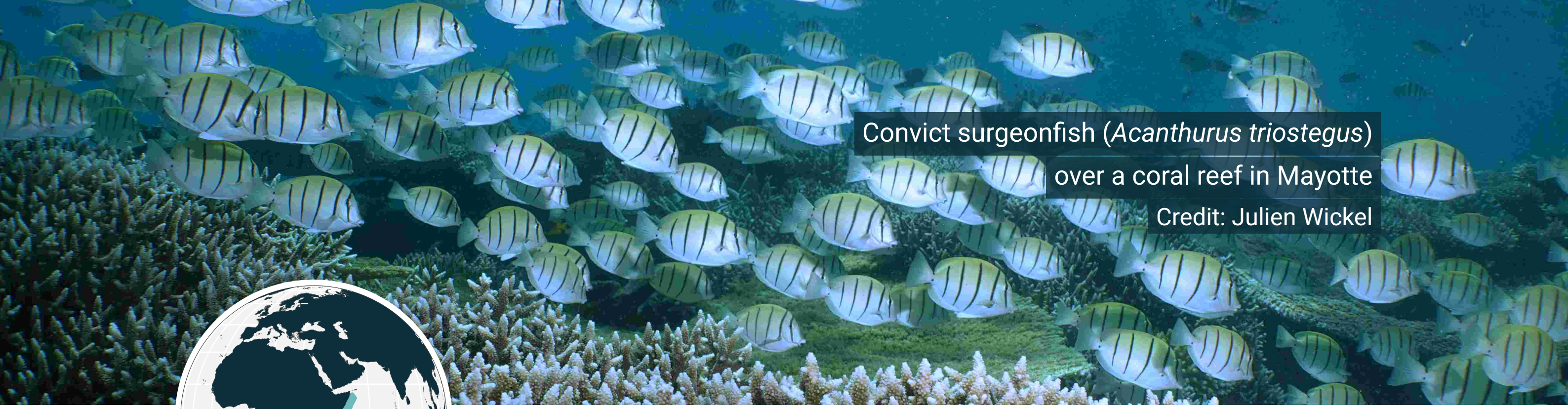

Convict surgeonfish (*Acanthurus triostegus*) over a coral reef in Mayotte
Credit: Julien Wickel

WESTERN INDIAN OCEAN

Authors

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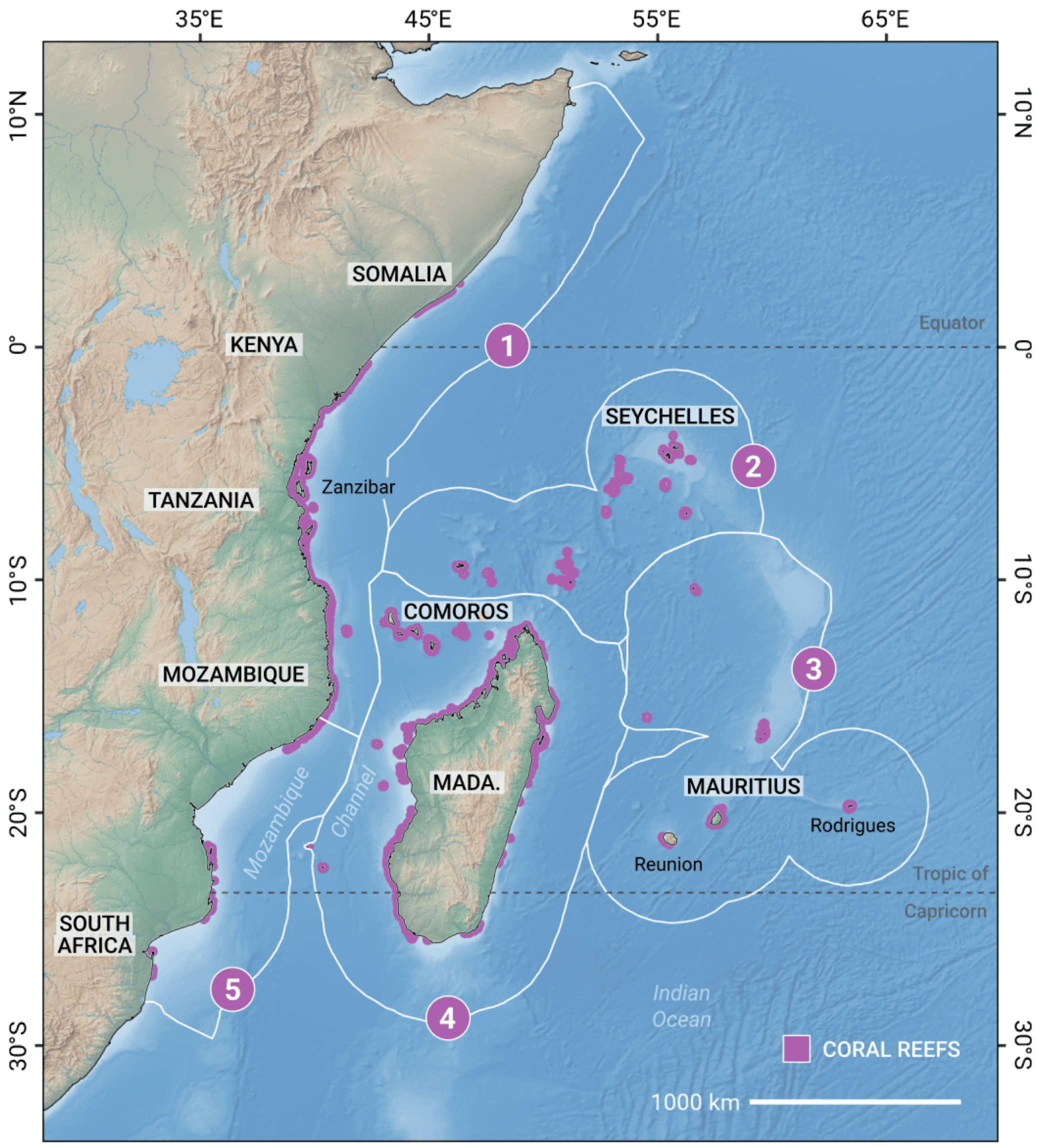
INTRODUCTION

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].

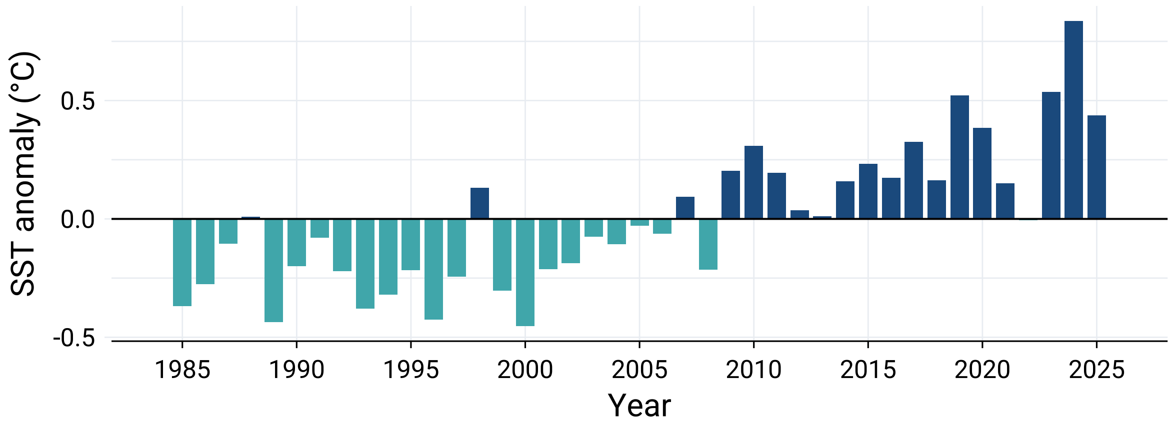
THREATS AND DISTURBANCES

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

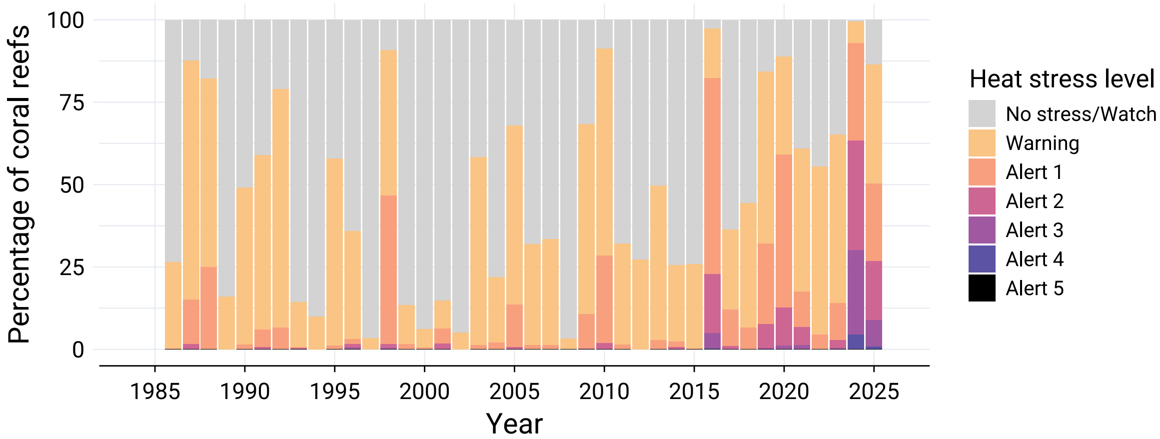
From 1980 to 2025, a total of 151 cyclones passed within 100 km of coral reefs in the Western Indian Ocean, averaging 3.4 $\pm$ 2 SD cyclones per year (Figure 2.10.4).

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, $0 < \text{DHW} < 4$); 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).

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].

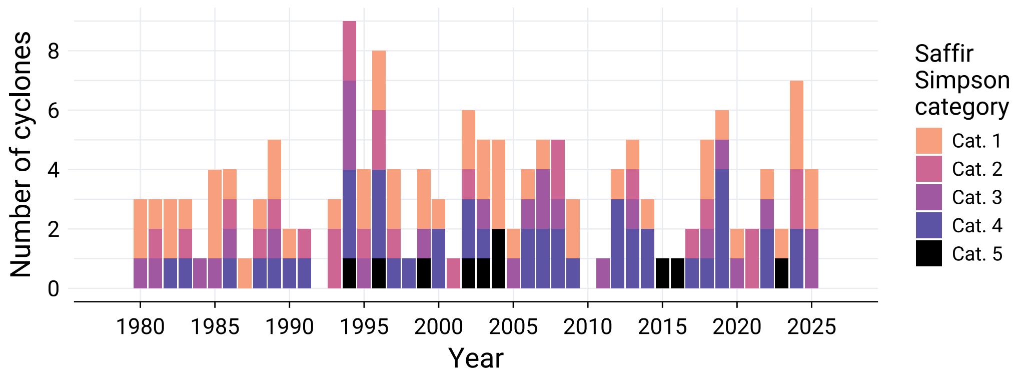
Coral disease and growth anomalies have also been reported with increasing frequency across the region, particularly following major disturbances [464]. As intervals between disturbance events shorten, these stressors act synergistically with climate-driven pressures, compounding cumulative impacts on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

STATUS OF CORAL REEFS IN THE REGION

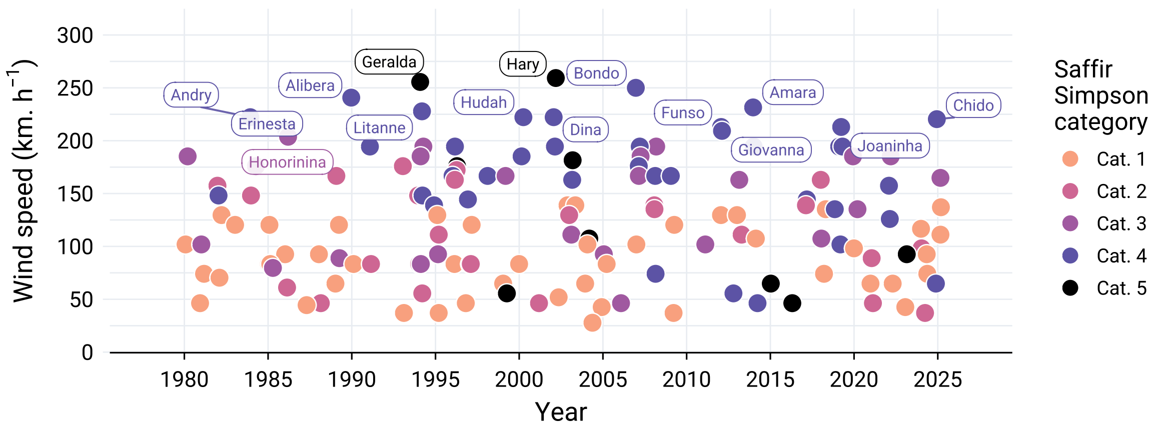
Hard coral cover

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% [$\text{CrI}_{95\%}$]: 30.1 – 37.4%) in 2023, before declining sharply to 25.9% ($\text{CrI}_{95\%}$: 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.

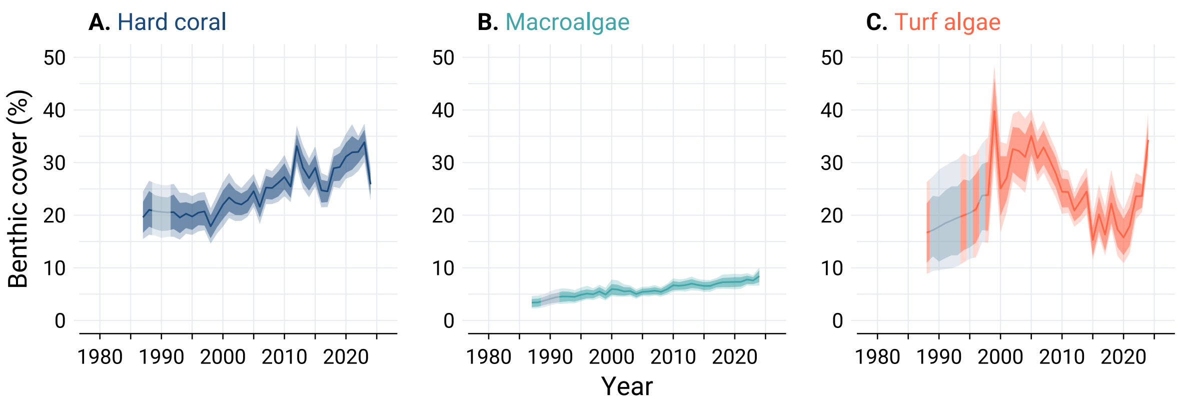
The 1998 global bleaching event, driven by strong positive phases of the El Niño Southern Oscillation and Indian Ocean Dipole, caused severe coral mortality across the region, with published average regional decline from ~40% to 30% hard coral cover, representing a 25% reduction [466]. The modelled data 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 approximately 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

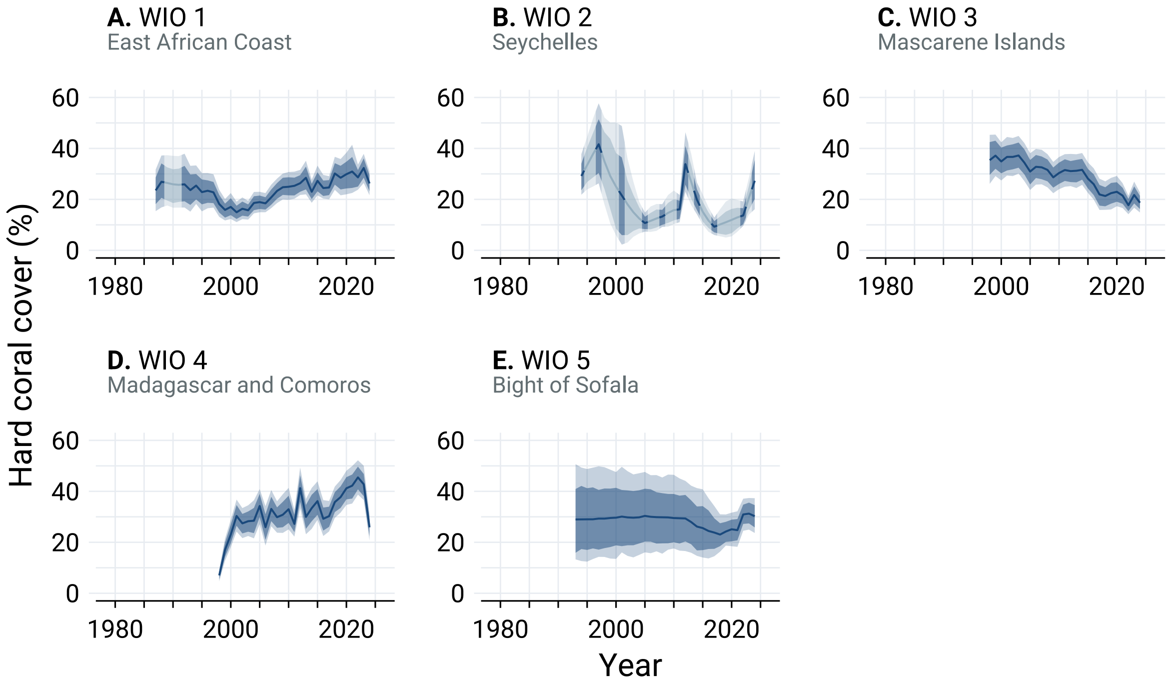
The three benthic indicators — hard coral, macroalgal, and turf algal cover — reveal dynamic and interconnected responses to disturbance, though longer-term trajectories should be interpreted with caution due to the nature of the dataset in the study. The modelled post-1998 recovery of hard coral cover 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

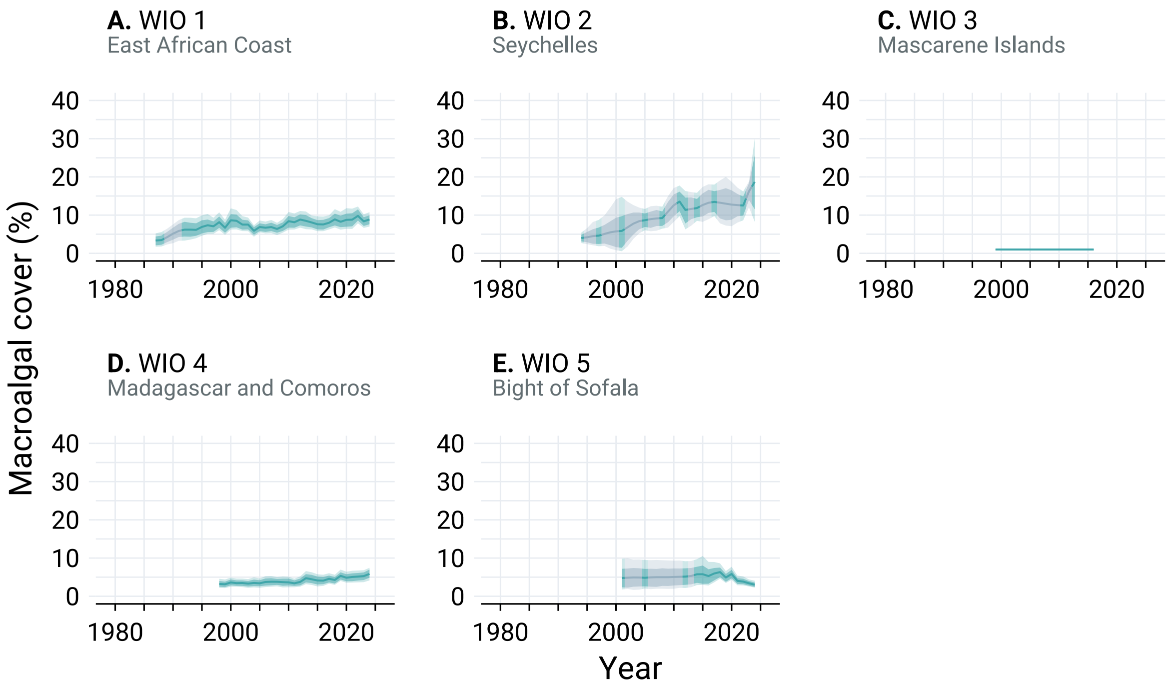
The Seychelles experienced severe hard coral cover loss post-1998, declining from 41.7% ($CrI_{95\%}$: 26.3 – 57.7%) in 1997 to 10.7% ($CrI_{95\%}$: 7.4 – 14.8%) by 2005 (Figure 2.10.7B). Coral recovery was slow, with no evidence of change in hard coral cover between the 2000s and the 2010s. Reefs retaining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

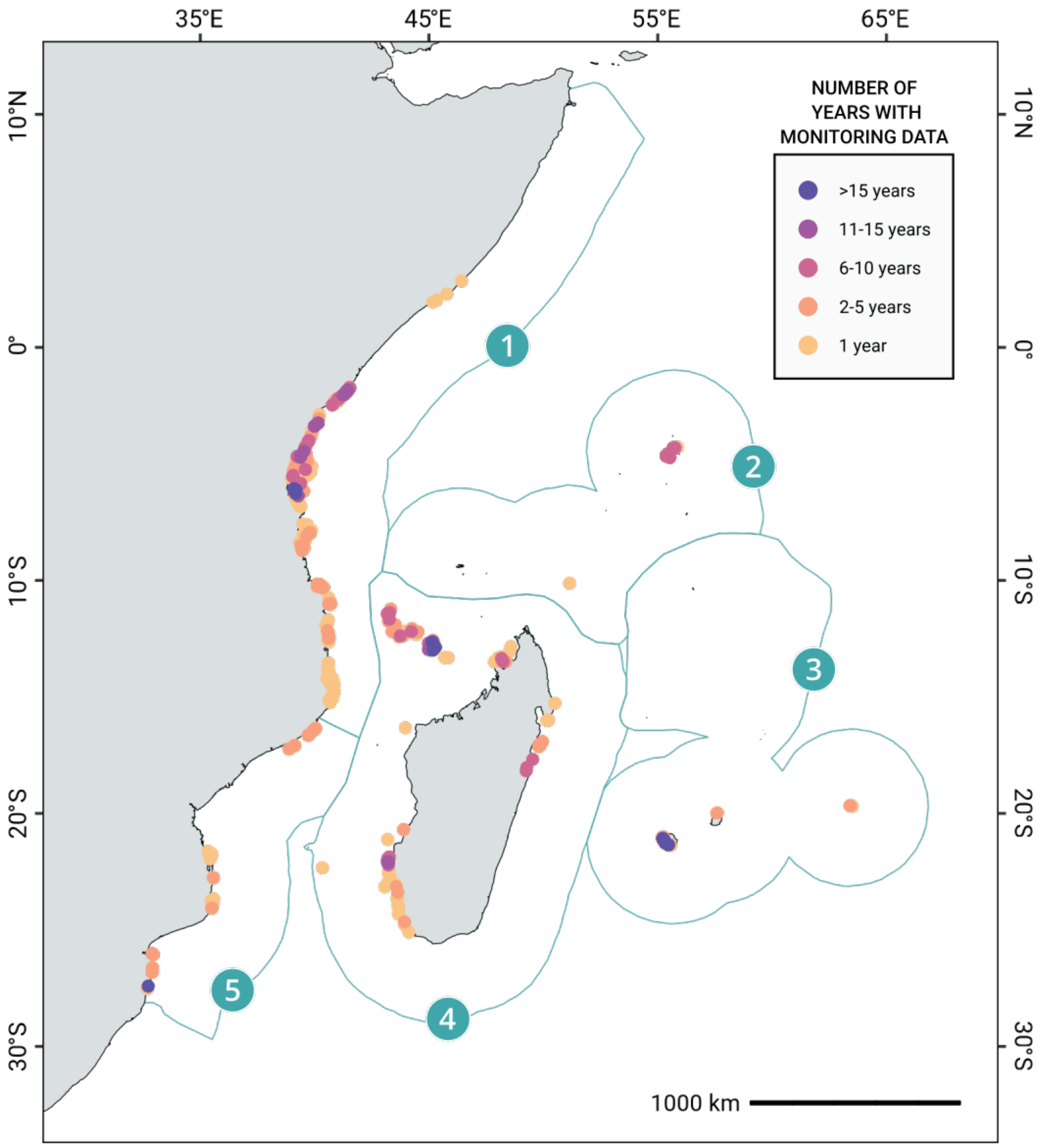
Western Indian Ocean 5 - Bight of Sofala and Delagoa

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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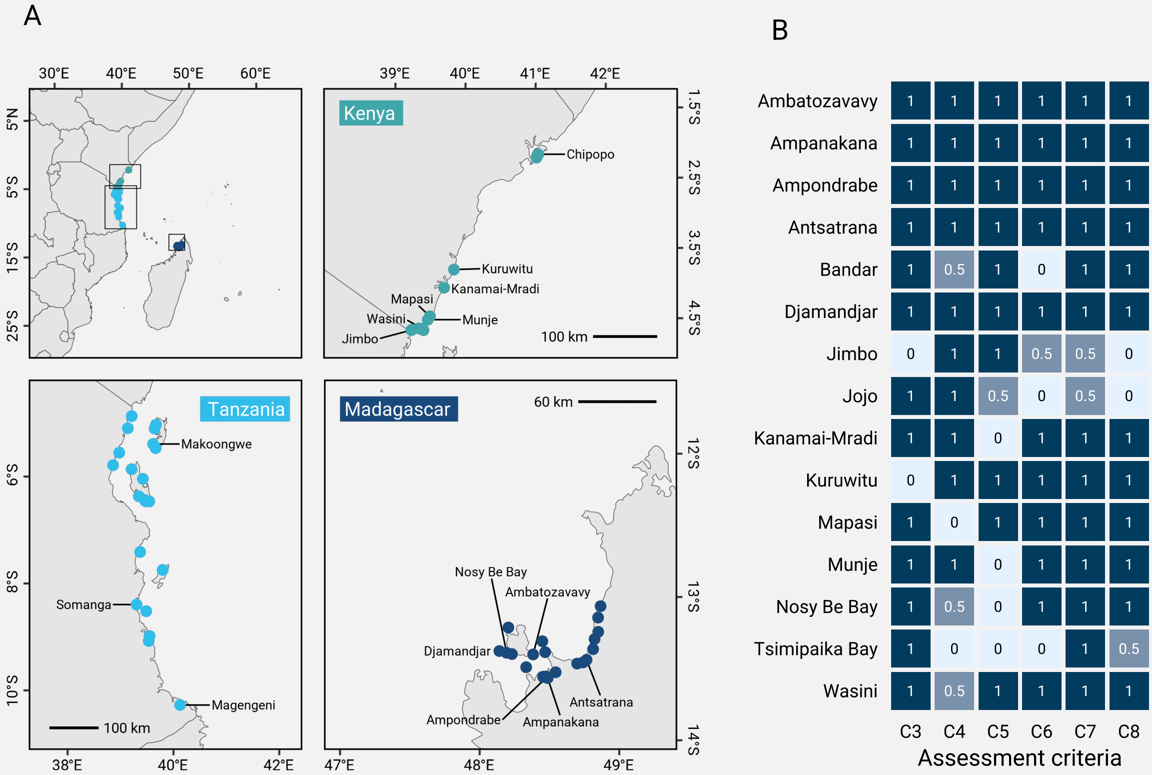
Coastal communities across the Western Indian Ocean (WIO), like millions of people globally, depend on coral reefs for food, livelihoods, coastal protection, and cultural value [99]. Yet, these reefs continue to face intensifying pressures from overfishing, destructive gear, and climate change [486]. 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, land-owning 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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