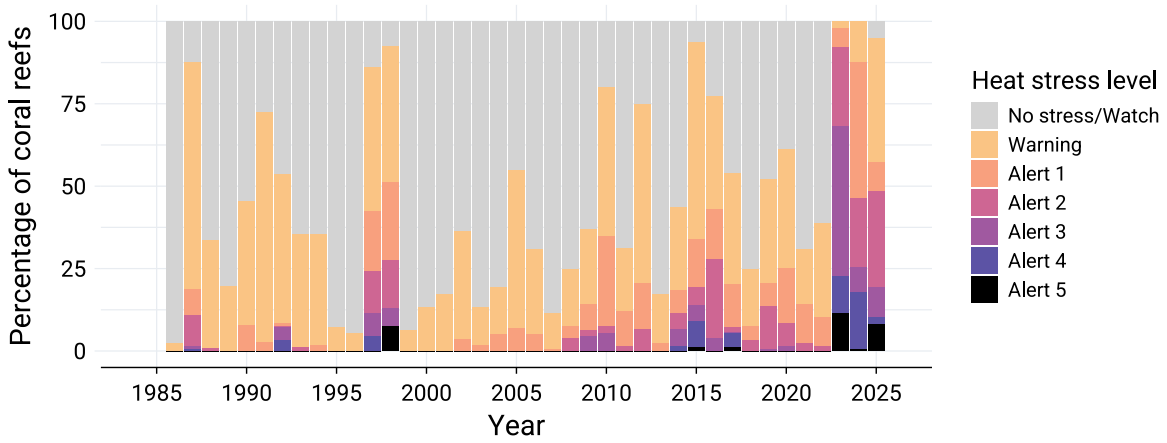
Inter-annual SST variations linked to the ENSO cycle remain the primary drivers of hard coral cover decline and recovery in the Eastern Tropical Pacific. Generally, larger positive SST anomalies (Figure 2.5.2) increase the percentage of reefs affected by heat stress, which can lead to extensive coral bleaching and mortality (Figure 2.5.3).

While thermal impacts are spatially variable, the most severe, widespread El Niño events occurred in 1982–1983, 1997–1998, 2015–2016 and 2023–2024. Across all these events, reefs near the Eastern Tropical Pacific latitudinal limits (Galápagos Islands, Southern Baja California, and continental Mexico) suffered the greatest and most widespread coral mortality and long-term consequences, including increased bioerosion, particularly after the 1982–1983 event [282]. However, impacts can also be highly localized. During the 2015–2016 event, high coral mortality was observed in the Gulf of Chiriquí (Panama), Islas Marías (Mexico), Clipperton Atoll, and Costa Rica, but was considerably less severe elsewhere. Additionally, anthropogenic climate change exacerbates El Niño impacts by raising baseline ocean temperatures, thereby increasing bleaching frequency and severity [282–284].

Conversely, La Niña introduces unique but similarly impactful stressors. Plunging temperatures can trigger cold-water bleaching, which caused significant coral mortality during the 2007–2008 event in the Galápagos and in 2008 and 2011 in the Gulf of California [285]. Furthermore, La Niña frequently drives low sea-level stands, increasing the probability of subaerial exposure during extreme low tides and resulting in massive coral mortality on shallow reef flats [284].



▲ **Figure 2.5.2.** Annual mean sea surface temperature (SST) anomalies (°C) over the coral reefs of the Eastern Tropical Pacific region from 1985 to 2025. SST anomalies were calculated based on the long-term average over the period 1985–2025. Positive values (red bars) indicate SST warmer than the long-term average, whereas negative values (blue bars) indicate SST cooler than the long-term average. Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).



▲ **Figure 2.5.3.** Percentage of the coral reefs within the Eastern Tropical Pacific region experiencing heat stress levels per year, from 1986 to 2025. No stress/Watch (DHW = 0); Warning (possible bleaching, $0 < \text{DHW} < 4$); Alert 1 (reef-wide bleaching, $4 \leq \text{DHW} < 8$); Alert 2 (reef-wide bleaching and mortality of heat-sensitive corals, $8 \leq \text{DHW} < 12$); Alert 3 (multi-species mortality, $12 \leq \text{DHW} < 16$); Alert 4 (severe multi-species mortality, $16 \leq \text{DHW} < 20$); and Alert 5 (near-complete mortality, $\text{DHW} \geq 20$). Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).

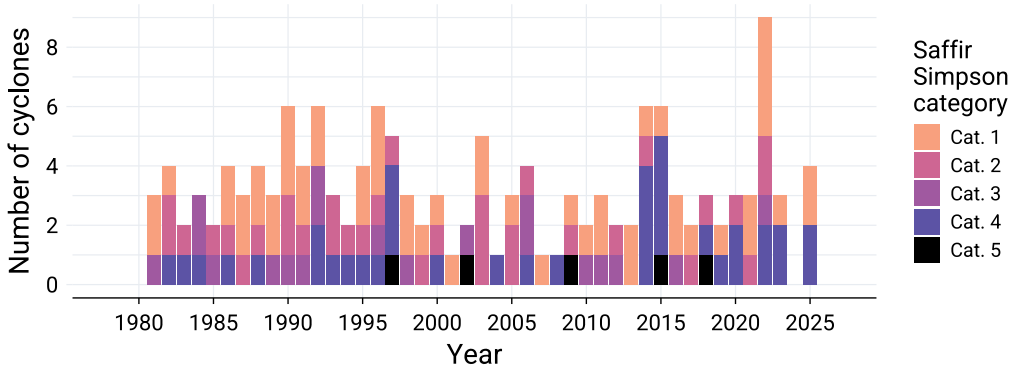
Cyclones

Cyclones in the Eastern Tropical Pacific originate north of 10° N and west of 90° W, and move towards the north-west, affecting mostly the oceanic islands of Revillagigedo and Clipperton, and the continental coasts of Nicaragua, El Salvador, Guatemala, and Mexico. The southern half of the region, including Costa Rica, Panama, Colombia, Ecuador, and their oceanic islands, is not affected by cyclones [286]. From 1980 to 2025, a total of 146 cyclones passed within 100 km of Eastern Tropical Pacific coral reefs, averaging 3.2 $\pm$ 1.8 SD cyclones per year (Figure 2.5.4). The cyclone with the highest sustained wind speed during this period was the cyclone Linda (September 1997) which passed 24 km from a reef with a wind speed of 274 $km.h^{-1}$ (Figure 2.5.5). Although the 100 km threshold was selected, cyclones passing at a greater distance, not shown in Figure 2.5.5, may also impact coral reefs [108].

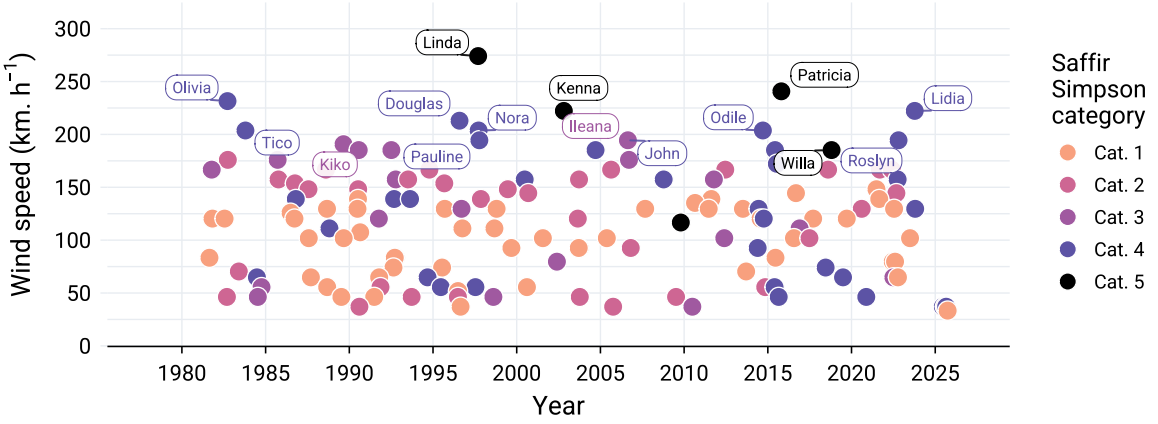
Other regional pressures

Shallow coral reef flats are subaerially exposed during extreme low tides across much of the equatorial Eastern Tropical Pacific, including in Ecuador, Colombia, Panama, and Costa Rica. With tidal amplitudes reaching up to 6 m, reefs in Panama and Colombia are particularly susceptible. At Saboga Island (Panama), for instance, at least 10 significant emersion events have occurred over a 24-year period [287]. These events, often intensified by La Niña's lower sea level stands [284], cause noticeable coral bleaching and mortality through desiccation, extreme heating, intense solar radiation, and osmotic stress from torrential rain during exposure [288, 289].

Upwelling significantly shapes Eastern Tropical Pacific coral reefs, acting as both a critical stressor and a potential refuge [290, 291]. Driven by seasonal trade winds that displace surface waters and force the shoaling of a shallow thermocline, upwelling introduces cool, low-oxygen deep waters into shallow habitats. The associated influx of nutrients and CO_2 lowers local pH, driving ocean acidification. Such conditions pose a severe threat to Eastern Tropical Pacific coral reefs [292] by lowering aragonite saturation states and inhibiting calcification. Reduced calcification not only stunts coral growth,



▲ **Figure 2.5.4.** Number of cyclones that passed within 100 km of the coral reefs of the Eastern Tropical Pacific region from 1980 to 2025. Colors correspond to the Saffir–Simpson category of the cyclone along its entire track. Derived from IBTrACS data (Knapp *et al.* [118]; see Materials and Methods).



▲ **Figure 2.5.5.** Maximum sustained wind speed of cyclones that passed within 100 km of a coral reef in the Eastern Tropical Pacific region between 1980 and 2025. Colors correspond to the Saffir–Simpson category of the cyclone along its entire track. However, the values of sustained wind speed are extracted from the nearest cyclone position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir–Simpson scale (*i.e.* 119 km.h^{-1}). Derived from IBTrACS data (Knapp *et al.* [118]; see Materials and Methods).

but also weakens the submarine cementation necessary to maintain reef frameworks [293, 294].

Finally, as coastal human populations expand, reefs face escalating localized threats, including overfishing, pollution, eutrophication, and increased sedimentation driven by coastal development [295].

STATUS OF CORAL REEFS IN THE REGION

The Eastern Tropical Pacific presents one of the most marginal environments for coral reef development in the world, yet coral reefs in the region exhibit remarkable resilience, consistently achieving rapid recovery in the wake of recurrent disturbances [284, 296]. Coral reefs in the Eastern Tropical Pacific are characterized by their low coral species diversity, yet they have a high potential for

growth, often attaining high coral cover before being impacted by recurrent, large-scale natural disturbances like El Niño events and tidal emersion events. Consequently, live hard coral and algal cover are often highly dynamic. In addition, coral reefs are scarce, small, and patchy, making hard coral cover data highly spatially variable.

Hard coral cover

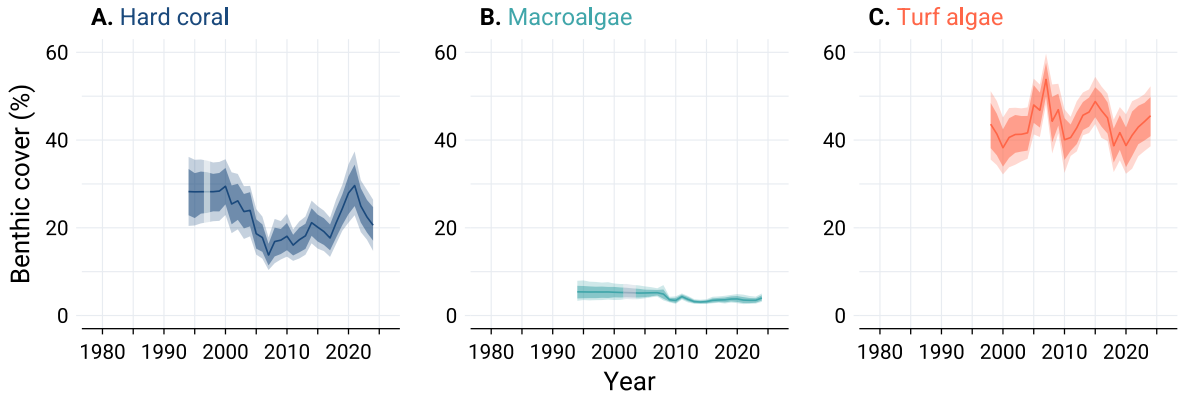
The data show no evidence of a change in hard coral cover between 1994–2009 and 2020–2024. However, this comparison masks periods of decline and increase, as hard coral cover throughout the Eastern Tropical Pacific has been highly dynamic over the last three decades (Figure 2.5.6A).

Hard coral cover declined from 28.1% in 1994 to 13.8% in 2007, corresponding to a loss of 14.3 percentage points, representing a 50.8% relative decline. This decline was followed by a fluctuating recovery, with strong evidence of hard coral cover increase from 2007 to 2021, rising from 15.9 percentage points over the period. Subsequently, however, hard coral cover was reduced to 20.6% by 2024 (Figure 2.5.6A).

Historically, periods of hard coral cover decline in this time series have been driven by recurrent natural disturbances, notably heat stress from the 1997–1998 El Niño (which most severely impacted reefs at the region's latitudinal edges) and significant tidal emersion events in the Panama Bight [289]. Mirroring these historical patterns, the most recent downturn (2021 to 2024) was driven by an extremely strong marine heatwave during the 2023–2024 El Niño, exacerbated by severe tidal emersion events at the beginning of both years.

Macroalgal cover

In contrast to hard coral cover, macroalgal cover has remained consistently low in the region, with a median of around 4.4% from 1994 to 2024 (Figure 2.5.6B). This low macroalgal cover is likely due both to the dominance of turf algae (Figure 2.5.6C) and to intense herbivory. Macroalgal cover remained around 5.2% between 1994 and 2008, declined to 3.4% in 2010, and has since ranged between 3.1% and 4.3% (Figure 2.5.6B).



▲ **Figure 2.5.6.** Modeled temporal trends of benthic cover in the Eastern Tropical Pacific region from 1980 to 2024 for hard coral (A), macroalgae (B), and turf algae (C). The solid line represents the estimated median benthic cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

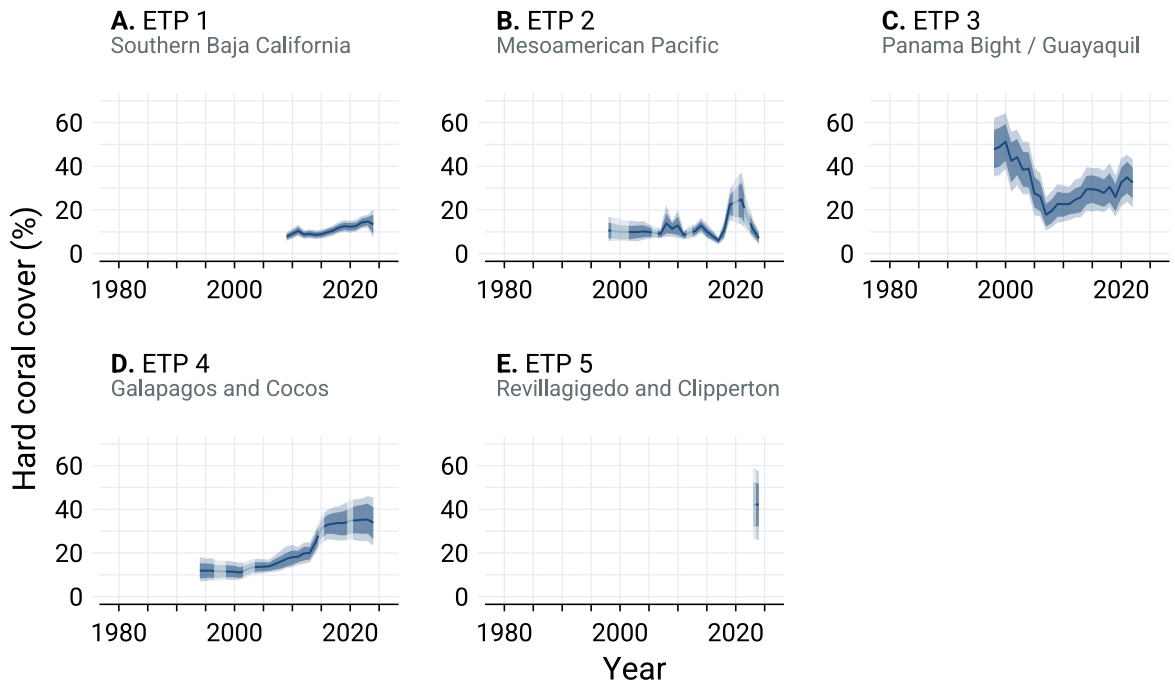
Turf algal cover

In the Eastern Tropical Pacific, turf algal cover has been several orders of magnitude higher and much more dynamic compared to macroalgal cover (Figure 2.5.6C). From 1994 to 2024, turf algal cover ranged between 38.2% and 53.8%, and showed no evidence of a change between the 1994–2009 and 2020–2024 periods (Figure 2.5.6C). There was strong evidence that turf algal cover increased between 1994 and 2007, rising by 10.2 percentage points over the period. This was followed by a period of decline between 2007 and 2014, with an absolute decline of 7.3 percentage points. Turf algal cover then remained relatively stable between 2014 and 2017, at around 46.7%. There was strong evidence of an increase in turf algal cover between 2018 and 2024, rising from 38.7% in 2018 to 45.5% in 2024, corresponding to an absolute gain of 6.6 percentage points (a 17.4% relative change).

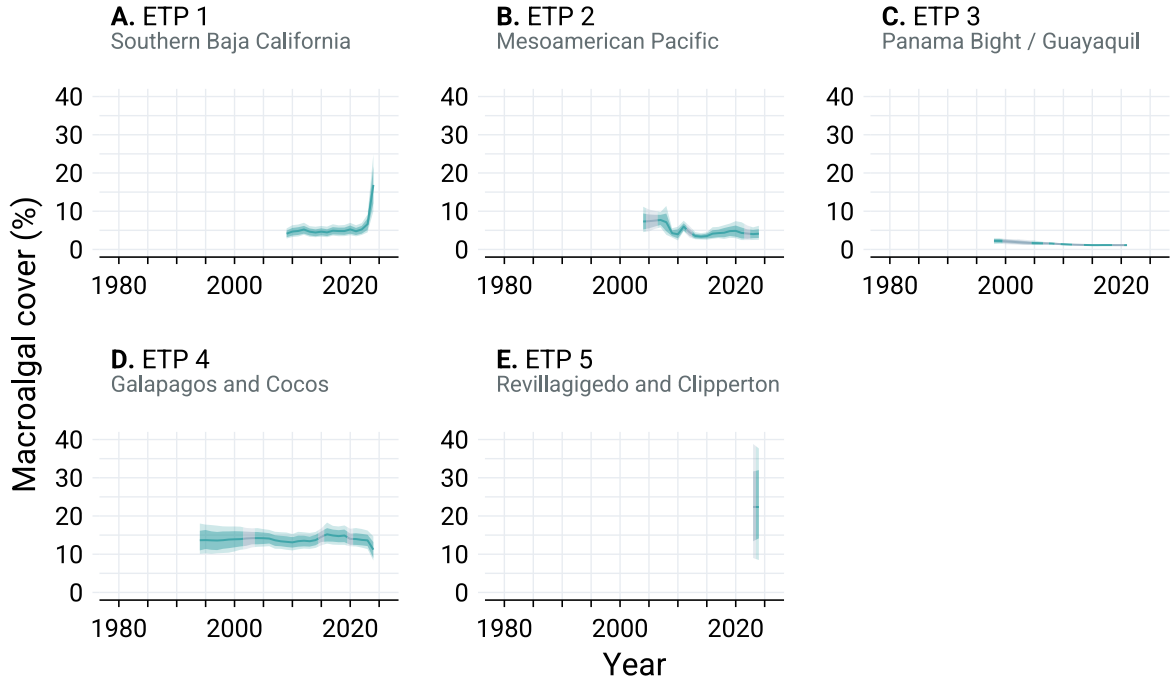
STATUS OF CORAL REEFS IN THE SUBREGIONS

ETP 1 - Southern Baja California

Hard coral cover within the Southern Baja California subregion demonstrates an atypical but general upward trend over the last 15 years. There was strong evidence that hard coral cover increased from 2009 to 2020–2024, rising from 7.8% in 2009 to a peak of 14.7% in 2023 (Figure 2.5.7A). This represents an absolute gain of 5.6 percentage points, corresponding to a 73.4% relative increase. Although largely consistent, this recovery experienced several minor interruptions, including a decline associated with cold-water bleaching in 2011–2012, a growth plateau around 2021, and a recent drop to 13.5% in 2024 following the severe 2023–2024 El Niño event.



▲ **Figure 2.5.7.** Modeled temporal trends of hard coral cover in Eastern Tropical Pacific (ETP) subregions from 1980 to 2024. The solid line represents the estimated median hard coral cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.



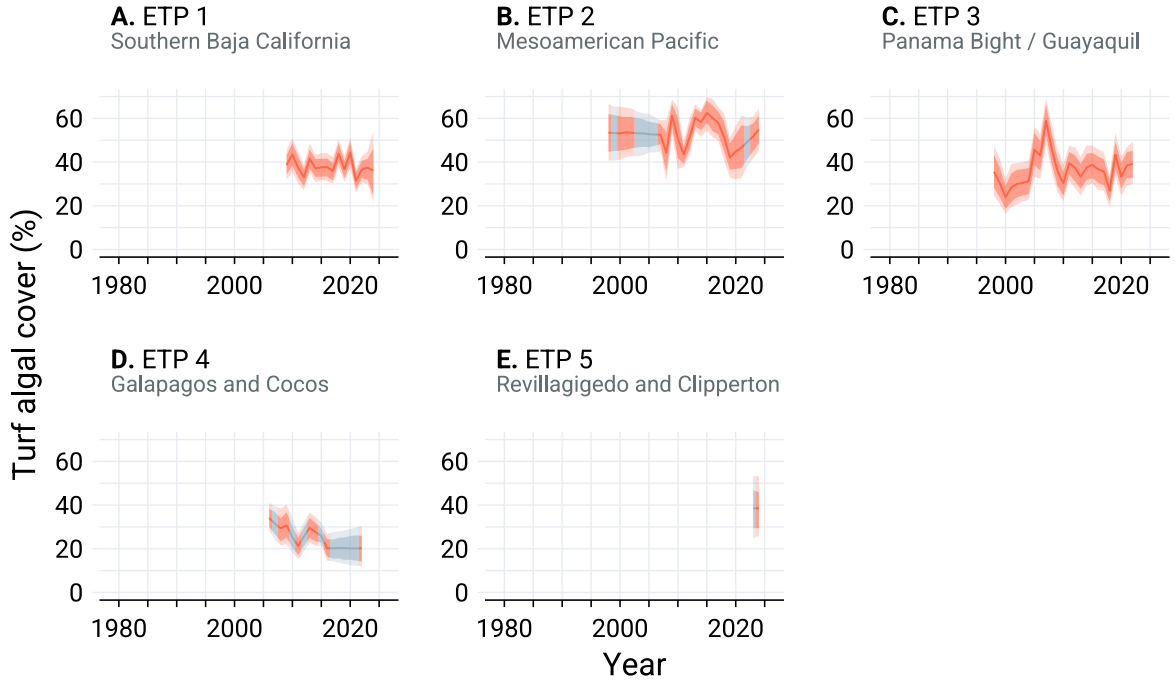
▲ **Figure 2.5.8.** Modeled temporal trends of macroalgal cover in Eastern Tropical Pacific (ETP) subregions from 1980 to 2024. The solid line represents the estimated median macroalgal cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

Although mass coral mortality events since 2010 freed space for algal colonization, macroalgal cover has remained relatively low and stable at around 4.7% between 2009 and 2021 (Figure 2.5.8A). However, there was strong evidence that macroalgal cover increased between 2021 and 2024, rising from 4.7% to 16.9% over this period. The steepest part of this increase occurred between 2023 and 2024, concurrent with the recent El Niño and severe bleaching-driven hard coral decline (Figure 2.5.8A).

Turf algal cover remained relatively consistent, fluctuating interannually around 38.0%, without exhibiting a clear long-term trend or an apparent response to the 2015 and 2023–2024 bleaching events. Accordingly, there was no evidence of a change in turf algal cover between 2009 and 2020–2024 (Figure 2.5.9A). The relative stability of both macroalgal and turf algal cover likely reflects the consistently abundant herbivorous communities (fish and sea urchins) in the subregion [297, 298]. This supports the hypothesis that the macroalgal peak observed in 2024 may be transient.

ETP 2 - Mesoamerican Pacific

Hard coral cover in the Mesoamerican Pacific subregion generally remained stable with values fluctuating around 10.2% from 1998 to 2018, with no evidence of change over the period (Figure 2.5.7B), though this apparent stability at the subregional scale masks distinct local dynamics. Following a marked subregional low in median hard coral cover of 6.0% in 2017, there was strong evidence of an increase until 2021, when cover reached 24.9%, corresponding to an absolute gain of 18.9 percentage points. However, this transient peak is widely considered to be a sampling artifact resulting from heterogeneity in data collection. Beyond these artifacts, observed changes appear to be strongly driven by heat stress.



▲ **Figure 2.5.9.** Modeled temporal trends of turf algal cover in Eastern Tropical Pacific (ETP) subregions from 1980 to 2024. The solid line represents the estimated median turf algal cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

There was strong evidence that macroalgal cover declined between 1998–2009 and 2020–2024, with a relative decrease of 39.4%. Macroalgal cover reached its highest values between 1998 and 2008, averaging around 7.4% (Figure 2.5.8B).

Turf algal cover in the subregion remained stable at around 53.1% from 1998 to 2007, but subsequently fluctuated, declining to 42.1% in 2019 before rising to a maximum of 55.0% in 2024 (Figure 2.5.9B). However, there was no evidence of a change in turf algal cover between 1998–2009 and 2020–2024.

ETP 3 - Panama Bight/Guayaquil

Hard coral cover in the Panama Bight/Guayaquil subregion experienced a pronounced decline from 47.7% in 1998 to a historical low of 17.9% in 2007 (Figure 2.5.7C). Accordingly, there was strong evidence of a decline in hard coral cover from the 2000s to the 2010s, corresponding to a decrease of 6.0 percentage points. Hard coral cover subsequently recovered until 2014, reaching 29.5%, a trajectory consistent with observations from Gorgona Island, Colombia [289]. More broadly, there was strong evidence of an increase in hard coral cover between the 2010s and the 2020s, with a gain of 6.1 percentage points.

Macroalgal cover in the Panama Bight/Guayaquil subregion showed strong evidence of a decline between 1998–2009 and 2020–2022, with an absolute decrease of 0.7 percentage points, corresponding to a relative decline of 37.9%. However, the decline from 1.8% to 1.1% also underscores the consistently low levels of macroalgal cover throughout the entire period (Figure 2.5.8C).

Conversely, turf algal cover increased almost continuously from 35.5% in 1998 to a peak of 58.8% in 2007 (Figure 2.5.9C). Between 2007 and 2013, there was strong evi-

dence of a decline in turf algal cover, which fell to 33.2% before fluctuating around 36.7% until 2022.

ETP 4 - Galápagos and Cocos

The Galápagos and Cocos subregion exhibits a remarkable long-term recovery of hard coral cover, facilitated by its remote location shielding it from localized anthropogenic stressors. There was strong evidence of an increase in hard coral cover between 1994–2009 and 2020–2024, with a 21.7 percentage point absolute increase over the period (Figure 2.5.7D). Following an initial period of stability between 1994 and 2007, where hard coral cover averaged at 12.5%, there was strong evidence of an increase from 2007 to 2014. Indeed, hard coral cover rose from 15.0% to 24.9% between these two years, representing a 9.9 percentage point increase (a 66.3% relative change). This trend continued through 2017, when hard coral cover reached 33.4%. From 2018 to 2024, hard coral cover remained relatively stable, with no evidence of a change over the period. In the Galápagos Islands, hard coral cover recovery was exclusively confined to the northern Darwin and Wolf islands. Indeed, coral reefs were functionally lost across most of the archipelago after the catastrophic 1982–1983 El Niño, with their remaining frameworks reduced to rubble by the bioeroding sea urchin *Eucidaris galapagensis* [299].

Macroalgal cover in the Galápagos and Cocos subregion was the highest recorded across the entire Eastern Tropical Pacific region. However, there was no evidence of a long-term change in macroalgal cover between 1994–2009 and 2020–2024, with cover remaining around 13.8% throughout the period (Figure 2.5.8D). Nevertheless, there was strong evidence of a decline in macroalgal cover between 2018 and 2024, from 14.7% to 11.2%, respectively. The long-term stability in macroalgal cover was likely driven by conditions in the Northern Galápagos, where consistently high herbivorous fish biomass and strong hydrodynamic energy promote algal dislodgement [300].

There was no evidence of a change in turf algal cover between 1994 and 2007, during which it remained relatively stable at around 33.9%. Between 2007 and 2011, however, turf algal cover declined to 21.3%, mirroring the increase in hard coral cover observed over the same period. Following a transient increase to 29.6% in 2013, turf algal cover stabilized at around 20.3% from 2016 through 2024 (Figure 2.5.9D). Overall, there was strong evidence that turf algal cover declined between 1994–2009 and 2020–2024, corresponding to an absolute decrease of 12.9 percentage points.

ETP 5 - Revillagigedo and Clipperton

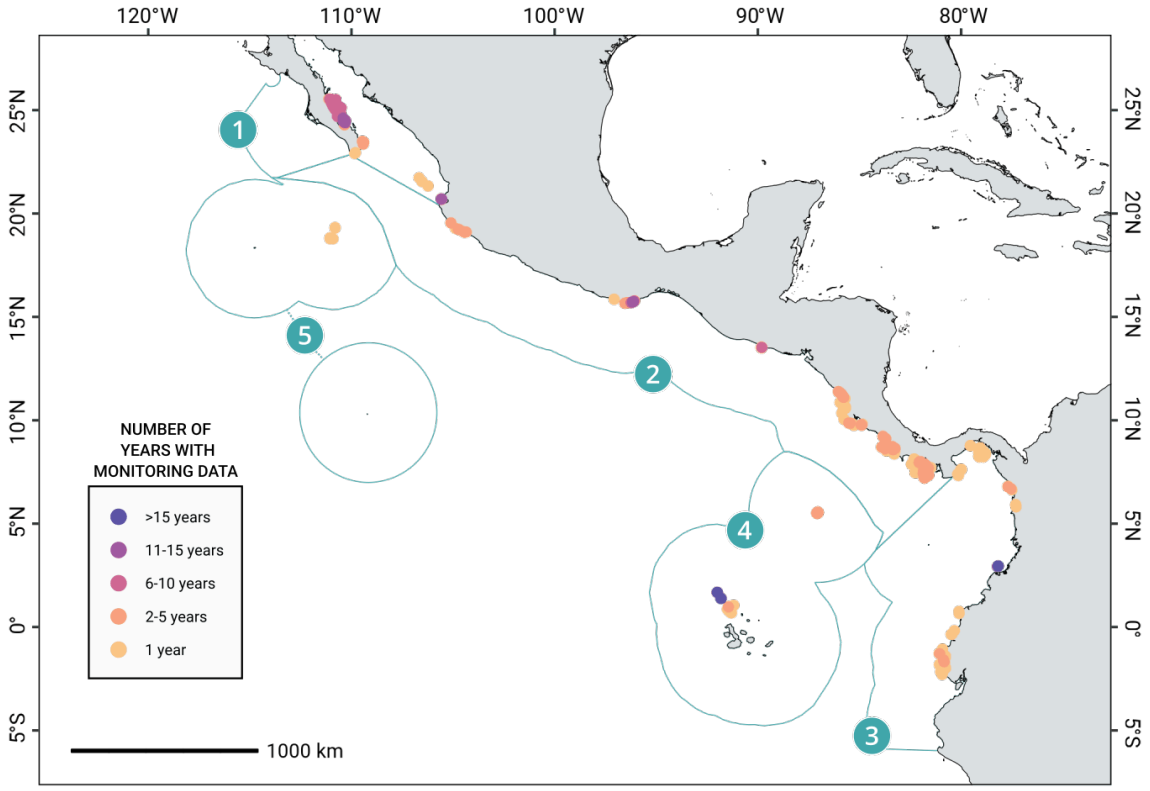
A single 2024 sampling in the Revillagigedo Islands provides the subregion's only data record, revealing median hard coral cover of 42.2% (Figure 2.5.7E), turf algal cover of 38.6% (Figure 2.5.9E), and an Eastern Tropical Pacific record macroalgal cover of 22.4% (Figure 2.5.8E). At Clipperton Atoll, hard coral cover approaches 100% in stable, deep-water refugia [301]. Despite periodic declines from recurrent ENSO anomalies and violent cyclones, overall hard coral cover steadily increased over the last two decades. Estimates of 42.3% at 5 m and 81.3% at 16 m in 1994 [301] rose to 48.7% in 2005 and 58.3% in 2019 [302]. However, severe global disturbances drove a major shift, transforming the coral reef from stress-susceptible branching *Pocillopora* to a highly resilient coral assemblage dominated by massive *Porites* [302].

DESCRIPTION OF MONITORING DATA

Temporal trends in the percentage cover of benthic categories were estimated using data from five datasets, encompassing 781 monitoring sites. These data were collected from 2,666 surveys conducted between 1994 and 2025, with 90% of surveys conducted after 2007 (Figure A.1). Of all surveys carried out in the region between 1994 and 2025, 36.8% (representing 49.4% of surveyed sites) were conducted in subregion 1, whereas only 0.4% (0.6% of sites) were conducted in subregion 5 (Table A.4).

CONCLUSIONS

The Eastern Tropical Pacific stands out as a comparatively resilient region amid the broader global coral reef crisis. Despite suboptimal environmental conditions and severe recurrent disturbances, including ENSO-driven marine heatwaves, cyclones, and extreme tidal emersion events, hard coral cover has shown an upward trajectory in three of the five subregions (Southern Baja California, Panama Bight/Guayaquil, and Galápagos and Cocos). However, this positive trend was likely disrupted by the severe 2023–2024 El Niño event.



▲ **Figure 2.5.10.** Spatio-temporal distribution of benthic cover monitoring sites across the Eastern Tropical Pacific region and respective subregions. Light blue polygons and numbers represent subregions. Subregion 1 = Southern Baja California, Subregion 2 = Mesoamerican Pacific, Subregion 3 = Panama Bight/Guayaquil, Subregion 4 = Galápagos and Cocos, Subregion 5 = Revillagigedo and Clipperton. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region.

Given the Eastern Tropical Pacific has experienced ocean warming rates comparable to those in other regions, the region's resilience can be attributed to its highly dynamic oceanography. The high natural environmental variability of the Eastern Tropical Pacific appears to mitigate the physiological impacts of extreme thermal fluctuations, effectively pre-conditioning corals to endure stress. Furthermore, local adaptation has played a critical role in driving this recovery. Evidence indicates that at some localities, *Pocillopora*, the dominant reef-building coral genus in the region, is shifting its symbiont algal composition toward species of the genus *Durusdinium*, conferring greater tolerance to high-temperature stress [303].

Ultimately, the Eastern Tropical Pacific demonstrates that coral reefs can adapt and persist despite severe environmental pressures. The long-term recovery observed across several subregions reinforces the region's status as a potential biological refuge, offering a uniquely positive perspective and valuable lessons for global reef conservation. However, whether reefs in the Eastern Tropical Pacific will continue to adapt and persist at a pace sufficient to keep up with ongoing global change remains uncertain.

DATA CONTRIBUTORS

Andrés Felipe Acosta-Chaparro, David Alejandro Alonso-Carvajal, Juan José Alvarado, Imelda G. Amador-Castro, Andrea Arriaga-Madrigal, Arturo Ayala-Bocos, William Bensted-Smith, Luis Calderon-Aguilera, Margarita Brandt, Jose Julio Casas-Maldonado, Luis Chasqui, Amilcar Leví Cupul-Magaña, Graham J. Edgar, Jan Freiwald, Diana Isabel Gómez-López, Ollin T. Gonzalez-Cuellar, Rebeca Granja-Fernández, Cristian Jitala, Inti Keith, Mathieu Leray, Andres Lopez-Perez, Priscilla Carlota Martínez-Panizo, Jenny Mihaly, Nicolas Moity, Raúl Navas-Camacho, Elizabeth Oh, Manuel Alejandro Olán-González, Damien Olivier, Sophie Osborn, Caroline Palmer, Georgina Ramírez-Ortiz, Héctor Reyes-Bonilla, Fernando E. Rivera, Favio J. Rivera-Martinez, Alma Paola Rodríguez-Troncoso, Jenny Carolina Rodríguez-Villalobos, Fabián A. Rodríguez-Zaragoza, Laura Sánchez-Valencia, Johanna Segovia-Prado, Rick D. Stuart-Smith, Omar Valencia-Mendez, Fernando A. Zapata.

Access to online supplementary materials

The modelled benthic cover data, detailed outputs from the data analyses, and the CRediT author contribution statement are available on Zenodo.

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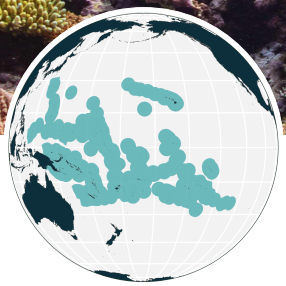
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Coral reef in Ouvéa, New Caledonia
Credit: Sylvain Corbel



PACIFIC

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Citation

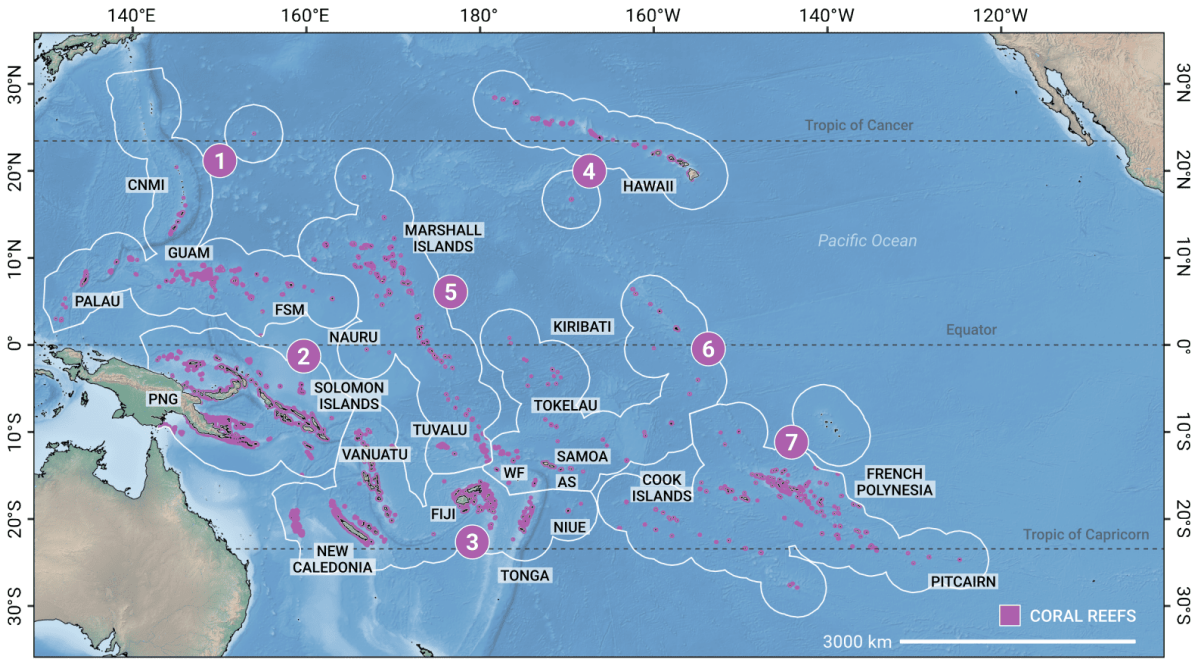
Wicquart, J., Ford, A., Towle, E. K., Heintz, T., *et al.* (2026). Chapter 6 – Pacific. *In González-Rivero, M., et al. Status of Coral Reefs of the World: 2025. Global Coral Reef Monitoring Network and International Coral Reef Initiative.* https://doi.org/10.59387/LFPR6347.PAC

INTRODUCTION

The GCRMN Pacific region extends over a maritime area of around 31.1 million km^2, ranging across 63° of latitude and 106° of longitude. The region hosts 65,350 km^2 of coral reefs, which represent 26.2% of the world's coral reef extent, and is the second largest GCRMN region in terms of coral reef extent. Coral reefs in the Pacific region reside within the exclusive economic zones of 23 countries and territories, ranging from Hawai'i in the North to French Polynesia in the South, and from Palau in the West to the Pitcairn Islands in the East (Figure 2.6.1). For the purpose of this report, the GCRMN Pacific region was divided into seven subregions (Figure 2.6.1). Subregion 2 (Eastern Coral Triangle) hosts 31.2% of coral reefs in the region (20,409 km^2) whereas subregion 6 (Central Polynesia) hosts only 3.5% of coral reefs (2,258 km^2) (Table A.1).

Intense volcanic activity in the Pacific has contributed to shaping a wide diversity of reef geomorphological structures [304, 305]. These range from fringing reefs surrounding the youngest volcanic islands (*e.g.* O'ahu in the main Hawaiian Islands), atolls formed around the oldest island systems (*e.g.* Abaiang in Kiribati), and barrier reefs enclosing lagoons (*e.g.* Babeldaob in Palau).

Due to the vast geographic extent of the Pacific, coral reefs in the region develop under a broad spectrum of environmental conditions, including variability in sea surface temperature, wave energy, and irradiance [306].



▲ **Figure 2.6.1.** The distribution of coral reefs within the Pacific region and respective subregions. White polygons and numbers represent subregions. Subregion 1 = Tropical Northwestern Pacific, Subregion 2 = Eastern Coral Triangle, Subregion 3 = Tropical Southwestern Pacific, Subregion 4 = Hawaiian archipelago, Subregion 5 = Marshall, Gilbert, and Ellis Islands, Subregion 6 = Central Polynesia, Subregion 7 = Southeast Polynesia. Coral reefs are delineated with a 20 km buffer to enhance interpretation, and only coral reefs within the region are represented. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region. AS = American Samoa, CNMI = Commonwealth of the Northern Mariana Islands, FSM = Federated States of Micronesia, PNG = Papua New Guinea, WF = Wallis and Futuna.

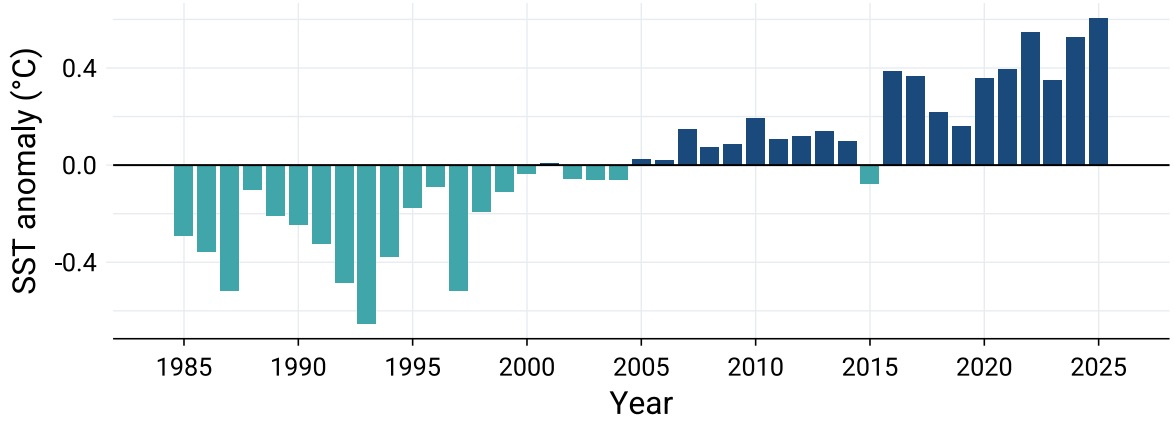
Compared to other GCRMN regions, coral reefs throughout the Pacific generally develop in clear, low-turbidity waters, as much of the region is composed of small islands with limited river catchments, reducing terrigenous sediments and nutrient runoff. Hard coral species richness exhibits a pronounced longitudinal gradient, ranging from highly diverse reef systems in the Eastern Coral Triangle (subregion 2) to markedly lower species richness in the easternmost Pacific reefs, including the Hawaiian archipelago (subregion 4) and Pitcairn Islands (within subregion 7) [307].

As of 2020, approximately 4.35 million inhabitants were living within 5 km of Pacific coral reefs. Between 2000 and 2020, the region experienced a 28.7% increase in populations residing within 5 km of coral reefs, corresponding to an average annual growth of about 50,000 inhabitants. The Eastern Coral Triangle (subregion 2) exhibited the largest absolute population increase over the 2000–2020 period and accounted for the highest number of inhabitants in 2020 (Table A.2). In 2020, and in comparison to other GCRMN regions, the Pacific recorded the highest proportion of human population living in proximity to coral reefs globally [2]. In addition to socio-economic factors, this high degree of population proximity to coral reefs can be explained by geographical constraints, including the limited extent of emergent land in many island states (with the notable exception of Papua New Guinea) and the presence of steep central mountain ranges on high islands, which concentrate settlements along narrow coastal zones (*e.g.* Moorea, in French Polynesia; [308]).

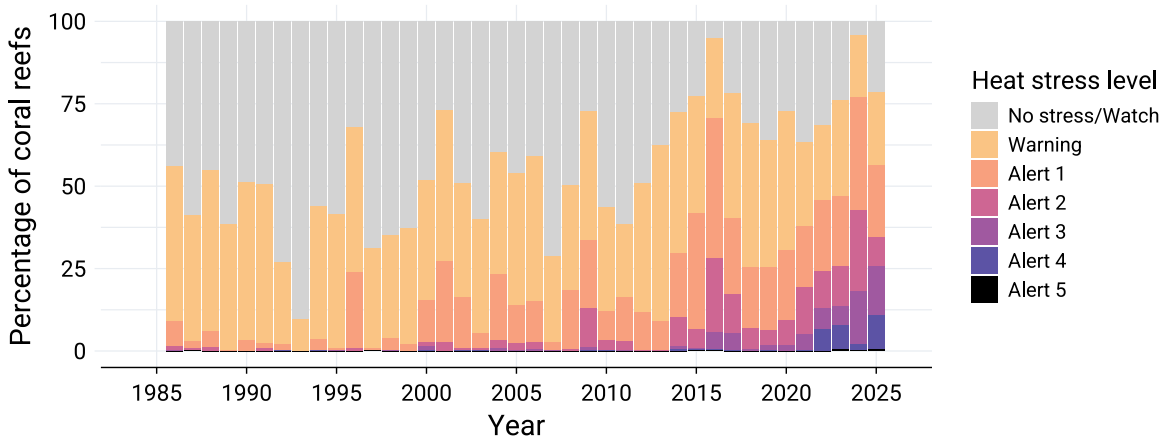
THREATS AND DISTURBANCES

Heat stress

From 1985 to 2025, the long-term average of sea surface temperatures (SST) recorded over coral reefs within the Pacific region was 27.71 °C. During this period, the SST over coral reefs in the region increased by 0.88 °C, corresponding to an average warming rate of 0.22 °C per decade (Table A.3). Increasing warming rates were previously evidenced by Heron *et al.* [309] over a period from 1985–2012, with an estimated warming rate of approximately 0.15 °C per decade across Pacific coral reefs.



▲ **Figure 2.6.2.** Annual mean sea surface temperature (SST) anomalies (°C) over the coral reefs of the Pacific region from 1985 to 2025. SST anomalies were calculated based on the long-term average over the period 1985–2025. Positive values (red bars) indicate SST warmer than the long-term average, whereas negative values (blue bars) indicate SST cooler than the long-term average. Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).



▲ **Figure 2.6.3.** Percentage of the coral reefs within the Pacific region experiencing heat stress levels per year, from 1986 to 2025. No stress/Watch (DHW = 0); Warning (possible bleaching, $0 < \text{DHW} < 4$); Alert 1 (reef-wide bleaching, $4 \leq \text{DHW} < 8$); Alert 2 (reef-wide bleaching and mortality of heat-sensitive corals, $8 \leq \text{DHW} < 12$); Alert 3 (multi-species mortality, $12 \leq \text{DHW} < 16$); Alert 4 (severe multi-species mortality, $16 \leq \text{DHW} < 20$); and Alert 5 (near-complete mortality, $\text{DHW} \geq 20$). Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).

Between 1986 and 2025, major episodes of heat stress affecting Pacific coral reefs occurred in 2000–2001, 2004–2006, 2009–2011, 2014–2017, and 2022–2025 (Figure 2.6.3). Due to the large geographical extent of the Pacific region, these events rarely occur within a single calendar year, unlike in most other GCRMN regions. Instead, heat stress events may begin earlier or persist later across different Pacific subregions. The region's geographic extent also means that reefs are not uniformly exposed to heat stress and therefore experience different levels of risk and impact [31]. Although heat-stress events historically coincided mainly with El Niño phases [310], an increasing proportion of reefs are now exposed to heat stress on an annual basis.

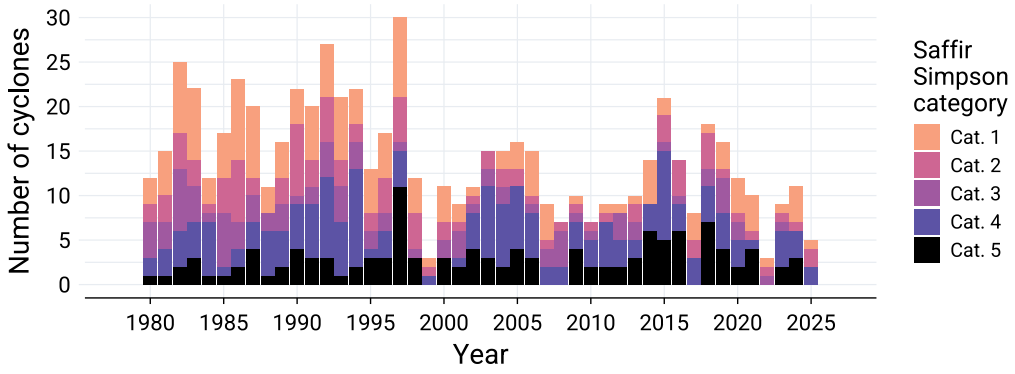
Cyclones

From 1980 to 2025, a total of 649 cyclones passed within 100 km of coral reefs in the Pacific region, representing an average of 14.2 $\pm$ 6.2 SD cyclones per year (Figure 2.6.4). Although the 100 km threshold was selected, strong cyclones passing at greater distances from a coral reef, not represented within Figure 2.6.5, may still have affected reefs through the extreme waves they can generate [108].

Not all coral reefs within the region are exposed to the same risk of cyclones. Coral reefs localised near the equator, such as those in Kiribati and Nauru (Figure 2.6.1), are rarely impacted by cyclones [108]. In contrast, coral reefs in the Tropical Northwestern and Southwestern Pacific (subregions 1 and 3, respectively) are the most exposed to cyclones in the region [44, 108].

Crown-of-thorns starfish

In addition to climate-related disturbances, coral reefs of the Pacific are also impacted by predator outbreaks of crown-of-thorns starfish (CoTS, *Acanthaster* spp.). Several CoTS population outbreaks have been reported throughout the Pacific region over the last decades [311–313]. During these CoTS population outbreaks, densities of CoTS can exceed 1,000 individuals per hectare, a stark contrast to non-outbreak periods which record approximately one individual per hectare [314, 315]. For example, on the coral reefs of Moorea, French Polynesia, a

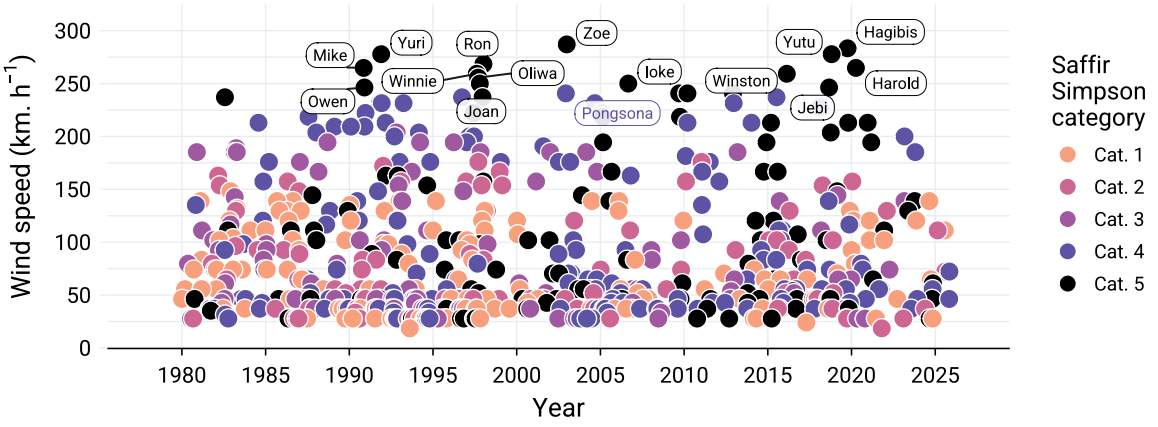


▲ **Figure 2.6.4.** Number of cyclones that passed within 100 km of the coral reefs of the Pacific region from 1980 to 2025. Colors correspond to the Saffir–Simpson category of the cyclone along its entire track. Derived from IBTrACS data (Knapp *et al.* [118]; see Materials and Methods).

CoTS outbreak led to a decrease in hard coral cover from 40% in 2005 to less than 5% in 2010 [314], with Pratchett *et al.* [316] estimating that the outbreak caused hard coral cover to decline from 42.4% to 19.1% within a single year. In another example from Guam, it took 12 years for hard coral species richness, cover, and composition to return to levels comparable to those before the 1968–1969 CoTS population outbreaks [317]. Similarly, surveys conducted in Rarotonga, Cook Islands, estimated that hard coral cover required approximately 17 years to recover to pre-CoTS levels observed in the 1990s [318].

Coastal development and land-based pollution

Over the past few decades, human activities have degraded water quality on Pacific coral reefs. Coastal development, insufficient wastewater treatment, some agricultural practices, and mining have contributed to this decline by increasing the input of nutrients and sediments, as well as introducing pollutants such as heavy metals, pesticides, and plastics into reef environments and marine food webs [319]. Elevated nutrient concentrations can promote the development of turf algae and macroalgae that compete with corals for space and thus modify the composition of the benthic assemblages [320]. Benthic cyanobacterial mats, which have also been associated with negative effects on hard corals and coral recruitment, have emerged as a conspicuous component on some Pacific Island reefs, likely in part due to declining water quality [321]. Additionally, suspended sediments decrease light availability for benthic organisms while sedimentation can decrease coral recruitment [48]. Golbuu *et al.* [322] showed that hard coral cover in Ngermeduu Bay, in Palau, was negatively correlated with the concentration of suspended sediments. Mining activities can deliver high loads of sediment, as well as associated heavy metal, to coastal waters. For example, at the Lihir Island Group, in Papua New Guinea, coral reefs located less than two kilometres from a gold mine waste disposal site were characterised by a lower hard coral species richness and hard coral cover 15 times lower than control sites [323]. In New Caledonia, fringing reefs downstream of nickel mining sites exhibit coral lesions associated with sedimentation, yet hard coral cover can remain relatively high and diverse [324].



▲ **Figure 2.6.5.** Maximum sustained wind speed of cyclones that passed within 100 km of a coral reef in the Pacific region between 1980 and 2025. Colors correspond to the Saffir–Simpson category of the cyclone along its entire track. However, the values of sustained wind speed are extracted from the nearest cyclone position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir–Simpson scale (*i.e.* 119 km.h^{-1}). Derived from IBTrACS data (Knapp *et al.* [118]; see Materials and Methods).

Unsustainable fishing

Fishing is essential for food security and livelihoods across many Pacific countries and territories, and holds strong social and cultural significance for coastal communities [325]. However, certain practices, such as blast fishing, night-time spearfishing, or harvesting individuals before they reach sexual maturity, undermine the long-term sustainability of fisheries [326, 327]. Overexploitation of key trophic groups, particularly herbivorous fishes, reduces grazing pressure and promotes the proliferation of macroalgae and turf algae at the expense of hard corals [328]. The impacts of overfishing extend beyond reef fishes, as the depletion of some invertebrates can trigger cascading ecological effects. For instance, sea cucumbers (*beche-de-mer*) have been heavily overharvested throughout many Pacific reefs [329], and their decline has been linked to increased incidence of diseases affecting hard corals [330].

Tsunamis

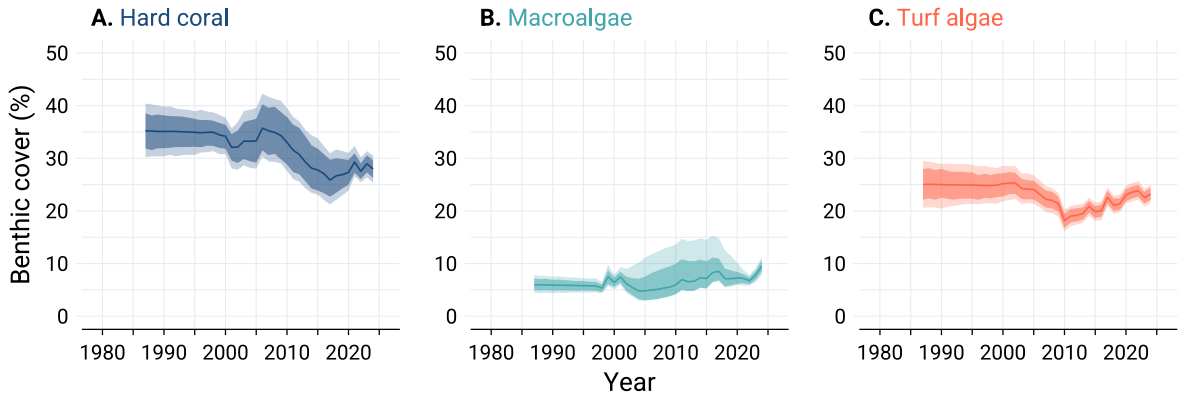
Pacific coral reefs are also impacted by tsunamis, which can generate physical disturbances comparable to those caused by cyclones. For example, the eruption of the Hunga Tonga–Hunga Ha'apai volcano in Tonga [331] triggered a tsunami that caused widespread hard coral loss across much of the country [332].

STATUS OF CORAL REEFS IN THE REGION

Hard coral cover

Hard coral cover declined from 34.5% (credible interval at 95% [CrI $_{95\%}$]: 31.1 – 38.5%) in 1987–2009 to 28.2% (CrI $_{95\%}$: 26.2 – 30.4%) in 2020–2024, a change supported by strong evidence (Figure 2.6.6A). This represents an absolute decrease of 6.3 percentage points, corresponding to a relative decline of 18.3% in hard coral cover. However, hard coral cover did not decline consistently between 1987 and 2024 but instead exhibited alternating phases of stability, decline, and recovery.

From 1987 to 2000, hard coral cover remained relatively stable at approximately 34.9% (CrI $_{95\%}$: 30.7 – 39.3%), a phase characterised by the absence of regional heat stress events (Figure 2.6.3).



▲ **Figure 2.6.6.** Modeled temporal trends of benthic cover in the Pacific region from 1980 to 2024 for hard coral (A), macroalgae (B), and turf algae (C). The solid line represents the estimated median benthic cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

Hard coral cover then declined by about 2 percentage points in 2001, likely associated with the first global bleaching event. This was followed by a recovery phase, with hard coral cover increasing to 35.7% by 2006 (Figure 2.6.6A).

A second and more prolonged phase of decline occurred between 2007 and 2017, during which hard coral cover gradually fell to 25.9%, representing a loss of nearly 10 percentage points (Figure 2.6.6A). This decline coincided with the global bleaching events of 2009–2011 and 2014–2017 (Figure 2.6.3). Although the 2014–2017 global bleaching event was unprecedented in intensity and spatial extent [31, 310], hard coral mortality varied substantially among Pacific reefs [333, 334]. However, the gradual decline in hard coral cover from 2007 to 2017 suggests a multifactorial origin, with heat stress likely acting alongside localized disturbances in some subregions, such as CoTS outbreaks [335] or particularly severe cyclones [336].

A second recovery phase occurred from 2018 to 2021, after which hard coral cover remained broadly stable through 2024 (Figure 2.6.6A). However, because monitoring data for 2023–2024 remain limited (Figure A.1), the full impact of the fourth global bleaching event may not yet be fully captured in these estimates. Field observations indicate variable impacts across the region, with limited hard coral mortality in New Caledonia (S. Job and T. Heintz, *pers. obs.*), Fiji (S. Mangubhai and A. Ford, *pers. obs.*), and Vanuatu (J. Johnson, *pers. obs.*).

Macroalgal cover

In contrast to hard coral cover, macroalgal cover in the region increased from 6.0% ($CrI_{95\%}$: 4.6 – 8.0%) in 1987–2009 to 7.7% ($CrI_{95\%}$: 6.7 – 9.1%) in 2020–2024 (Figure 2.6.6B), a change supported by strong evidence. This corresponds to an absolute increase of 1.7 percentage points, or a relative increase of 29.8%. Macroalgal cover remained generally stable at approximately 5.8% between 1987 and 1998 (Figure 2.6.6B). The end of this period was marked by an increase in macroalgal cover in 1998 followed by a macroalgal cover decline in 2002, coinciding with decline and subsequent recovery in hard coral cover after the 1998 global bleaching event. From 2003 to 2024, macroalgal cover exhibited a slow but persistent upward trend, interrupted only by a short period of relative stability between 2018 and 2021 (Figure 2.6.6B). The drivers of this gradual in-

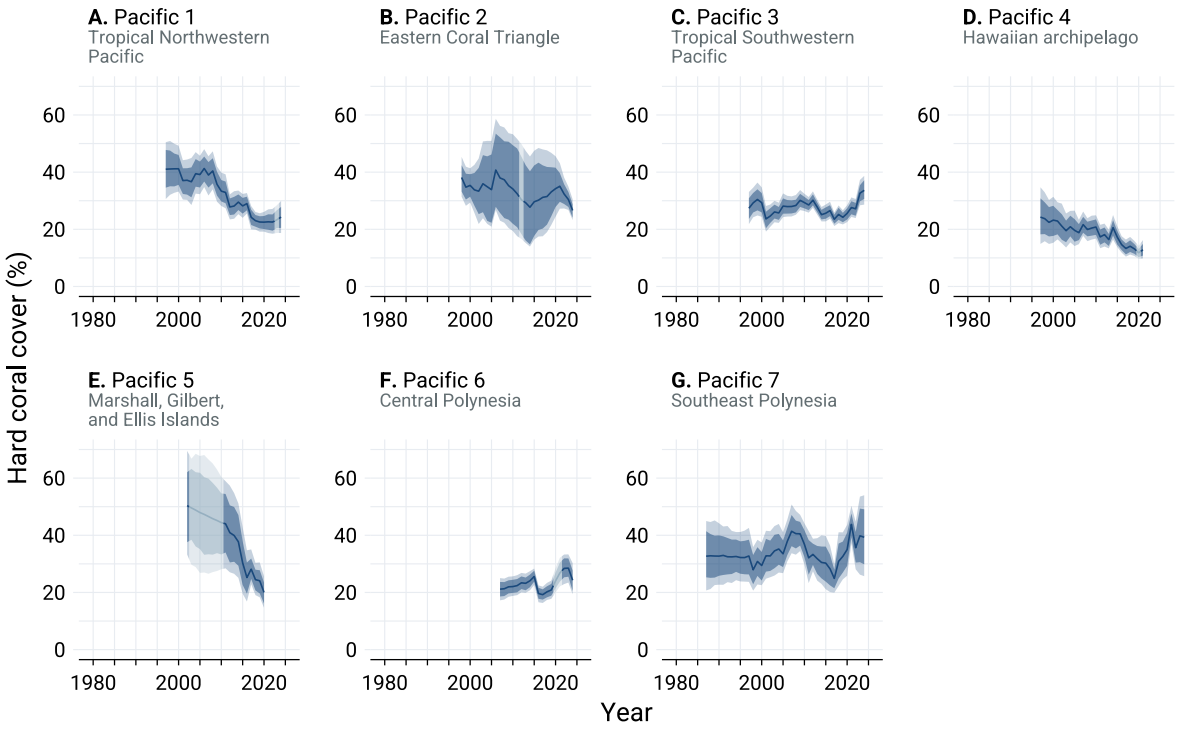
crease in macroalgal cover are likely multifactorial [337], including reduced hard coral cover that creates available substrate for algal colonisation, overfishing of herbivorous fish, and increased nutrient inputs in coastal waters that promote algal growth.

Turf algal cover

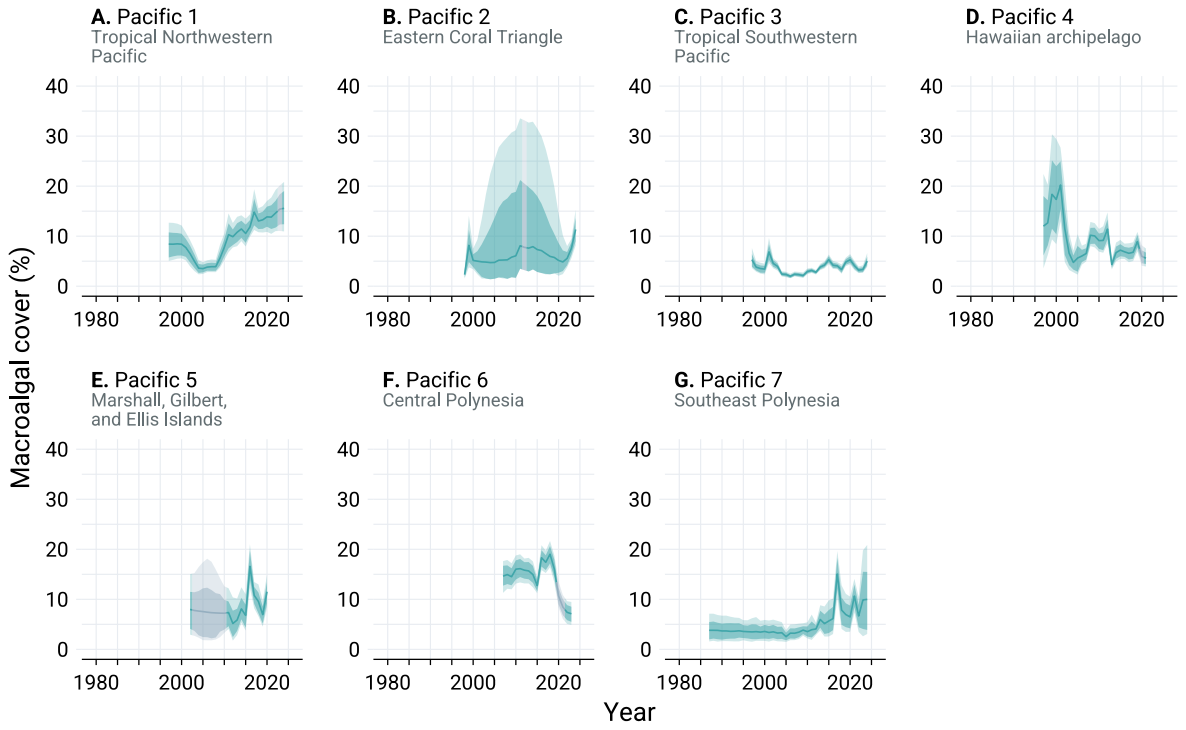
Turf algal cover showed no evidence of change between 1987–2009 and 2020–2024 (Figure 2.6.6C). In contrast, the 2011–2024 period was characterized by an increasing trend, with turf algal cover reaching 23.2% in 2024, coinciding with the decline in hard coral cover over the same period. Estimated values before 2011, and particularly the apparent decline between 2002 and 2010, should be interpreted with caution. Turf algae have not been recorded consistently across coral reef monitoring programs, especially in surveys conducted before 2011. Due to limited data availability at the subregion scale, inconsistencies in categorization among monitoring programs, and following consultation with regional experts, trends in turf algal cover were not considered at this scale.

STATUS OF CORAL REEFS IN THE SUBREGIONS

As coral reefs of the Pacific span thousands of kilometers, all reefs are not impacted by the same disturbance at the same time. For this reason, the regional trends in hard coral and macroalgal cover hide different patterns across the seven subregions of the Pacific. For instance, hard coral cover ranges from long-term decline in the Hawaiian archipelago subregion (Figure 2.6.7D) to overall increase in Central Polynesia (Figure 2.6.7F).



▲ **Figure 2.6.7.** Modeled temporal trends of hard coral cover in Pacific subregions from 1980 to 2024. The solid line represents the estimated median hard coral cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.



▲ **Figure 2.6.8.** Modeled temporal trends of macroalgal cover in Pacific subregions from 1980 to 2024. The solid line represents the estimated median macroalgal cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

Pacific 1 - Tropical Northwestern Pacific

Coral reefs in the Tropical Northwestern Pacific subregion occur across four countries and territories: Palau, the Commonwealth of the Northern Mariana Islands, Guam, and the Federated States of Micronesia.

In the Tropical Northwestern Pacific subregion, the results showed strong evidence that hard coral cover declined between 1997–2009 and 2020–2024 (Figure 2.6.7A). Median hard coral cover decreased from 39.3% ($CrI_{95\%}$: 33.0 – 45.2%) in the first period to 23.0% ($CrI_{95\%}$: 19.2 – 26.9%) in the second period, representing an absolute loss of 16.3 percentage points. Major declines in hard coral cover occurred in 2000, 2008, 2011, and 2016. The declines in hard coral cover in 2000, 2011, and 2016 coincide with three global bleaching events and are consistent with the bleaching severity and subsequent reductions in hard coral cover reported more locally in Palau [338, 339], Guam [340], the Federated States of Micronesia [341], and the Commonwealth of the Northern Mariana Islands [342]. The sharp decline in 2008, however, is more difficult to explain and may reflect local disturbances or dataset limitations.

Contrary to hard coral cover, macroalgal cover increased between 1997–2009 and 2020–2024, a change supported by strong evidence (Figure 2.6.8A). Median macroalgal cover increased from 6.0% ($CrI_{95\%}$: 4.5 – 8.0%) in the first period to 14.7% ($CrI_{95\%}$: 11.1 – 18.5%) in the second period, representing an absolute gain of 8.7 percentage points. Most of this increase took place between 2008 and 2017 and coincides with the pronounced decline in hard coral cover over the same period.

Pacific 2 - Eastern Coral Triangle

The Eastern Coral Triangle subregion encompasses coral reefs in Papua New Guinea and the Solomon Islands.

The data show no evidence of a change in hard coral cover between 1998–2009 and 2020–2024 for coral reefs of the Eastern Coral Triangle (Figure 2.6.7B). However, the limited number of surveys (Table A.4), reflected in the wide credible intervals, may have obscured potential changes over the full 1998–2024 period.

Similarly to hard coral cover, there was no evidence of a change in macroalgal cover in the Eastern Coral Triangle subregion between 1998–2009 and 2020–2024 (Figure 2.6.8B). The 95% credible intervals remained large throughout 2003–2023, with up to a 30 percentage points difference between the lower and upper bounds.

Pacific 3 - Tropical Southwestern Pacific

The Tropical Southwestern Pacific subregion encompasses coral reefs in New Caledonia, the Santa Cruz Islands (Solomon Islands), Vanuatu, Fiji, Tonga, and Niue.

There is no evidence of a change in hard coral cover in the Tropical Southwestern Pacific subregion between 1997–2009 and 2020–2024 (Figure 2.6.7C). Despite this overall stability, hard coral cover alternated between periods of decline (2000 and 2013–2017) and recovery (2003–2009 and 2018–2024). The two decline periods coincide with the first and third global bleaching events and correspond to elevated heat stress levels during those years, which have affected coral reefs in combination with other disturbances such as CoTS outbreaks (*e.g.* in New Caledonia, [343]) and cyclones (*e.g.* in Fiji, [344]). The increase in hard coral cover between 2018 and 2024 may reflect the absence of region-wide disturbances during this period, although local disturbances still occurred. For example, the January 2022 eruption of the Hunga Tonga–Hunga Ha'apai volcano in Tonga triggered a tsunami that caused widespread coral loss across much of the country [332]. In contrast, relatively low levels of bleaching and coral mortality were observed in Fiji (S. Mangubhai, *pers. obs.*), Vanuatu (J. Johnson, *pers. obs.*), and New Caledonia (T. Heintz, and S. Job, *pers. obs.*) in 2023 and 2024 despite the fourth global bleaching event. These examples highlight that regional trends in hard coral cover can mask contrasting trajectories at the local scale.

Macroalgal cover increased between 1997–2009 and 2020–2024, a change supported by strong evidence (Figure 2.6.8C). However, the magnitude of the change was limited, rising from 3.5% ($CrI_{95\%}$: 2.9 – 4.2%) in the first period to 4.2% ($CrI_{95\%}$: 3.6 – 4.9%) in the second period. This pattern is consistent with field observations conducted in New Caledonia, indicating that macroalgae remain a minor benthic component at most monitored sites (T. Heintz, *pers. obs.*). Nationwide, macroalgal cover was estimated at 1.9% in Tonga in 2022 [332], and 4.3% in Vanuatu in 2023 [345].

Pacific 4 - Hawaiian archipelago

The Hawaiian archipelago subregion can be divided into two areas: the Main Hawaiian Islands, home to over a million residents and welcoming more than eight million tourists a year, making local coral reefs heavily used; and the Northwestern Hawaiian islands, a collection of protected and mostly uninhabited atolls, islands, and banks. South of the subregion lies the Johnston Atoll [346].

In the Hawaiian archipelago, the results showed strong evidence that hard coral cover declined between 1997–2009 and 2020–2021 (Figure 2.6.7D). Median hard coral cover declined from 21.5% ($CrI_{95\%}$: 17.0 – 26.5%) in the first period to 12.7% ($CrI_{95\%}$: 9.6 – 16.0%) in the second

period, representing an absolute loss of 8.8 percentage points. The decline in hard coral cover was particularly marked around 2015, due to back-to-back bleaching events in 2014 and 2015 in the Main Hawaiian islands [347, 348] and a severe bleaching event in the Northwestern Hawaiian islands in 2014 [346, 349]. While less severe, an acute heat stress and bleaching event was also observed in 2019 across the Hawaiian archipelago [350].

Macroalgal cover also showed strong evidence of a decline between 1997–2009 and 2020–2021, dropping from 11.1% ($CrI_{95\%}$: 8.2 – 14.4%) in the first period to 5.7% ($CrI_{95\%}$: 4.1 – 7.5%) in the second (Figure 2.6.8D).

Pacific 5 - Marshall, Gilbert, and Ellis Islands

Coral reefs in the Marshall, Gilbert, and Ellis Islands subregion occur in the Marshall Islands, Gilbert Islands (Kiribati), Tuvalu, Nauru, Rotuma (Fiji), and Wake Island.

In the Marshall, Gilbert, and Ellis Islands subregion, the results showed strong evidence that hard coral cover declined between 2002–2009 and 2020 (Figure 2.6.7E). Median hard coral cover dropped from 47.9% ($CrI_{95\%}$: 28.6 – 64.9%) in the first period to 20.0% ($CrI_{95\%}$: 14.3 – 26.4%) in the second period, representing an absolute loss of 27.9 percentage points. Shifts in the South Equatorial Current during El Niño conditions have led the more equatorial Gilbert Islands to experience more frequent and prolonged heat stress than the Marshall and Ellis Islands to the north and south, respectively [351]. El Niño-driven heat stress caused mass coral bleaching in 2004–2005 and 2009–2010 across the Gilbert Islands, but only affected the northernmost atolls in the Ellis Islands [351].

Macroalgal cover showed no evidence of a change between 2002–2009 and 2020 (Figure 2.6.8E). The dominant algal taxa vary with exposure and local stressors across the Gilbert and Marshall Islands, with the calcifying algae *Halimeda* less common at disturbed sites [352].

Pacific 6 - Central Polynesia

The Central Polynesia subregion includes coral reefs in the Phoenix and Line Islands (Kiribati), the Northern Cook Islands, American Samoa, Samoa, Niua Islands (Tonga), Tokelau, and Wallis and Futuna, as well as several remote U.S. territories (*e.g.* Palmyra atoll).

Hard coral cover increased in the Central Polynesia subregion between 2007–2009 and 2020–2024, a change supported by strong evidence (Figure 2.6.7F). An abrupt decline in hard coral cover was recorded in 2015, coinciding with the third global bleaching event and aligning with reports of severe bleaching and substantial coral mortality more locally in American Samoa [353] and Jarvis Island [310]. In the Phoenix Islands, hard coral cover declined following the 2015–2016 heatwave but less than after the 2002–2003 heat stress event reported by Obura & Mangubhai [354], despite heat stress during 2015–2016 being approximately twice as intense [355]. At Palmyra Atoll, although the proportion of bleached corals was high during the third global bleaching event, resulting mortality remained limited [356, 357]. At the subregional scale, hard coral cover subsequently recovered between 2017 and 2023 (Figure 2.6.7F). In Samoa, this recovery is reflected by a near doubling of mean hard coral cover over five years, increasing from 10.6% in 2017 to 20.9% in 2022 [358].

Macroalgal cover declined between 2007–2009 and 2020–2023, a change supported by strong evidence (Figure 2.6.8F). It dropped from 14.7% ($CrI_{95\%}$: 12.0 – 17.4%) in the first period to 8.2% ($CrI_{95\%}$: 6.2 – 10.3%) in the second pe-

riod. Most of this macroalgal cover decline occurred from 2018 to 2023, in parallel with the increase in hard coral cover, as reported in Samoa [358].

Pacific 7 - Southeast Polynesia

The Southeast Polynesia subregion encompasses coral reefs in the southern Cook Islands, French Polynesia, and Pitcairn Islands.

In the Southeast Polynesia subregion, the data showed weak evidence that hard coral cover increased between 1987–2009 and 2020–2024 (Figure 2.6.7G). Median hard coral cover rose from 33.8% ($CrI_{95\%}$: 28.1 – 40.1%) in the first period to 38.8% ($CrI_{95\%}$: 31.8 – 46.5%) in the second period, representing an absolute gain of 5.0 percentage points. However, over the full 1987–2024 period, hard coral cover was highly dynamic, fluctuating between 20.0% and 37.3%. These pronounced temporal dynamics have also been reported at the local scale, notably in French Polynesia [335, 359] and in the Southern Cook Islands [318].

Macroalgal cover also showed strong evidence of an increase between 1987–2009 and 2020–2024, rising from 3.6% ($CrI_{95\%}$: 2.3 – 5.1%) in the first period to 8.9% ($CrI_{95\%}$: 5.5 – 13.7%) in the second (Figure 2.6.8G). It remained relatively stable from 1987 to 2005, then progressively increased through 2024. A pronounced peak occurred in 2017, when macroalgal cover reached 15.0%, coinciding with the decline in hard coral cover that same year (Figure 2.6.8G).

DESCRIPTION OF MONITORING DATA

Temporal trends in the percentage cover of benthic categories were estimated using data from 56 datasets encompassing 8,677 monitoring sites. These data were collected through 16,706 surveys conducted between 1987 and 2025, with 90% of surveys taking place after 2006 (Figure A.1). Of all surveys carried out in the region between 1987 and 2025, 36.4% (representing 26% of surveyed sites) were conducted in the Tropical Southwestern Pacific subregion, whereas only 2% (3.6% of sites) were conducted in the Marshall, Gilbert, and Ellis Islands subregion (Table A.4).

The limited number of surveys conducted in certain subregions hindered the assessment of benthic cover changes with sufficient evidence, underscoring the need to implement long-term monitoring programs in these subregions. In particular, the Eastern Coral Triangle, which includes Papua New Guinea and Solomon Islands, appears to be under-monitored given the large coral reef extent hosted by this subregion (*i.e.* 31.2% of coral reefs of the Pacific).

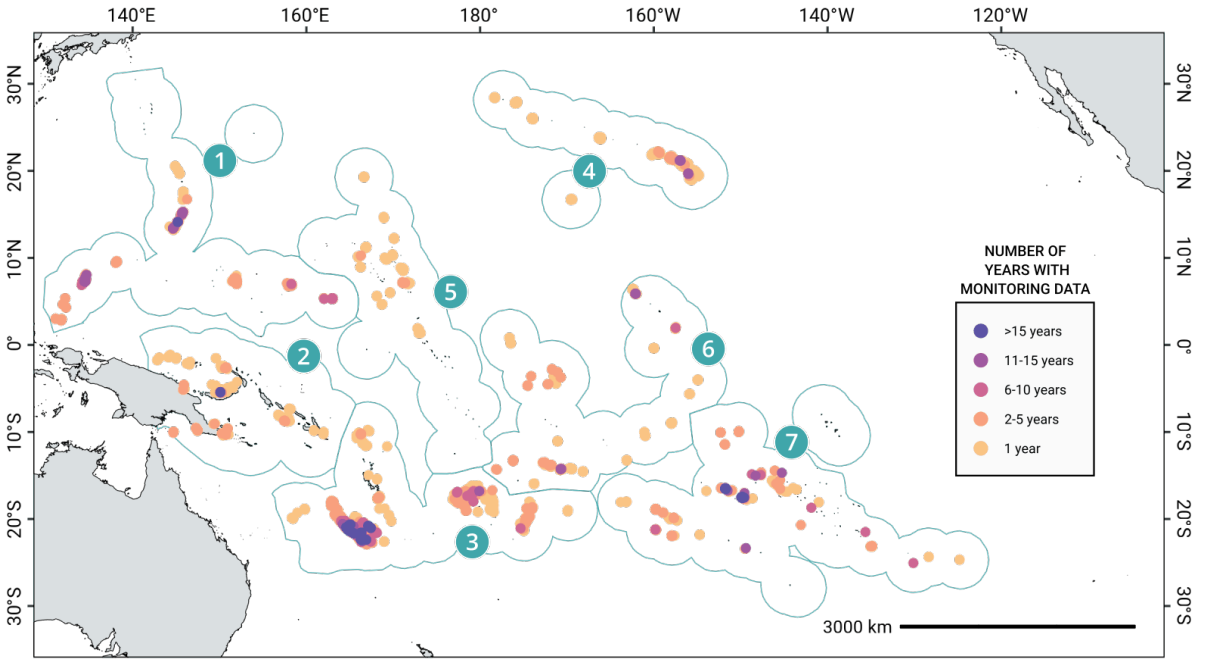
CONCLUSIONS

Overall, hard coral cover experienced a decline of approximately six percentage points (a 18% relative loss) across the Pacific region between 1987–2009 and 2020–2024, with the most pronounced losses occurring between 2007 and 2017. These declines align with successive global bleaching events driven by periods of intense heat stress. In contrast, macroalgal cover increased by around two percentage points over the same period. Turf algal cover remained relatively stable over the long term, although it has shown a noticeable increase over the past decade.

However, these regional-scale patterns mask substantial variability at subregional scale. Hard coral cover has declined in the three subregions located in the northern Pacific, while remaining stable or even increasing in southern Pacific subregions. This contrast reflects the immense geographic scale of the region, where coral reefs are characterised by diverse benthic communities and

are exposed to highly heterogeneous disturbance regimes. Differences in the timing, intensity, and frequency of stressors, such as heat stress, cyclones, or CoTS outbreaks, result in asynchronous responses in benthic cover trajectories among subregions.

Comparison with the *Status and Trends of Coral Reefs of the Pacific: 1980–2023* GCRMN report [44] shows that trends in macroalgae and turf algae are broadly consistent between assessments. However, in contrast to the present study, the previous regional report indicated relative stability in hard coral cover over the 1980–2023 period. This discrepancy likely arises from differences in modelling approaches, particularly in how data limitations were addressed. A key source of uncertainty is the limited availability of data for the Eastern Coral Triangle subregion, which contains nearly one-third of the Pacific's coral reef extent. These inconsistencies highlight the critical need to expand monitoring efforts, enhancing spatial representativeness and thereby strengthening the reliability of coral reef assessments across the Pacific.



▲ **Figure 2.6.9.** Spatio-temporal distribution of benthic cover monitoring sites across the Pacific region and respective subregions. Light blue polygons and numbers represent subregions. Subregion 1 = Tropical Northwestern Pacific, Subregion 2 = Eastern Coral Triangle, Subregion 3 = Tropical Southwestern Pacific, Subregion 4 = Hawaiian archipelago, Subregion 5 = Marshall, Gilbert, and Ellis Islands, Subregion 6 = Central Polynesia, Subregion 7 = Southeast Polynesia. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region.

DATA CONTRIBUTORS

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Access to online supplementary materials

The modelled benthic cover data, detailed outputs from the data analyses, and the CRediT author contribution statement are available on Zenodo.

CASE STUDY 6

Evaluating eDNA surveys for future monitoring strategies

Author

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As coral reefs face accelerating degradation globally, we need innovative tools to guide conservation and management efforts based on early detection of decline. Environmental DNA (eDNA) is reshaping biodiversity monitoring by enabling the detection of species through genetic traces from DNA shed into the water by individual organisms. Not only does eDNA enhance the resolution of biodiversity assessments, but it also promotes a more holistic understanding of ecosystem dynamics. This approach holds great promise for coral reef monitoring by offering a scalable, non-invasive survey method that expands taxonomic coverage, increases sensitivity to elusive or nocturnal species, and complements traditional diver-based surveys [360, 361]. For the GCRMN, which plays a pivotal role in assessing reef health worldwide, eDNA represents an opportunity to modernize and enrich monitoring programs, but it also introduces new technical, logistical, and interpretative challenges that will need to be carefully managed [362].

Unlike Underwater Visual Censuses (UVCs), which require taxonomic expertise, significant resources, and are limited by diver bias, eDNA analyses offer a complementary, standardized perspective. eDNA can detect organisms across multiple trophic levels, including cryptic and rare species that are easily overlooked by visual methods [363]. Incorporating monitoring data acquired through eDNA into its various data integration, standardization and analytical processes would therefore represent a highly valuable leap forward in coral reef monitoring. It would enable the network to not only monitor changes in visible benthic composition or potentially fish biomass, but to also track shifts across the entire biological community, including microbes and invertebrates, as well as the early signs of invasive or pathogenic species to inform timely management actions, with a level of precision and consistency previously unattainable [360]. In this way, eDNA acts as a biological time capsule that enhances our capacity to reconstruct ecological changes and assess long-term reef trajectories.

In practical terms, this means that beyond simply knowing how much live coral remains or how many herbivorous fishes are present, eDNA will document the hidden layers of reef ecosystems: the small reef cryptobenthic fishes that maintain ecological balance, or the microbial communities that often herald broader reef stress [364]. Where divers or cameras might miss early changes, eDNA can provide sensitive alerts of community turnover or novel species introductions long before they become visible. Moreover, through proper storage and control processes, collected eDNA samples can be continuously sampled over time, allowing for comparison between historic samples and recent samples. By tapping into this genetic "fingerprint of place", eDNA effectively extends the monitoring data both taxonomically (across the full spec-

trum of life) and temporally, as it can detect species that may have recently passed through an area even if they are not physically present at the exact moment of a dive survey [361]. In theory, this approach could fill gaps associated with UVCs.

Within the current monitoring context dominated by visual or photographic assessments, eDNA offers a fundamentally different kind of data. This additional data dramatically enhances the scope of biodiversity monitoring, capturing hidden dimensions that would otherwise remain undetected [360]. For coral reef monitoring, this means adding a layer of biological depth and sensitivity that can reveal subtle shifts in ecosystem structure, often preceding more obvious shifts in more traditionally reported indicators, such as declines in coral cover or fish biomass.

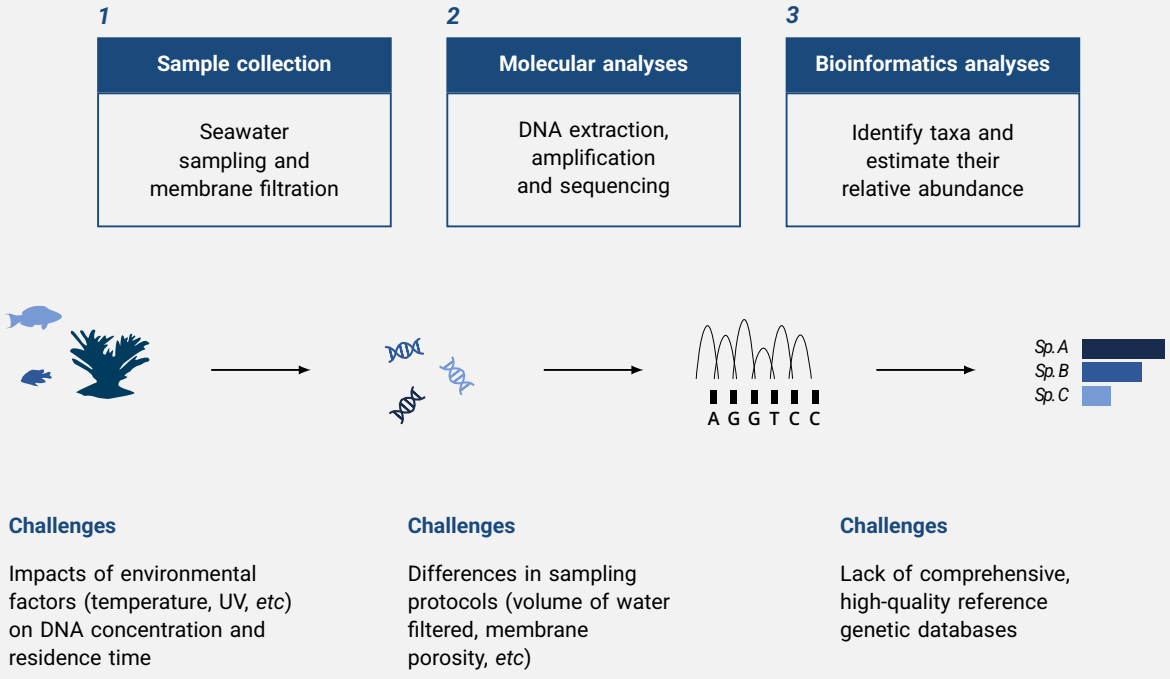
However, deploying eDNA at scale is not without substantial challenges (Figure C6.1). Three key issues currently limit its widespread implementation:

- **The dependence on comprehensive, high-quality reference genetic databases**. While databases for fish, especially those with markers such as the 12S MiFish gene, are relatively well-developed, genetic references for corals, algae, and many invertebrates are underrepresented, meaning a significant fraction of DNA sequences cannot yet be assigned to species [362]. This limits the immediate utility of eDNA for fully characterizing reef communities and calls for major investments in regional barcoding initiatives to establish robust reference libraries.
- **Environmental factors complicate interpretation**. The persistence and transport of eDNA are influenced by temperature, UV exposure, currents, and water chemistry. Although eDNA can reveal which species are present in a broader area, pinpointing their precise habitat associations remains complex, complicating direct links to metrics like coral cover or local fish biomass traditionally used by the GCRMN.
- **Methodological choices introduce further complexity**. The volume of water filtered, the choice of membrane porosity, and the number of sampling and PCR replicates all affect species detection [360, 362]. Establishing standardized protocols is thus essential to ensure comparability among sites and over time. Moreover, while eDNA metabarcoding is powerful for detecting presence, it does not reliably quantify biomass or abundance. Absolute counts remain elusive, and relative read counts can be skewed by variable DNA shedding rates and PCR amplification biases [363].

Overall, integrating eDNA into global coral reef monitoring programs will require overcoming the numerous interpretation challenges alongside investing in expanding genetic reference libraries, standardizing protocols to ensure comparability across regions, and securing local laboratory and bioinformatics capacity. It will also entail navigating regulatory landscapes, including compliance with the Nagoya Protocol and local laws governing genetic resource use [362]. eDNA should thus be viewed as a transformative complement rather than a replacement for traditional monitoring methods, particularly during this transitional phase of methodological refinement.

The future, however, looks highly promising. Similar to biomedical science, where genetic screening is now routine, the remaining challenges of eDNA-based monitoring are likely to be resolved rapidly. As costs decrease and analytical tools improve, eDNA will become a cornerstone of next-generation reef monitoring. In this light, eDNA's integration into coral reef monitoring strategies will be highly advantageous, at least starting with eDNA collection as it can be preserved for future analysis — allowing samples collected today to be analyzed in the future, as challenges are overcome and technological advancements continue. By integrating eDNA

into global coral reef monitoring programs, monitoring data will be capable of building a more complete, nuanced picture of reef biodiversity, and of detecting early warning signs of declining coral health, and thus help safeguard coral reefs in the face of accelerating global change.



▲ **Figure C6.1.** Schematic representation of the main steps in environmental DNA analysis and the associated challenges for its use as a large-scale monitoring tool.

CASE STUDY 7

The economic value of U.S. coral reef ecosystem services

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Coral reefs are critical natural assets that support livelihoods and economies worldwide, but they are also systems at risk from complex pressures. Assessing the value of coral reef ecosystem services helps translate their importance into terms that inform evidence-based conservation and restoration policies. This study estimates those values to highlight both the benefits coral reefs provide and the significant losses society faces if they are degraded or lost.

The National Oceanic and Atmospheric Administration's (NOAA) Coral Reef Conservation Program conducts valuation studies across all United States of America's (U.S.) coral reef jurisdictions. Preliminary results from 2021–2026 provide updated economic estimates of U.S. coral reef ecosystem services, incorporating non-monetary values such as cultural identity and spiritual significance. Because limited coral reef valuation data exist, benefit transfer methods were used to estimate values where primary data are lacking [365].

In 2013, the annual total economic value of U.S. coral reefs was estimated at USD 3.4 billion [366], whereas preliminary estimates from this study assessing six U.S. jurisdictions suggest an annual total economic value of USD 8.2–9.6 billion; a higher value is expected with completion of the valuation of the remaining two U.S. jurisdictions in 2026. It is important to note that the total values reflect conservative estimates developed for multiple categories of services that vary by jurisdiction (Table C7.1).

U.S. coral reefs provide over USD 2.6 billion annually in coastal protection by reducing erosion, flooding, and storm damage to buildings and infrastructure. Despite widespread recognition of this benefit, degradation of reef structures reduces their ability to dissipate wave energy, potentially leading to increased coastal flooding during storms and heightened risks to coastal properties. Pacific Island coral reefs hold deep social-cultural significance. In Guam, traditional reef fishing is worth USD 3.8 million annually and is crucial for sustaining social customs, such as the sharing of catch. For many communities, certain reef-fish species are central to cultural beliefs or practices, such as in origin stories or traditional harvesting events. In Hawai'i, consumptive/subsistence fishing is valued at USD 9.3–16.7 million per year, emphasizing the vital role of coral reefs in local nutrition and food security.

▼ **Table C7.1.** Preliminary total economic value (2024 USD million.yr^{-1}) estimates of U.S. coral reefs.

U.S. coral reef state or territory	Total economic value (2024 USD million.yr^{-1})*
Florida	681–1,370
Puerto Rico	1,870
U.S. Virgin Islands	426.0–814.3
Flower Garden Banks	*Pending*
American Samoa	50.4
Guam	71.5–82.9
Hawai'i	5,150–5,360
Commonwealth of the Northern Mariana Islands	*Pending*
Total	**8,200–9,600**

*These estimates do not include indirect or induced values.

Coral reefs are foundational to recreation and tourism industries, serving as key drivers of local economies. In the U.S. Virgin Islands, for instance, tourists spend approximately USD 597.4 million a year on reef-related activities; recreational fishing is worth USD 3.4 million per year, with each fishing household valuing coral reefs at an estimated USD 265.50 annually. In Florida, reef-related scuba diving and snorkelling are worth approximately USD 646.2 million annually and contribute USD 1.1 billion to the economy.

These findings show that coral reefs deliver benefits far beyond the economy to support livelihoods, cultures, and societies. By incorporating often-overlooked non-monetary values, this study provides a more complete understanding of their true worth. Using frameworks such as IPBES [367] and emerging pathways that integrate ecological, cultural, and economic data, it offers a new approach for linking plural valuation to global conservation priorities. The resulting ecosystem service values can be integrated into natural capital accounts (*e.g.* the United Nations System of Environmental-Economic Accounting), cost-benefit analyses, environmental impact assessments, and spatial planning, or used to help guide investment and resource allocation, incentivize stewardship and private sector engagement, strengthen community integration, and support global frameworks and reporting.

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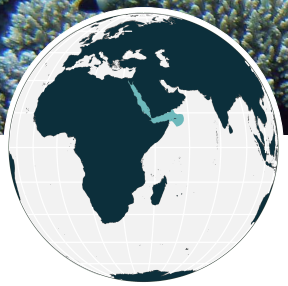
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Ras Baghdadi Reef, Wadi El-Gemal National Park, Egypt
Credit: Maher Amer



RED SEA AND GULF OF ADEN

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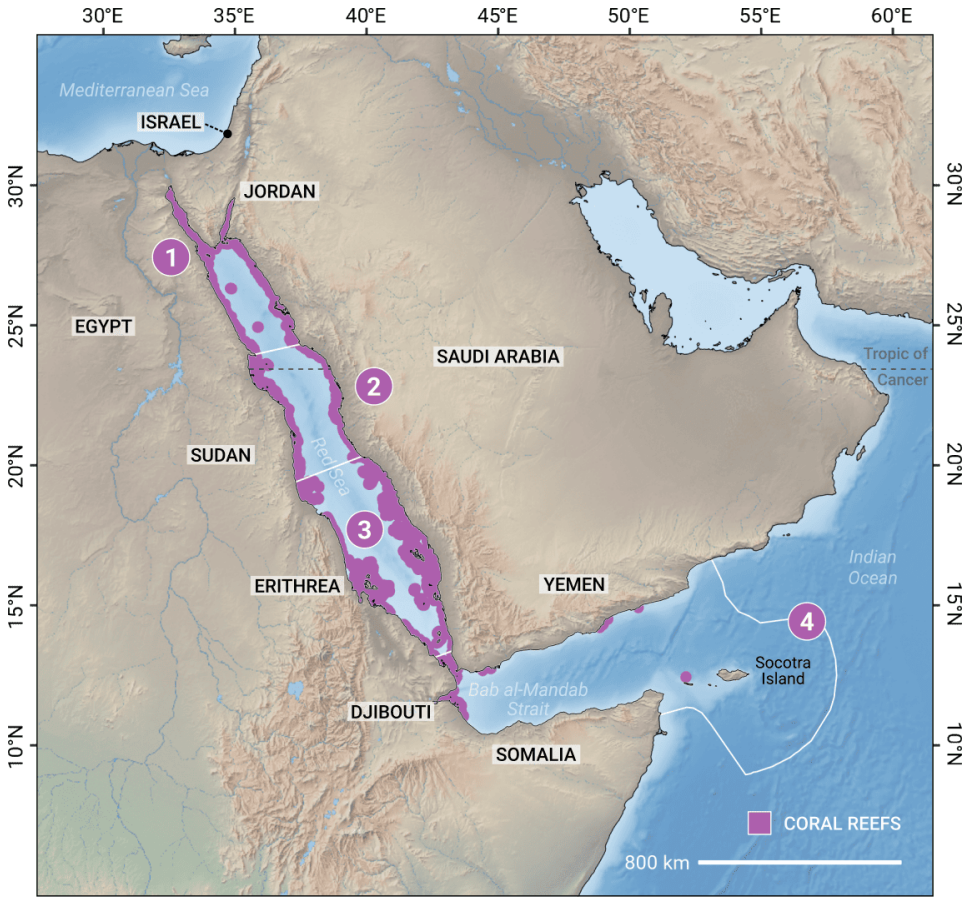
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INTRODUCTION

The Red Sea and Gulf of Aden GCRMN region extends over a maritime area of around 1.1 million km^2, ranging across 22° of latitude and 27° of longitude, forming one of the world's most diverse coral reef provinces [307]. Covering an estimated 12,590 km^2, or about 5.0% of global coral reef extent, it is the sixth-largest GCRMN region. There are, in addition, many other reef types that contribute to the great diversity of this system within the region and hence support an enormous reef-associated biota. Coral reefs lie within the exclusive economic zones of nine countries: Egypt, Sudan, Eritrea, Djibouti, Saudi Arabia, Yemen (including Socotra), Somalia, Jordan, and Israel. Coral reefs extend from the northern Red Sea, through the Gulfs of Suez and Aqaba, along the Red Sea coastline, and into the Gulf of Aden (Figure 2.7.1). For the purpose of this report, the GCRMN Red Sea and Gulf of Aden region was divided into four subregions: Northern Red Sea (subregion 1, 29.2% of coral reefs extent in the region), Central Red Sea (subregion 2, 26.6%), Southern Red Sea (subregion 3, 37.4%) and Gulf of Aden (subregion 4, 6.8%) (Table A.1).



▲ **Figure 2.7.1.** The distribution of coral reefs within the Red Sea and Gulf of Aden region and respective subregions. White polygons and numbers represent subregions. Subregion 1 = Northern Red Sea, Subregion 2 = Central Red Sea, Subregion 3 = Southern Red Sea, Subregion 4 = Gulf of Aden. Coral reefs are delineated with a 20 km buffer to enhance interpretation, and only coral reefs within the region are represented. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region.

The Red Sea is one of the warmest and most saline seas in the world, creating extreme reef environments. Corals in the Red Sea therefore tolerate conditions that are lethal elsewhere, reflecting long-term selection for heat- and stress-tolerant genotypes [368], which likely underpins the reef resilience and recovery observed at some sites.

The Red Sea is a narrow, elongated basin that exhibits pronounced north-to-south environmental gradients, with cooler, less saline waters in the north and warmer, saltier conditions in the south [369–372]. Northern Red Sea reefs (subregion 1) are generally well-developed narrow fringing systems along steep slopes, while the Gulf of Suez and Gulf of Aqaba host patch reefs and narrow fringing systems. Southward, reefs broaden, diversify, and form longitudinal barrier reefs along the Saudi Arabian and African coasts, extending hundreds of kilometers, with additional patch reefs and atoll-like structures such as Sanganeb Atoll in Sudan [373]. South of 20° N, reefs from the Central Red Sea (subregion 2) are less vertically developed, often in turbid waters, with coastal fringing reefs replaced by mangroves, sand beaches, and patch reefs. In the Gulf of Aden (subregion 4), fringing reefs occur along ~25% of the Yemeni coast [374]. Seasonal macroalgae can dominate shallow substrates in turbid areas, while offshore reefs, including platform, patch, barrier reefs, coral cays, and islands such as Farasan and Dahlak, support the greatest diversity [375]. Central areas (Sudan, Eritrea) host extensive fringing and offshore complexes, whereas southern Saudi reefs extend over 1,800 km, and the Socotra Archipelago contains highly diverse reefs with Indian Ocean affinities [376]. Smaller reefs in Djibouti and northern Somalia remain ecologically connected to the Red Sea corridor (subregion 4) [376].

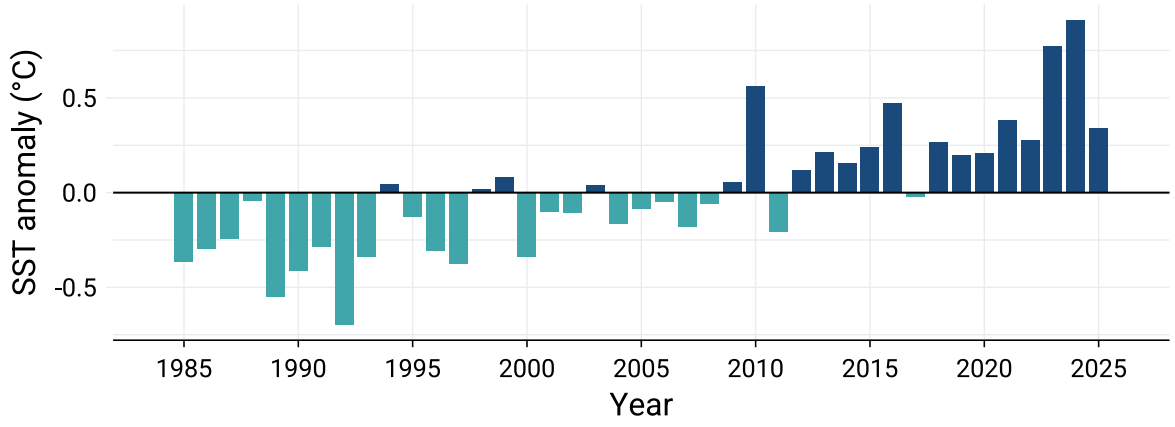
The coral reefs of the Red Sea support livelihoods for over 28 million people, contributing ~USD 230 million annually from fisheries and ~USD 6 billion from reef-based tourism globally [371, 377]. Coral reef tourism has historically accounted for nearly 2% of Egypt's gross domestic product (GDP), with similar growth anticipated in Saudi Arabia under Vision 2030 [17]. Rapid population growth near reefs (3.27 million within 5 km in 2020, a 69.4% increase since 2000), combined with coastal development, shipping, and land-based pollution, increasingly impacts reef condition, even as reefs continue to provide coastal protection from waves and sea-level rise.

THREATS AND DISTURBANCES

Heat stress

From 1985 to 2025, the long-term average sea surface temperature (SST) over coral reefs within the Red Sea and Gulf of Aden region was 27.85 °C (Table A.3). SST increased by 0.9 °C over this period, at an average rate of 0.23 °C per decade (Table A.3). SST anomalies were largely negative from the late 1980s until around 2009, before shifting to positive values and rising to approximately +0.25 °C during 2020–2024 (Figure 2.7.2). While these are regional averages, variability exists. In cooler years, the northern Red Sea was occasionally warmer than the southern Red Sea, whereas in warmer years the southern Red Sea consistently recorded higher temperatures, increasing heat stress in that subregion [378].

Major heat stress events in the Red Sea and Gulf of Aden region occurred in 1998 and 2010, with prolonged heat stress during 2015–2016, and renewed warming in 2020. Extreme heat stress peaked in 2023–2024, resulting in severe bleaching before stress levels declined in 2025 (Figure 2.7.3). Coral bleaching from heat stress was confirmed by in situ observations [379]. Bleaching risk in the Red Sea increases from north to south, with central and southern reefs exposed to higher temperatures and more frequent extreme events [108].



▲ **Figure 2.7.2.** Annual mean sea surface temperature (SST) anomalies (°C) over the coral reefs of the Red Sea and Gulf of Aden region from 1985 to 2025. SST anomalies were calculated based on the long-term average over the period 1985–2025. Positive values (red bars) indicate SST warmer than the long-term average, whereas negative values (blue bars) indicate SST cooler than the long-term average. Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).

The year 2024 marked the warmest year on record globally, exceeding 1.5 °C above pre-industrial levels [380], coinciding with the largest SST anomalies in the region (Figure 2.7.2). Severe mass bleaching was recorded for the first time in the northern Red Sea (north of 26° N) and along the Gulf of Aqaba reefs [381].

Regional disturbances

In 2024, three physical factors intensified bleaching risk in the northern Red Sea and Gulf of Aqaba. Altered currents reduced mixing of cooler deep water with surface layers, extreme low sea levels exposed shallow corals to sunlight and air, and calm winds limited replenishment of cooler surface water, prolonging heat stress [382]. Together, these localized disturbances likely amplified bleaching severity, highlighting subregional drivers of heat stress not always evident in broader regional analyses.

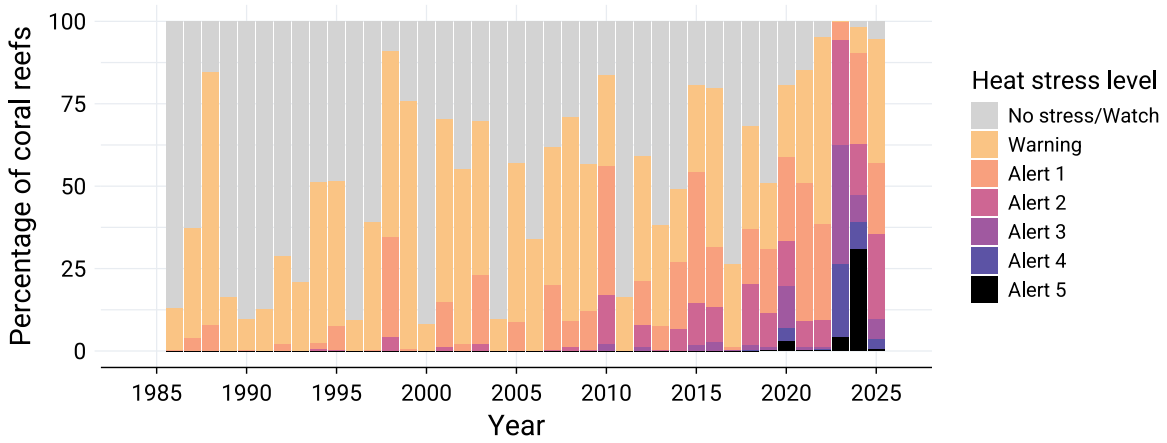
In addition to physical stressors, coral reefs in the Red Sea and Gulf of Aden are affected by localized anthropogenic stressors [371]. Coastal development and shoreline infrastructure alter water flow and increase sedimentation, while shipping operations contribute to localized pollution, anchor damage, and chemical stress [383]. These anthropogenic pressures, when interacting with natural disturbances, such as low tides, altered currents, and calm wind conditions, exacerbate bleaching risk and reduce reef resilience.

STATUS OF CORAL REEFS IN THE REGION

Long-term modeled benthic cover estimates for the Red Sea and Gulf of Aden region from 1997 to 2024 reveal alternating periods of stability, decline, and recovery in coral reef condition. These patterns reflect the combined influence of localized human impacts, including coastal development, tourism, fishing, and sedimentation, together with episodic thermal stress events that have driven repeated coral bleaching across the region.

Hard coral cover

Hard coral cover has consistently remained the dominant benthic component in the Red Sea and Gulf of Aden region, with median cover ranging from 20.1% to 37.8% between



▲ **Figure 2.7.3.** Percentage of the coral reefs within the Red Sea and Gulf of Aden region experiencing heat stress levels per year, from 1986 to 2025. No stress/Watch (DHW = 0); Warning (possible bleaching, 0 < DHW < 4); Alert 1 (reef-wide bleaching, 4 ≤ DHW < 8); Alert 2 (reef-wide bleaching and mortality of heat-sensitive corals, 8 ≤ DHW < 12); Alert 3 (multi-species mortality, 12 ≤ DHW < 16); Alert 4 (severe multi-species mortality, 16 ≤ DHW < 20); and Alert 5 (near-complete mortality, DHW ≥ 20). Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).

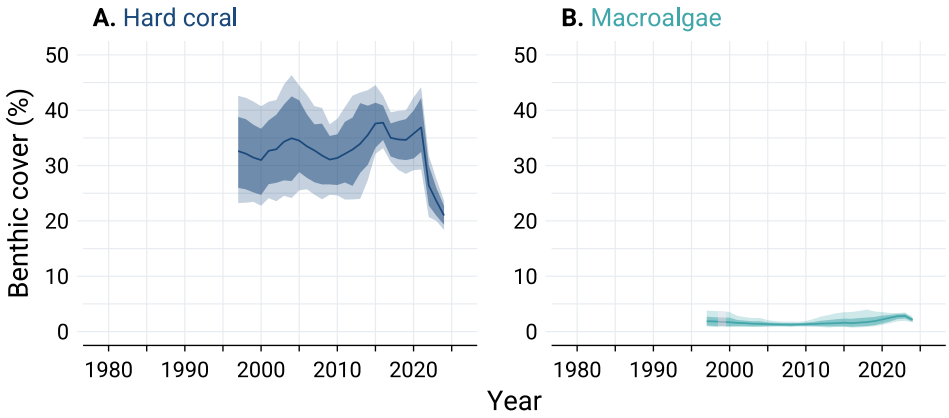
1997 and 2024 (Figure 2.7.4A).

Coral reefs in the region were exposed to elevated thermal conditions prior to the global coral bleaching event of 1998 (Figure 2.7.3). Despite this early heat stress, median hard coral cover remained relatively high, at 32.1% during 1997–1999 (Figure 2.7.4A). This stability during a period when widespread bleaching occurred globally suggests that reefs in the Red Sea and Gulf of Aden possess notable thermal tolerance. This resilience is likely supported by long-term exposure to naturally high and variable temperatures, as well as the presence of heat-tolerant coral genotypes in the northern basin [368].

From 2000 to 2008, hard coral cover remained relatively stable, with median values ranging between 31.0% and 34.9% (Figure 2.7.4A). Several years within this period, particularly 2003–2005, recorded upper credible intervals approaching or exceeding 45%, indicating that many reef sites maintained strong coral dominance despite environmental variability.

The early 2010s were marked by another period of heat stress associated with bleaching events, although modeled median hard coral cover remained relatively stable at around 31.5% between 2009 and 2011 (Figure 2.7.4A). Following this disturbance, reefs showed a clear recovery phase, and by 2016, median hard coral cover had increased to 37.8%; the highest values observed in the modelled time series. This recovery likely reflects favourable post-disturbance conditions, successful coral recruitment, and ecological controls such as herbivory that limit algal growth.

From 2017 to 2021, coral reefs in the region experienced a period of relative stability. Median hard coral cover remained consistently high, ranging from 34.6% to 36.9% (Figure 2.7.4A). This pattern suggests a period of relatively minor disturbance impacts, during which reefs have retained the capacity to recover hard coral cover to maintain historical levels. At the same time, human pressures along the Red Sea coastline were gradually increasing. Expanding coastal development, tourism infrastructure, and associated activities contribute to localized coral stress through sedimentation and physical disturbance [384]. Despite these pressures, hard coral cover



▲ **Figure 2.7.4.** Modeled temporal trends of benthic cover in the Red Sea and Gulf of Aden region from 1980 to 2024 for hard coral (A) and macroalgae (B). The solid line represents the estimated median benthic cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

remained relatively stable during this period, indicating that reefs were able to withstand chronic but sub-threshold stress without major losses in benthic cover.

The year 2021, however, marked a clear turning point in reef condition. Renewed ocean warming beginning in 2021, consistent with global increases in SST, preceded one of the most intense heat stress periods recorded for the region. Summer heat stress intensified further during 2023–2024 (Figure 2.7.3), leading to widespread and severe bleaching across many reef areas. From 2021 to 2024, median hard coral cover declined rapidly, from 36.9% to 21.0% (Figure 2.7.4A). This represents an absolute loss of 15.9 percentage points, corresponding to a relative decline of 43.1% within only a few years. Lower credible intervals in 2024 approached 18%, indicating that some reef sites experienced particularly severe localized impacts, while other reef sites retained relatively higher levels of hard coral cover. This rapid decline contrasts sharply with the slower fluctuations observed in earlier decades and suggests the frequency and intensity of heat-stress events may now be exceeding frequencies that prevent full recovery [369].

Macroalgal cover

Despite the substantial loss of hard coral cover during the study period, macroalgal cover did not increase proportionally. Median macroalgal cover rose only slightly, from approximately 1.5% in 1997–2009 to 2.5% in 2020–2024 (Figure 2.7.4B). This limited response reflects the long-standing pattern of relatively low macroalgal dominance across Red Sea reefs [385, 386]. Environmental conditions such as limited nutrient input, high salinity, intense solar radiation, and strong herbivory pressure may restrict large-scale macroalgal proliferation and moderate competitive interactions with corals [83, 387].

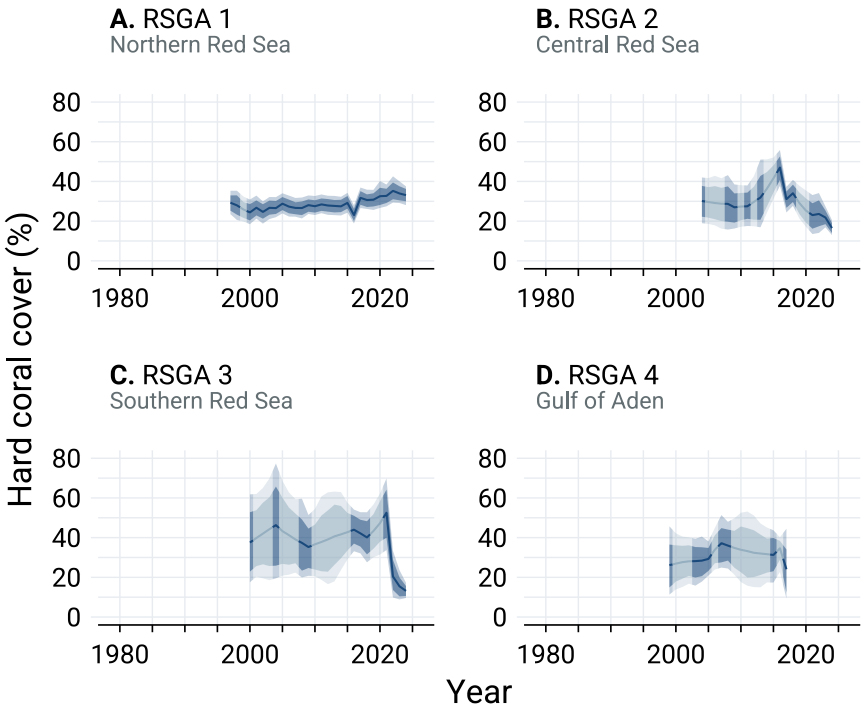
The absence of a clear macroalgal phase shift suggests that other opportunistic benthic groups may be occupying space made available by coral mortality. In particular, turf algae are increasingly recognized as potential competitors on degraded reefs. Increased turf algal growth has been documented in parts of the Central Red Sea under conditions of nutrient enrichment and environmental disturbance, where it can expand rapidly and occupy available substrate [388]. Monitoring these benthic groups is therefore important for detecting early changes in reef

community structure that may not be visible when examining macroalgal cover alone.

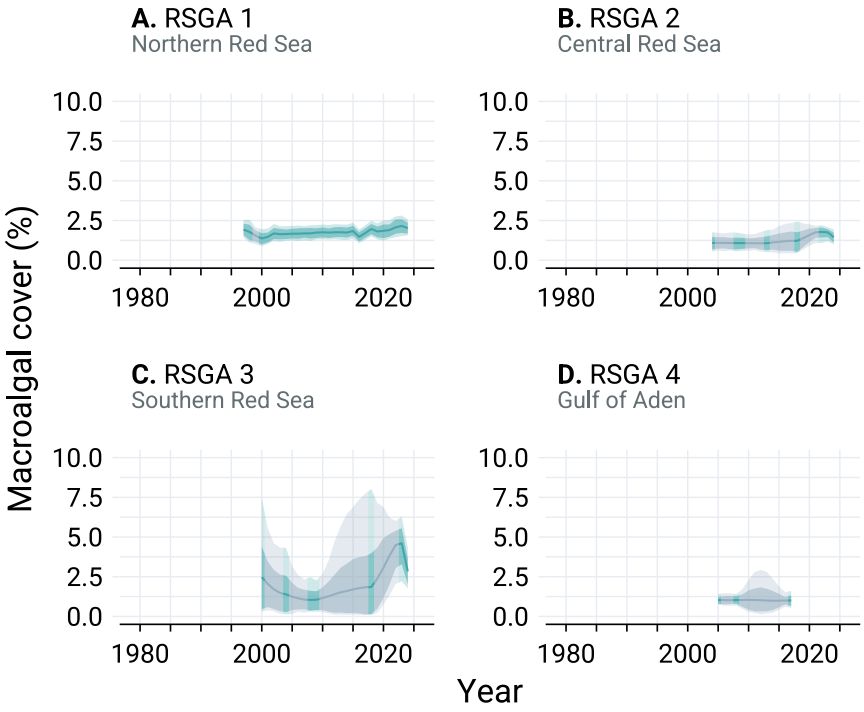
STATUS OF CORAL REEFS IN THE SUBREGIONS

RSGA 1 - Northern Red Sea

Coral reefs from the Northern Red Sea (subregion 1) showed consistent hard coral cover throughout 1997–2024, ranging from 23.0% to 35.3%. Hard coral cover declined after the 1998 coral bleaching event to 24.4% in 2000 (ranging from 18.5% to 30.9%). The 2016 mass bleaching event was the most significant event impacting this region, with the lowest recorded values reported (23.1%, ranging from 18.8% to 28%). However, after the 2016 mass bleaching event, hard coral cover has notably increased to 33.5% in 2020–2024, indicating long-term persistence and recovery potential (Figure 2.7.5A). Even during the 2010 and 2023–2024 heat stress events, declines were modest, and coral remained the dominant benthic component on Northern Red Sea reefs. Macroalgal cover remained very low at around 1.7% over the study period, suggesting limited phase shift potential. This sustained stability likely reflects naturally moderated SSTs, lower local disturbance, and adaptive traits in the coral assemblages that enhance tolerance to episodic heat stress, supporting the concept of a northern refugia [389].



▲ **Figure 2.7.5.** Modeled temporal trends of hard coral cover in Red Sea and Gulf of Aden (RSGA) subregions from 1980 to 2024. The solid line represents the estimated median hard coral cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.



▲ **Figure 2.7.6.** Modeled temporal trends of macroalgal cover in Red Sea and Gulf of Aden (RSGA) subregions from 1980 to 2024. The solid line represents the estimated median macroalgal cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

RSGA 2 - Central Red Sea

Coral reefs in the Central Red Sea (subregion 2) experienced fluctuations over the 2004–2024 study period. Heat stress events in 2015–2016 and 2023–2024 led to a decline, with hard coral cover dropping by 15.9 and 5.3 percentage points, respectively (Figure 2.7.5B). However, macroalgal cover remained low at around 1.3% throughout the study period despite coral loss, suggesting that other benthic components, such as turf algae, may have occupied the newly available substrate [390]. The variability in coral cover likely reflects a combination of higher exposure to heat stress, localized human pressures, and site-specific ecological differences that influence recovery potential, emphasizing the vulnerability of these central coral reefs to recurrent stress.

RSGA 3 - Southern Red Sea

Coral reefs in the Southern Red Sea (subregion 3) are geographically isolated and less connected to neighbouring subregions [389]. The Southern Red Sea consistently had the highest median hard coral cover of all subregions, with an average of 38.5% from 2000 to 2024, with some years exceeding 45% (Figure 2.7.5C), highlighting strong coral dominance. Nonetheless, the extreme heat stress event of 2023–2024 caused severe declines in hard coral cover, which fell to 13.1% in 2024, highlighting the sensitivity of these reef habitats to prolonged and extreme warming. Macroalgal cover increased by 0.9% from 2018 to 2024, averaging at 2.8% in 2024 (ranging between 1.8% and 4.3%), but hard corals continued to dominate benthic cover. The Southern Red Sea region's prior high hard coral cover and subsequent sharp declines suggest that while favourable environmental conditions

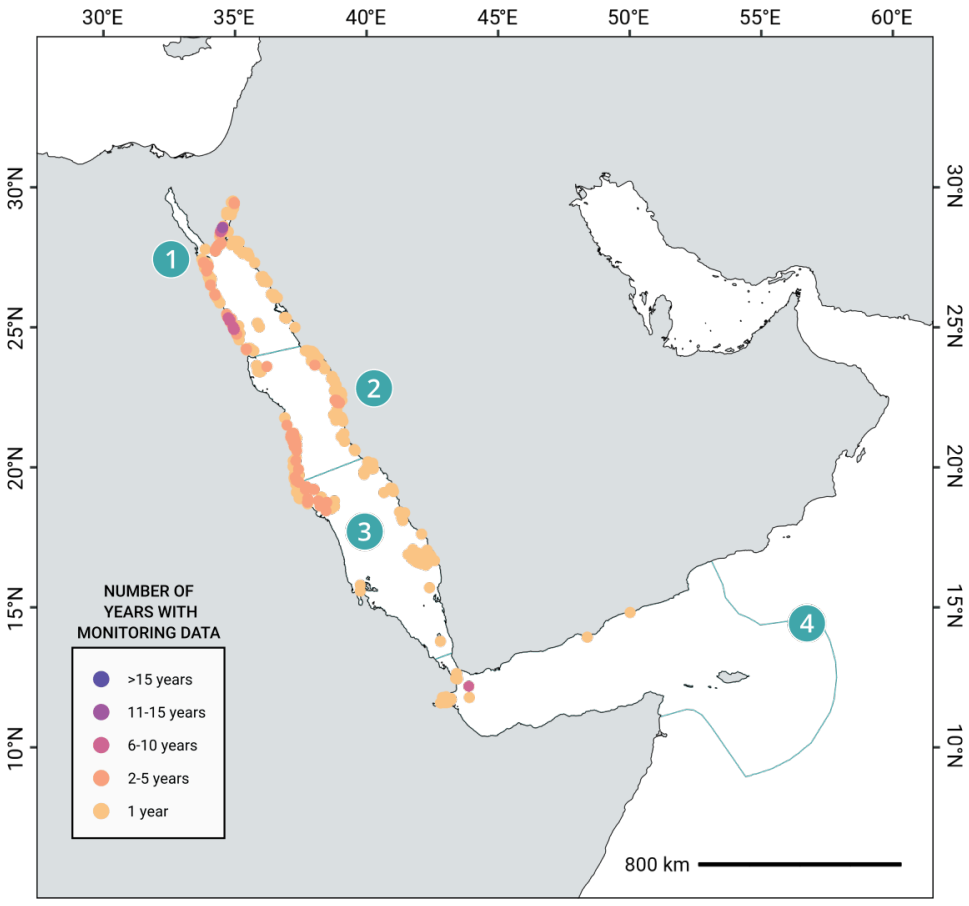
supported recovery after earlier bleaching events, extreme heat stress can override historical resilience, suggesting potential limits of coral persistence in the southern basin [390].

RSGA 4 - Gulf of Aden

Coral reefs in the Gulf of Aden (subregion 4) are less structurally developed than those within the Red Sea basin [383]. Available monitoring data for this subregion are limited, spanning 1999–2017, with no recent observations. During this period, hard coral cover ranged from 24.1% to 37.2% (Figure 2.7.5D), while macroalgal cover remained consistently low, averaging 1.0% (Figure 2.7.6D). The sparse temporal coverage and absence of post-2017 data constrain the assessment of current reef condition and longer-term trends.

DESCRIPTION OF MONITORING DATA

Trends in benthic cover across the Red Sea and Gulf of Aden region were derived from 14 datasets encompassing 502 monitoring sites and 903 surveys conducted between 1997 and 2024 (Table A.4).



▲ **Figure 2.7.7.** Spatio-temporal distribution of benthic cover monitoring sites across the Red Sea and Gulf of Aden region and respective subregions. Light blue polygons and numbers represent subregions. Subregion 1 = Northern Red Sea, Subregion 2 = Central Red Sea, Subregion 3 = Southern Red Sea, Subregion 4 = Gulf of Aden. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region.

Spatial coverage of surveys was uneven among subregions: 67% of surveys (representing 53.2% of surveyed sites) were conducted in Northern Red Sea (subregion 1), whereas only 3.8% (5.6% of surveyed sites) were conducted in Gulf of Aden (subregion 4, Table A.4). The majority of surveys focused on shallow reef areas, at depths of less than 10 m (Figure A.2). This uneven effort shapes interpretation of benthic trajectories, emphasizing more confidence in the patterns observed in the Northern Red Sea (subregion 1) compared with the sparsely sampled Gulf of Aden (subregion 4).

Temporal trends in monitoring data across the four Red Sea and Gulf of Aden subregions also reflect these regional differences. The Northern Red Sea (subregion 1) has the most complete monitoring data, with consistent monitoring from the late 1990s through 2024 (Figure 2.7.7), providing a strong basis for evaluating long-term reef dynamics, resilience, and coral responses to repeated disturbance events over nearly three decades. Monitoring in Central Red Sea (subregion 2) and Southern Red Sea (subregion 3) began later and became more consistent from the early to mid-2000s, allowing assessment of reef condition across multiple environmental stress periods, although some gaps remain where surveys were not conducted. Gulf of Aden (subregion 4) has the most limited record, with observations only from the late 1990s to the late 2010s and no recent data, restricting evaluation of current reef status or comparisons with other subregions. Despite these limitations, the available observations provide a historical context for understanding regional benthic trends, highlighting areas of resilience and vulnerability, and emphasizing the need for expanded, sustained monitoring to support effective management and conservation strategies across all Red Sea and Gulf of Aden subregions.

CONCLUSIONS

The benthic cover results across the four Red Sea and Gulf of Aden subregions reveal distinct spatial patterns in hard coral and macroalgal dynamics from 1997 to 2024. Median hard coral cover was highest in the Southern Red Sea (subregion 3) and the Central Red Sea (subregion 2) during the early 2000s, exceeding 40% and 30%, respectively, whereas the Northern Red Sea (subregion 1) and Gulf of Aden (subregion 4) maintained lower but more stable hard coral cover, generally around 25–29% (Figure 2.7.5). Notably, reefs in the Northern Red Sea (subregion 1) remained comparatively stable even during major bleaching events that affected other parts of the Red Sea. This resilience likely reflects environmental buffering, cooler baseline temperatures, or the presence of heat-tolerant coral genotypes [389].

Macroalgal cover remained low across all subregions, generally below 2% cover, with only a slight increase in the Southern Red Sea (subregion 3) during 2020–2023 (Figure 2.7.6). These patterns suggest that coral declines did not result in a widespread shift to macroalgal dominance at the regional scale.

Overall, the modelled benthic trajectories from 1997 to 2024 highlight several important ecological patterns. First, hard coral cover exhibited alternating phases of stability, recovery, and recent rapid decline, reflecting the cumulative effects of repeated thermal stress events. Second, consistently low macroalgal cover indicates that large-scale shifts toward algal dominance have not occurred, although changes in other benthic groups may still be emerging. Third, the broad credibility intervals observed in several years point to strong spatial variability in reef condition, with some reefs maintaining relatively high hard coral cover while others experienced substantial declines.

Together, these results emphasize the complex interactions between climatic disturbances and local environmental conditions shaping reef trajectories across the Red Sea and Gulf of Aden. Although reefs in the region have historically shown resilience to bleaching events, the severe coral losses recorded during 2023–2024 demonstrate that even these systems are vulnerable to increasingly frequent and intense thermal stress. Continued long-term monitoring and adaptive management will therefore be essential to support the future condition and stability of these globally important reef ecosystems.

DATA CONTRIBUTORS

Mohammad S. Al-Tawaha, Khadija Y. I. Abaker, Swaleh Aboud, Mustafa K. Altaib, Maher Amer, Tamer Attalla, Chico Birrell, Graham J. Edgar, Elamin M. Elamin, Sheikheldin M. Elamin, Mohammed Elsayed, Jan Freiwald, Matthew Fynes, Jamie Hayes, Magda M. Ismael, Gareth John, Mohammed M. Kotb, Anderson B. Mayfield, Jenny Mihaly, Karin Moejes, David Obura, Elizabeth Oh, Osama S. Saad, Rick D. Stuart-Smith, Oliver Taylor.

Access to online supplementary materials

The modelled benthic cover data, detailed outputs from the data analyses, and the CRediT author contribution statement are available on Zenodo.

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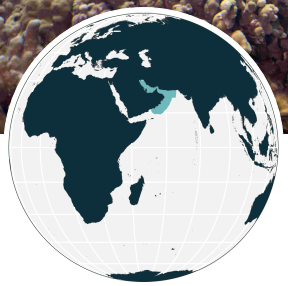
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Coral reef in the Gulf of Oman
Credit: Christophe Chellapermal



ROPME SEA AREA

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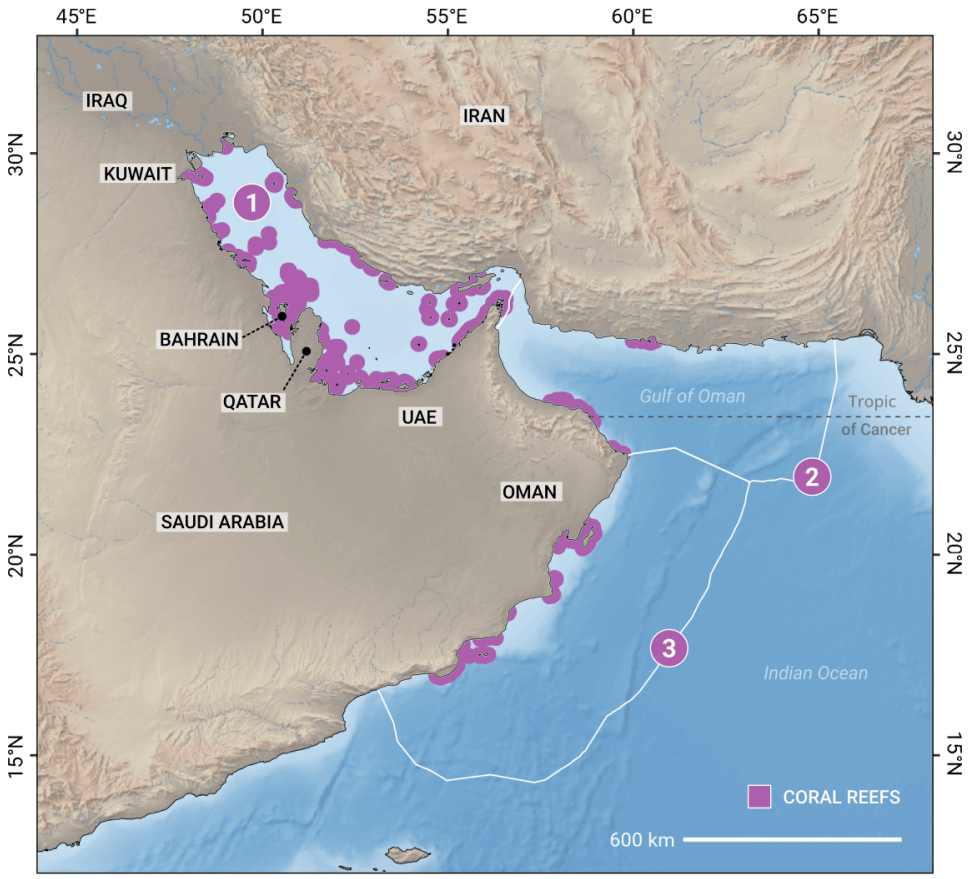
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INTRODUCTION

The Regional Organization for the Protection of the Marine Environment (ROPME) Sea Area GCRMN region extends over a maritime area of around 1 million km^2, ranging across 17° of latitude and 19° of longitude. The region hosts 1,808 km^2 of coral reefs, representing 0.7% of the world's coral reef extent, ranking it as the eighth largest GCRMN region based on total coral reef extent. Coral reefs reside within the exclusive economic zones of eight countries in the region, ranging from Kuwait and Iraq in the northwest, to Iran in the East, and Oman in the South (Figure 2.8.1)[1].

For the purpose of this report, the ROPME Sea Area region was divided into three subregions (Figure 2.8.1), with subregion 1 (Persian/Arabian Gulf), subregion 2 (Gulf of Oman), and subregion 3 (Southern Oman) hosting 73.2% (1,324 km^2), 9.9% (178 km^2) and 16.9% (306 km^2) of the regional coral reefs extent, respectively (Table A.1).



▲ **Figure 2.8.1.** The distribution of coral reefs within the ROPME Sea Area region and respective subregions. White polygons and numbers represent subregions. Subregion 1 = Persian/Arabian Gulf, Subregion 2 = Gulf of Oman, Subregion 3 = Southern Oman. Coral reefs are delineated with a 20 km buffer to enhance interpretation, and only coral reefs within the region are represented. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region.

[1] The area of reef extent is slightly lower in this report compared with the 2020 GCRMN report due to a shift in the southern border to the Oman-Yemen border, whereas the 2020 report included parts of eastern Yemen in the outer ROPME Sea Area.

Coral reefs represent an important socio-economic resource for coastal communities in the ROPME Sea Area, as well as a growing focus for scientific research on climate adaptation. As of 2020, a total of 3.4 million people were living within 5 km of coral reefs across the ROPME region (Table A.2). From 2000 to 2020, the region recorded over a threefold (208.2%) increase in human populations residing within 5 km of coral reefs, representing an average annual increase of approximately 114,000 inhabitants. The greatest population increase over this period was recorded in subregion 1 (Persian/Arabian Gulf), which hosted the vast majority (95.9%) of regional reef-adjacent inhabitants in 2020 (Table A.2).

THREATS AND DISTURBANCES

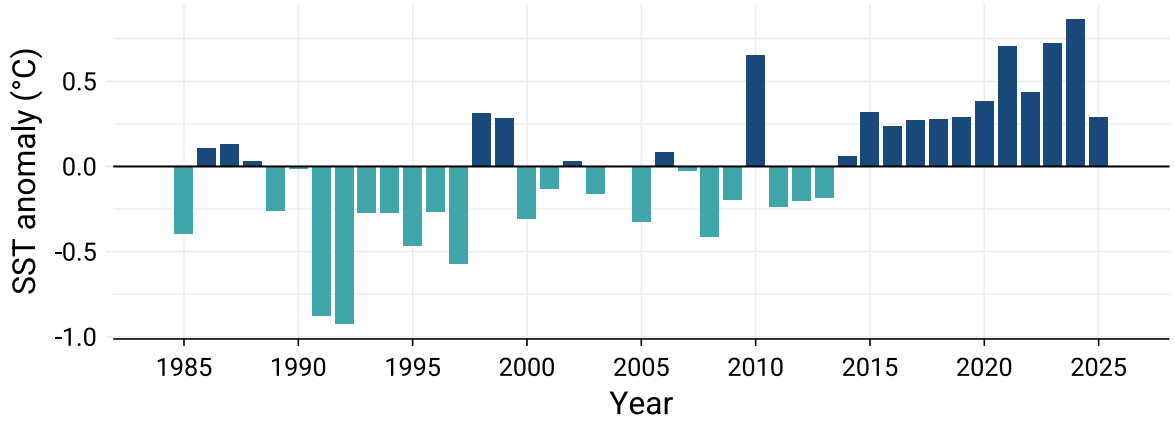
Heat stress

Between 1985 and 2025, the long-term average of sea surface temperature (SST) recorded over coral reefs was 26.76 °C across the entire ROPME region. During this period, the SST over regional coral reefs increased by 0.5 °C, corresponding to an average warming rate of 0.12 °C per decade (Table A.3).

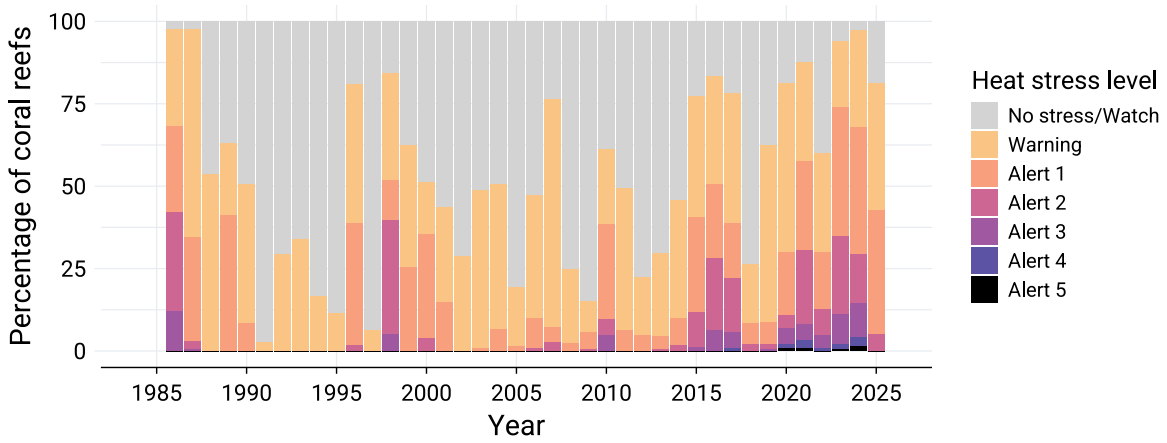
Between 1986 and 2025, major heat stress (Bleaching Alert 1–5) affected ROPME reefs in distinct pulses: the mid-1980s (1986–1987), a strong late-1990s/early-2000s cluster (1996–2002, peaking in 1998), and renewed events around 2010 in multiple years (Figure 2.8.3). During the third global bleaching event from 2014–2017, regional heat stress became more frequent and began reaching higher alert levels across a substantial fraction of reefs. Since 2020, severe heat stress has occurred annually, with a larger proportion of reefs being exposed to Alert 2–4, and the maximum Alert 5 heat stress level occurring in most years, consistent with accelerating regional heat stress exposure [391].

Coastal development

Coastal development represents a major non-climatic pressure on reefs in the ROPME Sea Area, particularly in subregion 1 (Persian/Arabian Gulf), where coastal urbanized area expanded by 55% between 2001 and 2021 [392].



▲ **Figure 2.8.2.** Annual mean sea surface temperature (SST) anomalies (°C) over the coral reefs of the ROPME Sea area region from 1985 to 2025. SST anomalies were calculated based on the long-term average over the period 1985–2025. Positive values (red bars) indicate SST warmer than the long-term average, whereas negative values (blue bars) indicate SST cooler than the long-term average. Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).



▲ **Figure 2.8.3.** Percentage of the coral reefs within the ROPME Sea area region experiencing heat stress levels per year, from 1986 to 2025. No stress/Watch (DHW = 0); Warning (possible bleaching, 0 < DHW < 4); Alert 1 (reef-wide bleaching, 4 ≤ DHW < 8); Alert 2 (reef-wide bleaching and mortality of heat-sensitive corals, 8 ≤ DHW < 12); Alert 3 (multi-species mortality, 12 ≤ DHW < 16); Alert 4 (severe multi-species mortality, 16 ≤ DHW < 20); and Alert 5 (near-complete mortality, DHW ≥ 20). Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).

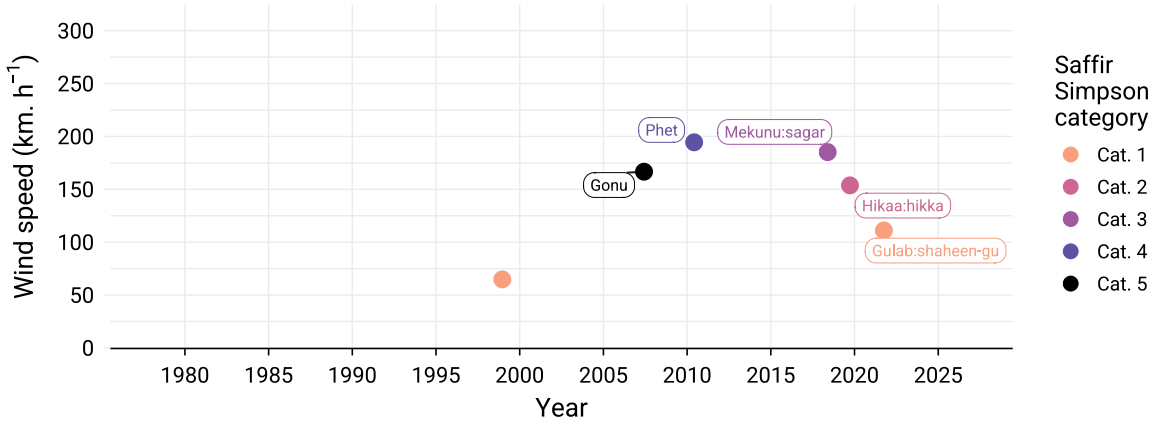
Large-scale reclamation, dredging, and channelization have resulted in the direct loss of reef habitat and extensive sediment plumes that degrade adjacent reef systems [393]. Such developments have also occurred in subregions 2 and 3, but at far lower intensities and with more limited reef impacts, except around the Muscat capital area in Oman [394].

Harmful algal blooms

Harmful algal blooms have emerged as an episodic but consequential disturbance. While typically confined to lagoons and embayments, the persistent 2008–2009 harmful algal bloom and associated anoxia extended across much of subregion 2 and into subregion 1, causing widespread mortality from which many reefs have yet to fully recover [395]. Increasing coastal wastewater input and rising SSTs suggest that harmful algal bloom frequency and extent are likely to intensify [396].

Fishing pressure

Fishing pressure affects reefs across the region, but uneven monitoring effort prevents reliable inference about where its impacts are greatest. Regionally common fishing gear, including *gargoors* (traditional wire fish traps), seine nets, drift nets, and others, can cause direct physical damage to reef habitats through placement, dragging, and incidental breakage during removal [397, 398]. Beyond these direct effects, fishing pressure may influence reef resilience through the selective removal of key functional groups, especially herbivorous and predatory fishes, with potential consequences for benthic recovery trajectories and ecosystem functioning [397, 398]. These indirect effects are likely to interact with disturbance history and local environmental conditions, rather than acting in isolation. Additional understudied pressures, including pollution, localized hypoxia, and location-specific seasonal macroalgal and turf algal overgrowth, further compound the vulnerability of coral reefs across the ROPME region.



▲ **Figure 2.8.4.** Maximum sustained wind speed of cyclones that passed within 100 km of a coral reef in the ROPME Sea Area region between 1980 and 2025. Colors correspond to the Saffir–Simpson category of the cyclone along its entire track. However, the values of sustained wind speed are extracted from the nearest cyclone position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir–Simpson scale (*i.e.* 119 km.h^{-1}). Derived from IBTrACS data (Knapp *et al.* [118]; see Materials and Methods).

Cyclones and storms

Between 1980 and 2025, a total of six cyclones passed within 100 km of coral reefs in the ROPME Sea Area (average 0.2 ± 0.4 SD cyclones per year, Figure 2.8.4). Cyclone tracks and impacts are exclusively confined to ROPME subregions 2 and 3 (Gulf of Oman and, particularly, Southern Oman). The category 5 cyclone Gonu in 2007 had the greatest impacts on the coral reefs, which led to severe wave-related damage to exposed reefs across the UAE and Oman's Gulf of Oman coasts and islands, with some reefs also experiencing cyclone-induced sedimentation impacts [394]. It is also possible that cyclones have impacted reefs in southern Oman in ROPME subregion 3, but limited reef monitoring activities in that area hamper interpretation. In addition to cyclones, several tropical storms have also impacted coral reefs in the ROPME Sea Area [399].

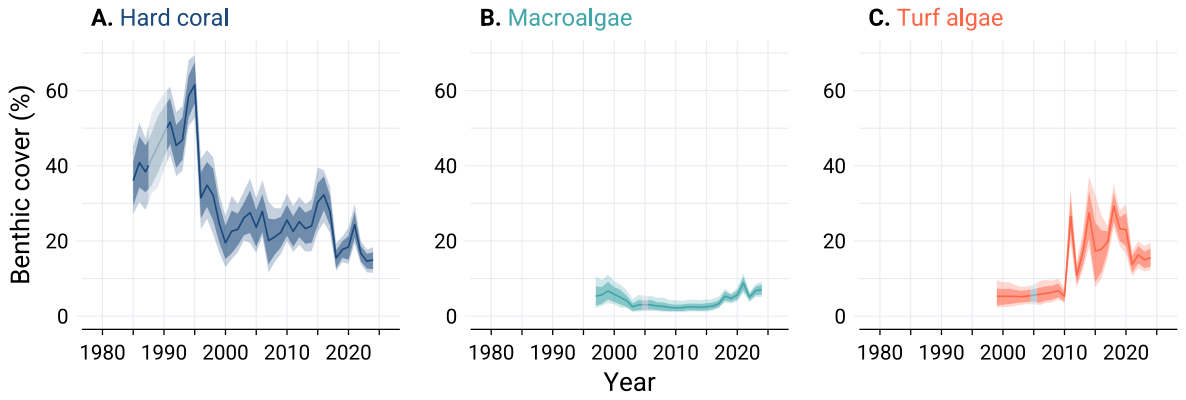
STATUS OF CORAL REEFS IN THE REGION

The modeled trajectories of benthic cover across the ROPME region show how hard coral and macroalgal communities changed from 1985 to 2024 (Figure 2.8.5). They reveal repeated disturbance-recovery cycles, followed more recently by persistently low hard coral cover, driven primarily by recurrent marine heatwaves and bleaching, with subregional influence from cyclones, harmful algal blooms, and coastal pressures.

Hard coral cover

High hard coral cover in the 1980s and early 1990s

The reconstructed timelines indicate hard coral cover was high across the region during the 1980s, averaging nearly 40% of the benthos in 1985 and increasing into the early 1990s (Figure 2.8.5A). Although monitoring data from this period are limited (see Description of monitoring data), the estimates are consistent with historical accounts describing exten-



▲ **Figure 2.8.5.** Modeled temporal trends of benthic cover in the ROPME Sea area region from 1980 to 2024 for hard coral (A), macroalgae (B), and turf algae (C). The solid line represents the estimated median benthic cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

sive coral assemblages across subregions 1 and 2 before major late-1990s SST extremes (Figure 2.8.2 and Figure 2.8.3) [400–404]. Much of this baseline reflected widespread *Acropora*-dominated communities, particularly in subregion 1, which were structurally important but highly sensitive to heat stress [34, 400]. The early 1990s therefore likely represent a pre-disturbance regional baseline before recurrent marine heatwaves became a defining feature of the system.

Abrupt decline following back-to-back bleaching events (1996 and 1998)

The most pronounced feature of the trajectory is the sharp decline in hard coral cover beginning in the mid-1990s. Over the five years to 2000, regional hard coral cover more than halved, from 61% in 1995 to 23% in 2000 (Figure 2.8.5A). This collapse reflects the severe impacts of the 1996 and 1998 marine heatwaves (Figure 2.8.3) associated with the first global bleaching event [34], which caused widespread mortality across subregion 1 (Persian/Arabian Gulf) and the near-extirpation of branching corals in many areas [405–409]. The magnitude and timing of the modeled decline closely match field observations, and mark the first major regional shift in benthic community structure during the modern warming era. This reflects the concentration of extreme late-1990s heat stress in subregion 1, whereas comparable impacts were less evident in subregions 2 and 3.

Partial recovery and a suppressed post-disturbance equilibrium (2000–2016)

Following the late 1990s collapse, hard coral cover recovered gradually, but incompletely, during the 2000s and early 2010s, exceeding 30% by 2015 but never returning to the >50% levels of the early 1990s (Figure 2.8.5A). Field observations indicate that this recovery was often driven by heat-tolerant massive and encrusting corals rather than the formerly dominant branching *Acropora*, especially in the southern Persian/Arabian Gulf [98, 407, 410]. This partial rebound occurred during a relative lull between the late-1990s heat-stress cluster and the more frequent severe events after 2016, with localized heat stress around 2010 (Figure 2.8.3) producing only modest impacts on regional cover [411]. The period thus represents the longest interval without major regional mortality in the monitoring record, yet the failure to fully recover to high former hard coral cover levels suggests that earlier bleaching had already reduced structural complexity and recovery capacity, shifting reefs toward

a lower-cover state.

Renewed decline following the 2017 and 2021 bleaching events

From 2016 to 2024, the modeled trajectory enters a renewed decline that aligns with more frequent high-level heat stress during the mid-2010s and annual severe heat stress after 2020 (Figure 2.8.3). The downturn intensified in 2017, when subregion 1 (Persian/Arabian Gulf) experienced its hottest summer on record, with reef temperatures exceeding 37 °C at several monitored sites [412]. This event caused widespread mortality across the southern and western Gulf, including reefs where *Acropora* had previously regained dominance, and drove average regional hard coral cover below 20% for the first time on record in 2018 (Figure 2.8.5A).

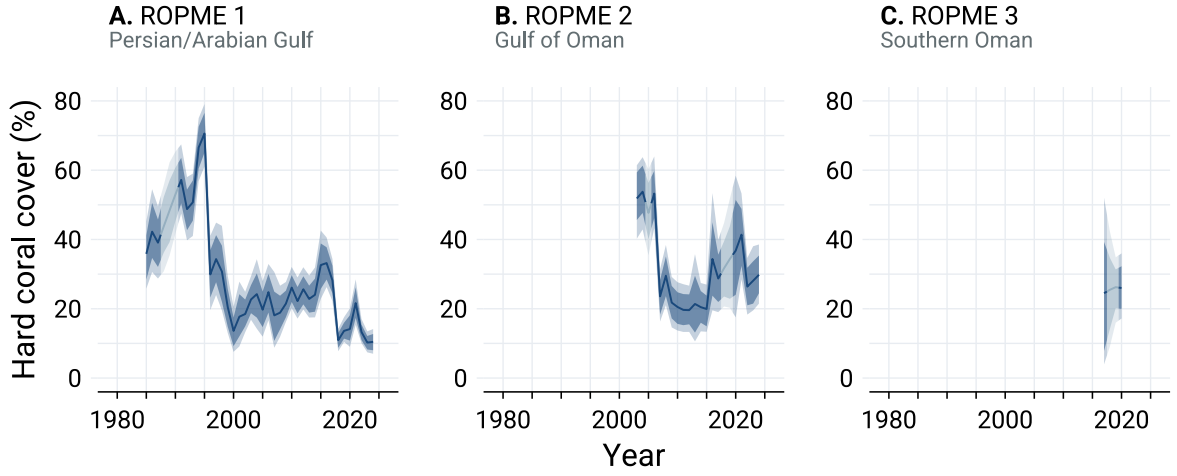
A further decline followed the severe 2021 SST anomaly (Figure 2.8.2), which affected much of subregion 1 and parts of subregion 2 (Gulf of Oman). Particularly strong impacts fell on remnant *Acropora* refugia, showing that thermally sensitive branching assemblages remained the principal source of regional vulnerability (*e.g.* Sir Bu Nair Island on the UAE's Gulf coast and Khor Fakkan on the UAE's Gulf of Oman coast). These stands, already greatly reduced by earlier disturbances but partially recovered, suffered renewed mortality during the 2021 marine heatwave [413]. The modeled trajectory captures this decline, with hard coral cover of 24% in early 2021 falling by roughly one-third the following year.

Additional bleaching in 2023, observed in both the southern Gulf and the Muscat area (subregions 1 and 2), further affected already depleted coral populations. A more extreme event followed in 2024 (Figure 2.8.2 and Figure 2.8.3), beginning in the Gulf of Oman in July and extending across most reefs in the southern Arabian Gulf in subsequent months. By then, monitoring data showed the lowest average hard coral cover yet reported for the region, with no evidence of rebound (median regional hard coral cover in 2024: 15%) (Figure 2.8.5A). Collectively, these trends indicate recovery intervals are now shorter than disturbance intervals under near-annual heat stress, and point to a declining regional capacity for resilience.

Macroalgal cover

Low and stable algal cover through the 1990s and 2000s

Macroalgal cover was very low (<5%) in the mid-1990s, which is consistent with the high hard coral cover, and thus limited substrate availability, during this period. Although monitoring data are not available, literature descriptions indicate algal communities primarily consisted of macroalgae and turf, and encrusting calcareous algae occupied minor space as well; transient seasonal macroalgal blooms were common on regional reefs (*e.g.* during spring in subregion 1 and during monsoonal upwelling in subregion 3) [398, 414]. After the 1996 and 1998 bleaching events, macroalgal cover increased modestly to ~5% (Figure 2.8.5B), consistent with early successional dynamics in which turf and fleshy algae were reported to colonize newly dead coral skeletons [398, 414]. However, the increase in macroalgae was negligible in magnitude relative to the substantial hard coral loss that occurred during this period, reflecting observations at that time of limited algal colonization of dead reef framework [407]. This indicates that, at the regional scale, turf and macroalgal expansion in the wake of these early bleaching events through the 2000s was limited by environmental conditions and/or grazing pressure.



▲ **Figure 2.8.6.** Modeled temporal trends of hard coral cover in ROPME Sea area subregions from 1980 to 2024. The solid line represents the estimated median hard coral cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

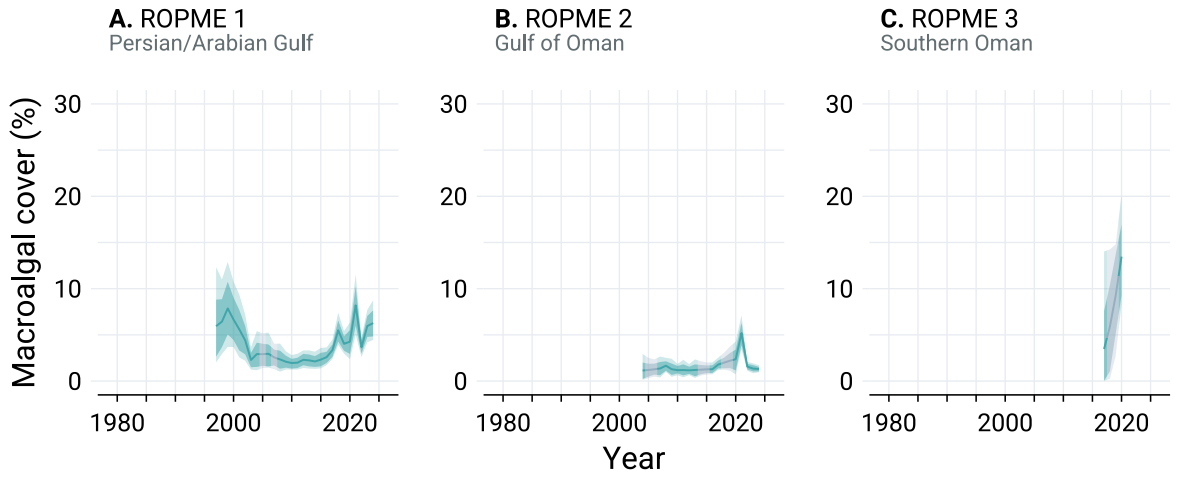
Gradual upward trajectory after 2010, without a region-wide phase shift

A feature of the modeled estimates is the gradual, low-magnitude rise in macroalgal cover from 2010 onward, and an abrupt rise in turf algae (Figure 2.8.5B and Figure 2.8.5C). Macroalgal cover increased steadily into the early 2020s, with a particular upward inflection after the 2017 bleaching event, with a peak macroalgal cover of ~9% in 2021 (Figure 2.8.5B). This trend suggests that, while regional macroalgal abundance has increased over time, particularly following 2017 and 2021 coral mortality events, there has not been a widespread transition to macroalgal dominance at the scale of the entire ROPME region, despite substantial declines in live hard coral cover at the regional scale. That said, turf algae, rather than macroalgae, has become an increasingly common member of the benthic community in parts of the region (Figure 2.8.5C), rapidly occupying coral skeletons in the wake of bleaching. Although variable, turf algae since 2011 has hovered between 15 and 30% coverage, compared with ~5% in the early 2000s, representing a substantial component of the benthos in recent years, and often exceeding the amount of regional live hard coral cover (Figure 2.8.5A).

Overall, the trajectory suggests that macroalgal cover has responded to repeated coral mortality events but remains a minor component of regional benthic structure. Despite substantial hard coral cover loss, the continued absence of region-wide macroalgal dominance indicates that herbivory may still play a role in limiting macroalgal expansion, although the relative contributions of grazing versus environmental constraints cannot be resolved with the available data. Further integration of herbivorous fishes and urchins into regional monitoring is warranted. Turf algae, however, has become a dominant member of the benthos, likely impairing coral recovery via direct space occupation and trapping of sediments that reduce larval coral settlement rates [415].

Synthesis and implications for resilience

When considered together, the modeled trends for hard coral and algae illustrate that the ROPME Sea Area as a whole is undergoing prolonged ecological restructuring. Average hard coral cover has declined from



▲ **Figure 2.8.7.** Modeled temporal trends of macroalgal cover in ROPME Sea area subregions from 1980 to 2024. The solid line represents the estimated median macroalgal cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

a high of >50% in the 1990s to ~15% by 2024, with the partial recovery that occurred in the mid-2000s giving way to a second, more persistent and extreme decline after 2017.

In contrast to the trajectory following the 1996 and 1998 bleaching events, when hard coral cover rebounded to ~30%, the most recent downturn shows no sign of recovery, showing instead a near continual decline over the past decade, reflecting more frequent marine heatwaves superimposed on reefs already weakened by earlier bleaching, harmful algal bloom impacts, cyclones, and local pressures.

Macroalgal cover, while increasing gradually over the monitoring period, remains comparatively low, while turf algae has become increasingly common on regional reefs since 2010. The steady upward trend in macroalgae and, particularly, turf algae suggests growing competitive pressure on juvenile coral recruitment and highlights the likely importance of herbivore communities in maintaining reefs in non-algal-dominated states. Alongside heat stress, changes in grazing communities may further influence recovery potential by modifying key ecological processes, even where reefs have not transitioned to macroalgal dominance.

Taken together, these patterns depict a regional coral reef system that has undergone multiple disturbance cycles and is currently facing increasing recovery challenges, resulting in lower region-wide hard coral cover than in the past. The declining recovery ceiling, compounded by recent recurrent severe bleaching events (2017, 2021, 2023, 2024), suggests regional resilience is being eroded by chronic exposure to extreme temperatures. The trajectories modeled here therefore reinforce the conclusion that the ROPME Sea Area, despite hosting the world's most thermally tolerant corals [416], is increasingly vulnerable as warming trends accelerate and the frequency of extreme events continues to rise with climate change.

STATUS OF CORAL REEFS IN THE SUBREGIONS

The modeled trajectories show clear subregional contrasts in reef condition and recovery pathways (Figure 2.8.6 and Figure 2.8.7). Subregion 1 largely determines the regional pattern because it contains the most reef

area and monitoring effort, and shows the strongest long-term decline: high hard coral cover in the 1980s, collapse after the 1996 and 1998 bleaching events, partial recovery through the 2000s, and renewed decline after 2017. By contrast, subregion 2 exhibits a later and more punctuated disturbance-recovery sequence, with declines linked first to cyclone Gonu and the 2008–2009 harmful algal bloom event, followed by recovery through the 2010s and renewed losses under recent bleaching. Subregion 3 remains too data-limited for robust temporal comparison, as only two datasets are available; therefore interpretation is necessarily qualitative.

ROPME 1 - Persian/Arabian Gulf

Subregion 1 most closely mirrors, and largely drives, the region-wide trajectory, consistent with the dominance of this subregion in the overall dataset (28 of 34 datasets). Hard coral cover was high during the 1980s, before declining sharply in the late 1990s following the 1996 and 1998 bleaching events (Figure 2.8.6A). This collapse, one of the most consequential in the history of Gulf reef systems, resulted in widespread mortality of branching corals and shifts toward thermally tolerant massive coral taxa [98, 407]. The modeled decline is substantial and sustained, with hard coral cover stabilizing at 20 to 30% through much of the 2000s and early 2010s.

The partial recovery during this period reflects both the resilience of surviving coral taxa and the absence of major Gulf-wide disturbance events. However, as seen in the regional trend, hard coral cover did not return to its earlier high levels, suggesting a lower post-disturbance equilibrium. Following the 2017 bleaching event, hard coral cover in subregion 1 declined again, with losses continuing after widespread bleaching-induced mortality in 2021 (Figure 2.8.6A). The 2024 bleaching event likely contributed to the protracted downward trajectory. By this time, hard coral cover reached its lowest recorded levels (~10%, the lowest of all subregions), highlighting the cumulative impact of repeated heat stress in a subregion already experiencing the highest summer sea temperatures globally.

Macroalgal cover in subregion 1 remains low across the entire time series (Figure 2.8.7A). Slight increases occurred in 2000 and between 2017 and 2021, likely reflecting the colonization of dead coral substrate that followed episodic bleaching events. However, the macroalgal cover did not exceed an average of 5% in modeled estimates. This is consistent with the historically low macroalgal abundance reported for Gulf reefs [407, 410], where localized herbivory, paired with extreme environmental conditions, may limit macroalgal growth. Following 2017 and 2021, a modest increase in macroalgal cover occurred that mirrors regional patterns, but with no transition to macroalgal dominance.

ROPME 2 - Gulf of Oman

Unlike subregion 1, the Gulf of Oman exhibits a distinct trajectory shaped first by cyclone and impacts from harmful algal blooms [394, 398], and only later by severe bleaching [413] (Figure 2.8.6B). The time series for hard coral cover began in 2003, when average hard coral cover exceeded 50%, reflecting the high diversity and variety of growth forms that is typical for this subregion. A pronounced decline in subregion 2 hard coral cover follows shortly thereafter, driven first by cyclone Gonu in 2007, and then by the 2008–2009 harmful algal blooms-associated anoxia event. In the wake of these events, the lowest hard coral cover of ~25% was recorded in 2012.

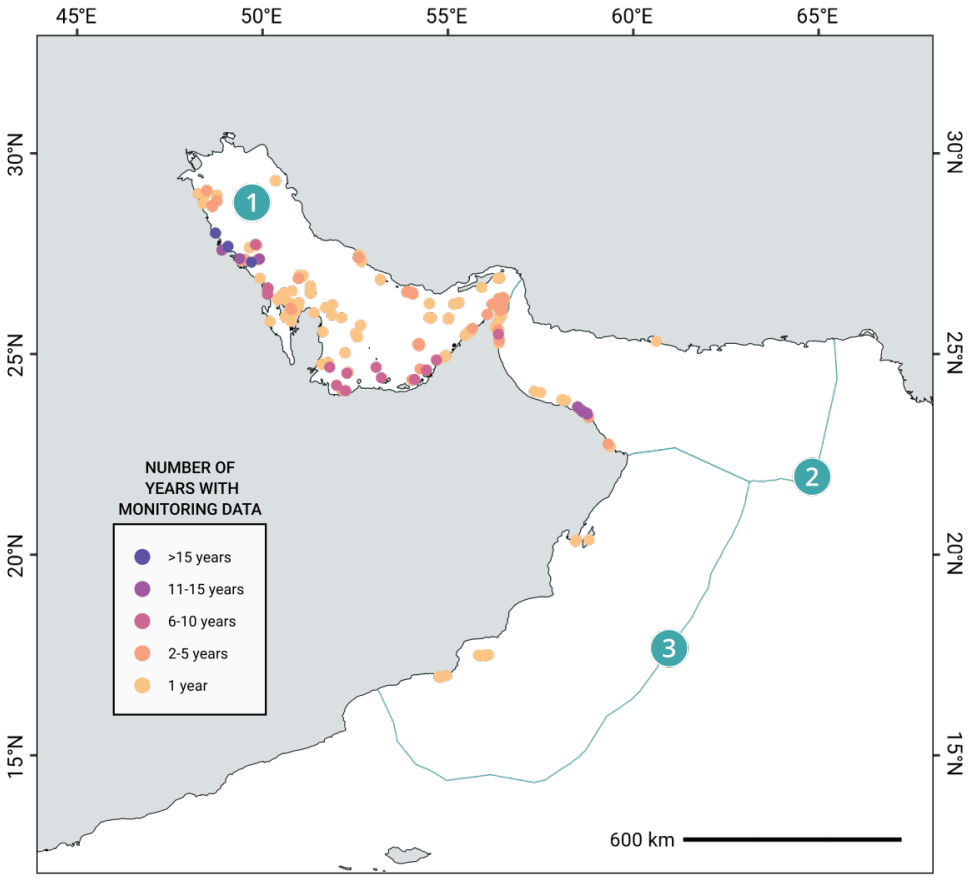
Recovery was evident through the 2010s, with the average hard coral cover increasing to over 35% by 2020 (Figure 2.8.6B). This resurgence likely reflects both natural recovery processes and the generally

cooler summer temperatures associated with upwelling, which historically reduced the propensity for bleaching in the Gulf of Oman [394]. However, this recovery trajectory was interrupted in the early 2020s (Figure 2.8.6B). Significant coral bleaching occurred in 2021, with profound impacts on the reefs throughout the Gulf of Oman, particularly in areas dominated by branching corals such as *Acropora* [413]. Although the magnitude of decline is more limited than in subregion 1 (from a median of ~41% to 30% between 2021 and 2024), the trend shows clear sensitivity to thermal anomalies, suggesting that even upwelling-influenced systems are increasingly vulnerable to intensified warming.

In subregion 2, macroalgal cover remains very low (averaging ~1–2%) throughout most of the time series, with only a modest rise after 2015 (Figure 2.8.7B). The subdued trajectory may reflect high grazing pressure and the episodic nature of disturbances. The model does not indicate a sustained subregional-scale macroalgal proliferation, despite being more nutrient-enriched compared to subregion 1 [394, 398].

ROPME 3 - Southern Oman

In contrast to subregions 1 and 2, subregion 3 cannot yet be robustly compared because interpretation is constrained by limited data, with only two time points (2017 and 2020) informing the model.



▲ **Figure 2.8.8.** Spatio-temporal distribution of benthic cover monitoring sites across the ROPME Sea Area region and respective subregions. Light blue polygons and numbers represent subregions. ROPME 1 = Persian/Arabian Gulf, ROPME 2 = Gulf of Oman, ROPME 3 = Southern Oman. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region.

The broad credible intervals reflect this uncertainty. Hard coral cover appears moderate (~20%), but highly uncertain, and macroalgal cover shows an apparent increase toward 2020, though this pattern should not be over-interpreted (Figure 2.8.6C and Figure 2.8.7C). Given the paucity of data, no firm spatial or temporal conclusions can be drawn. Increased monitoring surveys are warranted for this subregion.

DESCRIPTION OF MONITORING DATA

Temporal trends in the percentage cover of benthic categories discussed above were estimated using data from 34 datasets encompassing 379 monitoring sites (Figure 2.8.8). These data were collected through 862 surveys conducted between 1985 and 2024, with 90% of surveys taking place after 1993 (Figure A.1). The vast majority of sites (69.9%) were surveyed only within a single year (*i.e.* snapshot assessments), and only 4.2% of sites had survey data greater than a decade (Figure A.3); only three of the 379 reef sites had survey data extending beyond 15 years. While this indicates that long-term data availability remains a challenge, there have been dramatic improvements in recurrent monitoring in recent years, with 30.1% of sites now surveyed over 2–5 year time spans, compared with just 2.9% of sites in the 2020 regional assessment [417].

Of all surveys carried out in the region between 1985 and 2024, nearly three-quarters (73.1%, representing 76.8% of surveyed sites) were conducted in subregion 1 (Persian/Arabian Gulf), whereas only 1.7% (4.0% of sites) of surveys were conducted in subregion 3 (Southern Oman) (Table A.4). Subregion 1 also had the longest monitoring period, spanning four decades from 1985 to 2024, compared with the understudied subregion 3, where only two datasets are available in 2017 and 2020. Enhanced coral reef monitoring in Southern Oman is highly warranted, particularly given the unusual ecology of these pseudo-high-latitude reefs and the deficiencies in knowledge on status and trends [394, 398, 404].

CONCLUSIONS

Coral reefs of the ROPME Sea Area remain globally distinctive for their persistence under extreme environmental conditions, yet the evidence synthesized here indicates regional resilience is being progressively eroded. Across the region, hard coral cover has declined markedly from the high values observed in the 1980s and early 1990s, with repeated marine heatwaves and bleaching events driving stepwise losses, particularly in subregion 1 (Persian/Arabian Gulf), which dominates the regional signal. Subregion 2 (Gulf of Oman) follows a different but still concerning trajectory, shaped by cyclones, harmful algal blooms, and more recent thermal anomalies, while inference for subregion 3 (Southern Oman) remains constrained by limited monitoring. At the regional scale, macroalgal cover remains comparatively low, but the increasing prominence of turf algae indicates growing constraints on coral recovery and reinforces concern over future benthic restructuring. These trajectories point to a system increasingly characterized by shortened recovery intervals, recurrent disturbance, and declining structural complexity. Strengthened and more spatially balanced long-term monitoring, especially in Southern Oman, is now essential, as is management that reduces local stressors and protects herbivore and habitat function. Despite the exceptional thermal tolerance of these reefs, continued warming is likely to intensify vulnerability across the region.

DATA CONTRIBUTORS

Greta Smith Aeby, Jenan Al Asfoor, Suaad S. Al Harthi, Reem Al Mealla, Luai Muhammad Alhems, Andrew G. Bauman, Ivonne Bejarano, Radhouane Ben Hamadou, Rita Bento, John Burt, Michel Claereboudt, Sergey Dobretsov, Nigel Downing, Jan Freiwald, Amir Ghazilou, Jeneen Hadj-Hammou, Emily Howells, Thadickal Joydas, Antoine Leduc, Reynaldo Lindo, Elayne Looker, Diego Lozano-Cortés, Alyssa Marshell, Daniel Mateos, Jenny Mihaly, Rebekka Pentti, Ali M. Qasem, Pedro Range, Mohammad Reza Shokri, Dixit Sudhanshu.

Access to online supplementary materials

The modelled benthic cover data, detailed outputs from the data analyses, and the CRediT author contribution statement are available on Zenodo.

CASE STUDY 8

Data diplomacy in the world's hottest sea

Authors

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Introduction

The ROPME Sea Area, including the Arabian/Persian Gulf, Gulf of Oman, and Southern Oman, supports coral reefs occurring in some of the world's most extreme marine conditions. Summer sea temperatures exceed 35°C and salinities surpass 42 PSU, compared with 29°C and 35 PSU typical of tropical systems, and coastal development is intense. These reefs also face mounting pressures from marine heatwaves, and recent assessments show a 40% decline in coral cover since the mid-1990s, and exceeding 80% in some areas [418]. However, the 2020 Global Coral Reef Monitoring Network (GCRMN) report emphasized gaps in the scientific record that inhibited the use of the ROPME Sea Area data for management and conservation. Data gaps include limited time series, geographic omissions, and inconsistent methodologies [417].

To address these limitations, scientists from seven coastal nations (Bahrain, Iran, Kuwait, Oman, Qatar, Saudi Arabia, and the United Arab Emirates) launched a "data diplomacy" initiative in an effort to share, standardize, and jointly interpret coral reef monitoring data across borders. This case study describes the initiative's development, scientific outputs, and broader relevance.

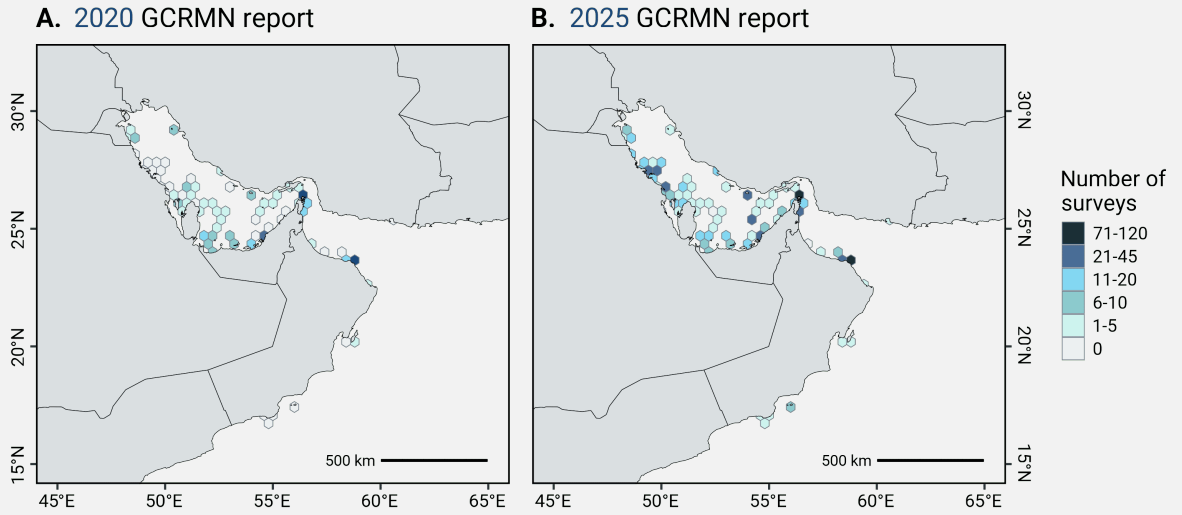
Genesis of data diplomacy

The 2020 ROPME regional assessment for GCRMN was based on only 68 sites, most surveyed only once, and using varied or undocumented methods [417]. Recognizing the need for consistency, stakeholders from across the regional nations endorsed three principles: (1) methodological convergence via standardized photo-quadrat protocols and metadata; (2) open but protected data access, with formal credit to data contributors; and (3) neutral stewardship by a trusted regional institution, with open-access co-authorship to ensure transparency.

Building a regional repository

A 2023 workshop at New York University (NYU) Abu Dhabi convened 46 stakeholders (all ROPME Sea Area nations except Iraq) who agreed to adopt a unified protocol. In 2024, a follow-up workshop trained participants in the MERMAID and ReefCloud platforms, piloting a data workflow from image collection to centralized archiving. These tools enabled AI-supported annotation and standardized data sharing with GCRMN. A formal data-sharing agreement was signed, with contributors guaranteed authorship in a public scientific dataset. By March 2025, over 2,000 georeferenced transects from 379 sites (1985–2024) had been compiled, five times

the number in 2020, increasing not only the temporal depth of data but additionally the spatial extent of coverage across the region (Figure C8.1). New elements including reef complexity and fish assemblage data for ecosystem-level analysis are also increasingly being incorporated.



▲ **Figure C8.1.** Map showing the spatial distribution of ROPME region coral reefs (polygons), shaded by the number of reef monitoring surveys that were contributed for the 2020 (A) versus 2025 (B) GCRMN assessment.

Scientific and policy insights

The expanded dataset has clarified spatial-temporal patterns of reef condition. While 2017 and 2023 bleaching events caused high mortality in the southern Gulf, reefs in the northern Gulf and Strait of Hormuz fared better. These differences, linked to local wind-driven cooling and hydrodynamic mixing, were previously undocumented. These analyses were only possible due to methodological harmonization, and this dataset will be critical for supporting management and conservation decisions, including the first regional IUCN Red List of Ecosystems (RLE) for coral reefs. Preliminary findings suggest strong signals under RLE Criteria A (reduction in geographic distribution) and C (abiotic environmental degradation). The architecture also supports evaluation of protected areas and transboundary conservation planning.

Lessons for other regions

Five lessons emerge from the ROPME Sea Area initiative: 1) neutral custodianship builds trust; 2) early methodological agreement prevents data incompatibility; 3) reciprocal credit encourages participation; 4) alignment with international frameworks ensures policy relevance; and 5) digital infrastructure supports scalability. Implementing these principles transformed fragmented efforts into a shared regional asset.

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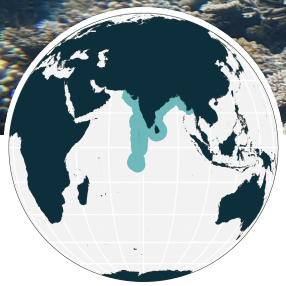
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Coral reefs in the Ari Atoll, Maldives
Credit: Irene Pancrazi



SOUTH ASIA

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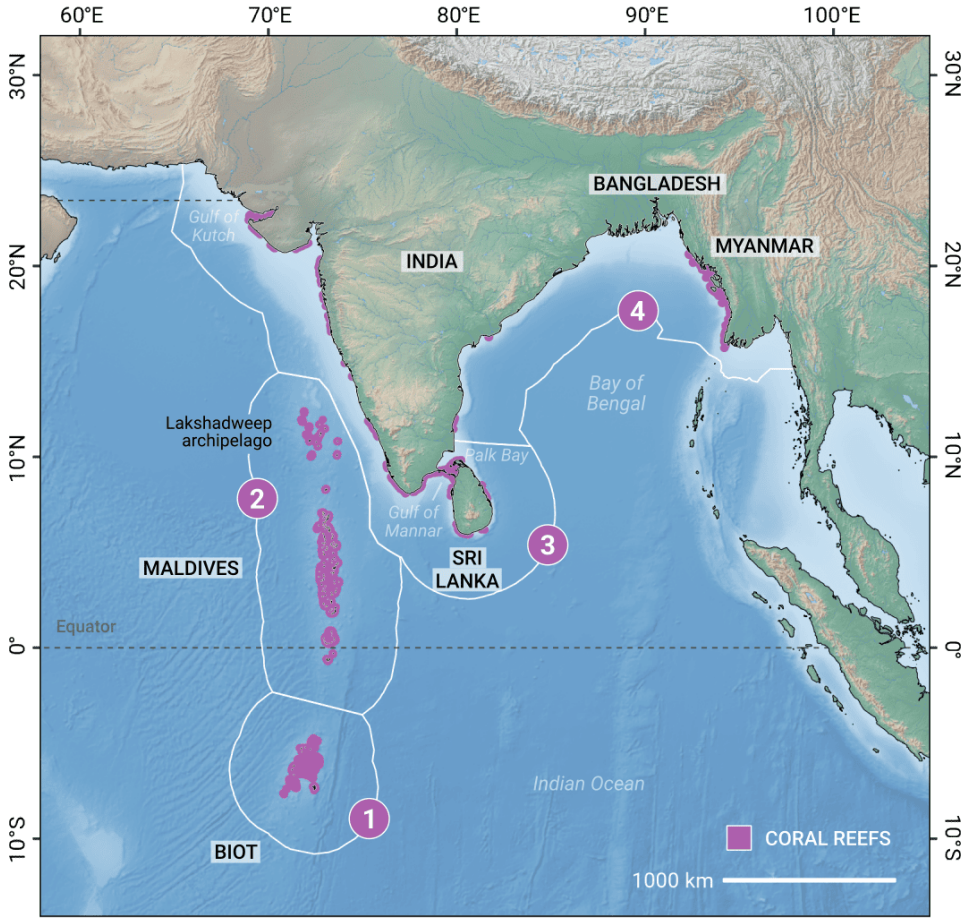
Citation

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On 14 May 2026, while this chapter was still in preparation, our co-author, Professor Monica Montefalcone, tragically passed away following a diving accident. Monica was an esteemed marine scientist whose work made significant contributions to the understanding and conservation of coral reefs in the Maldives. Her passing is deeply mourned by her colleagues, students, collaborators, and the wider marine science community. We dedicate this chapter to the memory of Monica Montefalcone, and the other divers who lost their lives as a result of the accident and the subsequent recovery operation.

INTRODUCTION

The GCRMN South Asia region extends over a maritime area of around 4.1 million km^2, ranging across 36° of latitude and 33° of longitude. The region hosts 10,755 km^2 of coral reefs, which represent 4.3% of the world's coral reef extent, making South Asia the seventh largest GCRMN region in terms of coral reef extent. Coral reefs are found within the exclusive economic zones of six countries and territories within the region, ranging from India in the north to the Chagos Archipelago in the south, and from India in the west to Myanmar in the east (Figure 2.9.1). For the purpose of this report, the GCRMN South Asia region was divided into four subregions (Figure 2.9.1). The Maldives subregion contains the largest proportion of coral reefs in the region, accounting for 58.5% (6,297 km^2) of the regional reef extent, whereas the Bay of Bengal subregion contains the smallest proportion, with only 7.2% (770 km^2, Table A.1).



▲ **Figure 2.9.1.** The distribution of coral reefs within the South Asia region and respective subregions. White polygons and numbers represent subregions. Subregion 1 = Chagos, Subregion 2 = Maldives, Subregion 3 = West and South Indian Shelf, Subregion 4 = Bay of Bengal. Coral reefs are delineated with a 20 km buffer to enhance interpretation, and only coral reefs within the region are represented. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region. BIOT = British Indian Ocean Territory.

The Chagos subregion contains one of the world's largest atoll structures called the Great Chagos Bank, where reefs are protected from fishing within a no-take MPA (Sheppard *et al.* [419], but see Collins *et al.* [420]). The Maldives subregion comprises the Republic of Maldives, with 26 atolls rising steeply from abyssal depths [421], as well as the Lakshadweep archipelago under India's jurisdiction. The West and South Indian Shelf includes coral reefs in the Gulf of Kutch, the Gulf of Mannar, Palk Bay, and around Sri Lanka (Figure 2.9.1). Finally, the Bay of Bengal subregion mostly includes coral reefs in Myanmar and those around St. Martin's Island in Bangladesh.

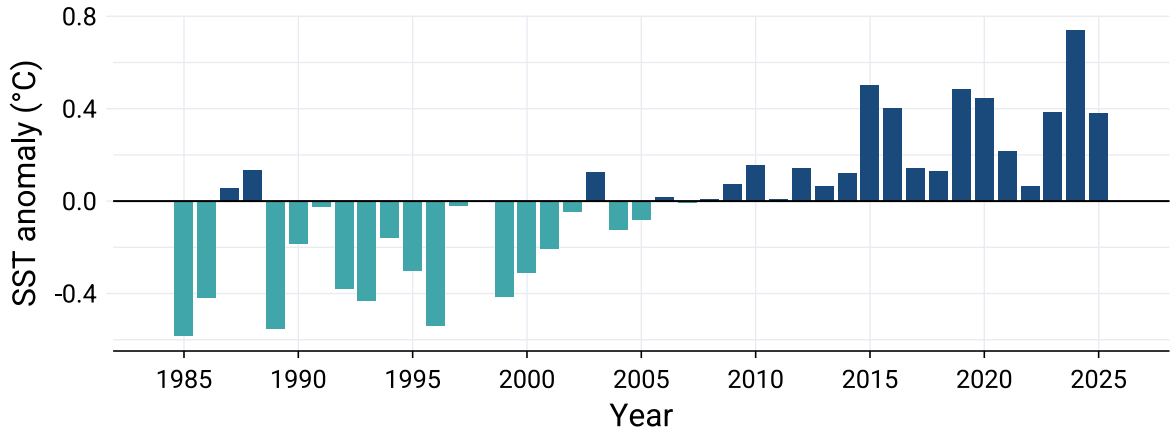
As of 2020, approximately 7.7 million people were living within 5 km of South Asia coral reefs. Between 2000 and 2020, the human populations residing within 5 km of coral reefs increased by 21.8%, corresponding to an average annual growth of approximately 70,000 inhabitants. The West and South Indian Shelf subregion exhibited the largest population increase between 2000 and 2020, with an additional 1 million inhabitants, and also accounted for the highest number of inhabitants in 2020 (Table A.2). The Chagos subregion has no permanent population, although Diego Garcia hosts a military presence.

THREATS AND DISTURBANCES

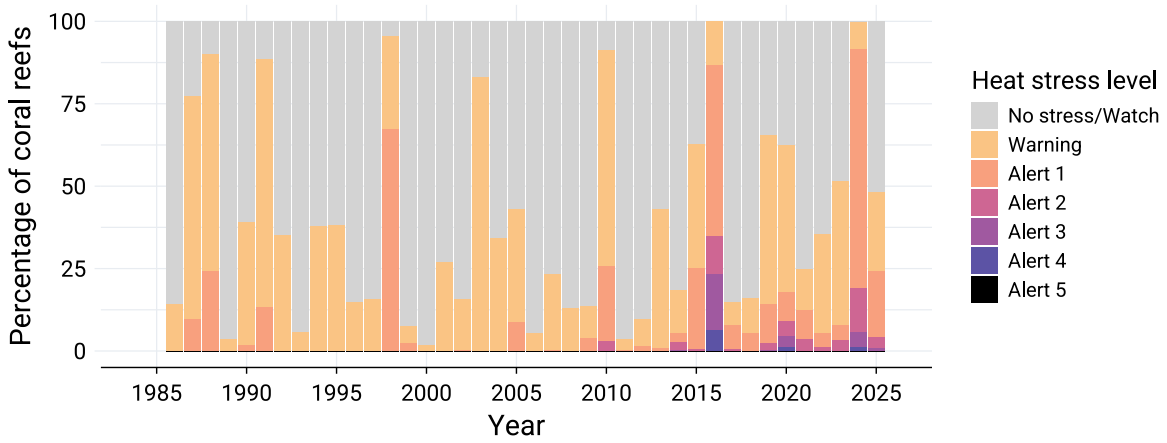
Heat stress

From 1985 to 2025, the long-term average of sea surface temperatures (SST) recorded over coral reefs within the South Asia region was 28.58 °C. During this period, the SST over coral reefs in the region increased by 0.79 °C due to anthropogenic climate change, corresponding to an average warming rate of 0.20 °C per decade (Figure 2.9.2, Table A.3).

Between 1986 and 2025, major episodes of heat stress affecting coral reefs in South Asia occurred in 1998, 2010, 2016, 2020, and 2024 (Figure 2.9.3). The heat stress events in 1998, 2010, 2016, and 2024 coincided with the four global bleaching events. In contrast to the severe impacts observed in 1998 and 2016, the 2010 event resulted in relatively limited hard coral mortality in the region [422–424].



▲ **Figure 2.9.2.** Annual mean sea surface temperature (SST) anomalies (°C) over the coral reefs of the South Asia region from 1985 to 2025. SST anomalies were calculated based on the long-term average over the period 1985–2025. Positive values (red bars) indicate SST warmer than the long-term average, whereas negative values (blue bars) indicate SST cooler than the long-term average. Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).



▲ **Figure 2.9.3.** Percentage of the coral reefs within the South Asia region experiencing heat stress levels per year, from 1986 to 2025. No stress/Watch (DHW = 0); Warning (possible bleaching, 0 $<$ DHW < 4); Alert 1 (reef-wide bleaching, 4 $\leq$ DHW $<$ 8); Alert 2 (reef-wide bleaching and mortality of heat-sensitive corals, 8 $\leq$ DHW $<$ 12); Alert 3 (multi-species mortality, 12 $\leq$ DHW $<$ 16); Alert 4 (severe multi-species mortality, 16 $\leq$ DHW $<$ 20); and Alert 5 (near-complete mortality, DHW $\geq$ 20). Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).

More recently, significant declines in hard coral cover have been reported in parts of the Maldives following the fourth global bleaching event [47, 425].

Cyclones and tsunamis

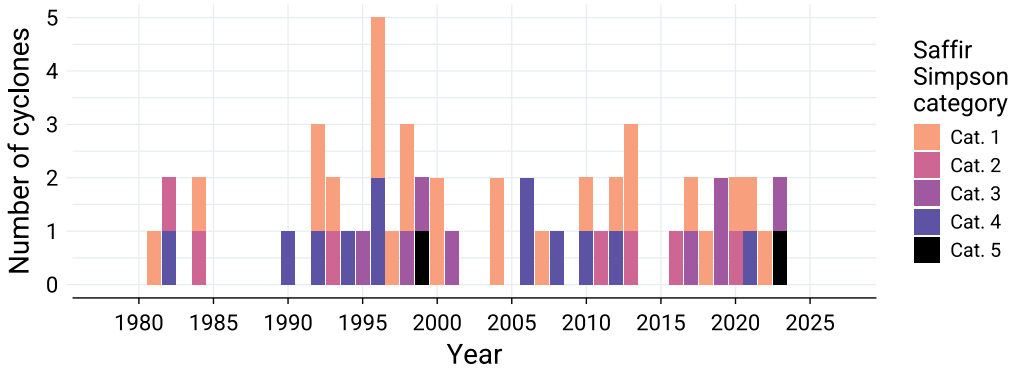
From 1980 to 2025, a total of 54 cyclones passed within 100 km of coral reefs in the South Asia region, representing an average of 1.2 $\pm$ 1.1 SD cyclones per year (Figure 2.9.4). The cyclone with the highest sustained wind speed over the studied period was the cyclone Giri, in October 2010, with sustained wind speed of 250 $km.h^{-1}$ (Figure 2.9.5). Coral reefs located near the equator, such as those in the southern Maldives, are rarely affected by cyclones [108].

In addition to cyclones, tsunamis represent another natural disturbance capable of causing physical damage to coral reefs. In particular, the 2004 Sumatran earthquake and tsunami had limited but spatially heterogeneous impacts on coral reefs, with some of the most significant effects observed in Sri Lanka [426–428].

Regional disturbances

In addition to climatic disturbances, coral reefs in South Asia are also affected by local human pressures linked to growing coastal populations and expanding tourism [429]. While coral mining was historically practiced in the Maldives for construction [430], it has now largely been discontinued. However, land reclamation and dredging continue to pose significant threats by increasing sedimentation and causing habitat destruction [421]. More broadly, with the exception of the uninhabited Chagos Islands, reefs across the region are impacted by coastal development, eutrophication, pollution, and unsustainable fishing [421]. Land-based pollution, particularly from wastewater, remains an understudied but important threat to coral reefs in South Asia [431].

Additional disturbances, partly driven by human activities, have also been reported. In the Gulf of Mannar, blooms of *Noctiluca scintillans* have caused hypoxic conditions and coral mortality [432]. In the Lakshadweep archipelago, coral diseases have been observed alongside the overgrowth of hard corals by algae and sponges [433].



▲ **Figure 2.9.4.** Number of cyclones that passed within 100 km of the coral reefs of the South Asia region from 1980 to 2025. Colors correspond to the Saffir–Simpson category of the cyclone along its entire track. Derived from IBTrACS data (Knapp *et al.* [118]; see Materials and Methods).

Outbreaks of *Acanthaster planci* (crown-of-thorns starfish) have also been documented in the Maldives [434, 435].

Together, these local stressors are likely to weaken the resilience of coral reefs to climate-related impacts, highlighting the urgent need for both improved local management and sustained global climate action [91].

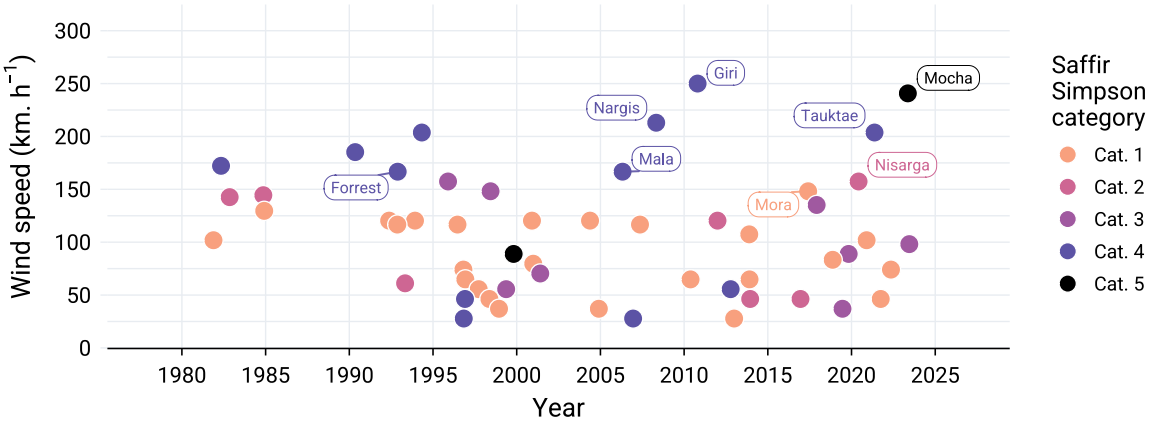
STATUS OF CORAL REEFS IN THE REGION

Hard coral cover

Hard coral cover in the South Asia region increased from 30.1% (credible interval at 95% [CrI $_{95\%}$]: 26.0 – 34.5%) in 1997–2009 to 35.0% (CrI $_{95\%}$: 30.7 – 39.5%) in 2020–2024, a change supported by strong evidence (Figure 2.9.6A). This represents an absolute increase of 4.9 percentage points, corresponding to a relative increase of 16.2% in hard coral cover. Hard coral cover showed two periods of major decline in 1997–1998 and 2015–2016, and two subsequent periods of recovery.

The first period of decline in hard coral cover, which occurred in 1997–1998, led to an absolute loss of 21.3 percentage points (Figure 2.9.6A). This decline was associated with the first global bleaching event, whose impacts were particularly high in the Maldives [422]. Hard coral cover then increased until 2013, when it reached 39.3%, approaching the level of hard coral cover recorded before the impact of the first global bleaching event. However, Montefalcone *et al.* [422] showed that the benthic cover of branching and tabular *Acropora* species only partially recovered, leading to a change in the hard coral assemblage. As reported in previous studies, the second global bleaching event in 2010 had only limited impacts on hard coral cover in the South Asia region, likely because the associated heat stress was less severe than during other major heat stress events [422, 423] (Figure 2.9.3).

The second period of decline in hard coral cover occurred in 2015–2016, due to the third global bleaching event. The absolute loss in hard coral cover was 12.8 percentage points, representing a smaller decline than that observed following the first global bleaching event (Figure 2.9.6A). However, similarly to the first global bleaching event, the heat stress event of 2015–2016 led to a change in the hard coral assemblage, with a relative decline of branching and digitate *Acropora* and a relative increase of massive hard coral species [424, 436].



▲ **Figure 2.9.5.** Maximum sustained wind speed of cyclones that passed within 100 km of a coral reef in the South Asia region between 1980 and 2025. Colors correspond to the Saffir–Simpson category of the cyclone along its entire track. However, the values of sustained wind speed are extracted from the nearest cyclone position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir–Simpson scale (*i.e.* 119 km.h^{-1}). Derived from IBTrACS data (Knapp *et al.* [118]; see Materials and Methods).

Macroalgal cover

Macroalgal cover in the region increased from 2.0% (CrI $_{95\%}$: 1.6 – 2.7%) in 1997–2009 to 2.8% (CrI $_{95\%}$: 2.3 – 3.5%) in 2020–2024, a change supported by strong evidence (Figure 2.9.6B). This corresponds to an absolute increase of 0.8 percentage points, or a relative increase of 36.4%. Despite this high relative increase, macroalgae remain minor benthic components in South Asian coral reefs.

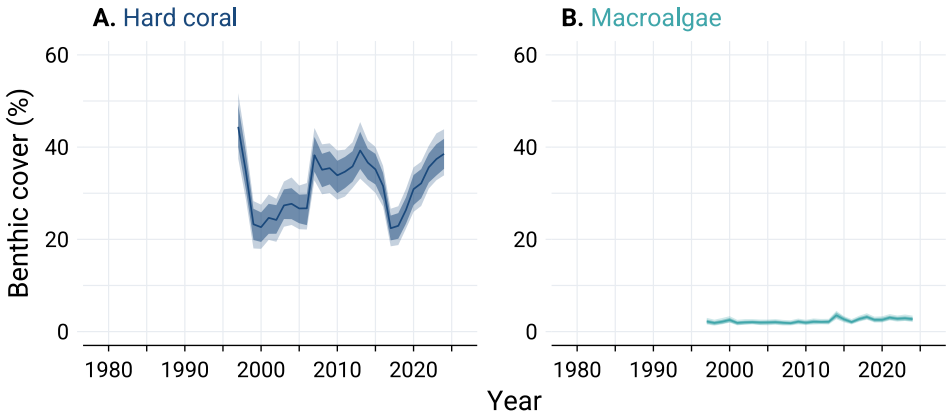
STATUS OF CORAL REEFS IN THE SUBREGIONS

South Asia 1 - Chagos

Coral reefs in the Chagos subregion occur exclusively within the Chagos Archipelago, which comprises seven atolls and several banks located at varying depths. The Archipelago was designated as a no-take marine protected area in 2010 [419].

There is no evidence of a change in hard coral cover in the Chagos subregion between 2010–2014 and 2020–2023 (Figure 2.9.7A). However, this apparent long-term stability masks pronounced interannual variability. Although our estimates do not extend prior to 2010, previous studies indicate that hard coral cover declined by approximately 50 percentage points following heat stress associated with the 1998 global bleaching event [437], before gradually recovering over the subsequent decade [423]. Hard coral cover then declined from 40.9% in 2010 to 13.0% in 2017, with most of the loss occurring in 2015 and 2016 (Figure 2.9.7A). This represents an absolute decline of 27.9 percentage points, largely driven by the third global bleaching event [423, 438, 439]. Hard coral cover subsequently recovered to 37.9% by 2023, approaching levels observed in 2010 (Figure 2.9.7A). Although the available data do not cover this period, a recent study indicates a 23% relative decline in hard coral cover in 2024, following the fourth global bleaching event [440].

Similarly to hard coral cover, there is no evidence of a change in macroalgal cover in the Chagos subregion between 2010–2014 and 2020–2023 (Figure 2.9.8A). Macroalgal cover peaked at around 5.0% in 2018, although the drivers of this increase remain unclear.



▲ **Figure 2.9.6.** Modeled temporal trends of benthic cover in the South Asia region from 1980 to 2024 for hard coral (A) and macroalgae (B). The solid line represents the estimated median benthic cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

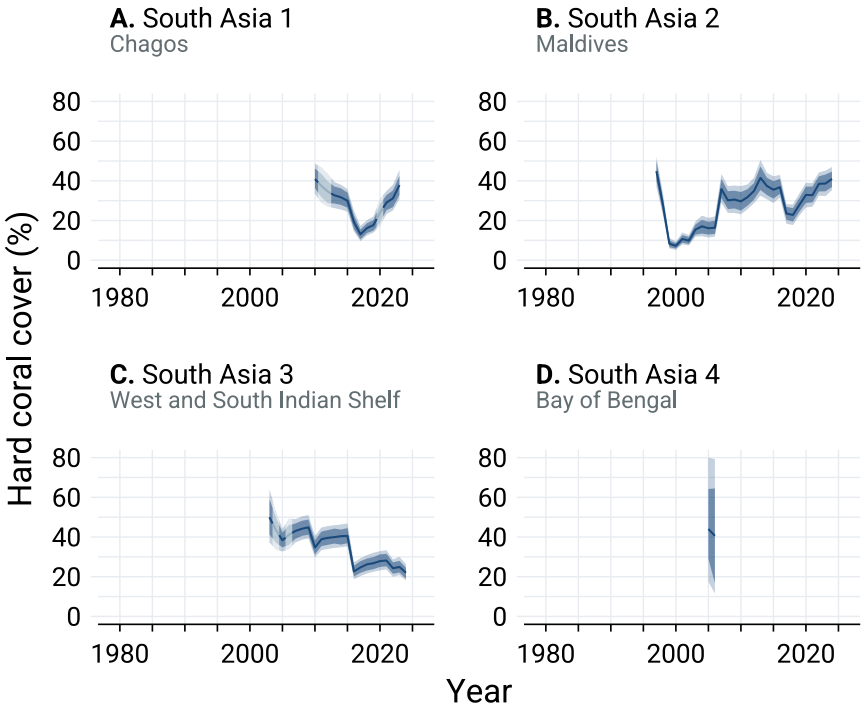
South Asia 2 - Maldives

The Maldives subregion encompasses coral reefs in the Republic of Maldives and the Indian archipelago of Lakshadweep. In this subregion, results provide strong evidence that hard coral cover increased between 1997–2009 and 2020–2024 (Figure 2.9.7B). However, this long-term increase masks pronounced temporal variability, with alternating periods of decline (2000 and 2013–2017) and recovery (2003–2009 and 2018–2024). A major decline followed the first global bleaching event, during which hard coral cover decreased from 44.9% in 1997 to 7.3% in 2000, corresponding to an absolute loss of 37.6 percentage points. In the Lakshadweep archipelago, Arthur [441] reported bleaching prevalence of 82% and hard coral mortality of 26% following this heat stress event. Hard coral cover subsequently increased until 2007 and remained relatively stable until 2015, reaching 35.5% (Figure 2.9.7B). A second decline occurred in 2016 in association with the third global bleaching event [442], although the magnitude of loss was lower than that observed following the first global bleaching event. This event also modified hard coral assemblages, with a relative decline in branching and tabular *Acropora* [422, 436]. As the study period does not include 2025, the impacts of the fourth global bleaching event (2023–2024) could not be assessed. However, Montefalcone *et al.* [425] reported a 40% loss in hard coral cover in the central atolls of the Maldives, but no detectable mortality in the southern part of the archipelago, following the fourth global bleaching event.

Macroalgal cover increased by 0.6 percentage points between 1997–2009 and 2020–2024, a change supported by strong evidence (Figure 2.9.8B). Peaks in macroalgal cover occurred around 3.0% in 2000 and 4.5% in 2014, broadly coinciding with declines in hard coral cover (Figure 2.9.8B).

South Asia 3 - West and South Indian Shelf

Coral reefs in the West and South Indian Shelf subregion occur along the western and southern coasts of India and in Sri Lanka. In this subregion, data provide strong evidence that hard coral cover declined between 2003–2009 and 2020–2024 (Figure 2.9.7C). Although our estimates do not extend prior to 2003, Arthur [441] reported limited bleaching prevalence and mortality following the 1998 global bleach-



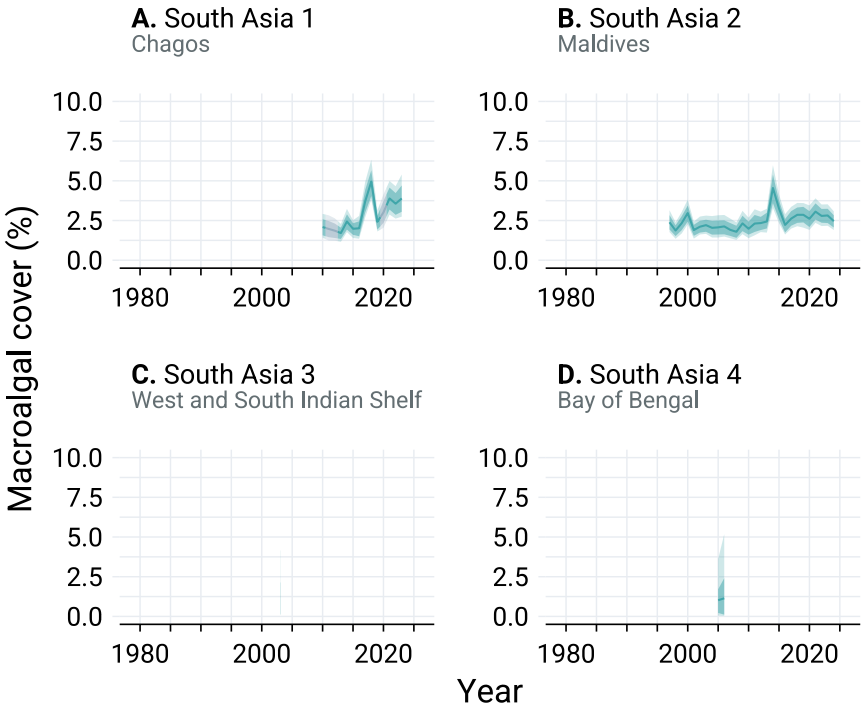
▲ **Figure 2.9.7.** Modeled temporal trends of hard coral cover in South Asia subregions from 1980 to 2024. The solid line represents the estimated median hard coral cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

-ing event in the Gulf of Kutch, but 23% bleaching-related coral mortality in the Gulf of Mannar. From 2003 onward, hard coral cover exhibited a stepwise decline, with pronounced absolute decreases in 2003–2005 (-11.5 percentage points), 2009–2010 (-10.2 percentage points), and 2015–2016 (-17.9 percentage points). The first decrease coincides with the Sumatran-Andaman tsunami of December 2004 [443], whereas the 2009 and 2015 declines align with the second and third global bleaching events. Raj *et al.* [444] reported approximately 71% bleaching prevalence and 8% mortality in coral reefs of the Malvan Marine Sanctuary on the western Indian coast following the third bleaching event. In the Gulf of Mannar, this third global bleaching event led to an approximately 50% relative decline in mean hard coral cover [424]. In 2019, an algal bloom of *Noctiluca scintillans* led to a hypoxia phenomenon which caused a mortality of hard coral colonies in the Gulf of Mannar [432].

Owing to the absence of macroalgal cover data integrated into our analyses for this subregion, long-term trends for this benthic category could not be estimated. However, Raj *et al.* [424] reported an increase in total algal cover following the decline in hard coral cover during 2015–2016.

South Asia 4 - Bay of Bengal

The Bay of Bengal subregion encompasses coral reefs along the eastern coast of India, the coast of Myanmar, and around Saint Martin's Island in Bangladesh. Due to the very limited number of surveys available for this subregion, long-term changes in hard coral and macroalgal cover between 1980 and 2024 could not be estimated (Figure 2.9.7D and Figure 2.9.8D). In 2005–2006, hard coral and macroalgae accounted for 42% and 1% of the benthic cover, respectively.



▲ **Figure 2.9.8.** Modeled temporal trends of macroalgal cover in South Asia subregions from 1980 to 2024. The solid line represents the estimated median macroalgal cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

DESCRIPTION OF MONITORING DATA

Temporal trends in the percentage cover of benthic categories were estimated using data from 15 datasets encompassing 636 monitoring sites. These data were collected through 2,825 surveys conducted between 1997 and 2024, with 90% of surveys taking place after 2005 (Figure A.1). Of all surveys carried out in the region between 1997 and 2024, 57% (representing 75% of surveyed sites) were conducted in the West and South Indian Shelf subregion, whereas only 0.1% (0.3% of sites) were conducted in the Bay of Bengal subregion (Table A.4). The temporal distribution of surveys shows a substantial increase in monitoring effort over time, reflecting the expansion of reef monitoring programs across the region in the early 2000s (Figure A.1).

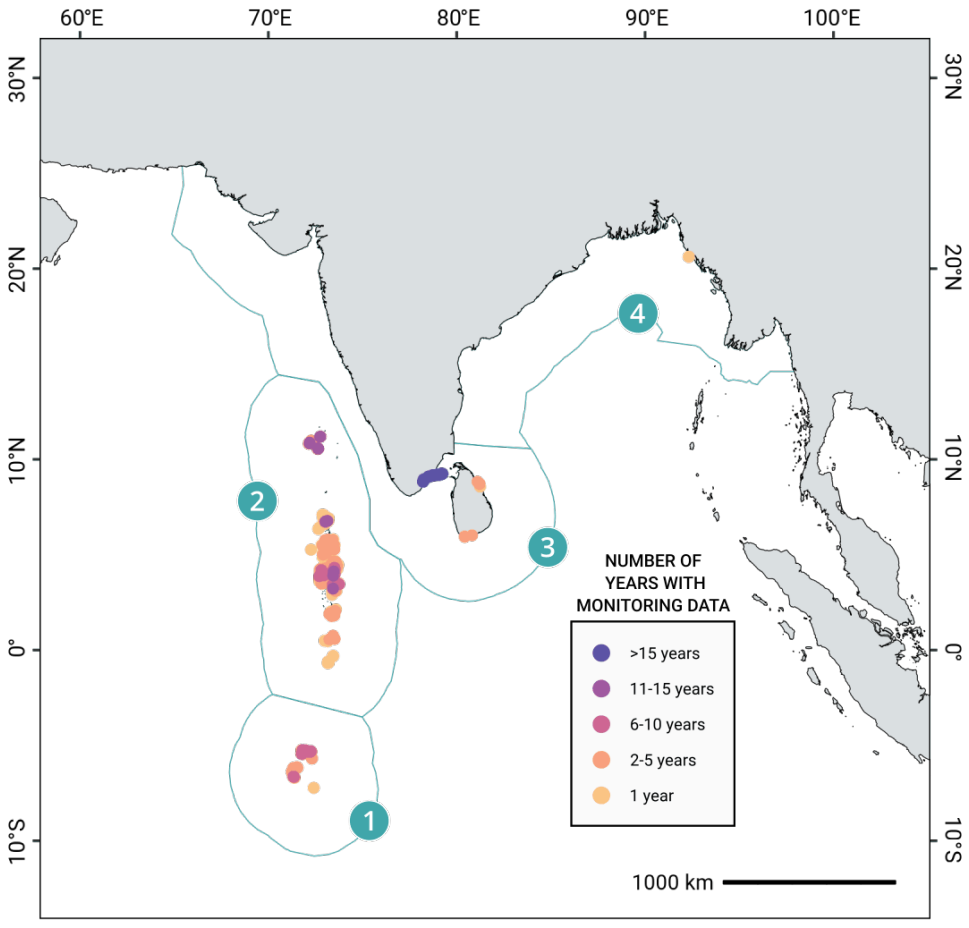
CONCLUSIONS

The results indicate a 4.9 percentage point increase in hard coral cover in the South Asia region between 1997–2009 and 2020–2024. However, this increase largely reflects the low baseline values observed during the first period, and the rapid recovery recorded in the second period. Over the full time series, hard coral cover remained highly dynamic, with a difference of 20 percentage points between its lowest and highest values. Macroalgal cover increased by only 0.8 percentage points over the same periods, and continues to represent a minor component of South Asian coral reef benthic communities.

Regional trends, particularly the overall increase in hard coral cover, were largely driven by the Chagos and Maldives subregions, which together account for 83% of the total regional coral reef extent. In contrast, the West and South Indian Shelf subregion

shows a decline in hard coral cover from 2003 to 2024, suggesting continental reef systems may be more affected by global and local anthropogenic pressures than oceanic archipelagos such as Chagos, the Maldives, and Lakshadweep. This pattern is consistent with the region's demographic context, as the West and South Indian Shelf subregion concentrates the largest proportion of the human population living within 5 km of coral reefs in South Asia (Table A.2).

While reefs in Chagos, central Maldives, Lakshadweep, and the Gulf of Mannar appear to be sufficiently monitored across space and time to support robust assessments of reef condition, monitoring remains spatially uneven across much of the region. In particular, several coral reef systems within the West and South Indian Shelf remain poorly represented. More concerningly, data from the Bay of Bengal subregion are insufficient to support a long-term assessment of hard coral cover, highlighting a critical need for improved monitoring efforts in this subregion.



▲ **Figure 2.9.9.** Spatio-temporal distribution of benthic cover monitoring sites across the South Asia region and respective subregions. Light blue polygons and numbers represent subregions. Subregion 1 = Chagos, Subregion 2 = Maldives, Subregion 3 = West and South Indian Shelf, Subregion 4 = Bay of Bengal. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region.

DATA CONTRIBUTORS

Khadeeja Ali, Cassandra E. Benkwitt, Carlo Nike Bianchi, Graham J. Edgar, Ida Elizabeth George, Mohamed F.M. Fairoz, Jan Freiwald, Javier Gonzalez-Barrios, Nicholas Graham, Rachel Gunn, Shaha Aminath Hashim, Titus Immanuel, Vibitha M. A., Jenny Mihaly, Monica Montefalcone, Carla Morri, Radhika Nair, Elizabeth Oh, Edward J.K. Patterson, Morgan Pratchett, Abdul Raheem C. N., Rajkumar Rajan, Laura Richardson, Ronan Roche, Jyodee Sannassy Pilly, Mohamed Shimal, Danielle Spring, Rick D. Stuart-Smith, Nina Tabitha Simon, John Turner, Swetha V. Rajan, Shaun Wilson.

Access to online supplementary materials

The modelled benthic cover data, detailed outputs from the data analyses, and the CRediT author contribution statement are available on Zenodo.

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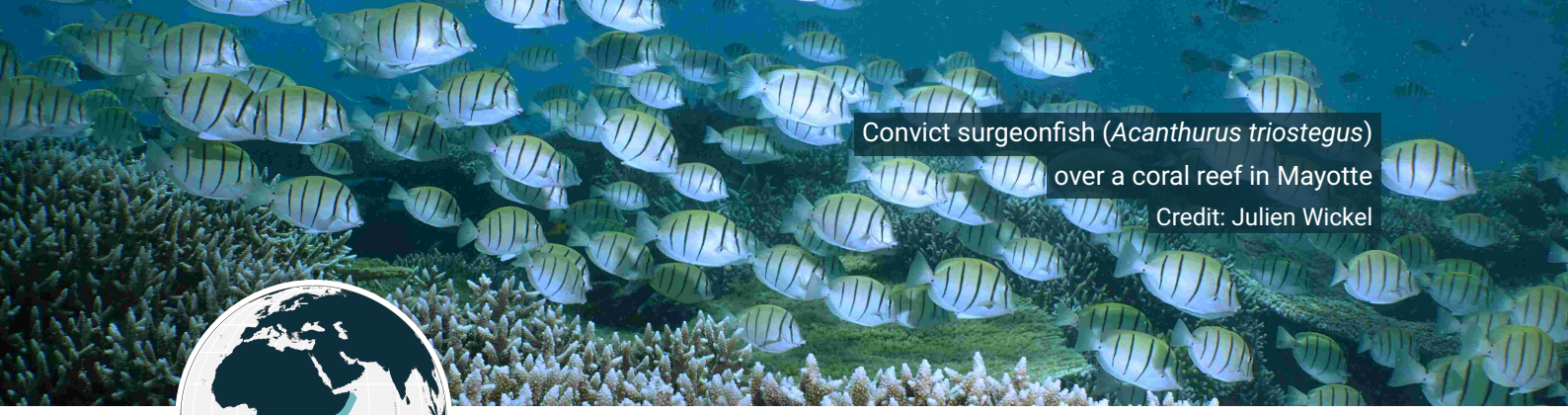
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Convict surgeonfish (*Acanthurus triostegus*) over a coral reef in Mayotte
Credit: Julien Wickel



WESTERN INDIAN OCEAN

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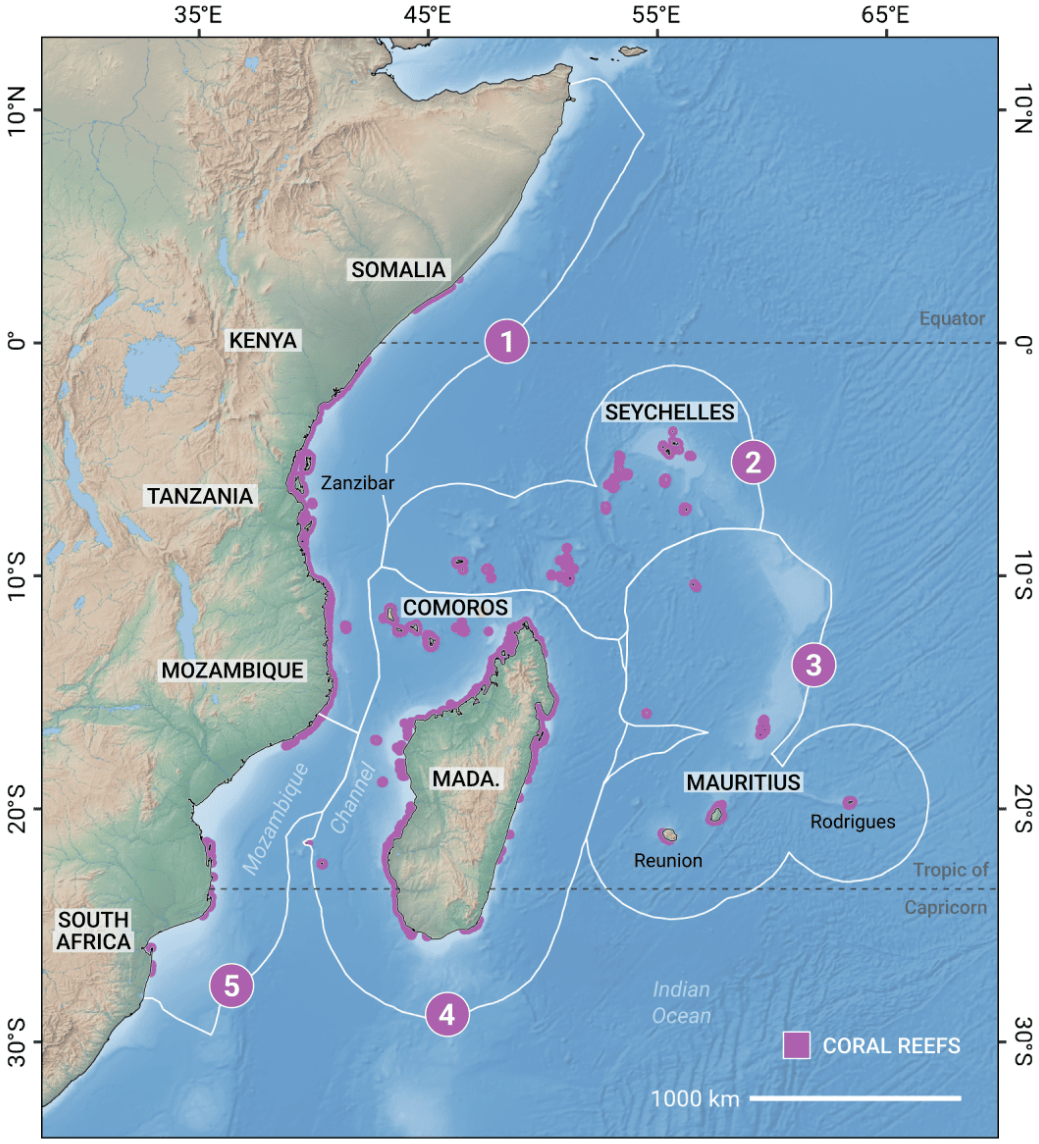
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INTRODUCTION

The Western Indian Ocean GCRMN region extends over a maritime area of approximately 6.4 million km^2, ranging across 41° of latitude and 34° of longitude. The region contains 14,682 km^2 of coral reefs, representing 5.9% of the world's coral reef extent, making it the fifth largest GCRMN region in terms of coral reef extent. These coral reefs are recognised as globally important for marine biodiversity [445, 446] and are distributed across the exclusive economic zones of 16 countries and territories, ranging from the Federal Republic of Somalia in the north to South Africa in the south, and from the Republic of Mauritius in the east to mainland Africa in the west (Figure 2.10.1).



▲ **Figure 2.10.1.** The distribution of coral reefs within the Western Indian Ocean region and respective subregions. White polygons and numbers represent subregions. Subregion 1 = East African Coast, Subregion 2 = Seychelles, Subregion 3 = Mascarene Islands, Subregion 4 = Madagascar and Comoros, Subregion 5 = Bight of Sofala. Coral reefs are delineated with a 20 km buffer to enhance interpretation, and only coral reefs within the region are represented. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region. MADA. = Madagascar.

The Western Indian Ocean region is divided into five subregions (Figure 2.10.1). The East African Coast (subregion 1) is the largest subregion by coral reef extent, hosting 42.8% (6,285 km^2) of the region's coral reefs. Although there are numerous Marine Protected Areas (MPAs), reserves, and conservation areas in this subregion, management effectiveness varies [447, 448]. The Madagascar and Comoros subregion (subregion 4) contains 35.3% (5,176 km^2) of the regional coral reef extent, with a substantial proportion of reefs located within MPAs, although management effectiveness also varies across this subregion [449]. The Seychelles subregion (subregion 2), hosts 13.0% (1,904 km^2) of the region's coral reefs, of which more than 95% of coral reefs are found within designated protected areas under the Seychelles Marine Spatial Plan. This includes 18.5% in strict no-take zones, with the remainder managed as Sustainable Use Areas [450]. The Mascarene Islands (subregion 3) contains 6.9% (1,014 km^2) of the region's coral reefs, distributed across volcanic islands and low-lying coral atolls [451] and is considered one of the highest centres of marine endemism in the region [452]. Although the Bight of Sofala subregion (subregion 5) hosts only 2.1% (302 km^2) of coral reefs in the region, it supports notable biodiversity and protected areas. This includes the iSimangaliso Wetland Park, a UNESCO World Heritage Site and MPA, the Primeiras and Segundas Environmental Protection Area, and Bazaruto National Park [453].

Between 2000 and 2020, the human population residing within 5 km of coral reefs in the region increased by 84.2%, representing an average annual increase of around 240,000 inhabitants. By 2020, a total of 10.5 million inhabitants lived within 5 km of coral reefs across the region. The greatest population increase over this period was recorded in the East African Coast subregion, which also had the highest number of inhabitants in the region reported in 2020 (Table A.2).

Coral reefs provide critical contributions to nature and people across the region. They support livelihoods, food security, and cultural identity for coastal communities, underpin national and local economies, and host the biodiversity necessary to support these benefits [25]. However, a recent IUCN Red List of Ecosystems assessment classified coral reefs in the region as vulnerable to collapse, jeopardising the interconnected ecosystem services on which coastal communities depend [99].

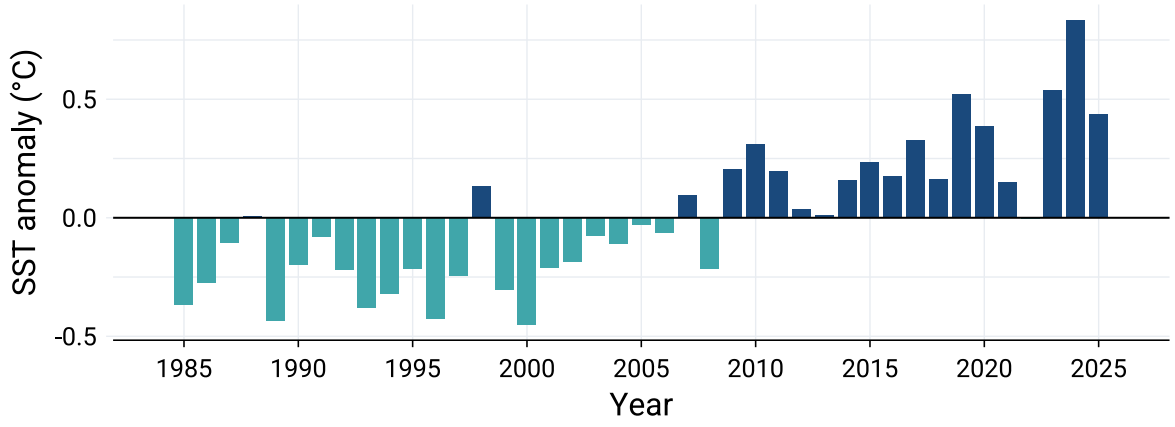
THREATS AND DISTURBANCES

Heat stress

From 1985 to 2025, long-term average sea surface temperature (SST) recorded over coral reefs within the region was 27.19 °C. SST increased by 0.74 °C over the period, corresponding to an average warming rate of 0.18 °C per decade (Table A.3). Conditions warmer than the long-term average were rare prior to 2009, but have occurred almost yearly since (Figure 2.10.2), with positive SST anomalies accumulating and driving increasingly recurrent and severe coral bleaching events [454].

From 1986 to 2025, the Western Indian Ocean experienced five major regional heat stress events (Figure 2.10.3). In 1998, 47% of coral reefs in the region were exposed to more than 4 Degree Heating Weeks (DHW), reaching 77% for the East African Coast subregion (subregion 1). In 2010, a heat stress event exposed 29% of reefs to more than 4 DHW, with exposure reaching 89% in the Seychelles subregion (subregion 2).

By 2016, during the longest sustained period of global heat stress on record (2014–2017) [31], 82% of reefs experienced more than 4 DHW. Values exceeding 16 DHW were recorded in the region for the first time, affecting 4% of reefs in the Mascarene Islands subregion (subregion 3). A subsequent two-year heat stress event in 2019–2020, associ-



▲ **Figure 2.10.2.** Annual mean sea surface temperature (SST) anomalies (°C) over the coral reefs of the Western Indian Ocean region from 1985 to 2025. SST anomalies were calculated based on the long-term average over the period 1985–2025. Positive values (red bars) indicate SST warmer than the long-term average, whereas negative values (blue bars) indicate SST cooler than the long-term average. Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).

ated with one of the strongest positive Indian Ocean Dipole (IOD) events on record [455], exposed 32% of reefs to more than 4 DHW in 2019, increasing to 59% in 2020.

The most severe heat stress event occurred during 2024–2025, driven by the combined effects of El Niño, a positive IOD, and record global ocean temperatures [456]. During this global bleaching event, 93% of reefs experienced more than 4 DHW (Figure 2.10.3), and values exceeding 20 DHW were documented in the region for the first time.

Heat stress events have also been linked to major cyclonic activity, with heat-stressed corals becoming more structurally vulnerable to physical damage [109].

Cyclones

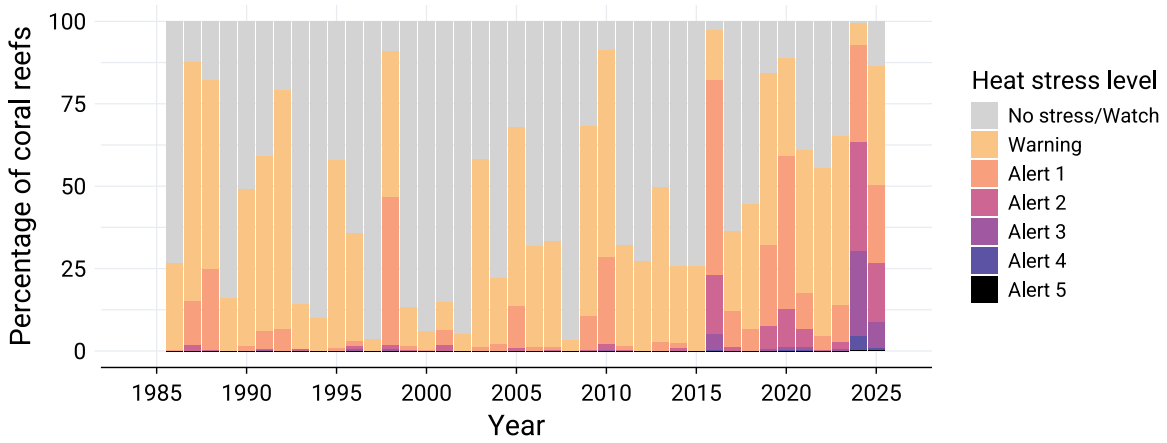
From 1980 to 2025, a total of 151 cyclones passed within 100 km of coral reefs in the Western Indian Ocean, averaging 3.4 $\pm$ 2 SD cyclones per year (Figure 2.10.4).

The cyclone with the highest sustained wind speed over the studied period was Cyclone Hary in March 2002, which passed over reefs in northern Madagascar with sustained wind speeds of 259 km.h^{-1} (Figure 2.10.5). The years 2024 and 2025 saw a rapid succession of cyclones impacting the region, with cumulative coral mortality of 66% and a 35% loss of hard coral cover reported for Mayotte [457], following the combined impacts of the 2024 bleaching event and Cyclone Chido (Figure 2.10.3 and Figure 2.10.5).

Synergistic impacts between heat stress and cyclone damage compound reef degradation, and cyclones passing beyond the 100 km threshold may still cause notable damage to reefs not captured in this analysis [108].

Regional disturbances

Coral reefs in the Western Indian Ocean region face multiple pressures that compound their vulnerability. Unsustainable and destructive fishing practices reduce fish biomass, alter community structure, and directly degrade reef habitat [458]. The overfishing of key herbivorous fishes diminishes ecosystem recovery capacity by reducing grazing pressure on algae, allowing turf and fleshy macroalgae to colonise substrate that would otherwise be available for coral recruitment [459]. Overfishing of predatory



▲ **Figure 2.10.3.** Percentage of the coral reefs within the Western Indian Ocean region experiencing heat stress levels per year, from 1986 to 2025. No stress/Watch (DHW = 0); Warning (possible bleaching, 0 < DHW < 4); Alert 1 (reef-wide bleaching, 4 ≤ DHW < 8); Alert 2 (reef-wide bleaching and mortality of heat-sensitive corals, 8 ≤ DHW < 12); Alert 3 (multi-species mortality, 12 ≤ DHW < 16); Alert 4 (severe multi-species mortality, 16 ≤ DHW < 20); and Alert 5 (near-complete mortality, DHW ≥ 20). Derived from NOAA Coral Reef Watch data (Skirving *et al.* [107]; see Materials and Methods).

fishes can drive elevated sea urchin densities, increasing bioerosion of reef flats [460], while localised declines in sea urchin populations, particularly *Diadema setosum*, often following bleaching events, have reduced grazing pressure and facilitated macroalgal growth [461, 462]. Overfishing of mesopredators, often in combination with nutrient loading, has been linked to crown-of-thorns starfish (*Acanthaster* spp.) outbreaks, which have been intermittently reported, and can cause rapid, localised loss of hard coral cover [313].

Periodic river flooding of nearshore reefs, often amplified by land-use change, is another major disturbance in the region, delivering increased sediment, freshwater, and nutrient loads, and contributing to localised coral mortality or slowing post-disturbance recovery [48, 463].

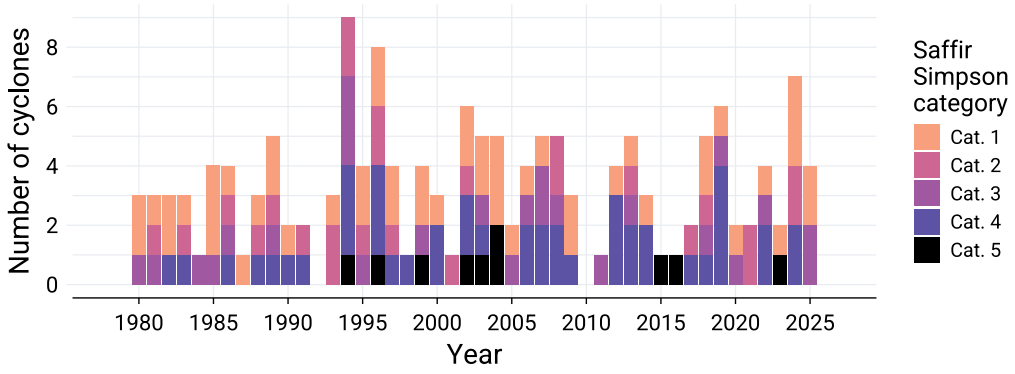
Coral disease and growth anomalies have also been reported with increasing frequency across the region, particularly following major disturbances [464]. As intervals between disturbance events shorten, these stressors act synergistically with climate-driven pressures, compounding cumulative impacts on coral reefs.

STATUS OF CORAL REEFS IN THE REGION

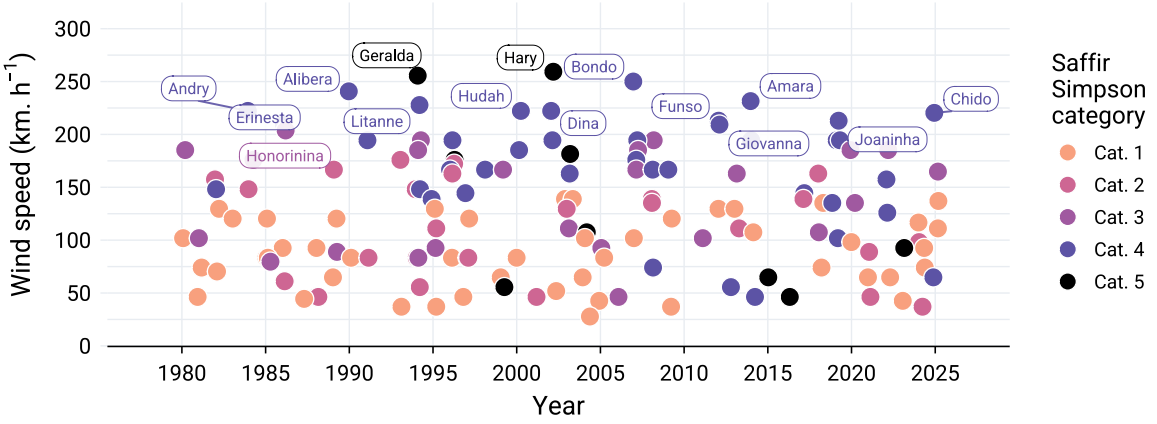
Hard coral cover

The modelled data showed an apparent broadly positive trajectory in hard coral cover in the Western Indian Ocean since 1987, peaking at 33.9% (credible interval at 95% [$CrI_{95\%}$]: 30.1 – 37.4%) in 2023, before declining sharply to 25.9% ($CrI_{95\%}$: 22.8 – 29.1%) in 2024 (Figure 2.10.6A). A comparison of the reference period (1987–2009) with modern hard coral cover estimates (2020–2024) from the model demonstrated strong evidence of an increase. However, this apparent long-term trend should be interpreted with caution in light of the structural characteristics of the underlying dataset and broader scientific literature.

Early modelled estimates (1987–1997) indicated an average of 20.3% hard coral cover. However, this baseline carries uncertainty due to the sparsity of available data.



▲ **Figure 2.10.4.** Number of cyclones that passed within 100 km of the coral reefs of the Western Indian Ocean region from 1980 to 2025. Colors correspond to the Saffir–Simpson category of the cyclone along its entire track. Derived from IBTrACS data (Knapp *et al.* [118]; see Materials and Methods).



▲ **Figure 2.10.5.** Maximum sustained wind speed of cyclones that passed within 100 km of a coral reef in the Western Indian Ocean region between 1980 and 2025. Colors correspond to the Saffir–Simpson category of the cyclone along its entire track. However, the values of sustained wind speed are extracted from the nearest cyclone position from a coral reef. For this reason, some sustained wind speed values are below the lower threshold of category 1 Saffir–Simpson scale (*i.e.* 119 km.h^{-1}). Derived from IBTrACS data (Knapp *et al.* [118]; see Materials and Methods).

Published regional analyses report pre-1998 hard coral cover of up to 40% [465, 466], suggesting the modelled baseline may underestimate historical cover. Additionally, 59% of the survey data were derived from the Mascarene Islands subregion (subregion 3, 6.9% of regional coral reef extent), while the East African Coast (subregion 1) and Madagascar and Comoros (subregion 4), accounting for 78% of regional reef extent, contributed just 30.1% of the survey data (Table A.1 and Table A.4). Critically, over 90% of sites in the study had only a single year of monitoring data (Figure A.3). The apparent net increase may therefore reflect partial recovery from a more severely degraded state than the modelled baseline demonstrates, and the addition of a large number of new sites in the dataset.

The 1998 global bleaching event, driven by strong positive phases of the El Niño Southern Oscillation and Indian Ocean Dipole, caused severe coral mortality across the region, with published average regional decline from ~40% to 30% hard coral cover, rep-

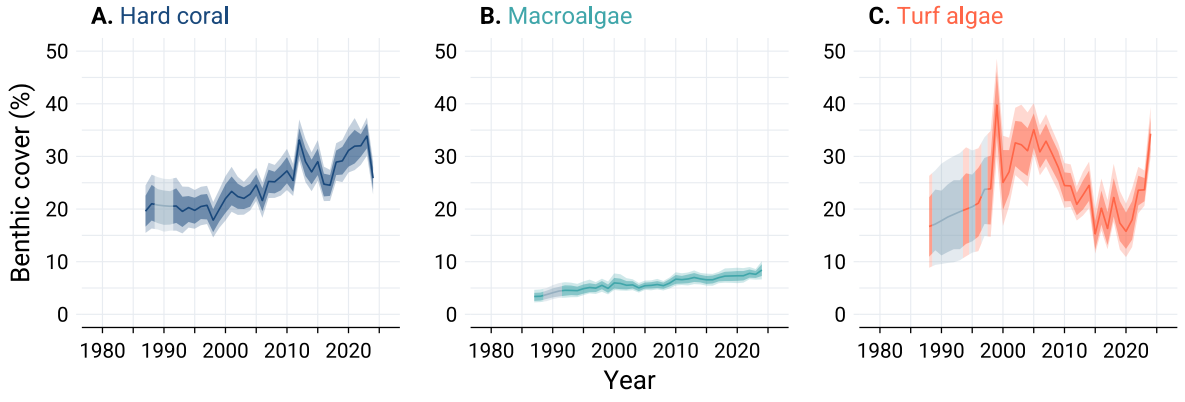
resenting a 25% reduction [466]. The modelled data suggest sustained apparent recovery following this event, with hard coral cover increasing from 22.0% in 2000 to a peak of 33.9% ($CrI_{95\%}$: 30.1 – 37.4%) by 2023. Short-term declines in 2013–2014 were followed by further impacts during the third global bleaching event (2014–2017), which exceeded the severity of prior events in the region [31]. The modelled data did not demonstrate evidence of change between 2015 and 2018, likely reflecting localised impacts in subregions with limited coral reef extent.

The fourth global bleaching event in 2024 resulted in an unprecedented extent of severe heat stress [47, 277], and the largest and most rapid coral decline observed in the modelled data, with regional hard coral cover falling to levels comparable to 1999 by 2024; effectively erasing two decades of hard coral cover recovery. Given the spatial scale of thermal anomalies and the post-2020 increase in surveys (Figure A.1), this decline in hard coral cover likely represented a genuine and severe ecological impact. The severity and speed of this reversal raise serious concerns about the capacity of the region's coral reefs to recover under the accelerating frequency, intensity, and spatial extent of heat stress events, driven by committed climate change.

Macroalgal cover

The modelled data showed a long-term increase in macroalgal cover across the Western Indian Ocean, from 3.4% ($CrI_{95\%}$: 2.3 – 4.7%) in 1987 to 8.4% ($CrI_{95\%}$: 6.6 – 10.1%) by 2024 (Figure 2.10.6B), with strong evidence of a net increase when comparing the reference period (1987–2009) to the most recent period (2020–2024). As with the hard coral cover findings, early macroalgal cover estimates carry some uncertainty due to data sparsity and potential inconsistencies in algal categorisation, though the uncertainty is relatively narrow and there is no strong published evidence suggesting that pre-1998 macroalgal cover differed substantially to the modelled baseline [465, 466].

Prior to 1998, modelled macroalgal cover increased gradually from 3.4% ($CrI_{95\%}$: 2.3 – 4.7%) in 1987 to 5.0% ($CrI_{95\%}$: 3.9 – 6.1%) in 1997.



▲ **Figure 2.10.6.** Modeled temporal trends of benthic cover in the Western Indian Ocean region from 1980 to 2024 for hard coral (A), macroalgae (B), and turf algae (C). The solid line represents the estimated median benthic cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

A modest rise coincided with the 1998 global bleaching event, followed by a return to near-baseline levels before increasing to 6.0% (CrI $_{95\%}$: 4.6 – 7.8%) in 2000. This limited post-bleaching response may reflect the role of grazing pressure, environmental conditions, or competitive interactions with other benthic groups in restricting macroalgal expansion [275].

Macroalgal cover declined between 2000 and 2004, consistent with competitive displacement as corals recolonised substrate, before increasing gradually and persistently, reaching 7.6% (CrI $_{95\%}$: 6.6 – 8.6%) by 2023. Macroalgae are structurally complex, typically slower to establish than turf algae, but are more persistent once established, requiring specific browsing herbivores for removal [467, 468]. The steady upward trajectory may reflect reductions in browsing herbivory, alongside nutrient enrichment from coastal population growth and poorly managed wastewater [469].

A sharp acceleration occurred in 2024, with macroalgal cover expanding to 8.4% (CrI $_{95\%}$: 6.6 – 10.1%), representing the largest single-year increase across the study period, and coinciding with the decline in hard coral cover following the 2024 bleaching event. The scale of hard coral cover loss may have allowed for macroalgal colonisation of bare substrate at a level not previously seen in the modelled data. Whether this represents a temporary condition of the reefs, as appears to have been the case post-1998, or the beginning of a more persistent phase shift, will depend on environmental conditions, the capacity of herbivore populations to constrain macroalgal expansion, and critically, the interval before the next major heat stress event.

Turf algal cover

Turf algal cover showed large apparent fluctuations over time, ranging from 12.7% (CrI $_{95\%}$: 6.5 – 22.4%) in 1987 to 34.3% (CrI $_{95\%}$: 29.8 – 39.4%) in 2024 (Figure 2.10.6C), however, there is no evidence of a change in turf algal cover between the reference period (1987–2009) and the recent period (2020–2024). Unlike macroalgae, turf algae were more responsive to disturbance events [131]. As short, fast-growing filamentous assemblages, turf algae rapidly colonise bare substrate following coral mortality, and remain controlled under adequate grazing pressure, allowing displacement by coral recruits [131]. As turf algae respond quickly to changes in substrate availability [275], short-term fluctuations in the modelled data are likely relatively robust, despite the dominance of single-year sites in the dataset.

Between 1987 and 1997, turf algal cover increased steadily from 12.7% (CrI $_{95\%}$: 6.5 – 22.4%) to 23.8% (CrI $_{95\%}$: 14.9 – 33.6%), remaining at a similar level in 1998. The following year saw the greatest single-year increase in the study period, with cover reaching 39.8% (CrI $_{95\%}$: 29.9 – 48.6%), consistent with a lag between coral mortality and the turf algal colonisation [275]. Turf algal cover dropped to 25.1% (CrI $_{95\%}$: 16.8 – 33.9%) in 2000 before rising again through the early 2000s. Generally, the 2000s represented the period with the highest sustained turf algal cover, reflecting substrate availability following the 1998 coral bleaching event. From 2005 onward, the data demonstrated strong evidence of decline through to the 2010s, suggesting progressive displacement of turf algae, potentially by recovering hard coral, though this should be considered in the context of the underlying dataset. From 2015, turf algal cover became highly variable with strong evidence of an increase between 2015 and 2018. This trend in turf algal cover contrasts with the comparatively stable macroalgal trend over the same time period, suggesting greater turf algal sensitivity to repeated short-term disturbance [275]. In 2023–2024, there was a dramatic increase in turf algal cover from approxi-

mately 23.7% (CrI $_{95\%}$: 20.6 – 27.3%) in 2023, to 34.3% (CrI $_{95\%}$: 29.8 – 39.4%) in 2024, mirroring the post-1998 coral bleaching event.

Due to limited data availability at the subregion scale, inconsistencies in categorisation among monitoring programs, and following consultation with regional experts, trends in turf algae were not considered at the subregion scale.

Regional synthesis

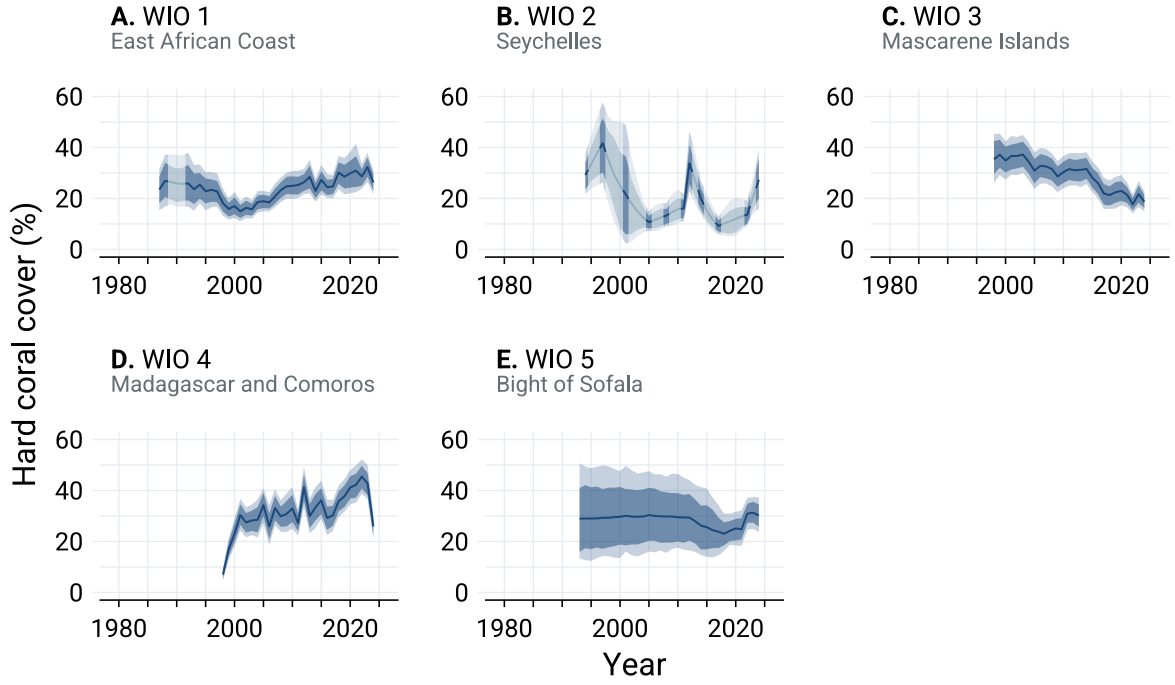
The three benthic indicators — hard coral, macroalgal, and turf algal cover — reveal dynamic and interconnected responses to disturbance, though longer-term trajectories should be interpreted with caution due to the nature of the dataset in the study. The modelled post-1998 recovery of hard coral cover coincided with a decline in turf algal cover, but not in macroalgal cover, suggesting the system recovered from the acute disturbance without halting a gradual shift toward higher macroalgal cover.

The broadly concurrent increases in both hard coral and macroalgal cover through the 2000s and 2010s may point towards shifts in herbivore community structure. Grazing herbivores effectively control turf algae, but browsing species are required to control established macroalgae [467, 470]. Where overfishing has reduced fish biomass, sea urchins may partially compensate for turf control, but cannot substitute for the browsing function, and urchin-dominated grazing regimes are associated with reduced recovery capacity in the Western Indian Ocean [467, 471]. The steady increase in macroalgal cover despite repeated disturbances that affected coral and turf algal cover, may be consistent with a regional decline in browsing herbivory, compounded by bottom-up nutrient enrichment from coastal population growth and limited wastewater management [469], and therefore potentially shifting competitive dynamics in favour of fleshy macroalgae, even where herbivory is maintained [469, 472]. Dense turf algal assemblages can further inhibit coral recruitment, even before macroalgae become dominant [473].

The 2023–2024 period is unprecedented in the modelled dataset. Both turf and macroalgal cover increased sharply, while the region experienced its largest decline in hard coral cover. Previous heat stress events were followed by increases in turf algal cover and inconsistent or moderate responses in macroalgal cover. In contrast, the fourth global bleaching event and the associated hard coral loss appear to have allowed both algal groups to increase drastically. By 2024, the combined algal cover accounted for around half of the benthos, while hard coral cover plummeted to levels comparable to 1999. A striking decline from the peak hard coral cover of 33.9% (CrI $_{95\%}$: 30.1 – 37.4%) in 2023, and potentially an alarming sign of a shift from coral to algal dominance.

STATUS OF CORAL REEFS IN THE SUBREGIONS

Within the Western Indian Ocean region, subregional trends in hard coral cover (Figure 2.10.7) and macroalgal cover (Figure 2.10.8) reflect heterogeneous exposure to disturbances, differing recovery capacities, and localised stressors. The two largest subregions, East African Coast (subregion 1) and Madagascar and Comoros (subregion 4), drove much of the regional signal by reef extent weighting, though their trajectories diverged from smaller subregions.



▲ **Figure 2.10.7.** Modeled temporal trends of hard coral cover in Western Indian Ocean (WIO) subregions from 1980 to 2024. The solid line represents the estimated median hard coral cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

Western Indian Ocean 1 - East African Coast

The modelled data showed hard coral cover along the East African Coast was relatively stable between 22.8–26.9% from 1987 to 1997 (Figure 2.10.7A). However, this is below previously reported cover of 25–50% [474]; some country-level historical baselines document cover of 49–70% at multiple Tanzanian reefs [475], including up to 80% at Tutia Reef in Mafia prior to 1991 [476]. The 1998 global bleaching event, compounded by extreme rainfall causing major river flooding and high sediment and freshwater discharge to nearshore reefs, caused mass coral mortality [466, 474]. Coral recovery following this heat stress event was spatially heterogeneous, persisting despite crown-of-thorns starfish (CoTS) outbreaks across Kenya and Tanzania (2002–2006) [477]. Hard coral cover then peaked in 2023, before declining sharply following the fourth global bleaching event in 2024. Macroalgal cover increased gradually over the study period, with strong evidence of an increase between 1987–2009 and 2020–2024 periods, although cover remained relatively stable through 2023–2024 (Figure 2.10.8A). This pattern is consistent with a delayed response, and with field observations that post-disturbance benthic shifts have favoured corallimorpharians, sponges, and soft corals [462].

Western Indian Ocean 2 - Seychelles

The Seychelles experienced severe hard coral cover loss post-1998, declining from 41.7% ($CrI_{95\%}$: 26.3 – 57.7%) in 1997 to 10.7% ($CrI_{95\%}$: 7.4 – 14.8%) by 2005 (Figure 2.10.7B). Coral recovery was slow, with no evidence of change in hard coral cover between the 2000s and the 2010s. Reefs retaining herbivore biomass showed partial re-

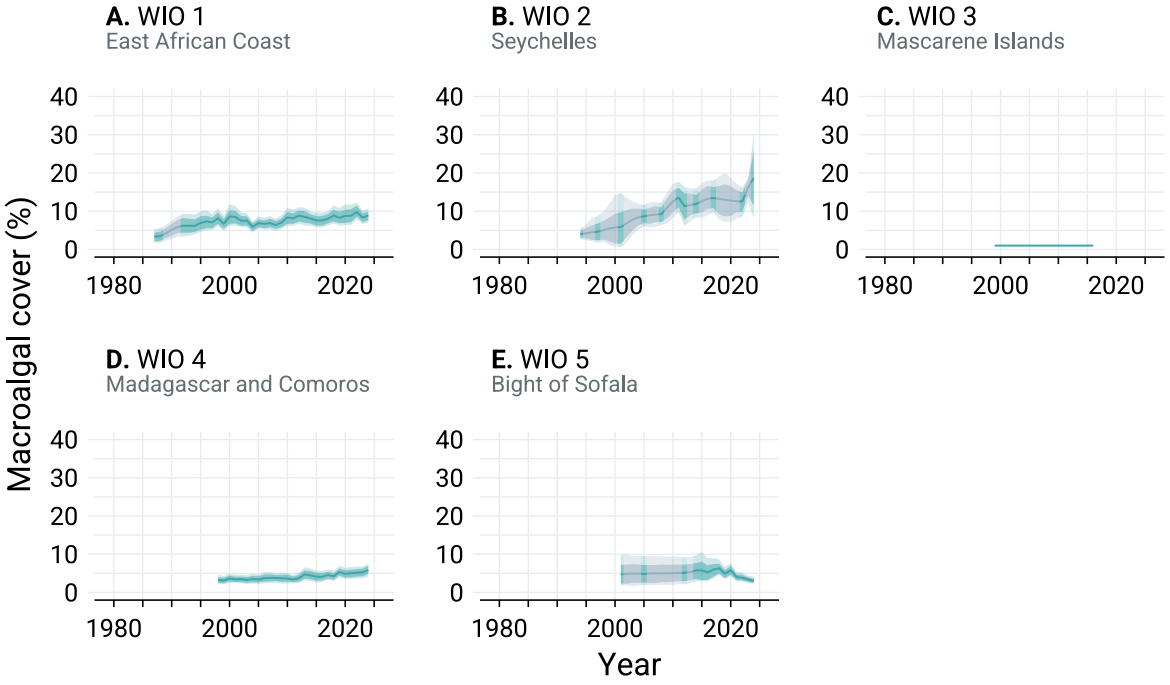
covery by 2012, while others shifted to persistent macroalgal-dominated states [478]. The third global bleaching event caused further coral declines, with hard coral cover dropping to 9.3% (CrI $_{95\%}$: 6.0 – 12.9%) by 2017. Although strong evidence of recovery followed, with faster recovery rates compared to post-1998 disturbance, hard coral cover remained markedly below historical baselines. Macroalgal cover had the largest proportional increase in the Seychelles compared with any other subregion, likely driven by the prolonged post-1998 period of low hard coral cover (Figure 2.10.8B). Nevertheless, trends should be interpreted with caution given data sparsity.

Western Indian Ocean 3 - Mascarene Islands

Mascarene Islands coral reefs are particularly vulnerable to disturbance due to their small size, coastal proximity, and urbanization of upstream watersheds [479]. Monitoring since the late 1990s indicates a steady decline in hard coral cover from 35.4% (CrI $_{95\%}$: 26.1 – 45.3%) in 1998 to 17.7% (CrI $_{95\%}$: 14.2 – 21.5%) by 2022 — the lowest value in the time series — with strong evidence of decline across consecutive decades and no discernible recovery phases (Figure 2.10.7C). Episodes of hard coral cover loss largely coincided with local bleaching events, with the 2025 bleaching preceding Cyclone Garance, compounding impacts [480]. Geographic isolation, upstream of dominant current systems, likely limits coral recruitment from neighboring reefs [445, 481]. While early-successional coral communities can recover rapidly following climate-driven bleaching, chronic water quality degradation linked to coastal development, has a marked impact on reef assemblages. In these conditions, benthic algal cover increases, hard coral cover decreases, and reefs shift from *Acropora*-dominated communities to more opportunistic, stress-tolerant genera [479]. Macroalgal cover remained low throughout the study period, potentially due to strong hydrodynamic conditions limiting the establishment of erect algae on outer reef slopes (Figure 2.10.8C). In addition, large macroalgae have historically been uncommon in coral reef habitats of the Mascarene Islands, which may partly explain these patterns.

Western Indian Ocean 4 - Madagascar and Comoros

Across the Madagascar and Comoros subregion, coral reef trajectories showed pronounced temporal variability in hard coral cover, with periods of decline following disturbances and partial, spatially heterogeneous recovery (Figure 2.10.7D). From 7.0% (CrI $_{95\%}$: 4.7 – 9.6%) hard coral cover in 1998, coral recovered strongly to 45.5% (CrI $_{95\%}$: 38.7 – 52.2%) by 2022, with strong evidence of sustained increases across consecutive decades. The magnitude of the modelled recovery exceeds published estimates for the subregion and likely reflects, in part, expansion of monitoring coverage. Locally constrained recovery by cyclone-driven river discharge, generating sediment and turbidity on nearshore reefs along western Madagascar (notably cyclones Cynthia, Geralda, and Leon-Eline), while cyclones Kenneth (2019) and Chido/Dikeledi (2024/2025) caused localised damage in the Comoros archipelago, compounding heat stress impacts [482, 483]. In Comoros, hard coral cover in Ngazidja and Mwali fell from 39% in 2010 to 10–20% in 2016 [484]. The 2024–2025 bleaching event produced the sharpest subregional reversal of the broadly positive trajectory, with hard coral cover falling to 25.9% (CrI $_{95\%}$: 20.7 – 31.2%) by 2024. Macroalgal cover increased gradually but almost continuously over the study period, rising from 3.2% (CrI $_{95\%}$: 2.2 – 4.6%) in 1998 to 5.9% (CrI $_{95\%}$: 4.1 – 7.5%) in 2024 (Figure 2.10.8D).



▲ **Figure 2.10.8.** Modeled temporal trends of macroalgal cover in Western Indian Ocean (WIO) subregions from 1980 to 2024. The solid line represents the estimated median macroalgal cover while darker and lighter ribbons represent 80% and 95% credible intervals, respectively. Grey areas represent periods during which no field data were available.

Western Indian Ocean 5 - Bight of Sofala and Delagoa

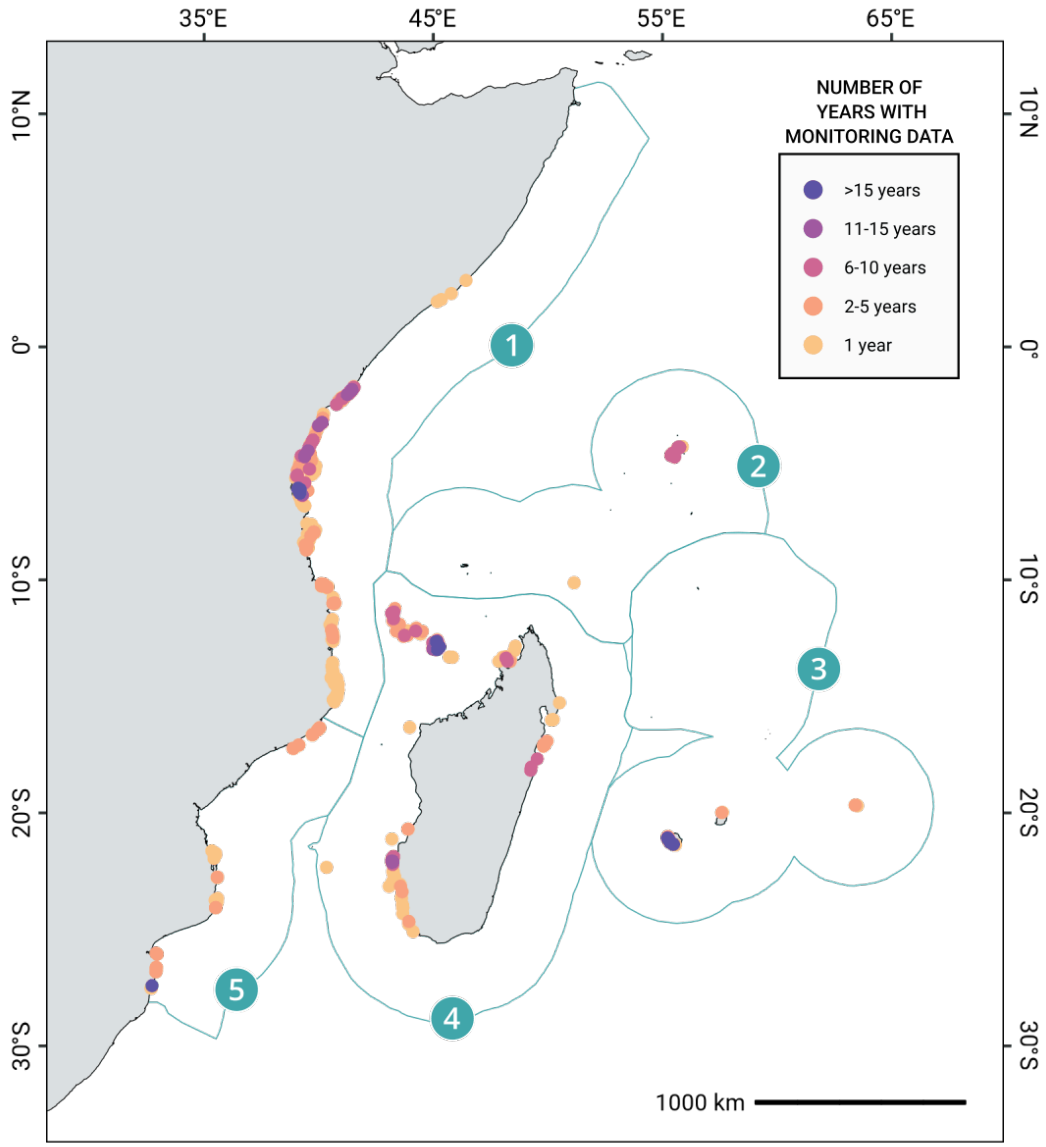
Hard coral cover in the Bight of Sofala and Delagoa subregion demonstrated long-term stability, with no evidence of change from 1993–2009 and 2020–2024 periods (Figure 2.10.7E). From the 1993 baseline of 29.0% (CrI $_{95\%}$: 13.3 – 50.7%), hard coral cover increased before declining from 2011 to 2018, coinciding with the 2010 and 2016 heat stress events [466], though this may be potentially confounded by the establishment of new monitoring sites from 2013. Thereafter, hard coral cover recovered, peaking at 31.3% (CrI $_{95\%}$: 25.0 – 37.4%) in 2023. Over the study period, hard coral cover varied by around 8 percentage points, with current levels about 2% higher than in the early 1990s. Cyclone-driven flooding from Eline (2000) and Idai (2019) discharged freshwater and sediment onto nearshore reefs, contributing to localised coral mortality [463] and likely suppressing coral recruitment. Macroalgal cover showed an inverse relationship with hard coral cover, increasing from 4.9% (CrI $_{95\%}$: 1.3 – 10.8%) in 1993 to 6.3% (CrI $_{95\%}$: 4.6 – 8.6%) in 2018, before declining to a low of 3.0% (CrI $_{95\%}$: 2.3 – 4.1%) in 2024 as coral recovered. This is the only Western Indian Ocean subregion that demonstrated recent macroalgal decline, with strong evidence of decline between 2018 and 2024, and cover remaining consistently below the 2018 peak (Figure 2.10.8E).

DESCRIPTION OF MONITORING DATA

Temporal trends in the percentage cover of benthic categories were estimated using data from 48 datasets encompassing 3,753 monitoring sites. These data were collected from 5,633 surveys conducted between 1987 and 2025 (Table A.4). Although

the Mascarene Islands subregion (subregion 3) accounts for 6.9% of the region's coral reef extent, 59.3% of surveys included in the study were conducted in this subregion. Conversely, only 30.1% of surveys were conducted in the East African Coast (subregion 1) and Madagascar and Comoros (subregion 4) subregions, which together account for 78% of the region's coral reef extent (Table A.4). Critically, 92% (*n* = 3,436) of sites had only a single year with monitoring data, with just 3% (*n* = 123) of sites with monitoring data for 6 years or more (Figure A.3).

The regional dataset is dominated by single year survey contributions rather than long-term, repeated measurements, leading to both a spatial and temporal imbalance in the underlying dataset.



▲ **Figure 2.10.9.** Spatio-temporal distribution of benthic cover monitoring sites across the Western Indian Ocean region and respective subregions. Light blue polygons and numbers represent subregions. Subregion 1 = East African Coast, Subregion 2 = Seychelles, Subregion 3 = Mascarene Islands, Subregion 4 = Madagascar and Comoros, Subregion 5 = Bight of Sofala. The scale bar provides distance at the equator and may not accurately represent distance over the entire latitudinal range of the region.

This may reflect the realities of coral reef monitoring programmes with limited funding and differing local objectives, resulting in a mismatch with the objectives of global reporting, which depend on long-term, spatially and temporally representative datasets. Addressing this gap will require sustained, targeted investment in regional scale monitoring to complement local programmes.

CONCLUSIONS

The outlook for recovery is uncertain. The macroalgal baseline is now substantially higher than it was following the 1998 bleaching event, meaning corals must recover against the backdrop of a stronger, more persistent competitor. The capacity of herbivore populations to constrain algal expansion is another key uncertainty. Regional data suggest a shift toward the dominance of smaller-bodied herbivores and detritivores in fish assemblages, with overfishing of larger functional groups, including scrapers and browsers, further constraining algal control and coral recruitment [467, 468, 471]. Most critically, the intervals between major heat stress events have shortened. The post-1998 recovery benefited from a relatively long period of reduced heat stress, a respite that may not be repeated following 2024. The IUCN Red List of Ecosystems assessment for Western Indian Ocean coral reefs projected that heat stress conditions historically associated with mass bleaching will exceed two major events per decade across much of the region within the coming decades [55, 99], a frequency at which coral recovery becomes severely constrained. Global analyses further suggest that current local-scale thermal refugia will largely disappear at 1.5 °C of warming [485]. In the context of elevated algal baselines, reduced recovery windows, and intensifying heat stress, the question of how the region's coral reefs will respond remains to be seen in the coming years.

DATA CONTRIBUTORS

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Access to online supplementary materials

The modelled benthic cover data, detailed outputs from the data analyses, and the CRediT author contribution statement are available on Zenodo.

CASE STUDY 9

From local to global: how community-managed areas are paving the way for other effective area-based conservation measures

Authors

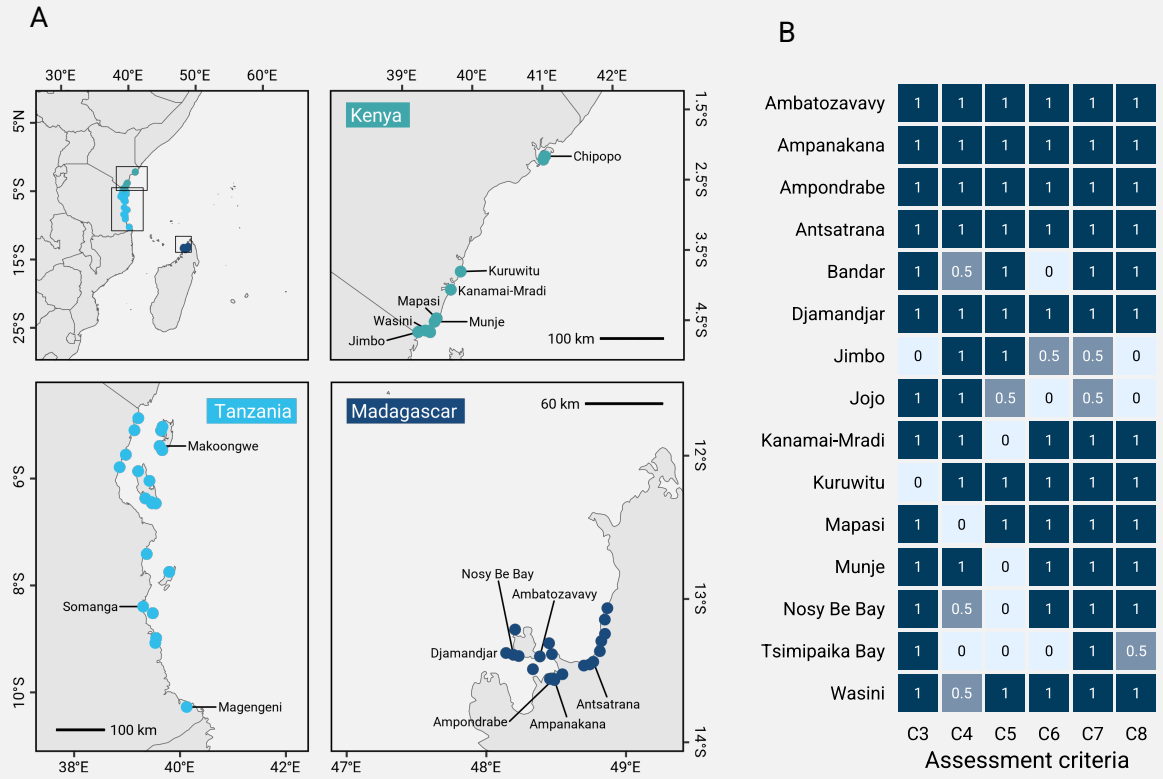
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Coastal communities across the Western Indian Ocean (WIO), like millions of people globally, depend on coral reefs for food, livelihoods, coastal protection, and cultural value [99]. Yet, these reefs continue to face intensifying pressures from overfishing, destructive gear, and climate change [486]. In the WIO, Locally Managed Marine Areas (LMMAs) have emerged as community-led initiatives to safeguard these critical coastal ecosystems [487]. Defined as nearshore waters and coastal resources managed at the local level by communities, landowning groups, partner organizations, and/or collaborative government representatives [447], LMMAs aim to maintain the ecological, social, and economic integrity of the marine ecosystem. They are typically led by community institutions to ensure sustainable resource use [488].

Coral reefs are the dominant ecosystem within most WIO LMMAs, and their condition is closely tied to the food security and livelihoods these areas are designed to protect. Yet reefs across the region are now vulnerable to ecosystem collapse under the combined pressure of ocean warming, overfishing, and destructive gear [99]. Effective area-based management is therefore central to securing positive reef outcomes. In the WIO, community-led fisheries closures and gear restrictions have been shown to raise reef fish biomass, average fish size, and catch value, particularly where compliance and enforcement are strong [489]. By embedding such measures in locally legitimate governance, LMMAs offer a practical mechanism for managing reefs for both biodiversity and the people who depend on them, and recognition as OECMs can strengthen and formalise that protection at national and global scales.

As countries work toward Target 3, known as "30x30", of the Kunming-Montreal Global Biodiversity Framework (GBF), LMMAs are among the potential area-based conservation approaches that align with the Other Effective Area-based Conservation Measures (OECMs) framework under the Convention on Biological Diversity (CBD), offering a ready pathway for formal recognition [490]. OECMs are geographically defined areas outside formal protected areas that are governed and managed to deliver sustained *in situ* biodiversity outcomes alongside ecosystem services and cultural values [491]. Recognising eligible LMMAs as OECMs would globally formalise community stewardship, unlock financing for community-led conservation, and support countries in reporting progress toward the GBF targets [492].



▲ **Figure C9.1.** Map of the Western Indian Ocean region highlighting LMMAs in Kenya, Tanzania, and Madagascar, which were assessed against the IUCN OECM criteria (A). Potential OECM candidates with OECM assessment criteria (3–8) across selected coastal sites. Fully compliant sites show complete adherence across all criteria (B). In subfigure B, the numbers represent the level of compliance, where 1 corresponds to the highest level of compliance.

Assessing WIO LMMAs for OECM potential

In a regional assessment, 69 LMMAs across five WIO countries were reviewed against the IUCN OECM site assessment criteria (Figure C9.1A). The assessment applied Criteria 3–8 to evaluate community perceptions of ecological integrity, governance mechanisms, equity, and community benefits [493], with an additional category (Criterion 9) assessing community benefits and values from LMMAs. Designed as an indicative overview, the assessment aimed to identify promising OECM candidates and highlight challenges and opportunities for recognition under the GBF Target 3.

Of the LMMAs assessed, most were relatively small, but generally well-defined: 65% covered less than 5 km^2, 63% had physical markers (*e.g.* buoys), and 64% were bounded by natural features. However, only 28% had boundaries mapped and publicly accessible in digital form, and 32% reported boundary disputes, underscoring the need for participatory spatial planning and clear boundary definition processes, consistent with Target 1 of the GBF.

Governance structures were in place for all sites (Criterion 5), and 95% reported achieving, or expecting to achieve, in situ conservation outcomes (Criterion 6). Perceptions of sustainability were high (83%) with respondents citing governance capacity, community awareness, and observed conservation benefits as key drivers (Criterion 7). Equity indicators were similarly positive, with 88% of respondents affirming that recognition, procedural inclusion, and fair benefit distribution are addressed (Criterion 8). The high level of community engagement in LMMA

governance reflects the growing legitimacy and social capital of locally managed conservation initiatives in the WIO region [488, 494].

On biodiversity significance, respondents linked effectiveness to site size and configuration: 58% cited size as critical for encompassing diverse habitats, 27% valued sites for their role in protecting key breeding areas, and 14% emphasised that proximity to adjacent communities enhanced functionality and relevance. Coral reefs are the primary ecosystem encompassed by WIO LMMAs, and typically measures of biodiversity use standard coral reef monitoring indicators. Thus LMMAs and OECMs are an effective tool for managing and protecting coral reefs, notably for fish diversity through local fishery regulations [489].

Despite strong results on governance, equity, and perceived sustainability outcomes, evidence for biodiversity values (Criterion 4) was largely qualitative, with limited quantitative species or habitat monitoring data. Long-term security (Criterion 9) was also variable, with many sites lacking formal legal recognition or policy integration to guarantee permanence. Overall, only 15% of LMMAs fully met the OECM criteria, 51% partially met them, and 34% did not meet them (Figure C9.1B).

From local stewardship to OECM recognition

The assessment shows that LMMAs deliver multidimensional community benefits, including improved fish stocks, enhanced livelihoods, and increased community empowerment, while contributing to broader biodiversity targets such as Targets 2 (restoration), 3 (area-based conservation), and 10 (sustainable use) of the GBF. However, scaling recognition of LMMAs and integrating community-led conservation into policy are constrained by limited quantitative documentation on biodiversity and livelihood outcomes [487]. Closing these gaps through stronger institutional capacity, clearer legal frameworks, robust monitoring systems, and better spatial documentation will be critical to elevating LMMAs from promising local initiatives to recognised OECMs that contribute meaningfully to national and global biodiversity goals. The structured, criteria-based approach applied here, screening community-managed areas against the IUCN OECM criteria and using standard reef monitoring indicators to evidence biodiversity outcomes, is readily transferable to other GCRMN regions where community-based management is widespread. Applying a common assessment framework across regions would allow comparable identification of OECM candidates worldwide and help translate locally led reef stewardship into measurable progress toward GBF Target 3.

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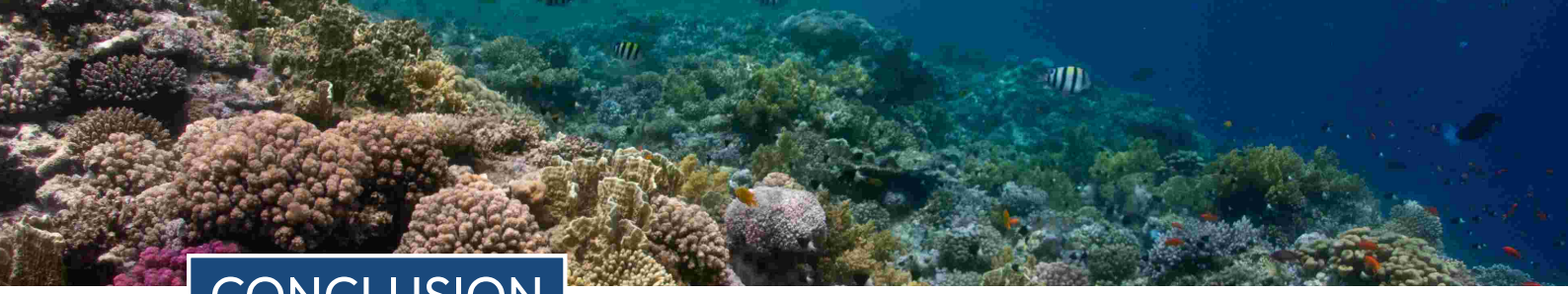
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CONCLUSION

THE STATUS OF CORAL REEFS OF THE WORLD

The global assessment of coral reef status reveals a clear and concerning long-term decline in coral reef condition over the past four decades (1980–2024). Global hard coral cover has decreased substantially, with an estimated average loss of ~9.5% (ranging from 4 to 14.5%) relative to the reference period (1980–2009), and a pronounced stepwise decline since 2010. This pattern closely tracks the increasing frequency, intensity, and spatial extent of marine heatwaves driven by anthropogenic climate change, which have resulted in four global bleaching events since 1998. In addition to more frequent and intense global bleaching events, coral reefs at regional and local scales have been and continue to be severely impacted by environmental and anthropogenic pressures, leading to a steeper decline in hard coral cover (see Part 2 for details).

Each global bleaching event has caused significant losses in hard coral cover, while shortening recovery windows between events limit the capacity of coral reefs to rebound. Together, these dynamics indicate a systemic, global-scale degradation of coral reef ecosystems, rather than localised impacts or cyclical patterns of decline and recovery.

In parallel, macroalgal cover has steadily increased globally, rising by approximately 44% since the reference period (ranging from 28% to 59%), particularly following major coral mortality events. Although global average macroalgal cover remains relatively low, this trend is concerning because even moderate increases at regional and local scales can suppress coral recruitment and recovery. The combined signal of declining hard coral cover and increasing macroalgal cover indicates a gradual compromise in recovery potential and a growing risk of a continued downward trend in hard coral cover and a shift toward less desirable ecosystem states.

Importantly, global averages obscure substantial regional variability. While several regions, including the Caribbean and the ROPME Sea Area, now exhibit critically low hard coral cover levels that may constrain future recovery, other regions have remained relatively stable or show limited signs of recovery. These contrasting trajectories reflect how exposure to global and local pressures, ecological context, and management effectiveness vary across regions and sites.

THE FUTURE OF THE WORLD'S CORAL REEFS

Climate-driven marine heatwaves, compounded by local human and environmental pressures, are pushing coral reefs toward a sustained global decline, with shrinking recovery windows, making continued degradation likely without timely climate mitigation and local management action.

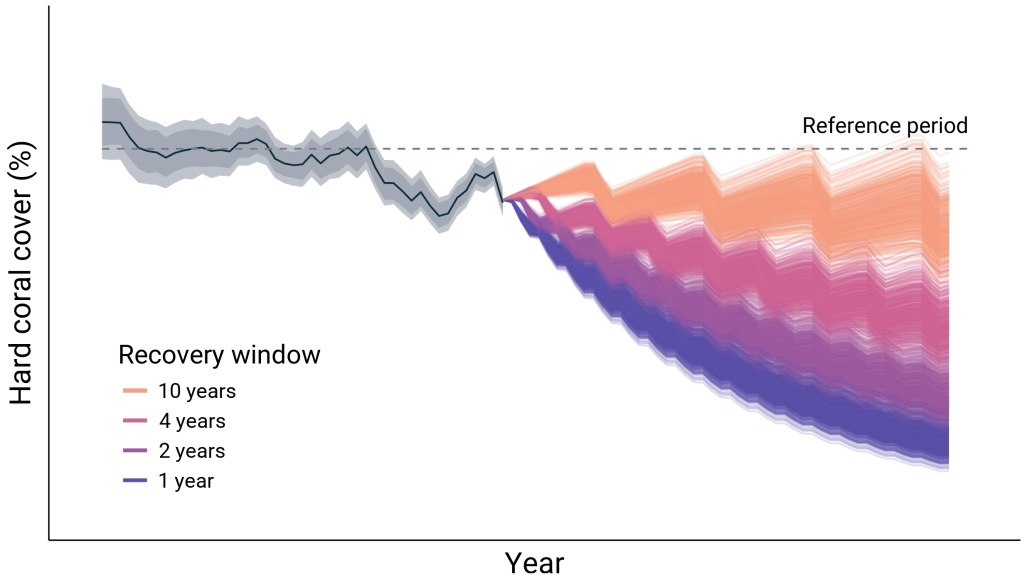
Over the last four decades, coral reefs have exhibited a concerning decline, primarily driven by episodic marine heatwaves intensified by climate change, and compounded by localised pressures from environmental (*e.g.* tropical cyclones, crown-of-thorns starfish) and anthropogenic (*e.g.* eutrophication, overfishing, coastal development) sources.

Without significant reductions in greenhouse gas emissions, warming will surpass 2 °C [495] and marine heatwaves will persist and intensify, driving further coral reef decline. The years 2023 and 2024 were the warmest on record, with sea surface temperatures overshooting 1.5 °C above pre-industrial levels [496, 497]. At this level, 87% of reefs faced unprecedented heat stress, with over 40% of reefs experiencing mortality-levels of heat stress in 2024 (DHW $>$ 8 °C-weeks), leading to the largest annual decline in hard coral cover on record (Figure 1.4C).

The concerning trajectory of the ratcheting down of global hard coral cover suggests that a rebound to reference levels is unlikely under future climate scenarios. Furthermore, a continued decline in hard coral cover is expected with shorter recovery periods. To illustrate the importance of the length of recovery periods (*i.e.* the number of years between major disturbances that cause loss of hard coral cover), a simple forecast model simulates future trajectories under four different scenarios (major disturbance every 1, 2, 4 or 10 years, see Materials and Methods for details, Figure 3.1).

In recent years, coral reefs have seen a significant decrease in recovery intervals, shifting from about one major disturbance per decade to every 5–6 years after 2010. Simulations projecting future trends with varying recovery times suggest that an increase in the frequency of major disturbances — exceeding decadal events (*i.e.* more than one event every 10 years) — will lead to a continual decline in global hard coral cover (Figure 3.1). When the frequency of major disturbances increases beyond previously observed frequencies (*e.g.* every 1–2 years), the decline in hard coral cover is expected to be more severe than currently documented in this report. Importantly, these illustrative simulations do not account for the complex, non-linear mechanisms that underpin coral reef dynamics. For example, as hard coral cover declines, reproductive output and fertilisation success may decline non-linearly, potentially creating an Allee effect that constrains recovery and disproportionately accentuates patterns of decline [97, 498]. Therefore, the simulations from this model are rather conservative in their predictions of future trajectories.

Evidence for a concerning future outlook for coral reefs is also provided by the International Union for Conservation of Nature (IUCN) Red List of Ecosystems assessments at regional (*e.g.* Caribbean [499] and the Western Indian Ocean [99]), and local (*e.g.* Mesoamerican Barrier Reef [500]) scales, which consistently suggest coral reefs are vulnerable to collapse across geographies.



▲ **Figure 3.1.** Possible future trajectories of global-average hard coral cover, based on historical trends between 1980 and 2024 (grey line and ribbons; see Part 1 for details), under future scenarios that vary the length of recovery windows (*i.e.* the number of years between major disturbances). Each coloured line represents an individual simulation under a different scenario. The dashed grey line represents the historical reference, estimated as the median hard coral cover during the reference period (1980–2009).

These results are also supported by a comprehensive, albeit localised, modelling study to forecast the trajectories of hard coral cover in the Great Barrier Reef, which highlights that recovery is only possible if warming remains below 2 °C, allowing thermal adaptation to keep pace [501].

Future coral reef trajectories are strongly influenced by the complexity of these ecosystems and the heterogeneity in pressures and ecological responses across scales and geographies. Beyond the major indicators highlighted in this report (*i.e.* hard coral and macroalgal cover), coral reefs are likely to demonstrate changes across a myriad of ecological dimensions (*e.g.* recovery rates, compositional change, thermal tolerance adaptation, *etc.*) and with diverse implications for their resilience and the services offered to local human communities [502, 503]. However, while changes beyond high-level indicators have been demonstrated at local scales [504, 505], the lack of consistency and taxonomic depth in monitoring severely constrains regional or global assessments of the severity and magnitude of changes beyond hard coral cover [147].

IMPLICATIONS FOR SCIENCE, MANAGEMENT, AND POLICY

Securing a sustainable future for coral reefs requires integrated mitigation of global and local stressors, and strong monitoring and reporting systems, because environmental uncertainty and social–political challenges demand adaptive, multi-scale policy and management responses. These same challenges also create opportunities for transformative action, innovation and collaboration.

Securing a future in which coral reefs retain their biodiversity, ecological functions, and ecosystem services under changing climate conditions is a political, ecological, and social challenge. Uncertainty in climate trajectories, societal insecurity and change, variability in ecological responses across geographies, and the cumulative impacts of multiple stressors acting at different scales mean a range of policy and management responses must be considered to adaptively address the challenges facing coral reefs [506].

This also means understanding how best to manage ecological, social, and political dynamics at a variety of spatial and temporal scales, and embedding solutions within broader agendas for sustainable development, livelihoods, and climate resilience, is now integral to addressing the escalating problems that confront coral reefs [506].

Globally, coral reefs are ecosystems on the front lines of climate change. However, coral reefs have not crossed a point of no return, and global coral reef collapse is not inevitable. The results in this report suggest that coral reefs still retain recovery potential, but long-term trends are concerning without accelerated mitigation of stressors. Keeping warming well below 2 °C is vital to substantially improve the prospects for reef resilience and recovery [495]. Integrating climate and biodiversity policies is essential for positive biodiversity conservation outcomes under the Global Biodiversity Framework and for maximising the effectiveness and co-benefits of conservation actions.

While climate mitigation policies and commitments are important for achieving the required change in a timely manner [1, 507], coral reefs will continue to be impacted even under the most ambitious climate mitigation scenarios, as inherent delays in the Earth's climate system mean that future warming will continue for some time [508]. Nevertheless, evidence shows that coral reefs subjected to lower local pressures are better able to resist, survive, and recover from climate impacts [88]. Consequently, addressing local and global stressors in tandem is critical to generate positive ecological and societal outcomes from coral reef conservation. Management interventions such as effectively enforced marine protected areas, sustainable fisheries management (including herbivorous fish protection), integrated coastal zone management, improved land-sea planning, and reductions in land-based sources of pollution are therefore essential complements to climate mitigation, and represent proven, readily deployable management tools for supporting reef resilience.

Within this context, identifying climate refugia — reef systems less exposed to the effects of climate change — has emerged as a key conservation priority [88, 509–511]. These refugia offer strategic opportunities for targeted local management approaches to protect less-vulnerable, well-connected coral reefs that may serve as source populations, helping to repopulate degraded areas while climate mitigation actions take effect [510, 511]. Protecting and investing in these areas can buy crucial time for coral reefs globally. Critically, realis-

ing the potential of climate refugia depends heavily on substantially reducing local human pressures through strong governance, compliance, and community engagement [88]. In some contexts, carefully applied, active interventions such as reef restoration, species translocation, or assisted recovery may further support resilience when integrated with effective management and emissions-reduction strategies.

As innovation in conservation practices, including at-scale restoration and adaptation, advances in addressing the ecological, social, and political challenges of managing nature under climate change, long-term monitoring and adequate reporting are cornerstones of adaptive conservation efforts. Well-designed, implemented, and sustained monitoring programs, together with integrated regional and global reporting frameworks underpinned by robust analytical mechanisms, such as fundamental and applied science, play critical roles in the conservation and management of coral reefs. These systems provide the evidence base needed to learn from successes, adjust course where necessary, scale what works, and demonstrate that progress is both possible and measurable, bolstering confidence in sustained investment and action.

RECOMMENDATIONS FOR CONTINUAL IMPROVEMENT IN LONG-TERM CORAL REEF MONITORING AND REPORTING

Standardised coral reef monitoring and reporting efforts have significantly improved the collective understanding of coral reef trends, but further expansion — especially in high-priority, vulnerable regions and using a broader set of indicators — is essential to guide more effective conservation.

Long-term monitoring and reporting are essential for tracking progress, identifying emerging risks, demonstrating success, and enabling adaptive management that improves policy and practice. Sustaining long-term monitoring requires financial incentives, capacity building, and institutional support, particularly in developing countries [512]. Securing the future of coral reefs demands that these commitments are not made in isolation, but that timely action is coordinated across climate, biodiversity, and monitoring agendas.

Together with the previous GCRMN report on the *Status of Coral Reefs of the World: 2020* [30], this report marks the second time the GCRMN has produced a quantitative assessment of the status and trends of coral reefs worldwide and regionally (https://gcrmn.net/resources/). While labour-intensive [513], the GCRMN approach has yielded step-changes in understanding global coral reef trajectories, as well as improving: (1) collaboration across regional GCRMN nodes; (2) transparency in reporting by providing science-based evidence; (3) acknowledgment of data contributions; (4) innovation in data analysis to best represent trends across geographies; (5) visibility of the value of monitoring efforts; and (6) data standardisation and integration [513, 514]. Further to the value of integrated monitoring and reporting, innovations in data platforms, standardised observation frameworks, and monitoring technologies have significantly improved our ability to track coral reef status and trends globally [147, 514]. A demonstration of these achievements is provided by comparing data contributions between the two most recent reports (2020 and 2025), which shows a threefold increase in available data and more accurate estimates of

long-term trends in this report (Box 1).

Building on the lessons learned from this report, the following priorities emerge as recommendations for improving global coral reef monitoring efforts: (1) increase resource allocation to coral reef monitoring, in particular for developing nations; (2) increase efforts to build temporal and spatial coverage of monitoring, especially in regions that represent a large proportion of coral reefs and remain understudied; (3) guide the standardisation of monitoring practices; (4) improve data discoverability and interoperability while promoting adequate recognition of data ownership under the FAIR data principles [515]; (5) increase taxonomic resolution and diversity of indicators, as per the Essential Ocean Variables [516], to better understand changes in coral reefs beyond hard coral cover; and (6) integrate ecological and socio-economic monitoring indicators to better align the interpretation of the results in a whole-system framework.

BOX 2

The fourth global coral bleaching event

Authors

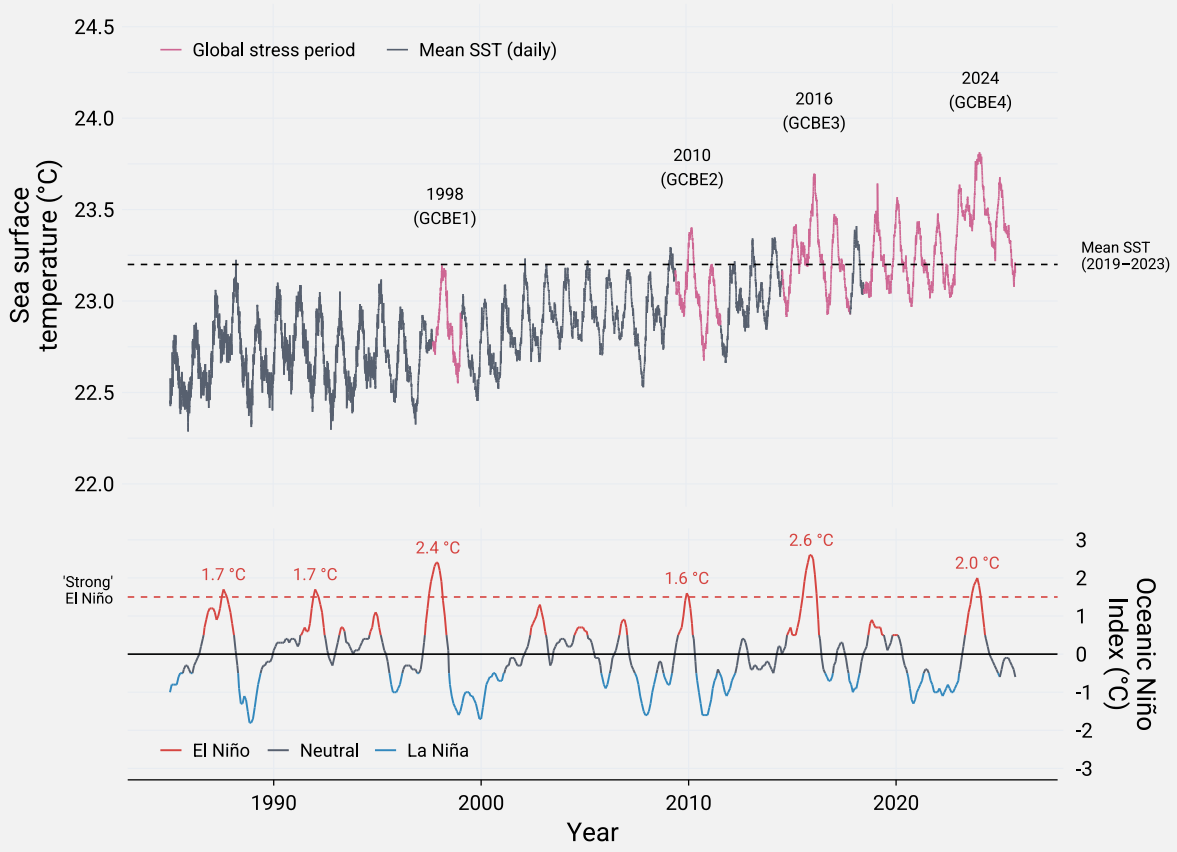
Blake L. **Spady**[1,2], Kristen T. **Brown**[1,2], Erick F. **Geiger**[1,2], Graham **Kolodziej**[1,2], Gang **Liu**[2], Morgan **Pomeroy**[1,2], Derek P. **Manzello**[2], Nicole **Ryan**[3], James **Gilmour**[3,4], Claire L. **Ross**[4,5], Richard D. **Evans**[4,5], Swaleh Ali **Aboud**[6], Chico **Birrell**[6], Mohammed **Dosoky**[6], Khaled **El-Haddad**[6], Mohammed **Elsayed**[6], Patricia **Kramer**[7,8], Lisa **Carne**[9], Tyler B. **Smith**[10], Lauren K. **Olinger**[10], Israel A. **Muñiz-Castillo**[11], Aldo **Croquer**[12]

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Introduction

The first confirmation of multiple, regional-scale mass coral bleaching events occurring simultaneously or subsequently throughout the Atlantic, Indian, and Pacific Oceans during a single continuous period was in 1998, marking the first documented global-scale coral bleaching event (GCBE) [42, 43]. Additional GCBEs were documented in 2010 (GCBE2) and 2014–2017 (GCBE3), each more widespread, enduring, and consequential than the last. Historically, there have been strong correlations between the timing of GCBEs and El Niño phases of the El Niño Southern Oscillation (ENSO) [29]. Since 1998, every El Niño event with a peak Oceanic Niño Index value $\geq$ 1.5˚C (considered a "strong" El Niño) [517] has been associated with declared global coral bleaching events. However, as general surface ocean temperatures increase, the role of periodic warming mechanisms such as strong El Niño events may no longer be a prerequisite for global-scale heat stress accumulations ([47]; Figure B2.1).

In April 2024, the U.S. National Oceanic and Atmospheric Administration (NOAA) announced the presence of the fourth global event (GCBE4), starting in February 2023, resulting from synoptic-scale marine heatwaves in all three tropical ocean basins. While bleaching confirmations are necessary to confirm and declare a GCBE, satellite-derived heat stress metrics can be used to describe and increase our understanding of these events. Here, using methods and results from [47], we describe trends in global satellite-derived heat stress patterns since the last GCRMN report [30], including during GCBE4 (2023–2025). We also compare this period to previous heat stress accumulation trends and global bleaching events. Analyses and comparisons are primarily calculated using the Global Coral Heat Stress Index, which identifies periods of global-scale coral heat stress using conditions achieved during the 1998 event (GCBE1) as a baseline [47].

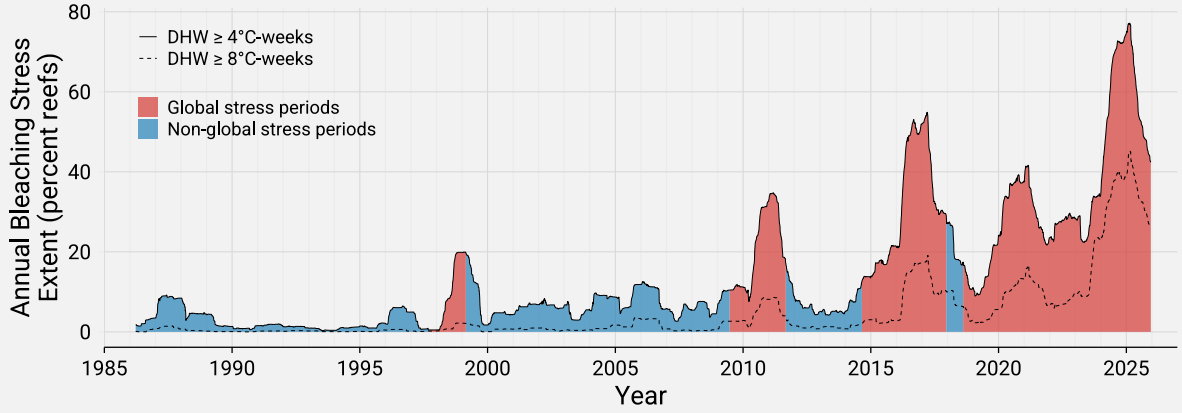


▲ **Figure B2.1.** Trends in global sea surface temperature (SST) and El Niño Southern Oscillation (ENSO) strength from 1985–2025. Daily mean global (all ocean pixels from 45˚N to 45˚S) SST is shown in the upper time-series (left y-axis). Intervals in which the SST time-series is purple indicate global stress periods. The lower time-series (right y-axis) shows the ENSO strength based on the Oceanic Niño Index (ONI) within the Niño 3.4 region (3-month running mean; NOAA ERSST.v5). The ENSO time-series is coloured blue during La Niña periods, grey during ENSO neutral periods, and red during El Niño periods. Peak SSTs corresponding to both a confirmed global-scale coral bleaching event (GCBE) and a strong El Niño (ONI $\geq$ 1.5˚C; dotted red line; peak ONI values labelled in red) and are labelled with the year and corresponding GCBE number. The dotted black line indicates the recent baseline surface ocean temperature, calculated as the mean global SST from June 2019 to March 2023 (a period consisting of ENSO neutral or La Niña phases). Figure reproduced with permission from Spady *et al.* [47].

The first three periods of global stress identified by the Global Coral Heat Stress Index closely align with the initial three GCBEs. Following the global stress period associated with GCBE3 (August 2014 – December 2017), there was a roughly eight-month period of comparative relief before the next global heat stress period initiated in August 2018. This state of global stress has remained consistently present for over seven years ([47]; Figure B2.2). From August 2018 to December 2025, ~87% of reef areas (5 km satellite pixels containing coral reefs) globally experienced Degree Heating Week (DHW) accumulations of $\geq$ 4˚C-weeks (*i.e.* bleaching-level heat stress) at least once. Unlike the three previous global stress periods, which were driven in part by strong El Niño events, the ongoing fourth global stress period was dominated by La Niña and neutral phases of ENSO up to early 2023. The mean global (45˚N – 45˚S) sea surface temperature (SST) during this interval is on par with that of the peak daily mean global SST during the 1998 El Niño, driving relentless global heat stress despite lacking the periodic warming mechanism that El Niño typically provokes ([47]; Figure B2.1). This global stress period driven by shifted-baseline surface ocean temperatures was then met with the development of

the strong 2023/2024 El Niño event. Consequently, the most extensive and severe global heat stress accumulations on record emerged, leading to the unprecedented fourth global-scale coral bleaching event.

Spady *et al.* [47] demonstrates that during GCBE4, 77% of reefs globally experienced bleaching-level heat stress within a 365-day period (February 2024 to February 2025), greater than the previous record high of 55% achieved during GCBE3, and almost four times that reached during GCBE1 (20%). The annual extent of bleaching-level heat stress was also record-setting in each of the individual tropical ocean basins. In the Atlantic Ocean, 99.9% of reef areas were affected from June 2023 to June 2024. While the extent of bleaching-level heat stress during GCBE4 has been unprecedented, the average intensity of heat stress accumulations is perhaps even more noteworthy. From February 2023 to September 2025, the median of maximum DHWs among all reef pixels was 8.2˚C-weeks, with more than half of the reef areas globally having experienced heat stress generally considered capable of eliciting mortality. The median of maximum DHWs during the first three GCBEs was 1.4, 2.7, and 5.5˚C-weeks respectively. Further, extreme heat stress accumulations of $\geq$ 20˚C-weeks, expected to result in near-complete, reef-wide mortality was measured across significant areas of several key reef systems including Western Australia, the Red Sea, and in more than 25% of all reef areas within the Atlantic Ocean basin.



▲ **Figure B2.2.** Daily values (25 March 1986 - 17 December 2025) of the annual bleaching-level heat stress extent, as calculated by the Global Coral Heat Stress Index, showing the percentage of reef locations globally that had accumulated Degree Heating Weeks (DHW) $\geq$ 4˚C-weeks in the previous 365 days (solid black line). Sections under this line shaded in red depict periods of global stress on coral reefs as defined by the Global Coral Stress Index (see Spady *et al.* [47]). Sections shaded in blue depict periods during which accumulated heat stress did not meet the criteria of a global stress period. The black dashed line shows the percentage of reef locations globally that had accumulated DHW $\geq$ 8˚C-weeks in the previous 365 days. Figure reproduced with permission from Spady *et al.* [47].

Unprecedented mass bleaching on Australia's west coast

Western Australia's (WA) coral reefs are diverse and abundant, ranging from oceanic to fringing reefs spanning thousands of kilometres of coastline. Some WA reefs suffered mass bleaching during the previous Global Coral Bleaching Events (1998, 2010, 2016) and El Niño conditions, whereas others bleached only during La Niña conditions (2011–2013) or have not suffered mass bleaching. However, from August 2024 to June 2025, WA experienced its most widespread (~2,000 km) coral bleaching on record. Heat stress remained persistent for 80–160 days along much of the coastline, with peak Degree Heating Week (DHW) accumulations ranging 11–17˚C-weeks offshore, 8–21˚C-weeks in the far north, and 20–30˚C-weeks in central regions.

At remote oceanic atolls, bleaching and mortality was moderate ($\approx$ 30%) at Ashmore and Scott reefs, but extreme ($>$75%) at the Rowley Shoals. At the inshore Kimberley region, bleaching was widespread and severe ($>$60%) at most intertidal and shallow communities. Most Pilbara reefs experienced extreme ($>$75%) bleaching and mortality, while coral communities at world-heritage Ningaloo Reef recorded the most severe bleaching ($\approx$ 30–75%) on record. Post-bleaching mortality and wider ecosystem impacts will take years to assess, with recovery rates uncertain given predicted increases in ocean temperatures.

Impacts of the fourth global bleaching event on Saudi Red Sea reefs

The Fourth Global Coral Bleaching Event affected the Red Sea, where above-average sea surface temperatures (SST) triggered coral bleaching across multiple regions. A similar event occurred in 2015–2017, but during 2023–2024, bleaching reached the Northern Red Sea in summer 2024, where thermal stress exceeded 28 °C-weeks. Regional thermal stress varied due to local conditions, and the heat stress of 2025 did not cause bleaching, indicating changes were driven by 2023-2024. Despite these extremes, many corals recovered, with thermally sensitive species affected and stress-tolerant corals relatively more abundant, supporting the refugia hypothesis.

Bleaching occurred throughout the Red Sea, with early observations recorded regionally in 2023. SHAMS surveys reported bleaching in the Southern Red Sea from July 2024, and in the Central and Northern Red Sea between August to November 2024. Mortality varied regionally, highest in the Central Red Sea and lowest in the Northern Red Sea. These extreme events restructured coral assemblages, showing increased dominance of resilient taxa and shifts in overall composition.

These patterns highlight the urgent need for regional collaboration to strengthen reef resilience. Standardized monitoring, data sharing, and localized bleaching response networks are critical for timely detection, management, and mitigation of future bleaching across the Red Sea.

Record thermal stress on Caribbean reefs and management interventions

The 4th GCBE caused unprecedented, basin-wide impacts across the Caribbean, with extreme heat stress driving widespread bleaching and coral mortality, reversing decades of natural growth and restoration progress. Record thermal exposure caused major losses of reef-building coral species, with $>$30% mortality at many sites. These impacts were amplified by the preceding devastation from the regional stony coral tissue loss disease outbreak in $>$33 countries/territories, which had already weakened coral communities and reduced recovery capacity.

In the U.S. Virgin Islands, ~53% of corals bleached across 33 surveyed reefs and coral cover declined by 47%. Coral cover at 70% of Dominican Republic reefs declined to $<$5%. In Belize, heat stress caused catastrophic impacts where $>$90% of wild and restored acroporids died and long monitored sites collapsed to $<$5% coral. Across the Mesoamerican Region, ~30% of colonies died and Puerto Morelos experienced 100% mortality of *A. palmata*. *Acropora* were most affected, with wild and restored populations suffering rapid tissue loss and near complete collapse in several countries.

In response, managers implemented emergency interventions such as relocating nursery stock to cooler water, deploying shading structures, expanding land-based rescue, and prioritizing thermally tolerant donors; while new studies showed assisted sexual reproduction corals had higher resilience than adult colonies of the same species, with ~90% remaining healthy under heat stress. Despite severe region-wide losses due to accumulated heat stress, the increase in coordinated management actions including emergency interventions, expanded rescue and rehabilitation efforts, and science-based adaptive management offer a hopeful path for strengthening the Caribbean's future reef recovery.

Conclusions

While the extent and average intensity of GCBE4 have been truly unparalleled, a more long-term concern is the increasing frequency of bleaching-level heat stress accumulations around the world. By the end of GCBE3, the average time between severe bleaching events globally was only six years [40]. From 2019 to 2025, more than 25% of reef areas globally experienced DHW $\geq$ 4˚C-weeks four times or more [47], meaning near-annual bleaching stress has become a reality for much of the world's coral reefs. The ability of reefs to adapt at a rate and scale sufficient to overcome the current and projected rate of warming has not been demonstrated [518–520]. With an expectation of longer-lasting heat stress events and potentially annual marine heat extremes [40], corals will face increased pressure for acclimatization and/or adaptation and, if unable to do so, their survival under accelerating warming is increasingly threatened [34, 485].



MATERIALS AND METHODS

SPATIAL DATA

Several spatial datasets were used to perform the analyses and generate the maps presented in this report. The sources of these datasets, along with the processing steps and modifications applied, are detailed below.

Background maps

For the background representation of the regional maps, we used the high-resolution *Cross-Blended Hypsometric Tints (Hypso + Shaded Relief + Ocean Bottom + Drainages) v6* dataset from Natural Earth [521].

Regions, subregions, and ecoregions

Although several spatial classifications have been proposed for coral reef ecology (*e.g.* Veron *et al.* [307]), we adopted the Marine Ecoregions of the World (hereafter referred to as ecoregions) framework developed by Spalding *et al.* [522] to define our spatial units. These entities were organized within a hierarchical structure (Figure 4.1), designed to support analyses at multiple spatial scales while ensuring both ecological relevance and alignment with governance frameworks.

While the original delineations were largely retained, two modifications were implemented. First, ecoregion 87 ("Northern and Central Red Sea") was subdivided into two distinct ecoregions — "Northern Red Sea" and "Central Red Sea" — to better capture latitudinal differences in coral reef ecosystems within the Red Sea. Second, the boundary between ecoregion 89 ("Gulf of Aden") and ecoregion 92 ("Western Arabian Sea") was adjusted to coincide with the Yemen–Oman exclusive economic zone (EEZ) boundary which was obtained from the *World EEZ v12* dataset [523]. This adjustment was made to ensure consistency with the regional delineations used by the United Nations Environment Programme (UNEP). Following these modifications, a total of 110 ecoregions were defined.

These ecoregions were subsequently aggregated into 53 subregions. This aggregation was based on expert judgment, with the objective of preserving ecological coherence across the resulting units. The number of ecoregions combined within each subregion ranged from one to six. Although most ecoregions retained the names used by Spalding *et al.* [522], some were renamed for the purposes of this report.

Finally, the subregions were further aggregated into a total of 10 regions. In defining these regions, particular attention was paid not only to ecological homogeneity but also to the alignment with existing regional cooperation and governance structures, in order to enhance the operational applicability of the results. For example, the Pacific region aligns with the Secretariat of the Pacific Regional Environment Programme (SPREP), South Asia with the South Asia Co-operative Environment Programme (SACEP), and the Caribbean with the Cartagena Convention – SPAW Protocol (CAR-SPAW). Together, these 10 regions were aggregated to define the global level.

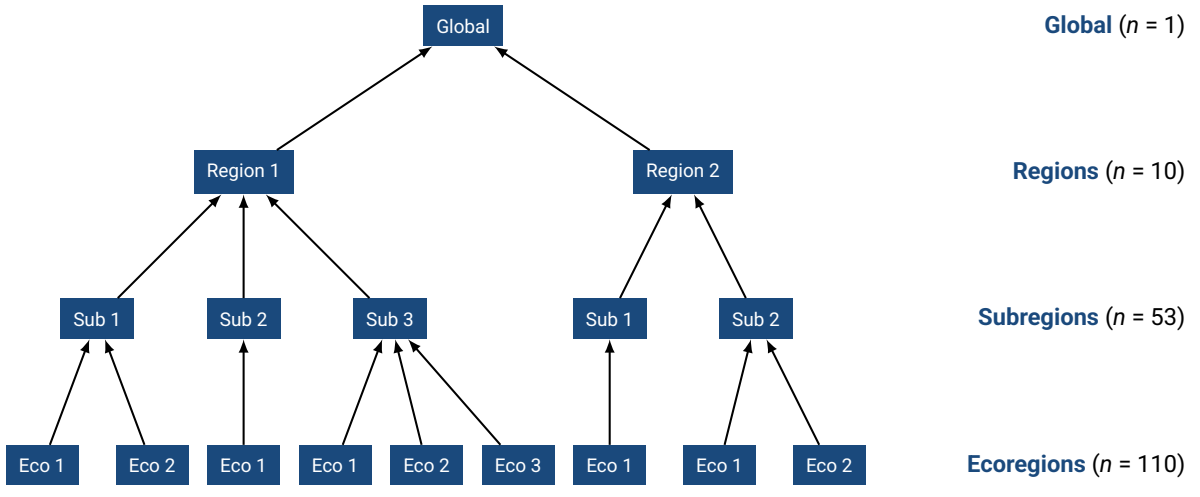
All scripts used to modify the ecoregions and to perform the aggregation into subregions and regions are publicly available at gcrmn_regions. The corresponding shapefiles for ecoregions, subregions, and regions can be downloaded through this repository.

Coral reef distribution

We used a modified version of the *Tropical Coral Reefs of the World* dataset [524], developed by the World Resources Institute, as the spatial representation of global coral reef distribution. This dataset provides a shapefile of shallow-water coral reef locations at a 500 m spatial resolution. Following expert recommendations, we updated the dataset by adding and removing coral reef locations (*i.e.* coral reef pixels) where necessary.

For Australia, we added a coral reef pixel for Norfolk Island, where the presence of coral reefs has been documented but was not included in the original dataset [525]. We also implemented substantial revisions in the Eastern Tropical Pacific region. Specifically, we removed all coral reef locations for the Galápagos Archipelago, except those associated with Darwin and Wolf Islands [526]. In addition, we added coral reef pixels based on sites where regional contributors have conducted surveys and directly confirmed the presence of coral reefs. Added coral reef locations were generated by buffering points to match the original 500 m spatial resolution.

The modified *Tropical Coral Reefs of the World* dataset (hereafter named the "coral reef distribution dataset") was subsequently used to map global coral reef distribution and to support several analyses, including the estimation of coral reef extent (see Coral reef extent) and the derivation of weights for benthic cover estimates (see Spatial weights).



▲ **Figure 4.1.** Schematic representation of the spatial hierarchy used for the report. The numbers in parentheses on the right correspond to the number of entities for each spatial scale.

BOX 3

Why reef maps matter for global status assessments

Author

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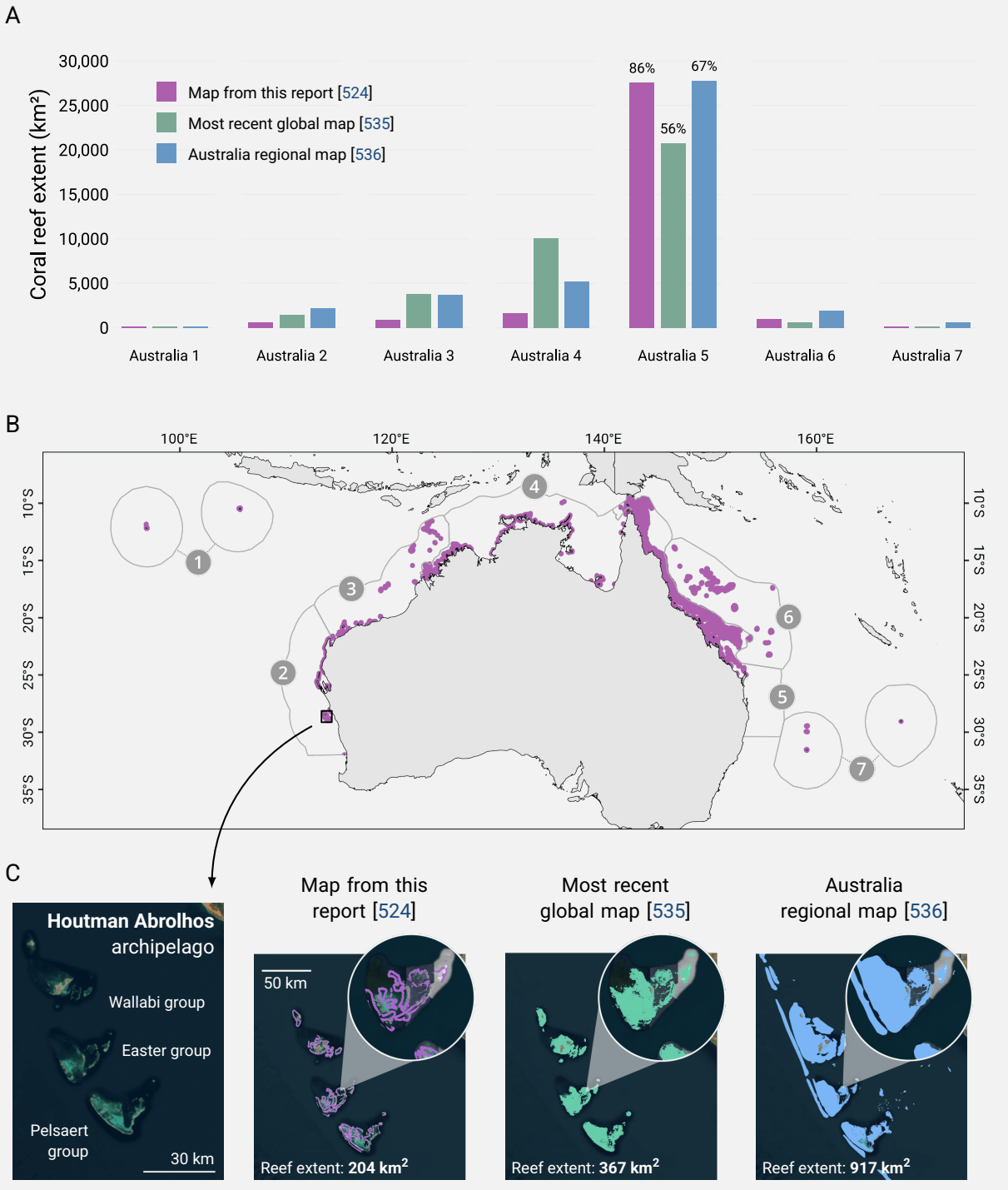
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Maps and monitoring go hand in hand. Shallow tropical reef extent maps underpin how monitoring data are weighted and summarised in the *Status of Coral Reefs of the World* reports. Maps also form the basis of ecosystem service valuations [17], assessments of reef use [527], threat modelling [528–530], and conservation prioritisation [510]. Despite the value of maps to coral reef science, consistently mapping underwater features at global scales remains challenging.

The first global reef extent map was compiled by Darwin from nautical charts from expeditions in the 1800s [531]. Aerial photography later enabled reefs to be mapped from above, and by the 1990s advances in computing allowed the World Resource Monitoring Centre to collate and digitise existing reef maps into a harmonised global GIS layer, The World Atlas of Coral Reefs [532]. A major leap forward in the early 2000s, The Millennium Coral Reef Mapping Project [533] used Landsat imagery to map ~80% of the world's reefs, becoming a key foundation for the composite basemap used in this report [524]. Since then, improvements in satellite data accessibility, spatial resolution and image processing have enabled a new generation of global reef maps [534], including the first global map of coral reef habitat, the Allen Coral Atlas [535].

With each mapping iteration, our understanding of coral reef extent and distribution improves, but important spatial inconsistencies remain in our global maps, with some geographies and deeper depths still less well resolved. This is particularly true for deeper, remote, or turbid areas.

Figure B3.1 illustrates how map choice can influence outcomes. Global products like the WRI Global Coral Reef map [524] (in purple) and the Allen Coral Atlas [535] (in green) support consistent inter-regional comparisons; the former, however, has inconsistencies due to being a composite of multiple data sources, while the latter is still limited by sensor capabilities and may under-resolve coral reefs in deeper or turbid environments. Trade-offs in processing mean regional products, such as the recent Australian Tropical Reef Features extent map [104] (in blue), are often better resolved and subject to stronger quality control, improving local representation but reducing global comparability. The "Houtman Abrolhos" example shows how earlier basemaps under-represent reef features — particularly in remote or understudied areas — compared to newer global and national products.



▲ **Figure B3.1.** Estimates of coral reef extent across GCRMN Australia subregions based on three commonly used coral reef distribution datasets (A). For Australia 5, values above the bars indicate the proportion of the total Australia region coral reef extent represented by this subregion. Map of the Australia subregions and coral reef extent derived from the WRI 2007 dataset, displayed with a 20 km buffer (B). Comparison of coral reef extent maps for the Houtman Abrolhos, the southernmost true coral reef system in Western Australia, located within the Australia 2 subregion (C).

Differences among reef extent products introduce spatial biases that influence how monitoring data are scaled and interpreted. Across the GCRMN Australia ecoregions, mapped reef area varies substantially among products, ranging from ~31,868 km^2 (WRI 2011) to ~36,911 km^2 (ACA 2021), and ~41,466 km^2 (NESP 2025, mapped beyond 20 m depth). These differences redistribute the proportional reef area assigned to each region and therefore alter their weighting in regional assessments. For example, the Great Barrier Reef and Torres Strait (Australia 5) contribute 86% of mapped area in this report, but 56% in newer mapping that better resolves turbid inshore reefs across northern and western Australia.

Moving beyond reef extent to incorporate geomorphic zones and habitats — strongly linked to coral cover, benthic diversity and ecosystem function [537] — will be essential for proper global accounting, and will help us better represent survey locations.

The maps shown in Figure B3.1 should not be viewed as competing products, but as complementary knowledge that, when integrated — for example by the Global Ecosystem Atlas that's ingesting ACA and NESP products — builds our knowledge of reef extent and distribution. Continued refinement of global reef extent and habitat maps — and consideration of geomorphic zones and habitat classifications in future *Status of Coral Reefs of the World* analyses [535] — will be critical to improving global reef accounting and strengthening future status assessments.

CORAL REEF EXTENT

The coral reef distribution dataset (see Coral reef distribution) was used to quantify coral reef extent at the global, regional, and subregional levels. Coral reef extent was estimated by performing a spatial intersection between the coral reef distribution dataset and each previously defined spatial entity (see Regions, subregions, and ecoregions). For each entity, total coral reef extent was calculated using the EPSG:4326 coordinate reference system. Coral reef extent was further expressed as a proportion of (1) the total global coral reef extent and (2) the corresponding total regional coral reef extent.

Differences between our estimates and those derived from other datasets primarily reflect variations in how coral reefs are defined. For example, Lyons *et al.* [538], using the Allen Coral Atlas dataset, estimated the global extent of shallow coral reefs at 348,361 km^2, compared with our estimate of 249,677 km^2 from the modified *Tropical Coral Reefs of the World* dataset. Discrepancies in coral reef extent estimates can be particularly pronounced at the national or territorial scale (Box 3).

HUMAN POPULATION

Indicators related to human population were derived from the *Gridded Population of the World v4.11 – Population Count* dataset [95], which provides estimates of human population counts for the years 2000, 2005, 2010, 2015, and 2020 at a spatial resolution of 30 arc-seconds (approximately 1 km at the equator).

A spatial intersection was performed between the coral reef distribution dataset (see Coral reef distribution) and the GCRMN regions, to associate each coral reef polygon with its corresponding region. Using the resulting spatial layer, we generated 5 km buffers around coral reefs within each region using Google Earth Engine. Although partially arbitrary, the 5 km distance was selected as a reasonable threshold representing the spatial extent over which human populations are likely to exert direct (*e.g.* fishing, coastal development) or indirect (*e.g.* sedimentation, nutrient runoff) pressures on coral reefs [308]. This distance may also serve as a proxy for the dependency of nearby populations on coral reefs, particularly for subsistence fishing [2].

Using Google Earth Engine, we then summed up the total human population within each buffer and for each reference year (2000, 2005, 2010, and 2020). The same process was also used with the subregions in place of regions.

Based on these extracted data, we computed three indicators of human population:

1. The total number of inhabitants in 2020 living within 5 km of coral reefs,
2. The absolute change in the number of inhabitants living within 5 km of coral reefs between 2000 and 2020,
3. The relative change in the number of inhabitants living within 5 km of coral reefs between 2000 and 2020.

The Gridded Population of the World dataset does not account for non-permanent human populations. Consequently, human population estimates are likely underestimated in certain subregions, such as South Asia 1, where military personnel are present on Diego Garcia atoll. Furthermore, global gridded population datasets, including the *Gridded Population of the World v4.11 – Population Count* dataset, have been reported to systematically underestimate rural populations [539].

CYCLONE OCCURRENCE

Tropical storms, and more particularly cyclones (also named hurricanes in the Caribbean region and typhoons in the East Asian Seas region), are major disturbance events affecting coral reefs [108]. To examine their occurrence on coral reefs of the world over the past four decades, we used version 4.01 of the *International Best Track Archive for Climate Stewardship* (IBTrACS) dataset [118, 540]. This dataset provides 3-hourly storm positions at a spatial resolution of 0.1° (approximately 10 km). We applied four successive filters to the IBTrACS dataset. First, we retained only cyclones, defined as events with maximum sustained wind speeds exceeding 119 $km.h^{-1}$ (*i.e.* Category 1 or higher on the Saffir–Simpson scale; see Table 4.1), across their entire tracks. It should be noted, however, that wind speeds may fall below this threshold at certain points along a cyclone's track. Second, we restricted the dataset to cyclones occurring after 1 January 1980, corresponding to the reporting period of interest. Third, we excluded cyclones lacking sustained wind speed records for all positions along their tracks. Finally, using the coral reef distribution dataset (see section Coral reef distribution), we generated a 100 km buffer around coral reefs within Google Earth Engine. Only cyclones intersecting this buffer were retained; however, acknowledging that hurricane impacts on coral reefs may extend beyond this distance [108, 541], this occurrence value might underestimate the actual number of tropical cyclones impacting coral reefs.

Following these filtering steps, each cyclone was assigned a Saffir-Simpson scale category (Table 4.1) based on the maximum sustained wind speed recorded along its track. For each cyclone, we then extracted (1) the minimum distance between its trajectory and coral reefs, and (2) the sustained wind speed at the point on the track closest to a coral reef. These extractions were conducted separately for each region and subregion.

HEAT STRESS INDICATORS

To characterize the heat stress on coral reefs of the world, we used the *CoralTemp v3.1* dataset [107], developed by the NOAA Coral Reef Watch program. CoralTemp provides global daily sea surface temperature (SST) data and derived metrics at a spatial resolution of 5 km, available from January 1, 1985, to the present.

▼ **Table 4.1.** The different categories of the Saffir-Simpson scale and their associated sustained wind speed. Note that Wehner & Kossin [542] proposed the addition of a sixth category to the Saffir-Simpson scale.

Type	Category	Sustained wind speed ($km.h^{-1}$)
Tropical storm		64 to 118
Cyclone	1	119 to 153
	2	154 to 177
	3	178 to 210
	4	211 to 251
	5	$>$ 251

Long-term SST trends

We extracted daily SST averages from 1 January 1985 to 31 December 2025 over coral reef areas within each GCRMN region and subregion, using the coral reef distribution dataset (see Coral reef distribution). Generalized least squares regressions were then applied to the resulting SST time series. Generalized least squares was preferred over ordinary least squares regression because it explicitly accounts for temporal autocorrelation and yields more reliable estimates of standard errors [543, 544]. The regression slopes, representing annual SST trends, were multiplied by ten to derive decadal warming rates for each region and subregion, and by 40, which corresponds to the number of years between 1985 and 2025, to estimate the mean long-term SST change over the study period. Finally, long-term SST averages were calculated by averaging all SST values within each extracted time series.

SST anomaly

Using the daily SST averages from 1 January 1985 to 31 December 2025 we calculated long-term SST averages for the study period by averaging all values within each region or subregion. Daily SST anomalies were subsequently calculated by subtracting these long-term means from the corresponding daily SST values, using 1985–2025 as the reference period. Finally, daily SST anomalies were averaged per year to obtain annual SST anomalies.

Degree heating weeks

We used degree heating weeks (DHW) as a proxy of coral bleaching. DHW quantifies the cumulative heat stress experienced by hard corals by integrating both the intensity and duration of anomalously warm SST. DHW accumulate daily temperature anomalies exceeding 1 °C above the climatological maximum (HotSpot) over a rolling 12-week (84-day) period and are expressed in °C-weeks [107].

Using the annual maximum DHW product from Coral Reef Watch, we quantified the distribution of coral reef pixels across DHW values for each year from 1986 to 2024. This assessment was performed for all GCRMN regions and subregions using the coral reef distribution dataset (see Coral reef distribution). DHW values were then classified into seven heat stress levels following the categories defined by the NOAA Coral Reef Watch program (Table 4.2), with the "No Stress" and "Bleaching Watch" classes merged into a single "No stress/Watch" category for clarity. Finally, we computed the annual relative frequency of coral reef pixels within each heat stress level for every GCRMN region and subregion.

In the regional chapters, the term "Percentage of coral reefs" was used in the heat stress level figures (*e.g.* Figure 2.8.3) for the sake of simplicity. However, the more technically accurate expression is "Percentage of 5 km coral reef pixels." This distinction is important because a 5 km pixel classified as containing coral reefs may include only a small fraction of coral reef area (*e.g.* 1% of the pixel), whereas another pixel may be almost entirely covered by coral reefs.

Oceanic Niño Index

To visualize the alternation between the two phases of the El Niño Southern Oscillation (ENSO) in Figure 1.4, we used the Oceanic Niño Index v2.0 [545] provided by NOAA. The Oceanic Niño Index is calculated as a three-month running mean of ERSST.v5 SST anomalies in the Niño 3.4 region (5° N-5° S, 120°-170° W), relative to the 1971–2000 baseline period. Thresholds of +0.5 °C and +1.5 °C were used to identify El Niño and strong El Niño conditions, respectively. Conversely, thresholds of −0.5 °C and −1.5 °C were used to identify La Niña and strong La Niña conditions.

▼ **Table 4.2.** Definitions of heat stress levels based on the NOAA Coral Reef Watch program [546]. Contrary to the categories defined by the NOAA Coral Reef Watch, the categories "No Stress" and "Bleaching Watch" were grouped under the category "No stress/Watch". The HotSpot is the difference between the daily SST and the maximum monthly mean, defined as the highest climatological monthly mean SST over the 1985–2012 reference period [107].

Heat stress level	Definition	Potential coral bleaching and mortality
No stress / Watch	$0 \leq$ HotSpot < 1	
Warning	$1 \leq$ HotSpot and $0 <$ DHW < 4	Risk of Possible Bleaching
Alert 1	$1 \leq$ HotSpot and $4 \leq$ DHW < 8	Risk of Reef-Wide Bleaching
Alert 2	$1 \leq$ HotSpot and $8 \leq$ DHW < 12	Risk of Reef-Wide Bleaching with Mortality of Heat-Sensitive Corals
Alert 3	$1 \leq$ HotSpot and $12 \leq$ DHW < 16	Risk of Multi-Species Mortality
Alert 4	$1 \leq$ HotSpot and $16 \leq$ DHW < 20	Risk of Severe, Multi-Species Mortality ($>$ 50% of corals)
Alert 5	$1 \leq$ HotSpot and $20 \leq$ DHW	Risk of Near Complete Mortality ($>$ 80% of corals)

BENTHIC COVER

Although numerous ecological indicators are monitored within the world's coral reefs [547], measurements of benthic cover are the most widely collected, both spatially and temporally. The main reasons are that most monitoring methods of benthic cover are relatively simple, inexpensive, and require limited skills in taxonomy [548, 549]. Early measurements of the percentage cover of benthic organisms started before 1980 [550, 551], and new monitoring methods have little impact on comparison with data acquired using historical methods. For example, different methods used to estimate benthic cover, such as point-intersect transects and photo-quadrats, lead to similar results [552].

This comparability of measurements acquired through different methods makes it possible to combine data from multiple monitoring programs to estimate temporal trends of percentage cover of benthic organisms at large spatial scales. This type of analysis, called full data analysis, allows for greater precision in results compared to other types of synthetic studies, like meta-analyses or systematic reviews [553]. A full data analysis thus makes it possible to estimate complex temporal trends rather than a simple slope, as would be the case with a meta-analysis through a meta-effect size [553]. Moreover, since full data analysis is based on raw data, it allows the use of both published and unpublished data, compared to meta-analyses and systematic reviews, which rely on published studies only.

Among the wide range of benthic categories used by monitoring programs globally, in this report we include only hard coral, macroalgal, and turf algal cover. In contrast to the *Status of Coral Reefs of the World: 2020* GCRMN report [30], we present macroalgae and turf algae separately rather than as total algal cover, as total algal cover — which combines macroalgae, turf algae, and coralline algae — can mask important differences in the functional roles of these groups [554]. Note, however, that trends in turf algal cover are mostly presented at

the regional scale. Turf algal cover was not consistently included in subregional analyses because data availability at that spatial scale was insufficient to produce robust estimates across many subregions. Additionally, coralline algae were excluded from this analysis as they are largely cryptic and therefore poorly detected by the two-dimensional survey methods used in most monitoring programs, to avoid reporting potentially biased patterns that would not accurately reflect ecological change.

Data integration

A full data analysis of benthic cover in coral reefs is only possible if a homogeneous dataset is available. Since a homogeneous dataset was not available across all monitoring programs, it was necessary to create one by applying a data integration step before beginning the data analysis. The goal of the data integration was to assemble a homogeneous synthetic dataset on coral reef benthic cover through the aggregation of multiple heterogeneous individual datasets. The data integration was performed using an improved version of the workflow developed by Wicquart *et al.* [513] in the frame of the production of the *Status of Coral Reefs of the World: 2020* GCRMN report [30]. The main workflow enhancements compared to this previous GCRMN report were the additions of quality checks to provide a more reliable assurance of data quality. The data integration workflow used for the present report comprises five main steps, which are illustrated in Figure 4.3 and described in the following sections.

Data collation

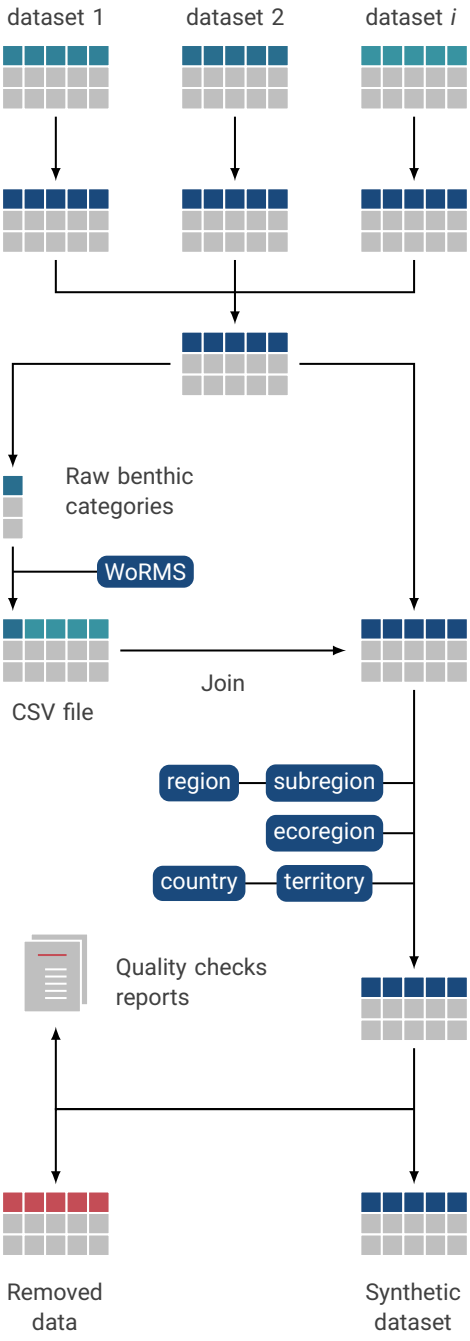
We contacted previous data contributors to the *Status of Coral Reefs of the World: 2020* GCRMN report [30] to obtain permission to reuse their data and to request updated versions of the datasets they had previously provided. In addition, an open call for data contributions was launched in May 2024 through both the GCRMN and ICRI networks, including the ICRI membership, its partners and communication channels. Finally, data compilation was further supplemented by identifying datasets available in scientific articles, data papers, and coral reef–dedicated data platforms (*e.g.* ReefCloud, MERMAID), and inviting the respective data owners to contribute. Specifically, the following datasets were collated: Wilkinson *et al.* [555], Jeffrey *et al.* [556], Reef Check Foundation [557], Montilla *et al.* [558], Mallela *et al.* [559], Great Barrier Reef Marine Park Authority [560], Edgar *et al.* [561], CRIOBE SNO CORAIL [562], California State University Northridge [563], Enercal Energies Nouvelles [564], Office de l'Eau Martinique and Impact Mer [565], Belize National Coral Reef Monitoring Network [566], Ali *et al.* [567], Elamin *et al.* [568], Abaker [569], two datasets from CREMP [570, 571], and nine datasets from NOAA [572–580].

Data standardization

Individual datasets gathered during the data collation step were heterogeneous in data formats (*e.g.* CSV, Excel), data structure, variable names, and units (*e.g.* meters vs feet for the depth). For this reason, a standardization step was necessary to enable these individual datasets to be merged into a single synthetic dataset. We created an R script for each individual dataset, and applied specific modifications to each of them to match the data format and variables that have been defined in (Table 4.3). These modifications included merging of Excel spreadsheets, joining of the main dataset with its metadata (*e.g.* site coordinates, benthic codes), selecting and renaming variables, converting units, Coordinates Reference System (CRS) and date formats, deriving benthic cover values from number of points per transect or photo quadrats (Figure 4.3 - Step 1).

All the data standardization was conducted using code, ensuring a clear and transparent record of all changes made. Since all the R scripts are publicly available, it is also possible for data owners to check the validity of modifications applied to their datasets (see gcrmndb_benthos).

At the end of the data standardization step, individual datasets were merged to create a single raw synthetic dataset (Figure 4.3 – Step 2).



1 Standardization

Standardize all individual datasets to the same format

2 Grouping

Bind all standardized individual datasets together

3 Taxonomic recategorization

Correct, recategorize, and find upper taxonomic levels

4 Spatial attribution

Assign GCRMN region, subregion, ecoregion, country, and territory to each site

5 Quality checks

Control data quality and remove incorrect rows

▲ **Figure 4.3.** Illustration of the five main steps of the data integration workflow used to create the synthetic dataset on coral reefs benthic cover.

▼ **Table 4.3.** Description of variables included in the gcrmndb_benthos synthetic dataset. Variable names (except region, subregion, ecoregion, category, subcategory, and condition) correspond to DarwinCore terms. Variables 2 to 13 are associated with spatial dimension, variables 14 to 17 are associated with temporal dimension, and variables 20 to 28 are associated with taxonomy.

Nb	Variable	Type	Description
1	datasetID	Factor	ID of the dataset
2	region	Factor	GCRMN region
3	subregion	Factor	GCRMN subregion
4	ecoregion	Factor	GCRMN ecoregion
5	country	Factor	Country
6	territory	Character	Territory
7	locality	Character	Site name
8	habitat	Factor	Habitat
9	parentEventID	Integer	Transect ID
10	eventID	Integer	Quadrat ID
11	decimalLatitude	Numeric	Latitude (decimal, EPSG:4326)
12	decimalLongitude	Numeric	Longitude (decimal, EPSG:4326)
13	verbatimDepth	Numeric	Depth (m)
14	year	Integer	Four-digit year
15	month	Integer	Integer month
16	day	Integer	Integer day
17	eventDate	Date	Date (YYYY-MM-DD, ISO 8601)
18	samplingProtocol	Character	Method used to acquire the measurement
19	recordedBy	Character	Person who acquired the measurement
20	category	Factor	Benthic category
21	subcategory	Factor	Benthic subcategory
22	condition	Character	Condition for hard corals
23	phylum	Character	Phylum
24	class	Character	Class
25	order	Character	Order
26	family	Character	Family
27	genus	Character	Genus
28	scientificName	Character	Species
29	measurementValue	Numeric	Percentage cover

Taxonomic re-categorization

Raw benthic categories recorded by data owners were manually re-categorized into standardized taxonomic and descriptive variables (variables 20 to 28 in Table 4.3). Where categories were broad and non-taxonomic (*e.g.* "Turf"), we assigned the appropriate 'category' and 'subcategory' values directly; lower-level taxonomic variables were left blank. Where categories contained sufficient taxonomic information (*e.g. Acropora*), we verified spellings against the *World Register of Marine Species* (WoRMS), accounted for any taxonomic updates, and populated the taxonomic variables accordingly.

The variables 'category' and 'subcategory' were then assigned in R using the following attribution rules:

- **Hard coral** — 'order' = Scleractinia, or 'family' = Milleporidae or Helioporidae. Excludes dead corals; may include partially or fully bleached corals.
- **Macroalgae** — 'class' = Phaeophyceae, Florideophyceae, or Ulvophyceae, and 'order' ≠ Corallinales.
- **Coralline algae** — 'order' = Corallinales.
- **Turf algae** — assigned where contributors recorded "turf algae", regardless of taxonomic information.
- **Cyanobacteria** — 'phylum' = Cyanobacteria.
- **Other fauna** — 'phylum' = Porifera, Chordata, Echinodermata, Bryozoa, Annelida, Mollusca, or Arthropoda; or 'order' = Actiniaria, Alcyonacea, Zoantharia, Corallimorpharia, or Antipatharia; or 'subclass' = Octocorallia.
- **Seagrass** — 'phylum' = Tracheophyta.

The following are additional specific cases encountered during taxonomic re-categorization and the rules we followed for each case:

- **Mixed categories** — where a single raw category combines two or more distinct benthic types. We assigned the lowest common denominator as the 'Category' variable. For example, "Macroalgae and turf algae" was recorded as "Algae".
- **Stacked categories** — where a raw category describes one benthic type overlying another. We assigned the category that covered the other. For example, "Algae on dead coral" was recorded as "Algae".
- **Homonym taxa names** — where a raw category name refers to more than one taxonomic group. We either contacted the data owner to resolve the ambiguity during data standardization (*e.g.* "Turbinaria" replaced with "Turbinaria - coral" or "Turbinaria - macroalgae"), or, where this was not possible, left the 'Category' variable blank, resulting in the removal of those rows from the dataset.
- **Unrequired or unintelligible categories** — where a raw category was either irrelevant to benthic composition or could not be interpreted. The 'Category' variable was left blank, resulting in removal of those rows. Examples include "Shadow", "Other", and "Unknown".

Following recategorization, we joined the tables containing the 'organismID' variables and the new variables (variables 20 to 28 in Table 4.3) with the main data table. Finally, we removed the temporary variable 'organismID' and all rows containing missing values for the variable 'category'. Note that rows with missing 'category' values were removed without rescaling the remaining percentage cover values. As a result, the total percentage cover within a sampling unit (*e.g.* quadrat, transect) may sum to less than 100%. Finally, percentage cover values were summed by a sampling unit (*i.e.* quadrat, or transect) to remove any duplicated benthic categories that may have occurred following taxonomic re-categorization.

The recategorization of raw benthic categories into standardized categories through variables 20 to 28 of Table 4.3, can be publicly accessed in the file "03_tax-recategorisation.csv" on the gcrmndb_benthos GitHub repository.

Spatial attribution

Using site coordinates, we added five additional spatial-dimension variables to the dataset (Figure 4.3 – Step 4). We performed a spatial intersection to associate GCRMN 'region', 'subregion' and 'ecoregion' variables to each site using the previously defined spatial entities (see Regions, subregions, and ecoregions). We also added the 'country' and 'territory' variables by using the variables 'SOVEREIGN1' and 'TERRITORY1' from the *World EEZ v12* dataset [523]. Some sites located near coastlines fell outside their designated EEZ due to the limited spatial resolution of the *World EEZ v12* dataset, resulting in missing values for the 'country' and 'territory' variables. To minimize this, we performed the spatial assignment in two steps: first, a standard spatial intersection between all sites and the *World EEZ v12* dataset; then, for sites that remained unassigned, a second spatial intersection using EEZ polygons buffered by 1 km.

Quality checks

Aggregating large datasets without sufficient quality verification risks compromising data integrity and introducing bias. This has been demonstrated in previous studies, where insufficient verification of large synthetic datasets has led to biased interpretations of ecological patterns and processes [581]. For this reason, we added a step dedicated to the assessment of data quality (Figure 4.3 – Step 5). We defined a set of nine quality checks that were inspired by Vandepitte *et al.* [582] (Table 4.4). Quality checks take the form of questions associated with one or several variable(s) and can easily be transcribed into code.

▼ **Table 4.4.** List of quality checks used for the gcrmndb_benthos synthetic dataset.

Nb	Variable	Question
1	decimalLatitude decimalLongitude	Are the latitude and longitude available?
2	decimalLatitude	Is the latitude within its possible boundaries (*i.e.* between -90 and 90)?
3	decimalLongitude	Is the longitude within its possible boundaries (*i.e.* between -180 and 180)?
4	decimalLatitude decimalLongitude	Is the site within the coral reef distribution area (100 km buffer)?
5	decimalLatitude decimalLongitude	Is the site located within a GCRMN region?
6	decimalLatitude decimalLongitude	Is the site located within an EEZ (1 km buffer)?
7	year	Is the year available?
8	measurementValue	Is the sum of the percentage cover of benthic categories within the sampling unit greater than 0 and lower than 100?
9	measurementValue	Is the percentage cover of a given benthic category (*i.e.* a row) greater than 0 and lower than 100?

The sum of the percentage cover of benthic categories within a sampling unit can be slightly greater than 100 due to rounding of decimal places [583]. Thus, to avoid removing too much data, rows for which the sum of the percentage cover of benthic categories within a sampling unit was between 100 and 101 were normalised relative to 100. This step was done before applying the quality check 8 (Table 4.4).

All data rows failing to meet the quality checks were removed, however, to maximize data retention, all efforts were made to identify and correct systematic errors before data was removed.

Code availability

The data integration process, including data collation, data standardization and data cleaning, was done through the gcrmndb_benthos GitHub repository. This repository serves as a shared resource for GCRMN reports and was not specifically established or maintained for the purposes of the current report. The data integration steps were performed using R software version 4.6.0 [584] and predominantly packages *tidyverse* [585], *mermaidr* [586], and *sf* [587]. The code used for the data integration of benthic cover monitoring data used for this report can be accessed at https://github.com/GCRMN/gcrmndb_benthos and was archived in a GitHub release (version 1.0.0).

Data analysis

Data selection

Several filters were applied to the gcrmndb_benthos synthetic dataset prior to analysis. First, we retained only records for benthic categories hard coral, macroalgae, and turf algae, and restricted observations to depths of 0–30 m, corresponding to shallow coral reefs [588]. Observations with missing depth information were retained to avoid excessive data loss.

Additionally, dataset-specific filters were applied based on methodological compatibility and regional expert review. Data from the XL Catlin Seaview Survey (datasetID *0009*) were excluded as surveys were conducted along kilometre-long transects, which are not comparable to the standard benthic monitoring transects of a few tens of metres. Aberrant data points, or those influencing model outputs in ways inconsistent with published literature and field observations, were removed based on the expert advice of regional node coordinators. The dataset-specific filters that were applied are detailed in the following paragraphs.

Australia region. Data from the AIMS dataset (datasetID *0003*) collected in 1992 were excluded because they were compiled during the preparatory phase of the long-term monitoring program and were considered unreliable. Additionally, data from the Reef Check Foundation dataset (datasetID *0015*) for the Houtman ecoregion prior to 2000 and for the Arnhem Coast to Gulf of Carpentaria, Bonaparte Coast, and Coral Sea ecoregions prior to 2010; and data from the DBCA dataset (datasetID *0112*) collected before 1994 in the Exmouth to Broome ecoregion, were excluded as values were considered either unrealistically high or low by regional experts.

Brazil region. All Reef Check Foundation data (datasetID *0015*) were excluded from analysis because they were already incorporated into the RCBrasil dataset (datasetID *0208*), which compiles national Reef Check data. Data from the Brazil 4 subregion (Trindade and Martin Vaz Islands) were also excluded as these reefs are considered rocky reefs rather than true coral reefs. In addition, data from one site from the RCBrasil dataset (datasetID *0208*) and one site from the Reef Life Survey dataset (datasetID *0038*) were excluded due to aberrant values identified by regional experts.

Caribbean region. Macroalgal cover data from the CZMU dataset (datasetID *0092*) collected before 1985 in the Caribbean 2 subregion (Eastern Caribbean) were excluded. Similarly, macroalgal cover data from the Wilkinson *et al.* [555] dataset (datasetID *0133*) collected before 1990 were also excluded from the Caribbean 5 subregion (Greater Antilles) and the Bahamian ecoregion. These values were considered unrealistically high by regional experts.

Eastern Tropical Pacific region. Data collected before 2006 for the Cortezian ecoregion were excluded because values were considered excessively high by regional experts.

Pacific region. Data from the Reef Check Foundation dataset (datasetID *0015*) collected before 2000 in the Samoa Islands ecoregion and before 1999 in the West Caroline Islands ecoregion were excluded due to unusually high hard coral cover values.

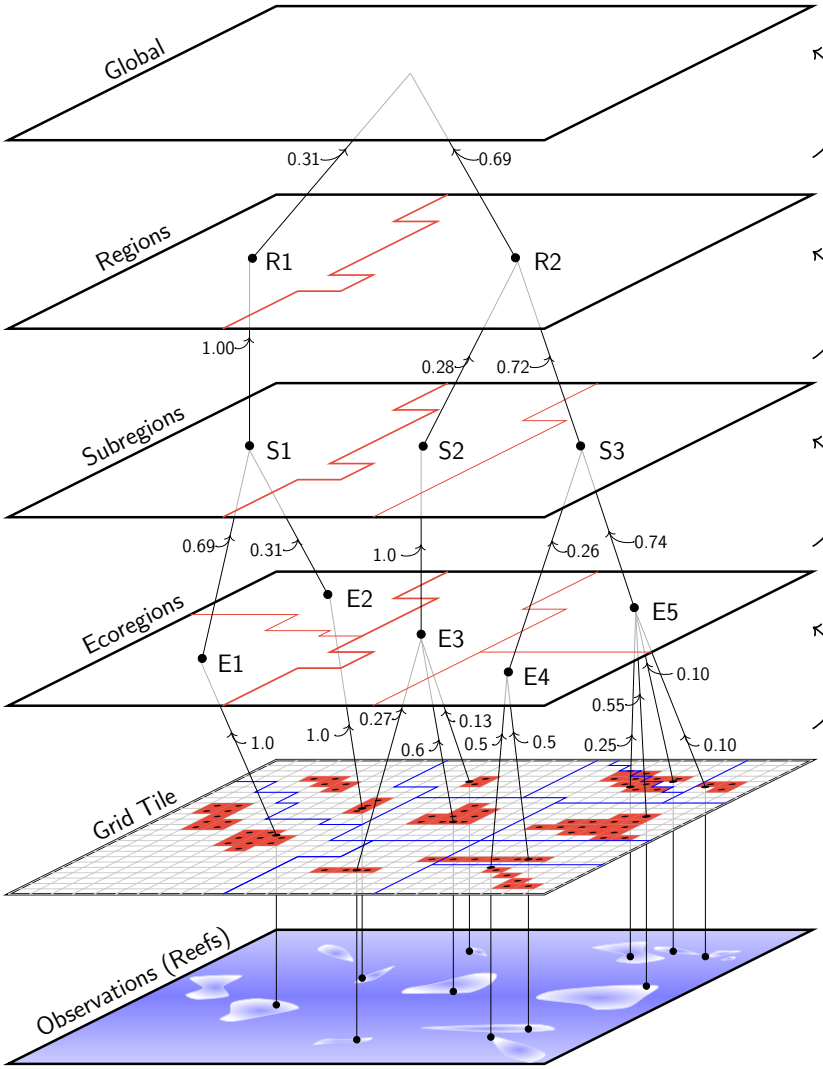
Western Indian Ocean region. Data from the UTOPIAN dataset (datasetID *0230*) for the year 2023 were excluded because they generated an anomalous peak in model estimates.

Modelling considerations

Sampling designs for data collection across our synthetic dataset span a spectrum from permanently fixed to randomly selected survey sites, reflecting the varying purposes, resources, and logistical constraints of individual monitoring programs. While fixed sites serve as their own baselines over time, providing relatively efficient means of detecting temporal trends, there is a risk the selected locations may not be representative of the broader area. By contrast, randomly selected sites reduce this potential bias and improve spatial representation, but typically require a considerably higher number of sites (*i.e.* sample size) to detect temporal trends above the noise.

The challenge, and opportunity, for the current report is to estimate the status and trends of coral reefs at broad spatio-temporal scales using a collection of datasets from a large number of disparate monitoring programs from around the world. Whenever multiple datasets are integrated (particularly if individual datasets represent disparate geographies), the issues of representativeness and bias are exacerbated. Firstly, quantitative estimates are always driven by sample sizes. Within any well-designed monitoring program, efforts are made to ensure the design remains relatively balanced. However, when aggregating multiple datasets up to a broader scale, sample sizes will differ across programs, thus introducing imbalance into the broader design. It is therefore important to control for varying sample sizes to minimise the risks of bias towards more heavily replicated datasets, especially since sampling intensity is likely a function of the relative prosperity of the surrounding populations and proximity to major population centers, rather than the density and distribution of the reefs themselves.

Large (and complex) spatio-temporal models that employ full positional encoding to evaluate spatial patterns between all possible pairs of sampling units (sites). These models can facilitate the transfer of information from the fine, observation level measurements to the broader levels of this report. By assuming that a response (such as percentage live hard coral cover) varies continuously over an entire two-dimensional surface, such models are able to leverage trends in areas of relatively high sample density to estimate the trends in neighbouring areas of sparse sampling density — albeit with greater uncertainty. However, such models proved to be too computationally burdensome and were incredibly difficult to tune to ensure they yield sensible outcomes. They also assumed that changes over space were relatively gradual and thus, can easily smooth over what would otherwise be considered abrupt local changes.



Region Weights

Region	Area	Weight
R1	26	0.31
R2	54	0.69

Subregion Weights

Region	Subregion	Area	Weight
R1	S1	26	1.00
R2	S2	15	0.28
R2	S3	39	0.72

Ecoregion Weights

Subregion	Ecoregion	Area	Weight
S1	E1	18	0.69
S1	E2	8	0.31
S2	E3	15	0.28
S3	E4	10	0.26
S3	E5	29	0.74

Grid Tile Weights

Ecoregion	Tile ID	Area	Weight
E1	T11-20	18	1.00
E2	T16-18	8	1.00
E3	T05-11	4	0.27
E3	T13-11	9	0.60
E3	T21-13	2	0.13
E4	T05-05	5	0.50
E4	T06-04	5	0.50
E5	T15-03	16	0.55
E5	T21-03	3	0.10
E5	T20-07	3	0.24
E5	T22-05	7	0.10

▲ **Figure 4.4.** Schematic diagram illustrating the hierarchical structure relating the hypothetical observations (bottom layer) to the level of a 10×10 km grid tile, ecoregion, region, and global scale. The grid tile layer depicts the 10×10 km tile containing reef (red) and the voronoi polygons (blue lines) used to partition area zones of sample unit influence. The ecoregion layer illustrates five fictitious ecoregions which are aggregated into two regions in the layer above. Vertical lines illustrate the aggregation of data along the hierarchy and the numbers along these paths represent the aggregation weights (also tabulated).

Furthermore, incorporating information about the spatial distribution of reefs as well as physical barriers to auto-correlative processes are far from trivial.

As an alternative, we explored hierarchical models in which sampling units are progressively aggregated with their neighbours into larger and larger units. For example, neighbouring quadrats are grouped together into sites, sites into global grid locations (see below) and so on up to the level of the whole globe. This represents a pseudo-spatial model in that although the influence of neighbouring data does deteriorate along the hierarchy, it does so in increments relating to group membership rather than as a continuous function of spatial distance. Hence in the case of an area comprising of ten sub-areas, each of the sub-areas will share some information with the other sub-areas even though any one of them might

be geographically closer to a member of another area than most of the sub-areas in its designated area (the classic nearest *vs* average neighbour conundrum). In any case, all attempts to fit full, global hierarchical models with the very disparate datasets proved very difficult to stabilise.

To address these data challenges, we therefore used smaller hierarchical models (Figure 4.4), fitted separately to each ecoregion, and integrated within a spatially-weighted aggregation hierarchy in which individual model posteriors (annual estimates) were propagated up through the hierarchy. Although this approach retains some pseudo-spatial hierarchy — allowing data-poor areas to leverage patterns from data-rich areas, the leveraging is constrained to within ecoregions, where ecological processes are more likely to be homogeneous and trends therefore more likely to be consistent with the observed data.

Spatial hierarchy

The pseudo-spatial hierarchy outlined above necessitates incremental jumps in scale from the level at which observations are collected up to the global (or even regional) scale. If the increments are too large, the information (temporal patterns) shared across neighbouring spatial units might be driven by very different underlying conditions and thus not appropriate.

The original datasets collated in this study were provided at scales of either quadrat/transect or spatial aggregations thereof. These can be naturally grouped into sites (or individual reefs) as the first incremental scale jump, however, subsequent increments are less obvious. While there are numerous ways of grouping coral reef locations into broader geographic areas (*e.g.* Veron *et al.* [307]), we used the ecoregions, that were further grouped up into subregions and then regions to provide additional modelling and reporting granularity (Figure 4.4).

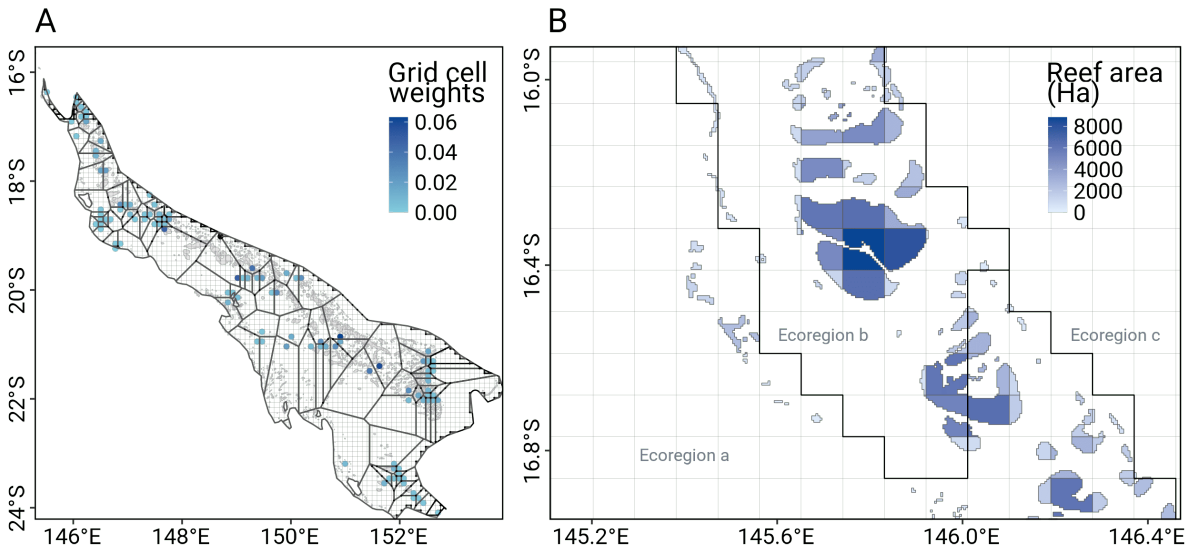
The jump in spatial scale from Site to ecoregion can be very large and span a wide range of influential processes and drivers. We therefore included an additional intermediate scale by creating an array (grid) of cells of a constant size across the globe. The application of this grid layer has the added benefit of reconciling design differences between fixed and randomly selected sites, allowing the incorporation of multiple sampling designs in one model.

Spatial weights

To maximise the likelihood that hierarchical aggregations are reflective of broad spatial patterns and not heavily biased by sampling effort alone, the aggregations were weighted by the proportion of coral reef extent represented by each spatial unit.

However, estimating the global distribution and extent of coral reefs remains challenging, as the granularity and accuracy of coral reef mapping vary substantially among regions (Box 3). To address this, we used the *Tropical Coral Reefs of the World* global coral reef distribution dataset ([524]; see Coral reef distribution).

The intermediate spatial scale between observed sites and ecoregions was provided by generating a 10×10 km grid of tiles across the entire globe and assigning a unique identifier to each tile. All observed site level locations were assigned to a grid tile on the basis of nearest neighbour within 10 km. To estimate the amount of coral reef extent within each ecoregion that was represented by each of the observed sites, voronoi polygons were generated from the unique site locations and overlaid onto the grid (Figure 4.5A). The coral reef extent associated with each voronoi cell was then expressed as a proportion of the total ecoregion coral reef extent, thereby representing the relative weight that each grid tile should carry in the analyses.



▲ **Figure 4.5.** Illustration of voronoi polygons (A) and grid tile weights (B). In subfigure A, the voronoi polygons are overlaid on the 10×10 km grid and reefs (grey). Shaded grid tiles represent grid tiles containing observed sites and hue of the grid tile shading represents the relative weights (proportion of coral reef extent in each grid tile). In subfigure B, the relative coral reef extent is represented by each 10×10 km grid tile within three ecoregions. The hue of reef fill is proportional to the relative coral reef extent in the ecoregion.

When aggregating estimates from ecoregion to subregion scale, each ecoregion was weighted by its proportional reef extent within the subregion (Figure 4.5B). The same approach was applied at higher scales: subregions were weighted by their proportional reef extent within their region, and regions by their proportional reef extent globally.

Statistical model

Hard coral, macroalgal, and turf algal cover were calculated by summing observation level data across associated taxonomic groupings.

Separate ecoregion Bayesian hierarchical models were constructed within the stan statistical modelling platform [589] via the *rstan* interface [590]. Each model comprised a model matrix representing year dummy coded as cell means contrasts, a model matrix representing Dataset coded as sum to zero contrasts as well as varying effects representing the hierarchical structure of Sites nested within grid tiles (Figure 4.4). Weights were also applied to the grid tiles in order to allow the influence of each grid tile to be proportional to the relative coral reef extent of each grid tile represented.

Separate models were fitted to explore trends in hard coral, macroalgal, and turf algal cover. In each case, cover was modelled against a beta distribution (logit link). Cover values of either 0 or 1 were first shrunk by 0.01 for compatibility with the beta distribution. Weakly informative priors were applied to the beta shape parameters as well as the varying effects parameters and their standard deviations.

In order to impute missing year combinations and smooth over short-term oscillations in estimates resulting from short-term fluctuations in sampling designs and data availability, priors on Year effects (except for the first observed year of data in an ecoregion) were assigned weakly informative normal priors centred around the posterior of either previous Year (in the case of Years after the initial observed year) or after (in the case of Years prior to the initial observed year). For the initial observed Year, standard (zero centred) weakly informative priors were applied.

The Dataset effects were included to act as proxies for the many and varying ways that different datasets differed, including depth, sampling unit type (quadrats, transects, *etc.*), and observer experience. Weakly informative normal priors were applied to the Dataset effects.

The statistical models can be summarised as:

$$Y \sim \text{Beta}(\mu\phi, (1-\mu)\phi) \tag{3.1}$$

$$\text{logit}(\mu) = \beta_y \text{X}_y + \gamma_d \text{X}_d + \omega\gamma_t \text{Z}_t + \gamma_s \text{Z}_s + \gamma_r \text{Z}_r \tag{3.2}$$

$$\gamma_t = \sigma_t \, \text{Z}_t \tag{3.3}$$

$$\gamma_s = \sigma_s \, \text{Z}_s \tag{3.4}$$

$$\gamma_r = \sigma_r \, \text{Z}_r \tag{3.5}$$

$$\phi \sim \Gamma(0.01, 0.01) \tag{3.6}$$

$$\text{Z}_t, \text{Z}_s, \text{Z}_r \sim \mathcal{N}(0, 1) \tag{3.7}$$

$$\sigma_t, \sigma_s, \sigma_r \sim t(3, 0, 1) \tag{3.8}$$

$$\sigma_y \sim t(3, 0, 0.5) \tag{3.9}$$

$$\beta_y \sim \mathcal{N}(y, \sigma_y) \tag{3.10}$$

$$\beta_{F_y} \sim \mathcal{N}(O_y, 1) \tag{3.11}$$

$$y \begin{cases} y < I_y \sim \mathcal{N}(y_{y+1}, 0.1) \\ y = I_y \sim \mathcal{N}(\beta_{y+1}, 0.1) \\ y = D_y \sim \mathcal{N}(y_{y-1}, \sigma_y) \\ y = G_{ys} \sim \mathcal{N}\left(\frac{\beta_{y-1} + y_{y+1}}{2}, 0.1\right) \\ y = G_y \sim \mathcal{N}\left(\frac{y_{y-1} + y_{y+1}}{2}, 0.1\right) \\ y = G_{ye} \sim \mathcal{N}\left(\frac{y_{y-1} + \beta_{y+1}}{2}, 0.1\right) \\ y = F_y \sim \mathcal{N}(O_y, 0.1) \\ y > F_y \sim \mathcal{N}(y_{y-1}, 0.1) \end{cases} \tag{3.12}$$

$$\gamma_d \sim \mathcal{N}(0, 0.1) \tag{3.13}$$

where:

- Y is a vector of the cover of the benthic group (*e.g.* either hard coral, macroalgae, turf algae, *etc.*).
- β_y and X_y represent the effects of Year and cell-means (no intercept) Year model matrices respectively.
- β_{F_y} represents the effect of the final observed year.
- γ_d and X_d represent the effects of Dataset ID and the associated model matrix such that the β parameters apply to the average Dataset (rather than the first level).
- γ_d is mean centered to ensure it sums-to-zero so that the β parameters apply to the average Dataset (rather than the first level).
- γ_t, γ_s, and γ_r are the sum-to-zero varying effects.
- Z_t, Z_s, and Z_r represent the Grid_id, Site and Transect/Replicate codes respectively and are the grid-level weights.
- y represents year hyperparameters and y is a year iterator.
- F_y and O_y represents the final observed year and associated mean observed response within the ecoregion.
- I_y represents the first observed year within the ecoregion.
- D_y represents the years for which there are observed data.
- G_{ys}, G_y, and G_{ye} represent the years marking the start, middle and end of a temporal gap within the observed data for the ecoregion.
- Weakly informative priors (on the link scale) are applied to the σ hyperpriors as well as γ_d (dataset effects).
- A weakly informative prior is assigned to the β associated with the final observed year. The mean of this prior is based on the median of the raw data for the final year and the standard deviation of 1 on the logit scale is relatively large.
- Priors for all other β are derived from the latent year hyperparameters.
- Since y are hyperparameters on the expected β and are based on lags from surrounding estimates, the priors had narrower spread so as to offer some regularisation in the expected change in cover over a single year.
- Similarly, the prior on the sum-to-zero effect of Dataset ID (γ) was more informative since there were numerous incidences in which different Datasets did not overlap in space and/or time. Hence without less weak priors, there was a strong possibility of the models doubling up on trying to account for differences in Datasets, space and time resulting in highly inflated uncertainties.

Estimates of annual benthic cover thereafter consisted of inverse-link transformed β estimates for the years for which there were observed data and Y estimates were used to fill in the gaps (years for which there were no observed data). This methodology enabled us to generate benthic cover estimates that were unsmoothed for data years (thereby permitting rapid change) while still providing estimates of cover during the temporal gaps that transitioned between the bookends of the gaps and with uncertainty increasing distal to the bookends.

An alternate version in which annual benthic cover is derived from inverse-link transformed y values is also provisioned. As this version uses the generative year hyperpriors, it effectively applies running one-year smooth over the estimates.

All models were run with 10,000 no-u-turn iterations, a warmup of 5,000 and a thinning rate of 10 for each of three chains (each with random initial values). The adaptive delta was elevated to 0.99 to reduce the incidences of post-warmup divergences. Diagnostics indicated that all chains converged on stable, well mixed posteriors (Rhat values < 1.05) and low MCMC sample autocorrelation (< 0.2). The models were run in the *STAN* probabilistic language [591] via the *cmdstan* interface for R 4.6.0 [584].

Hierarchical aggregation

The full posteriors for the Year effects (on the logit scale) of each ecoregion were averaged together within each subregion (Figure 4.4). The resulting posteriors were then summarised by back-transforming to the response scale (inverse logit transform in the case of beta models) and calculating the medians and highest probability (density) credibility intervals (80% and 95%). Similarly, the unstandardised subregion posteriors were aggregated (with weights) up to region and then global level (Figure 4.4).

Contrasts and evidence levels

Contrasts (changes in hard coral cover) between specific or groups of years were achieved by multiplying a matrix of contrast coefficients (reflecting the specific contrasts) by the posteriors of the annual marginal estimates. The contrast coefficients for any single contrast summed to zero and the positive and negative coefficients summed to one and negative one, respectively. For instances where a contrast represents a group of years, the estimate represents the average over those years. Changes between years (or groups of years) were expressed as both absolute and relative (percentage) change by using posteriors on the response and log scales respectively. Evidence for increases or decreases in cover was quantified using Bayesian exceedance probabilities, calculated as the proportion of posterior samples greater than zero ($\Pr(\Delta > 0)$) or less than zero respectively. Exceedance probabilities were also mapped onto a categorical evidence scale as indicated in Table 4.5.

Data visualisation

While we compiled benthic cover data spanning 1973 to 2025 (Table A.4), modelled temporal trends are shown only for the 1980–2024 period. We chose not to display model outputs for 1973–1979 and 2025 due to the limited number of surveys available for those years (Figure B1.1C). Although these periods were associated with greater uncertainty, reflected in wider credible intervals, we considered the corresponding estimates insufficiently robust for visualisation. Nevertheless, the hierarchical Bayesian models were fitted using the full 1973–2025 dataset, thereby informing estimates across the 1980–2024 period.

▼ **Table 4.5.** Comparison of the four levels of evidence, which correspond to the confidence in whether there was a change in benthic cover between two periods, based on Bayesian exceedance probabilities.

Level	Probability threshold
Strong evidence	$\Pr(\Delta > 0) \leq 1.00$
Evidence	$\Pr(\Delta > 0) < 0.95$
Weak evidence	$\Pr(\Delta > 0) < 0.90$
No evidence	$\Pr(\Delta > 0) < 0.85$

Modelling future reef trajectories

To illustrate the effect of shortening recovery periods on the future trajectories of hard coral cover at global scales, historical rates of gain and loss of hard coral cover between 1980 and 2024 were used to parametrise a simple mathematical model and simulate future trajectories under different scenarios of varying recovery windows (*i.e.* the number of years between major disturbances).

An Ordinary Differential Equation (ODE) was used to define annual rates of growth and decline in hard coral cover (Eq. 3.16). To parametrise these rates of growth and decline, historical annual rates were extracted from posteriors of the statistical model described in the Statistical model section. Periods of decline were identified around the peaks of global bleaching events (1998, 2010, 2016, and 2024), and rates of decline (m) were estimated by calculating the relative change between hard coral cover estimation on the year before the event and two years after each event (divided by the number of years). This assessment acknowledges that global bleaching events occur over Austral and Boreal summers and have a legacy impact on hard coral cover. Growth rates (r) were estimated as the relative change in hard coral cover (divided by the number of years), based on the posteriors of the models, during periods of recovery between major disturbances (1984–1995, 1999–2003, 2014–2015, and 2017–2019).

$$G = rC\left(1 - \frac{C}{K}\right) \tag{3.14}$$

$$M = mC \tag{3.15}$$

$$\left(\frac{\delta C}{\delta t}\right) = G - M \tag{3.16}$$

where:

- C represents coral cover at a given time
- K represents a carrying capacity coefficient ($K = 0.50$)

To simulate future hard coral cover trajectories, the model was initialised at 26% hard coral cover and ran iteratively over 100 years using 1,000 simulations randomly drawing posteriors from the Bayesian model to parametrise rates of growth and decline. Four recovery windows were simulated to account for different periods of recovery (*i.e.* 1, 2, 4, and 10 years). During recovery periods, the mortality rate (m) was set to zero. Other years, when disturbances occurred, growth rate (r) was set to zero to simulate the annual rate of decline from global bleaching events.

CODE AVAILABILITY

All the scripts used to produce values, tables, figures, and maps presented in this report can be consulted at https://github.com/GCRMN/global_2025.

Google Earth Engine [592] was used for some data analysis scripts using the JavaScript language. Hierarchical Bayesian models were run on an RStudio Server while all other R scripts were run on a personal computer Dell Precision 7780 of 32 GB RAM, operated on Windows 11.

Most of the analyses were performed using R software version 4.6.0 [584] and predominantly packages *tidyverse* [585], *terra* [593], *tidyterra* [594], *patchwork* [595], and *sf* [587]. Versions of all R packages used are provided within the README of the GitHub repository (https://github.com/GCRMN/global_2025).

BOX 4

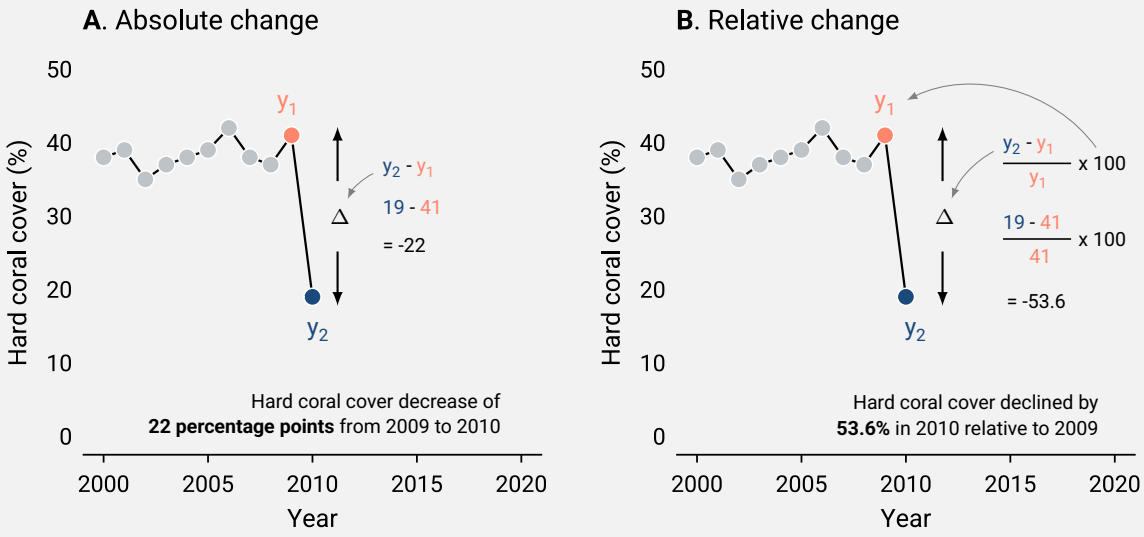
What is the difference between absolute and relative change in percentage cover?

Authors

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To understand how hard coral cover has changed between two years or two different time periods, it is necessary to quantify this change. Change in hard coral cover can be measured in two ways: as an ***absolute change*** or a ***relative change***. Absolute change is simply the difference in hard coral cover between two time points; whereas relative change expresses that same difference as a proportion of the starting value.



▲ **Figure B4.1.** Schematic representation of the difference between absolute (A) and relative change (B) in hard coral cover between two years.

For example, to calculate the change in hard coral cover between 2009 and 2010 where hard coral cover decreased from 41% of the benthos to 19%, we calculate the difference between the two values for an absolute change (decrease) in hard coral cover of 22 percentage points (Figure B4.1.A); the relative change is 53.6% (because 22 is just over half of the original 41%) (Figure B4.1.B). A relative change could also be expressed as a fold change, such that a 53.6% decline represents a halving in hard coral cover.

Since hard coral cover is itself expressed as a percentage, calculating relative change means taking a percentage of a percentage, and thus these two measures are easily conflated. However, absolute change and relative change describe different things and they should not be used interchangeably. Absolute change allows us to express the actual loss of hard coral cover between the two time points, whereas relative change allows us to compare changes between different coral reefs that may not have similar initial hard coral cover.

The differences between absolute and relative change values can be substantial. For example, a decrease in cover from 4% to 1% represents an absolute change of only -3 percentage points, however, this same decrease corresponds to a relative change of -75%, which is substantive.



GLOSSARY

Coral bleaching

A stress response in which corals expel the microscopic algae living inside their tissues. These algae give coral its colour and provide most of its food. This happens when corals are under stress due to an environmental disturbance. Coral bleaching is usually triggered by heat stress and UV radiation, but can also be caused by other factors, such as changes in water quality.

Coral reef

A biogenic marine ecosystem primarily built by hard corals through the accumulation of calcium carbonate structures. For the purposes of this report, the definition is limited to shallow coral reefs, which are located at depths of less than 30 m, although coral reef ecosystems may extend beyond this depth (*e.g.* mesophotic coral reefs).

Credible intervals

Intervals that represent the range within which a parameter value lies with a specified posterior probability (*e.g.* 95%), reflecting the uncertainty surrounding the estimate. A 95% credible interval means there is a 95% probability that the true value lies within the stated range, given the data and model. Wider intervals indicate greater uncertainty in the estimate. Credible intervals are used in Bayesian statistics and are conceptually similar to confidence intervals in frequentist statistics.

Crown-of-thorns starfish (CoTS)

Crown-of-thorns starfish are large marine invertebrates which feed on coral as adults. They occur naturally throughout the Indo-Pacific region, on coral reefs from the Red Sea to the west coast of the Americas. Genetic studies show there are at least four species of crown-of-thorns starfish. These are the North and South Indian Ocean species (*Acanthaster planci* and *Acanthaster mauritiensis*), a Red Sea species (*Acanthaster benziei*) and a Pacific species (*Acanthaster* cf. *solaris*). The starfish can grow up to 80 cm in diameter. Despite their large adult size, they are often difficult to find on a reef, preferring to hide under ledges or in the reef structure.

Crown-of-thorns starfish (CoTS) outbreak

CoTS are natural predators in the Indo-Pacific and Red Sea, but outbreaks occur when the population density of these invertebrates (*Acanthaster spp.*) reaches critical levels — typically defined as 15 or more starfish in a one-hectare area — so that they consume live coral tissue faster than the coral can grow, causing significant impact on coral reefs.

Crustose coralline algae (CCA)

Calcifying red algae (Rhodophyta), widespread in coastal marine habitats across the globe, typically forming pink, purple, or red encrusting veneers on substrates. In coral reefs, CCA are a key component of the ecosystem, acting as a natural cement

that binds and stabilises coral rubble, and inducing the settlement and metamorphosis of scleractinian coral larvae. In this report, CCA was excluded from both the macroalgae and turf algae benthic categories.

Degree heating weeks (DHWs)

A widely used tool for measuring accumulated heat stress in an area and predicting coral bleaching risk across tropical coral reef ecosystems. It is calculated by combining the intensity of daily temperature extremes with the total time during which daily temperatures exceed the bleaching threshold over the previous three months. Significant coral bleaching is predicted when DHW is above 4 °C-weeks, and coral mortality is expected above 8 °C-weeks.

El Niño

Refers to the extensive warming of the central and eastern tropical Pacific Ocean, which leads to a major shift in weather patterns across the Pacific. El Niño events typically occur every two to seven years.

El Niño Southern Oscillation (ENSO)

A term used to describe the oscillation between the El Niño phase and the La Niña, or opposite, phase.

Global bleaching event

A period of widespread, severe coral bleaching occurring across all major ocean basins — the Pacific, Atlantic, and Indian Oceans — affecting large portions of the world's coral reefs due to cumulative marine heat stress and rising global temperatures. Global bleaching events peaked in 1998, 2010, 2016, and 2024, although their impacts may have extended before or after these years, depending on the region.

Hard corals

Corals that produce a calcium carbonate skeleton and contribute to building the reef framework. Although primarily represented by species from the order Scleractinia (true hard corals), this category also includes reef-building species from the families Milleporidae (*e.g. Millepora complanata*) and Helioporidae (*e.g. Heliopora coerulea*). Soft corals also bleach, but they are not captured by the hard coral cover indicator used in this report.

Macroalgae

Large multicellular algae visible to the naked eye, including fleshy and calcified forms that occupy benthic substrates. For the purposes of this report, macroalgae includes algae of the classes Phaeophyceae, Florideophyceae, and Ulvophyceae, but excludes crustose coralline algae.

Marine heatwaves (MHWs)

A discrete period of prolonged anomalously warm water at a particular location. Quantitative definitions are based on ocean temperatures exceeding a fixed, seasonally varying, or cumulative threshold. Typically, MHWs are identified when temperatures exceed the 90th percentile of sea surface temperature observations for the same time of year over a 30-year reference period, for at least 5 consecutive days. MHWs are caused by a combination of local oceanic and atmospheric processes. The global annual count of MHW days has increased relative to the historical record, and this trend is projected to continue under climate change [596].

Reef extent

The distribution of shallow coral reefs that can generally be seen from satellites. Reef extent is estimated by data fusion approaches that extend shallow tropical coral

reefs into adjacent areas that were otherwise too deep or turbid to confidently assign a geomorphic zone or dominant benthic substrate class. Reef extent includes areas down to a maximum depth of about 30 m, though this is not globally consistent due to water optical properties [538].

Sea surface temperature (SST)

Defined as the temperature of the top few millimetres of the ocean. This temperature directly or indirectly affects the rates of all physical, chemical, and most biological processes occurring in the ocean. SST is globally monitored by sensors on satellites, buoys, ships, ocean reference stations, and other technologies.

Statistical evidence

The degree of certainty associated with a change in benthic cover between two years or groups of years, as estimated from the Bayesian model (see section Contrasts and evidence levels).

Turf algae

Assemblages of short, filamentous algae forming dense mats on reef surfaces (*e.g.* dead hard coral, coral rubble), typically less than a few centimetres in height.



ANNEXES

▼ **Table A.1.** Comparison of coral reef extent indicators across the 53 GCRMN subregions and associated regions. EAS = East Asian Seas, ETP = Eastern Tropical Pacific, RSGA = Red Sea and Gulf of Aden, WIO = Western Indian Ocean.

Region	Subregion	Coral reef extent (km^2)	Coral reef extent relative to the region (%)	Coral reef extent relative to the world (%)
Australia	All	42,283		16.94
Australia	Australia 1	175	0.42	0.07
Australia	Australia 2	915	2.16	0.37
Australia	Australia 3	1,694	4.01	0.68
Australia	Australia 4	2,952	6.98	1.18
Australia	Australia 5	34,503	81.60	13.82
Australia	Australia 6	1,918	4.53	0.77
Australia	Australia 7	126	0.30	0.05
Brazil	All	1,178		0.47
Brazil	Brazil 1	10	0.83	0.00
Brazil	Brazil 2	347	29.47	0.14
Brazil	Brazil 3	670	56.89	0.27
Brazil	Brazil 4	19	1.61	0.01
Brazil	Brazil 5	132	11.20	0.05
Caribbean	All	24,671		9.88
Caribbean	Caribbean 1	5,095	20.65	2.04
Caribbean	Caribbean 2	1,709	6.93	0.68
Caribbean	Caribbean 3	1,093	4.43	0.44
Caribbean	Caribbean 4	6,914	28.02	2.77
Caribbean	Caribbean 5	8,089	32.79	3.24
Caribbean	Caribbean 6	1,770	7.18	0.71
EAS	All	75,765		30.35
EAS	EAS 1	32,019	42.26	12.82
EAS	EAS 2	19,347	25.54	7.75
EAS	EAS 3	6,497	8.58	2.60
EAS	EAS 4	4,107	5.42	1.65
EAS	EAS 5	6,104	8.06	2.44
EAS	EAS 6	5,405	7.13	2.16
EAS	EAS 7	2,286	3.02	0.92
ETP	All	593		0.24
ETP	ETP 1	29	4.86	0.01
ETP	ETP 2	263	44.39	0.11
ETP	ETP 3	272	45.87	0.11
ETP	ETP 4	19	3.24	0.01
ETP	ETP 5	10	1.64	0.00
Pacific	All	65,350		26.17

▼ **Table A.1.** (*continued*). Comparison of coral reef extent indicators across the 53 GCRMN subregions and associated regions.

Region	Subregion	Coral reef extent (km^2)	Coral reef extent relative to the region (%)	Coral reef extent relative to the world (%)
Pacific	Pacific 1	6,299	9.64	2.52
Pacific	Pacific 2	20,409	31.23	8.17
Pacific	Pacific 3	17,749	27.16	7.11
Pacific	Pacific 4	4,077	6.24	1.63
Pacific	Pacific 5	7,980	12.21	3.20
Pacific	Pacific 6	2,258	3.45	0.90
Pacific	Pacific 7	6,577	10.06	2.63
ROPME	All	1,808		0.72
ROPME	ROPME 1	1,324	73.21	0.53
ROPME	ROPME 2	178	9.86	0.07
ROPME	ROPME 3	306	16.93	0.12
RSGA	All	12,590		5.04
RSGA	RSGA 1	3,676	29.19	1.47
RSGA	RSGA 2	3,351	26.62	1.34
RSGA	RSGA 3	4,702	37.35	1.88
RSGA	RSGA 4	861	6.84	0.34
South Asia	All	10,755		4.31
South Asia	South Asia 1	2,697	25.08	1.08
South Asia	South Asia 2	6,297	58.55	2.52
South Asia	South Asia 3	991	9.21	0.40
South Asia	South Asia 4	770	7.16	0.31
WIO	All	14,682		5.88
WIO	WIO 1	6,285	42.81	2.52
WIO	WIO 2	1,904	12.97	0.76
WIO	WIO 3	1,014	6.91	0.41
WIO	WIO 4	5,176	35.26	2.07
WIO	WIO 5	302	2.06	0.12
Global		249,677		

▼ **Table A.2.** Comparison of human population living within 5 km from coral reefs across the 53 GCRMN subregions and associated regions. EAS = East Asian Seas, ETP = Eastern Tropical Pacific, RSGA = Red Sea and Gulf of Aden, WIO = Western Indian Ocean.

Region	Subregion	Population (2020)	Absolute change in population between 2000 and 2020	Relative change in population between 2000 and 2020 (%)
Australia	All	198,593	-183	-0.09
Australia	Australia 1	2,607	7	0.25
Australia	Australia 2	381	0	0.00
Australia	Australia 3	1,988	0	-0.01
Australia	Australia 4	74,401	148	0.20
Australia	Australia 5	117,244	109	0.09
Australia	Australia 6	0	0	0.07
Australia	Australia 7	1,972	-447	-18.49
Brazil	All	3,582,576	954,162	36.30
Brazil	Brazil 1	3,370	1,318	64.21
Brazil	Brazil 2	1,936,021	627,424	47.95
Brazil	Brazil 3	1,704,525	351,055	25.94
Brazil	Brazil 4	0	0	0.00
Brazil	Brazil 5	9,004	2,060	29.66
Caribbean	All	15,683,270	4,756,532	43.53
Caribbean	Caribbean 1	393,788	106,580	37.11
Caribbean	Caribbean 2	1,364,715	165,580	13.81
Caribbean	Caribbean 3	794,328	225,181	39.56
Caribbean	Caribbean 4	1,362,140	651,019	91.55
Caribbean	Caribbean 5	10,078,803	3,284,771	48.35
Caribbean	Caribbean 6	1,689,496	323,401	23.67
EAS	All	54,662,176	15,949,020	41.20
EAS	EAS 1	30,053,097	7,438,165	32.89
EAS	EAS 2	4,432,443	1,937,365	77.65
EAS	EAS 3	5,992,605	2,407,768	67.17
EAS	EAS 4	8,029,951	2,725,238	51.37
EAS	EAS 5	2,354,825	868,507	58.43
EAS	EAS 6	2,033,491	429,454	26.77
EAS	EAS 7	1,925,105	161,566	9.16
ETP	All	284,565	145,256	104.27
ETP	ETP 1	48,300	36,935	324.98
ETP	ETP 2	215,588	102,246	90.21
ETP	ETP 3	20,677	6,075	41.61
ETP	ETP 4	0	0	94.13
ETP	ETP 5	0	0	0.00
Pacific	All	4,354,775	971,063	28.70
Pacific	Pacific 1	336,206	-11,300	-3.25
Pacific	Pacific 2	1,434,947	507,206	54.67
Pacific	Pacific 3	989,729	172,909	21.17
Pacific	Pacific 4	952,088	183,297	23.84
Pacific	Pacific 5	201,959	53,514	36.05
Pacific	Pacific 6	196,106	8,122	4.32

▼ **Table A.2.** (*continued*). Comparison of human population living within 5 km from coral reefs across the 53 GCRMN subregions and associated regions.

Region	Subregion	Population (2020)	Absolute change in population between 2000 and 2020	Relative change in population between 2000 and 2020 (%)
Pacific	Pacific 7	243,739	57,316	30.74
ROPME	All	3,381,313	2,284,078	208.17
ROPME	ROPME 1	3,241,731	2,279,628	236.94
ROPME	ROPME 2	128,479	4,451	3.59
ROPME	ROPME 3	11,103	-1	-0.01
RSGA	All	3,275,730	1,342,023	69.40
RSGA	RSGA 1	392,352	121,148	44.67
RSGA	RSGA 2	1,623,683	729,573	81.60
RSGA	RSGA 3	648,439	259,021	66.51
RSGA	RSGA 4	612,667	232,865	61.31
South Asia	All	7,712,887	1,378,431	21.76
South Asia	South Asia 1	0	0	0.00
South Asia	South Asia 2	654,174	329,443	101.45
South Asia	South Asia 3	6,566,582	1,024,097	18.48
South Asia	South Asia 4	492,131	24,892	5.33
WIO	All	10,502,290	4,800,830	84.20
WIO	WIO 1	7,370,443	3,561,990	93.53
WIO	WIO 2	105,138	25,468	31.97
WIO	WIO 3	759,424	102,778	15.65
WIO	WIO 4	2,199,460	1,088,510	97.98
WIO	WIO 5	67,824	22,084	48.28
Global		103,633,805	32,580,234	45.85

▼ **Table A.3.** Comparison of thermal stress indicators across the 53 GCRMN subregions and associated regions. EAS = East Asian Seas, ETP = Eastern Tropical Pacific, RSGA = Red Sea and Gulf of Aden, WIO = Western Indian Ocean.

Region	Subregion	Average change in SST from 1985 to 2025 (°C)	Warming rate (°C.decade^{-1})	Long-term SST average from 1985 to 2025 (°C)
Australia	All	0.78	0.20	26.41
Australia	Australia 1	0.75	0.19	27.6
Australia	Australia 2	0.53	0.13	23.01
Australia	Australia 3	0.66	0.17	27.52
Australia	Australia 4	0.58	0.14	28.26
Australia	Australia 5	0.81	0.20	26.33
Australia	Australia 6	0.84	0.21	26.53
Australia	Australia 7	0.57	0.14	22.07
Brazil	All	0.76	0.19	26.65
Brazil	Brazil 1	0.72	0.18	27.51
Brazil	Brazil 2	0.79	0.20	27.43
Brazil	Brazil 3	0.75	0.19	25.99
Brazil	Brazil 4	0.36	0.09	25.48
Brazil	Brazil 5	0.99	0.25	27.65
Caribbean	All	1.03	0.26	27.51
Caribbean	Caribbean 1	0.97	0.24	26.45
Caribbean	Caribbean 2	1.05	0.26	27.64
Caribbean	Caribbean 3	1.09	0.27	27.32
Caribbean	Caribbean 4	1.16	0.29	28.07
Caribbean	Caribbean 5	0.98	0.25	27.87
Caribbean	Caribbean 6	1.04	0.26	26.67
EAS	All	0.83	0.21	28.64
EAS	EAS 1	0.88	0.22	28.77
EAS	EAS 2	0.85	0.21	28.8
EAS	EAS 3	0.68	0.17	28.89
EAS	EAS 4	0.72	0.18	28.42
EAS	EAS 5	0.75	0.19	29.01
EAS	EAS 6	0.85	0.21	28.01
EAS	EAS 7	0.93	0.23	25.36
ETP	All	0.79	0.20	27.85
ETP	ETP 1	1.52	0.38	25.66
ETP	ETP 2	0.66	0.16	28.52
ETP	ETP 3	0.87	0.22	27.43
ETP	ETP 4	0.77	0.19	26.95
ETP	ETP 5	0.72	0.18	27.38
Pacific	All	0.88	0.22	27.71
Pacific	Pacific 1	0.95	0.24	29.02
Pacific	Pacific 2	1.05	0.26	28.79
Pacific	Pacific 3	0.92	0.23	26.48
Pacific	Pacific 4	1.05	0.26	24.93
Pacific	Pacific 5	0.72	0.18	28.77
Pacific	Pacific 6	0.65	0.16	28.58

▼ **Table A.3.** (*continued*). Comparison of thermal stress indicators across the 53 GCRMN subregions and associated regions.

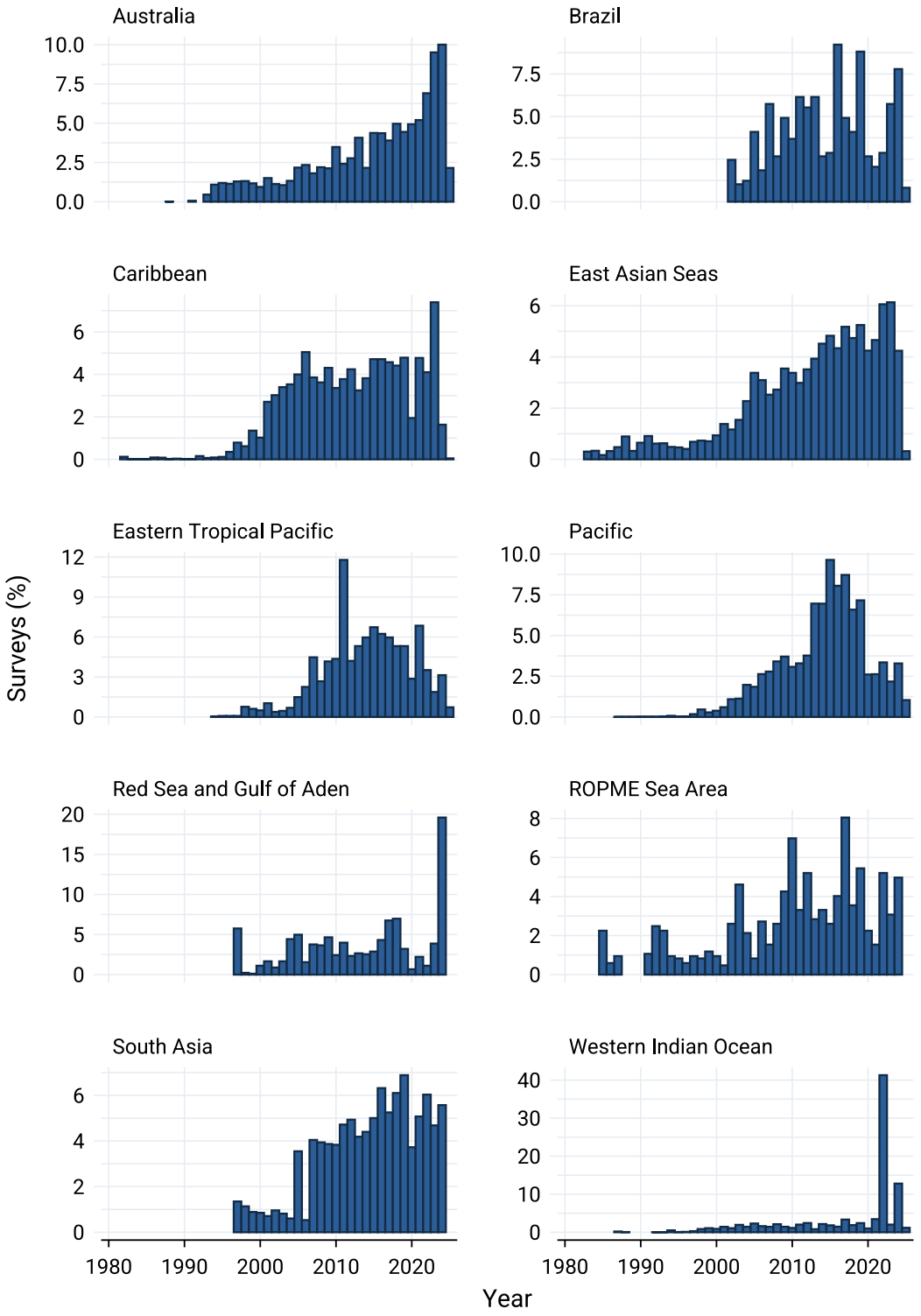
Region	Subregion	Average change in SST from 1985 to 2025 (°C)	Warming rate (°C.decade^{-1})	Long-term SST average from 1985 to 2025 (°C)
Pacific	Pacific 7	0.33	0.08	27.19
ROPME	All	0.5	0.13	26.76
ROPME	ROPME 1	0.50	0.12	26.81
ROPME	ROPME 2	0.89	0.22	27.62
ROPME	ROPME 3	0.65	0.16	25.96
RSGA	All	0.92	0.23	27.85
RSGA	RSGA 1	1.05	0.26	25.45
RSGA	RSGA 2	0.98	0.24	27.81
RSGA	RSGA 3	0.91	0.23	29.51
RSGA	RSGA 4	0.70	0.17	28.73
South Asia	All	0.79	0.20	28.58
South Asia	South Asia 1	0.85	0.21	28.28
South Asia	South Asia 2	0.74	0.19	28.88
South Asia	South Asia 3	0.76	0.19	27.66
South Asia	South Asia 4	1.03	0.26	28.25
WIO	All	0.74	0.18	27.19
WIO	WIO 1	0.78	0.20	27.34
WIO	WIO 2	0.82	0.20	27.46
WIO	WIO 3	0.94	0.24	26
WIO	WIO 4	0.65	0.16	27.14
WIO	WIO 5	0.51	0.13	26.52
Global		0.81	0.20	27.74

▼ **Table A.4.** Comparison of benthic cover monitoring indicators across the 53 GCRMN subregions and associated regions. A site is defined as a unique geographic location characterized by a specific latitude-longitude coordinate pair. A survey refers to a distinct monitoring event conducted at a specific site at a given point in time (date or year). EAS = East Asian Seas, ETP = Eastern Tropical Pacific, RSGA = Red Sea and Gulf of Aden, WIO = Western Indian Ocean.

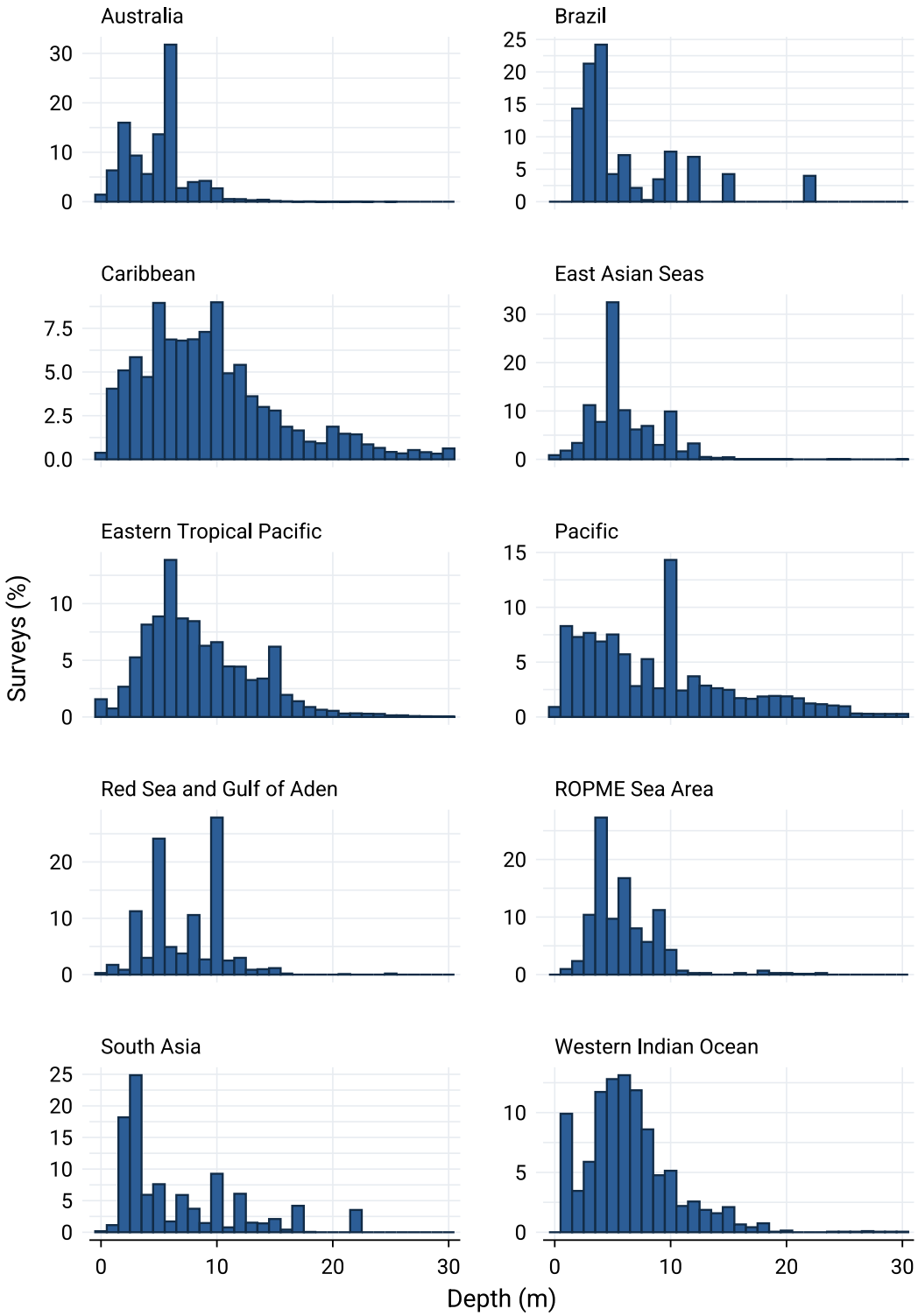
Region	Subregion	Datasets	Sites	Surveys	Range
Australia	All	14	2,645	13,211	1988-2025
Australia	Australia 1	3	52	169	1997-2023
Australia	Australia 2	4	197	1,198	1988-2024
Australia	Australia 3	5	374	1,399	1994-2024
Australia	Australia 4	2	92	159	2008-2024
Australia	Australia 5	8	1,547	9,438	1993-2025
Australia	Australia 6	3	275	554	2012-2024
Australia	Australia 7	3	108	294	2009-2025
Brazil	All	4	77	488	2002-2025
Brazil	Brazil 1	3	31	181	2002-2024
Brazil	Brazil 2	2	16	172	2003-2025
Brazil	Brazil 3	4	30	135	2002-2024
Caribbean	All	72	14,076	24,011	1973-2025
Caribbean	Caribbean 1	7	803	1,177	1982-2024
Caribbean	Caribbean 2	27	5,845	8,373	1982-2024
Caribbean	Caribbean 3	11	424	1,263	1973-2025
Caribbean	Caribbean 4	16	1,623	3,254	1985-2024
Caribbean	Caribbean 5	20	3,686	4,994	1982-2024
Caribbean	Caribbean 6	15	1,695	4,950	1984-2024
EAS	All	29	5,360	19,994	1983-2025
EAS	EAS 1	8	1,777	4,371	1986-2024
EAS	EAS 2	3	299	493	1997-2024
EAS	EAS 3	14	1,352	4,830	1986-2024
EAS	EAS 4	4	231	473	1987-2024
EAS	EAS 5	8	312	442	1988-2025
EAS	EAS 6	7	275	774	1997-2024
EAS	EAS 7	7	1,114	8,611	1983-2025
ETP	All	5	781	2,666	1994-2025
ETP	ETP 1	3	145	980	2009-2024
ETP	ETP 2	4	387	828	1998-2025
ETP	ETP 3	3	148	442	1998-2022
ETP	ETP 4	3	96	406	1994-2024
ETP	ETP 5	1	5	10	2024-2024
Pacific	All	56	8,677	16,706	1987-2025
Pacific	Pacific 1	7	1,404	2,448	1997-2024
Pacific	Pacific 2	10	384	683	1998-2025
Pacific	Pacific 3	29	2,257	6,079	1997-2025
Pacific	Pacific 4	5	2,065	2,451	1997-2021
Pacific	Pacific 5	5	311	338	2002-2020
Pacific	Pacific 6	13	1,811	2,216	2007-2024
Pacific	Pacific 7	11	445	2,491	1987-2024
ROPME	All	34	379	862	1985-2024

▼ **Table A.4.** (*continued*). Comparison of benthic cover monitoring indicators across the 53 GCRMN subregions and associated regions.

Region	Subregion	Datasets	Sites	Surveys	Range
ROPME	ROPME 1	28	291	630	1985-2024
ROPME	ROPME 2	14	73	217	2003-2024
ROPME	ROPME 3	2	15	15	2017-2020
RSGA	All	14	502	903	1997-2024
RSGA	RSGA 1	6	267	605	1997-2024
RSGA	RSGA 2	10	120	165	2004-2024
RSGA	RSGA 3	8	87	99	2000-2024
RSGA	RSGA 4	3	28	34	1999-2017
South Asia	All	15	636	2,825	1997-2024
South Asia	South Asia 1	3	63	224	2010-2023
South Asia	South Asia 2	10	477	989	1997-2024
South Asia	South Asia 3	3	94	1,610	2003-2024
South Asia	South Asia 4	1	2	2	2005-2006
WIO	All	48	3,753	5,633	1987-2025
WIO	WIO 1	25	448	1,050	1987-2025
WIO	WIO 2	4	125	244	1994-2024
WIO	WIO 3	3	2,905	3,343	1998-2025
WIO	WIO 4	15	196	692	1998-2025
WIO	WIO 5	9	79	304	1993-2024
Global		267	36,886	87,299	1973-2025



▲ **Figure A.1.** Percentage of benthic cover surveys conducted annually across the 10 GCRMN regions from 1980 to 2025, relative to all surveys conducted in each region during this period.



▲ **Figure A.2.** Percentage of benthic cover surveys by depth across the 10 GCRMN regions from 1980 to 2025, relative to all surveys conducted in each region between 0 and 30 m.



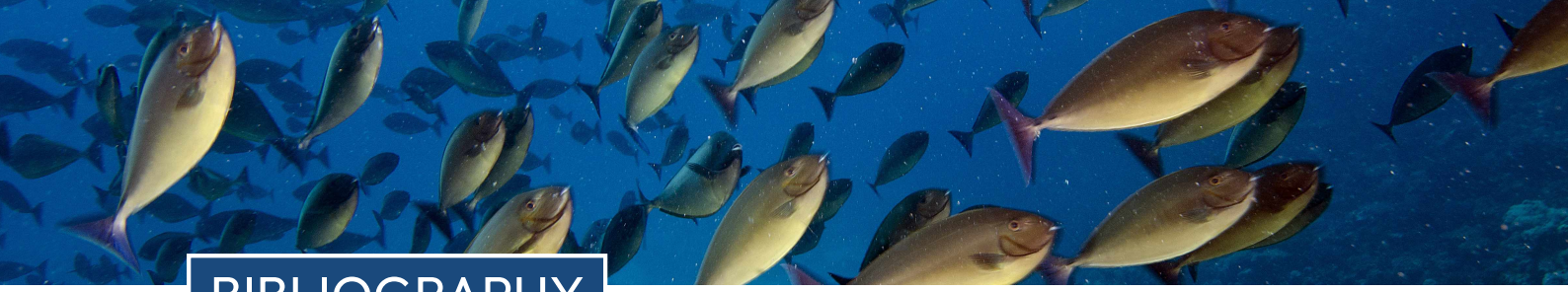
▲ **Figure A.3.** Comparison of the percentage of sites per number of years with monitoring data across the 10 GCRMN regions. A site is defined as a unique geographic location characterized by a specific latitude-longitude coordinate pair. A survey refers to a distinct monitoring event conducted at a specific site at a given point in time (date or year).

▼ **Table A.5.** List of individual datasets used to estimate benthic cover trends in the 10 GCRMN regions. EAS = East Asian Seas, ETP = Eastern Tropical Pacific, RSGA = Red Sea and Gulf of Aden, WIO = Western Indian Ocean. Each dataset is referenced by its name, with the corresponding dataset identifier provided in parentheses. Additional details are available in the gcrmndb_benthos GitHub repository.

Region	Datasets
Australia	AIMS (0003), Reef Check Foundation (0015), Reef Life Survey (0038), DBCA (0112), Tebbett et al, 2022 (0118), Kimberley Marine Parks (0134), Coral Sea Marine Park (0136), Mallela CI-CKI (0150), GBRMPA (0175), AIMS Western Australia (0182), CSIRO (0229), Citizens of the Reef (0248), RSRC_Heron_LTM (0268), Coral Reef Health - LHI and NI (0272)
Brazil	Reef Life Survey (0038), RCBrasil (0208), Cordeiro et al. (0209), Mies, 2024 (0233)
Caribbean	USVI - Yawzi and Tektite (0001), USVI - Random (0002), Reef Check Foundation (0015), Reef Life Survey (0038), Kayal and Dromard (0047), Ankay Conservation (0060), Reef Renewal Bonaire (0062), Nature Foundation SXM (0066), ODE Martinique (0067), Puntacana Foundation (0078), TCRMP (0079), FUNDEMAR (0080), UWI DBML (0081), Saba (0082), RNSM (0083), Curacao (0084), BREAM (0085), Fundacion Cap Cana (0086), Titè - ONF (0087), Grand Cayman (0088), Coral Cay Conservation (0089), Montilla et al, 2021 (0090), AGRRA 1.0 (0091), CZMU (0092), Puerto Rico CRMP (0093), CECIMAR (0094), CIMAR - UCR (0095), Bouchon and Bouchon-Navaro (0096), SECREMP (0097), Steneck (0098), Meesters et al. (0099), NCRMP - FGB (0101), NCRMP - Puerto Rico (0102), NCRMP - USVI (0103), NCRMP - Florida (0104), DTCREMP (0106), CREMP (0107), WLC (0108), Kemenes-Varadero (0109), INVEMAR (0110), NEPA (0111), Barcolab (0113), STENAPA (0114), CCMI (0115), SFS (0116), González Díaz et al. (0117), Alemu I (0130), Wilkinson et al, 2013 (0133), Garza et al, 2022 (0135), Hawkins and Roberts, 1995 (0138), Steneck and Torres (0145), STINAPA Bonaire (0146), USF (0147), Maréchal et al. (0148), Mallela 2007 (0149), pre-NCRMP (0151), Forrester (0152), Iberostar (0153), McField, 1997 (0154), McField, 1999 (0155), FGB LTMP - Random Transect (0163), FGB LTMP - Repetitive Quadrat (0164), McLeod et al. (0165), FUNDEMAR (2011-2016) (0166), Operation Wallacea (0176), Antonius and Weiner, 1982 (0178), Bright et al, 1984 (0179), Dodge et al, 1982 (0180), Edmunds and Bruno, 1996 (0181), Caballero-Aragón et al, 2022 (0235), AMMP (0267), AGRRA 2.0 (0270)
EAS	Reef Check Foundation (0015), Biosphere Foundation (0016), Reef Life Survey (0038), Sulubaaï (Shark Fin Bay project) (0077), Coral Cay Conservation (0089), Raghunathan and Mondal (0139), Operation Wallacea (0176), Voolstra et al. (0220), EAS 2025 - Brunei (0240), EAS 2025 - Cambodia (0241), EAS 2025 - China (0242), EAS 2025 - Myanmar (0244), EAS 2025 - Singapore (0245), EAS 2025 - Thailand (0246), EAS 2025 - Vietnam (0247), EAS 2020 - Hong Kong (0250), EAS 2025 - Hong Kong (0251), EAS 2020 - Taiwan (0252), EAS 2025 - Taiwan (0253), EAS 2020 - Brunei (0256), EAS 2020 - Cambodia (0257), EAS 2020 - Singapore (0260), EAS 2020 - Thailand (0261), EAS 2020 - Vietnam (0262), EAS 2020 - Japan (0263), EAS 2020 - Malaysia (0265), EAS 2025 - Japan (0269), EAS 2025 - Malaysia (0271), ASEAN - AUS (0274)
ETP	Reef Check Foundation (0015), Reef Life Survey (0038), ETP regional dataset (0168), Moity (0169), Innoceana (0187)

▼ **Table A.5.** (*continued*). List of individual datasets used to estimate benthic cover trends in the 10 GCRMN regions.

Region	Datasets
Pacific	CRIOBE - MPA (0004), CRIOBE - Polynesia Mana (0005), CRIOBE - Tiahura (0006), CRIOBE - ATPP barrier reef (0007), CRIOBE - ATPP outer slope (0008), 2013-2014_Koro Island, Fiji (0010), NCRMP - American Samoa (0011), NCRMP - CNMI and Guam (0012), NCRMP - Hawaii (0013), NCRMP - PRIA (0014), Reef Check Foundation (0015), Biosphere Foundation (0016), KNS (0017), Kiribati (0018), SLN (0019), PACN (0020), RORC NC (0021), MCRMP (0022), PA-NC (0023), Laurent WANTIEZ (0024), 2011_Southern Bua (0025), 2012_Western Bua (0026), 2009-2011_Kubulau (0027), C2O Pacific (a) (0028), Kimbe Bay (0029), PNG BAF 2019 (0030), 2017_Northern Lau (0031), 2013-2014_Vatu-i-Ra (0032), 2019_Dama Bureta (0033), 2020_NamenaAndVatuira (0034), Lau Seascape Surveys (0035), SI_Munda (0036), Khen et al, 2022 (0037), Reef Life Survey (0038), MMR (0039), Smallhorn-West et al, 2019 (0040), Hydro-Paalo (0041), Living Ocean Foundation (0042), 100 Island Challenge (0043), PICRC (0044), SRMR and Combe Reef (0045), 2023-24 Fiji GCRMN sites (0046), Kayal, Penin, and Adjeroud (NC) (0048), Kayal, Penin, and Adjeroud (Mo.) (0049), FEO (0050), Phoenix Islands (0051), Vava'u Ocean Initiative 2017 (0052), Vava'u Ocean Initiative 2022 (0053), 100 Island Challenge (SLI) (0054), Samoa Ocean Strategy (0055), C2O Pacific (b) (0061), Coral Sea Marine Park (0136), NGS - Solomon Islands (0237), NGS - Palau (0238), Smallhorn-West et al, 2025 (0239), Sea Women of Melanesia (0249)
ROPME	Reef Check Foundation (0015), SBN_UAE_2023 (0056), Kuwait_2014 (0058), Bahrain_2011 (0059), Qatar_2015-2017 (0063), UAE_Musandam (0064), SBNvsKF_UAE_2021-2022 (0065), AlHiel_UAE_2023 (0068), UAE_Musandam_2022 (0069), Claereboudt, 2015 (0070), Aeby et al, 2022 (0071), Al Mealla, 2022 (0072), Howells et al, 2020 (0073), Shokri, 2021 (0074), Aeby et al, 2020 (0075), KFUPM - Saudi Aramco (0076), Abu Dhabi 2017-2019 (0100), UAE_Oman_2008-2012 (0105), UAE_2019_Mateos (0119), Oman_2017_2020 (0120), Qatar_2014 (0137), UAE_2006-2014 (0144), Al-Abdulkader et al, 2019 (0156), Vogt, 1994 (0157), Leduc (0167), Benzoni et al, 2006 (0171), Vousden, 1995 (0172), Ghazilou, 2024 (0221), Leduc, 2024 (0222), Qatar, 2022 (0223), Al Harthi, 2024 (0224), Al Mealla, 2024 (0225), Kuwait_1987 (0226), UAE_2023-2024 (0227)
RSGA	Reef Check Foundation (0015), Reef Life Survey (0038), AIDE (0128), CORDIO (Comoros) (0129), Attalla, 2024 (0173), One Ocean LLC (0174), Al-Tawaha (0177), SHAMS (0183), Saad et al. (0184), Ali et al. 2017 and 2018 (0216), Elamin et al. 2018 (a) (0217), Elamin et al. 2018 (b) (0218), Abaker 2021 (0219), RedSea_WIO_Oman_2024_bleach (0232)
South Asia	Reef Check Foundation (0015), Reef Life Survey (0038), Montefalcone et al. (0057), Maldives Resilient Reefs (0131), Maldives Resilient Reefs (RC) (0132), Fairoz (0140), SDMRI (0141), Benkwitt et al. (a) (0142), Benkwitt et al. (b) (0143), Sannassy Pilly et al, 2024 (0170), Rajan et al. (0194), MMRI (0203), Nair et al. (1998-2023) (0210), Nair et al. (2007-2020) (0211), Nair et al. (2024) (0212)
WIO	Reef Check Foundation (0015), Reef Life Survey (0038), CORDIO (Kenya) (0121), A Rocha Kenya (0122), REEFolution (0123), KMFRI (0124), WRTI (0125), Dahari (0126), Moheli MPA (0127), AIDE (0128), CORDIO (Comoros) (0129), Bloomberg Coral Bleaching (0158), Bahari ni Urithi (0159), BAF (0160), Vibrant Ocean Initiative 2022 (0161), GFCR Survey 2022 (0162), Operation Wallacea (0176), Sh Aba et al. (0185), ORI (0186), IMS (0188), CORDIO (Tanzania) (0189), CHICOP (0190), SUZA (0191), Elma (0192), Under the Wave (0193), CNRO (0195), Blue Ventures (0196), PRÎSM (0197), YSO (0198), IHSM (0199), WWF Madagascar (0200), CORDIO (Madagascar) (0201), KORAI (0202), Memba-Mossuril Baseline (0204), PSESPA Monitoring2WWF (0205), Inhambane LMMAs (0206), Vamizi Island Monitoring (0207), PSESPA 2019 (0213), CTV (0214), BECA (0215), Likhulu (0228), UTOPIAN (0230), GCRMN Réunion (0231), RedSea_WIO_Oman_2024_bleach (0232), Graham et al, 2024 (0234), Love the Oceans (0236), Bee Ecological (0266), PNMM (0273)



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