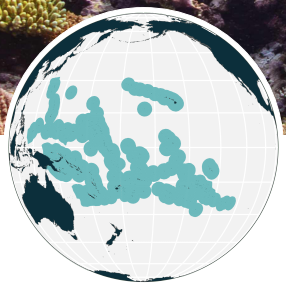
Coral reef in Ouvéa, New Caledonia
Credit: Sylvain Corbel



PACIFIC

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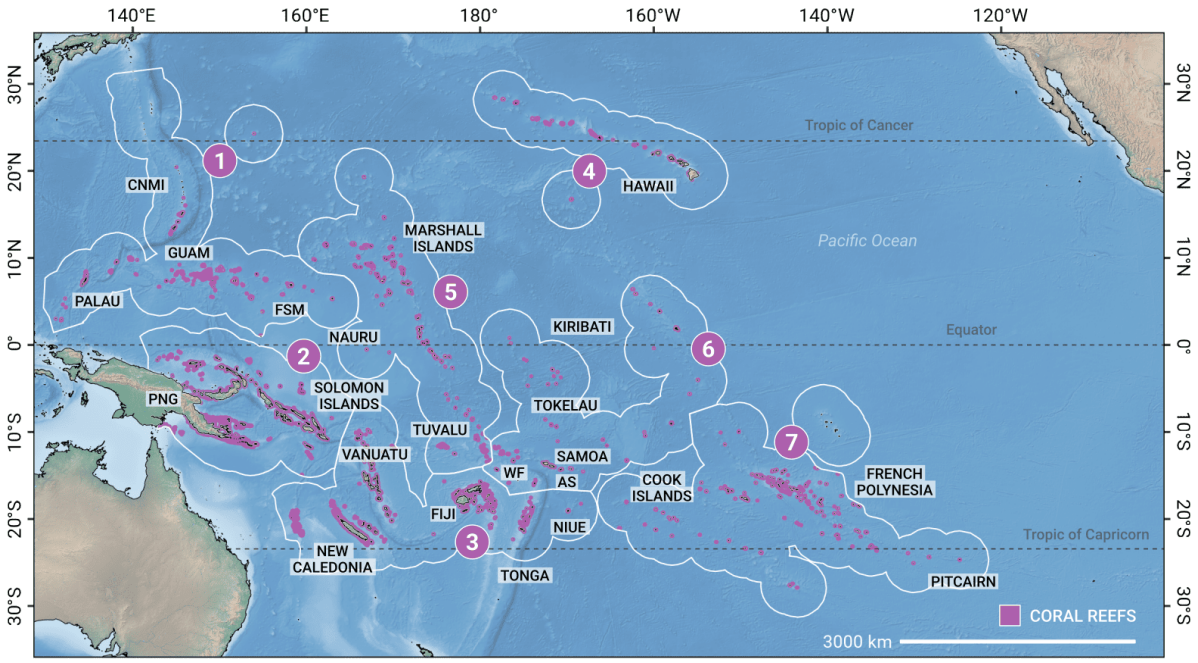
INTRODUCTION

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.

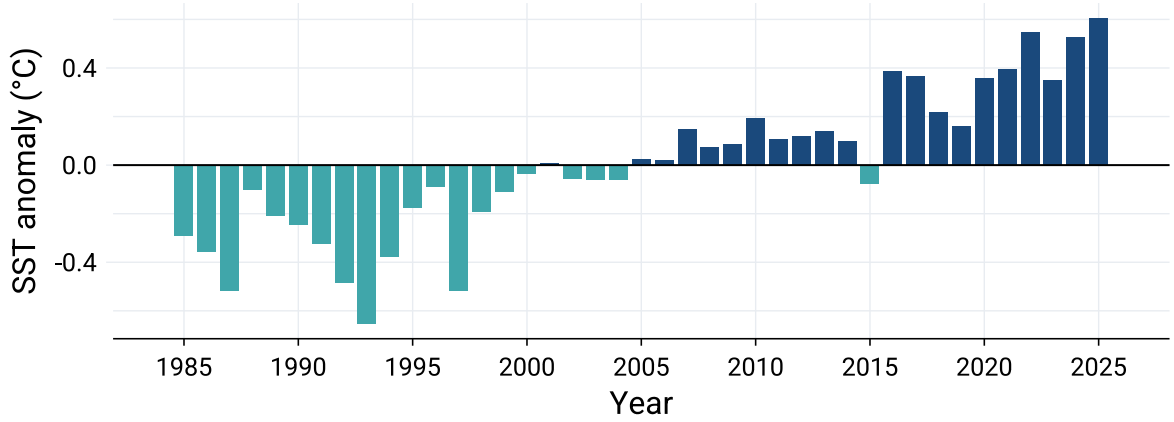
Compared to other GCRMN regions, coral reefs throughout the Pacific generally develop in clear, low-turbidity waters, as much of the region is composed of small islands with limited river catchments, reducing terrigenous sediments and nutrient runoff. Hard coral species richness exhibits a pronounced 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]).

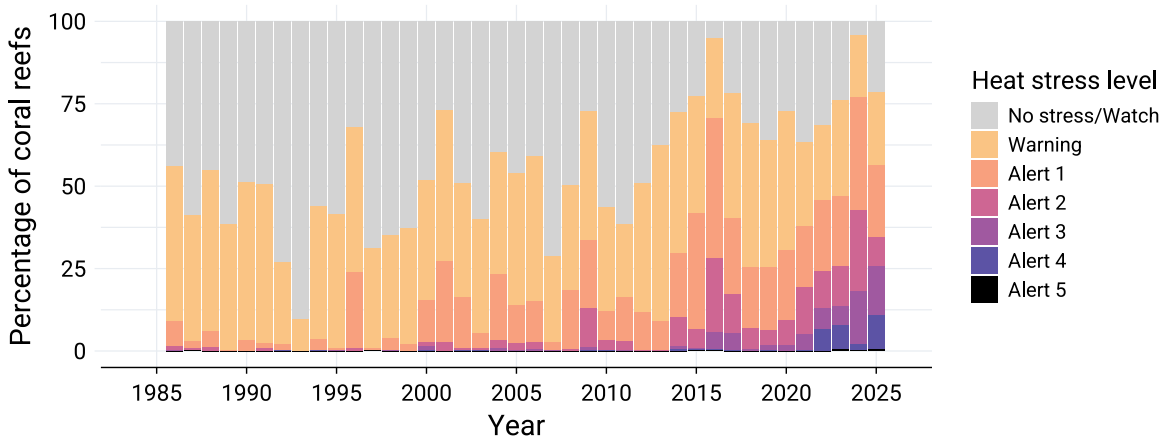
THREATS AND DISTURBANCES

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.

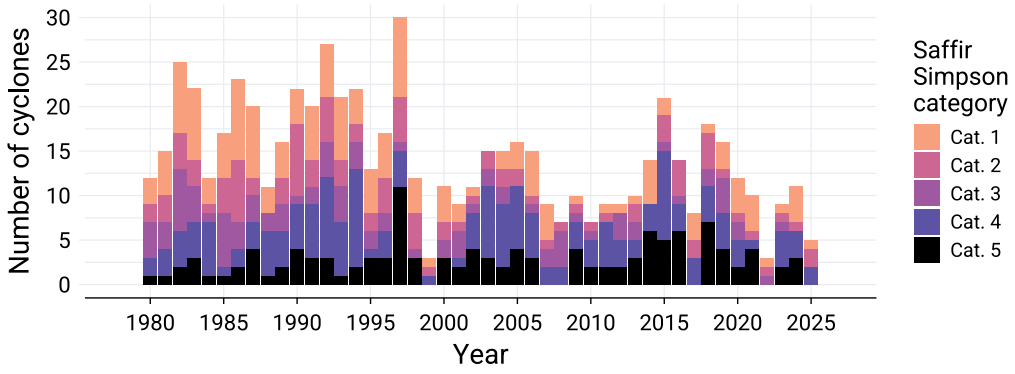
Cyclones

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

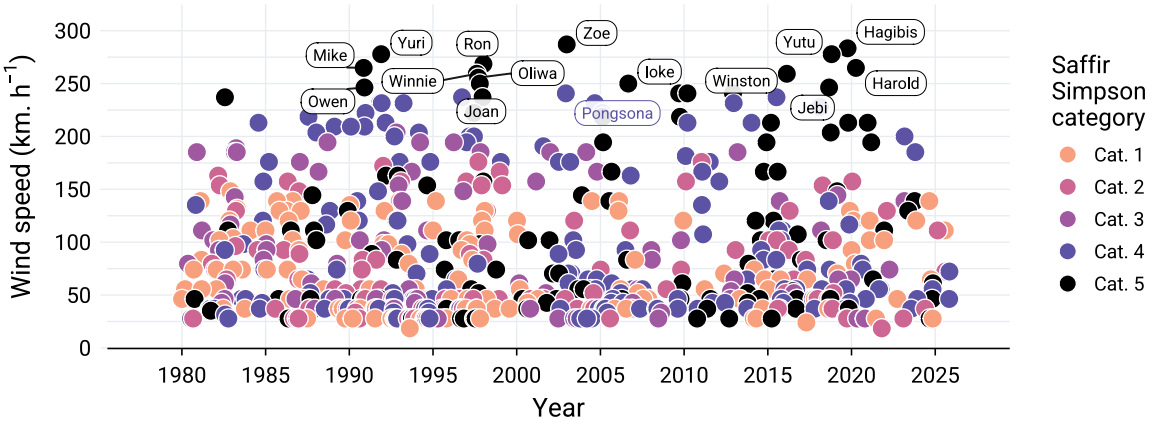


▲ **Figure 2.6.4.** Number of cyclones that passed within 100 km of the coral reefs of the Pacific region from 1980 to 2025. Colors correspond to the Saffir–Simpson category of the cyclone along its entire track. Derived from IBTrACS data (Knapp *et al.* [118]; see Materials and Methods).

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

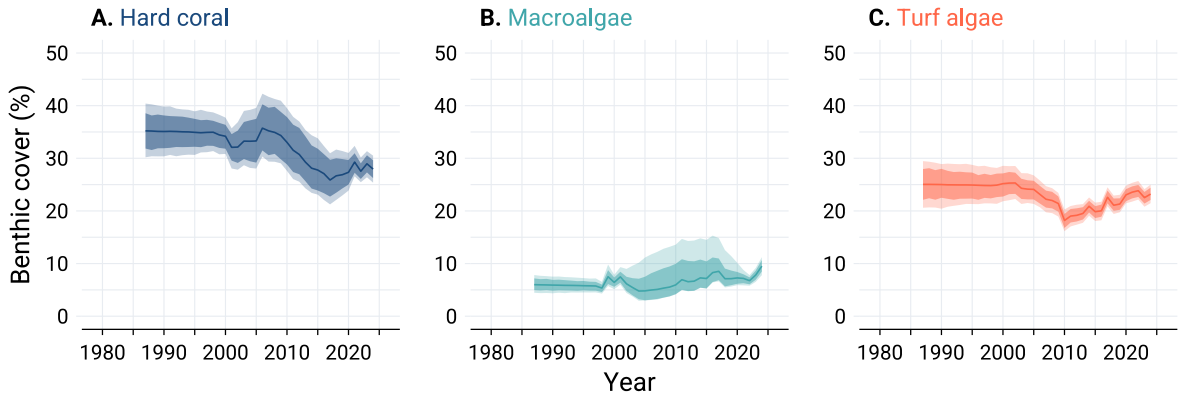
Pacific coral reefs are also impacted by tsunamis, which can generate physical disturbances comparable to those caused by cyclones. For example, the eruption of the Hunga Tonga–Hunga Ha'apai volcano in Tonga [331] triggered a tsunami that caused widespread hard coral loss across much of the country [332].

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-

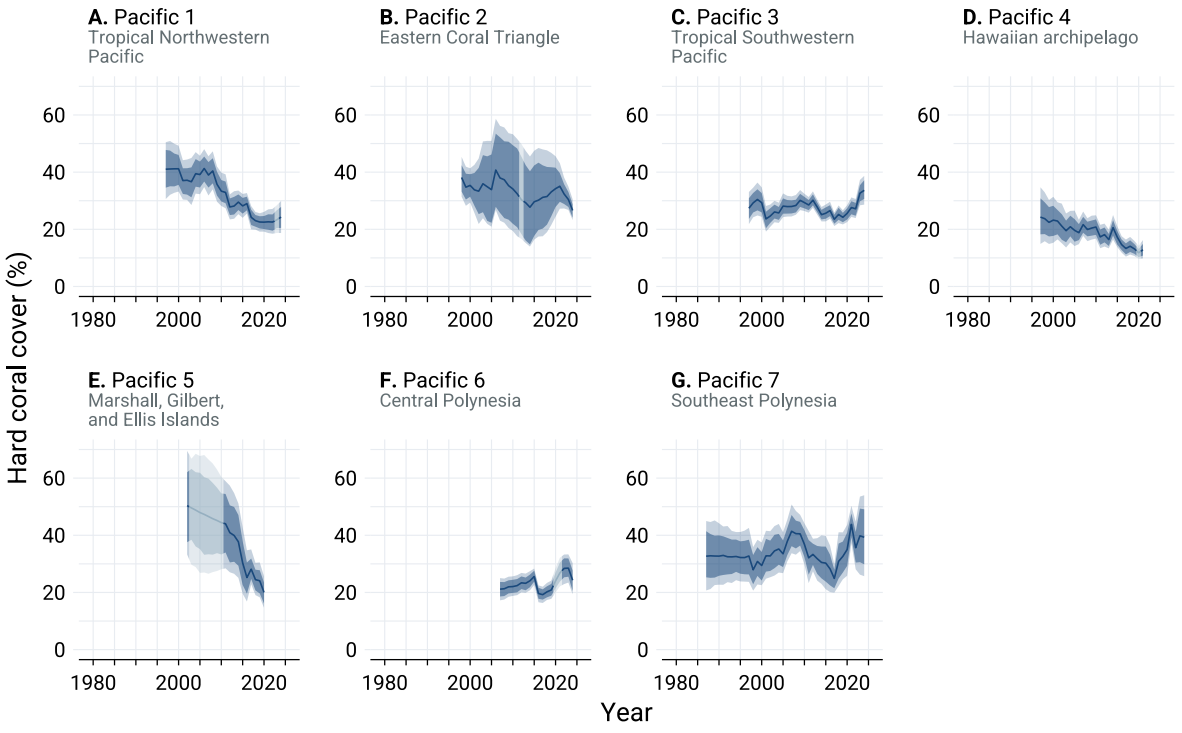
crease in macroalgal cover are likely multifactorial [337], including reduced hard coral cover that creates available substrate for algal colonisation, overfishing of herbivorous fish, and increased nutrient inputs in coastal waters that promote algal growth.

Turf algal cover

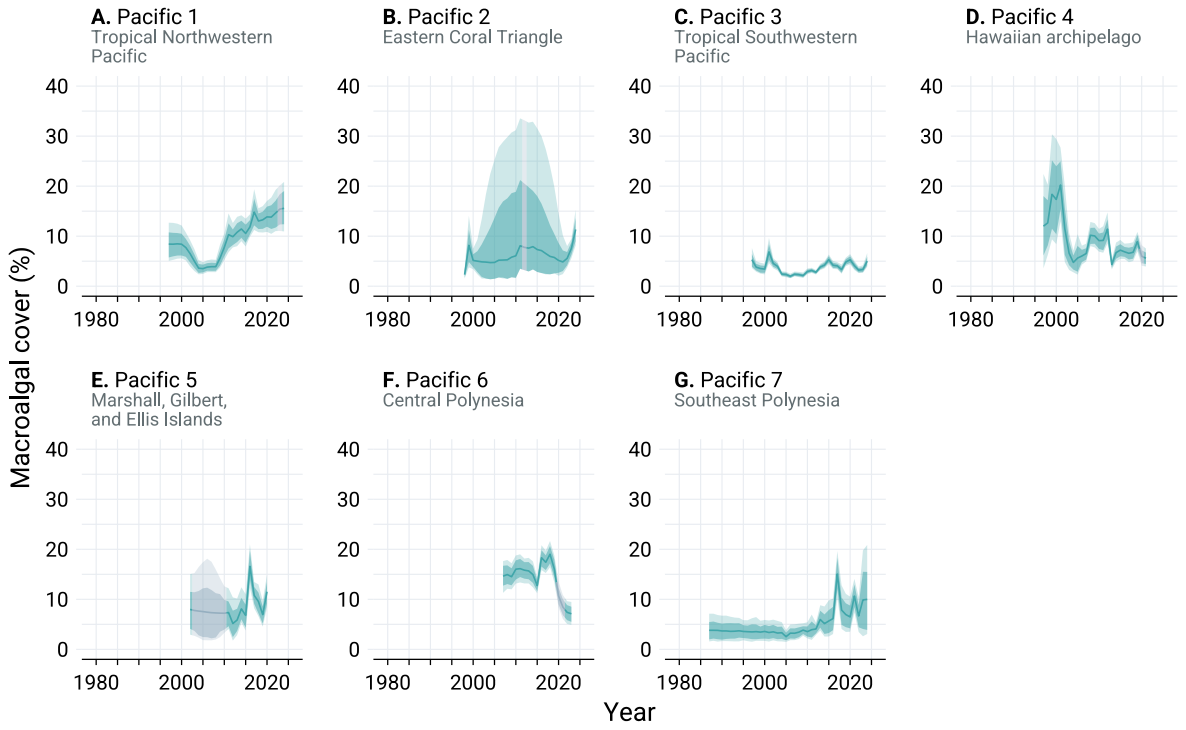
Turf algal cover showed no evidence of change between 1987–2009 and 2020–2024 (Figure 2.6.6C). In contrast, the 2011–2024 period was characterized by an increasing trend, with turf algal cover reaching 23.2% in 2024, coinciding with the decline in hard coral cover over the same period. Estimated values 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).

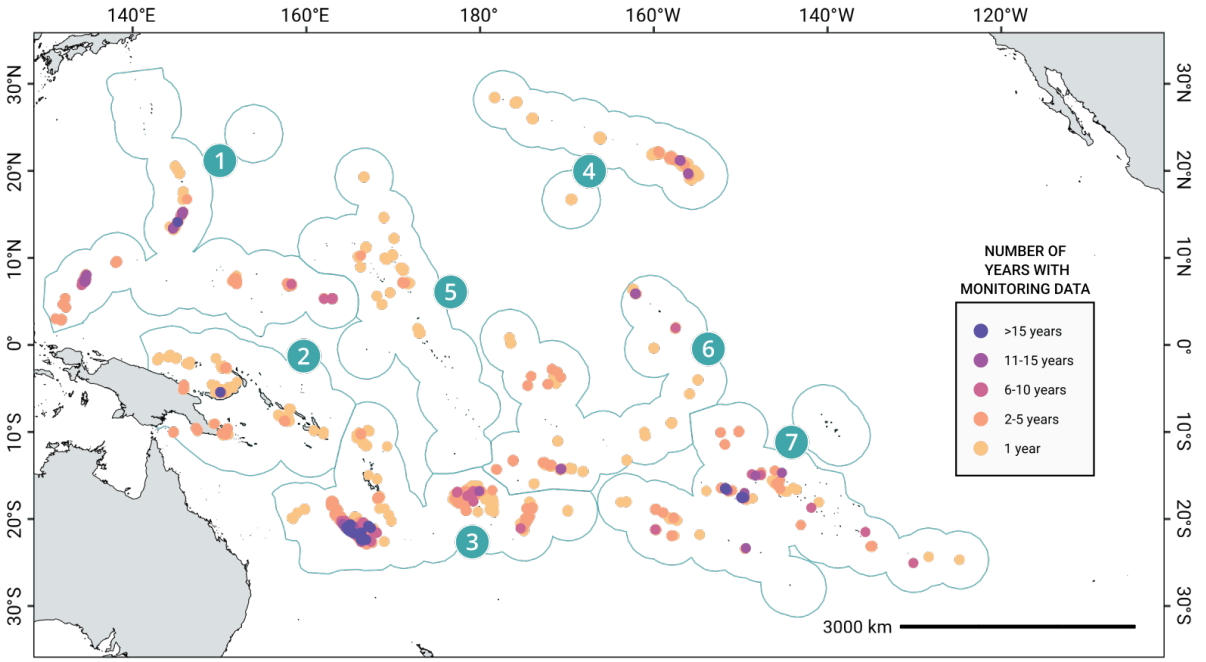
CONCLUSIONS

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

Mehdi Adjeroud, Abigail Alling, Phoebe Argyle, Gou Ava, Mary Bonin, Jonathan Booth, Tracey Boslogo, Sara Cannon, Yannick Chancerelle, Liam Clegg, Anne Cohen, Kitty Currier, Orla Doherty, Simon Donner, Janelle Eagle, Graham J. Edgar, Andy Estep, Viliami Fatongiatau, Martin Finau, Michael Fox, Jan Freiwald, Beverly French, Antoine Gilbert, Nicolas Guillemot, Tevita Havea, Tom Heintz, Andrew Hoey, Eryn Hooper, Peter Houk, Alec Hughes, Sandrine Job, Johanna Johnson, Jacinta Jonathan, Geoffrey Jones, Stacy Jupiter, Malnoa Karo, Mohsen Kayal, Adi Khen, Kelly Kozar, Uali Kula, July Kuri, Didier Labrousse, Florian Le Bail, Andy Lewis, Enelio Liufau, Naomi Longa, Sosefo Malau, Sangeeta Mangubhai, Ateliana Maugateau, Juan Mayorga, Mark McCormick, Sheila McKenna, Jenny Mihaly, Carol Milner, Bradley Moore, Kirby Morejohn, Elizah Nagombi, Stephen Neale, Victor Nestor, Mike Neuman, Poasi Ngaluafe, David Obura, Elizabeth Oh, Selma Pamalok, Michael Parrish, Nicole Pedersen, Lucie Penin, Alain Pidjot, Serge Planes, Tiana Reimann, Geraldine Rengiil, Randi Rotjan, Stuart Sandin, Annisah Sapul, Gilles Siu, Patrick Smallhorn-West, Jennifer Smith, Lucy Southworth, Maya Srinivasan, Karen Stone, Rick D. Stuart-Smith, Heather Summers, Helen Sykes, Elizabeth Thompson, Erica K. Towle, Visesio Uluika, Laurent Wantiez, Jane Waterhouse, David Welch, Yvonne Wong, Lindsay Young, Ute Zischka, Schannel Van Dijken.

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

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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.

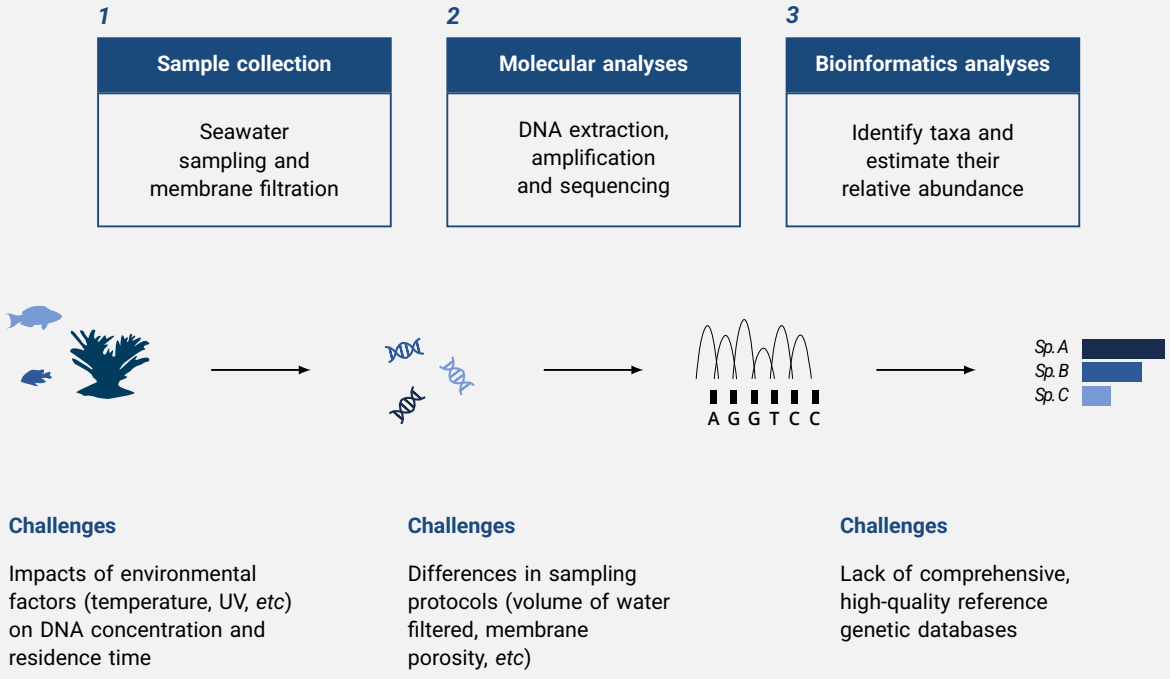
However, deploying eDNA at scale is not without substantial challenges (Figure C6.1). Three key issues currently limit its widespread implementation:

- **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

Author

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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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