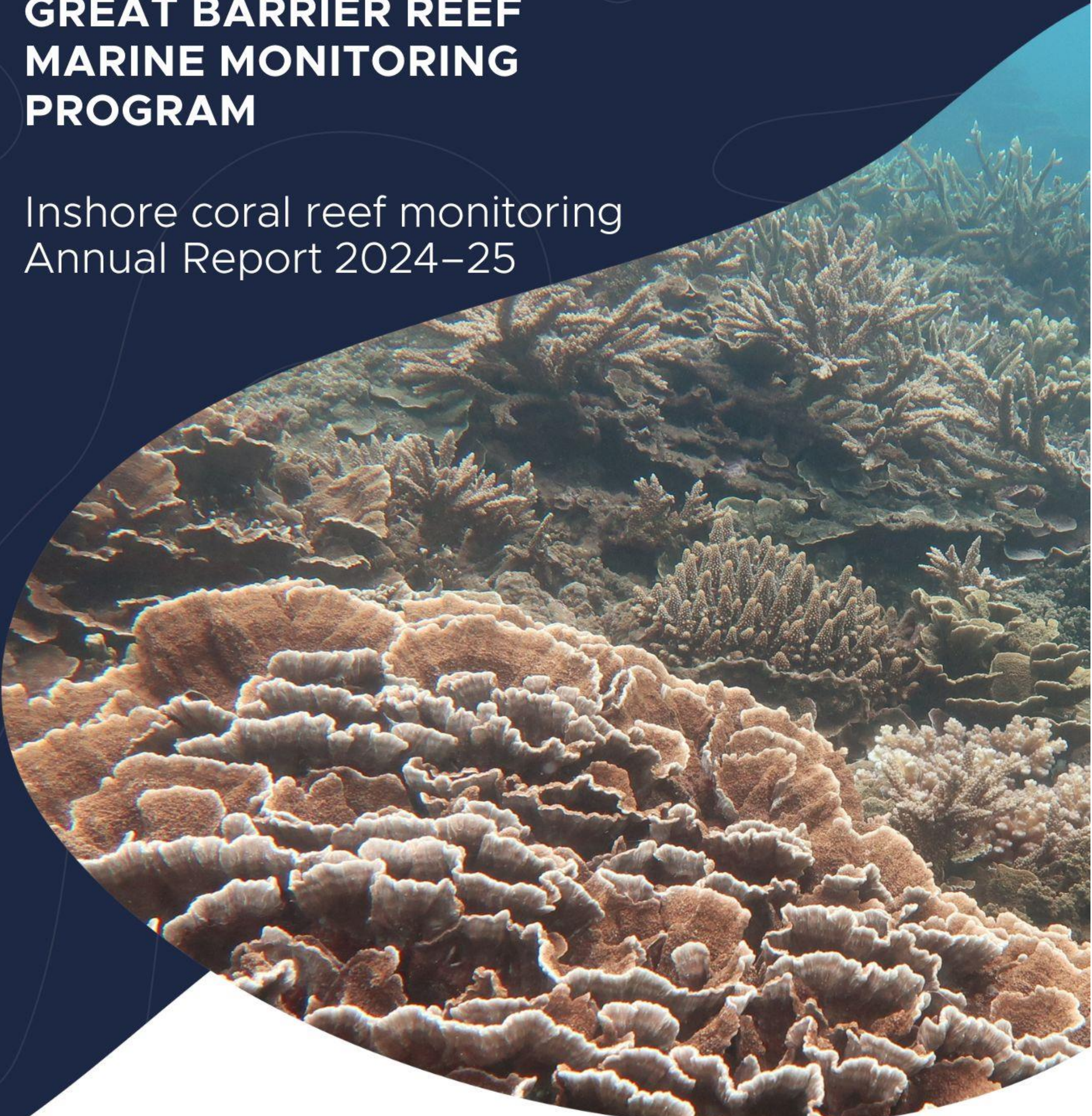
Australian Government
Great Barrier Reef Marine Park Authority

Reef Authority

# GREAT BARRIER REEF MARINE MONITORING PROGRAM

## Inshore coral reef monitoring Annual Report 2024–25



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Front cover photo: Healthy *Montipora* and *Acropora* colonies on the shallow slope at Franklands East. © Australian Institute of Marine Science, Photographer: Johnston Davidson

The Great Barrier Reef Marine Park Authority acknowledges the continuing Sea Country management and custodianship of the Great Barrier Reef by Aboriginal and Torres Strait Island Traditional Owners whose rich cultures, heritage values, enduring connections and shared efforts protect the Reef for future generations.



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## Commonly used abbreviations and acronyms

| | |
|---|---|
| AIMS | Australian Institute of Marine Science |
| Reef Authority | Great Barrier Reef Marine Park Authority |
| BoM | Australian Bureau of Meteorology |
| Chl *a* | Chlorophyll *a* |
| CSIRO | Commonwealth Scientific and Industrial Research Organization |
| LTMP | Long-Term Monitoring Program |
| MMP | Marine Monitoring Program |
| NOAA | National Oceanic and Atmospheric Administration |
| Reef 2050 WQIP | Reef 2050 Water Quality Improvement Plan |
| The Reef | Great Barrier Reef |
| TSS | Total suspended solids |

## Acknowledgements

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We acknowledge the Traditional Owners of the AIMS sites and Sea Country in which this work was undertaken and pay our respects to them as the first scientists in this land and acknowledge the important contribution this unique wisdom provides to contemporary scientific conversations and collaborations. We acknowledge the deep and timeless connection between Aboriginal and Torres Strait Islander people, their land and sea country and pay our respects to Elders past and present and acknowledge the future leaders of tomorrow.

# EXECUTIVE SUMMARY

This report details the condition of 30 inshore coral reefs monitored under the Great Barrier Reef Marine Monitoring Program and 6 inshore coral reefs monitored by the Australian Institute of Marine Science's Long-Term Monitoring Program. Results are presented in the context of the pressures faced by the ecosystem and their ramifications for the long-term health of inshore coral reefs.

The overall decline in the condition of inshore coral communities over the 20 years of monitoring demonstrates that the frequency and severity of acute disturbances have outstripped the capacity of communities to recover.

The overall condition of inshore reefs remains 'poor' but has improved slightly since reaching the lowest value recorded over the last 20 years in 2024 (Figure 1). Influential in the improvement has been the acceleration in recovery of reefs in the Mackay–Whitsunday region after an initially slow recovery from the severe impacts of cyclone Debbie. This improvement is, however, countered by declines in the Wet Tropics and Burdekin regions due primarily to the impacts of flooding over the last 22 years. In addition, the full impact of the severe marine heat wave in early 2024 in the Fitzroy Region has been realised with the condition of coral communities declining to 'very poor' condition.

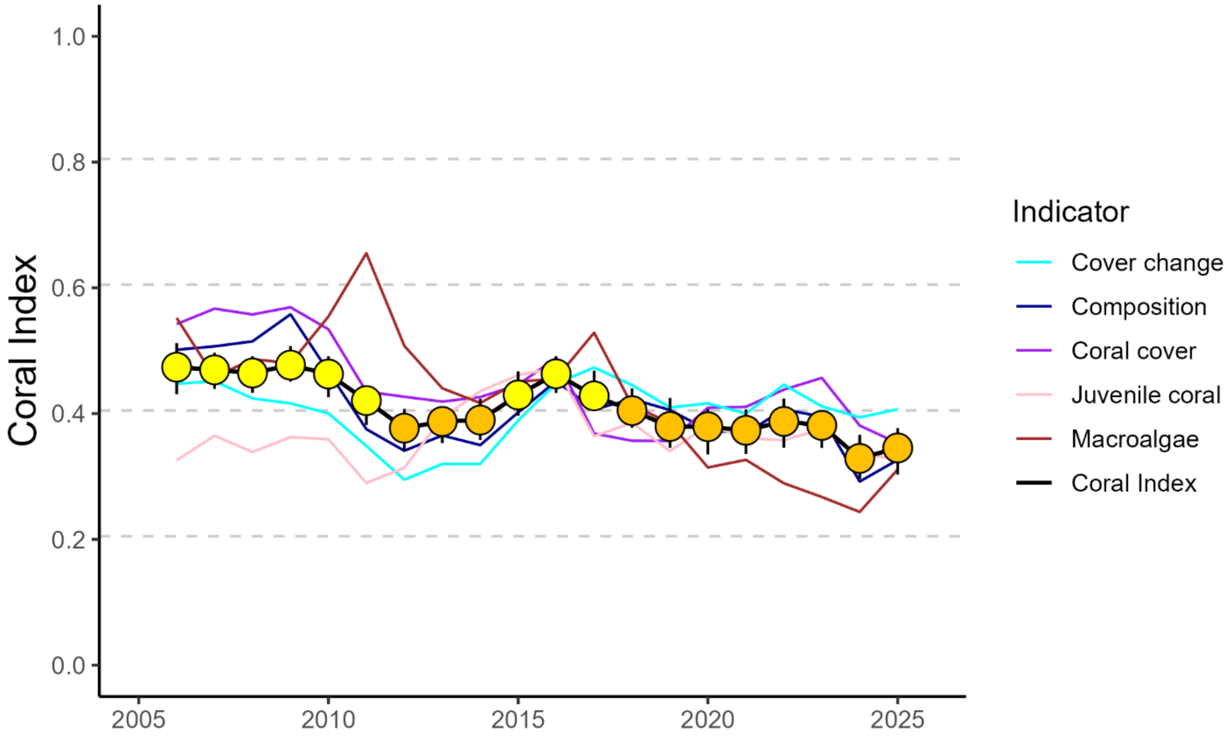



Figure 1 Trends in the Coral Index and contributing indicator scores for the inshore Reef. Coral Index scores are coloured according to Reef Water Quality Report Card categories: orange = 'poor', yellow = 'moderate'.

The Coral Index was developed by the Marine Monitoring Program as a way of expressing coral community condition that recognises coral communities are naturally dynamic. Coral communities in good condition must be resilient, that is, able to resist or recover from environmental pressures. The Coral Index is a composite of 5 indicators. Each indicator represents different processes that contribute to resilient coral reef communities. Indicators are in bold, followed by an explanation for their selection:

- **Coral cover** as an indicator of both corals' ability to resist the cumulative environmental pressures to which they have been exposed, and also the relative size of the population of corals as a source of larvae,

- **Macroalgae** proportion within the algal community as an indicator of the risk of competition with corals – *as high levels of macroalgae are detrimental to coral community resilience scores for this indicator decline with increasing levels of macroalgae*,
- **Juvenile coral** density as an indicator of the success of early life history stages in the replenishment of coral populations,
- Rate of coral **Cover change** as an indicator of the recovery potential of coral communities due to growth,
- Hard coral community **Composition** as an indicator of selective pressures imposed by the environmental conditions at a reef.

*Note: throughout the report capitalisation of the first letter of each indicator is used to make it explicit that it is the values for the indicator that are being referred to.*

The Coral Index scores average across the scores for the 5 indicators with scores in the ranges: 0 to2, considered 'poor', 0.21 to 0.4 'poor', 0.41 to 0.6 'moderate', '0.61 to 0.8' good, and greater than 0.8 'very good'. Coral Index scores contribute to marine condition assessments in the Reef Water Quality Report Card and regional report cards for the Wet Tropics, Burdekin Dry Tropics, Mackay–Whitsunday–Isaac, and Fitzroy regions. Coral Index scores are based primarily on Marine Monitoring Program data but also include data from inshore reefs monitored by the Australian Institute of Marine Science's Long-Term Monitoring Program. The regional report cards variously incorporate additional locally relevant data sources into estimates of the Coral cover indicator.

Very high rainfall, during February 2025, in the Burdekin and southern parts of the Wet Tropics caused severe flooding with loss of coral cover at several reefs in the Herbert–Tully sub-region and Burdekin region attributed to exposure of corals to low salinity flood-plumes. High winds during the active monsoon period also contributed to loss of coral cover on some reefs in the Burdekin region where clear evidence of storm damage was observed.

Elevated populations of corallivorous crown-of-thorns starfish were again present on reefs in the Johnstone Russell–Mulgrave sub-region of the Wet Tropics region. 'Outbreak' densities were observed at Fitzroy Island, High Island and in the Frankland Group. The impact of these starfish on corals continues to be reduced by culling undertaken by the Reef Authority's Crown-of-thorns Starfish Control Program.

Summer water temperatures in 2024-2025 were above average especially in the Wet Tropics and parts of the Burdekin region where coral bleaching was probable, as assessed by the Degree Heating Week product supplied by the National Oceanic and Atmospheric Administration (NOAA). However, *in situ* temperature loggers maintained at the coral monitoring sites suggested less extreme summer temperatures and no loss of coral cover was attributed to beaching over the 2024-2025 summer. In the Fitzroy region coral cover continued to decline as the longer-term impacts of heat stress in early 2024 compounded losses observed in 2024.

Improvement of Coral Index scores between 2011 and 2016 demonstrated the capacity of inshore coral communities to recover. However, between 2016 and 2025, the cumulative pressures imposed by cyclones and flooding, high seawater temperatures leading to coral bleaching in 2017, 2020, 2022 and 2024, and high densities of crown-of-thorns starfish, have contributed to a period of decline.

Overall, negative relationships between changes in Coral Index scores and discharge from the catchment in the Wet Tropics, Burdekin and Fitzroy regions demonstrate that loads entering inshore waters during high rainfall periods are reducing the resilience of inshore coral communities. In addition, the higher prevalence of macroalgae in areas of poor water quality highlights the increased potential for phase shifts to algae-dominated states in the more nutrient-rich areas of the inshore Great Barrier Reef (the Reef). While these results do not provide clear guidance in terms of load reductions required to improve Coral Index scores in the inshore Reef, they do support the premise of the Reef 2050 Water Quality Improvement Plan that the loads entering the Reef during high rainfall periods are reducing the resilience of these communities. The recent increase in disturbance frequency due to the increasing frequency and severity of marine heat waves only reinforces the importance of managing local pressures to ensure the balance between damage to coral communities caused by acute disturbances and their subsequent recovery supports the long-term resilience of these communities.

The following sections summarise the condition of coral communities in mid-2025 in each Natural Resource Management region in which inshore reefs are monitored.

**Wet Tropics region coral community condition**

Coral communities remain in 'moderate' condition in this region, despite condition having declined in the Barron–Daintree and Herbert–Tully sub-regions.

- In the Barron–Daintree sub-region, the Coral index score remains 'poor' and has declined to the lowest level recorded. Reefs in 2024 were severely impacted by freshwater inundation and waves associated with the passage of cyclone Jasper. The most impacted reef was Snapper South, where all corals were killed. Buoying the Coral Index score in 2024 were 'good' scores for the Macroalgae and Cover change indicators, and the timing of surveys for one reef, Low Isles, which was surveyed prior to the passage of cyclone Jasper. A further decline for 2025 was due to: the resurvey of Low Isles that revealed this reef was also impacted by the 2024 events, the initial rate of hard coral recovery has led to a reduction in the Cover change score, and the recolonisation of parts of the reef by macroalgae has reduced the Macroalgae indicator score. *It should also be noted that the score reported this year for 2024 of 0.40 ('poor') has been revised down from that reported last year of 0.42 'moderate'. This change was due to the decision to set the Composition indicator score to zero for reefs at which there is zero cover of hard corals.*
- In the Johnstone Russell–Mulgrave sub-region, the Coral Index remained 'moderate' and stable since declining between 2021 and 2024. The recent declines were due to the cumulative impacts of coral bleaching, cyclone Jasper and crown-of-thorns starfish that variously reduced coral cover across the region. Since 2022, the Macroalgae indicator has declined from 'good' to 'poor', and this is likely influencing the ongoing 'poor' Juvenile indicator score. Despite the ongoing removal of crown-of-thorns starfish by the Crown-of-thorns Starfish Control Program, outbreak densities were again observed on the eastern aspects of each island, or island group.
- In the Herbert–Tully sub-region, the Coral Index score has continued to decline but remains 'moderate'. The decline in 2025 largely captures the exposure of shallow sites at Dunk Island and Bedarra Island to low salinity floodwaters. The floods killed corals and resulted in reduced scores for the Coral cover, Juvenile coral and Composition scores but improved scores for the Macroalgae indicator as macroalgae were also impacted by the flood. In other areas such improvements in Macroalgae scores following a disturbance event have proven short-lived and it is likely scores for this indicator will decline next year as the macroalgae community is reestablished.

**Burdekin region coral community condition**

While the Coral Index score remains 'moderate', it has declined from 0.49 to 0.41 since 2020. In early 2025 reefs were exposed to both low salinity floodwaters and wave damage associated with a localised storm, both stemming from a particularly active monsoon. These events caused reductions in the Coral cover score at both 2 m and 5 m depths, and at 2 m depth, also included the scores for the Composition and Juvenile coral indicators. The influence of these impacts was moderated by improved scores for the Macroalgae indicator as levels of macroalgae were reduced by the summer's events. As with the Herbert–Tully sub-region, this marked improvement in the Macroalgae indicator score is likely to be temporary, as macroalgae typically recolonise following a disturbance event.

Burdekin reefs incurred minor impacts from thermal stress in 2020 and 2024, along with cyclone Kirrily also in 2024. Despite these events, regional-scale coral cover remained reasonably stable, with these events slowing further recovery rather than causing additional declines. Recovery was further constrained by regionally ‘poor’ scores for the Juvenile coral and Macroalgae indicators, suggesting water quality related pressures continue to limit the recovery potential of these reefs.

**Mackay–Whitsunday region coral community condition**

The Coral Index score increased to ‘moderate’ as the recovery of coral communities from the severe impacts of cyclone Debbie accelerates. Driving this recovery has been the continued increase in the density of juvenile corals, with scores for the Juvenile indicator increasing within the ‘moderate range’ since 2022. While still ‘poor’, the Coral cover indicator has consistently improved as juvenile and surviving corals grow. However, the Cover change score remains ‘poor’ as the rate of increase in coral cover remains slightly below modelled expectations. It should be noted here that the Cover change indicator is a rolling mean of interannual scores over the last 4 years and as such the current score is partially influenced by slower recovery in recent years. The Macroalgae indicator score remains ‘poor’ with high levels of macroalgae persisting at several reefs; however, this indicator showed the most improvement over the last year and is close to ‘moderate’.

**Fitzroy region coral community condition**

The marine heatwave of 2024 had a severe impact on coral communities in this region, reflected by the Coral Index declining to ‘very poor’ in 2025. During surveys in 2024 many corals were still bleached and initial declines in coral cover were evident. However, the full impact of the heat stress only became clear in 2025, when surveys revealed a 57% loss of hard coral cover compared to 2023 levels. A large proportion of the coral killed was of the genus *Acropora,* which resulted in the Composition Indicator being downgraded to ‘very poor’. Of added concern are the current ‘very poor’ scores for Juvenile and Macroalgae indicators, both of which have been consistently rated as ‘poor’ or ‘very poor’ in this region, signifying a clear bottleneck for coral community recovery.

# 1 INTRODUCTION

The proximity of inshore reefs to the coast makes them highly accessible, resulting in social, economic and cultural importance that is disproportionate to their relatively small share of the Great Barrier Reef World Heritage Area's coral estate (Great Barrier Reef Marine Park Authority 2024). Unfortunately, this proximity also exposes inshore reefs to increased pressures of elevated turbidity, high nutrient levels and low salinity flood plumes compared to their offshore counterparts.

Reefs globally are under pressure as the effects of climate change are superimposed onto the natural disturbance and recovery cycles of coral communities (Osborne *et al.* 2017, Hughes *et al.* 2018, Emslie *et al*. 2024). The increasing pressures facing coral reefs makes it even more important that the Reef is managed to optimise the potential for coral communities to resist or recover from inevitable disturbance events (Bellwood *et al.* 2004, Marshall & Johnson 2007, Carpenter *et al.* 2008, Mora 2008, Hughes *et al.* 2010).

## 1.1 Conceptual basis for the coral monitoring program

Disentangling the complex interactions between benthic communities and environmental pressures shaping coral reef condition is reliant on accurate, long-term, field-based observations of how these communities respond to a range of exogenous pressures. To this end, the Australian Institute of Marine Science (AIMS) and the Great Barrier Reef Marine Park Authority (the Reef Authority) have co-invested to provide inshore coral reef monitoring under the Great Barrier Reef Marine Monitoring Program (MMP) since 2005.

A key component to the outputs of the MMP is the synthesis and communication of information to a range of stakeholders. The primary communication tool for the coral component of the MMP is the Coral Index, which contributes to the Reef Water Quality Report Card. The Coral Index captures key aspects of coral community condition and resilience. It is used to track trends over time and highlight where and when condition is poor.

The Coral Index is based on the general understanding that healthy and resilient coral communities exist in a dynamic equilibrium, with communities periodically in a state of recovery, punctuated by acute disturbance events. Common acute disturbances to inshore reefs include cyclones (often producing flooding), high water temperatures and, rarely, outbreaks of crown-of-thorns starfish, all of which can result in widespread mortality of corals (e.g., Sweatman *et al.* 2007, Osborne *et al.* 2011). While, nutrients carried into the system as run-off may compound the influences of acute disturbances by increasing the susceptibility of corals to disease (Bruno *et al.* 2003, Haapkylä *et al*. 2011, Kuntz *et al*. 2005, Kline *et al*. 2006, Haapkylä *et al*. 2011, Weber *et al.* 2012, Vega Thurber *et al.* 2013), exacerbating outbreaks of crown-of-thorns starfish (Wooldridge & Brodie 2015), and potentially magnifying the impacts of thermal stress (Wooldridge & Done 2004, Negri *et al.* 2011, Wiedenmann *et al.* 2013, Fisher *et al.* 2019, Brunner *et al.* 2021, Cantin *et al.* 2021). It is the potential for pollutants in run-off to suppress the recovery of coral communities (Schaffelke *et al.* 2017) that is a key focus of this monitoring and reporting program.

The replacement of hard corals (order Scleractinia) lost to disturbance is reliant on both the recruitment of new colonies and regeneration of remaining colonies (Smith 2008, Diaz-Pulido *et al.* 2009). Elevated concentrations of nutrients and pesticides, together with increased turbidity can negatively affect coral reproduction (reviewed by Fabricius 2005, van Dam *et al.* 2011, Erftemeijer *et al.* 2012, Luo *et al.* 2022). High rates of sediment deposition and accumulation on reef surfaces can negatively affect larval settlement (Babcock & Smith 2002, Baird *et al.* 2003, Fabricius *et al.* 2003, Ricardo *et al.* 2017) and smother juvenile corals (Harrison & Wallace 1990, Rogers 1990, Fabricius & Wolanski 2000). The density of juvenile hard corals is included as a key indicator of the success of recruitment processes. Relationships between high nutrient and organic matter availability and higher incidence or severity of coral disease (Bruno *et al.* 2003, Haapkylä *et al*. 2011, Weber *et al.* 2012, Vega Thurber *et al.* 2013) highlight the cumulative pressure that poor water quality places on corals already stressed by recent disturbances.

The cover of macroalgae is monitored and reported on because macroalgae are more abundant in areas with high water column Chlorophyll (Chl *a*) concentrations, reflecting higher nutrient availability (De'ath & Fabricius 2010, Petus *et al.* 2016). A high abundance of macroalgae may suppress reef resilience (e.g., Hughes *et al.* 2007, Foster *et al.* 2008, Cheal *et al.* 2010) through increased competition for space or by changing the microenvironment into which corals settle and grow (e.g., McCook *et al.* 2001, Hauri *et al.* 2010). Macroalgae have been documented to suppress fecundity (Foster *et al.* 2008) and reduce overall recruitment of hard corals (Birrell *et al.* 2008a, Diaz-Pulido *et al.* 2010, Smith *et al.* 2023, Burgo *et al.* 2025), although chemical cues from some species appear to promote the settlement of coral larvae (Morse *et al.* 1996, Birrell *et al.* 2008b). Macroalgae have also been shown to diminish the capacity for growth among local coral communities as direct competitors for space and light (Fabricius 2005) or as a result of allelopathic alteration of the microbial communities of the coral holobiont (Morrow *et al*. 2012, Vega Thurber *et al.* 2012, Clements & Hay 2023).

Corals derive most of their energy from the photosynthesis of their symbiotic algae but can also obtain energy by feeding on ingested particles and planktonic organisms (heterotrophic feeding). The extent to which corals can compensate for reduced photosynthetic energy through heterotrophic feeding–for instance, when light is limited in turbid waters (Bessell-Browne *et al.* 2017)–varies among species (Anthony 1999, Anthony & Fabricius 2000, Houlbrèque & Ferrier-Pagès 2009). Similarly, the energy required to shed sediment varies among species due to differences in the efficiencies of passive (largely depending on growth form) or active (such as mucus production) strategies for sediment removal (Rogers 1990, Stafford-Smith & Ormond 1992, Duckworth *et al.* 2017). The balance between energy gained via heterotrophic feeding and energy expended to remove sediment in turbid environments will influence the ability of coral species to thrive. The taxonomic composition of hard coral communities is monitored as an indication of the selective pressure of water quality on coral communities, evident as changes in community composition along environmental gradients (De'ath & Fabricius 2010, Thompson *et al.* 2010, Uthicke *et al.* 2010, Fabricius *et al*. 2012, Luo *et al.* 2022).

A precursor, and more responsive indication, of selective pressures imposed by water quality is the rate that coral cover recovers following disturbances. Reductions in energy supplied by their symbionts, as well as increased competition for space, can slow coral growth or increase their susceptibility to disease (Vega Thurber *et al.* 2013). Building on this concept, Thompson *et al.* (2020) developed an indicator based on the expected rate of post-disturbance coral cover increase.

## 1.2 Purpose of this report

The purpose of this report is to provide the data, analyses and interpretation underpinning Coral Index scores included in the Reef Water Quality Report Card. This report includes results from coral reefs monitored by AIMS as part of the MMP until July 2025 with inclusion of data from inshore reefs monitored by the AIMS Long-Term Monitoring Program (LTMP) from 2005 to 2025. The Coral Index and indicator scores reported here were also supplied to regional bodies responsible for the Wet Tropics, Burdekin Dry Tropics and Mackay–Whitsunday–Isaac regional report cards.

To relate changes in the condition of coral reefs to variations in local water quality, the coral component of the MMP has the overarching objective to "*quantify the extent, frequency and intensity of acute and chronic impacts on the condition and trend of inshore coral reefs and their subsequent recovery*". The specific objectives are to monitor, assess and report:

i. the condition and trend of Great Barrier Reef inshore coral reefs in relation to desired outcomes (expressed as Coral Index scores) along identified or expected gradients in water quality,
ii. the extent, frequency and intensity of acute and chronic impacts on the condition of Great Barrier Reef inshore coral reefs, including exposure to flood plumes, sediments, nutrients, and pesticides,
iii. the recovery in condition of Great Barrier Reef inshore coral reefs from acute and chronic impacts including exposure to flood plumes, sediments, nutrients, and pesticides,

iv. trends in incidences of coral mortality attributed to coral disease, crown-of-thorns starfish, *Drupella* spp., *Cliona orientalis*, cyclones and thermal bleaching.

The program was developed with an express intent to report on the condition of coral reefs adjacent to Wet Tropics, Burdekin, Mackay–Whitsunday and Fitzroy Natural Resource Management regions (Figure 2). This regional focus is reflected in the structure of both the results and discussion sections of the report where current state and trends in coral communities are presented at regional scales before more general, Reef-wide, responses to environmental conditions are explored.

# 2 METHODS

This section provides an overview of the source and manipulation of climate and environmental pressure data, the sampling of coral communities, and the methods used to analyse these data.

## 2.1 Climate and environmental pressures

A range of environmental pressure variables are incorporated into this report as a basis for interpreting spatial and temporal trends in coral communities. The sources and use of these data are summarised in Table 1.

### *2.1.1 River discharge*

Daily records of river discharge in megalitres (ML) were obtained from Queensland Government Department of Natural Resources and Mines (DNRM) river gauge stations for the major rivers draining to the Great Barrier Reef (the Reef). For the Reef and each (sub-)region, total annual discharge estimates for each Water-year (1 October to 30 September) were based on those reported by MMP Water Quality (Gruber *et al*. 2026, Table A5), these values include a correction factor applied to gauged discharges to account for ungauged areas of the catchment.

For each (sub-)region, time-series of daily discharge were estimated as the sum of gauged values from gauging stations nearest to the mouths of the major rivers (Table A1).

Total annual river discharge for each region was used as a covariate in analysis of change in Coral Index scores. For this analysis, the biennial changes in Coral Index scores were considered due to the underlying sampling design of the program through to 2020 (Table 3). To match this sampling frequency, the maximum of the total annual discharge from all rivers discharging into a given region for each 2-year period between 2006 and 2025 was calculated.

### *2.1.2 River nutrient and sediment loads*

Loads of particulate nitrogen (PN), dissolved inorganic nitrogen (DIN) and total suspended sediment delivered by rivers were also sourced from MMP Water Quality (Gruber *et al.* 2026). Their methods state:

"The DIN loads for the basins of the Wet Tropics and Haughton Basin were calculated using the model originally developed in Lewis *et al*. (2014) which uses a combination of the annual nitrogen fertiliser applied in each basin coupled with basin discharge (calculated as per section 2.1.1). DIN loads for the Burdekin, Pioneer and Fitzroy basins were taken from those reported in the Great Barrier Reef Catchment Loads Monitoring Program. If the measured data for the most recent years in these basins were unavailable, a mean of the long-term annual mean concentration from the previous monitoring data was coupled with the discharge to calculate a load. DIN loads for the remaining basins were calculated using an annual mean concentration which was multiplied by the corresponding basin discharge calculations. The annual mean concentration for each basin was informed using a combination of available monitoring data and Source Catchments model outputs (McCloskey *et al.* 2021). The pre-development DIN loads were calculated using a combination of the estimates from the Source Catchments model as well as available monitoring data from largely 'pristine' locations.

The sediment and PN loads were similarly determined through a stepwise process. For the basins where the Great Barrier Reef Catchment Loads Monitoring Program captured >95% of the basin area (e.g., Burdekin, Pioneer, and Fitzroy) the measured/reported sediment and PN loads were used. If the measured data for the most recent years were unavailable, a mean of the long-term annual mean concentration from the previous monitoring data was coupled with the discharge to calculate a load. For other basins with monitoring data, the range of annual mean concentrations were compiled and compared with the latest Source Catchment modelling values. From these data a 'best estimate' of an annual mean concentration was produced and applied with the annual discharge data to calculate loads. Finally, for the basins that have little to no monitoring data, the annual mean concentration from the Source Catchments data was examined along with nearest neighbour monitoring data to determine a 'best estimate' concentration to produce the load."

### 2.1.3 Sea temperature

To assess variability in temperature within and among regions, temperature loggers were deployed at each coral monitoring reef at both 2 m and 5 m depths and routinely exchanged at the time of the coral surveys (i.e., every 12 or 24 months). Exceptions were Snapper South, Fitzroy East, High East, Franklands East, Dunk South and Palms East (see sections 4.3.1 and 4.3.2, for maps of sites) where loggers were not deployed due to the proximity of those sites to the sites on the western or northern aspects of these same islands, where loggers were deployed. No temperature loggers were deployed at Middle Island as the proximity of loggers at surrounding locations was considered sufficiently close to confidently reflect the water temperature at Middle Island.

Over the duration of the program a variety of temperature loggers have been used as summarised in Table A2.

For presentation and analysis, the data from all loggers deployed within a (sub-)region were averaged to produce a time-series of mean average water temperature. From these time-series a seasonal climatology for each (sub-)region was estimated as the mean temperature for each day of the year over the period 2005 to 2015. This baseline climatology excludes the high temperatures that led to coral bleaching in 2016 and 2017. In the Fitzroy region, 2006 data were also excluded due to severe coral bleaching in that year. Temperature data for each (sub-)region are plotted as anomalies, estimated as the mean difference between daily observations within a (sub-)region and the seasonal climatology.

### 2.1.4 Temperature stress

We present 2 estimates of seasonal temperature anomalies, which provide an indication of the potential temperature stress experienced by corals.

Degree heating weeks (DHW) product was downloaded from the National Oceanic and Atmospheric Administration (NOAA) Coral Reef Watch portal. The product sourced was the maximum DHW estimate for each ~5 km square pixel in a calendar year. DHW estimates accumulated thermal stress by summing the number of weeks in which sea-surface temperatures exceed the mean temperature of the hottest month from a location's climatology by more than 1 degree (Liu *et al*. 2014). For each pixel globally the seasonal climatology was estimated from 1985-2012 to identify the hottest month of the year. The mean temperature of this month, averaged across the years 1985-1990 and 1993, was used as the baseline summer maximum temperature. DHW estimates are the accumulation of temperature anomalies exceeding this baseline by at least 1 degree Celsius, calculated over a rolling 12-week window.

We also calculate an *in-situ* estimate of degree heating weeks based on the temperature logger time series. For these estimates the mean monthly maximum temperature was derived from each logger timeseries prior to 2016. Excluded from this baseline period were 1998 and, in the Fitzroy region only, 2006 as severe coral bleaching was observed at those times. From this baseline $Obs.DHW$, similarly accumulated temperature anomalies over a 12-week rolling window with the annual maximum value recorded. However, to keep the satellite derived DHW product and the *in situ* $Obs.DHW$ estimates on comparable scales, the $Obs.DHW$ estimates accumulate temperature anomalies greater than 0.4 degrees above their respective references, similar to the approach promoted by Whitaker & DeCarlo (2024).

$$Obs.DHW = \sum (Ti - (T_m + 0.5))7$$

Where, $T_m$ is the mean temperature of the hottest month over the baseline period for a location and $Ti$ is observed mean daily temperature. Only positive anomalies over the preceding 12 weeks are summed with the result divided by 7 to return the summed daily anomalies to the weekly scale.

Table 1. Summary of climate and environmental data considered in this report.

| | Data range | Method | Usage | Data source |
|---|---|---|---|---|
| *Climate* | | | | |
| Riverine discharge | 1980 – 2025 | Water gauging stations closest to river mouth both as raw daily volumes and annual estimates adjusted for ungauged area of catchment | regional discharge plots and table, covariate in analysis of temporal change in Coral Index | DNRME, annual estimates adjustment as tabulated by Gruber *et al.* (2026) |
| Riverine DIN, sediment and PN loads | 2006 – 2025 | Annual load data from DNRME monitoring stations closest to river mouths with data in-fill and adjustments for ungauged areas | covariate in analysis of temporal change in Coral Index | MMP Water Quality (Gruber *et al*. 2026) |
| Sea temperature | 2005 – 2025 | *in situ* sensor at coral sites | regional plots, thermal bleaching disturbance categorisation, *in situ* DHW estimates | MMP Inshore Coral monitoring/ AIMS temperature monitoring program |
| DHW | 2006 – 2025 | remote sensing | informing attribution of thermal stress, thermal stress maps | National Oceanographic and Atmospheric Administration |
| Cyclone tracks | 2005 – 2025 | | informing attribution of storms as cause of observed coral loss, cyclone track maps | BoM |
| *Environment at coral monitoring sites* | | | | |
| Wet season Chl *a* and total suspended solids (TSS) | 2021 – 2025 | Concentration estimated by multiplying the proportion of the wet-season (Nov-Apr) pixels adjacent to coral monitoring sites classified were into 4 broad water-types by the distribution of niskin measured concentrations within each water-type | Mapping. Chl *a* and TSS concentrations and for Chl *a* as covariate in analysis of variability in Coral Index and indictor current state | MMP Water Quality (Gruber et al. 2026) |
| Non-algal particulate | 2002 – 2018 | remote sensing adjacent to coral sites, resolution ~1 $km^2$ | Macroalgae and Composition metric thresholds | BoM |
| Chl *a* | 2002 – 2018 | remote sensing adjacent to coral sites, resolution ~1 $km^2$ | Macroalgae and Composition metric thresholds | BoM |
| K490 light attenuation coefficient | 2020 – 2024 | Remote sensing adjacent to coral sites, resolution ~1$km^2$ | Covariate in analysis of variability in Coral Index and indictor current state | IMOS |
| Sediment grain size | 2006 – 2017 | optical and sieve analysis of samples from coral sites | Macroalgae metric thresholds | MMP Inshore Coral monitoring |
| Chl *a* and Turbidity | 2021 – 2025 | Estimates from FLNTU loggers at 5 m depths at a subset of coral monitoring sites | Covariate in analysis of variability in Coral Index and indictor current state | MMP Water Quality |
| Particulate N and P, Phosphate and $NO_x$ | 2021 – 2025 | Means concentration from niskin samples adjacent to a subset of coral monitoring sites | Covariate in analysis of variability in Coral Index and indictor current state | MMP Water Quality |

### 2.1.5 Cyclone tracks

Cyclone tracks and intensity were downloaded from the BoM at https://www.bom.gov.au/cyclone/history/index.shtml. These tracks were primarily used to validate damage categorised as being caused by cyclones at the time of coral surveys. They are also presented in graphical form to illustrate the proximity of cyclones to the reefs monitored.

### 2.1.6 Water quality

Wet-season (1 December–30 April) water-type exposures were estimated based on the methods developed by the water quality component of the MMP (Petus *et al*. 2016, Gruber *et al*. 2026). In brief, Sentinel satellite data were used to classify waters into 21 Forel-Ule colour classes that were then aggregated into 4 reef water-types (Table 2). The water-type exposure for each pixel for the period 2021–2025 was estimated as the mean of the annual proportional exposures to each water-type over that period.

Wet-season concentrations of Chl *a* and TSS within each colour class were estimated based on distributions of Chl *a* and TSS measured from near-surface water samples, following the sampling methods outlined in Gruber *et al.* (2026). Each wet-season water sample was matched by date and location to a satellite derived water-type classification. The measured water quality estimates used were restricted to those taken within Open coastal, Mid-shelf or Offshore water bodies to guard against extreme values that can occur in enclosed coastal or macro-tidal habitats in which none of the coral monitoring occurs. The distributions of measured water quality within each water-type are summarised in Table 2.

For mapping, the median values of Chl *a* and TSS for each pixel were derived from a 2000 row, weighted distribution constructed by randomly sampling from the distributions of measured concentrations in each water-type (Table 2), proportionate to the wet-season water-type exposures for that pixel.

For reef-level estimates of Chl *a* and TSS concentrations, a set of 9 pixels were selected in open waters adjacent to each coral monitoring site. Estimates of annual median Chl *a* and TSS concentrations for each pixel were derived from a 2000 row weighted distribution constructed by randomly sampling from the distributions of measured concentrations in each water-type (Table 2), proportionate to the wet-season water-type exposures for each pixel. The resulting 9 distributions (one per pixel) were combined, and the annual wet-season estimate extracted as the median of this combined distribution. Reef level Chl *a* and TSS concentrations were estimated as the mean the last 5 annual estimates.

A second set of remotely sensed water clarity data, the diffuse attenuation coefficient at the 490 nm wavelength, K490 was source from daily satellite imagery curated by IMOS[1]. Daily estimates from pixels adjacent to each monitoring site were extracted from IMOS curated time series using the GBR Data Management System, https://pygeoapi.reefdata.io/collections/imos-srs-aqua-oc-k490. The diffuse attenuation coefficient estimates how strongly light intensity is attenuated within the water column due to the presence of scattering particles. K490 estimates the attenuation coefficient of light at 490 nm wave-length, i.e. visible light in the blue to green spectrum. Water clarity is inversely related to K490. For each monitoring location the median value of K490 over the period July 2020 to June 2024 was extracted, as the K490 timeseries for 2024-2025 was incomplete at the time of reporting.

For the subset of coral monitoring locations at which there are adjacent MMP water quality monitoring locations (Table 3) mean concentrations of Chl *a*, TSS, dissolved N, dissolved P and the ratio of both dissolved and total fractions of N and P from niskin samples were estimated. These estimates were

[1] Data were sourced from Australia's Integrated Marine Observing System (IMOS) – IMOS is enabled by the National Collaborative Research Infrastructure Strategy (NCRIS). It is operated by a consortium of institutions as an unincorporated joint venture, with the University of Tasmania as Lead Agent

derived from all samples over the period July 2021 to June 2025 and used as explanatory variables for variation in Coral Index, indicator scores and the coral community composition, macroalgae proportion and cover that underpin the Composition and Macroalgae indicators.

Table 2. Water types estimated from Sentinel imagery. Descriptions and data supplied by Caroline Petus, MMP Water Quality. Distributions based on the random resampling (2000 times) form the original number of observations (obs)

| Reef water-type | Forel-Ule (FU) colour classes | Description | Distribution | Chl *a* µg L$^{-1}$ | TSS mg L$^{-1}$ |
|---|---|---|---|---|---|
| WT1 | FU ≥ 10 | Brownish to brownish-green turbid waters typical of inshore regions of the Reef that receive land-based discharge and/or have high concentrations of resuspended sediments during the wet season. In floodwaters, this water-type typically contains high sediment and dissolved organic matter concentrations resulting in reduced light levels. It is also enriched in coloured dissolved organic matter and phytoplankton concentrations and has elevated nutrient levels. | 10th | 0.27 | 1.2 |
| | | | Median | 0.835 | 4.3 |
| | | | 90th | 2.715 | 22 |
| | | | # obs | 462 | 465 |
| WT2 | FU 6–9 | Greenish to greenish-blue turbid water typical of coastal waters with colour dominated by algae (Chl a) but also containing dissolved organic matter and fine sediment. This water-type is often found in open coastal waters of the Reef as well as in the mid-water plumes where relatively high nutrient availability and increased light levels due to sedimentation favour coastal productivity (Bainbridge *et al.* 2012). | 10th | 0.17 | 0.4 |
| | | | Median | 0.46 | 2.4 |
| | | | 90th | 1.15 | 10 |
| | | | # obs | 1220 | 1191 |
| WT3 | FU 4–5 | Greenish-blue water corresponding to waters with slightly above ambient suspended sediment concentrations and highlight penetration typical of areas towards the open sea. This water-type includes the outer regions of river flood plumes, fine sediment resuspension around reefs and islands and marine processes such as upwelling. Type III waters are associated with low land-sourced contaminant concentrations, and the ecological relevance of these conditions is likely to be minimal although not well researched. The Type III areas have a low magnitude score in the Reef exposure assessment. | 10th | 0.1 | 0.154 |
| | | | Median | 0.254 | 1.2 |
| | | | 90th | 0.732 | 5.019 |
| | | | # obs | 575 | 570 |
| WT4 | FU <4 | Bluish marine waters with high light penetration | 10th | 0.1 | 0.05 |
| | | | Median | 0.23 | 0.827 |
| | | | 90th | 1.947 | 3.87 |
| | | | # obs | 75 | 74 |

## 2.2 Coral monitoring

This section details the sampling design and sampling methods used to monitor and report coral community condition.

### *2.2.1 Sampling design*

Monitoring of benthic communities occurred at inshore reefs adjacent to 4 of the 6 natural resource management regions with catchments draining into the Reef: Wet Tropics, Burdekin, Mackay–Whitsunday and Fitzroy (Table 3, Figure 2). Sub-regions were included in the Wet Tropics region to align reefs more closely with the combined catchments of the Barron and Daintree rivers, the Johnstone and Russell-Mulgrave rivers, and the Herbert and Tully rivers.

No reefs are included adjacent to Cape York due to logistical and occupational health and safety issues relating to diving in coastal waters in this region. Limited development of coral reefs in nearshore waters adjacent to the Burnett Mary region precluded sampling there.

### *2.2.2 Site selection*

Initial selection of sites was jointly decided by an expert panel chaired by the Reef Authority. The selection was based on 2 primary considerations:

1. Within the Reef, strong gradients in water quality exist with increasing distance from the coast and exposure to river plumes (Larcombe *et al.* 1995, Brinkman *et al.* 2011). The selection of reefs for inclusion in the sampling design was informed by the desire to include reefs spanning these gradients to help assess the impact of water quality associated impacts.
2. There was either an existing coral community or evidence (in the form of carbonate-based substratum) of past coral reef development.

Exact locations were selected without prior investigation. Once a section of reef had been identified that was of sufficient size to accommodate the sampling design, a marker was deployed from the surface and transects established at the desired depth adjacent to this point.

In the Wet Tropics region, where few reefs exist in the inshore zone and well-developed reefs exist on more than one aspect of an island, separate reefs on windward and leeward aspects were included in the design. The benthic communities can be quite different on these 2 aspects even though the surrounding water quality is relatively similar. Differences in wave and current regimes determine whether materials such as sediments, freshwater, nutrients or toxins accumulate or disperse and hence moderate the exposure of benthic communities to environmental stresses. In addition to reefs monitored by the MMP, data from inshore reefs monitored by the AIMS LTMP have been included in this report.

Since the program began in 2005 there have been several changes to the selection of reefs sampled. In 2005 and 2006, three mainland fringing reef locations were sampled along the Daintree coast. Concerns over increasing crocodile populations in this area led to the cessation of sampling at these locations. In 2015, a revision of the marine water quality monitoring component of the MMP resulted in modifications to the sampling design for water quality. This included a concentration of sampling effort along a gradient away from the Tully River mouth. To better match the water quality sampling to the coral reef sampling in the Herbert–Tully sub-region, a new reef site was initiated at Bedarra and sampling at King Reef discontinued. Also influencing the discontinuation of sampling at King Reef was that the substrate was primarily composed of abiogenic rock rather than biogenically derived carbonate, suggesting this was not a coral reef. The substrate at Peak Island also lacked any substantive carbonate structure and sampling discontinued in 2020. As the MMP sites at Middle Reef in the Burdekin region were co-located with LTMP sites, this reef was removed from the MMP sampling schedule in 2015. Subsequent revision of the LTMP sampling design resulted their discontinuation of monitoring of Middle Reef, Green Island and Langford and Bird Islands in 2022, noting the last survey of Middle Reef was in 2013.

The current sites monitored by the MMP and LTMP and reported herein are presented in Figure 2.

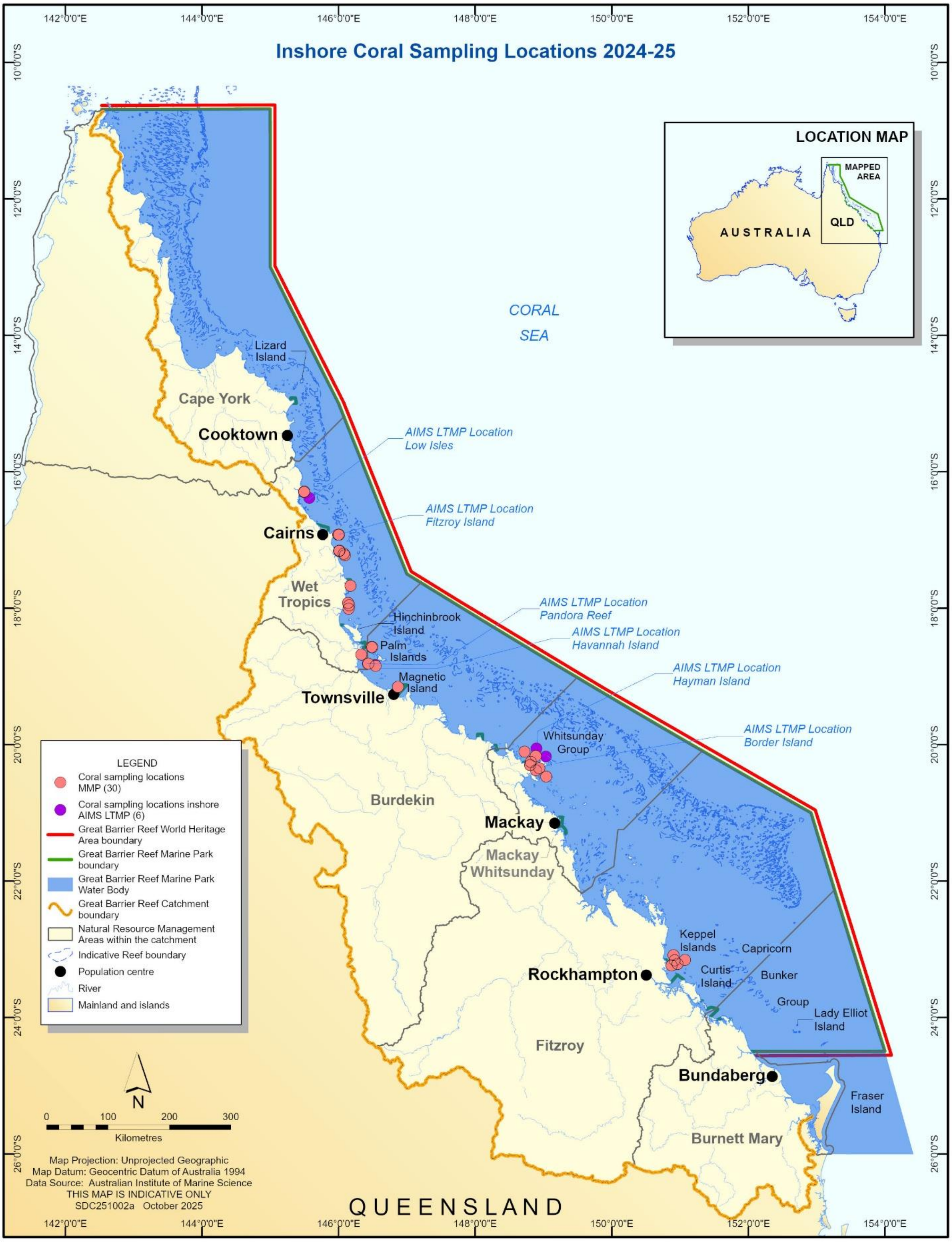



Figure 2. Coral sampling locations 2025. Map provided by the Reef Authority.

### 2.2.3 Depth selection

Within the turbid inshore waters of the Reef the composition of coral communities varies strongly with depth due to differing exposure to pressures and disturbances (e.g., Sweatman *et al.* 2007). For the MMP, transects were established at 2 depths. The lower limit for the inshore coral surveys was selected at 5 m below lowest astronomical tide (LAT) datum. Below this depth, coral communities rapidly diminish at many inshore reefs. A shallower depth of 2 m below LAT was selected as a compromise between a desire to sample the reef crest and logistical reasons, including the inability to use the photo point intercept technique in very shallow water and the potential for site markers to create a danger to navigation. The AIMS LTMP sites are not as consistently depth-defined as those of the MMP, with most sites set in the range of 5–7 m below LAT. Middle Reef was the exception with sites at approximately 3 m below LAT.

### 2.2.4 Site marking

At each reef, 2 sites separated by at least 250 m were selected along a similar aspect. These sites are permanently marked with steel fence posts at the beginning of each of 5 20 m-long transects and smaller steel rods (10 mm-diameter) at the midpoint and the end of each transect. Compass bearings and measured distances record the transect path between these permanent markers. Transects were set initially by running 2 60 m fibreglass tape measures out along the desired depth contour. Digital depth gauges were used along with tide heights from the closest location included in '*Seafarer Tides'* electronic tide charts produced by the Australian Hydrographic Service to set transects as close as possible to the desired depth. Consecutive transects were separated by 5 metres. The position of the first picket of each site was recorded by GPS. Site directions and waypoints are stored electronically in AIMS databases.

### 2.2.5 Sampling timing and frequency

Coral reef monitoring was undertaken predominantly over the months May–July, as this allows most of the influences resulting from summer disturbances, such as cyclones and thermal bleaching events, to be realised. Although the acute events occur over summer, the stress incurred can cause ongoing mortality for several months. The winter sampling also protects observers from potential risk from marine stingers over the summer months. The exception was Snapper Island, where sampling occurred typically in the months August–October.

The frequency of surveys has changed gradually over time (Table 3) due to budgetary constraints. In 2005 and 2006, all MMP reefs were surveyed. From 2007 through to 2014, a subset of reefs at which there were co-located water sampling sites were classified as 'core' reefs and sampled annually. The remaining reefs were classified as 'cycle' and sampled only in alternate years, with half sampled in odd-numbered years (i.e., 2009, 2011 and 2013) and the remainder in even-numbered years.

When an acute disturbance was suspected to have impacted cycle reefs during the preceding summer, they were resurveyed irrespective of their odd or even year classification. This provided both a timely estimate of the impact of the acute event and a baseline for the recovery period. Further funding reductions necessitated the adoption of a biennial sampling cycle for all reefs in 2015, although a contingency for the out-of-phase resampling of reefs impacted by acute disturbance was maintained.

In 2021 the program returned to annual sampling of all reefs.

Table 3. Coral monitoring samples. Black dots mark reefs surveyed as per sampling design, the "+" symbol indicates reefs surveyed out of schedule to assess disturbance. LTMP Reefs surveyed by the AIMS LTMP, WQ, indicates reefs at which water quality monitoring was undertaken, * indicates WQ was ceased in 2014, and ** indicates WQ was begun in 2015. Blank cells indicate when reefs were not surveyed. Grey fill indicates when reefs were removed from the sampling design of the program.

| (sub-) region | Reef | 2005 | 2006 | 2007 | 2008 | 2009 | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 | 2024 | 2025 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Barron–Daintree | Cape Tribulation Nth | ● | ● | | | | | | | | | | | | | | | | | | | |
| | Cape Tribulation Mid | ● | ● | | | | | | | | | | | | | | | | | | | |
| | Cape Tribulation Sth | ● | ● | | | | | | | | | | | | | | | | | | | |
| | Snapper North (WQ*) | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | + | ● | + | ● | + | ● | ● | ● | ● | ● |
| | Snapper South | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | + | ● | + | ● | ● | ● | ● | ● | ● |
| | Low Isles LTMP | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | ● | ● | ● | ● |
| Johnstone Russell–Mulgrave | Green LTMP | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | | | | |
| | Fitzroy West LTMP | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | ● | ● | ● | ● |
| | Fitzroy West (WQ) | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | + | ● | + | ● | + | ● | ● | ● | ● | ● |
| | Fitzroy East | ● | ● | + | ● | | ● | + | ● | | ● | | ● | | ● | | ● | ● | ● | ● | ● | ● |
| | High East | ● | ● | ● | | ● | | ● | | ● | | ● | + | ● | + | ● | + | ● | ● | ● | ● | ● |
| | High West (WQ) | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | | ● | + | ● | + | ● | ● | ● | ● | ● | ● |
| | Frankland East | ● | ● | ● | | ● | | ● | | ● | | ● | + | ● | + | ● | + | ● | ● | ● | ● | ● |
| | Frankland West (WQ) | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | | ● | + | ● | | ● | ● | ● | ● | ● | ● |
| Herbert–Tully | Barnards | ● | ● | ● | | ● | | ● | | ● | | ● | | ● | + | ● | + | ● | ● | ● | ● | ● |
| | King | ● | ● | | ● | | ● | | ● | | ● | | | | | | | | | | | |
| | Dunk North (WQ) | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | | ● | + | ● | | ● | ● | ● | ● | ● | ● |
| | Dunk South | ● | ● | | ● | | ● | + | ● | | ● | | ● | + | ● | + | ● | ● | ● | ● | ● | ● |
| | Bedarra | | | | | | | | | | | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● |
| Burdekin | Palms West (WQ) | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | + | ● | + | ● | + | ● | ● | ● | ● | ● |
| | Palms East | ● | ● | | ● | | ● | + | ● | | ● | | ● | | ● | + | ● | ● | ● | ● | ● | ● |
| | Lady Elliot Reef | ● | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | ● | ● | ● | ● | ● |
| | Pandora North LTMP | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | ● | ● | ● | ● |
| | Pandora (WQ) | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | | ● | + | ● | | ● | ● | ● | ● | ● | ● |
| | Havannah North LTMP | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | + | ● | ● | ● | ● | ● |
| | Havannah | ● | ● | ● | | ● | | ● | | ● | | ● | + | ● | + | ● | + | ● | ● | ● | ● | ● |
| | Middle Reef LTMP | ● | | ● | | ● | | ● | | ● | | | | | | | | | | | | |
| | Middle Reef | ● | ● | ● | | ● | | ● | | ● | | | | | | | | | | | | |
| | Magnetic (WQ) | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | + | ● | + | ● | + | ● | ● | ● | ● | ● |
| Mackay–Whitsunday | Langford LTMP | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | | | | |
| | Hayman LTMP | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | ● | ● | ● | ● |
| | Border LTMP | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | ● | ● | | ● |
| | Double Cone (WQ) | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | | ● | + | ● | + | ● | ● | ● | ● | ● | ● |
| | Hook | ● | ● | | ● | | ● | | ● | | ● | | ● | | ● | | ● | ● | ● | ● | ● | ● |
| | Daydream (WQ*) | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | | ● | + | ● | | ● | ● | ● | ● | ● | ● |
| | Shute Harbour | ● | ● | | ● | | ● | | ● | | ● | | ● | + | ● | | ● | ● | ● | ● | ● | ● |
| | Dent | ● | ● | ● | | ● | | ● | | ● | | ● | | ● | | ● | + | ● | ● | ● | ● | ● |
| | Pine (WQ) | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | | ● | + | ● | + | ● | ● | ● | ● | ● |
| | Seaforth (WQ**) | ● | ● | ● | | ● | | ● | | ● | | ● | | ● | | ● | + | ● | ● | ● | ● | ● |
| Fitzroy | North Keppel | ● | ● | ● | | ● | | ● | | ● | + | ● | | ● | | ● | + | ● | ● | ● | ● | ● |
| | Middle | ● | ● | | ● | | ● | | ● | | ● | + | ● | | ● | + | ● | ● | ● | ● | ● | ● |
| | Barren (WQ*) | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | | ● | | ● | + | ● | ● | ● | ● | ● |
| | Keppels South (WQ*) | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | + | ● | | ● | ● | ● | ● | ● | ● |
| | Pelican (WQ*) | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | ● | | ● | | ● | ● | ● | ● | ● | ● |
| | Peak | ● | ● | | ● | | ● | + | ● | | ● | + | | ● | | ● | | | | | | |

## 2.3 Coral community sampling methods

Three sampling methodologies were used to describe the benthic communities of inshore coral reefs (Table 4).

Table 4. Survey methods used by the MMP and LTMP to describe coral communities.

| Survey method | Information provided | Transect dimension | |
|---|---|---|---|
| | | MMP (20 m long transects) | LTMP (50 m long transects) |
| Photo point intercept transects | Percentage cover of the substratum of major benthic habitat components. | Approximately 34 cm wide belt (dive slate length) along upslope side of transect sampled at 50 cm intervals from which 32 frames are sampled. | Approximately 34 cm wide belt along upslope side of transect sampled at 1 m intervals from which 40 frames are sampled. |
| Juvenile coral transects | Size structure and density of juvenile coral communities. | 34 cm wide belt (dive slate length) along the upslope side of transect. Size classes: 0–2 cm, 2–5 cm | 34 cm wide belt along the upslope side of the first 5 m of transect. Size class: 0–5 cm. |
| SCUBA search transects | Cause of any current or recent coral mortality | 2 m wide belt centred on the transect line | 2 m wide belt centred on the transect line |

### *2.3.1 Photo point intercept transects*

Estimates of the composition of benthic communities were derived from the identification of organisms on digital photographs taken along the permanently marked transects. The method closely followed the Standard Operation Procedure Number 10 of the AIMS LTMP (Jonker *et al.* 2008). In short, digital photographs were taken at 50 cm intervals along each 20 m transect. Estimates of proportional cover of benthic community components (benthic cover) were derived from the identification of the benthos lying beneath 5 fixed points digitally overlaid onto these images. Thirty-two images were randomly selected and analysed from each transect. Poor quality images were excluded and replaced by an image chosen randomly from those not originally selected. The AIMS LTMP utilised longer 50 m transects sampled at 1 m intervals, from which 40 images were selected.

For hard and soft corals, identification to genus level was mostly achieved. Identifications for each point were entered directly into a data-entry front-end to an Oracle-database, developed by AIMS. This system allows the recall of images and checking of any identified points.

### *2.3.2 Juvenile coral transects*

These surveys provide an estimate of the number of both hard and soft coral colonies that have successfully survived early life-cycle stages culminating in visible juvenile corals. In the first year of this program, juvenile coral colonies were counted as part of a demographic survey that counted the number of all individuals falling into a broad range of size classes that intersected a 34-cm wide (data slate length) belt along the upslope side of the first 10 m of each 20-m marked transect. As the focus narrowed to just juvenile colonies, the number of size classes was reduced, allowing an increase in the spatial coverage of sampling. From 2006 coral colonies less than 10 cm in diameter were counted within a belt 34 cm wide along the full length of each 20 m transect. Each colony was identified to genus and assigned to a size class of >0–2 cm, >2–5 cm or >5–10 cm. In 2019, recording of the 5–10 cm size class was discontinued as reporting focused on the <5 cm size class, and the age of larger colonies becomes increasingly uncertain. Importantly, this method aims to record only small colonies that were assessed as juveniles resulting from larval settlement and subsequent survival and growth. It does not include small coral colonies considered to have resulted from fragmentation or partial mortality of larger colonies. In 2006, the LTMP also introduced juvenile surveys along the first 5 m of each transect and focused on the single size-class of 0–5 cm. In practice, corals <~ 0.5 cm are unlikely to be detected in AIMS surveys.

### *2.3.3 SCUBA search transects*

SCUBA search transects document the incidence of coral disease and other agents of coral mortality and damage. Tracking of these agents of mortality is important as declines in coral community condition linked to them may be associated with increased exposure to nutrients or turbidity (Morrow *et al*. 2012, Vega Thurber *et al.* 2013). The resulting data are used primarily for interpretive purposes and help to identify both acute events such as a high proportion of damaged corals following storms, coral bleaching, high densities of coral predators or periods of chronic stress as inferred from high levels of coral disease.

This method closely follows the Standard Operation Procedure Number 9 of the LTMP (Miller *et al.* 2020). For each 20 m transect a search was conducted within a 2 m wide belt centred on the marked transect line. Within this belt, any colony exhibiting a scar (bare white skeleton) was identified to genus and the cause of the scar categorised as brown band disease, black band disease, white syndrome (a catch-all for unspecified disease), *Drupella* spp. (in which case the number of *Drupella* spp. snails was recorded), crown-of-thorns starfish feeding scar, bleaching (when the colony was bleached and partial mortality was occurring) or unknown (when a cause could not be confidently assumed). Scarring caused by fish bites was not recorded as it is deemed to be neither indicative of poor coral health nor likely to result in significant loss of coral cover. In addition, the number of crown-of-thorns starfish and their size-class were counted, and the number of coral colonies being overgrown by sponges was also recorded.

Finally, an 11-point scale (Table 5) was used to record the proportion of corals that were bleached and the proportion that had been physically damaged (as indicated by toppled or broken colonies). The physical damage category may include anchor as well as storm damage. The category ranges were derived from the 6 categories 0 to 5 used to score benthic cover from manta tow surveys by the LTMP with addition of + and – to include more differentiation within these categories.

Table 5. Categories used to record proportion of corals bleached or physically damaged.

| Recorded Category | Proportion of colonies effected |
|---|---|
| 0+ | Individual colonies |
| 1- | 1% to 5% |
| 1+ | 6% to 10% |
| 2- | 11% to 20% |
| 2+ | 21% to 30% |
| 3- | 31% to 40% |
| 3+ | 41% to 50% |
| 4- | 51% to 63% |
| 4+ | 63% to 75% |
| 5- | 76% to 87% |
| 5+ | >87% |

## 2.4 Calculating Reef Water Quality Report Card coral scores

Coral community condition is summarised as the Coral Index that aggregates scores for 5 indicators of reef ecosystem state (Thompson *et al.* 2020). The Coral Index score is the basis of coral community grades reported by the Reef Water Quality Report Card and the various regional report cards. The Coral Index is formulated around the concept of community resilience. The underlying assumption is that a 'resilient' community should show clear signs of recovery after inevitable acute disturbances, such as cyclones and thermal bleaching events, or, in the absence of disturbance, maintain a high cover of corals and successful recruitment processes. Each of the 5 indicators of coral community condition represents a different process that contributes to coral community resilience and is potentially disrupted by poor water quality:

- **Coral cover** as an indicator of corals' ability to resist the cumulative environmental pressures to which they have been exposed,
- Proportion of **Macroalgae** in algal cover as an indicator of competition with corals,
- **Juvenile coral** density as an indicator of the success of early life history stages in the replenishment of coral populations,
- Rate of hard coral **Cover change** as an indicator of the recovery potential of coral communities due to growth, and
- Hard coral community **Composition** as an indicator of selective pressures imposed by the environmental conditions at a reef.

For each of these indicators, a metric has been developed to allow scoring of observed condition on a consistent scale (0–1). The aggregation of indicator scores provides the Coral Index score as a summary of coral community condition.

### *2.4.1 Coral cover indicator metric*

High coral cover is a highly desirable state for coral reefs, both in providing essential ecological goods and services related to habitat complexity, maintenance of biodiversity and long-term reef development, and from a purely aesthetic perspective with clear socio-economic advantages. In terms of reef resilience, although low cover may be expected following severe disturbance events, high cover reflects both the ability to recover from previous acute pressures but also implies a degree of resistance to any chronic pressures influencing a reef or the ability to recover from past disturbances. Of note, this resistance may have selected high cover of a relatively few, particularly tolerant species, necessitating some consideration of community composition when assessing high coral cover. Finally, high cover equates to a large brood-stock: a necessary link to recruitment and an indication of the potential for recovery of communities in the local area.

This metric scores reefs based on the level of coral cover derived from point intercept transects. For each reef the proportional cover of all hard (order Scleractinia) and soft (subclass Octocorallia) corals are defined as 2 groups: "HC" and "SC", respectively. The Coral cover indicator is then calculated as:

$$Coral\ cover_{ij} = HC_{ij} + SC_{ij}$$

Where $i$ = reef and $j$ = time.

The threshold values for scoring this metric were based on assessment of coral cover time-series observed at inshore reefs from LTMP data (1992–2014), MMP data (2005–2014) and surveys from Cape Flattery to the Keppel Islands by Sea Research prior to 1998 (Ayling 1997), which identified a mean of >50% for combined coral cover on those inshore reefs. Due to the low likelihood of coral cover reaching 100%, the threshold for this indicator (where the score is a maximum of 1) has been set at 75%. This value captures the plausible level of coral cover achievable on inshore reefs and allows a natural break point for the categorisation of coral cover into the 5 reporting bands of the Reef Water Quality Report Card. Thus, the scoring for the Coral cover indicator is scaled linearly from zero when cover is 0% through to 1 when cover is at or above the threshold level of 75% (Figure 3). The decision to consider both hard and soft corals, rather than hard corals only recognises that the soft coral species present are a natural part of the diversity of inshore reef communities.

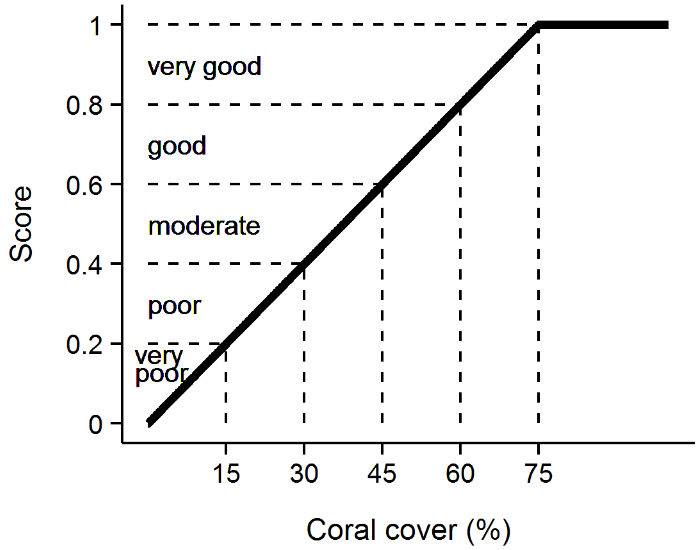



Figure 3,. Scoring diagram for the Coral cover indicator metric. Numeric scores and associated condition classifications based on observed coral cover are presented.

### 2.4.2 Macroalgae indicator metric

In contrast to coral cover, high cover of macroalgae on coral reefs is widely accepted as representing a degraded state. As opportunistic colonisers, macroalgae generally out-compete corals, recovering more quickly following physical disturbances. Macroalgae have been documented to suppress coral fecundity (Foster *et al*. 2008), reduce recruitment of hard corals (Birrell *et al*. 2008a, b, Diaz-Pulido *et al.* 2010) and diminish the capacity for growth among local coral communities (Fabricius 2005). The Macroalgae indicator metric considers the proportional representation of macroalgae in the algal community based on cover estimates derived from point intercept transects and is calculated as:

$$MAproportion_{ij} = MA_{ij} \,/\, A_{ij}$$

Where, $A$= percent cover of all algae, $i$ = reef, $j$ = time and $MA$ = percent cover of macroalgae.

Standardising the Macroalgae indicator against total cover of algae rather than the proportional cover of the substrate ensures this indicator is theoretically independent of coral cover. At high coral cover it is impossible to also have a high cover of macroalgae. However, when coral cover is high, a high proportion of macroalgae within the limited space available to algae can still be interpreted as imposing a downward pressure on coral resilience.

For calculating this metric, the collective term macroalgae defines a broad functional grouping that combines species clearly visible to the naked eye, although excluding crustose coralline and fine filamentous or “turf” forms, including cyanobacteria. In addition, as macroalgae show marked differences in abundance across the naturally steep gradient of environmental conditions within the inshore Reef, separate upper and lower thresholds were estimated for each reef and depth (Table A3). The use of separate thresholds ensures that the indicator is sensitive to changes likely to occur at a given reef.

The thresholds for each reef were determined based on predicted *MAproportion* from Generalised Boosted Models (Ridgeway 2007) that included mean *MAproportion* over the period 2005–2014 as the response and long-term mean Chl *a* concentration, suspended sediment concentration and proportion of clay and silt sized grains in reefal sediments as covariates (Thompson *et al.* 2016). Recognising the likelihood that the observed cover of macroalgae reflects a shifted baseline, an additional consideration in setting the upper threshold for *MAproportion* was the ecological influence of macroalgae on other indicators of coral community condition. Regression tree analyses that included *MAproportion* as the predictor variable indicated reduced scores for the Juvenile coral, Coral cover and Cover change indicators at higher levels of *MAproportion* (Thompson *et al*. 2016). These thresholds for ecological impacts caps informed the setting of upper bounds of *MAproportion* across all reefs at 23% at 2 m and at 25% at 5 m. The upper bounds for any reefs with predicted *MAproportion* higher than these caps were reduced to the cap level.

Scores for the Macroalgae indicator were scaled linearly from 0 when *MAproportion* is at or above the upper threshold through to 1 when *MAproportion* is at or below the lower threshold (Figure 4).

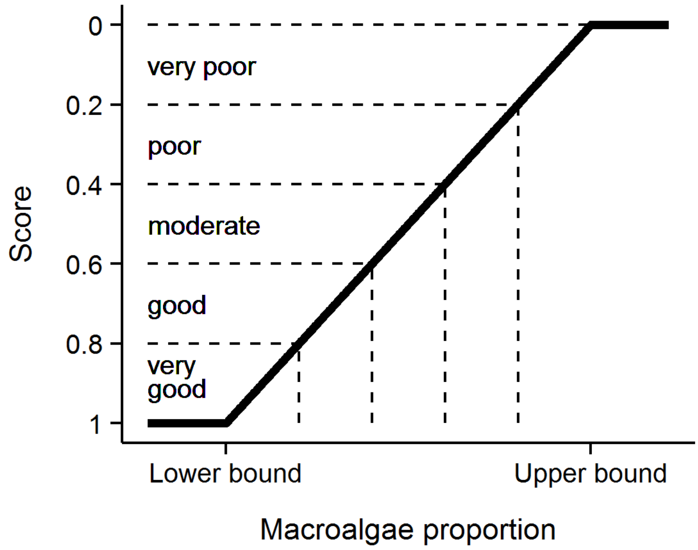



Figure 4. Scoring diagram for the Macroalgae indicator metric. Upper and lower threshold values are reef and depth specific. Numeric scores and associated condition classifications are presented. Note that for this metric the y-axis is inverted as high values reflect poor condition.

### *2.4.3 Juvenile coral indicator metric*

For coral communities to recover rapidly from disturbance events there must be adequate recruitment of new corals into the population. This metric scores the important recruitment process by targeting corals that have survived the early life stages. With the inclusion of LTMP data into the Coral Index, juvenile coral count data were subset to only include colonies up to 5 cm in diameter as this size class is common to both MMP and LTMP sampling. The Juvenile indicator scores the ratio between the number of juvenile corals observed and the area of transect occupied by algae. Specifically, counts of juvenile hard corals were converted to density per m$^2$ of space available to settlement as:

$$Juvenile\ density_{ij} = J_{ij} \ / \ AS_{ij}$$

Where, $J$= count of juvenile colonies < 5 cm in diameter, $i$ = reef, $j$ = time and $AS$ = area of transect occupied by any algae as estimated from the co-located photo point intercept transects.

Selection of thresholds for the scoring of this metric was based on the analysis of recovery outcomes for MMP and LTMP reefs up to 2014 (Thompson *et al*. 2016). From these time-series, a binomial model was fit to juvenile densities observed at times when coral cover was below 10% and categorised, based on recovery rate, as being either below or above the predicted lower estimate of hard coral cover increase as estimated by the Cover change indicator described below. This analysis identified a threshold of 4.6 juveniles per m$^2$ above which the probability that coral cover would subsequently increase at predicted rates outweighed the probability of lower than predicted rates of recovery.

Adding some weight to this result is that it was broadly consistent with the density of 6.3 juveniles per $m^2$, in the wider size range <10 cm, necessary for recovery in the Seychelles (Graham *et al.* 2015). As the upper density of juvenile colonies is effectively unbounded, it was desirable to set an upper threshold for scoring purposes. The density at which the probability was > 80% for coral cover to recover at predicted rates was 13 juveniles per $m^2$, and this density was chosen as the upper threshold. Based on this analysis, this metric was scored as follows: Juvenile coral score was scaled linearly from 0 at a density of 0 colonies per $m^2$ to 0.4 at a density of 4.6 colonies per $m^2$, then linearly to a score of 1 when the density was 13 colonies per $m^2$ or above (Figure 5).

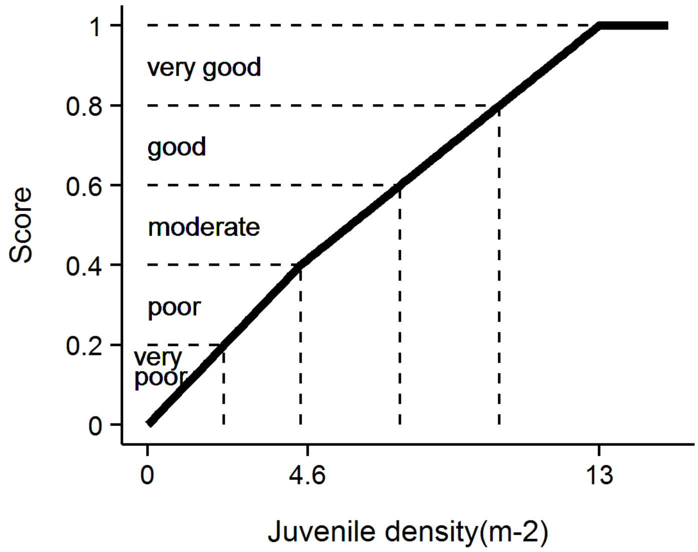



Figure 5. Scoring diagram for the Juvenile coral indicator metric. Numeric scores and associated condition classifications are presented.

### 2.4.4 Cover change indicator metric

A second avenue for recovery of coral communities is the growth of corals during periods free from acute disturbance (Gilmour *et al*. 2013). Chronic pressures associated with water quality may suppress the rate at which coral cover increases, indicating a lack of resilience. The Cover change indicator score is derived from the comparison of the observed change in hard coral cover between 2 visits and the change in hard coral cover predicted by Gompertz growth equations (Thompson & Dolman 2010) parameterised from time-series of coral cover available on inshore reefs from 1992 until 2007. Gompertz equations were parameterised separately for the fast-growing corals of the family Acroporidae and combined grouping of all other slower growing hard corals at each of 2 m and 5 m depths. Initial exploratory analysis provided no justification for further sub-setting of the coral communities as estimates for rate of change between visits did not vary among further subsets of the non-*Acropora* corals.

Years in which disturbance events occurred at a reef preclude the estimation of this indicator, as there is no expectation for increase in such situations. As such, estimates are only derived for annual or biennial periods during which no acute disturbances occurred.

A Bayesian framework was used to permit propagation of uncertainty through the predictions of expected coral cover increase from the 2 separately predicted coral types. The below formulae apply to the family Acroporidae (*Acr*) and have the same form as those applied for Other Corals (*OthC*) if these terms are exchanged where they appear in the equations.

$$\ln(Acr_{it}) \sim \mathcal{N}(\mu_{it}, \sigma^2)$$

$$\mu_{it} = vAcr_i + \ln(Acr_{it-1}) + \left(-\frac{vAcr_i}{\ln(estK_i)}\right) * \ln\,(Acr_{it-1} + OthC_{it-1} + Sc_{it-1})$$

$$vAcr_i = \alpha + \sum_{j=0}^{J} \beta_j Region_i \sum_{k=0}^{K} \gamma_k Reef_i$$

$$\alpha \sim \mathcal{N}(0, 10^6)$$

$$\beta_j \sim \mathcal{N}(0, \sigma^2_{Region})$$

$$\gamma_k \sim \mathcal{N}(0, \sigma^2_{Reef})$$

$$\sigma^2, \sigma^2_{Region}, \sigma^2_{Reef} = \mathcal{U}(0,100)$$

$$rAcr = \overline{vAcr_i}$$

Where, $Acr_{it}, OthC_{it}$ and $Sc_{it}$ are the cover of Acroporidae coral, other hard coral and soft coral, respectively, at a given reef at time $(t)$. $eskK$ is the community size at equilibrium (100) and $rAcr$ is the rate of increase (growth rate) in percent cover of Acroporidae coral. Varying effects of region and reef ($\beta_j$ and $\Upsilon_k$, respectively) were also incorporated to account for spatial autocorrelation. Model coefficients associated with the intercept, region and reef ($\alpha_i, \beta_j$ and $\Upsilon_k$, respectively) all had weakly informative Gaussian priors, the latter 2 with model standard deviation. The overall rate of coral growth $rAcr$ constituted the mean of the individual posterior rates of increase for $\overline{vAcr_i}$.

As model predictions relate to annual changes in hard coral cover, observed cover was adjusted to an estimated annual change since the previous observation ($Acr_{adj}$) prior to comparison to modelled estimates. Adjusted values, $Acr_{adj}$, were estimated as per the following formula:

$$Acr_{adj} = Acr_{i-1} + (Acr_i - Acr_{i-1}) * (365/(days\ between\ samples)$$

Where cover declined no adjustment was made and $Acr_{adj}$ assumed $Acr_i$.

Gompertz models were fitted in a Bayesian framework to facilitate combining growth rates and associated uncertainties across models. A total of 20,000 Markov-chain Monte Carlo sampling interactions across 3 chains with a warmup of 10,000 and thinned to every fifth observation resulted in well mixed samples from stable and converged posteriors (all r-hat (potential scale reduction factor) values less than 1.02). Model validation did not reveal any pattern in the residuals. Bayesian models were run in JAGS (Plummer 2003) via the R2jags package (Su & Yajima 2015) for R.

The posteriors of Acroporidae predicted cover and other hard coral predicted cover were combined into posterior predictions of total hard coral cover from which the mean, median and 95% Highest Probability Density (HPD) intervals were calculated.

As changes in hard coral cover from one year to the next are relatively small, the indicator value is averaged over valid estimates (inter-annual or biennial periods when cover was not impacted by an acute disturbance) for a four-year period culminating in the reporting year. If no valid observations were available in that four-year period, the most recent valid estimate is rolled forward.

To convert this indicator to a score the following process was applied (Figure 6):

- If hard coral cover declined between surveys, a score of 0 was applied.
- If hard coral cover change was between 0 and the lower HPD interval of predicted total hard coral cover change, scores were scaled to between 0.1 when no change was observed through to 0.4 when change was equal to the lower interval of the predicted change.
- If hard coral cover change was within the upper and lower HPD intervals of the predicted change the score was scaled from 0.4 at the lower interval through to 0.6 at the upper interval.
- If hard coral cover change was greater than the upper HPD interval of predicted change and less than double the upper interval, scores were scaled from 0.6 at the upper interval to 0.9 at double the upper interval.
- If change was greater than double the upper HPD interval, a score of 1 was applied.

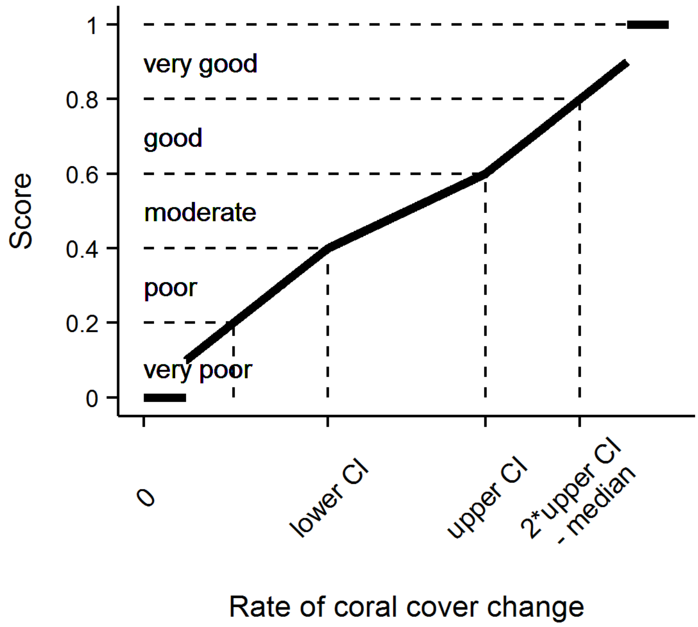



Figure 6. Scoring diagram for Cover change indicator metric.

### 2.4.5 Composition indicator metric

The coral communities monitored by the MMP vary considerably in the relative composition of hard coral species (Uthicke *et al*. 2010, Thompson *et al.* 2020). As demonstrated by Uthicke *et al*. (2010) and Fabricius *et al.* (2012), some of this variability can be attributed to differences in environmental conditions between locations, which implies selection for certain species based on environmental conditions. Coral communities respond to environmental conditions in a variety of ways. Most noticeably, they respond to acute shifts in conditions such as exposure to substantially reduced salinity (van Woesik 1991, Berkelmans *et al*. 2012), deviations from their normally experienced temperature profiles (Hoegh-Guldberg 1999) or extreme changes in their immediate hydrodynamic conditions (cyclones); all of which result in reductions in coral cover as susceptible species are killed. In contrast, the increased loads of sediments and nutrients entering the Reef carried in river discharge and/or land-based run-off due to land use practices in the adjacent catchments (Waters *et al*. 2014) may include a combination of acute conditions associated with flood events and then chronic change in conditions as pollutants are cycled through the system (Lambrechts *et al.* 2010). Chronic change in conditions, such as elevated turbidity or nutrient levels, could provide a longer period of selective pressures as environmental conditions disproportionately favour recruitment and survival of species tolerant to those conditions (see section 1.1).

This metric compares the composition of hard coral communities at each reef to a baseline composition at that reef (see below) and interprets any observed change as being representative of communities expected under improved or worsened water quality. A full description of this indicator is provided in Thompson *et al.* (2014). The basis of the metric is the scaling of cover for constituent genera (subset to broad morphological groups for the abundant genera *Acropora* and *Porites*) by weightings that correspond to the distribution of each genus along a water quality gradient. The location of each Reef along the water quality gradient was estimated as the reef's score along the first axis of a principal component analysis applied to observed turbidity and Chl *a* concentration. Genus weightings were derived from the location of each genus along the axis using these reef-level water quality scores as a constraining variable in a Canonical Analysis of Principal Coordinates (partial CAP; Anderson & Willis 2003) applied to MMP data (Thompson *et al.* 2020) as:

$$C_t = \sum_{i=1}^{n} H_{it} * G_i$$

Where, $C_t$ = the community composition location along the water quality gradient at time $t$,

$H_{it}$ = the Hellinger transformed (Legendre & Gallagher 2001) cover of genus $i$ at time $t$, and

$G_i$ = the score for genus $i$ taken from the constrained axis of the partial CAP.

Indicator scores are assigned based on the location of $C_t$ for the year of interest relative to a community specific baseline. The baseline for each community is bounded by the 95% confidence intervals about the mean $C_t$ from the first 5 years of observations of the community at each reef and depth. The scoring of the indicator is categorical being 0.5 when $C_t$ falls within the 95% confidence intervals for the location, 1 if beyond the confidence interval in a direction toward a community representative of lower turbidity and Chl *a* concentration and 0 if beyond the confidence interval in the direction of a community representative of higher turbidity and Chl *a* concentration (Figure 7).

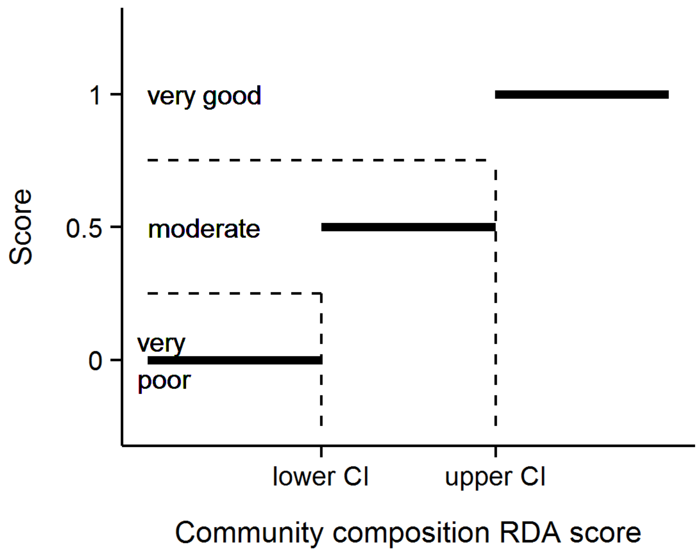



Figure 7. Scoring diagram for the Composition indicator metric

In 2022, AIMS adopted a series of revisions to the taxonomy of hard corals. For the most part, these changes resulted in the splitting or renaming of genera for which backward compatibility with prior genus-level taxonomy, used for the Composition indicator scores, was achieved. Rarely, some corals could not be identified to the level necessary to allow mapping to the genera on which the Composition indicator was based. This occurred both for the 2022 data and for blurred images from preceding years. Where corals could not be assigned to the required genera, they were excluded from the data prior to the estimation of Composition scores. An exception was the combined code used for the encrusting Pectiniidae when the differentiation between *Oxypora* and *Echinophyllia* could not be achieved. In this case corals were assigned the genus *Oxypora* as the more commonly occurring genus. The location of these genera along the constrained WQ axis ($G_i$) were very similar (0.008 and 0.002, respectively).

In 2025 scoring for this indicator was set to zero on occasions when there is no hard coral cover at a reef .

### 2.4.6 Aggregating indicator scores to Reef and regional scale assessments

In aggregating scores for various indicators into a single index, uncertainty should be considered. The degree of uncertainty in an index score derived for any spatial scale of interest will include uncertainty across multiple levels including, basic observational error, relevance of thresholds and variation in scores for different indicators or communities being assessed.

To derive Reef Water Quality Report Card scores for regions that propagated uncertainty through the double hierarchical aggregation of indicators and then reefs, a bootstrapping method was adopted. Firstly, for each indicator, a distribution of 10,000 observations was created by resampling (with replacement) from the observed scores for all reef and depth combinations within the region or sub-region of interest. Secondly, these 5 resulting distributions (one for each indicator) were added together and collectively resampled 10,000 times (with replacement) to derive a single distribution comprising 10,000 scores.

To generate estimates of precision (and thus confidence intervals) appropriate for the scale of the sampling design, the bootstrapped distribution of 10,000 scores was resampled once for every original input indicator score. Confidence intervals were calculated as the 2.5% and 97.5% quantiles of repeated estimates of the mean.

Mean Coral Index scores for each (sub-)region were estimated as the mean of observed mean scores for each indicator from all reefs and depths within the (sub-)region. Reef level scores as reported in the Reef Water Quality Report Card were estimated as the weighted mean of regional scores. Weightings applied reflect the relative proportion of inshore coral reef area within the 4 regions as: Wet Tropics (0.209), Burdekin (0.092), Mackay–Whitsunday (0.381) and Fitzroy (0.318). Lastly, Coral Index scores were converted to qualitative assessments by converting to a five-point rating and colour scheme with scores of:

- 0 to 0.2 were rated as ‘very poor’ and coloured red
- 0.21 to 0.4 were rated as ‘poor’ and coloured orange
- 0.41 to 0.6 were rated as ‘moderate’ and coloured yellow
- 0.61 to 0.8 were rated as ‘good’, and coloured light green
- 0.81 were rated as ‘very good’ and coloured dark green.

The indicators, associated thresholds, and scoring system utilised are summarised in Table 6. We note that the Composition indicator is likely to respond over longer time frames than the other indicators due to the inertia in community composition imposed by long-lived coral species.

Table 6. Threshold values for the assessment of coral reef condition and resilience indicators.

| Community attribute | Score | Thresholds |
|---|---|---|
| Combined hard and soft coral cover | Continuous between 0–1 | 1 at 75% cover or greater |
| | | 0 at zero cover |
| Proportion of algae cover classified as macroalgae | Continuous between 0–1 | ≤ reef specific lower bound and ≥ reef specific upper bound |
| Density of hard coral juveniles (<5 cm diameter) | 1 | > 13 juveniles per $m^2$ of available substrate |
| | Continuous between 0.4 and 1 | 4.6 to 13 juveniles per $m^2$ of available substrate |
| | Continuous between 0 and 0.4 | 0 to 4.6 juveniles per $m^2$ of available substrate |
| Rate of increase in hard coral cover (preceding 4 years) | 1 | Change > 2x upper 95% CI of predicted change |
| | Continuous between 0.6 and 0.9 | Change between upper 95% CI and 2x upper 95% CI |
| | Continuous between 0.4 and 0.6 | Change within 95% CI of the predicted change |
| | Continuous between 0.1 and 0.4 | Change between lower 95% CI and 2x lower 95% CI |
| | 0 | change < 2x lower 95% CI of predicted change |
| Composition of hard coral community | 1 | Beyond 95% CI of baseline condition in the direction of improved water quality |
| | 0.5 | Within 95% Confidence intervals of baseline composition |
| | 0 | Beyond 95% CI of baseline condition in the direction of declined water quality |

## 2.5 Data analysis and presentation

Observed coral community condition and relationships to variability in environmental conditions are presented at a range of spatial and temporal scales (Table 7).

Table 7. Format for presentation of community condition.

| Section | Scope | Scale | Covariates | Analyses/Presentation |
|---|---|---|---|---|
| 4.1 | Temporal trend in coral community condition | Reef | Major disturbances | Relative influence of major pressures over the time-series |
| 4.3 | Trends in Coral Index and individual indicators | (sub-)region | | Generalised linear mixed models; pairwise comparisons |
| 4.4.1 | Coral Index and indicator scores in 2025 | Reef and region | Chl *a*, Light attenuation coefficient, Suspended solids, N to P ratios (Total and dissolved fractions | Generalised linear mixed models, predicted responses |
| 4.4.2 | Temporal variability in Coral Index in relation to water quality | region | Regional riverine discharge | Generalised additive models, predicted responses |
| Appendix 1: Additional Information | Trends in benthic community composition. | reef/Depth | | Plots |
| | Summaries of 2025 observations | reef/Depth | | Observed values |

### 2.5.1 *Temporal trends in Coral indicators*

In each (sub-)region, a panel of plots provide temporal trends in observed values underlying each of the 5 indicators that contribute to the Coral Index.

Generalised linear mixed models were fit to the observed data to provide predicted annual means and 95% confidence intervals. For all indictors, models predicted values for the fixed effect of years and included a random effect for each reef and depth combination within the (sub-)region. Nested within the random term were sites within each reef and depth combination (Juvenile density model) and transects within sites for the coral cover and macroalgae proportion models. The inclusion of random locational effects helps to account not only for the sampling design that includes a mixture of annual and biennial sampling frequency (Table 3), but also the underlying differences in indicator values among the permanent sites and transects.

The assumed distributions of the data varied for each indicator:

- the community composition values were modelled as conforming to a Gaussian distribution,
- cover change estimates assumed a beta distribution,
- juvenile densities selected the better fit to observed values between models based on Poisson and negative binomial distributions,
- with coral cover and macroalgae proportion selecting the better fit between models was based on binomial and beta binomial distributions

More detailed summaries of taxonomic structure within benthic cover and juvenile communities at each reef and depth combination are presented as bar plots (Figure A1 to Figure A6). These additional plots break down cover and density of corals to the taxonomic level of Family. Genus level cover data for the current year only are included in Table A9 to Table A11.

### 2.5.2 *Analysis of change in Coral Index and indicator scores*

Differences in the Coral Index or individual indicator scores were estimated between focal years identified as local maxima or minima within the time-series of the Coral Index scores within each (sub-)region. Confidence in the magnitude of these differences is expressed as a probability that the mean difference in scores was greater or less than zero. Probabilities were estimated based on the location of zero (no difference) within the posterior distribution (n=1000) estimated from the mean and standard deviation of observed differences in scores between focal years. Probabilities were estimated separately for communities at 2 m and 5 m depths.

### 2.5.3 Response to pressures

The most tangible immediate effect of disturbances to coral communities is the loss of coral cover. A summary of disturbance history across all reefs and within each (sub-)region is presented as a bar plot of annual hard coral cover loss. The height of the bar represents the mean hard coral cover lost across all 2 m and 5 m sites within a region. Bars are segmented based on the proportion of loss attributed to different disturbance types. For each observation of hard coral cover at a reef and depth, the observation was categorised by any disturbance that had impacted the reef since the previous observation (Table 8) and the hard coral cover lost calculated as:

$$Loss = predicted - observed$$

where, *observed* is the observed cover of hard corals and *predicted* is the cover of hard corals predicted from the application of the coral growth models described for the Cover change indicator (section 2.4.4). The observed cover is adjusted to represent an annual time step, based on the period since the previous observation, to be consistent with the model predicted value. The proportion of coral cover lost per region for each disturbance type is subsequently calculated as:

$$proportional\ Loss = \left(\frac{Loss}{\sum Loss_r}\right)$$

Where, $\sum Loss_r$ is the overall cover lost at the scale of interest, either Reef or (sub-)region. It is important to note that for each loss attributed to a specific disturbance any cumulative impact of water quality is implicitly included.

For reference among (sub-)regions, the y axis of each plot was scaled to the maximum mean hard coral cover loss observed across regions in a single year (25.5% loss of coral cover within the Mackay–Whitsunday region in 2017).

Table 8. Information considered for disturbance categorisation.

| Disturbance | Description |
|---|---|
| Thermal bleaching | Consideration of DHW estimates and reported observations of coral bleaching |
| Crown-of-thorns starfish | SCUBA search revealing > 40 $ha^{-1}$ density of crown-of-thorns starfish during present or previous survey of the reef |
| Disease | SCUBA search observations of coral disease during present or previous survey of the reef |
| Flood | Discharge from local rivers sufficient that reduced salinity at the reef sites can reasonably be inferred. An exception was classification of a flood effect in the Whitsundays region based on high levels of sediment deposition to corals. This classification has been retained for historical reasons and would not be classified as a flood effect under the current criteria |
| Storm | Observations of physical damage to corals during survey that can reasonably be attributable to a storm or cyclone event based on nature of damage and the proximity of the reef to storm or cyclone paths. |
| Multiple | When a combination of the above occur |
| Chronic | In years that no acute disturbance was recorded a *Loss* was recorded when *observed* hard coral cover fell below the *predicted* cover and these losses classified as disturbance type 'Chronic'. This categorisation includes the cumulative impacts of minor exposure to any of the above disturbances along with chronic environmental conditions. Importantly, as estimates for each disturbance are a mean, and the disturbance categorisation "Chronic" includes all non-disturbance observations, any proportion of loss attributed to this category represents a mean under-performance in rate of cover increase for reefs not subject to an acute disturbance. |

### 2.5.4 Variation in Coral Index and indicator scores to gradients in water quality

The relationships between recent Coral Index or indicator scores and the location of reefs along water quality gradients presented in section 4.4.1 were explored using generalised linear mixed models (GLMM). Models were fit separately to each combination of Coral Index or indicator score, and depth, that included either mean Chl *a* concentration or k490 light attenuation coefficients as explanatory variable with region as a random effect. Statistical evidence for water quality influences on the coral community indicators were identified on the basis of a Likelihood Ratio Test that compared models to a null model fitting an intercept and regional factor only.

As scores are bound by 0 and 1, models assumed a Beta response distribution. Where the distribution of scores included 0 or 1, data were scaled as ((Score*0.998) + 0.001) prior to analysis to lie between 0 and 1 as defined by a beta distribution. Exceptions were the Composition indicator scores that were modelled using a probit regression due to their categorical response and the macroalgae indicator for which initial plotting of the data showed scores included a high proportion of zeros and that these were spread across water quality gradients in most regions making modelling unwarranted.

Both the Macroalgae and Composition indicators are designed to score communities based on expectations given their location along water quality gradients, thus enabling their sensitivity to change. As such the indicator values underlying these indicators: the proportion of algal cover categorised as macroalgae, and product of hard coral genus cover and water quality eigenvector weightings (Table A4), were also examined. Macroalgal proportion was also fit using a beta distribution, and a gaussian distribution was used for genus composition values.

GLMs were fit using the glmmTMB function within the glmmTMB package and the probit model for community composition was fit using the clm function in the ordinal package within the R Statistical and Graphical Environment (R Core Team 2023).

For the subset of reefs at which water quality is measured by the MMP water quality project GLM were fit to the same set of coral community responses coupled with: Chl *a* or ntu derived from *in situ* FLNTU loggers located at the coral monitoring site, or concentrations of Particulate N, Particulate P, Phosphate and $NO_x$ (the combined concentration of Nitrate and Nitrite) estimated from MMP water quality niskin samples in open water adjacent to the coral monitoring sites.

Water quality data were reef level averages of the period July 2021 to June 2025, noting the satellite derived Chl *a* estimates are wet season only. The only exception being k490 estimates that averaged data during the period July 2020 to June 2024 due to data availability.

Where linear relationships between the coral community index or indicators and the satellite derived water quality measures were indicated, Gradient Boosted Models of the same form as the GLM were fit to provide predictions for plotting that help visualise any nonlinearity in the response. Variability in the response was estimated by bootstrapping 100 models that randomly sampled, with replacement, n estimates of the response, where n represents the total number of reefs. To approximate the Beta distribution the response variables that are constrained to lie between 0 and 1 were rescaled to range between 0.01 and 0.99 before being transformed using the quartile logistic function, qlogis(), in R prior to analysis.

For the subset of reefs at which water quality was measured by the MMP, predictions were derived from the GLM models due to the limited sample size.

### *2.5.5 Relationship between Coral Index scores and environmental conditions*

The response of coral communities to variation in environmental conditions presented in section 4.4.2 was assessed by comparing changes in Coral Index scores to annual discharge and loads of dissolved inorganic nitrogen, particulate nitrogen and total suspended solids from catchments in each region.

For these analyses Generalised Additive Models (GAMs) were applied separately to results from each region. The response variable was the biennial change in the Coral Index score (*I*) at a given reef (r) from one year (y) to the year (y+2). Biennial changes were considered due to the biennial sampling design of the program.

$$\Delta I = I_{ry+2} - I_{ry}$$

Similarly, the discharge and load covariates selected were the greater of the preceding 2 water years. To reduce confounding between the response of the Coral Index scores to acute disturbances, observations of change in the Coral Index at reefs categorised as being influenced by an acute disturbance event in a given biennial period were excluded.

In the first instance, GAMs allowed for the fitting of non-linear responses using natural splines; when these models did not support non-linear response, simple linear models were used.

All GAMs were fit via the mgcv package (Wood 2019) and linear models were fit via the stats package within the R Statistical and Graphical Environment (R Core Team 2023).

# 3 PRESSURES INFLUENCING CORAL REEFS

The condition of coral reefs is affected by a range of environmental pressures. Interpreting the impact of pressures associated with water quality relies on first understanding the impacts of acute pressures such as cyclones, high seawater temperatures that lead to coral bleaching and predation by crown-of-thorns starfish. This section summarises the primary pressures imposed on inshore areas of the Reef in recent years. The impacts of these pressures are spatially variable and summarised at the Reef level in section 4.1 and (sub-)regional level in sections 4.3 to 4.6.

## 3.1 Cyclones

Tropical cyclones frequently cross the inshore Reef. Over the 2024–25 reporting period no cyclones produced damaging waves affecting the regions covered by this report (Figure 8). However, clear storm damage was observed at Pandora, Havannah and Orpheus East in the Burdekin Region. This damage was attributed to high winds associated with an active monsoon trough in early February 2025.

Since 2005, 4 intense systems caused region-wide damage to coral communities:

- cyclone Larry (2006) and cyclone Yasi (2011) both caused damage to Wet Tropics and Burdekin region reefs. The severely impacted reefs at Dunk North and the 2 m depth at Barnards in the Herbert–Tully sub-region are showing clear signs of recovery from these storms (Figure A3). Coral cover at the Barnards has largely returned to the high level observed in 2005. At Palms East in the Burdekin region cyclone Yasi removed almost all the previously high cover of soft corals. The recovery of coral cover at this reef has resulted in a shift in coral community composition with the current community dominated by hard corals of the family Acroporidae (Figure A4)
- cyclone Debbie (2017) caused severe coral loss on reefs in the Mackay–Whitsunday region (Figure 8). Signs of recovery of coral cover in the wake of this cyclone are variable (section 4.5).
- cyclone Jasper (2023) caused high levels of rainfall in Wet Tropics catchments that led to high levels of freshwater discharge and inundation of reefs close to the coast (Table A5). This was especially evident in the Baron-Daintree sub-region (Figure 16e, Table A5).

Numerous smaller cyclones have crossed the inshore Reef over the last decade (Figure 8), causing more moderate and localised damage, the most recent being cyclone Kirrily in 2024.

## 3.2 Sea temperature

Sea temperatures over the 2025 summer were above long-term averages and within the 6-8 degree heating weeks (DHW, NOAA 2018) for which coral bleaching was probable in the northern Wet Tropics and close to the coast in the Burdekin region (Figure 9). No major loss of coral on the reefs monitored were attributed to these 2025 sea temperatures. In contrast, extreme temperature anomalies, with a mean of 12.5 DHW, were recorded at reefs in the Fitzroy region in 2024. During surveys in May 2024 ongoing bleaching and recent mortality of corals was observed at most sites in the Fitzroy region classified as having been impacted by coral bleaching. Further loss of coral cover observed in 2025 was attributed to the mortality of corals bleached in 2024 (Figure 9, Figure 12, Figure 32e). Severe (8-10 DHW) heat stress in 2024 extended north into the Whitsunday Islands and close-inshore areas of the Burdekin region (Figure 9); however, minimal bleaching or loss of coral was observed during our surveys in 2024 in these regions (Figure 13, Figure 25).

Previously, high levels of heat stress occurred in inshore areas south of Hinchinbrook Island in 2020 (Figure 9), and widespread coral bleaching was observed at reefs in the Burdekin and Fitzroy regions during MMP surveys in 2020. High temperatures were also experienced across the MMP reporting area in 2017 but not 2016, when northern areas of the Reef experienced extreme temperatures (Figure 9).

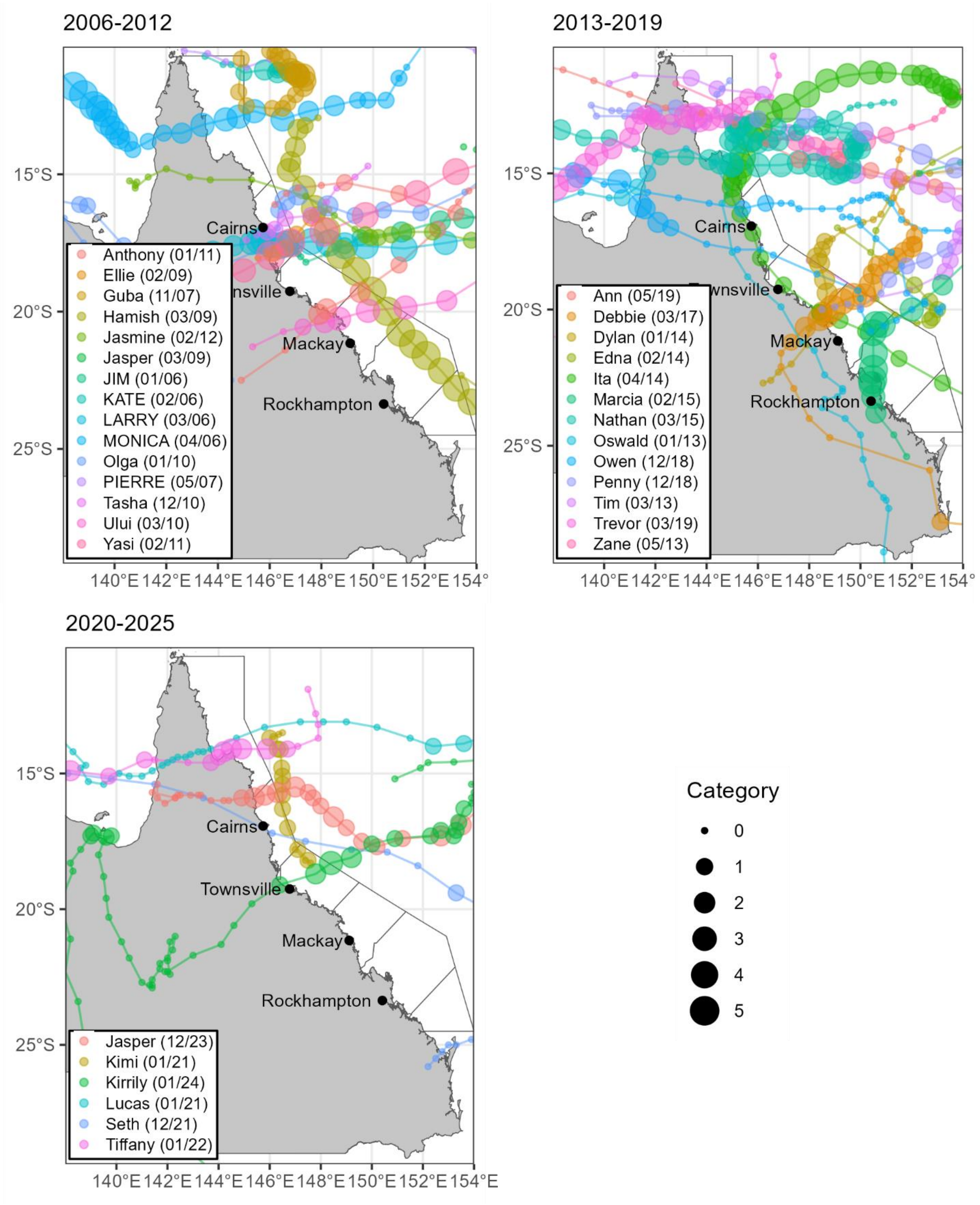
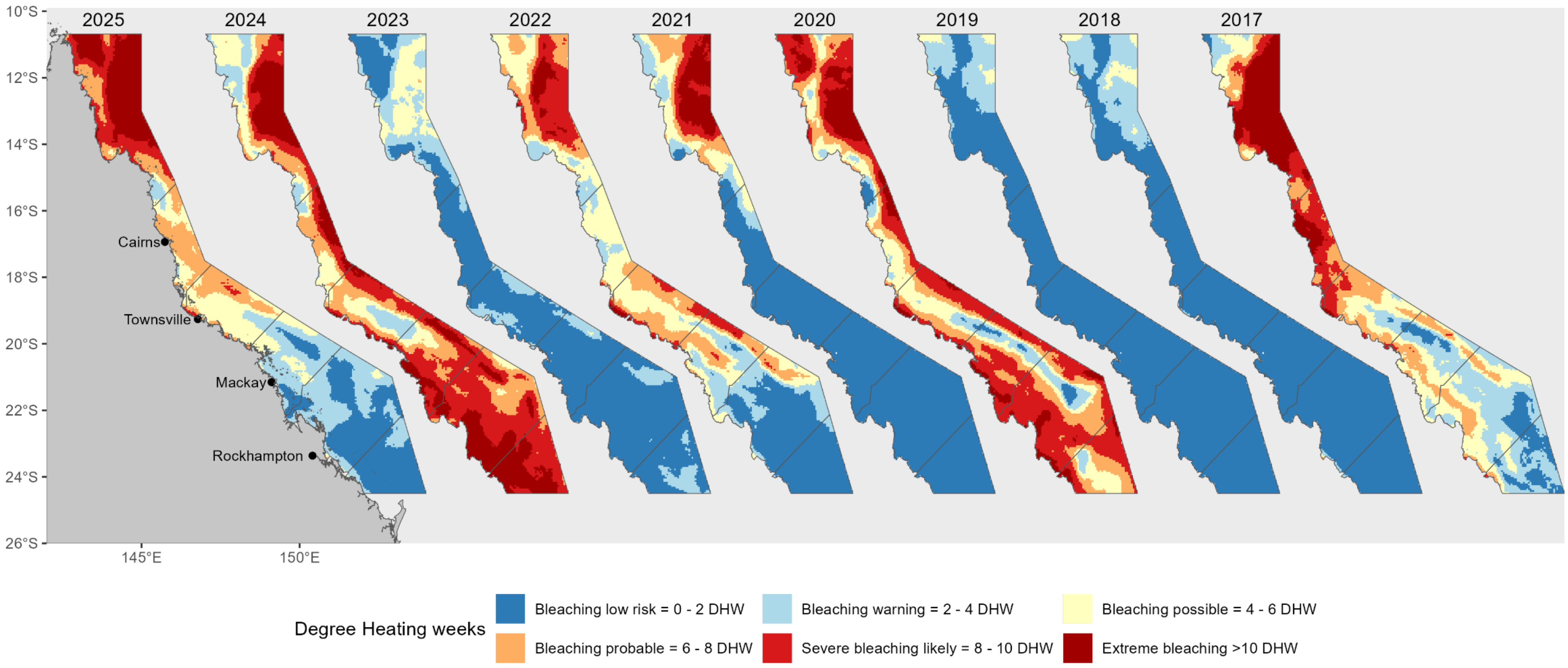



Figure 8. Cyclone tracks for systems crossing the inshore Reef since 2006. Tracks sourced from the BoM

Figure 9. Annual DHW estimates for the Reef. Data are the annual maximum DHW estimates for each ~25 km² pixel. Data were sourced from NOAA Coral Reef Watch.



## 3.3 Crown-of-thorns starfish

In 2025, the density of crown-of-thorns starfish was above outbreak levels at 3 of the 6 MMP reefs in the Johnstone Russell-Mulgrave sub-region (Table 9). A single large individual was also observed at Fitzroy West. Of the inshore reefs reported by the MMP, outbreak densities of crown-of-thorns starfish (30 $ha^{-1}$, following Harriot *et al.* 2003) have only been observed at reefs in the Barron Daintree and Johnstone Russell-Mulgrave focus areas of the Wet Tropics region. Within the Johnstone Russell-Mulgrave focus area, outbreak densities have been observed on at least 2 of the 6 MMP reefs since 2012 (Table 9). In other Regions, single large individuals were recorded at Palms West in both 2019 and 2024, Palms East in 2016 and at Langford Island in the Whitsundays in 2017. However, starfish have been present in the outer Whitsunday Islands in recent years as evidenced by culling data presented in Table 10.

Table 9. Numbers of crown-of-thorns starfish observed along scuba search transects. Numbers presented are the total number observed at the reef summed over sites and, for MMP, depths. Highlighted cells indicate where the density along transects exceeded the threshold of 30 $ha^{-1}$, indicating a population 'outbreak'. * Denotes reefs no longer surveyed by MMP or LTMP.

| | Barron Daintree | | | Johnstone Russell-Mulgrave | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Year | Snapper North | Snapper South | Low Isles | Green | Fitzroy West LTMP | Fitzroy West | Fitzroy East | High East | High West | Franklands East | Franklands West |
| 2005 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2006 | 0 | 0 | | | | 0 | 0 | 0 | 0 | 0 | 0 |
| 2007 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2008 | 0 | 0 | | | | 0 | 0 | | 0 | | 0 |
| 2009 | 0 | 0 | 0 | 0 | 0 | 0 | | 0 | 0 | 0 | 0 |
| 2010 | 0 | 0 | | | | 0 | 0 | | 0 | | 2 |
| 2011 | 0 | 0 | 0 | 7 | 0 | 0 | 0 | 0 | 0 | 0 | 1 |
| 2012 | 30 | 0 | | | | 22 | 14 | | 0 | | 6 |
| 2013 | 23 | 49 | 17 | 5 | 57 | 4 | | 0 | 0 | 0 | 2 |
| 2014 | 0 | 1 | | | | 3 | 3 | | 0 | | 7 |
| 2015 | 0 | 0 | 4 | 0 | 0 | 0 | | 3 | | 1 | |
| 2016 | 0 | 0 | | | | 5 | 1 | 3 | 0 | 6 | 6 |
| 2017 | 0 | 0 | 0 | 0 | 0 | 4 | | 2 | 0 | 7 | 6 |
| 2018 | 0 | 0 | | | | 0 | 0 | 5 | 4 | 3 | 2 |
| 2019 | 0 | 0 | 0 | 0 | 0 | 1 | | 2 | 0 | 3 | |
| 2020 | 0 | 0 | | | | 9 | 5 | 20 | 0 | 30 | 6 |
| 2021 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 3 | 1 | 7 | 3 |
| 2022 | 0 | 0 | 0 | * | 1 | 0 | 0 | 5 | 7 | 0 | 0 |
| 2023 | 0 | 0 | 0 | * | 0 | 0 | 1 | 0 | 0 | 6 | 0 |
| 2024 | 0 | 0 | 0 | * | 1 | 3 | 3 | 5 | 0 | 11 | 6 |
| 2025 | 0 | 0 | 0 | * | 0 | 0 | 4 | 4 | 1 | 9 | 0 |

Within the Johnstone Russell–Mulgrave sub-region, crown-of-thorns densities peaked at outbreak levels (> 30 individuals per hectare) at 5 of the 6 reefs monitored in 2020 (Table 9, Figure A9). The crown-of-thorns starfish, both observed by the MMP and removed by the Reef Authority's Crown-of-thorns Starfish Control Program, consistently ranged across several size cohorts indicating, the ongoing recruitment and survival of crown-of-thorns starfish over recent years (Table 11). In 2025, juvenile starfish were again present, demonstrating their continued recruitment.

Table 10. Number of crown-of-thorns starfish removed. Australian Government Crown-of-thorns Starfish Control Program data supplied by the Reef Authority, Eye on the Reef. Figures in bold are the number of individuals removed in financial year ending in the "Year" field. The catch rate per diver hour is given in brackets to provide an idea of relative population density.

| **Year** | Snapper Island | Low Isles | Fitzroy Island | Frankland Group | Pelorus Island | Border Island | Hayman Island | Hook Island |
|---|---|---|---|---|---|---|---|---|
| 2013 | | | **2564** (10.2) | | | | 1 (0.1) | 0 (0) |
| 2014 | **135** (16.2) | **537** (5.2) | **2546** (13.0) | | | | | |
| 2015 | | **225** (2.5) | **451** (2.4) | | | | | |
| 2016 | | **84** (2.5) | **370** (4.4) | | | | | |
| 2017 | | | **143** (1.0) | **500** (4.3) | | | | |
| 2018 | | | **4** (0.05) | **343** (3.0) | | | | |
| 2019 | | | | | | | | |
| 2020 | | | | | | | | |
| 2021 | | **2** (0.1) | **2999** (4.4) | **6379** (13.1) | | | | |
| 2022 | | | **206** (0.7) | **390** (1.4) | | | **7** (0.4) | **59** (0.7) |
| 2023 | | | **3** (0.03) | **134** (0.6) | | **11** (0.3) | **15** (0.2) | **120** (0.7) |
| 2024 | | | **18** (0.3) | **1450** (2.8) | **2** (0.06) | **1** (0.1) | **8** (0.1) | **109** (0.2) |
| 2025 | | **4** (0.03) | **71** (1.8) | **678** (8.7) | | | **8** (0.3) | **44** (0.3) |

Table 11. Size class distribution of crown-of-thorns starfish on inshore reefs in the Wet Tropics. Included are the percentages culled, as listed in Table 10, of cohorts 1–4, and percentage followed by number observed in parentheses observed during MMP scuba search surveys. Bold font provides a visual cue to the dominant cohort.

| Year | Crown-of-thorns Starfish Control Program | | | | MMP surveys | | |
|---|---|---|---|---|---|---|---|
| | Cohort 1 0-15 cm | Cohort 2 15-25 cm | Cohort 3 25-40 cm | Cohort 4 >40 cm | 0-15 cm | 15-25 cm | >25 cm |
| 2012 | | | | | **54 (39)** | 40 (29) | 6 (4) |
| 2013 | 33 | **45** | 19 | 3 | 17 (13) | **56 (44)** | 27 (21) |
| 2014 | 11 | **44** | 35 | 9 | **57 (8)** | | 43 (6) |
| 2015 | **57** | 17 | 17 | 8 | **75 (3)** | 25 (1) | |
| 2016 | **93** | 6 | 1 | 0 | **67 (15)** | 33 (7) | |
| 2017 | **75** | 23 | 2 | 0 | **55 (11)** | 45 (9) | |
| 2018 | 43 | **51** | 6 | 0 | 14 (2) | 36 (5) | **50 (7)** |
| 2019 | | | | | 33 (2) | **67 (4)** | |
| 2020 | | | | | 27 (19) | **49 (34)** | 24 (17) |
| 2021 | 23 | **63** | 13 | 1 | 6 (1) | 25 (4) | **69 (11)** |
| 2022 | 10 | **63** | 26 | 2 | 8 (1) | 23 (3) | **69 (9)** |
| 2023 | 21 | **74** | 5 | 0 | **57(4)** | 43(3) | |
| 2024 | **63** | 33 | 3 | 0 | **55 (16)** | 45(13) | |
| 2025 | 31 | **62** | 6 | 0 | 22 (4) | **61 (11)** | 17 (3) |

### 3.4 River discharge

Across the Reef, river discharge for the 2024-2025 water year (1st October to 30th September) exceeded 1.5 times the long-term median (Figure 10). Much of the additional discharge was delivered by catchments in the Burdekin Region where most catchments delivered more than 6 times their long-term medians (Table A5). All catchments in the Mackay Whitsunday Region produced annual discharges between 2 -3 times their long-term medians as did the Herbet and Barron rivers in the Wet Tropics (Table A5). All other rivers in the Wet Tropics and rivers in the Fitzroy Region exceeded median discharge levels to a lesser degree (Table A5).

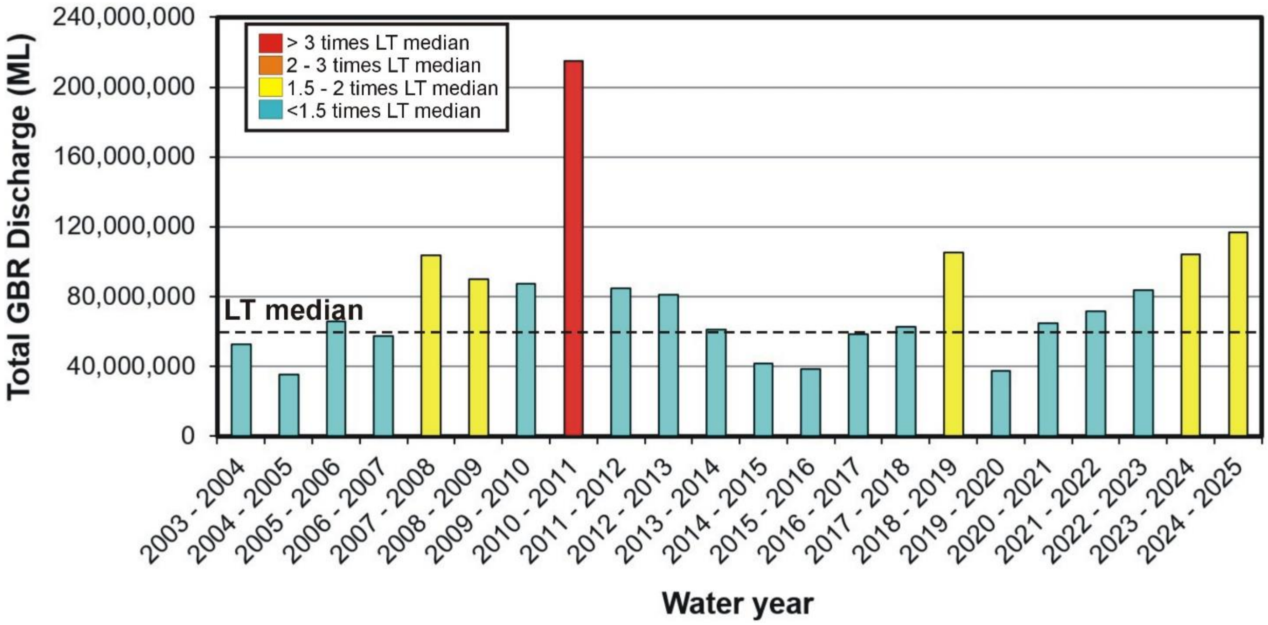



Figure 10. Annual total river discharge to the Reef. Annual estimates aggregate over the water year: 1 October to 30 September, for the 35 main Reef basins. Values are colour coded relative to proportion of long-term (LT) median (1986–2016) discharge. Figure source: Gruber *et al*. 2026, data source: DNRM, http://watermonitoring.dnrm.qld.gov.au/host.htm

Loss of coral attributed to exposure to low salinity floodwaters was observed at 2 m depths on Bedarra, Dunk North, Dunk South, Lady Elliot and Palms West. The proportion of hard coral cover killed at these reefs ranged from 90% at Lady Elliott (cover of hard corals declined from 28% in 2024 to 3% in 2025) through to 36% at Palms West (Figure A3, Figure A4). Lesser levels of coral cover loss attributed to these floods were observed at the 5 m depths on Bedarra, Lady Elliot and Palms West (Figure A3, Figure A4).

The MMP maintains a salinity logger at the Dunk North 2 m depth coral site 2. This logger captured incursions of freshwater to the depth of the coral transects particularly during periods of low tide with salinity reaching as low as 13 PSU (Figure 11). The tidal exposure suggests the more consistent presence of low salinity at shallower depths. During February 2025, the following cumulative exposures were observed: <15 PSU – 6hrs, <22 PSU – 2 days, <28 PSU – 8 days (Figure 11). These can be considered with reference to published salinity thresholds that kill *Acropora* of 3 days at 22 PSU through to 16 days at 28 PSU (Berkelmans *et al.* 2012). The location of the salinity logger is the lee of Dunk Island with sites at Dunk South and Bedarra being more directly exposed to flood plumes (Figure A3).

Previous records of extensive coral mortality include:

- Snapper South in 2024 when all coral at both 2 m and 5 m deep monitoring sites were killed, an event eclipsing the major flooding of the Daintree in 2019 that contributed to 38% of hard coral cover being killed at the 2 m depth only (Figure A1, Table A5).
- Flooding of the Fitzroy River in 2011 that caused high levels of mortality among corals at 2 m depth on reefs to the south of Great Keppel Island (Figure A6, Table A5).

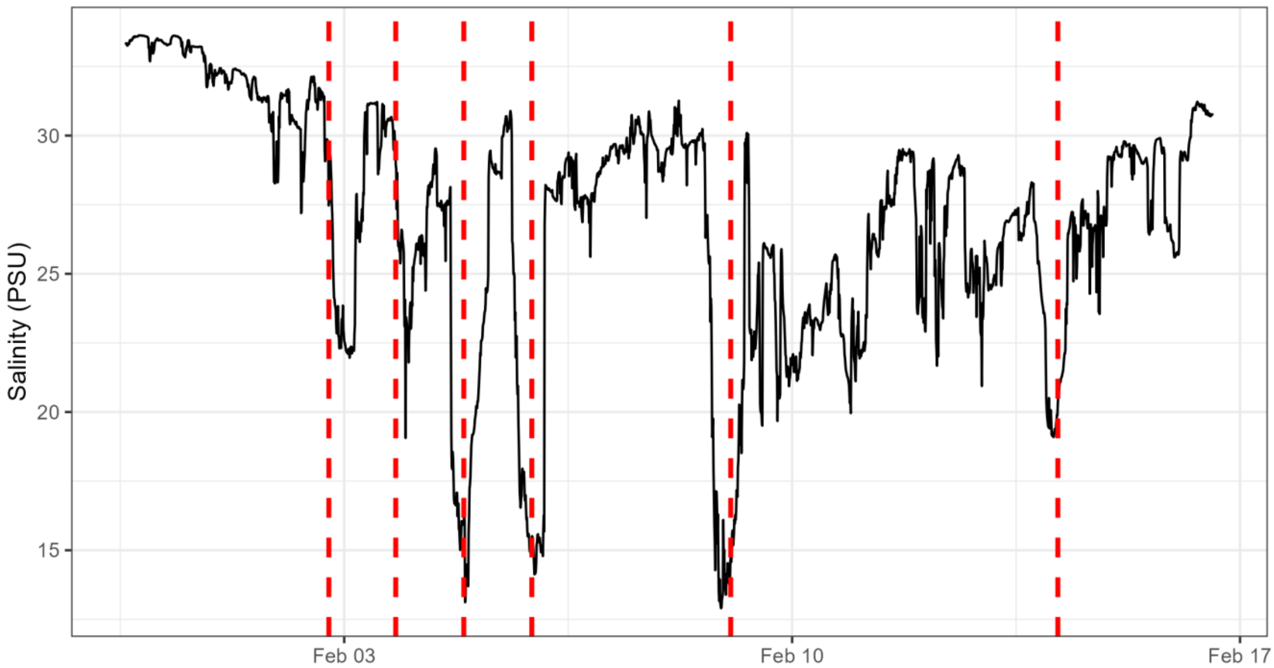



Figure 11. Salinity record for Dunk North 2m depth during February 2025 floods. The salinity profile (black) are 10-minute interval records from a Sea-Bird Electronics 37 salinity logger maintained by the MMP (see Gruber et al. 2026 for details). Red reference lines are positioned at daily lowest tide on days when salinity showed major dips.

The influence of high sediment and nutrient loads are not as overtly obvious as the mortality of corals exposed to freshwater and are explored in terms of suppression of coral recovery and variable condition of coral communities along water quality gradients in section 4.7.

### 3.5 Water quality

Summary plots of water quality (WQ) data for each sub-region or region in which coral monitoring occurs (Figure A11 to Figure A16) are adapted from the complimentrary annual MMP Inshore Water Quality annual report (Gruber *et al*. 2026). For full details of the methods used to create these plots the reader should refer to that report.

Salient points to note are:

- The *long-term* WQ Index relates to the sampling design implemented in the early years of the program — prior to 2015. To account for variation due to relatively few samples per year in the early design, a four-year running mean is applied to annual scores.
- The *annual condition* WQ Index is applied to the full sampling design implemented in 2015 and annual scores are the means for that year only.
- For both indices, each observation of the individual water quality indicators is scored relative to guideline values and aggregated hierarchically to derive Index scores at the scale of the sampling site, then sampling sub-region and region.
- The time-series of data presented for individual water quality indicators and their modelled predictions are based on observations that are detrended to account for the influence of tides, winds and season.

Within section 4 of this report, reference to trends in indicators or deviations from guidelines follow the convention applied by Gruber *et al*. (2026). Reference to trends in any water quality parameter relate to observation of a linear trend in genralised additive mixed models (GAMM) with a slope that deviates beyond zero as assessed by upper or lower 95% confidence interval of that slope. In contrast statements relating to current levels of a parameter relative to guidleline values are based on the observed mean, or median, (depending on the central tendency measure stipulated for each indicator in the guidelines) being above or below the annual guideline value.

# 4 CORAL COMMUNITY CONDITION AND TRENDS

Results are presented in the following sequence:

- Reef-wide coral community condition (Coral Index scores) and trend (4.1)
- Reef-wide relative impact of disturbances (4.2)
- Coral community condition (Coral Index scores) and trend in each (sub-)region (4.3)
- Coral community condition along water quality gradients (4.4)

Pressures and current coral community condition differ among and within regions. As such, temporal trends in community attributes are presented for each (sub-)region along with time-series of data relating to the primary pressures influencing coral communities.

Finally, site-specific data and tables with additional information are presented in Appendix 1. Time-series of community condition and composition for each reef monitored are also available online at http://apps.aims.gov.au/reef-monitoring/.

## 4.1 Reef-wide coral community condition and trend

At the whole of the inshore Reef-scale, the Coral Index score in 2025 remained ‘poor’ although improved slightly since reaching the lowest value recorded in 2024 (Figure 12). Spatially, the improvement in 2025 was driven by the Mackay–Whitsunday Region where coral communities have continued to recover from the severe impact of cyclone Debbie in 2017 (section 4.5). The magnitude of improvement in the Mackay–Whitsunday Region offset declines in Coral Index scores in each of the other regions (sections 4.3, 4.4, 4.7).

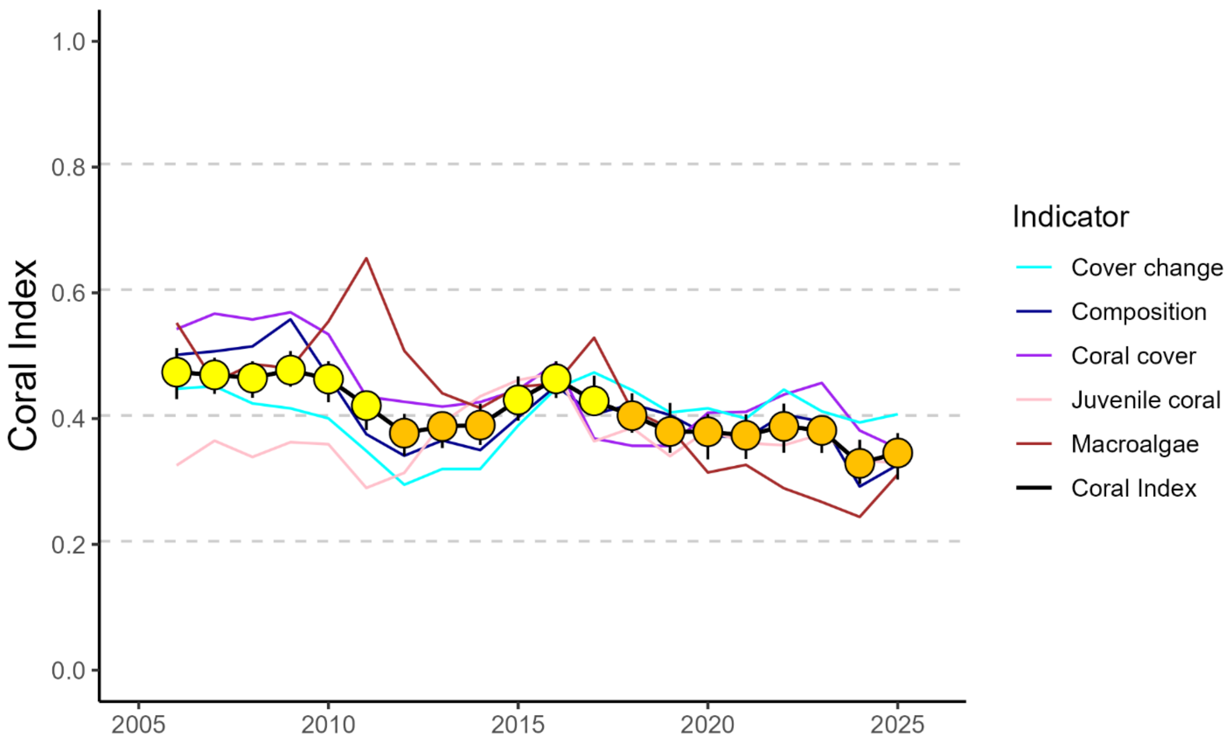



Figure 12. The Reef level trend in Coral Index and indicator scores. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = ‘poor’, yellow = ’moderate’. Error in Coral Index scores were derived from bootstrapped distribution of regional indicator scores weighted by the relative area of inshore coral reefs in each region.

The most improved indicator in 2025 was the Macroalgae score, again heavily influenced by marked improvement in the Mackay–Whitsunday Region (section 4.5). Macroalgae scores also improved in the Burdekin Region and Herbert–Tully sub-region and in both cases, these improvements may be temporary as both macroalgae and corals were impacted by flooding (sections 4.3.4, 4.4).

Previously, when the cover of macroalgae been reduced by an acute disturbance event, it has tended to rapidly rebound, as can be seen by the brief peaks in Macroalgae scores in 2011 and 2017 (Figure 12). Only scores for the Coral cover indicator declined at the Reef-scale, reflecting losses attributed to exposure to low salinity flood plumes in early 2025 in the Herbert–Tully sub-region and Burdekin Region, with storm damage also contributing to losses in the Burdekin Region (sections 4.3.4, 4.4). In the Fitzroy region, the further loss of coral cover reflected the longer-term impact of high water temperatures that led to severe coral bleaching and loss of coral cover in 2024 (section 4.6).

Leading up to 2016, the recovery of coral condition demonstrated the inherent resilience of the inshore coral communities following a period punctuated by the impacts of cyclones and high discharge from the Reef's catchments. Since 2016, Coral Index scores declined in the face of multiple disturbances (Figure 12). Despite the current uptick in the Macroalgae score, the long-term decline in this indicator remains of particular concern as it suggests increasing downward pressure on coral community recovery (Figure 12).

Ultimately, the Reef level coral community condition reflects large-scale averages and overall responses of coral communities exposed to varied past and ongoing pressures. The following sections explore results at finer spatial resolution. However, what is clear from the Reef-level disturbance time-series is that, since 2005 inshore reefs have experienced impacts from a range of disturbance events that have outweighed the coral community's ability to recover.

## 4.2 Reef-wide relative impact of disturbances

The most directly observable impact of acute disturbance events is the loss of coral cover. Over the period of the MMP, cyclones and storms remain the primary cause of coral mortality, accounting for 37.2% of coral cover losses on inshore reefs (Figure 13,Table A 6). Unsurprisingly, the intense category 4 and 5 systems: cyclone Larry (Wet Tropics and Burdekin regions – 2006), cyclone Yasi (Wet Tropics and Burdekin regions – 2011) and cyclone Debbie (Whitsunday region – 2017), have caused the greatest losses. Minor loss of coral in recent years has been attributed to cyclone Jasper in 2023 and cyclone Kirrily in 2024, and rough conditions associated with the monsoon in the Burdekin Region in February 2025 (Figure 13).

When interpreting Figure 13 it is important to note that until 2021 both the LTMP and MMP included biennial sampling designs (Table 3). While the MMP did infill sampling in cases when acute disturbances were likely, missing samples can result in a lagged attribution of coral loss to disturbance events. For example, loss of coral cover attributed to cyclone Debbie (March 2017) is represented in 2017 when 6 of the 7 impacted MMP reefs were resurveyed, in 2018 when the final MMP reef was resurveyed, and in 2019 when the LTMP reefs in the region were resurveyed. In contrast, delayed response to bleaching events in 2017, 2020 and 2024 are represented by losses attributed to bleaching in 2018, 2021 and 2025, respectively (Figure 13). In these instances, corals were still bleached at the time of surveys in the months following severe thermal stress, and the subsequent loss of cover was attributed to a delayed response.

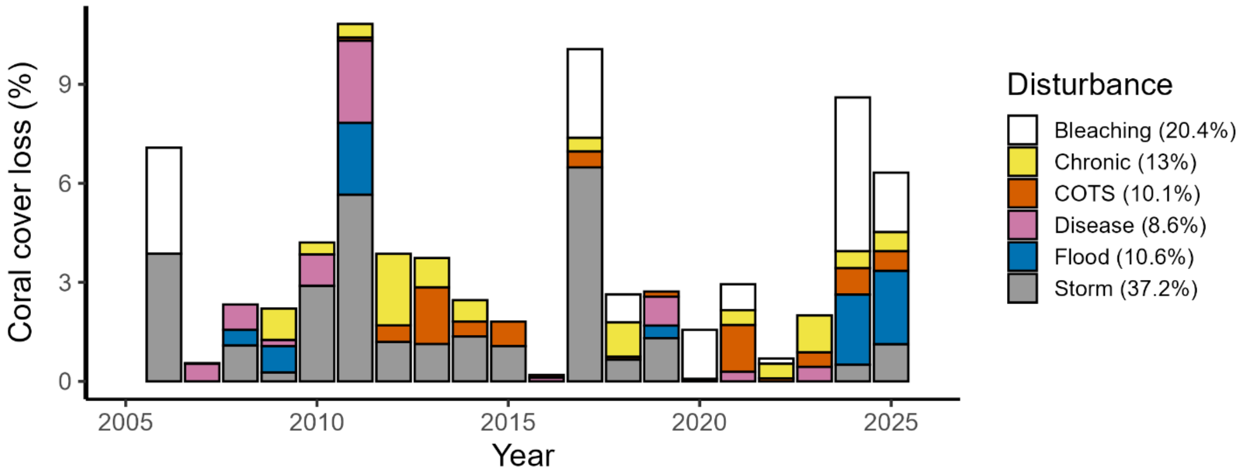



Figure 13. Hard coral cover loss by disturbance type across the inshore Reef. Length of bars represents the mean loss of cover across all reefs in each year. Colours represent the identified cause of cover loss. COTS = crown-of-thorns starfish

Thermal bleaching events have contributed to 20.4% of the coral cover losses since 2005 (Figure 13). This figure has increased from the 14.7% reported for 2023 due to the severe impacts observed in 2024, particularly in the Fitzroy region. Previous marine heatwaves resulting in reductions in coral cover occurred in 2006, 2017, 2020 and, to a lesser extent, 2022 (Figure 13). It is likely that some losses of cover recorded as disease and chronic stressors include the longer-term impacts of prior acute events.

While crown-of-thorns starfish have caused moderate losses (10.1%, Figure 13), their potential impact has been reduced by the removal of starfish by the Reef Authority's Crown-of-thorns Starfish Control Program (Table 10). These figures contrast with those from more offshore areas where crown-of-thorns starfish (Osborne *et al.* 2011, De'ath *et al.* 2012), and more recently thermal bleaching (Hughes *et al.* 2018), are recognised as major contributors to loss of coral cover.

Flooding of the Daintree River, associated with cyclone Jasper (2023), caused the single most extreme disturbance to inshore reefs documented by the MMP, with all coral killed at both 2 m and 5 m depths at Snapper Island South (Figure A1). In 2025 mortality of corals at several reefs in the Burdekin Region and Herbert–Tully sub-region was also attributed to exposure to low salinity floodwaters. In combination with the impacts of flooding of the Daintree River in 2024 the proportion of coral cover lost due to direct exposure to floodwaters has risen to 10.6% (Figure 13). Prior to this, loss of corals from direct exposure to low salinity floodwaters had been limited to 2 m depths on reefs closest to rivers during major flood events. This is unsurprising, as more frequent exposure would be expected to preclude reef development. Indeed, 2 of the 3 reefs most impacted, Peak Island and Pelican Island in the Fitzroy region, demonstrate minimal development of a carbonate substrate. It is for this reason that Peak Island was removed from the program in 2020. All other reefs included in the LTMP and MMP were selected to capture areas where development of a carbonate substrate provides evidence for historical reef building capacity of corals.

In combination, the acute disturbance events listed above contribute strongly to the declines in coral cover and, by extension, Coral Index scores in all regions.

## 4.3 Regional Coral Index and indicator trends

### *4.3.1 Wet Tropics*

Coral communities within inshore areas of the Wet Tropics remain in ‘moderate’ condition for 2025. However, the Coral Index score has declined each year since 2022, a trend consistent among all indicators (Figure 14). In 2025 the Juvenile coral indicator declined into the “poor” range for the first time since 2012 when juvenile scores were recovering from a low point in 2011. This 2011 low was caused by the impact of Cyclone Yasi that reduced not only the number of juveniles but also the cover of corals, with a corresponding increase in the cover of turf algae, causing a double blow to the juvenile score that is based on the density of juveniles per unit area of algae-covered substrate (Figure 14). The current reduction of Juvenile score is similarly influenced by both lower numbers of juveniles and the increase in the cover of algae due to recent loss of coral.

The relatively stable condition observed from 2016 to 2022 (Figure 14) masks differing trends among sub-regions with the over-all condition reflecting a range of minor disturbances that have variously impacted reefs among the sub-regions and prevented region-wide improvement, as detailed in the following sections.

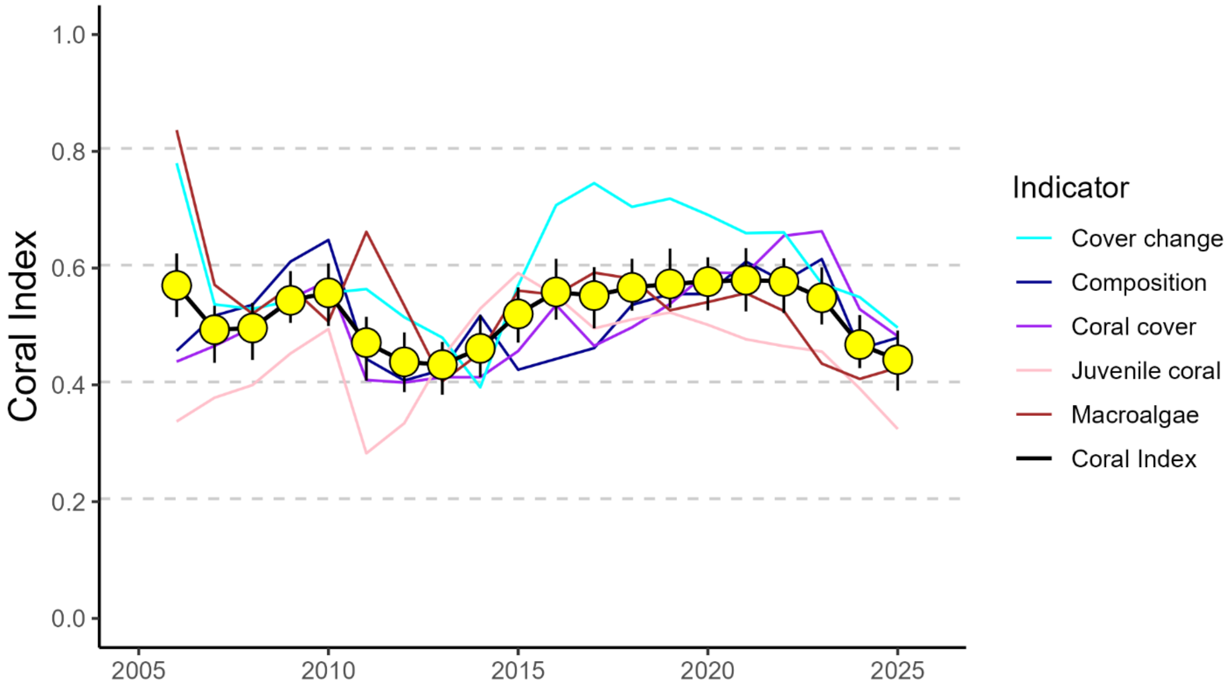



Figure 14. Coral Index and indicator trends for the Wet Tropics region. Coral Index scores are coloured by report card category: yellow = ’moderate’. Error in Coral Index scores was derived from bootstrapped distributions of indicator scores at individual reefs.

#### *4.3.1.1 Barron–Daintree sub-region*

The condition of coral communities remained ‘poor’ having declined slightly in 2025 (Figure 15). In December 2023, cyclone Jasper caused both physical damage to corals and precipitated extreme rainfall that caused major flooding (Figure 8). At Snapper Island all corals on the southern sites were killed as the reef was inundated by the freshwater plume from the Daintree River (Figure 8, Figure 16e, Table A5). The magnitude of these impacts on the Coral Index was not fully realised in 2024 due to a temporary improvement in the Macroalgae indicator score that has declined in 2025 as levels macroalgae increased (Figure 15, Table 12). Until 2023, Coral Index scores had been improving from a low point in 2019 following both coral bleaching in 2017 and exposure to floodwaters and cyclone Owen in 2019 (Figure 16e, Table 12, Table A 6).

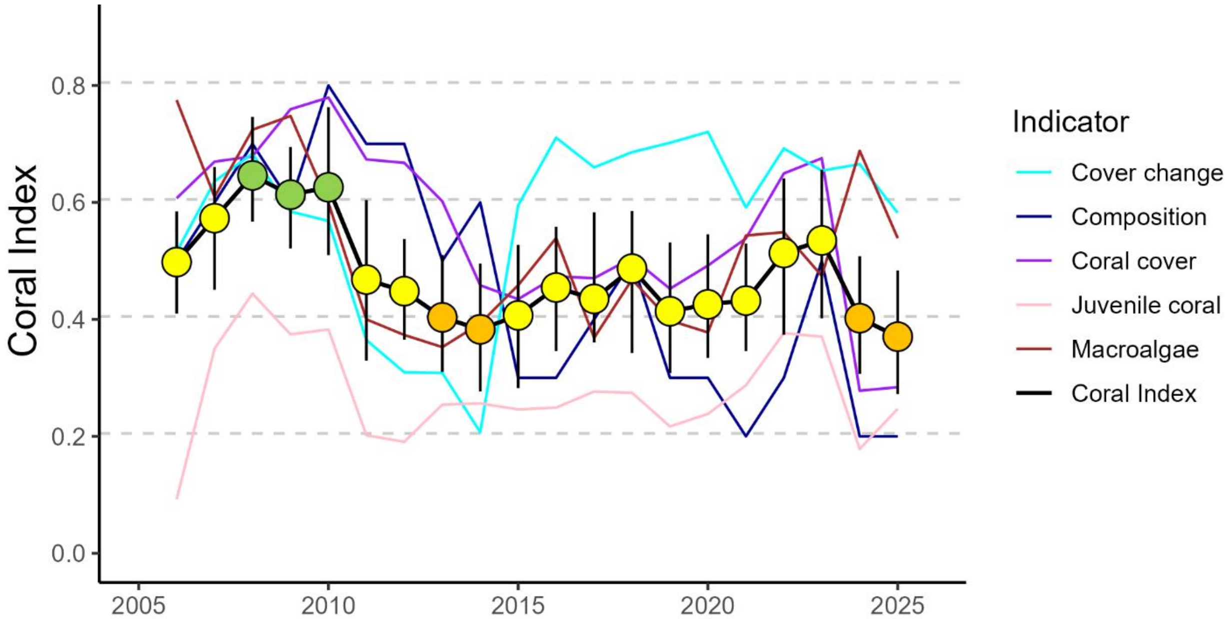



Figure 15. Coral Index and indicator trends in the Barron–Daintree sub-region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = ‘poor’, yellow = ’moderate’ and green = ’good’. Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at individual reefs.

Table 12. Coral Index and indicator score changes in the Barren–Daintree sub-region. Data represent the changes in scores between sub-regional maxima and minima in the Coral Index time-series (Figure 15). For the Coral Index, and each indicator, the observed change in the score and the probability that the change was greater or less than zero (no change) are presented. Shading is used as a visual aid to highlight the magnitude of the probability (P) that the score improved (blue shades) or declined (red shades). Probabilities are derived from the posterior distribution of observed score changes at each reef and depth.

| Period | Depth (m) | Coral Index | | Coral cover | | Macroalgae | | Juvenile coral | | Cover change | | Composition | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Score | P | Score | P | Score | P | Score | P | Score | P | Score | P |
| 2008 to 2014 | 2 | -0.21 | 0.89 | -0.36 | 0.71 | -0.17 | 0.76 | -0.41 | 0.93 | -0.62 | 0.99 | 0.50 | 1.00 |
| | 5 | -0.30 | 0.88 | -0.13 | 0.61 | -0.44 | 0.81 | -0.04 | 0.58 | -0.38 | 1.00 | -0.50 | 1.00 |
| 2014 to 2018 | 2 | -0.03 | 0.80 | 0.12 | 0.93 | -0.18 | 0.76 | -0.09 | 0.73 | 0.52 | 0.99 | -0.50 | 0.76 |
| | 5 | 0.19 | 0.97 | 0.00 | 0.51 | 0.24 | 0.75 | 0.09 | 0.70 | 0.45 | 0.95 | 0.17 | 0.73 |
| 2019 to 2023 | 2 | 0.19 | 0.83 | 0.26 | 1.00 | -0.08 | 0.76 | 0.48 | 0.79 | 0.02 | 0.72 | 0.25 | 0.77 |
| | 5 | 0.08 | 0.66 | 0.20 | 0.97 | 0.18 | 0.83 | -0.06 | 0.64 | -0.09 | 0.58 | 0.17 | 0.73 |
| 2023 to 2025 | 2 | -0.26 | 0.84 | -0.47 | 0.88 | 0.08 | 0.77 | -0.52 | 0.78 | -0.16 | 1.00 | -0.25 | 0.76 |
| | 5 | -0.10 | 0.63 | -0.34 | 0.76 | 0.05 | 0.75 | 0.14 | 0.66 | -0.01 | 0.51 | -0.33 | 0.72 |
| 2024 to 2025 | 2 | -0.01 | 0.70 | 0.03 | 0.77 | 0.00 | NA | 0.01 | 0.77 | -0.08 | 1.00 | 0.00 | NA |
| | 5 | -0.05 | 0.67 | -0.01 | 0.61 | -0.25 | 0.83 | 0.11 | 0.83 | -0.09 | 0.65 | 0.00 | NA |

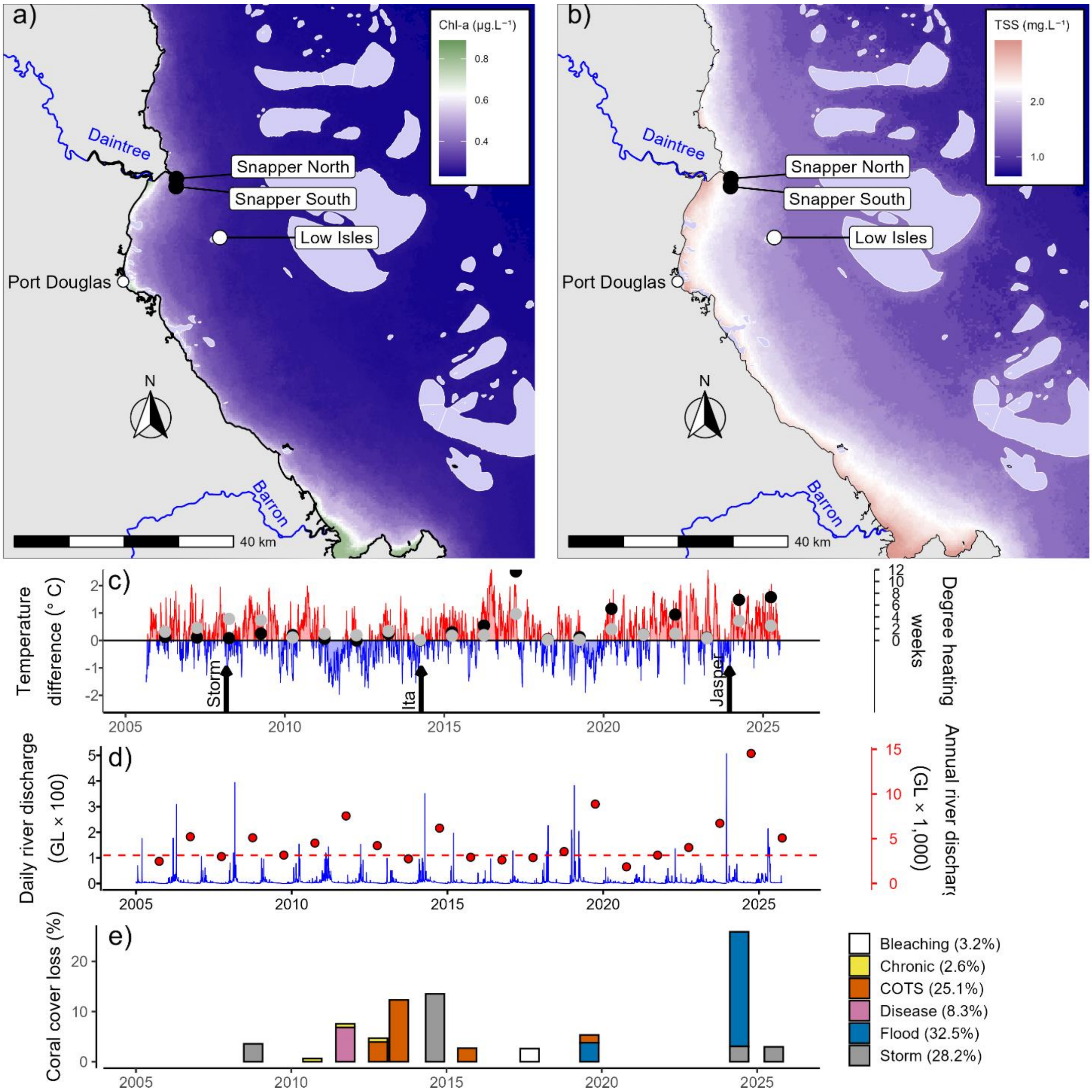



Figure 16. Environmental pressures in Barron–Daintree sub-region. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a), median wet season Chl *a* and b), median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2021–2025, white breaks in the colour gradients are set at wet-season guideline values for open coastal waters. c), Seasonally adjusted temperature deviation, timing of cyclones and storms indicated by black arrows, accumulated DHW over the summer period (1 December – 31 March) as reported by NOAA (black symbols) and derived from *in situ* loggers (grey symbols). d), Combined daily (blue) and annual water year – October to September (red) discharge for the Daintree and Barron basins, red dashed line represents long-term median discharge (1986–2016). e), break-down of hard coral cover loss by disturbance type; length of bars represents the mean loss of cover across all reefs in the sub-region.

In December 2023 cyclone Jasper crossed the North Queensland coast with associated floodwaters impacting reefs in the Barron–Daintree sub-region through to January 2024 (Figure 16e, Table A5). Across the 2023-24 wet season, all rivers in the Wet Tropics region exceeded their median flow by more than 1.5 times, with the greatest exceedance in this sub-region with the Daintree and Baron rivers exceeding median flows by 4.8 and 5.8 times respectively (Table A5). Loss of coral cover between 2023 and 2024 was attributed primarily to flood damage caused by the direct exposure of corals to low salinity floodwaters from the Daintree River, however, some storm damage also occurred (Table A 6). The storm damage recorded for 2025 occurred at Low Isles where the 2025 survey was the first survey of that reef since cyclone Jasper (Figure 16e).

The Coral cover indicator score remained categorised as ‘poor’ (0.28, Table A7, Figure 15), having decreased significantly from ‘good’ in 2023 and plateauing between 2024 and 2025 (Table 12, Figure 17a). From 2019 to 2023 this indicator had steadily improved across both depths (Table 12, Figure 16e). In 2024 all corals at both 2 m and 5 m depths at Snapper South were killed as the reef was inundated by freshwater associated with cyclone Jasper (Figure A1,Table A 6)

Corals at the 2 m depth at Snapper North were also impacted by cyclone Jasper through a combination of storm damage and freshwater exposure (Figure A1, Figure A7, Table A 6). Minor levels of coral bleaching were also observed. At a sub-regional level more than half the coral cover was lost, with mean cover of hard corals and soft corals combined declining from 52% in 2023 to 23% in 2024. The slight recovery in coral cover that occurred at Snapper North was masked by the overall lack of improvement in coral cover in 2025 once the impact of cyclone Jasper at Low Isles was quantified (Figure 17a). Coral cover remains zero at Snapper South where recovery will depend on the gradual recruitment and growth of juvenile corals.

The Cover change indicator declined to ‘moderate’ (0.58, Table A7) as the high rates of recovery observed until 2023 have been moderated by slower recovery at Snapper North and the lack of recovery at Snapper South over the last year (Table 12). The score at Snapper North 2 m depth declined to ‘moderate’ and Snapper South 5 m declined to ‘poor’ in 2025 (Table A7).

The Composition indicator remains ‘poor’ in 2025, showing no change from 2024 (0.33, Table A7, Figure 15). This result reflects declines at Snapper South 5 m depth where all coral was killed in 2024 (Figure A1, Table A9, Table A10). The scores did not change at other reefs and remained at 0.5, except at Snapper North (2 m depth), where the score was 0 (Table A7). That the score did not decline at the 2 m depth at Snapper South, despite the loss of all corals, is an artifact of the method. The score for this indicator is based on the relative abundance of corals that are more commonly found on reefs in less turbid, lower nutrient waters compared to those found in areas of poor water quality. The baseline condition at Snapper South was for a community that was neutral, with a mix of coral genera found across water quality gradients. Where there are no corals, the method also locates the community in a neutral state.

The Macroalgae indicator has declined to 'moderate’ (0.54, Table A7) after it briefly spiked to ‘good’ in 2024 (Figure 15). In 2024 the impacts of cyclone Jasper and associated floodwaters killed some of the macroalgae at Snapper Island sites causing a short improvement in the score for this indicator. This improvement was still evident in 2025 with an overall improvement of the Macroalgae indicator from 2023 to 2025 (Table 12). However, at Snapper South 5 m depth macroalgae cover had increased to over 17% in 2025 (Figure A1), demonstrating rapid colonisation over the last year, particularly among red macroalgae species (Table A 11).

The Juvenile coral indicator has improved over the last year to re-enter the ‘poor’ range (0.25, Table A7, Figure 15, Table 12). This improvement was largely driven by the density of juveniles recorded at Low Isles (Figure A1). In contrast, the density of juvenile corals remains very low at all Snapper Island locations where scores remain ‘very poor’ (Figure A1, Table A7).

2 back-to-back wet years have impacted Water Quality with the short-term water quality index falling to ‘moderate’ in 2025 (Figure A11b). The concentration of $NO_x$ continued to exceed guideline values but has remained stable since 2015 (Figure A11d). Over the past 2 years the concentration of Phosphate and Chl *a* increased to breach guideline values for the first time since 2017 (Figure A11c, e). While the trend of Chl *a* since 2015 remains stable, the trend for Phosphate and particulate N has deteriorated (Gruber *et al*. 2026). It should be noted these water quality data include sampling from before, during and after the wet season (Gruber *et al*. 2026). Over the period 2021–2025, wet-season concentrations of Chl *a* and TSS, as estimated from satellite imagery, were below wet-

season guideline values at all coral monitoring locations except TSS at Snapper North and South (Figure 16a, b, Table A8).

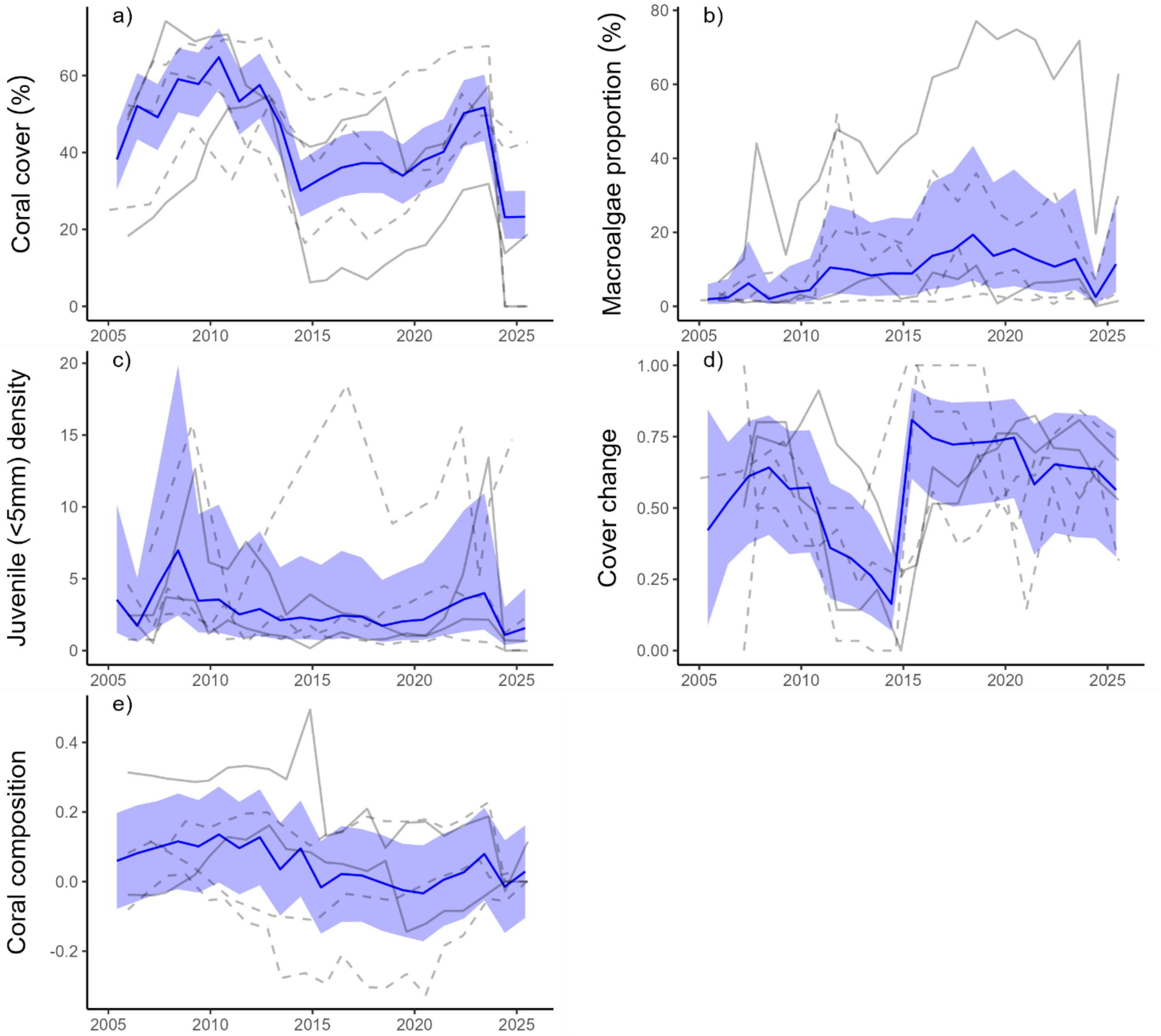



Figure 17. Indicator trends in the Barron–Daintree sub-region. Temporal trends in observed values of a), live coral cover, b), macroalgal proportion, c), juvenile coral density, e), coral composition that inform indicators and d), derived indicator score for change in cover. Data are sub-regional averages (blue lines) bound by 95% confidence intervals of those trends (shading). Grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

#### *4.3.1.2 Johnstone Russell-Mulgrave sub-region*

The 2025 Coral Index score remained 'moderate' and largely unchanged from 2024 (Figure 18,Table 13). The score in 2025 remains well below a 'good' categorisation, which was last recorded in 2021. Contributing to this categorisation were consistent declines in Macroalgae and Coral Cover indicators, Juvenile coral at 2 m depth, and Cover change at 5 m depth (Table 13). The decline in Coral cover was most noticeable in 2024 (Figure 18). Previously, the Coral Index had recovered from a low point in 2012, following severe damage to coral communities caused by cyclone Yasi, and high levels of coral disease (Figure 18, Figure 19e), and later stabilised between 'moderate' and 'good' from 2016 to 2021 (Figure 18).

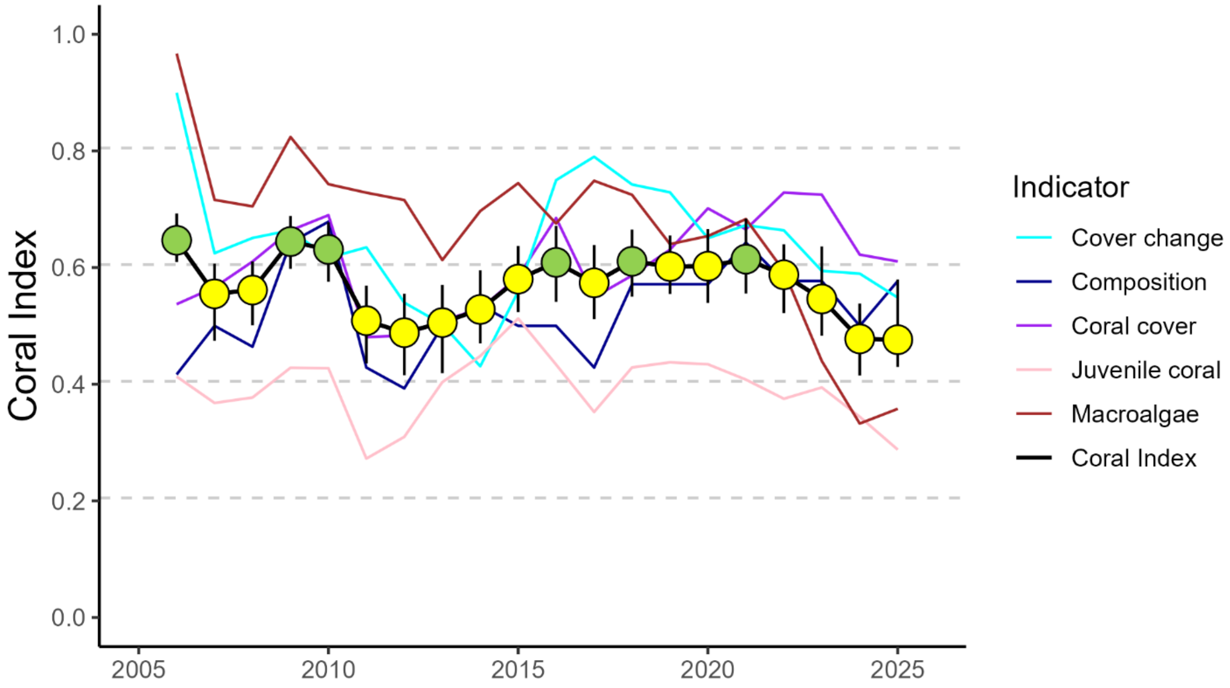



Figure 18. Coral Index and indicator trends in the Johnstone Russell–Mulgrave sub-region. Coral Index scores are coloured by Reef Water Quality Report Card categories: yellow = 'moderate' and green = 'good'. Error in Coral Index scores was derived from bootstrapped distributions of indicator scores at individual reefs.

Table 13. Coral Index and indicator score changes in the Johnstone Russell–Mulgrave sub-region. Data represent the changes in scores between sub-regional maxima and minima in the Coral Index time-series (Figure 18). For the Coral Index, and each indicator, the observed change in the sub-regional score and the probability that the change was greater or less than zero (no change) are presented. Shading is used as a visual aid to highlight the magnitude of the probability that the score improved (blue shades) or declined (red shades). Probabilities are derived from the posterior distribution of observed score changes at each reef and depth.

| Period | Depth | Coral Index | | Coral cover | | Macroalgae | | Juvenile coral | | Cover change | | Composition | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Score | P | Score | P | Score | P | Score | P | Score | P | Score | P |
| 2009 to 2012 | 2 | -0.21 | 0.93 | -0.24 | 0.85 | -0.21 | 0.70 | -0.12 | 0.80 | -0.21 | 0.70 | -0.25 | 0.73 |
| | 5 | -0.12 | 0.76 | -0.14 | 0.87 | -0.03 | 0.55 | -0.12 | 0.82 | -0.06 | 0.55 | -0.25 | 0.71 |
| 2012 to 2016 | 2 | 0.20 | 0.92 | 0.28 | 0.93 | 0.04 | 0.56 | 0.07 | 0.92 | 0.26 | 0.68 | 0.33 | 0.80 |
| | 5 | 0.05 | 0.66 | 0.14 | 0.77 | -0.10 | 0.73 | 0.16 | 0.82 | 0.22 | 0.69 | -0.06 | 0.54 |
| 2016 to 2021 | 2 | -0.04 | 0.57 | -0.02 | 0.53 | -0.02 | 0.51 | -0.02 | 0.66 | -0.14 | 0.67 | 0.00 | 0.50 |
| | 5 | 0.05 | 0.79 | -0.02 | 0.55 | 0.03 | 0.53 | -0.03 | 0.58 | -0.01 | 0.51 | 0.25 | 0.76 |
| 2021 to 2025 | 2 | -0.15 | 0.90 | -0.14 | 0.88 | -0.36 | 0.82 | -0.09 | 0.80 | -0.05 | 0.55 | -0.08 | 0.67 |
| | 5 | -0.16 | 0.94 | -0.05 | 0.73 | -0.40 | 0.87 | -0.06 | 0.63 | -0.21 | 0.77 | -0.07 | 0.65 |

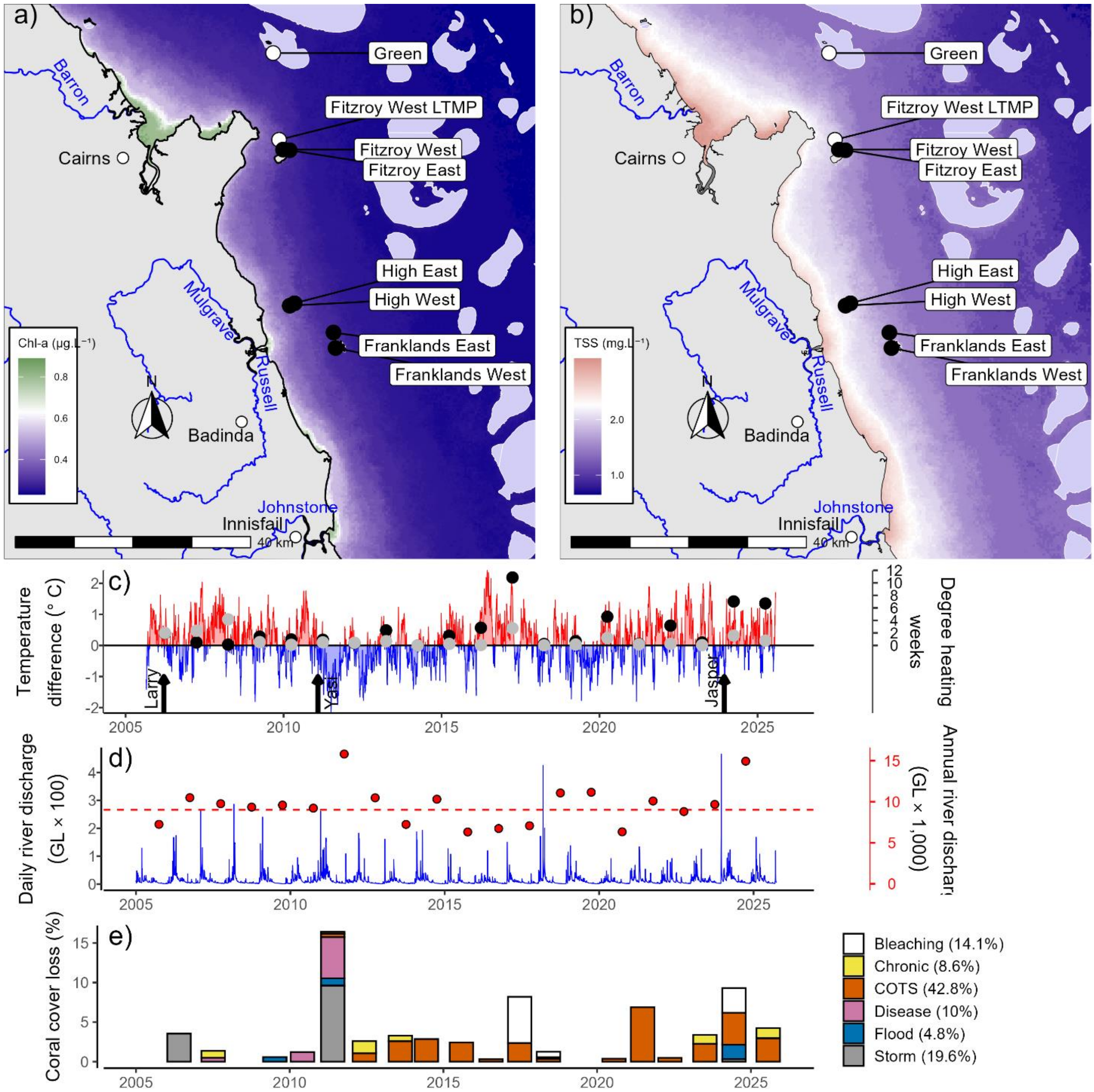



Figure 19. Environmental pressures in the Johnstone Russell–Mulgrave sub-region. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a), median wet season Chl *a* and b), median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2021–2025, white breaks in the colour gradients are set at wet-season guideline values for open coastal waters. c), Seasonally adjusted temperature deviation, timing of cyclones and storms indicated by black arrows, accumulated DHW over the summer period (1 December – 31 March) as reported by NOAA (black symbols) and derived from *in situ* loggers (grey symbols). d), Combined daily (blue) and annual water year – October to September (red) discharge for the North Johnstone, South Johnstone, Russell and Mulgrave basins, red dashed line represents long-term median discharge (1986–2016). e), break-down of hard coral cover loss by disturbance type; length of bars represents the mean loss of cover across all reefs in the sub-region.

The decreasing Coral Index in this sub-region since 2021 reflects the ongoing predation of corals by crown-of-thorns starfish and, in 2024, the combined effects of cyclone Jasper and elevated sea temperatures that caused coral bleaching (Figure 9, Figure 19e,Table A 6). Cyclone Jasper caused substantial damage to coral communities not only via physical action, but by the associated flooding, with the Russell-Mulgrave and Johnstone rivers exceeding their median flow by 1.6 and 1.7 times, respectively (Table A5). Where reductions in coral cover weren’t attributed to crown-of-thorns starfish

(Table A 6), the combination of these recent events likely contributed to the below expected rate of hard coral cover recorded as “Chronic” (Figure 19e).

In 2025, crown-of-thorns starfish were above outbreak levels at 3 reefs, Fitzroy East, High East and Franklands East (Table 9). At High West a single, large crown-of-thorns starfish was observed. Feeding scars were observed at all reefs in the subregion during the most recent surveys. The numbers seen in 2025 were down compared to 2024 (Table 9). During the 2024–2025 financial year the Crown-of-thorns Starfish Program removed 678 starfish from the Frankland group and 71 from Fitzroy Island (Table 10).

The Coral cover indicator score for 2025 remained categorised as ‘good’ (0.61, Table A7), despite an overall decline from 2021 (Figure 18, Table 13). Recent declines in coral cover were attributed to the persistent presence of crown-of-thorns starfish, a combination of flooding and bleaching in 2024, and chronic pressures that reduced the rate of coral cover increase, particularly between 2024 and 2025 (Figure 19e,Table A 6).

Between 2024 and 2025 the minimal change in subregional average coral cover masked variation in change among individual reefs. For example, there was a slight increase recorded at Fitzroy West that contrasted with declines at 5 m depth at both the High East and High West sites, and both depths at Franklands West where the decrease in coral cover coincided with an increase in the cover of macroalgae (Figure A2).

The increase in coral cover at Fitzroy West LTMP between 2023 and 2024 preceded the elevated sea temperatures and passage of cyclone Jasper over the 2023–24 summer (Figure A2).

The Cover change indicator score remained ‘moderate’ (0.56, Table A7) in 2025, albeit well below 2021 values when the Coral Index was at a recent high point, especially at 5 m depths (Figure 18, Table 13). In 2025, ‘poor’ or ‘very poor’ scores for Cover change were recorded for High East, Fitzroy East at 2 m, and the deeper slope of Fitzroy West monitored by the LTMP, indicating recent rates of increase in hard coral cover have been below modelled expectations (Table A7).

The Composition indicator has remained ‘moderate’ in 2025 (0.58, Table A7, Figure 18) indicating not consistent change in coral communities despite the ongoing presence of crown-of-thorns starfish. Scores of zero for this indicator tend to reflect reduced representation of *Acropora* compared to that observed at a reef in the first 5 years of the program — the baseline period for this indicator. In 2025 scores of zero were recorded at both depths of High West where the cover of *Acropora* cover was estimated at 0.4% at 2 m depth and 0.1% at 5 m depth (Table A9), below levels observed between 2005 and 2009 (Figure A2). A score of zero was also recorded at Fitzroy East 5 m where the proportional representation of *Acropora* was also down relative to baseline (Figure A2).

The Macroalgae indicator score remains ‘poor’ in 2025 (0.36, Table A7), following a significant decline for this indicator at both depths from 2021 to 2024 (Figure 18, Table 13). Across the sub-region, the cover of brown macroalgae is very low, particularly *Lobophora* and the family Sargassaceae, which are typical of many inshore reefs (Table A11). Low Macroalgae scores in this sub-region reflect dense mats of red macroalgae (Table A7, Table A11). Such mats have been a persistent feature at Franklands West and are more ephemeral elsewhere (Figure A2). Scores of zero for Macroalgae in 2025 at High East, Franklands West, and Franklands East reflect unusually high levels of red macroalgae relative to most years (Table A7, Figure A2).

The Juvenile coral indicator score has remained ‘poor’ (0.34, Table A7) having fluctuated between ‘moderate’ and ‘poor’ since 2016 (Figure 18). There is considerable variability in this indicator’s scores among both reefs and depths, ranging between ‘very good’ and ‘poor’ (Table A7). Despite this, there has been an overall decline at the 2 m depth since 2021 (Table 13).

In 2025, the concentrations of dissolved N and P ($NO_x$ and $PO_4$) particulate P, turbidity, and Secchi depth exceeded guideline values (Figure A12, Gruber *et al*. 2026). The trend in condition since the redesign of the sampling program for water quality in 2015 has been deteriorating for $PO_4$, particulate N, and P, and Secchi depth (Gruber *et al.* 2026). The short-term water quality index declined in 2025 but remained ‘moderate’, having sat close to the boundary of ‘good’ since 2019 (Figure A12b). Over the period 2021–2025, satellite derived estimates of wet-season concentrations of Chl *a* and TSS were below wet-season guideline values at all coral monitoring locations (Figure 19a, b, Table A8).

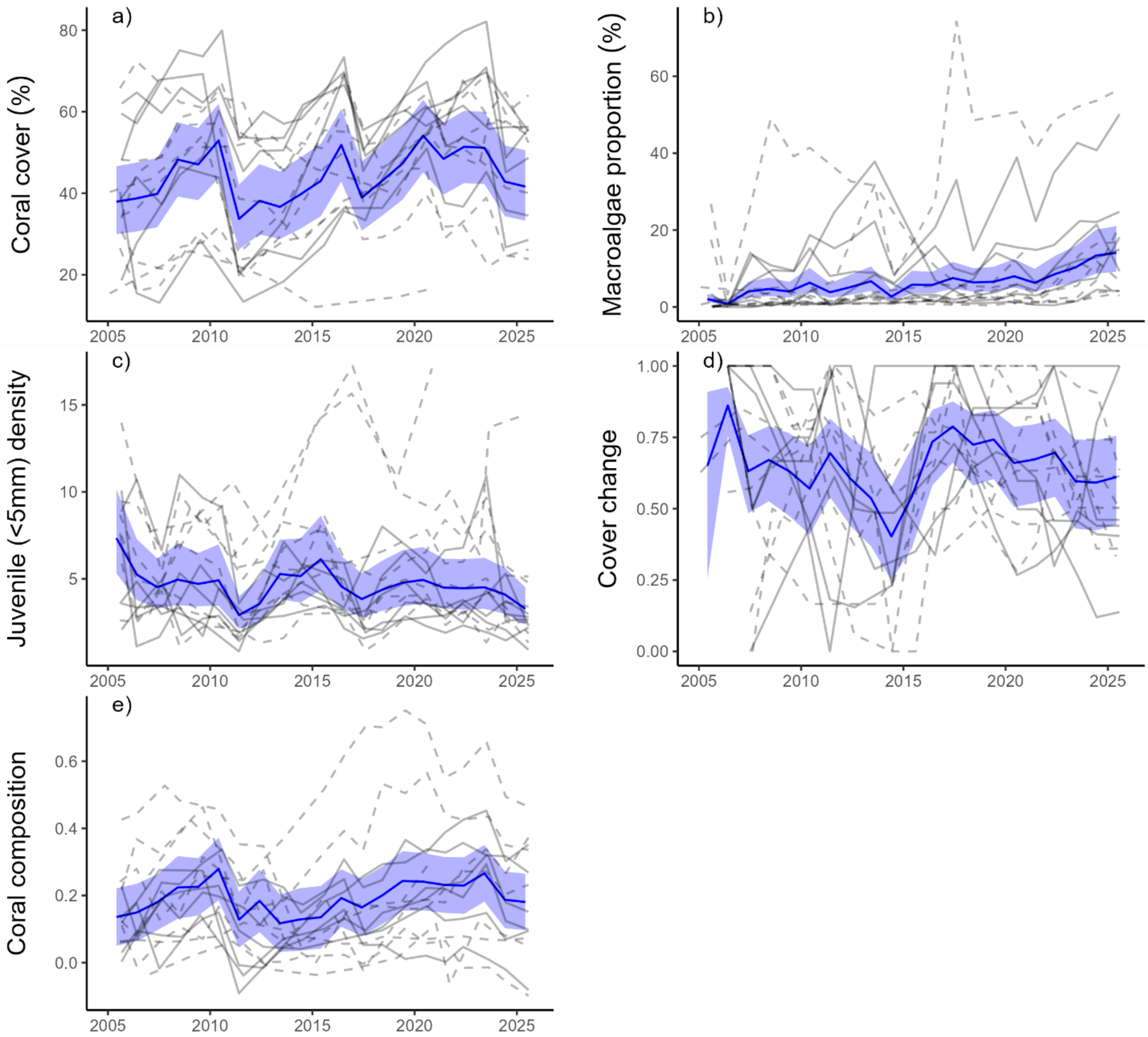



Figure 20. Indicator trends in the Johnstone Russell–Mulgrave sub-region. Temporal trends in observed values of a), live coral cover, b), macroalgal proportion, c), juvenile coral density, e), coral composition that inform indicators and d), derived indicator score for change in cover. Data are sub-regional averages (blue lines) bound by 95% confidence intervals of those trends (shading). Grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

#### *4.3.1.3 Herbert–Tully sub-region*

The Coral Index was categorised as 'moderate' in 2025, despite steadily declining since 2020 (Figure 21, Table 14). All indicators except Macroalgae continued to decline between 2024 and 2025 (Figure 21, Table 14, Table A7). The Macroalgae indicator score at 2 m depth improved in 2025 (Table 14); however, this may be temporary as it coincided with the exposure of most shallow sites to low salinity floodwaters.

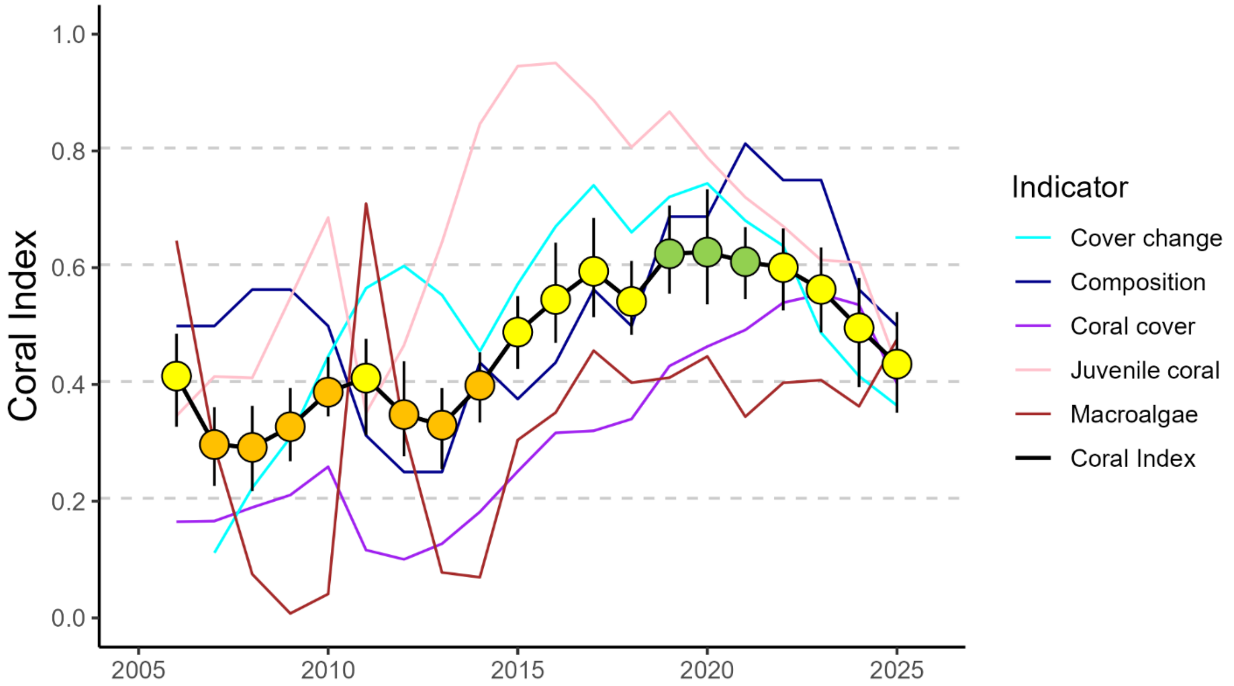



Figure 21. Coral Index and indicator trends in the Herbert–Tully sub-region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = 'poor', yellow = 'moderate' and green = 'good'. Error in Coral Index scores was derived from bootstrapped distributions of indicator scores at individual reefs

Table 14. Herbert–Tully sub-region Coral Index and indicator score changes. Data represent the changes in scores between sub-regional maxima and minima in the index time-series (Figure 21). For the Coral Index, and each indicator, the observed change in the sub-regional score and the probability that the change was greater or less than zero (no change) are presented. Shading is used as a visual aid to highlight the magnitude of the probability that the score improved (blue shades) or declined (red shades). Probabilities are derived from the posterior distribution of observed score changes at each reef and depth.

| Period | Depth | Coral Index | | Coral cover | | Macroalgae | | Juvenile coral | | Cover change | | Composition | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Score | P | Score | P | Score | P | Score | P | Score | P | Score | P |
| 2008 to 2011 | 2 | 0.10 | 0.76 | -0.08 | 0.75 | 0.67 | 0.92 | -0.05 | 0.64 | 0.34 | 0.95 | -0.38 | 0.93 |
| | 5 | 0.14 | 0.82 | -0.07 | 0.66 | 0.60 | 0.89 | -0.07 | 0.56 | 0.35 | 0.78 | -0.13 | 0.70 |
| 2011 to 2014 | 2 | 0.02 | 0.66 | 0.06 | 0.89 | -0.67 | 0.92 | 0.52 | 0.93 | -0.04 | 0.58 | 0.25 | 0.81 |
| | 5 | -0.05 | 0.64 | 0.07 | 0.90 | -0.61 | 0.90 | 0.46 | 0.97 | -0.17 | 0.82 | 0 | NA |
| 2014 to 2020 | 2 | 0.24 | 0.93 | 0.41 | 0.97 | 0.33 | 0.73 | -0.29 | 1.00 | 0.26 | 1.0 | 0.5 | 1.00 |
| | 5 | 0.27 | 0.97 | 0.28 | 0.87 | 0.41 | 0.77 | -0.03 | 0.76 | 0.33 | 0.99 | 0.33 | 0.87 |
| 2020 to 2025 | 2 | -0.18 | 0.99 | -0.15 | 0.93 | 0.18 | 0.67 | -0.36 | 1.00 | -0.33 | 0.96 | -0.25 | 0.81 |
| | 5 | -0.20 | 0.91 | 0.02 | 0.64 | -0.12 | 0.77 | -0.35 | 1.00 | -0.43 | 0.88 | -0.13 | 0.70 |
| 2024 to 2025 | 2 | -0.12 | 0.88 | -0.24 | 0.90 | 0.22 | 0.92 | -0.20 | 0.91 | -0.11 | 0.91 | -0.25 | 0.81 |
| | 5 | -0.01 | 0.64 | -0.03 | 0.66 | 0.01 | 0.51 | -0.15 | 0.97 | 0.01 | 0.57 | 0.13 | 0.71 |

Losses in hard coral cover since 2020 were attributed to high water temperatures that caused coral bleaching in 2020 and 2024, elevated levels of coral disease, and most significantly, exposure to low salinity floodwaters in 2025 (Figure 9, Figure 22c, e, Figure A8).

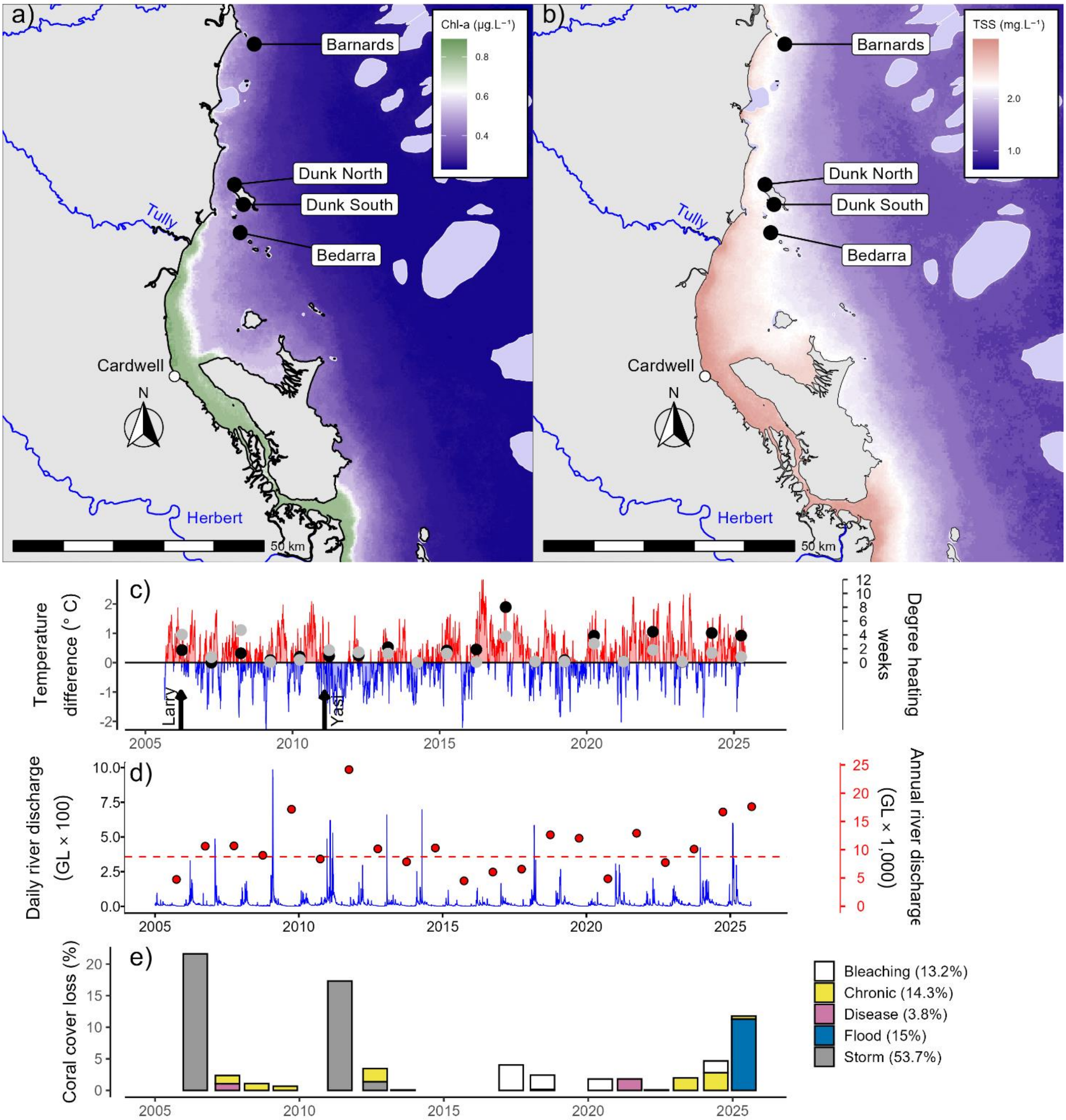



Figure 22. Environmental pressures in the Herbert–Tully sub-region. Maps show location of monitoring sites along with a), median wet season Chl *a* and b), median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2021–2025, white breaks in the colour gradients are set at wet-season guideline values for open coastal waters. c), Seasonally adjusted temperature deviation, timing of cyclones and storms indicated by black arrows, accumulated DHW over the summer period (1 December – 31 March) as reported by NOAA (black symbols) and derived from *in situ* loggers (grey symbols). d), Combined daily (blue) and annual water year – October to September (red) discharge for the Herbert, Murray and Tully basins, red dashed line represents long-term median discharge (1986–2016). e), break-down of hard coral cover loss by disturbance type; length of bars represents the mean loss of cover across all reefs.

The Coral cover indicator score was ‘poor’ in 2025 (0.40, Table A7). This indicator had shown steady improvement from ‘very poor’ in 2012 to ‘good’ in 2020, as coral cover recovered from the impact of Cyclone Yasi (Figure 21, Table 14, Figure 23a). In 2022 the score declined to ‘moderate’ and continued to fall, reaching ‘poor’ again in 2025 (Figure 21, Table A7). Declines were predominantly driven by losses of *Acropora* spp., which have been chronically affected by relatively high levels of disease since 2019 (Figure A3, Figure A8). These chronic impacts preceded a major loss caused by exposure to low salinity floodwaters at Bedarra, and the 2 m depths at Dunk North and Dunk South (Table A 6). Salinity records from Site 2 at Dunk Island North dropped as low as 13 PSU, including an accumulated exposure of 2 days at less than 22 PSU during February (Section 3.4, Figure 11), resulting in slight declines in coral cover from 32.7% in 2024 to 30.9% in 2025. By comparison, coral cover at Site 1 declined from 65.2% to 2.5%, suggesting higher exposure to low salinity waters.

The Cover change score declined to ‘poor’ (0.40) for the first time since 2009 following a significant decline since 2020 (Figure 21, Table 14). A ‘good’ score at Dunk North 5 m was a notable exception in this subregion, as scores for this indicator were ‘very poor’ at Bedarra 2 m, and ‘poor’ everywhere else (Table A7). Between 2020 and 2025, levels of coral disease were above median levels (Figure A8). In 2025, levels of disease at the Barnards were high enough to be classified as an acute disturbance, resulting in reduced growth or mortality of infected colonies. This disease, is likely the result of sub-lethal stress due to exposure to flood waters, and likely influenced the Cover change score, compounding reductions in the Coral cover score caused by the acute exposure to low salinity floodwaters at other reefs (Figure 22e).

The Composition score for this region remains ‘moderate’ (Table A7) having declined significantly since 2020, particularly at the 2 m depth (Table 14, Figure 21e).

The Macroalgae indicator score improved to ‘moderate’ in 2025 (Figure 21, Table A7).The scores for this indicator were highly variable, ranging from zero and 0.04 (‘very poor’), at the 5 m depth at Dunk South and 2 m depth at Bedarra respectively, through to 0.84 (‘very good’) at 5 m depth at Barnards (Table A7). At reefs classified as ‘very poor’, the macroalgal community is dominated by the brown algae *Lobophora* (Dunk South), and others from the family Sargassaceae (Bedarra) (Table A11). The improvement in this indicator between 2024 and 2025 was primarily attributable to flood impacts that reduced macroalgal cover and increased the cover of other algae on coral killed by the floods. Consequently, these changes have impacted both the numerator (macroalgae cover) and denominator (total algae cover) of the ratio used to score this indicator.

The Juvenile coral indicator has declined to ‘moderate’ (0.43, Table A7), following significant declines since 2014 (Table 14). The scores vary among depths ranging from ‘very poor’ or ‘poor’ at 2 m depths to ‘moderate’ or ‘good’ at 5m depths. Increases of juvenile corals in 2024 at Dunk South, Dunk North and Bedarra were short lived with subsequent declines recorded in 2025 at all these reefs (Figure A3). Bolstering scores for Juvenile coral between 2014 and 2021 were strong cohorts of *Turbinaria* (Family: Dendrophylliidae), which recruited in the years following cyclone Yasi. The subsequent decline in this indicator reflects that these corals have either died or grown beyond the juvenile size classes (Table 14, Figure 23c, Figure A3).

In 2025, most water quality parameters exceeded the guideline values, including $NO_x$ (Figure A13). However, the concentration of $NO_x$ has shown improvement since the redesign of the sampling program for water quality in 2015 (Figure A13, Gruber *et al.* 2026). Despite these exceedances, and a deterioration in particulate N since 2015, they were not sufficiently large to affect the short-term water quality index which remained ‘moderate’ (Figure A13). Over the period 2021–2025, satellite derived estimates of wet-season concentrations of TSS were marginally below wet-season guideline values at all coral monitoring locations (Figure 22a, b, Table A8).

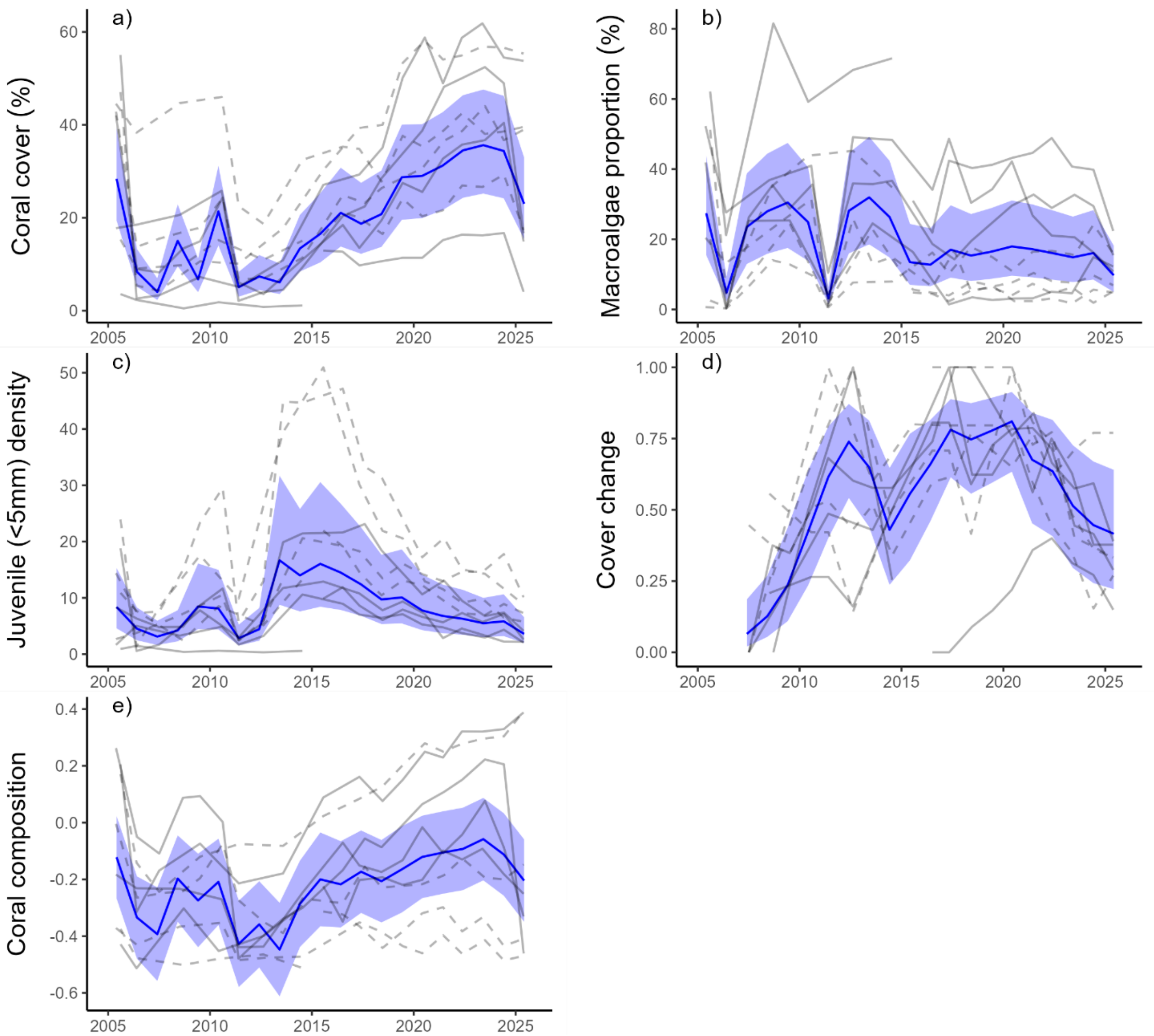



Figure 23. Indicator trends in the Herbert–Tully sub-region. Temporal trends in observed values of a), live coral cover, b), macroalgal proportion, c), juvenile coral density, e), coral composition that inform indicators and d), derived indicator score for change in cover. Data are sub-regional averages (blue lines) bound by 95% confidence intervals of those trends (shading). Grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

### *4.3.2 Burdekin region*

The Coral Index remained within the ‘moderate’ range in 2025 but has continued to decline from a high point observed in 2019 (Figure 24). All indicators except Coral cover and Macroalgae remain below the score for 2019 (Figure 24). The most consistent declines from 2019 occurred for Cover change, and, at 2 m depth, Composition (Figure 24, Table 15). Impacts of floods in early 2025 resulted in declines in the Coral Cover indictor scores and, at 2 m depths, declines in the Cover change and Composition scores (Figure 24, Table 15). In contrast, scores for Macroalgae improved from ‘poor’ to ‘moderate’ (Figure 24, Table 15).

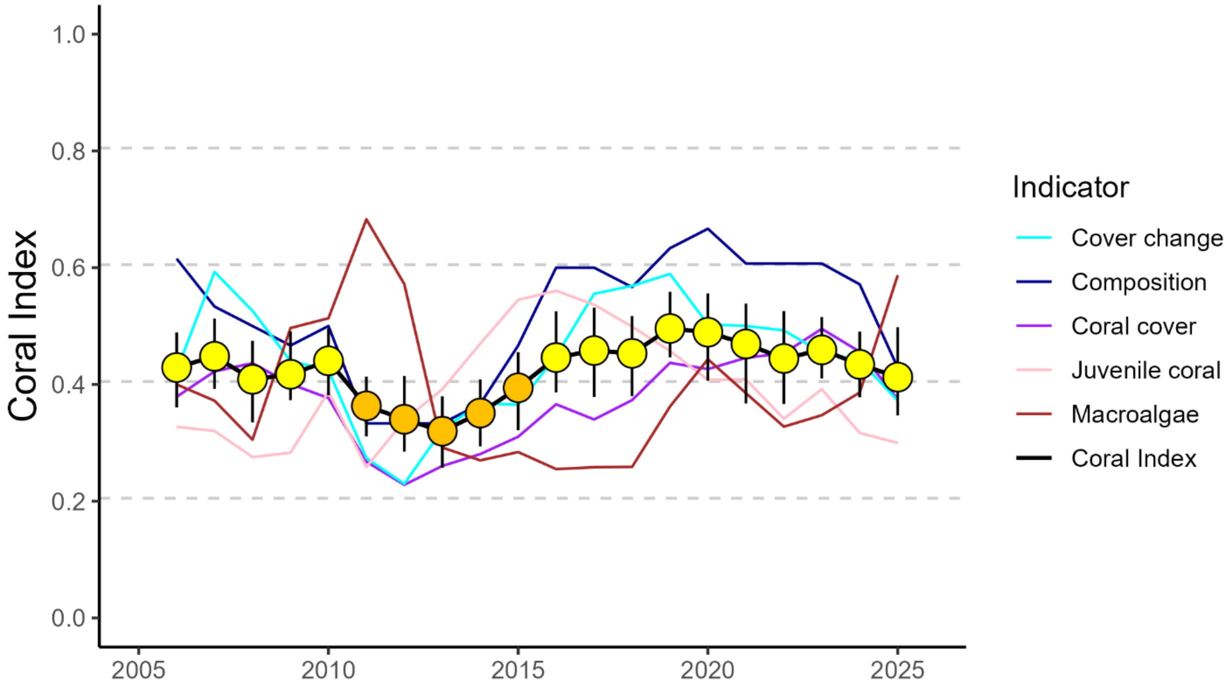



Figure 24. Coral Index and indicator trends in the Burdekin region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = ‘poor’, yellow = ’moderate’. Error in Coral Index scores was derived from bootstrapped distributions of indicator scores at individual reefs.

Table 15. Coral Index and indicator score changes in the Burdekin region. Data represent the changes in scores between regional maxima and minima in the index time-series (Figure 24). For the Coral Index, and each indicator, the observed change in the regional score and the probability that the change was greater or less than zero (no change) are presented. Shading is used as a visual aid to highlight the magnitude of the probability that the score improved (blue shades) or declined (red shades). Probabilities are derived from the posterior distribution of observed score changes at each reef and depth.

| Period | Depth | Coral Index | | Coral cover | | Macroalgae | | Juvenile coral | | Cover change | | Composition | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Score | P | Score | P | Score | P | Score | P | Score | P | Score | P |
| 2010 to 2013 | 2 | -0.08 | 0.70 | -0.09 | 0.64 | -0.17 | 0.71 | -0.04 | 0.61 | -0.05 | 0.54 | -0.07 | 0.57 |
| | 5 | -0.15 | 0.86 | -0.14 | 0.82 | -0.26 | 0.82 | 0.04 | 0.61 | -0.15 | 0.80 | -0.25 | 0.71 |
| 2013 to 2019 | 2 | 0.12 | 0.78 | 0.19 | 0.87 | 0.06 | 0.67 | -0.09 | 0.62 | 0.11 | 0.58 | 0.33 | 0.72 |
| | 5 | 0.24 | 0.93 | 0.22 | 0.09 | 0.11 | 0.72 | 0.17 | 0.81 | 0.40 | 0.95 | 0.31 | 0.73 |
| 2019 to 2025 | 2 | -0.10 | 0.82 | -0.11 | 0.74 | 0.15 | 0.62 | -0.08 | 0.69 | -0.20 | 0.73 | -0.25 | 0.73 |
| | 5 | -0.08 | 0.74 | -0.02 | 0.58 | 0.24 | 0.71 | -0.21 | 0.75 | -0.23 | 0.72 | -0.19 | 0.70 |
| 2024 to 2025 | 2 | -0.02 | 0.64 | -0.11 | 0.80 | 0.32 | 0.79 | -0.03 | 0.57 | -0.06 | 0.73 | -0.25 | 0.73 |
| | 5 | -0.02 | 0.61 | -0.06 | 0.91 | 0.11 | 0.82 | -0.01 | 0.52 | -0.08 | 0.68 | -0.06 | 0.58 |

Reefs in the Burdekin region were impacted by 2 acute disturbances over the 2024–25 summer, a localised storm that caused wave damage at some reefs, and flooding caused by a stationary tropical low weather system in February 2025 (Figure 25d, e, Table A 6).

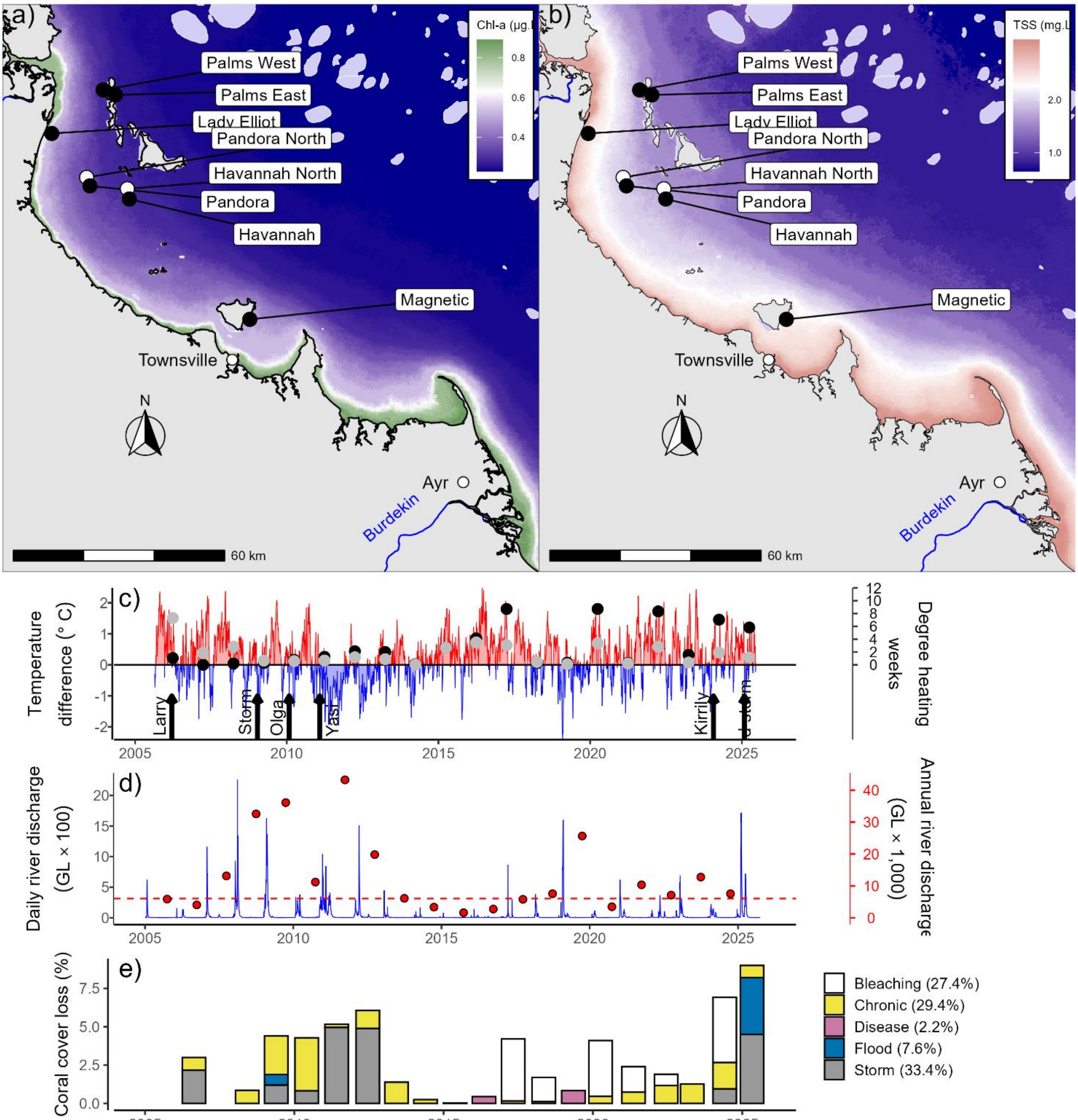



Figure 25. Environmental pressures in the Burdekin region. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a), median wet season Chl *a* and b), median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2021–2025, white breaks in the colour gradients are set at wet-season guideline values for open coastal waters. c), Seasonally adjusted temperature deviation, timing of cyclones and storms indicated by black arrows, accumulated DHW over the summer period (1 December – 31 March) as reported by NOAA (black symbols) and derived from *in situ* loggers (grey symbols). d), Combined daily (blue) and annual water year – October to September (red) discharge for the Black, Burdekin, Don and Haughton basins, red dashed line represents long-term median discharge (1986–2016). e), break-down of hard coral cover loss by disturbance type; length of bars represents the mean loss of cover across all reefs.

Annual freshwater discharge from the region's catchments ranged between 3.1 (Don River) and 8.6 (Ross River) times long-term medians (Table A5). Prior to the 2 acute events, reefs were exposed to marine heat wave conditions that caused coral bleaching in 2024, and to cyclone Kirrily that caused minor storm damage over the summer of 2023–24 (Figure 8, Figure 9, Figure 25e, Table A 6). The 2024 bleaching event adds to the accumulation of bleaching impacts resulting from marine heat waves in 2017 and 2020, and to a lesser degree 2022 (Figure 9, Figure 25e). Prior to the 2017 bleaching, reefs in this region were severely impacted by cyclone Yasi in 2011 (Figure 24, Figure 25e, Table A 6). Note that in Figure 25e the impacts from cyclone Yasi, and the 2017 and 2020 bleaching events span 2 years due to a combination of some reefs not being resurveyed in the winter immediately following the disturbance and, for bleaching, the full impacts not being realised until the following year.

The Coral cover indicator score declined to 'poor' (0.38, Table A7) between 2024 and 2025 (Figure 24, Table 15) reflecting reduced combined cover of hard and soft corals at all reefs except for 2 m depth at Magnetic (Figure 25a, Figure A4). The most impacted reef was Lady Elliot, where floodwaters killed 90% of the hard corals at 2 m depth (hard coral cover declined from 28.2% in 2024 to 2.9% in 2025) and 22% of hard coral at 5 m depth (Figure A4).

The Cover change indicator score has been declining since a peak in 2019 and is now categorised as 'poor' (0.37, Table A7) (Figure 24). This indicator was highly variable among reefs ranging from 0.17 to 0.56; however, the greatest disparity occurred between the depths at Magnetic that were 'very poor' (0.17) at 2 m depth and 'moderate' (0.55) at 5 m depth (Table A7).

The Composition indicator score remained 'moderate' (0.43, Table A7) but has decreased at 2 m depth across the region since 2019 (Figure 24, Table 15). A further decline was recorded at 2 m depth between 2024 and 2025 (Table 15). Declines in this indicator largely reflect reduced representation of *Acropora* within the hard coral community (Figure A4).

The Macroalgae indicator has improved at the 5 m depth since 2019, and at both depths between 2024 and 2025, moving to a score of 'moderate' in 2025 (0.59, Table A7, Table 15). The scores for this indicator vary substantially among reefs and between depths. Seven sites scored above 0.9 ('very good'), one site scored 0.61 ('good'), 2 sites were rated 'poor' (0.25, 0.29), and the remaining 4 sites were categorised as 'very poor' (0 - 0.1, Table A7). Sites of high macroalgal cover had communities dominated by species of large brown algae from the genus *Lobophora* and/or family Sargassaceae (Table A11). Marked improvements in Macroalgae indicator scores occurred at Pandora and 2m depth at Lady Elliot and Havannah where reduced macroalgal cover resulted from exposure to floodwaters and storm damage (Figure A4).

The Juvenile coral indicator remained 'poor' (0.29. However, the scores were highly variable among reefs, ranging from 0.1 'very poor' to 0.74 'good' (Table A7)). Juvenile density increased substantially between 2024 and 2025 at both depths at Magnetic and at 5m at Pandora North, while declines were evident at most other reefs (Figure A4). A key driver of the regional decline in juvenile densities at 5 m depth since 2020 has been the reduction in *Turbinaria* (Family: Dendrophylliidae). The strong cohorts that settled on some reefs following cyclone Yasi have died or grown beyond the juvenile size classes (Figure A4).

In 2025, turbidity, Secchi, and the concentrations of $NO_x$ and $PO_4$ quality parameters failed to meet guideline values (Figure A14d, e, f, Gruber *et al.* 2026). However, the concentration of $NO_x$ continues to show improvement since 2015, and despite a deterioration of $PO_4$ and turbidity over the same time period, both the short-term and long-term water quality index were classified as 'good' (Figure A14a, b, Gruber *et al*. 2026). In waters adjacent to the coral monitoring sites, mean wet-season TSS concentrations over the period 2021-2025 exceeded guidelines at Magnetic and Lady Elliot but not at any other location (Figure 25a, b, Table A 8).

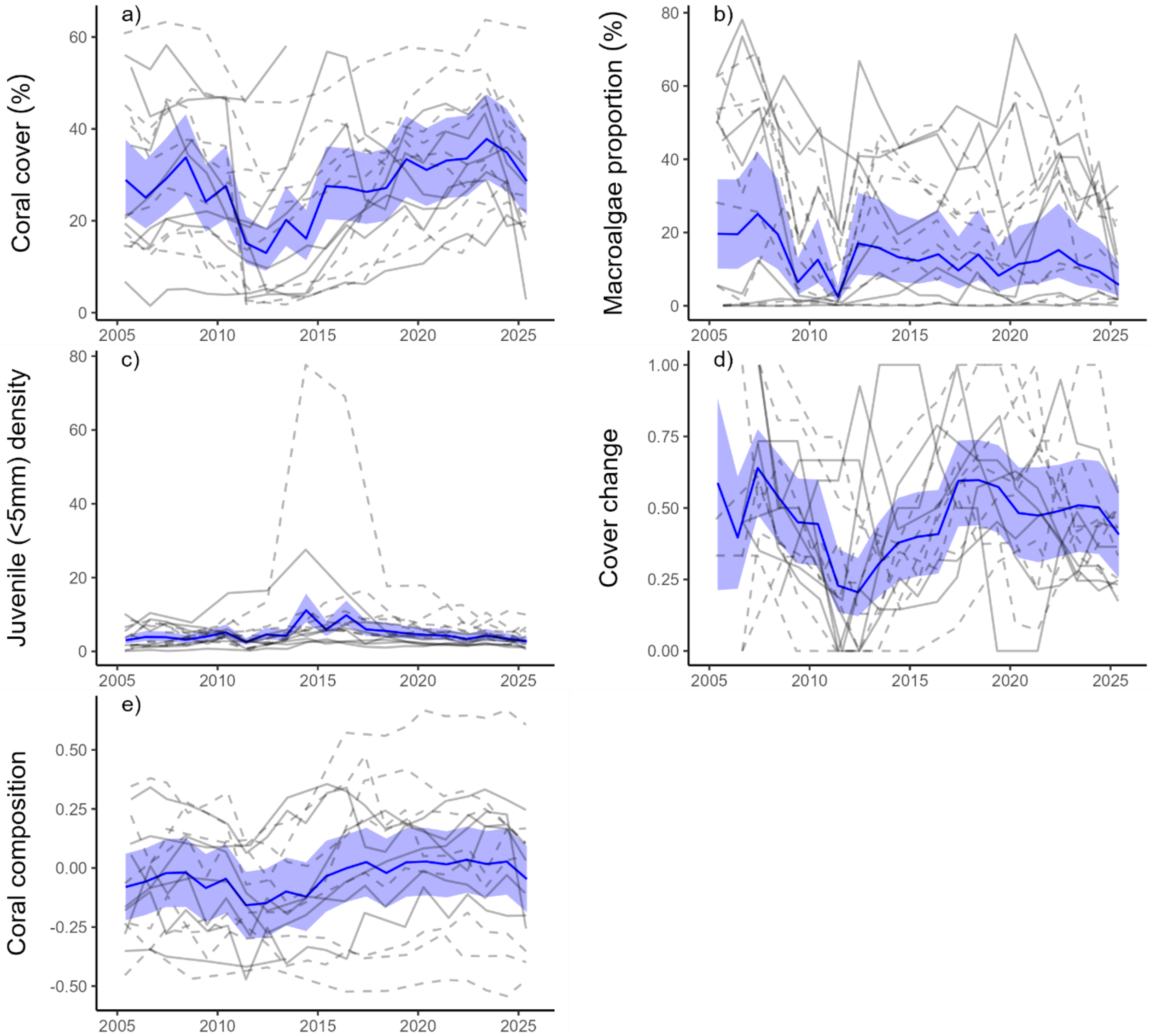



Figure 26. Indicator trends in the Burdekin region. Temporal trends in observed values of a), live coral cover, b), macroalgal proportion, c), juvenile coral density, e), coral composition that inform indicators and d), derived indicator score for change in cover. Data are regional averages (blue lines) bound by 95% confidence intervals of those trends (shading). Grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

### *4.3.3 Mackay–Whitsunday region*

In 2025 the Coral Index score improved to ‘moderate’ (Figure 27) as the recovery of coral communities since the impact of cyclone Debbie continued. All indicators have improved relative to scores in 2024. The Composition and Juvenile coral scores were both ‘moderate’, whereas scores for the other indicators remained within the ‘poor’ category (Figure 27, Table A7).

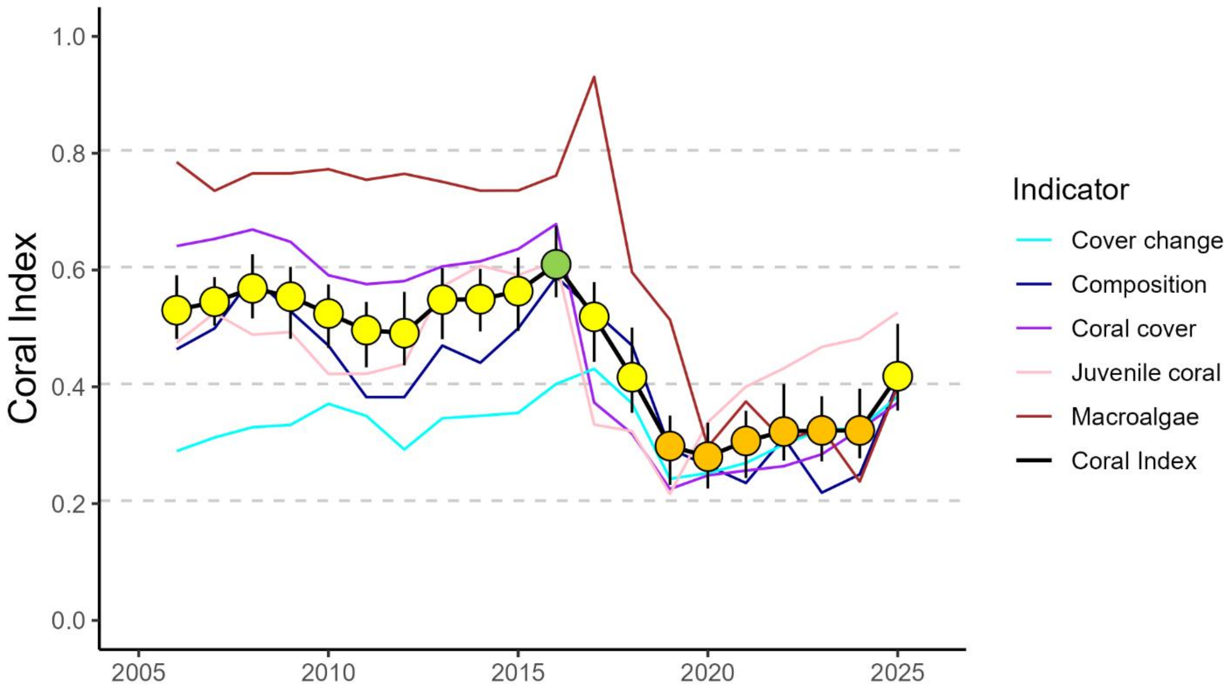



Figure 27. Coral Index and indicator trends in the Mackay–Whitsunday region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = ‘poor’, yellow = ’moderate’, green = ’good’. Error in Coral Index scores was derived from bootstrapped distributions of indicator scores at individual reefs.

Table 16. Coral Index and indicator score changes in the Mackay–Whitsunday region. Data represent the changes in scores between regional maxima and minima in the Coral Index time-series (Figure 27). For the Coral Index, and each indicator, the observed change in the regional score and the probability that the change was greater or less than zero (no change) are presented. Shading is used as a visual aid to highlight the magnitude of the probability that the score improved (blue shades) or declined (red shades). Probabilities are derived from the posterior distribution of observed score changes at each reef and depth.

| Period | Depth | Coral Index | | Coral cover | | Macroalgae | | Juvenile coral | | Cover change | | Composition | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Score | P | Score | P | Score | P | Score | P | Score | P | Score | P |
| 2012 to 2016 | 2 | 0.16 | 0.99 | 0.15 | 0.95 | 0.00 | NA | 0.18 | 0.86 | 0.20 | 0.76 | 0.29 | 0.86 |
| | 5 | 0.09 | 0.77 | 0.06 | 0.72 | -0.01 | 0.63 | 0.17 | 0.75 | 0.05 | 0.57 | 0.15 | 0.68 |
| 2016 to 2020 | 2 | -0.41 | 0.96 | -0.53 | 0.97 | -0.52 | 0.88 | -0.27 | 0.92 | -0.34 | 0.92 | -0.43 | 0.83 |
| | 5 | -0.27 | 0.92 | -0.36 | 0.95 | -0.42 | 0.83 | -0.28 | 0.86 | -0.06 | 0.57 | -0.25 | 0.76 |
| 2020 to 2025 | 2 | 0.09 | 0.7 | 0.11 | 0.86 | 0.04 | 0.54 | 0.14 | 0.90 | 0.06 | 0.57 | 0.07 | 0.59 |
| | 5 | 0.18 | 0.83 | 0.13 | 0.96 | 0.23 | 0.81 | 0.22 | 0.80 | 0.12 | 0.64 | 0.17 | 0.67 |
| 2024 to 2025 | 2 | 0.07 | 0.75 | 0.03 | 0.79 | 0.09 | 0.66 | 0.03 | 0.75 | 0.06 | 0.66 | 0.14 | 0.64 |
| | 5 | 0.11 | 0.99 | 0.06 | 0.91 | 0.22 | 0.83 | 0.06 | 0.69 | 0.05 | 0.66 | 0.17 | 0.76 |

No significant acute pressures influenced the reefs in this region over the 2024/25 summer. While water temperatures in early 2025 briefly exceed long term medians, NOAA heat stress estimates were 3-5 DHW, lower than those for 2024 (Figure 9, Figure 28c). Data from *in situ* temperature loggers confirms a pattern of less heat stress compared to previous marine heatwave events in 2020 and 2024 (Figure 28c). During surveys in July only a few individual colonies were bleached.

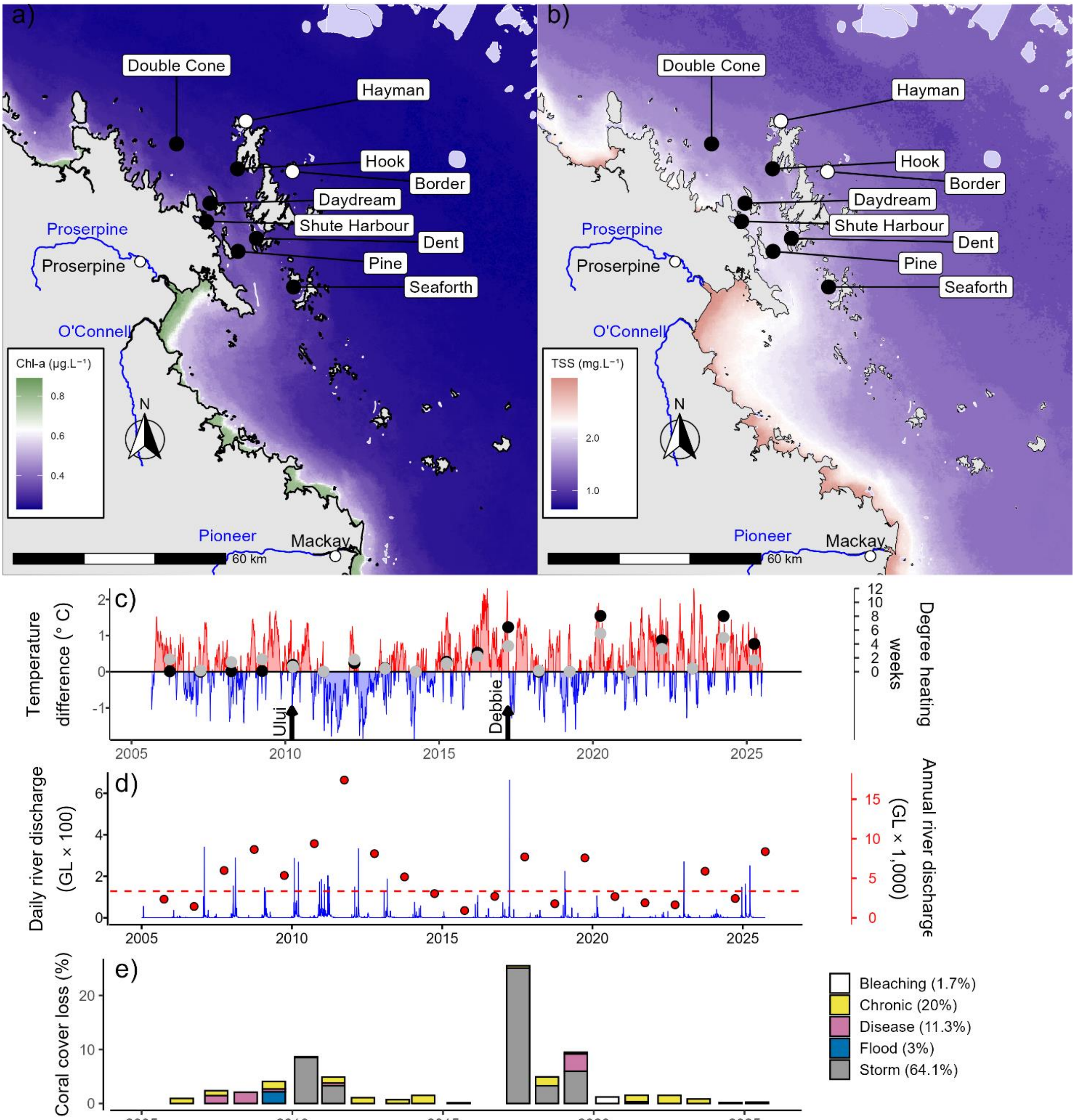



Figure 28. Environmental pressures in the Mackay-Whitsunday region. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a), median wet season Chl *a* and b), median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2021–2025, white breaks in the colour gradients are set at wet-season guideline values for open coastal waters. c), Seasonally adjusted temperature deviation, timing of cyclones and storms indicated by black arrows, accumulated DHW over the summer period (1 December – 31 March) as reported by NOAA (black symbols) and derived from *in situ* loggers (grey symbols). d), Combined daily (blue) and annual water year – October to September (red) discharge for the Carmila and Sandy creeks, Gregory, O'Connell and Pioneer rivers, red dashed line represents long-term median discharge (1986–2016). e), breakdown of hard coral cover loss by disturbance type; length of bars represents the mean loss of cover across all reefs.

The annual freshwater discharge from the region's catchments ranged between 2 and 2.9 times the long-term median (Table A5). Disease levels within the hard coral community remained below the long-term median level and unchanged from 2023 (Figure A8). There were no crown-of-thorns starfish observed in the period 2024–2025 (Figure A9).

The Coral cover indicator has gradually increased since the impacts of cyclone Debbie in 2017. There were improved scores at most reefs and depths over the last year, which continues a longer-term trend of improvement since the lowest Coral index score in 2020 (Table 16). The highest Coral cover score ('very good') in the region continues to be recorded at the 2 m depth at Shute Harbour where hard corals, predominantly *Acropora*, cover 50% of the substrate and soft corals a further 14% (Figure A5). 'Moderate' scores for Coral cover were maintained at the 5 m depths at Hook and Shute Harbour, and at both depths at Dent. Elsewhere, Coral cover scores were 'poor' or 'very poor' (Table A7).

Cover change has been 'poor' for most years (Figure 27). The lowest Cover change scores were recorded in 2019 and 2020 and, although regional scores have improved each year, the improvement was variable among reefs (Figure 27, Table 16, Figure 29d,). In 2025 Cover change scores ranged from 'good' at Daydream, 'moderate' at Hayman, Hook, Double Cone, and Dent (5 m) to 'poor' or 'very poor' at the remaining reefs (Table A7).

The extensive loss of Acroporidae corals following cyclone Debbie in 2017 (Figure A5) was reflected by the reductions in the Composition score (Figure 29e, Table 16). In 2025 improvements in the Composition score at Hayman, Hook (5 m), and Seaforth (2 m), transitioned the regional score from 'poor' to 'moderate'. (Figure 27, Table A7, Table 16).

In 2020, the Macroalgae indicator score declined substantially relative to the levels observed prior to cyclone Debbie (Figure 27). Since 2020 the regional Macroalgae score has remained 'poor' (Figure 26, Table 16, Table A7), although the proportion of macroalgae in the algal community varies markedly among reefs (Figure 29b). In 2025 macroalgae cover has declined across all reefs in the region (Figure A5). Macroalgae scores remained 'very good' at the more offshore sites at Hayman, Border and Hook (2 m) (Table A7). Some reefs improved markedly in 2025, with the cover of macroalgae being relatively low (Figure A5). Shute Harbour (5 m) was classified as 'very good'; Shute Harbour (2 m), Daydream (5 m) and Hook (5 m) as 'good'; and Dent (5 m) and Pine (5 m) as 'moderate'. In contrast scores were zero at 7 sites, reflecting the persistence of dense macroalgae since the impacts of cyclone Debbie in 2017 (Table A7, Figure A5)

While remaining 'moderate', the regional Juvenile coral indicator scores continue to improve as juvenile densities rebound following cyclone Debbie (Figure 27, Table 16). In 2025, juvenile coral densities at Hayman and Daydream remained sufficiently high to maintain 'good' to 'very good' scores (Table A7, Figure A5). Increased numbers of juveniles at Border also lifted its score to 'very good' (Table A7, Figure A5), while 'moderate' scores were recorded at Shute Harbour, Seaforth, and at the 2 m depth at Hook (Table A7). Conversely, Juvenile score declined to 'poor' at Pine 5 m and stayed 'poor' at the remaining reefs.

In 2025, the concentrations of most water quality parameters exceeded guideline values with only particulate N meeting the guidelines (Gruber *et al.* 2026). However, turbidity marginally improved over the 2015 to 2025 period(Figure A15f, Gruber *et al*. 2026). The short-term water quality index declined in 2025 after 7 years of improvement but remained 'moderate' (Figure A15b). From 2021–2025, wet-season concentrations of Chl *a* and TSS, as estimated from satellite imagery, were below wet-season guideline values at all coral monitoring locations (Figure 28a, b, Table A8).

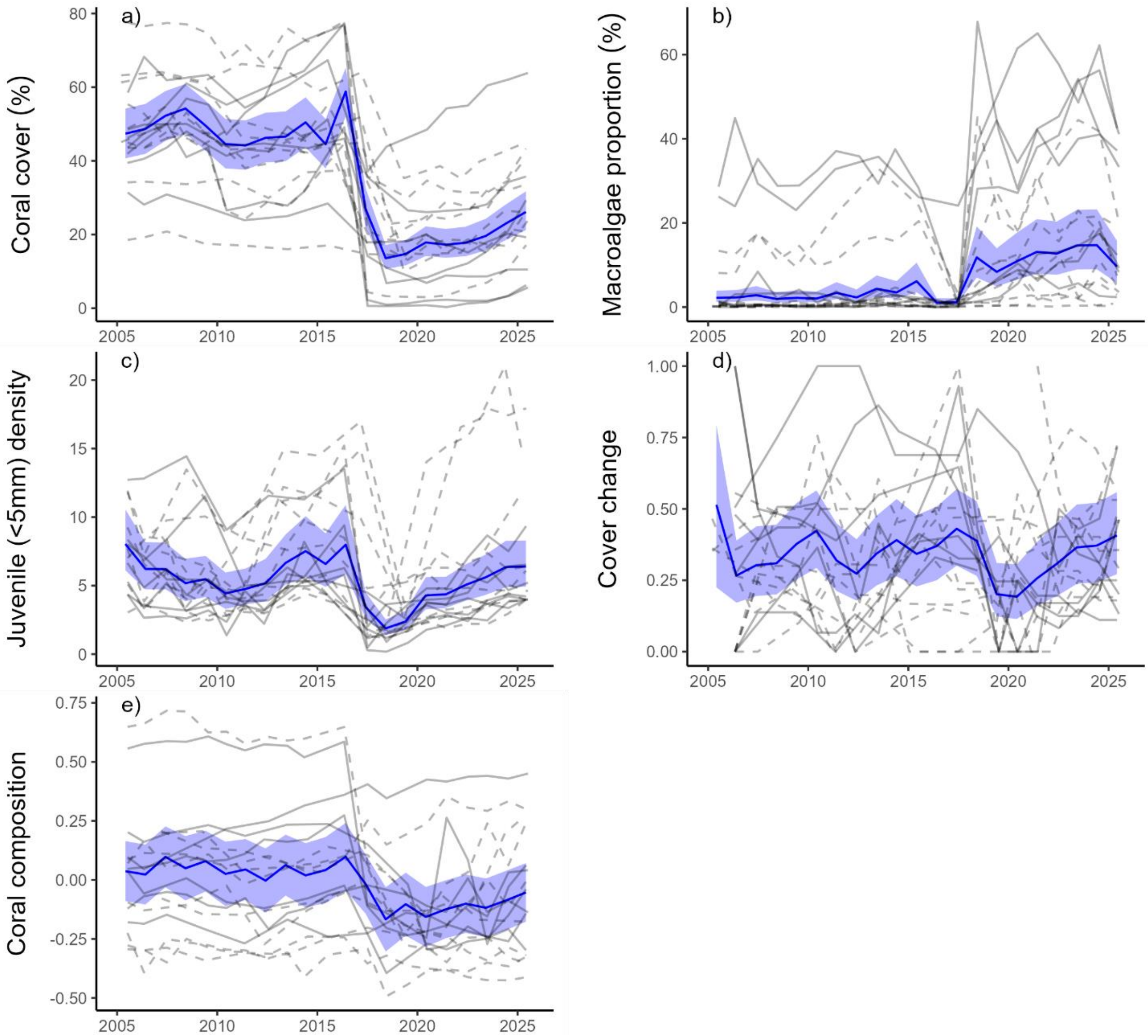



Figure 29. Indicator trends in the Mackay–Whitsunday region. Temporal trends in observed values of a), live coral cover, b), macroalgal proportion, c), juvenile coral density, e), coral composition that inform indicators and d), derived indicator score for change in cover. Data are regional averages (blue lines) bound by 95% confidence intervals of those trends (shading). Grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

### *4.3.4 Fitzroy region*

The Fitzroy region Coral Index score has continued to decline to ‘very poor’ in 2025, the first time this category has been reached since 2014 (Figure 30, Table 17). The ongoing decline was largely driven by reductions in the Coral cover indicator, which declined at both 2 m and 5 m depths at most reefs (Figure 30, Table 17). The Composition, Juvenile coral and Macroalgae indicators remained ‘very poor’ having declined since 2024 changes; however, scores were variable among reefs (Figure 30, Table 17, Table A7). The Cover change score remains ‘poor’ (Figure 30).

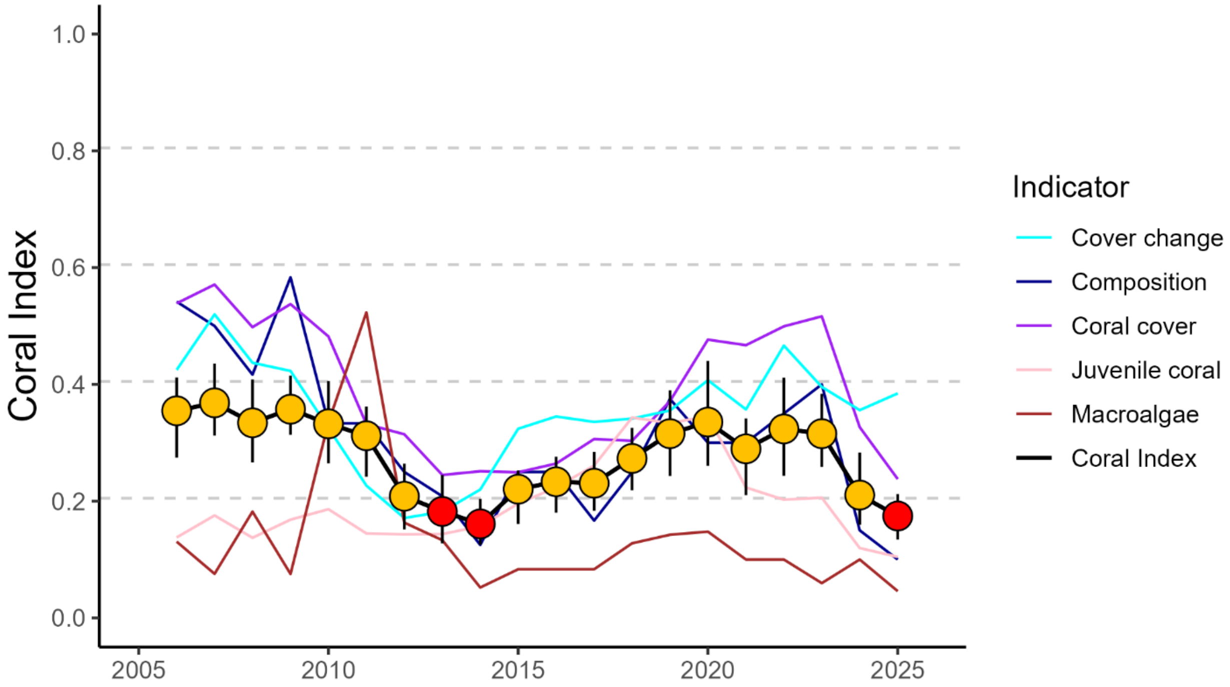



Figure 30. Coral Index and indicator trends in the Fitzroy region. Coral Index scores are coloured by Reef Water Quality Report Card categories: red =’ very poor’, orange = ‘poor’. Error in Coral Index scores was derived from bootstrapped distributions of indicator scores at individual reefs.

Table 17. Coral Index and indicator score changes in the Fitzroy region. Data represent the changes in scores between regional maxima and minima in the Coral Index time-series (Figure 30). For the Coral Index, and each indicator, the observed change in the regional score and the probability that the change was greater or less than zero (no change) are presented. Shading is used as a visual aid to highlight the magnitude of the probability that the score improved (blue shades) or declined (red shades). Probabilities are derived from the posterior distribution of observed score changes at each reef and depth.

| Period | Depth | Coral Index | | Coral cover | | Macroalgae | | Juvenile coral | | Cover change | | Composition | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Score | P | Score | P | Score | P | Score | P | Score | P | Score | P |
| 2007 to 2014 | 2 | -0.25 | 0.92 | -0.36 | 0.85 | -0.05 | 0.67 | -0.06 | 0.61 | -0.41 | 0.89 | -0.42 | 0.98 |
| | 5 | -0.15 | 0.92 | -0.28 | 0.93 | 0 | NA | 0.02 | 0.57 | -0.13 | 0.72 | -0.33 | 0.90 |
| 2014 to 2020 | 2 | 0.16 | 1.00 | 0.22 | 0.93 | 0.07 | 0.69 | 0.17 | 0.89 | 0.13 | 0.71 | 0.2 | 0.69 |
| | 5 | 0.21 | 0.98 | 0.22 | 0.90 | 0.10 | 0.71 | 0.22 | 0.81 | 0.23 | 0.90 | 0.3 | 0.71 |
| 2020 to 2023 | 2 | -0.01 | 0.53 | 0.08 | 0.69 | -0.08 | 0.68 | -0.12 | 0.75 | -0.01 | 0.51 | 0.1 | 0.69 |
| | 5 | -0.03 | 0.69 | 0.0 | 0.51 | -0.1 | 0.70 | -0.16 | 0.73 | -0.02 | 0.53 | 0.1 | 0.69 |
| 2023 to 2025 | 2 | -0.14 | 0.88 | -0.37 | 0.92 | -0.03 | 0.68 | -0.16 | 0.82 | 0.03 | 0.67 | -0.20 | 0.77 |
| | 5 | -0.14 | 0.99 | -0.19 | 0.91 | 0.00 | NA | -0.05 | 0.69 | -0.05 | 0.70 | -0.40 | 0.96 |
| 2024 to 2025 | 2 | -0.05 | 0.70 | -0.09 | 0.83 | -0.11 | 0.68 | -0.02 | 0.72 | 0.05 | 0.72 | -0.1 | 0.68 |
| | 5 | -0.02 | 0.59 | -0.09 | 0.87 | 0.00 | NA | -0.01 | 0.57 | 0.00 | 0.53 | 0 | 0.5 |

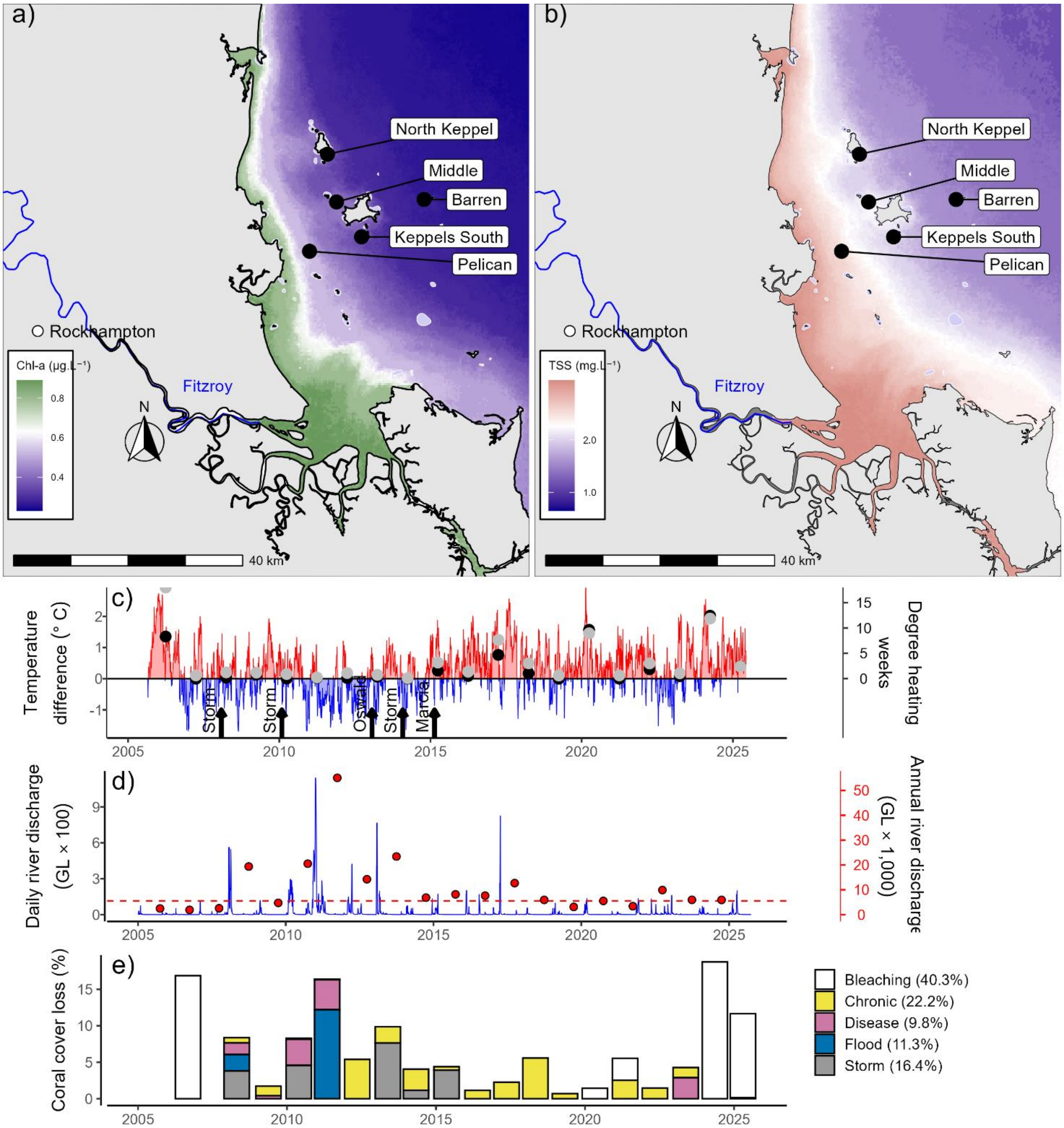



Figure 31. Environmental pressures in the Fitzroy region. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a), median wet season Chl *a* and b), median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2021–2025, white breaks in the colour gradients are set at wet-season guideline values for open coastal waters. c), Seasonally adjusted temperature deviation, timing of cyclones and storms indicated by black arrows, accumulated DHW over the summer period (1 December – 31 March) as reported by NOAA (black symbols) and derived from *in situ* loggers (grey symbols). d), Combined daily (blue) and annual water year – October to September (red) discharge for the Calliope and Fitzroy rivers and Waterpark Creek, red dashed line represents long-term median discharge (1986–2016). e), break-down of hard coral cover loss by disturbance type; length of bars represents the mean loss of cover across all reefs.

While summer sea surface temperatures again exceeded long-term medians, satellite derived estimates of coral heat stress in early 2025 ranged between 2 and 4 DHW, much lower than the extreme levels of > 10DHW recorded in 2024 (Figure 9, Figure 31c). Data from *in situ* temperature loggers at survey reefs confirm a pattern of lower heat stress, with an average DHW of 2.4 (Figure 31c). Bleaching was recorded for a few individual colonies at all reefs during this year's survey, but no widespread mass bleaching was observed. The annual freshwater discharge from the region's catchments approximated the long-term median and there was no flooding (Figure 28d, Table A5). Disease levels within the hard coral community remained below the long-term median level (Figure A8). There were no crown-of-thorns starfish and very low numbers of *Drupella* observed during 2025 surveys (Figure A9, Figure A10). While no acute pressures have impacted corals in the region since 2024, the 31% decline in hard coral cover (from 23% to 16%) was attributed to post-bleaching mortality following the 2024 marine heatwave (Figure 31e).

Across the region coral cover declined at all reefs except for Pelican (Figure 32a, Figure A6), where the Coral cover indicator scores remained 'poor' at 5 m depth and 'very poor' at 2 m (Table A7). Coral cover remains highest at Barren (5 m), the only reef where the Coral cover indicator was 'good' in 2025 (Table A7). Elsewhere, Coral cover scores in 2025 were either 'poor' (Barren 2 m, North Keppel 5 m and Pelican 5 m) or 'very poor' (Table A7). The overall loss of hard coral cover resulting from the 2024 mass bleaching event, incorporating both the initial mortality in 2024 and the subsequent mortality observed in 2025, was greatest at 2 m depths.

Reefs with relatively high *Acropora* cover in 2023 (i.e. all reefs other than Pelican), had lost between 31% at Barren 5 m and 86% at North Keppel 2 m of hard coral cover by 2025. These proportional losses translate to hard coral cover declining from 70.2% to 48.3% at Barren 5 m, and 48.5% to 6.7% at North Keppel 2 m (Figure A6).

Soft coral changes were minor in absolute terms due to their comparatively low cover relative to hard corals (Figure A6). At most reefs in the Fitzroy region the contribution of soft coral to the combined Coral cover metric is minor aside from Pelican (5 m), where 32% of the coral cover is comprised of soft corals, the majority (60%) of which are the family Sarcophytidae (Table A10).

The Cover change indicator score for the region remained 'poor' and unchanged in 2025. This indicator is based on the rate of change in hard coral cover over the last 4 years, but excludes changes caused by an acute pressure. With coral bleaching impacting hard coral cover estimates in both 2024 and 2025 at most reefs, scores for this indicator predominately reflect the rate of coral cover increase between 2021 and 2023. During this period, Barren was the only location at which hard coral cover was recovering in line with modelled expectations, the 'poor' (North Keppel and Keppels South) and 'very poor' (Middle) scores indicating slower rates of recovery (Table A7). The exception was Pelican (2 m) where the score of 0.71 (Table A7) includes changes occurring since 2024 and, although coral cover remains low, recent recovery is exceeding modelled expectations.

The regional Composition score has declined since 2023 (Figure 30, Table A7) due primarily to the disproportionate loss of *Acropora* cover following the 2024 bleaching event. In 2025, only North Keppel 2 m, where *Acropora* still comprised 99% of the hard coral community despite recent losses, and the more diverse community at Pelican (5 m), had Composition scores of 0.5, indicating community composition similar to those observed during the early years of the program (Table A7, Figure 32e).

The Macroalgae indicator has remained 'very poor' since 2012 (Figure 30). In 2025, the 'moderate' score at Barren (2 m) was the only score above zero (Table A7). The scores of zero at most reefs hide variations in the relative cover of macroalgae compared to other algae (Table A3). Across the region, the proportion of macroalgae increased by an average of 9.6% between 2024 and 2025. Macroalgal cover has increased everywhere except for Middle at 5 m, where cover is stable at 46%, and Pelican 5 m where cover declined slightly to 9% (Figure A6).

While the composition of algal communities varies among reefs, the presence of the brown macroalgae *Lobophora* is a common component everywhere. At North Keppel and Keppels South, *Lobophora* is by far the most common macroalgae (Table A 11). At Middle 2 m *Lobophora* is common and equally abundant as the family Sargassaceae. At both Middle 5 m and Pelican 2 m Sargassaceae becomes the dominant group. At Barren 5 m, *Lobophora* and red macroalgae are both key components of the macroalgal community, whereas at 2 m, the low cover of macroalgae is dominated by red macroalgae. Macroalgae cover is also relatively low at 5 m depth at Pelican (Figure A6), but here *Lobophora* is the most common group (Table A 11).

Juvenile numbers across this region have been low throughout the study. The regional Juvenile density indicator peaked in 2018, albeit with a ‘poor’ score. It has declined ever since and was categorised as ‘very poor’ in 2025, the lowest score to-date (Figure 30, Figure 32c ). Pelican (5 m) is the only reef in this region with a Juvenile density indicator score above ‘very poor’, having improved from ‘poor’ to ‘moderate’ in 2025, reflecting minor increases across a range of taxa including Dendrophylliidae (Table A7, Figure A6). The metric for the Juvenile indictor considers the ratio between the number of juvenile corals and the area of transects classified as algae. Since 2023, average juvenile numbers have declined by 30% while average algal cover has increased by 38%. Both changes have negatively impacted the Juvenile indicator.

*In situ* water quality monitoring was reinstated in 2021 after being discontinued in 2015, due to budget constraints. In 2015, the long-term water quality index was assessed as improving and scored ‘good’ (Figure A15a, Gruber *et al*. 2026). Conditions from 2021 to 2025 continued to be categorised as ‘good’ across the long-term water quality index (Figure A15a). In 2025, the short-term water quality index score was also ‘good’, with most water quality parameters being at or below guideline values (Figure A15). Parameters that exceeded guideline values in 2025 include dissolved inorganic forms of nitrogen ($NO_x$), turbidity, and Secchi depth (Gruber *et al*. 2026, Figure A15d, f). Between 2021 and 2025, wet-season Chl *a* concentrations were below guideline values at all coral monitoring sites, and the TSS guideline value was exceeded at only Pelican (Figure 32a,b, Table A8).

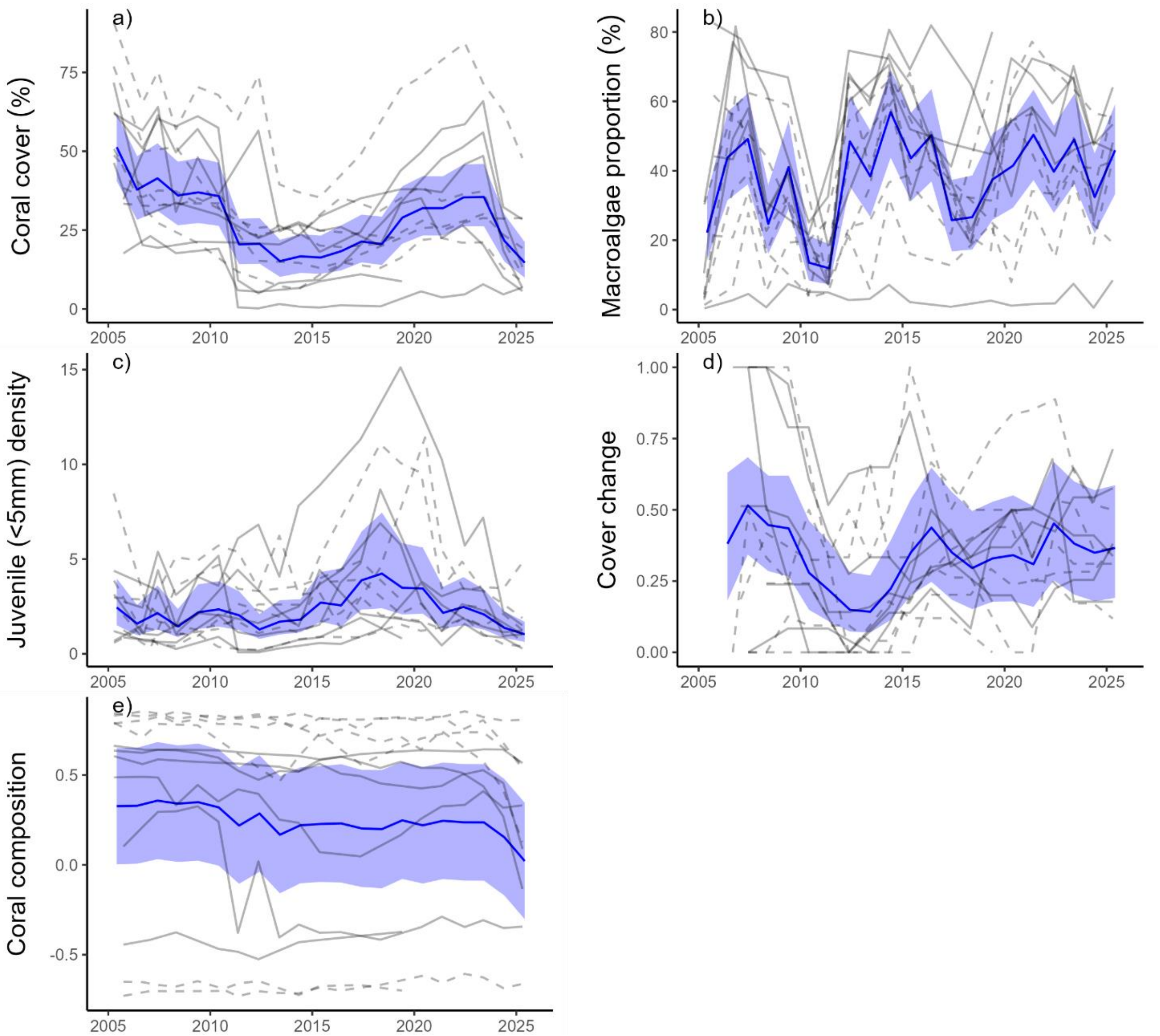



Figure 32. Indicator trends n the Fitzroy region. Temporal trends in observed values of a). live coral cover, b), macroalgal proportion, c), juvenile coral density, e), coral composition that inform indicators and d), derived indicator score for change in cover. Data are regional averages (blue lines) bound by 95% confidence intervals of those trends (shading). Grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

## 4.4 Response of coral communities to environmental conditions

### *4.4.1 Location along water quality gradients*

Relationships between Coral Index and Indicator scores, and values of macroalgae proportion and hard coral composition that underpin the Macroalgae and Composition scores, were assessed against 3 sets of water quality variables:

- Satellite derived estimates of the light attenuation coefficient kd490 (2020-2024) and wet season Chl *a* concentrations (2021-2025) available for all coral monitoring sites.
- Estimates of turbidity and Chl *a* measured at 10-minute intervals by loggers situated at the 5 m depths of a subset of coral monitoring sites.
- Estimates of particulate and dissolved forms of N and P adjacent to the above-mentioned logger sites.

The models used for these comparisons ae detailed in section 2.5.4.In 2025, Coral Index scores were negatively related to both wet season Chl *a* and light attenuation at 2 m depth, but not at 5 m depth (Table 18). While Coral Index scores are variable across the range of Chl *a* and k490 observed over recent years, predictions from the GBMs show that for the Burdekin and Wet Tropics regions in particular, "very poor" scores (below 0.2) are unlikely toward the lower end of the Chl *a* and light attenuation gradients, while "moderate" scores (above 0.4) become unlikely as conditions decline toward the higher end of observed Chl *a* concentrations and light attenuation (Figure 33).

Table 18. Relationships between coral reef communities and satellite derived estimates of water quality. Only combinations with statistically supported responses based on generalised linear models are presented. Environmental variables tested wet-season Chl *a* concentration between December 2021 and April 2025 and whole of year light attenuation k490 between July 2020 and June 2024. Tabulated values represent the predicted change in the response over the range of the environmental variable across monitoring reefs. Shading highlights responses that decline in response to declining water quality.

| Response | Depth | Environmental variable | Change across reefs | | |
|---|---|---|---|---|---|
| | | | mean | Credible intervals | |
| | | | | lowest | Highest |
| Coral Index score | 2 | Chl *a* | -0.2 | -0.34 | -0.06 |
| | | Light attenuation | -0.22 | -0.38 | -0.07 |
| Coral cover score | 2 | Chl *a* | -0.31 | -0.52 | -0.10 |
| | | Light attenuation | -0.38 | -0.59 | -0.16 |
| | 5 | Chl *a* | -0.26 | -0.49 | -0.02 |
| | | Light attenuation | -0.26 | -0.50 | -0.02 |
| Macroalgae cover | 2 | Chl *a* | 16 % | 0.5 % | 31.7% |
| | | Light attenuation | 17 % | 0.4% | 34% |
| Genus composition value | 2 | Chl *a* | -0.54 | -0.8 | -0.27 |
| | | Light attenuation | -0.59 | -0.88 | -0.31 |
| | 5 | Chl *a* | -0.92 | -1.23 | -0.62 |
| | | Light attenuation | -1.0 | -1.31 | -0.7 |

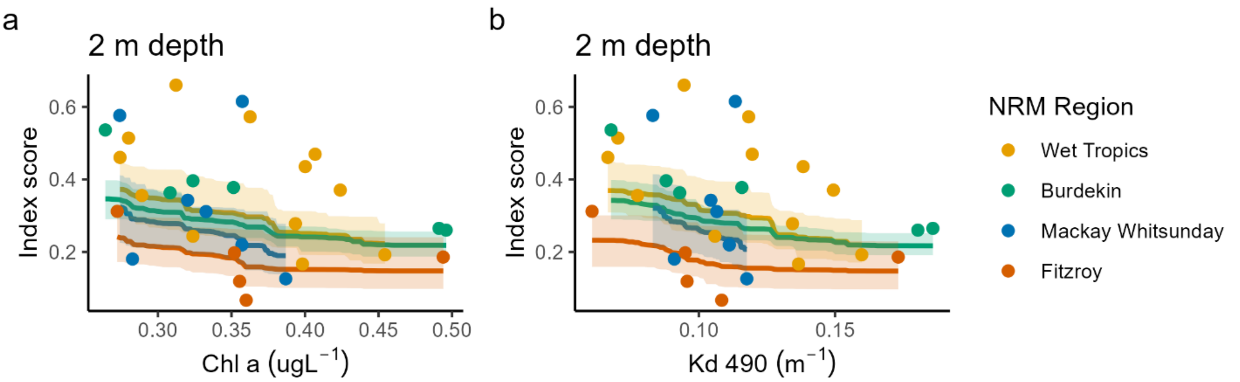



Figure 33. Relationships between Coral Index scores and satellite derived water quality. At 2 m depth relative to: a), concentration of Chl *a* and b), light attenuation (kd490)

Of the individual indicators, only Coral cover scores were negatively related to satellite derived water quality variables, a response observed at both 2 m and 5 m depths (Table 18). This relationship was most evident at 2 m depth, and particularly the Wet Tropics, where "very poor" scores were observed only at the higher end of the Chl *a* and light attenuation gradient (Figure 33).

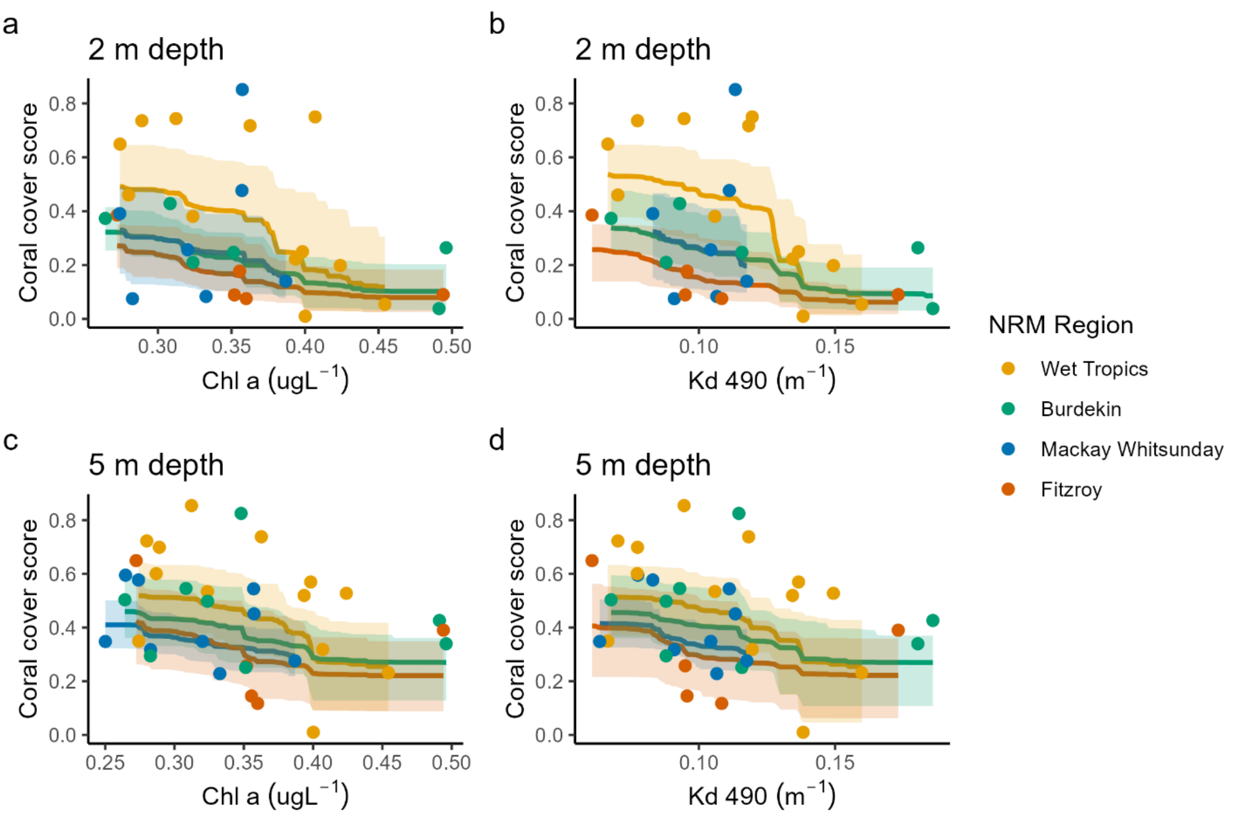



Figure 34. Relationships between Coral cover Indicator scores and satellite derived water quality. At 2 m depth relative to: a), concentration of Chl *a* and, b), light attenuation; and at 5 m depth relative to: c), concentration of Chl *a*, and d), light attenuation.

The Macroalgae indicator is scored against thresholds that vary along environmental gradients due to the recognised higher prevalence of these algae on shallow reefs in turbid inshore waters. While 2025 scores for the Macroalgae indicator showed no relationship to the satellite derived water quality estimates, the cover of macroalgae at 2 m depth was consistently low at the lowest values of Chl *a* and light attenuation compared to the variable cover at reefs across the rest of the water quality gradient (Figure 35).

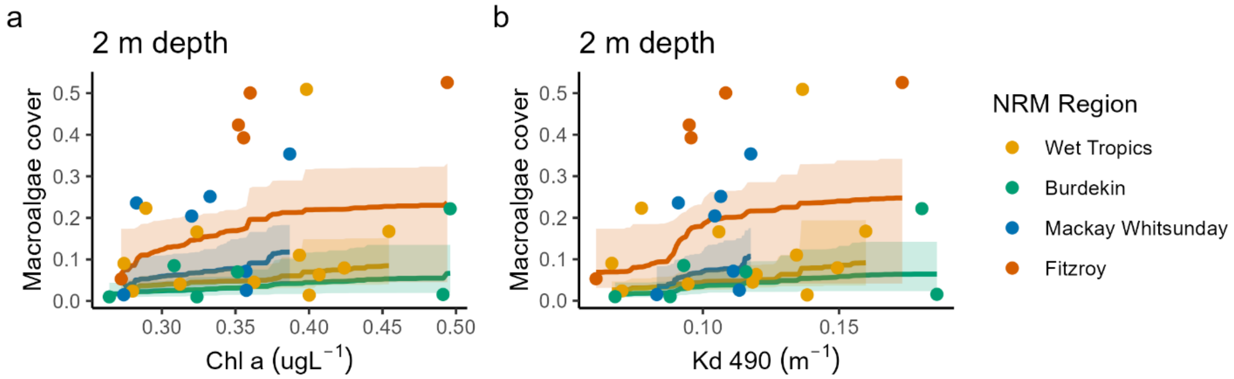



Figure 35. Relationships between Macroalgae cover and with satellite derived water quality. At 2 m depth relative to: a), Chl *a* concentration and b), light attenuation.

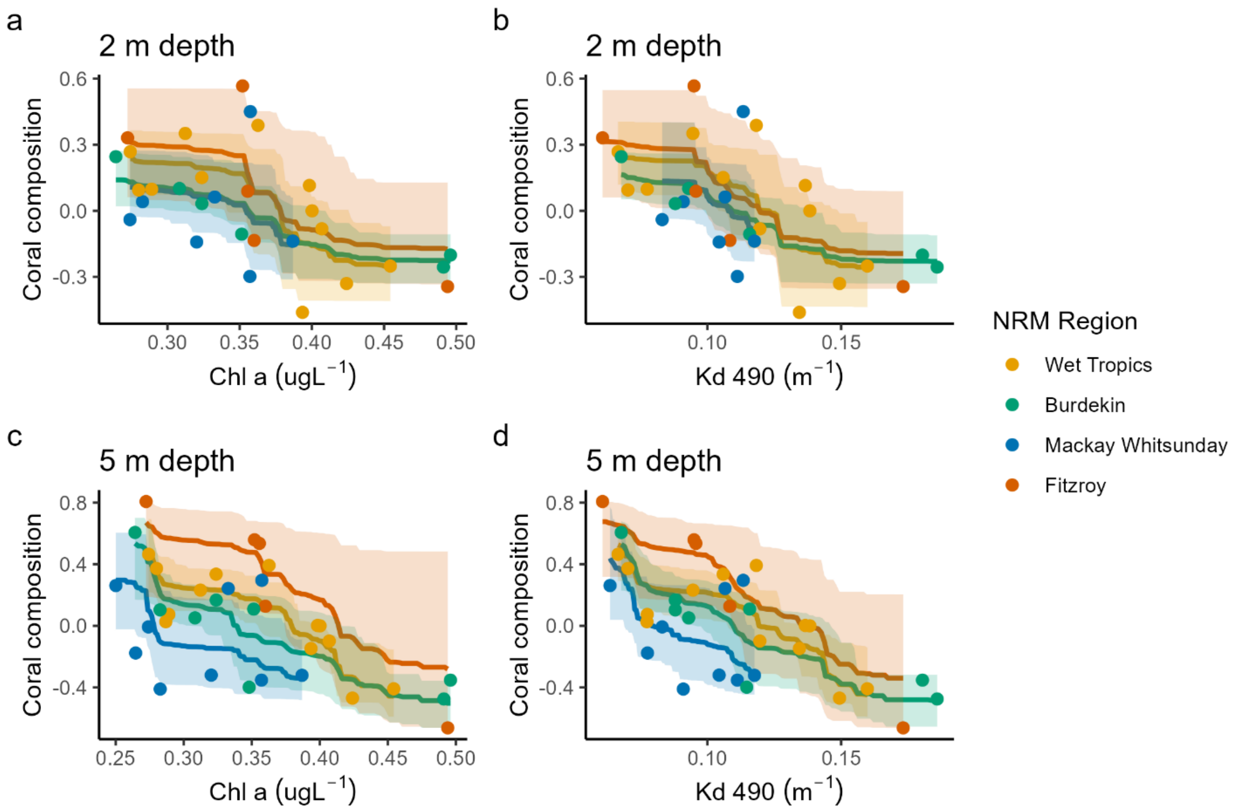



Figure 36. Relationship between hard coral community composition and satellite derived water quality. At 2 m depth relative to a), Chl *a* concentration and b), light attenuation; at 5 m depth relative to c), Chl *a* concentration and d), light attenuation. Coral composition values are the reef-level compositional estimates under-pinning the Composition indicator

The values for coral community composition that underpin the Composition indicator scores are based on the sum of covers for each hard coral genus multiplied by the genus score along a water quality vector (Table A4, see section 2.4.5 for detail). In 2025 community composition values at both 2 m (Figure 36a, b) and 5 m (Figure 36c, d) depth continued to vary along water quality gradients.

Relationships between coral reef community attributes at the subset of reefs at which water quality was physically monitored by the MMP were also apparent (Table 19, Table 20). The relationships observed were broadly consistent with those reported across all reefs and assessed against the satellite derived environmental gradients. Relationships between Coral Index or individual indicator scores and *in situ* logger based water quality estimates were more common at 2 m depth sites where scores for the Coral Index, Coral cover and Cover change scores decline with increasing turbidity (Table 19, Figure 37), with Coral cover and Cover change scores also declining with increasing Chl *a* concentration. Coral Index scores also declined across a gradient of increasing particulate N and P concentration, although the magnitude of change was lower than that estimated for the logger-based water quality measures (Table 19, Table 20).

Table 19. Relationships between coral reef communities and *in situ* logger derived water quality variables. Only combinations for which statistically supported responses based on generalised linear models are presented. Environmental variables tested were mean estimates of Chl *a* and Turbidity (ntu) over the period July 2021-June 2025. Reported values are the mean, lowest, and highest change in the coral community response variable across the range of water quality estimates among reefs. Shading highlights responses that decline in response with increasing levels of the water quality summaries.

| Response | Depth | Environmental variable | Change across reefs | | |
|---|---|---|---|---|---|
| | | | mean | Credible intervals | |
| | | | | lowest | Highest |
| Coral Index score | 2 | Turbidity | -0.21 | -0.37 | -0.05 |
| Coral cover score | 2 | Chl *a* | -0.46 | -0.64 | -0.29 |
| | | Turbidity | -0.34 | -0.50 | -0.19 |
| Juvenile score | 2 | Chl *a* | 0.31 | 0.16 | 0.47 |
| | | Turbidity | 0.22 | 0.06 | 0.38 |
| | 5 | Chl *a* | 0.55 | 0.32 | 0.77 |
| | | Turbidity | 0.42 | 0.17 | 0.66 |
| Cover change score | 2 | Turbidity | -0.62 | -0.89 | -0.36 |
| Community composition value | 2 | Chl *a* | -0.67 | -1.14 | -0.20 |
| | | Turbidity | -0.62 | -0.93 | -0.31 |
| | 5 | Turbidity | -0.40 | -0.74 | -0.05 |

Table 20. Relationships between coral reef communities and measured water quality variables. Only combinations for which statistically supported responses based on generalised linear models are presented. Environmental variables tested include mean values from niskin samples collected by the MMP over the period July 2021-June 2025; variables tested were concentrations of Particulate N, Particulate P, Phosphate and $NO_x$ (combined concentration of Nitrate and Nitrite). Reported values are the mean, lowest, and highest change in the coral community response variable across the range of water quality estimates among reefs. Shading highlights responses that decline in response with increasing concentration of nutrients.

| Response | Depth | Environmental variable | Change across reefs | | |
|---|---|---|---|---|---|
| | | | mean | Credible intervals | |
| | | | | lowest | Highest |
| Coral Index score | 2 | Particulate N | -0.18 | -0.34 | -0.02 |
| | | Particulate P | -0.18 | -0.37 | 0.01 |
| Macroalgae score | 2 | Particulate N | -0.44 | -0.69 | -0.19 |
| | | Particulate P | -0.38 | -0.70 | -0.05 |
| Macroalgae proportion | 2 | Particulate N | 0.42 | 0.19 | 0.64 |
| Macroalgae cover | 2 | Particulate N | 0.35 | 0.20 | 0.51 |
| | | Particulate P | 0.35 | 0.15 | 0.55 |
| Community composition value | 2 | Particulate N | -0.56 | -0.92 | -0.21 |
| | | Particulate P | -0.73 | -1.01 | -0.44 |
| | 5 | Particulate N | -1.06 | -1.42 | -0.70 |
| | | Particulate P | -1.22 | -1.51 | -0.93 |

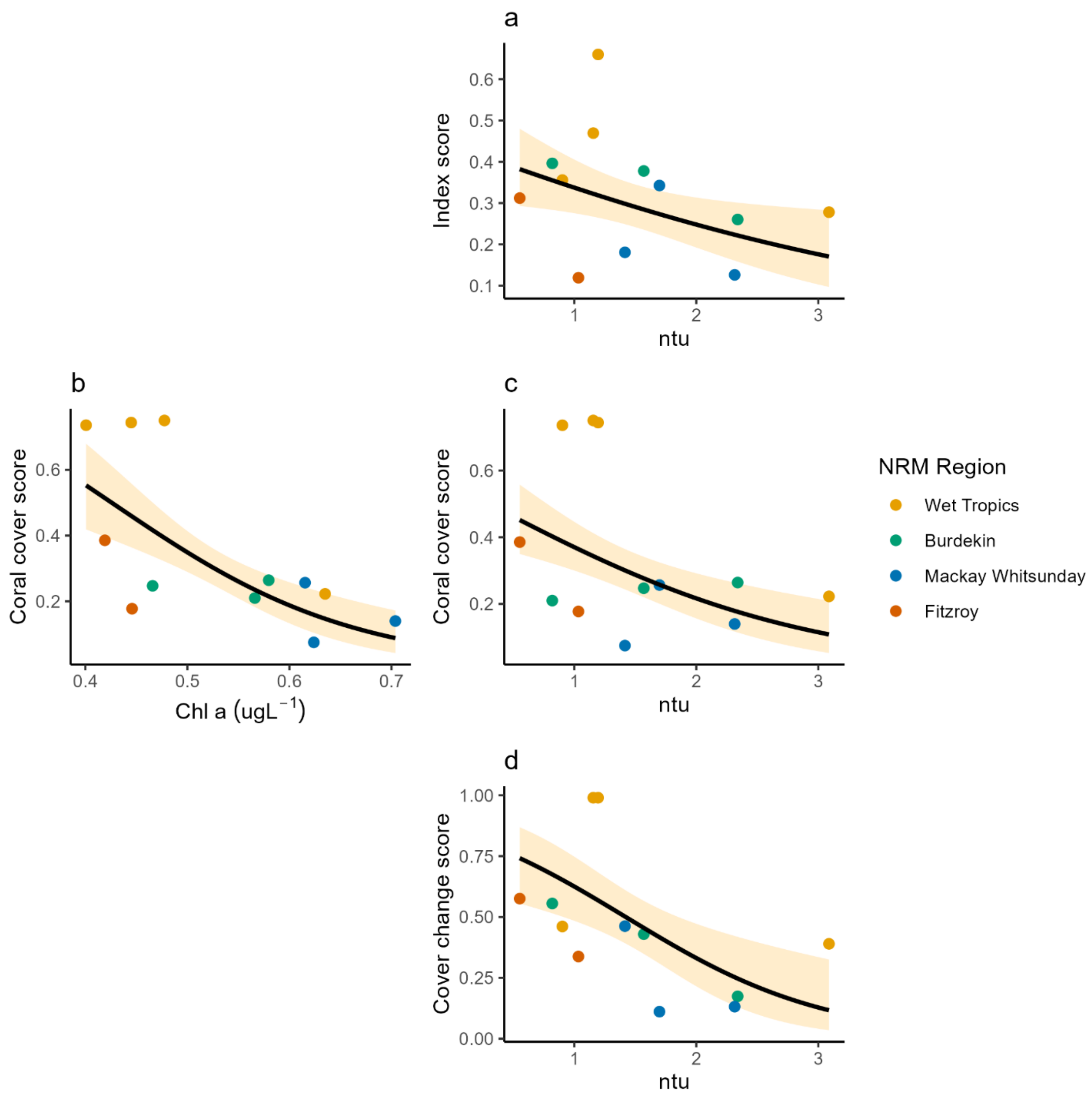



Figure 37. Relationships between coral community scores water quality FLNTU data. Plots show the relationships observed at 2 m depth between: a), Coral index scores and ntu, b), Coral cover scores and Chl *a* concentration, c), Coral cover scores and ntu, and d), Cover change scores and ntu. Trends represent predicted relationships with 95% credible intervales derived from GLM models. Points represent the observed data. Estimates of Chl *a* and ntu are mean of daily means over the 5 years to 30th June 2025 sourced from MMP FLNTU loggers at reef sites.

In contrast, scores for the Juvenile coral indicator currently show a positive relationship to concentration of Chl *a* and Turbidity at both 2 m (Figure 37e, f) and 5 m (Figure 38) depths.

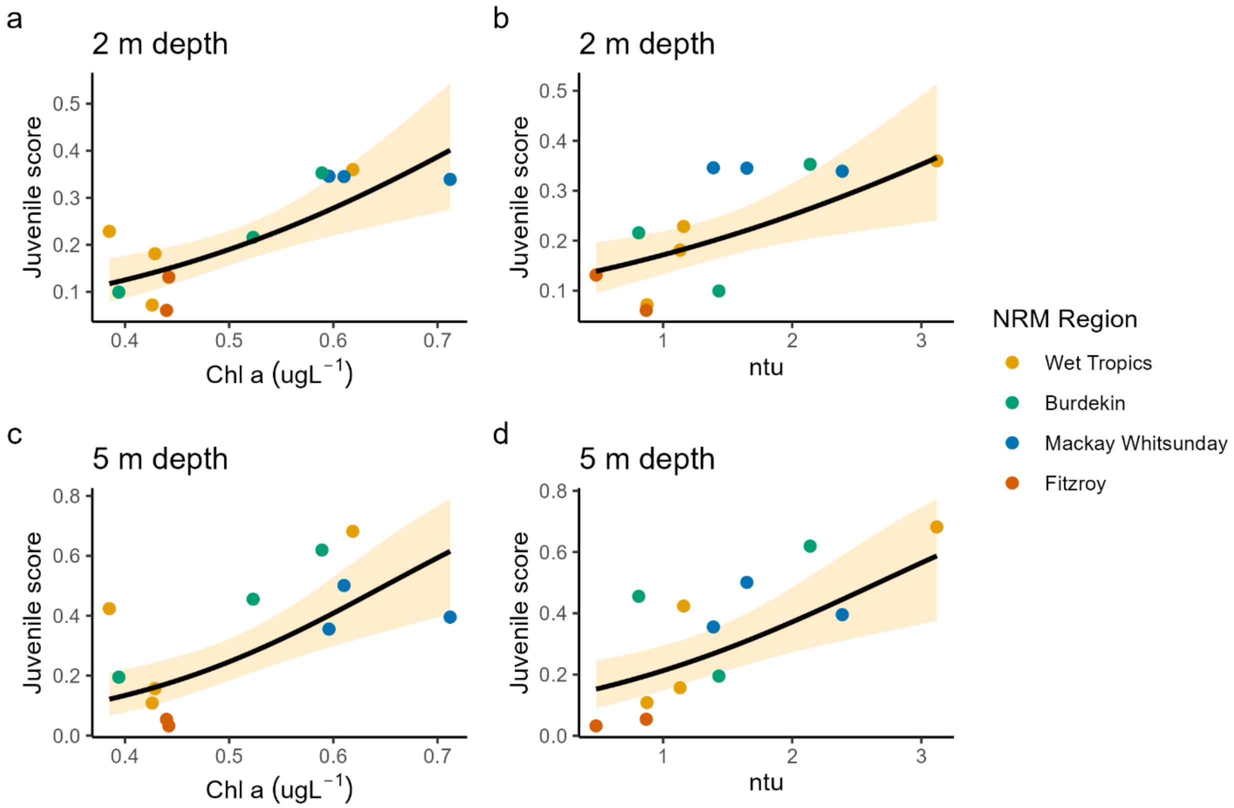



Figure 38. Relationships between Juvenile indicator scores and measured water quality. Trends represent predicted relationships with 95% credible intervals derived from GLM models. Points represent the observed data. Estimates of Chl *a* and ntu are mean of daily means over the 5 years to 30th June 2025 sourced from MMP FLNTU loggers at reef sites.

As observed in relation to the satellite derived water quality measures, the composition of coral communities varied along measured water gradients (Table 19, Table 20). At both 2 m and 5 m depths genus composition values varied along gradients of Turbidity (ntu) measured by FLNTU loggers and concentrations of particulate N and P in niskin samples. Only the relationship for ntu estimates are presented in Figure (39), as they are representative of those seen for the particulate nutrients.

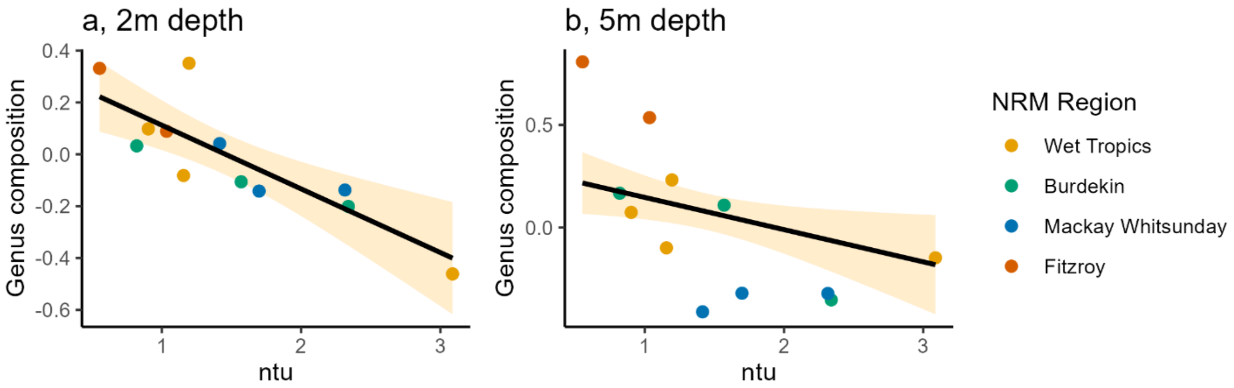



Figure 39. Relationships between coral community composition value and ntu. Trends represent predicted relationships with 95% credible intervals derived from GLM models. Points represent the observed data. Genus composition values are data underpinning estimates of the Composition score. Estimates of ntu are mean of daily means over the 5 years to 30th June 2025 sourced from MMP FLNTU loggers at reef sites.

Of the water quality variables tested, variability in the Macroalgae indicator and the underlying prevalence of macroalgae was evident in relation to concentrations of particulate N and P (Table 20). While minimum scores of zero were observed at a high proportion of sites, there was evidence for a decline in scores with increasing nutrient concentration (Figure 40a, b). This decline in score was reflected in the positive relationship between proportion of the reef substrate occupied by macroalgae (% cover, Figure 40c, d) and representation of macroalgae within the cover of all algae (macroalgae proportion, Figure 40e) – that underpins the Macroalgae indicator score.

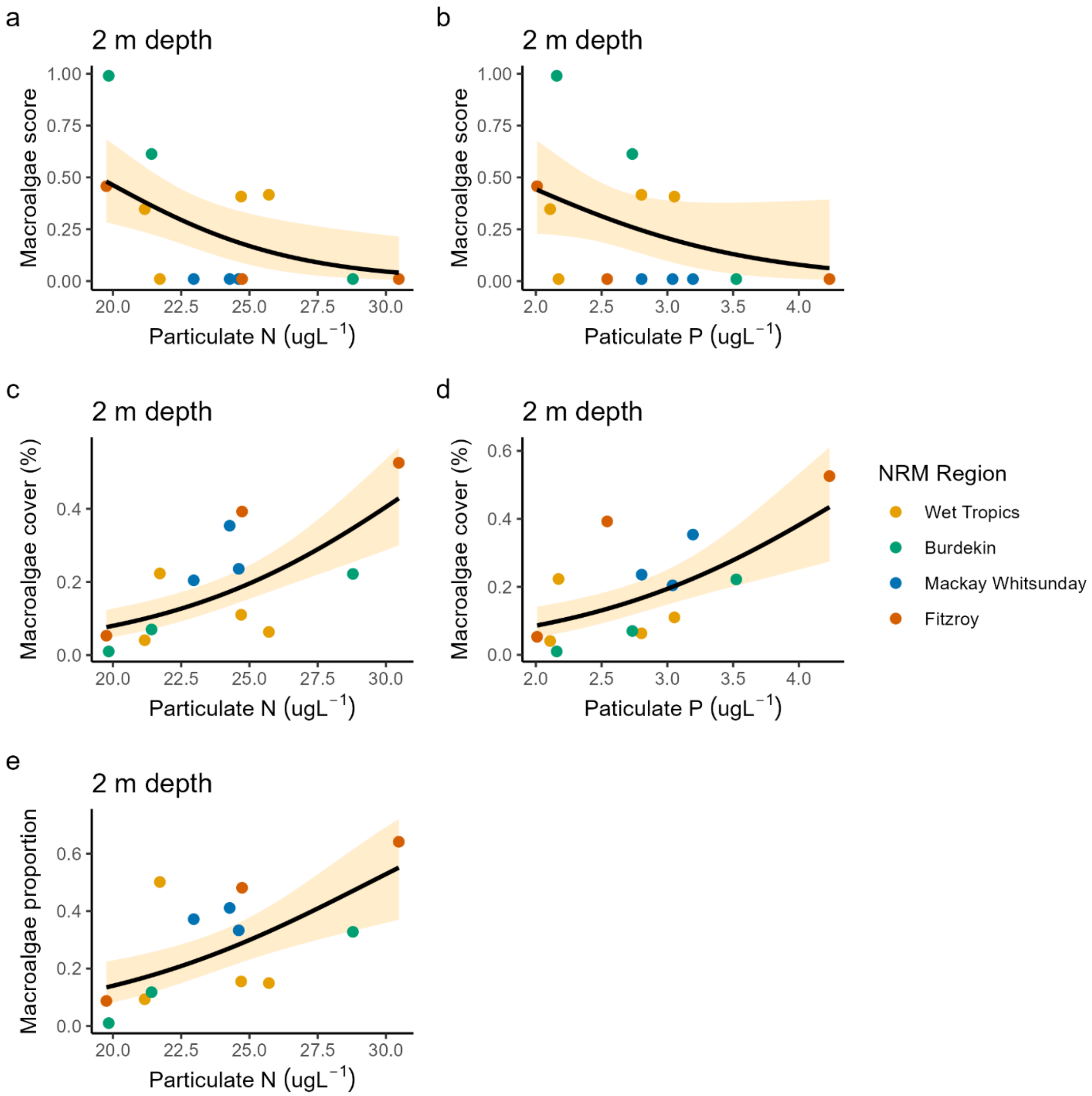



Figure 40. Relationships between macroalgae and measured nutrient concentrations. Trends in Macroalgae indicator scores and a), particulate N, and b), particulate P; macroalgae cover and c), particulate N and d), particulate P; e), the proportion of macroalgae cover in total algae cover and particulate N. Trends represent predicted relationships with 95% credible intervals derived from GLM models. Points represent the observed data.

### *4.4.2 Influence of discharge, catchment loads and water quality on reef recovery*

During periods free from acute disturbances (cyclones, thermal bleaching, crown-of-thorns starfish outbreaks or direct exposure to low salinity floodwaters), the recovery of reefs, as measured by biennial change in the Coral Index scores, was negatively related to discharge from the local catchments in each region other than Mackay–Whitsunday (Figure 41). Importantly, these relationships consider only the contemporary influence of environmental conditions on the indicators during recovery periods. Any influence of water quality on the severity of response to disturbance events will not be included. Similarly, lagged influences such as short-term reductions in the cover of macroalgae before their rapid recovery in the following year, that have been observed on several occasions following cyclones and floods, this will result in the underestimation of the response.

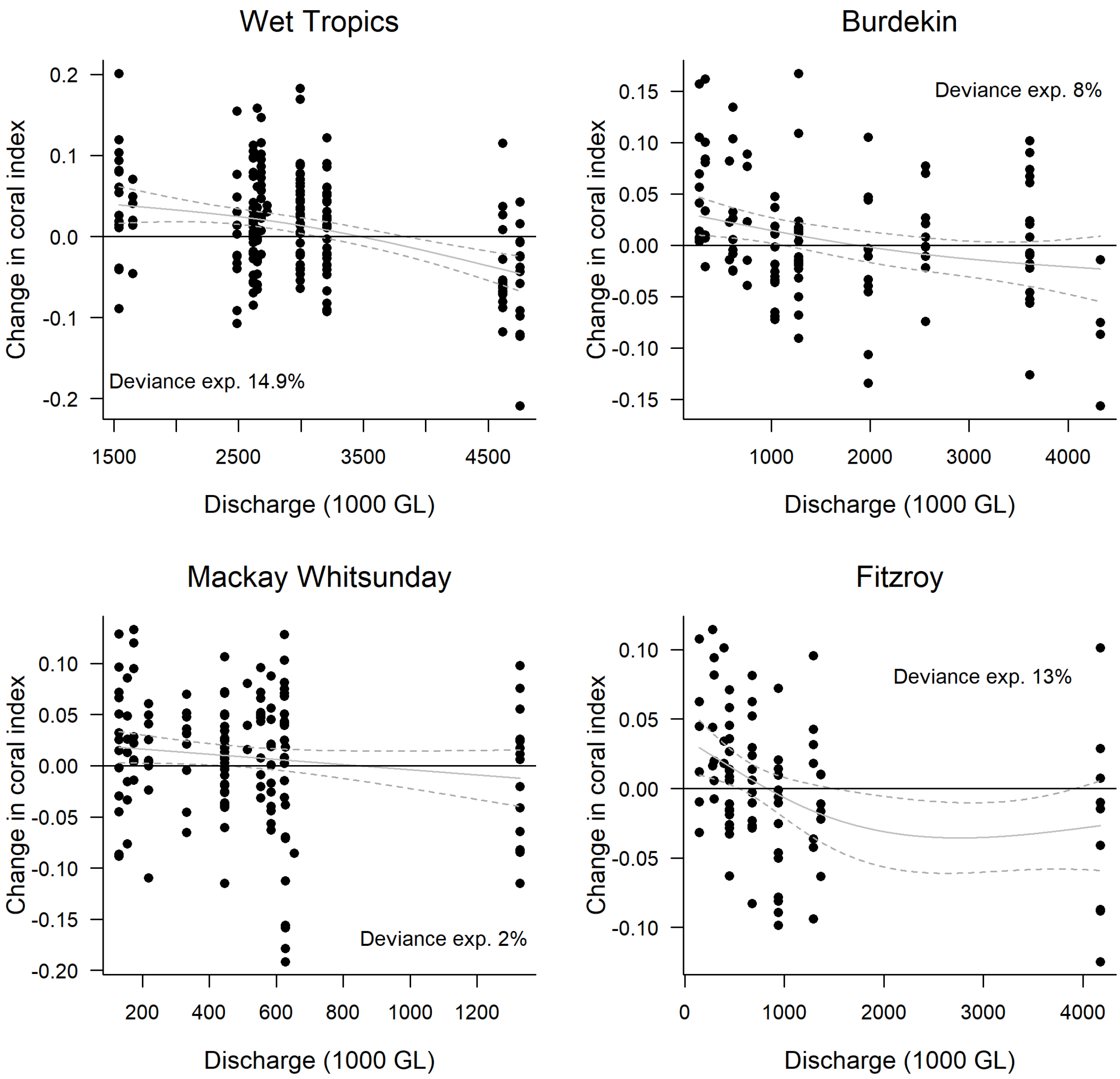



Figure 41. Relationship between the Coral Index and regional freshwater discharge. Plotted points represent observed change in the Coral Index score at each reef and depth over a 2-year period. Observations following years for which acute disturbances impacted communities in the period between samples were excluded. Discharge values represent the maximum annual discharge from the region's major rivers over the 2-year period corresponding to Coral Index changes. Trend lines represent the predicted change in Coral Index scores (solid line) and the 95% confidence intervals of the prediction (dashed lines).

# 5 DISCUSSION

As coral reefs are naturally dynamic systems that alternate between decline from impacts and periods of recovery (Connell 1978), it is critical for the persistence of coral communities that there is a long-term balance between these processes. This balance can only be achieved if there is sufficient time between disturbance events and favourable environmental conditions that promote recovery during intervening periods. The *Driver-Pressure-State-Impact-Response* framework (Maxim *et al.* 2009, Rehr *et al.* 2012) allows the identification of some of the key drivers and pressures influencing coral community condition with the potential to disrupt the disturbance recovery cycle.

In general, a desire for social and economic development are the primary *drivers* of human activities that can result in local scale *pressures* on Reef ecosystems, such as increased exposure to sediments, nutrients and toxicants, through to the global *pressure* of climate change. In this context, we consider both climate-related acute disturbances such as cyclones and marine heat waves, which are beyond the realm of management under the Reef 2050 Water Quality Improvement Plan (Reef 2050 WQIP), and those, such as water quality or crown-of-thorns starfish, which may be more locally manageable. A primary focus of this component of the MMP is assessing the role of water quality in the observed *state* of the Reef ecosystems.

The following sections summarise the *pressures* that have influenced the *state* of inshore coral communities. We consider *state* in terms of the resilience of coral communities in recognition that reefs will naturally be at different stages of a disturbance-recovery cycle and it is their ongoing capacity to resist, or recover from pressures, that is important information for managers. Assessing any *impact* this state may have on the socioeconomic services provided by inshore coral reefs and possible management *response* that may be warranted to mitigate these impacts are both beyond the scope of Marine Monitoring Program.

## 5.1 Pressures

### *5.1.1 Acute disturbances*

*Cyclones.* Since MMP surveys began in 2005, inshore reefs have been impacted by multiple acute disturbance events. Cyclones and storms remain the primary cause of hard coral cover losses on inshore reefs accounting for 37% of losses since 2005. Unsurprisingly it has been the intense category 4 and 5 systems, i.e., cyclone Larry (Wet Tropics and Burdekin regions – 2006), cyclone Yasi (Wet Tropics and Burdekin regions – 2011) and cyclone Debbie (Whitsunday region – 2017) that have caused the greatest losses with storm driven waves physically damaging or stripping corals from the reefs.

Since cyclone Debbie, observations of physical damage were recorded in 2024, associated with Cyclone Kirrily that caused minor damage to some reefs in the Burdekin region, and Cyclone Jasper that impacted reefs at Snapper Island and Fitzroy Island in the Wet Tropics. In 2025, physical damage was observed at reefs in the Burdekin Region, most notably Pandora Reef, and attributed to a localised storm within an active monsoon.

*Flooding.* In addition to the damage caused by waves, both Cyclone Jasper in December 2023 and the monsoon in early 2025 resulted in intense rainfall and extreme flooding. Exposure to low salinity floodwaters from the Daintree River in late 2023 killed all corals at both the 2 m and 5 m depth monitoring sites at Snapper South, marking the most severe impact of any acute event observed during the 20 years of the MMP. Flood damage was also recorded at the 2 m depth at High East.

Surveys in 2025 revealed further flood impacts. High levels of coral mortality at 2 m depths at Dunk Island, Bedarra Island, and Lady Elliot Reef, and to a lesser degree Palms West, were all consistent with exposure to low salinity floodwaters. Indeed, a salinity logger maintained by the MMP Water Quality group at Dunk Island recorded values below 15 PSU at one of the coral monitoring sites at Dunk North during low tides on several occasions in early February 2025, and a cumulative exposure to salinity below 22 PSU for 2.2 days over a 2-week period in early February. The response of corals to low salinity appears to follow a dose-time relationship, with a mortality threshold for *Acropora* of 3 days exposure to 22 PSU extending out 16 days at 28 PSU estimated by Berkelmans et al. (2012), noting this estimate considers survival several months after exposure and so includes mortality from both the initial exposure and subsequent stress. Immediate mortality of *Acropora* can occur in less than a day at PSU values below 15 (Balanay-Quiñones et al. 2009). It is highly likely that the reefs where greater impacts of the flood occurred would have been more exposed to low salinities than recorded by the logger, as the logger site is in the lee of Dunk Island in a location where the island wake effect (Wolanski et al. 1996) is likely to have resulted in upwelling of higher salinity waters during the flood event, in contrast to more severely impacted sites that face into the plume.

Previous losses of coral cover due to exposure to low salinity floodwaters recorded by this Program, have been limited to 2 m depths on reefs and include: south of Great Keppel Island in the Fitzroy region in 2008 and 2011, Snapper South in 2019 and High West in 2009 and 2011. In each case, these exposures coincided with maxima in the daily discharges from the adjacent catchments. More frequent exposure to low salinity waters will have limited the development of coral reefs closer to major rivers.

*Thermal stress.* A severe marine heat wave occurred in early 2024 resulting in the highest accumulation of heat stress yet recorded on reefs monitored by the MMP. The highest heat stress occurred in the Fitzroy region, where over half the coral at 2 m depth was killed by coral bleaching. Losses of coral cover attributed to coral bleaching in 2024 were also recorded in the Burdekin and Wet Tropics regions. Importantly, during surveys in May 2024, much of the surviving coral in the Fitzroy region was either partially or fully bleached and surveys in 2025 revealed further loss of coral cover with the proportion of cover lost at 2 m depth increasing to 73% compared to 2023 levels. Averaged over the 2 m and 5 m depths, 57% of coral cover was lost between 2023 and 2025. A similar scenario occurred at One Tree Island (Byrne *et al.* 2025). Delayed mortality in response to bleaching stress was observed in 2018 and 2021 following the 2017 and 2020 marine heatwaves, respectively.

This most recent bleaching event builds on previous events in 2006, 2017, 2020 and, to a lesser extent, 2022, with coral bleaching now accounting for 20% of coral cover loss on inshore reefs since 2005. Temperature reconstructions suggest that the 2017, 2020 and 2024 marine heat waves represent the warmest conditions in at least 400 years (Henley *et al.* 2024).

At moderate levels of heat stress, coral cover may not decline sufficiently for us to ascribe an acute disturbance event. However, the presence of bleached or partially bleached corals during surveys indicates some degree of stress. In such situations, it is possible that reduced Cover change scores resulted from slower growth of these stressed corals. Such lagged effects of disturbances, as well as the potential that the impact of acute events may be exacerbated by chronic pressures such as poor water quality (see below) will add some uncertainty to apportioning losses to specific pressures.

During the 2020 and 2024 bleaching events the proportion of coral lost due to bleaching was greater at the 2 m depth sites than at the adjacent 5 m depth sites. This observation is consistent with both previous reports of reduced severity of bleaching with depth (e.g., Muir *et al*. 2017, Cantin *et al.* 2021) and the finding that that oxidative stress increases with increasing irradiance Lesser (2024). In turbid water, reduced light intensity with increased depth and/or self-shading due to increased symbiont loads have been identified as mechanisms that provide some resistance to bleaching at deeper depths (Anthony *et al*. 2007, Sully and van Woesik 2020). Alternatively, differences in the susceptibility of corals based on taxonomic differences between depths may also play a role (Marshall & Baird 2000).

Except for reefs in the Fitzroy region, the inshore reefs monitored by the MMP have had lower loss of coral cover due to thermal stress than some offshore areas of the Reef (Hughes *et al.* 2018). Considering the magnitude of thermal stress across the Reef in 2016, 2017, 2020 and 2024, it seems clear that inshore reefs, other than in the Fitzroy region have, to date, been spared the magnitude of thermal stress that resulted in widespread mortality of corals elsewhere (Hughes *et al.* 2018). Worryingly, it is becoming clear that the frequency and severity of such events have increased, and are likely to continue to do so, as the climate continues to warm (van Hooidonk *et al*. 2017, Oliver *et al.* 2019, McWhorter *et al.* 2022, Emslie *et al.* 2024).

*Crown-of-thorns starfish*. Since 2005 the Wet Tropics is the only region in which crown-of-thorns starfish have been common on MMP sites. Individual starfish have also been recorded in the Burdekin at Palms East (2016) and Palms West (2019, 2024), and culling has occurred in the outer Whitsunday Islands. In 2025, outbreak densities of crown-of-thorns starfish were observed on the eastern aspects of Fitzroy Island, the Frankland Group and High Island. In recent years, the Crown-of-thorns Starfish Control Program has attempted to mitigate the impact of crown-of-thorns starfish by removing over 19,000 individuals[2] from Fitzroy Island and the Frankland Group since 2013, with over 2000 of these from the Frankland Group in the 2023-24 and 2024-25 financial years. Consistent across the cull data and MMP observations have been records of relatively high proportions of juvenile crown-of-thorns starfish in the population, signifying their ongoing recruitment and potential for future impacts.

In recent years, coral lost to crown-of-thorns starfish predation has contributed to keeping the Coral Index score below the ‘good’ range in the Johnstone Russell-Mulgrave sub-region.

In combination, acute disturbance events contribute strongly to the declines in the Coral cover (Lam *et al.* 2018) and Coral Index scores. The long-term maintenance of coral community condition requires that recovery processes keep pace with the impact of disturbances. For the MMP, it is important that acute disturbances are identified and quantified so that the potential for subsequent recovery can be assessed. The quantification of acute disturbance is largely based on changes in the cover of hard corals. Each of the remaining indicator metrics has been formulated to limit responsiveness to acute pressures and to focus, as directly as possible, on responses to chronic pressures, such as water quality during periods of reef recovery.

The reader must be aware, however, that while quantifying both acute and chronic pressures helps to focus on reef recovery processes, it is inevitable that acute and chronic pressures interact. In short, quantifying the impact of acute pressures will include the cumulative response of the identified pressure and any additional sensitivity of the coral community to that pressure because of local environmental conditions.

### *5.1.2 Chronic conditions – water quality*

Water quality is a summary term for a range of chemical and physical properties of marine waters that exert a fundamental influence on the processes governing ecosystem health. Water quality in the inshore Reef shows a strong gradient, improving with distance from the coast and from major river outfalls. Variation in benthic communities on coral reefs along these gradients provides clear evidence for the selective pressures imposed by water quality (van Woesik & Done 1997, van Woesik *et al.* 1999, Fabricius *et al.* 2005, DeVantier *et al.* 2006, De’ath & Fabricius 2008, Uthicke *et al.* 2010, Fabricius *et al.* 2012, Luo *et al.* 2022). The physical properties of the sites, such as hydrodynamic conditions and depth, also contribute to selective pressures (Browne *et al*. 2010, Thompson *et al.* 2010, Uthicke *et al.* 2010).

Such gradients are both a natural part of the Reef ecosystem, and the result of run-off-derived pollutants that have increased since European development of the Reef catchment (Belperio & Searle 1988, Waters *et al.* 2014). The premise underpinning the Reef 2050 WQIP is that anthropogenic contaminant loads delivered by rivers create conditions that suppress the health or resilience of the Reef’s inshore ecosystems. The core focus of the water quality monitoring component of the MMP (see separate report by Gruber *et al*. 2026) is the quantification of both the compounding influence of run-off on the naturally occurring gradients and any subsequent improvement due to the activities under the Reef 2050 WQIP.

[2] Australian Government Crown-of-thorns Starfish Control Program data supplied by Great Barrier Reef Marine Park Authority, Eye on the Reef.

For corals, the pressures relating to land management practices influence the 'state' of marine water quality. The MMP river plume monitoring and exposure mapping (see Gruber *et al.* 2026) clearly shows that inshore reefs are directly exposed to elevated loads of sediments and nutrients delivered by rivers. Such plumes may be considered acute pressures, especially when waters with lethally low levels of salinity reach corals, as has occurred in the Wet tropics and Burdekin regions over the last 2 years. For most inshore reefs, however, it is the chronic exposure to increased sediment and nutrient loads delivered to the Reef that is likely to influence the resilience of corals.

Turbidity in the Reef lagoon is strongly influenced by variations in the inflow of particles from the catchment but also resuspension by wind, currents and tides (Larcombe *et al.* 1995, Bainbridge *et al.* 2018). The additional flux of fine sediment imported by rivers may remain in the coastal zone for periods of months to years, leading to chronically elevated turbidity (Wolanski *et al.* 2008, Lambrechts *et al.* 2010, Brodie *et al*. 2012, Fabricius *et al.* 2013, Fabricius *et al.* 2014, Fabricius *et al.* 2016, Baird *et al*. 2019, Thompson *et al.* 2020). Long residence times for fine sediments and the potential for mobilisation of sediments by cyclones will blur direct relationships between river loads and marine conditions. However, any increase in turbidity associated with run-off will reduce the level of photosynthetically active radiation (PAR) reaching the benthos; PAR is a primary energy source for corals and so a key factor limiting coral productivity and growth (Cooper *et al.* 2007, Muir *et al.* 2015). Although it should be noted that corals can supplement their energy intake by heterotrophic feeding (Yu *et al.* 2023), a capacity that varies among species (Anthony 1999, Anthony & Fabricius 2000) and contributes to differences in coral community composition along water quality gradients.

In general, our observed relationships between more positive changes in Coral Index scores following relatively low discharge from local rivers are consistent with a reduction in pressures associated with run-off, such as those documented by Bruno *et al.* (2003), Kuntz *et al*. (2005), Kline *et al*. (2006), Voss & Richardson (2006), Kaczmarsky & Richardson (2010), Haapkylä *et al.* (2011, 2013), and Vega Thurber *et al.* (2013). The observed relationship between discharge and changes in the Coral Index implies that the cumulative impacts of river-delivered contaminants suppress the resilience of coral communities. Failure to observe a relationship between discharge and change in the Coral Index scores in the Mackay–Whitsunday region is likely due to the overwhelming influence of regular tidal resuspension and findings by Baird *et al.* (2019) that suggest fine catchment-derived sediment can form a turbid layer persisting for several years. This phenomenon will reduce the direct influence of acute run-off events on the variability in conditions and in particular, turbidity experienced by corals. There is strong vertical differentiation in benthic community composition at many Mackay–Whitsunday reefs, where communities at the deeper 5 m sites include a high representation of species tolerant to high turbidity, demonstrating the long-term selective pressure imposed by ambient conditions that also, potentially, limit their sensitivity to additional pressures imposed by run-off; a point raised by Morgan *et al.* (2016).

We are mindful, however, that interannual change in Coral Index scores was highly variable among reefs. This is expected as Coral Index scores at any point in space or time will reflect the cumulative responses of the communities to past disturbance events, variable exposure to water quality pressures, and natural stochasticity in the population dynamics of the diverse communities inhabiting these reefs. In combination, variable exposure to past events and location-specific pressures are also likely to have been selected for communities tolerant of those conditions (De Vantier *et al.* 2006). This means communities in different locations will have different susceptibility to water quality pressures (e.g., Morgan *et al.* 2016). It is precisely the inability to measure or predict accurately cumulative impacts across a diversity of exposures that supports the use of biological indicators, such as the coral and seagrass (Collier *et al.* 2021) indices in the MMP, as tools to identify where and when environmental stress is occurring (Karr 2006, Crain *et al.* 2008).

It is evident from the MMP marine water quality time series that there were gradual declines in water quality through to approximately 2018 in the wet Tropics and 2016 in the Burdekin and Whitsunday regions. A feature of the decline was a general increase in oxidised forms of dissolved nitrogen ($NO_x$) and dissolved organic carbon (DOC). Lønborg *et al.* (2015) suggest that these observations indicated changes in the carbon and nutrient cycling processes in the Reef lagoon, although the detailed understanding of these processes remains elusive. In 2025, concentrations for both these water quality parameters remain high, although $NO_x$ concentrations appear to have declined in recent years in most regions.

Of direct relevance to corals is that both increased DOC and nutrient concentrations have been shown to influence the microbiome of corals with potential to shift microbial fauna to a more pathogenic state (Kuntz *et al.* 2005, Kline *et al.* 2006, Vega Thurber *et al.* 2009). An emerging concept is that DIN enrichment can lead to an imbalance in the N:P ratios within the corals' symbiotic algae that reduces the provision of carbon to the coral. This, in turn, increases their susceptibility to thermal stress and reduces energy available for recovery (Morris *et al*. 2019). A potential mechanism is that elevated water column concentration of DOC during heat stress may decrease the threshold at which a disruption of the coral–algae symbiosis occurs by increasing coral-associated nitrogen fixation rates that further enhance the availability of N to algal symbionts (Rädecker *et al.* 2015, Pogoreutz *et al*. 2017).

Increased water column $NO_x$ concentrations may also promote growth in macroalgae. Work by Schaffelke and Klumpp (1998) demonstrated the potential for increased growth of the brown macroalgae *Sargassum* with the addition of inorganic N and P that were within levels measured by the MMP, and that either nutrient may be limiting depending on the time of year and concentrations present in the field. However, the water column $NO_x$ concentrations observed at MMP sites are low in comparison to P concentrations, suggesting increased $NO_x$ concentrations have the potential to increase the growth of *Sargassum* or possibly extend its range along the water quality gradient.

Over the period of the MMP, losses of coral cover attributed to disease and chronic pressures account for 22% of hard coral cover losses. These losses are likely to include the impacts of poor water quality as elevated levels of nutrients and fine organic sediments have been shown to increase the susceptibility of corals to disease (Bruno *et al.* 2003, Haapkylä *et al*. 2011, Kline *et al*. 2006, Kuntz *et al*. 2005, Weber *et al.* 2012, Vega Thurber *et al.* 2013). However, this figure is likely to be an underestimate, as losses attributed to acute disturbances will include any compounding impacts associated with chronic water quality pressures such as poor water quality magnifying the effects of heat stress events (Wiedenmann *et al.* 2013, Fisher *et al.* 2019, Cantin *et al.* 2021, Brunner *et al.* 2021). For example, the loss of coral cover in 2024 at A case in point was Fitzroy Island West was attributed to thermal stress based on observations of coral bleaching during summer bleaching surveys (Cantin *et al.* 2024). However, the losses were higher than might be expected from the moderate levels of heat stress recorded and coincided with exposure to floodwaters (Moran *et al.* 2025). Similarly, high levels of disease were observed amongst *Acropora* communities in the Herbert–Tully sub-region where reefs were exposed to moderate levels of heat stress and above median discharges from the local catchments.

The transport of coastal nutrients to the mid-shelf Reef remains a plausible factor enhancing the survival of crown-of-thorns starfish larvae, and so potentially extends the influence of run-off to large tracts of the Reef (Brodie *et al.* 2005, Fabricius *et al.* 2010, Furnas *et al.* 2013, Pratchett *et al*. 2014, Wooldridge & Brodie 2015, Brodie *et al.* 2017). However, the role of run-off in crown-of-thorns starfish outbreak dynamics remains unresolved (Pratchett *et al.* 2017). It is worth noting that the highest concentration of $NO_x$ has been consistently recorded in the Johnston Russell—Mulgrave subregion where there is a persistent presence of juvenile crown-of-thorns starfish.

## 5.2 Ecosystem state

### *5.2.1 Reef-wide coral community condition based on the Coral Index*

In 2025, the Reef-level Coral Index score improved slightly from the lowest recorded value in 2024. This improvement reflected the strong lift in condition observed in the Mackay–Whitsunday region with scores declining in other regions. It is to be expected that low points in the Coral Index will occur in the aftermath of disturbance events that cause a reduction in coral cover, such as recent flooding, cyclone Jasper and the severe marine heat wave that occurred in early 2024. However, the long-term persistence of coral communities requires a balance between the impacts of such events and the subsequent recovery of communities. Of concern is that at the scale of the inshore zone monitored by the MMP, impacts to coral communities over the last 20 years have outweighed their recovery. Prior to the impacts imposed by the 2023/24 summer the overall Coral Index had not recovered since declining to 'poor' in 2019. Although the Coral cover scores did improve over this period, they remained well below those observed in the early years of MMP. In addition to Coral cover, the Coral Index includes indicators aligned with community recovery potential. Of these, the Cover change and Juvenile indicator scores hovered around the juncture between "poor' and 'moderate' and Macroalgae scores declined within the 'poor' category. In combination these are of concern as they indicate reduced recovery potential of coral communities in recent years.

The cycle of disturbance and recovery, and the resulting coral community condition in 2025, does however vary among the regions as summarised below. These brief summaries should be considered in the context of section 5.3 where the interpretations of the individual indicators are presented.

### *5.2.2 Wet Tropics Region*

At the regional level, the Coral Index scores continued to decline, following on from a sharper decline to 2024, but remained 'moderate'. Scores for Coral cover, Cover change and Juvenile indicators all declined, contrasting with small improvements in Composition and Macroalgae scores. Despite these changes, there was no change to score classifications with the Juvenile indicator remaining 'poor' and all other indicators 'moderate'. Index scores declined in the Barron–Daintree and Herbert–Tully sub-regions and remained largely unchanged in the Johnstone Russell–Mulgrave subregion, as explained on below.

In general, most reefs have demonstrated a clear potential for recovery during periods free from acute disturbance events, with coral cover increasing across the region through to 2023. Notable exceptions have been Bedarra and Dunk South 5 m; the reefs with the highest Chl *a* and light attenuation coefficients across the region.

#### *5.2.2.1 Barron–Daintree sub-region*

In 2025, the Barron–Daintree sub-region score declined slightly and remained 'poor'. This ongoing decline follows the steep decline in 2024 caused by the severe impacts of Cyclone Jasper and associated flooding on the reefs at Snapper Island. The decline in 2025 primarily reflects reduced scores for the Macroalgae and Cover change indicators. Although the 2024 surveys for Low Isles occurred before the onset of cyclone Jasper, surveys in 2025 revealed there was little damage and Coral cover scores remained virtually unchanged, with the loss of coral cover recorded at Low Isles in 2025 compensated for by increased cover at Snapper Island North.

Cover change scores are averaged over a four-year window but exclude observations when hard coral cover was impacted by an acute disturbance. Further, the underlying model cannot predict a change in hard coral cover from a starting cover of zero, as occurred in 2024 at Snapper South. Hence the decline in this indicator from 'good' in 2024 to 'moderate' in 2025 does not assess the continued lack of hard coral at Snapper South.

In 2024 there was a steep improvement in the Macroalgae score as the impacts from cyclone Jasper and flooding reduced both the cover of macroalgae and corals, with the bare space created initially colonised by algal turfs. The decline in the indicator in 2025 captures the recolonisation by macroalgae, particularly at Snapper South 5 m depth.

Of concern is that the Juvenile coral score has remained in the lower end of the 'poor' range for most of the time since 2011, which will likely influence the potential for the rapid recovery of coral communities. The few instances where scores improved coincide with 2 strong cohorts of *Acropora* recruits at Snapper South. Here densities of juvenile corals are typically low, which suggests limited larval supply, a situation unlikely to be improved by the loss of corals as a local source of broodstock, and the increasing levels of macroalgae that are likely to further suppress coral recruitment.

#### *5.2.2.2 Johnstone Russell–Mulgrave sub-region*

The Coral Index remained 'moderate' and unchanged from that recorded in 2024.The decline from 'good' in 2021 to 'moderate' in 2024 reflected reduced scores for each indicator. In 2025 the Coral cover score remained 'good' with slight improvements at Fitzroy West and Franklands East since 2024, offset by a lack of change or minor losses elsewhere. In contrast, Coral cover scores in 2024 declined at most reefs, largely due to predation by crown-of-thorns starfish, especially at Frankland East, but also the combined impacts of coral bleaching, storm damage, and likely exposure to low salinity floodwaters.

In the absence of acute climatic pressures over the last year the overall lack of improvement in coral cover was largely attributed to the ongoing presence of crown-of thorns starfish, with feeding scars and starfish observed at most sites in 2025. That subregional coral cover has remained 'good' for more than a decade despite the ongoing presence of crown-of-thorns starfish, is testament to the success of the Reef Authority's Crown-of-thorns Starfish Control Program in maintaining hard coral cover. However, the ongoing outbreak densities and small size classes demonstrate the ongoing need for control of crown-of-thorns populations.

Also contributing to the maintenance of coral cover, has been that the rate that coral cover increased during the periods intervening acute climatic pressures or elevated populations of crown-of-thorns starfish. The Cover change score has remained 'good' for much of the last 20 years, with some declines to 'moderate' in the aftermath of cyclone Yasi and periods of heavy rainfall between 2012 and 2015 and again since 2023. Overall, this result demonstrates the ongoing capacity for coral cover recovery in this sub-region.

In contrast to the Coral cover and Cover change scores, which reflect ongoing potential for increases in cover via colony growth, the Macroalgae and Juvenile coral indicators remain 'poor'. In this sub-region, macroalgal communities are dominated by a range of relatively small red algae species that form dense mats both on dead coral colonies and rubble, and in the spaces between living coral branches. An important observation is that few juvenile hard corals are present where these algae occur. The Macroalgae indicator score marginally improved, but this was due mostly to improvements at Fitzroy Island where the cover of macroalgae is relatively low. The score was insensitive to changes at Franklands West and High East because scores in 2024 were already at their minimum value of zero. Consequently, despite increased prevalence of macroalgae in 2025, no change in score occurred.

#### *5.2.2.3 Herbert–Tully sub-region*

The Herbert–Tully sub-region score has continued to decline from a high point in 2020. The most recent decline captures the severe impact to coral communities at 2 m depths at Dunk Island and both 2 m and 5 m depths at Bedarra (see section 5.1.1). Until 2024, this decline was driven mainly by moderating Cover change and Juvenile coral scores, although these indicators remained 'moderate' and 'good', respectively.

The exposure of reefs to low salinity flood plumes in 2025 influenced the Coral Index scores in several ways. At reefs exposed to lethally low salinity, coral mortality reduced the Coral cover indicator to 'poor' for the first time since 2018, when coral communities were still recovering from the severe impacts of cyclone Yasi. At Dunk Island 2 m sites, the Composition score declined because *Acropora*–a group sensitive to poor water quality–was disproportionately affected by low salinity. Declines in the Juvenile indicator were driven by increases in the area of algae-covered substrate following reductions in coral cover, as well as a reduction in juvenile coral abundance. Together, these changes altered both the numerator and denominator of the ratio underlying the Juvenile score. However, the numbers of juvenile coral observed in 2025 declined at all reefs, not just those at which floodwaters impacted coral cover.

Although the Juvenile score has declined substantially over the last 6 years, it remains 'good'. The decline primarily mirrors the passing of strong cohorts of *Turbinaria* through the juvenile size class into adult sizes, and we do not consider this cause for undue concern. However, current juvenile communities at most reefs include a relatively high proportion of the family Merulinidae, a group that is collectively slow growing, and so is likely to remain in the juvenile size class for longer than fast growing taxa such as *Acropora,* adding a degree of positive bias to the score for this indicator.

Contrasting the declines in other indicators, the Macroalgae indicator score improved in 2025. This is likely a temporally response as macroalgae were also sensitive to low salinity floodwaters, and experience from other regions shows that macroalgae can rapidly recolonise after losing cover during acute disturbance events (see section 5.3.4).

Overall, there is a stark contrast between Barnards, Dunk North and Dunk South 2 m, where coral communities have shown a clear ability to recover during disturbance free periods, and Dunk South 5 m and Bedarra, where little capacity for recovery has been evident and high turbidity appears to be limiting coral performance.

### *5.2.3 Burdekin Region*

The Coral Index score remains 'moderate' but has continued to decline from a peak reached in 2020. However, within this decline the Coral cover indicator score in 2023 reached its highest level since surveys began in 2005. With coral cover continuing to increase through to 2024 at some reefs; for example hard coral cover at the 5 m depth of Pandora Reef reached 27%, the highest since MMP surveys began in 2005, providing the clearest evidence of that coral communities were recovering following large reductions in coral cover in 2001 (Done *et al.* 2007). Unfortunately, since 2023 reefs in the region were variously impacted by: heat stress and cyclone Kirrily in 2024, then exposure to low salinity floodwaters and a localised storm in 2025. In combination these disturbances caused coral cover losses at almost all reefs in the region, reducing scores for the Coral cover indicator, and due to the higher susceptibility, also the Composition indicator.

In contrast, Macroalgae indicator scores have improved since 2024 due to reductions of macroalgae cover by the same pressures that reduced coral cover. It is likely that this improvement will be short lived. Previous MMP results have revealed that macroalgae rapidly recolonises after being removed during acute disturbance events, as occurred in this region in the 2 years following cyclone Yasi. In 2024, there was a clear demarcation between the 'very good', scores at Palms East, Palms West, and 5 m sites at Lady Elliot, and the 'moderate' (Pandora 5 m) or 'very poor' scores elsewhere. Palms East and Palms West share regionally low levels of light attenuation and low Chl *a* concentration that may limit the proliferation of macroalgae. Conversely, the lack of macroalgae at the 5 m depth of Lady Elliot may be explained by the highly turbid water found there, which limits light availability at this site.

The longer-term decline in the Coral Index scores since 2020 reflects declines in the Composition, Coral cover, Cover change and Juvenile indicator scores, although changes over this period have been variable among reefs. It is the generally 'poor' scores for the Macroalgae and Juvenile indicators that are of concern, as they indicate a potential coral recruitment bottleneck for the resilience of reefs in this region.

The densities of juvenile corals have always been variable among reefs and depths, but the consistent decline in the Burdekin region since 2020 raises the potential for thermal stress to have impacted early life-history phases of corals, culminating in reduced recruitment and survivorship of juvenile corals. Studies by Ward *et al.* (2002) and Johnston *et al.* (2020) suggest thermal stress can lead to reduced reproduction in the subsequent spawning season. However, monitoring of coral settlement during the early years of the MMP (Davidson *et al*. 2019) indicated sporadic but generally low supply of larvae to this region. Preliminary hydrodynamic modelling (Luick *et al*. 2007, Connie 2.0[2]) and differences in population genetics of corals (Mackenzie *et al*. 2004) both indicate limited connectivity between Halifax Bay and reefs further offshore. We cannot tease apart the relative contributions of limited larval supply and coral fecundity over likely interactions with macroalgae (Viera 2020, Doropoulos *et al*. 2022) in explaining the recent low densities of juvenile corals.

The regional condition of reefs reached a low point following the impact of cyclone Yasi and associated high discharge from the catchment in 2011. A period of recovery ensued between 2013 and 2020 in which the Coral Index improved due to increases at both 2 m and 5 m depths for Coral cover, Macroalgae and Composition and increases at 5 m depths for Juvenile coral and Cover change. While coral cover increased at most reefs, it was the rapid increase in *Acropora* at Palms East that disproportionately contributed to improving Coral cover scores. Most other reefs had persistently low cover of fast-growing *Acropora*, resulting in slower increases in coral cover. The cover of *Acropora* also increased rapidly at Havannah 2 m and this was central to hard coral cover increasing from 15% in 2011 to 53% by 2015 at that reef. In the decade since, coral bleaching and high levels of disease reduced the cover of *Acropora* at Havannah 2 m from 44% in 2015 to a low of 13% in 2021. It appears several of the branching *Acropora* species that contributed to the very rapid recovery of coral cover at Havannah 2 m were particularly vulnerable to either thermal stress, high nutrient levels, or a combination of the 2, as predicted by Wooldridge (2020). In 2025 the cover of *Acropora* was around 21% but *A. pulchra,* a species common prior to 2017, is no longer present on the transects (*pers. obs*. Author).

### *5.2.4 Mackay–Whitsunday Region*

The Coral Index score improved to 'moderate' due to accelerating recovery of coral communities following the severe impacts of cyclone Debbie. Improvement in the Coral Index since 2024 reflects ongoing improvement in all 5 indicators; however, scores remain highly variable among reefs, reflecting clear differences in both the severity of impacts dealt by cyclone Debbie, and recovery trajectories.

Increases in macroalgae cover following disturbances is not uncommon, as algae quickly establishes on substrate made available following the loss of coral (McManus & Polsenberg 2004, Ceccarelli *et al.* 2020). Prior to cyclone Debbie, persistently high cover of macroalgae was only present at Seaforth and at 2 m depths at Pine. In 2025, 'very poor' scores for Macroalgae persisted at these reefs but was also recorded at Double Cone and the 2 m depths at Daydream and Dent. At most 5 m depths these scores reflect relatively low cover of macroalgae as threshold levels for scoring the Macroalgae indicator are low at 5 m depths that share silty substrates and turbid settings. However, at the 2 m depths most of these reefs have developed persistently high cover of macroalgae since cyclone Debbie that are very likely to have limited coral recruitment. Among those reefs with relatively high macroalgae cover, the presence of the family Sargassaceae and the genus *Lobophora* is worth noting as once established, these species often persist for long periods and have the potential to constrain coral recovery (see, Smith *et al*. 2023, Burgo *et al*. 2025) and trap benthic communities in a macroalgal dominated state (Mumby *et al.* 2013, Johns *et al*. 2018). Encouragingly, the cover of macroalgae declined at each of these locations between 2024 and 2025; however, as levels remain above reef specific thresholds, metric scores remain 0 and will not become sensitive to improvement until these thresholds are surpassed.

Both the Coral cover and Cover change scores remain 'poor'. For the Coral cover indicator low scores partly reflect the magnitude of losses caused by cyclone Debbie. Only at Shute Harbour 5 m has coral cover recovered to greater than 40.5%, the lower threshold for a 'moderate' score.

In the absence of acute disturbances since the minor bleaching impacts of 2020, the slow recovery of hard coral cover, indicated by the 'poor' Cover Change scores, is contributing to the current Coral Cover score. The Cover change score averages change in hard coral cover over a four-year period, and in 2024 it was only at Hayman, Daydream, and at the 5 m depth at Hook that hard coral cover had increased in line with modelled expectations. Encouragingly, added to this list in 2025 were the 2 m depth at Hook, the 5 m depth at Dent, and both depths at Double Cone Island, all showing the most convincing increase in coral cover following very little recovery in recent years.

It should be noted that prior to cyclone Debbie, a primary limitation to Coral Index scores was the persistently 'poor' score for the Cover change indicator. Conditions at the MMP monitoring sites in this region are generally characterised by high turbidity and high rates of sedimentation. In combination, these conditions have imposed strong selective pressures on corals. This is clearly illustrated by the marked differences in coral community composition between 2 m and 5 m depths at most reefs, with a shift from *Acropora* dominated communities at 2 m to a more mixed community of taxa tolerant of the highly turbid conditions at 5 m. These turbidity-tolerant corals tend to be slow growing. As the Cover change indicator is calibrated to account for this slower growth of non-*Acropora* species, the consistently low scores observed over the duration of the MMP indicate a particularly limited capacity for rapid recovery of coral cover, especially at the 5 m depths.

Water quality monitoring demonstrates the severe impact of cyclones on the water quality within the region, with declines in the long-term index following cyclone Ului, and both long and short-term indices declined to 'poor' condition following cyclone Debbie (Gruber *et al.* 2026). While both indices improved within the 'moderate' range through to 2024 during the, albeit slow, recovery of coral communities, there was a slight downturn in 2025, possibly reflecting the influence of above median discharges from the region's catchments over the 2024-25 wet season (Gruber *et al.* 2026).

### *5.2.5 Fitzroy Region*

Surveys in 2025 reveal the full impact of marine heat wave conditions in early 2024, with the coral communities declining to 'very poor' as the Coral Index score reached its lowest point since 2014. In May 2024, the severe impact of a marine heat wave over the preceding months was apparent with hard coral cover across the region reduced by more than a third relative to the 2023 levels. However, during the May 2024 surveys a high proportion of corals were still bleached and ongoing mortality occurring. By 2025 the full impact of the 2024 marine heat wave was revealed with mean hard coral cover across the region reduced to 15.6%, less than half of the 39.6% observed in 2023.

Contributing most to the decline in the Coral Index score since 2023 were declines in the Coral cover, Composition, and Juvenile coral scores. Coral communities at most reefs in this region are dominated by the genus *Acropora* and, in line with previous studies (e.g., Marshall & Baird 2000), this group was disproportionately impacted by the bleaching event. As discussed in section 5.2.8, the Composition indicator is particularly sensitive to changes in the relative cover of *Acropora*, and so it is not surprising that declines in the Composition indicator tracked the impact to corals more generally.

The decline in this indicator reflects both a 30% reduction in the number of juveniles compared to 2023, and the increase in cover of algae where it has colonised the areas of recently killed corals. The reduction in juvenile corals likely results from a combination of the loss of individuals susceptible to the heat stress that caused reductions in coral cover, diminished larval supply caused by reduced local broodstock, and potentially reduced fecundity of the surviving corals, following bleaching (Briggs *et al.* 2024).

Most reefs were impacted by the 2024 marine heat wave, with consequences playing out in both 2024 and 2025 surveys, the current score for the Cover change indicator largely reflects the rate of change in hard coral cover that occurred between 2021–2022 and 2022–2023. The only reef for which changes between 2024–2025 are informative was Pelican where, although coral cover remains very low at 2 m depth, the rate of recovery in recent years has exceeded modelled expectations. The current 'poor' score for this indicator largely captures underperformance of coral cover recovery prior to the 2024 bleaching event at North Keppel, Middle and Keppels South, but not Barren.

Scores for the Macroalgae indicator remain 'very poor'. The cover of macroalgae on reefs in this region remains persistently high, to the point that almost all reefs scored zero for this indicator in both 2023 and 2025. The persistently high levels of macroalgae on many reefs in this region make this indicator insensitive to further increases. In 2025 both macroalgae cover and the proportion of macroalgae in the total cover of algae increased.

Over the twenty years of monitoring by the MMP coral communities have been impacted by multiple disturbances, with regional hard coral cover never regaining the level of 48% observed in 2005. In early 2006, high water temperatures caused severe coral bleaching and loss of coral cover in the *Acropora* dominated communities at Barren, North Keppel, Middle, and Keppels South. Prior to the commencement of the MMP, Queensland Parks and Wildlife Service monitoring of reefs in Keppel Bay from 1993 to 2003 recorded substantial loss, and subsequent recovery, of coral cover following thermal bleaching events in 1998 and 2002 (Sweatman *et al.* 2007). Initial MMP surveys in 2005 documented ‘good’ to ‘very good’ hard coral cover on all the *Acropora*-dominated reefs, confirming the potential for recovery at these reefs when not subjected to additional pressures.

Between 2008 and 2015 physical damage caused by waves associated with cyclones Oswald and Marcia, along with unnamed storms, reduced coral cover at some reefs. During this period, flooding of the Fitzroy River impacted the coral communities in 2 primary ways. Corals in shallow waters, particularly those to the south of Great Keppel Island, were exposed to low salinity plumes that killed the corals (Jones & Berkelmans 2014), a phenomenon previously observed by van Woesik (1991). In addition, the negative relationship between the rate of change in Coral Index scores and discharge from the Fitzroy River demonstrates the wider impact of major flood events on coral community condition within Keppel Bay. Of note were elevated levels of disease following major flood events in 2008, 2010 and 2011, supporting hypotheses that either reduced salinity (Haapkylä *et al.* 2011) or increased nutrient enrichment (Vega Thurber *et al.* 2013) were sufficiently stressful to facilitate coral disease.

Since 2014, discharge from the Fitzroy River has been mostly at, or below, median levels with substantively greater than median flows occurring only in 2017 and 2022. Also, there have been no severe weather events causing damaging waves since 2015. Under these conditions some recovery of coral cover occurred despite marine heat wave conditions in early 2020 that caused minor losses in coral cover at some reefs. However, the rate of recovery has been slow with the regional Cover change indicator only achieving a ‘moderate’ score in 2021.

In addition to a loss of coral cover in 2006, the cover of macroalgae across the region increased dramatically (Diaz-Pulido *et al.* 2009, Ceccarelli *et al.* 2020). Although Diaz-Pulido *et al.* (2009) reported that this rapid increase in macroalgae cover was short lived, the MMP time-series demonstrates macroalgae have persisted at most reefs. Since 2006, the proportion of macroalgae cover within most coral communities has resulted in persistently ‘very poor’ Macroalgae scores. It was only in 2011 that the level of macroalgae declined sufficiently to lift the regional Macroalgae score into the ‘moderate’ range. In part, this occurred as macroalgae were also killed by exposure to low salinity floodwaters at some reefs. Most concerning is Middle Island, where, when first visited in 2005, *Acropora* cover was 70% and there was almost no macroalgae. The current macroalgae cover at Middle Island includes a high proportion of large brown algae of the family Sargassaceae and the genus *Lobophora.* The persistence of these macroalgae at Middle Island, where macroalgae cover reached 50% at 2m and 45% at 5 m depths in 2025, has almost certainly limited the recovery of coral cover. The timeseries of coral and macroalgae cover at Middle Island, in particular, support work that demonstrates high macroalgal cover can lead to positive feedback that reinforce macroalgae abundance while constraining coral recovery (Mumby *et al.* 2013, Clements *et al.* 2018, Johns *et al*. 2018).

One of the feedback mechanisms for locking reefs into a macroalgal dominated state is the impact of macroalgae on coral recruitment processes (Box & Mumby 2007, Birrell *et al.* 2008a, b, Forster *et al.* 2008, Johns *et al.* 2018). Although the Juvenile score had improved by 2018, it peaked in the ‘poor’ range and in 2025 is ‘very poor’. Adding to the limitations on coral recruitment imposed by high macroalgal cover, is the potential for limited larval supply. Following the loss of corals in 2011 there was a substantial decline in the settlement of coral larvae, especially at Pelican where the cover of potential brood-stock was effectively eradicated (Davidson *et al*. 2019). A final point warranting consideration is that much of the algae-covered substrate occurs on the basal sections of live staghorn *Acropora*, or the remnants of these colonies. Our observation across inshore reefs indicate that corals rarely recruit to these substrates (*pers. obs*. Author)

Considering the current reduced coral cover, depauperate juvenile coral density, and persistent high macroalgae cover, the capacity for rapid recovery of the coral communities within the Fitzroy region appears low.

## 5.3 Indicators

### *5.3.1 Coral cover*

For corals to persist in a location they need to be able to survive acute impacts but also maintain a competitive ability under the chronic pressures imposed by ambient conditions. The Coral cover indicator provides a clear assessment of the current state of the coral community. Obvious declines in the indicator identify the impact of acute pressures, while subsequent increases track the recovery of coral communities. In 2025, the overall Coral cover score declined to the lowest value observed over the 20 years of the MMP, marginally below 2018-2019 levels, demonstrating the imbalance between the impact of acute and chronic pressures and the ability of coral communities to recover.

The largest decline occurred in the Herbert–Tully subregion where exposure to low salinity flood plumes resulted in severe losses. Salinity logger data from the MMP water quality monitoring program at Dunk Island confirmed salinity was reduced to levels close to thresholds estimated by Berkelmans *et al.* (2012) as being lethal to *Acropora* corals. Given this logger is located on the north side of Dunk Island and somewhat sheltered from the flood plume, it is reasonable to assume lower salinities were experienced at the sites more directly exposed to the flood-plumes and where higher loss of coral was observed. Some reefs in the Burdekin region were similarly impacted by exposure to low salinities but also wave damage associated with the same monsoonal system that delivered the flooding rains. In the Fitzroy region ongoing loses compounded those observed in 2024, as the full extent of the impact of severe heat stress in early 2024 were realised.

In other sub-regions of the Wet Tropics, there was a very minor increase in coral cover in the Barron-Daintree, in contrast to a minor decline in Johnstone Russell–Mulgrave sub-region attributed to the ongoing presence of crown-of-thorns starfish. Despite the presence of outbreak densities of crown-of-thorns starfish on Johnstone Russell–Mulgrave reefs for more than a decade, the Coral cover score remains ‘good’, likely a testament to the success of the Authority’s crown-of-thorns control program.

Mackay–Whitsunday was the only region to see clear improvement in Coral cover scores in 2025 as coral communities continue to recover from the impact of cyclone Debbie.

In 2025, Coral cover scores at both 2 m and 5 m depths declined with increasing concentrations of total suspended solids and Chl *a* derived from satellite imagery. Influential in the relationship between Coral cover scores and satellite derived water quality estimates at 2 m depths have been the substantial losses of coral cover due to exposure to low salinity flood-plumes in either 2024 or 2025. These losses occurred at 6 of the 8 reefs with light attenuation coefficients greater than 0.13 $m^{-1}$ and 6 of the 9 reefs with Chl *a* exceeding 0.39 $\mu gL^{-1}$. The other 2 reefs in this set were Magnetic and Pelican where coral cover has been persistently low. At 5 m depths, although corals can be exposed to lethally low salinity waters under extreme circumstances, as occurred at Snapper Island South in 2024, the buoyancy of freshwater will generally ensure reduced exposure compared to shallower sites. As such, the reduced Coral cover scores observed at 5 m depth are likely to reflect a higher contribution of chronic pressures associated with poor water quality. At the subset of reefs for which *in situ* measurements of WQ are available, coral cover scores at 2 m depth also declined with increasing concentrations of Chl *a* and Turbidity derived from water quality loggers. At these reefs ‘good’ scores at the 3 water quality logger sites in the Johnstone Russell–Mulgrave subregion are influential as they far exceed the ‘poor’ or ‘very poor’ scores elsewhere and were within the lowest 4 (Chl *a*) or 6 (Turbidity) concentrations of water quality variables. Again, that these reefs have been spared major disturbances since cyclone Yasi in 2017 likely influenced this relationship.

It is not surprising that recent impacts can influence relationships between Coral cover scores and the more subtle pressures imposed by ambient water quality conditions. There is ample evidence from the data presented in this report, along with other studies (e.g., Sweatman *et al.* 2007, Browne *et al.* 2010, Morgan *et al.* 2016) that reefs in highly turbid and/or nutrient rich settings can support very high cover of coral species tolerant to those conditions. The emerging picture over the period of the MMP is that the tendency for lower coral cover on reefs with poor water quality reflects not only the additional pressure imposed by exposure to low salinity flood-plumes but also the slow, or lack of, recovery of coral communities following acute disturbance events on these reefs compared to those in cleaner waters.

### *5.3.2 Rate of change in coral cover*

The Cover change indicator assesses the rate of change in coral cover, predominantly as a measure of growth, during years free from acute disturbances. An adequate rate of coral cover increase is essential to ensure the long-term balance between cover lost to disturbances and that regained under ambient conditions. Within regions, the Cover change indicator scores are often highly variable. Such variability is likely due to communities at individual reefs being differentially exposed to pressures in both space and time, as well as due to sampling error. The scores for this indicator are averaged over a four-year period, intended to allow averaging over potential sampling error. Unfortunately, under the previous biennial sampling design or when multiple disturbances occur over sequential years, the scores over a four-year period may be derived from a single observation of cover change, or, when no valid estimates are available, carried forward from prior observations. It was partly to account for this issue that the program adopted a contingent sampling design to ensure visitation of reefs following disturbances, and more recently a return to annual sampling of all reefs to improve the data available from which to estimate scores for this indicator.

The issue of sampling error is most relevant where coral cover is very low and communities are predominantly comprised of slow growing species. In these situations, expected rates of increase are low relative to the precision of the sampling. In general, the Mackay–Whitsunday coral communities fall into this category with many currently having very low coral cover and, at 5 m depth in particular, communities with low representation of fast-growing Acroporidae. That despite this limitation, scores remaining poor in the Mackay–Whitsunday region is of concern, as it highlights the ongoing underperformance of these already slow growing corals since the severe impacts caused by cyclone Debbie in 2017.

A further issue arose in 2025 at Snapper South where all coral was killed in 2024. In this case the change in cover between 2024 and 2025 is impossible to estimate as the underlying model requires a non-zero level of coral cover from which to predict an increase. Until any coral returns to these sites the inter-annual change contributing to the four-year running mean of changes assessed by the Cover change indicator has been set to zero.

Over the period of the MMP, temporal trends in the Cover change scores can be generalised as having declined to low points between 2012 and 2014 followed by improvement through to 2017-2019 before stabilising or beginning to decline. The initial general decline in the Cover change indicator coincided with a period of high river discharge delivering high loads of sediments and nutrients to the Reef (Joo *et al.* 2012, Turner *et al.* 2012, 2013, Wallace *et al.* 2014, 2015). In each region, we noted peaks in coral disease over this period that corresponded to major flooding in the adjacent catchments. As discharge from local catchments returned to median levels or below, the Cover change indicator improved, suggesting a link between coral community recovery and catchment inputs and at least a partial release from chronic pressures related to catchment loads. Environmental conditions associated with the increased loads of sediments and nutrients delivered by these floods appear to have been sufficiently stressful to limit the recovery of coral cover and/or induce disease in susceptible species. This conclusion is consistent with previous observations linking nutrients and organic matter availability to higher incidence and severity of coral disease (Bruno *et al.* 2003, Haapkylä *et al*. 2011, Weber *et al.* 2012, Vega Thurber *et al.* 2013).

A notable exception to the above generalisation occurred in the Mackay–Whitsunday region where Cover change scores were consistently low prior to declining further following cyclone Debbie in 2017. The time-series of Cover change scores in the Mackay–Whitsunday region suggest ambient environmental conditions following cyclone Debbie, when the long-term water quality index declined to ‘poor’ (Moran *et al.* 2025), was suppressing coral growth for several years. Both the water quality index and Cover change scores have been improving over the last few years. In contrast, Cover change indicator scores improved between 2008 and 2011 in the Herbert–Tully sub-region when coral cover was rapidly recovering from the impacts of cyclone Larry, despite declining water quality over this period (Moran *et al.* 2025).

Complicating the use of this indictor is the subjectivity introduced as to when to categorise an acute pressure when levels of exposure are relatively low. As the indicator is only estimated for observations when no acute disturbance occurred, the designation, or not, of a disturbance can potentially bias the score for the Cover change indicator. For example, in the Johnstone Russell–Mulgrave sub-region, although remaining 'moderate' in 2025, the Cover change score has declined from a high point in 2017. Although crown-of-thorns starfish have been active across the region, the attribution of an acute disturbance due to crown-of-thorns starfish only occurred when hard coral cover declined. It is likely that feeding by these starfish at other times will have caused some loss of coral cover and lowered the Cover change score in recent years.

Current scores for the Cover change indicator aggregate changes that have occurred since 2021, meaning any low-level or protracted impacts of the 2022 and 2024 marine heat waves may have contributed to the declining Cover change scores in the Herbert–Tully sub-region, and Burdekin and Fitzroy regions. There is good evidence that high temperatures can impact coral growth. Following the 1998 bleaching event on the Reef there was a significant reduction in linear extension (~ 40%, D' Olivo 2013) and calcification rates (13%–18%, Cantin & Lough 2014) for *Porites* colonies, with recovery to pre-bleaching rates taking 2–4 years. Slower coral growth may also occur due to exposure to temperatures below those that would cause coral bleaching (Cantin *et al.* 2010, Anderson *et al.* 2018). This is perhaps not surprising given that studies on coral thermal optimum performance show that at least some species of corals perform best at, or slightly below, their local average temperature, with performance curves declining once the peak temperature is reached (Jokiel & Coles 1977, Jurriaans *et al.* 2021). Compounding any reduction in growth is that rates of mortality may be increased following exposure to thermal stress due to links between coral disease and elevated summer water temperatures (Selig *et al.* 2006, Heron *et al.* 2010, Ruiz-Moreno *et al*. 2012, Howells *et al.* 2020) that likely lead to subsequent mortality (Brodnicke *et al.* 2019). Indeed, within our dataset, losses in both the initial and subsequent observation following bleaching events have been categorised as acute bleaching impacts, especially where a high level of bleaching was evident during the initial post event surveys.

In 2025 the Cover change scores showed little relationship to water quality estimates. The only relationship observed was at 2 m depth for a subset of reefs at which water quality loggers are located, where scores declined with increasing turbidity. Influential were maximum scores of 1 at High West and Fitzroy West. Hard coral cover has been recovering well at Fitzroy West, with growth of the fast-growing *Acropora* driving the hard coral increase during years free from an acute pressure. In contrast, the coral community at High West is dominated by the slower growing Porites, meaning a lower rate of increase is required for comparable scores.

### *5.3.3 Community composition*

It is well documented that compositional differences in coral communities on the Reef occur along environmental gradients at a range of scales (Done 1982, van Woesik *et al.* 1999, Fabricius *et al.* 2005, Browne *et al*. 2010, De'ath & Fabricius 2010, Uthicke *et al*. 2010). The relationships between disease and altered environmental conditions, as discussed above, demonstrate the dynamic nature of coral community selection occurring on inshore reefs. Sensitive species may gain a foothold during relatively benign conditions only to be removed during periods when environmental conditions move beyond their tolerance.

Although the Composition scores do not vary along water quality gradients, coral community composition does, and this relationship is stronger at 5 m depths. Importantly, the measure of community composition reported here compares a single dimensional summary of community composition, derived from the distribution of each coral genus along water quality gradients that was observed in the early years of the MMP, and the relative cover of those genera in subsequent observations. Importantly, fast-growing *Acropora* score positively on this scale compared to the slower growing species of most other genera.

In 2025, the Composition indicator score declined in the Herbert–Tully sub-region of the Wet Tropics and the Burdekin and Fitzroy regions, where declines were largely proportionate to declines in Coral cover scores. Conversely, both Coral cover and Composition scores showed clear gains in the Mackay–Whitsunday region and a more modest improvement in the Johnstone Russell–Mulgrave sub-region. Scores for this indicator predominantly track the relative proportion of the genus *Acropora* relative to baseline observations at the monitored reefs (Thompson *et al*. 2022). In addition to being sensitive to poor water quality, *Acropora* are also susceptible to cyclones (Fabricius *et al*. 2008) and thermal bleaching (Marshall & Baird 2000) and are a preferred prey for the crown-of-thorns starfish (Pratchett 2007). As such, changes in the Composition indicator do not necessarily imply poor water quality as a causative agent. However, as a relatively fast-growing group, the maintenance of *Acropora* within the coral communities is essential for rapid recovery of coral cover following disturbances.

In the Fitzroy region, most reefs were dominated by branching *Acropora* in the early years of the MMP. While remaining 'poor' between 2010 and 2023, the Composition scores have improved, demonstrating the gradual recovery of this group. The 'very poor' condition in 2025 highlights the disproportionate loss of *Acropora* caused by the marine heat wave and resultant bleaching in early 2024.

Branching *Acropora* were one group identified by Roff *et al.* (2013) as showing reductions in contemporary communities, with reduced representation since the mid-20$^{th}$ century potentially linked to increased run-off from the adjacent catchments. While recovery of this group has been observed on many reefs, they remain sensitive to recent pressures and do not necessarily persist. For example, branching *Acropora* drove a rapid recovery of coral cover at Havannah Island between 2011 and 2015 before succumbing to disease and then coral bleaching in 2020 (AIMS Reef dashboard). While the Composition score in the Burdekin remains 'moderate' this result needs to be considered along with the generally low representation of *Acropora* at many reefs in the early years of the MMP that serve as the reference point for this indicator, compared to the higher representation of this genus historically (Done *et al.* 2007, Sweatman *et al*. 2007, Roff *et al*. 2013)

As this indicator tends to reiterate changes in coral cover due to its responsiveness to fluctuations in the cover of *Acropora,* it is partially redundant within the Coral Index. As the indicator is based on a constrained redundancy analysis, it is only sensitive to changes in the taxa that respond strongly to the univariate water quality gradient imposed on that analysis, meaning that changes in relative abundance of other taxa may go unnoticed. It is also apparent that the use of a three-level categorical system can result in large changes in score with very little actual change in community composition when communities are near categorical boundaries. The University of Queensland and AIMS have developed an indicator of community change that offers the ability to identify a greater range of changes in coral community composition (Gonzalez-Rivero *et al.* 2023a, b). This, however, does not currently apply any 'good' verses 'bad' interpretation of detected changes, and further consideration as to how this approach can be incorporated in the Coral Index is required.

### *5.3.4 Macroalgae*

For the thirteen reefs at which water quality is monitored, the 2025 scores for the Macroalgae indicator at 2 m depth declined with increasing concentrations of particulate nitrogen and phosphorus. However, the Macroalgae score did not show a relationship to measured water quality at 5 m depths, or the satellite derived environmental gradients available for all reefs. The limited relationship between Macroalgae scores and environmental gradients is influenced by the underlying metric for this indicator. The Coral Index has been designed to be responsive to change in environmental pressures with reef-level scores for each indicator having the potential to either improve or decline. This desire for a responsive index required setting location-specific thresholds for scoring both the Macroalgae and Composition indicators as water quality pressures unequivocally influence their underlying values. This setting of location-specific thresholds means that indicator scores must be considered in relative terms of improvement or decline as the baseline condition is likely to reflect communities that have been selected for by an already altered environment (van Woesik *et al.* 1999, Roff *et al.* 2013).

Relating the data underpinning the Macroalgae indicator to reef-level water quality demonstrates there is higher overall cover of macroalgae at 2 m depth on reefs exposed to relatively high concentrations of Chl *a* and low levels of light attenuation. At the 2 m depth on the subset of reefs where water quality is directly measured there is an increase in both cover and relative proportion of macroalgae within the algal community with increasing concentrations of particulate nitrogen and phosphorus. These results are consistent with findings that coral reef macroalgae generally benefit from increased nutrient availability due to run-off (e.g., Schaffelke *et al.* 2005, Adam *et al.* 2021) and link nutrient availability to reduced coral community resilience in inshore areas of the Reef. That these relationships do not extend to 5 m depths can be explained by the attenuation of light in turbid waters likely becoming a limiting factor.

There was a substantial improvement in the macroalgae indicator score in the Burdekin region, and to a lesser degree the Herbert–Tully sub-region in 2025. Following acute flood events, it is common to see short term improvements in Macroalgae indicator scores. The Macroalgae indicator is scored on the ratio between the cover of macroalgae to the cover of all algae combined. When both macroalgae and corals are sensitive to a disturbance, the macroalgae cover declines while the total cover of algae increases as more substrate is exposed following the death of corals and macroalgae, that is initially colonised by turf algae. This situation is often short-lived with macroalgae rapidly recolonising after disturbance events. This consideration does not extend to the Mackay-Whitsunday region where an improvement in the Macroalgae score more likely reflects real improvement. Of ongoing concern for the resilience of coral communities in 2025 is that despite these improvements the Macroalgae score remains ‘poor’.

Dense canopies of macroalgae, that compete with corals, continue to dominate the benthic communities at several reefs. Over the period of the MMP, increased cover of macroalgae has been precipitated by the loss of coral cover following floods, cyclones, and bleaching events. While such disturbances affect both coral and macroalgae, the inshore environment, with its availability of nutrients and lower abundance of herbivorous fish (Cheal *et al*. 2013), enables macroalgae to recolonise much faster than corals (McManus & Polsenberg 2004, Diaz-Pulido *et al.* 2007, Diaz-Pulido *et al.* 2009, Ceccarelli *et al.* 2020).

There are several pathways by which macroalgae competition occurs; from limiting the space or light available to corals (Tanner 1995, McCook *et al.* 2001, Birrell *et al.* 2005, 2008a, b, Hauri *et al.* 2010), physically damaging corals via abrasion (Clements *et al.* 2018), chemically interfering with coral recruitment process (Foster *et al.* 2008, Evensen *et al*. 2019, Monteil *et al.* 2020, Doropoulos *et al*. 2022), promoting bacterial communities pathogenic to corals (Smith *et al.* 2006, but see Clements and Hay 2023), and providing positive feedback to maintain communities in a macroalgae-dominated state (Mumby *et al.* 2013, Clements *et al.* 2018, Johns *et al.* 2018). The persistence of high macroalgae cover (notably the brown algal species *Lobophora* and the Sargassaceae) on several reefs within each region offers strong support for the presence of such negative feedback.

The variation among reefs in the recovery of coral communities illustrates the relationship between water quality and macroalgae in supressing coral community resilience. As an example, recovery of coral cover in the Fitzroy Region following coral bleaching in 2006 was inversely related to the persistence of macroalgae. At the 3 *Acropora*-dominated reefs (Keppels South, Middle and North Keppel), macroalgae cover (predominantly *Lobophora spp.*) rapidly increased and persisted at high densities; at the same time, the rate of change in coral cover remained low or coral cover continued to decline. In contrast, at Barren, where Chl *a* concentration is lower, the *Lobophora* bloom was less pronounced, and recovery of the coral community clearly progressed. Similarly, in the Mackay–Whitsunday region, macroalgae rapidly colonised the 2 m depths at Daydream, Double Cone, and Pine following severe impacts to coral communities caused by cyclone Debbie where they continue to supress coral recovery.

Schaffelke and Klump (1998) demonstrate nutrient limited growth for a species of Sargassaceae common to inshore reefs with a clear capacity for increased growth at dissolved inorganic concentration values within the range estimated by $NO_x$ values in most regions monitored by the MMP. However, it has been long accepted that biomass and cover of coral reef macroalgae is controlled by complex interactions of both biological (top-down controls such as grazing) and environmental (bottom-up controls such as nutrient levels) factors (e.g., Littler & Littler 2007). Wismer *et al.* (2009) and Rasher *et al.* (2013) demonstrate an inverse relationship between macroalgal cover and herbivore biomass, and Cheal *et al.* (2013) links this relationship to water quality by demonstrating a decline in herbivorous fish populations with increasing turbidity. Importantly, the reduction in herbivore biomass noted by Cheal *et al.* (2013) was observed on the LTMP survey reefs included in this report. The inshore reefs in the LTMP are located toward the mid-shelf end of the strong water quality gradient in inshore waters. The higher turbidity at most reefs surveyed as part of the MMP (Table A8) suggest an even lower biomass of herbivorous fishes.

Grazing is a key process for the control of macroalgal blooms and research demonstrates the importance of the maintenance of herbivore populations to avoid a phase-shift to a macroalgae-dominated state (e.g., Hughes *et al.* 2007, Rasher *et al.* 2013). Within the Burdekin region, Hughes *et al.* (2007) demonstrated that dense macroalgal communities could be supported in the absence of grazing on a reef with generally low cover of fleshy macroalgae, partly divorcing macroalgae biomass from a direct relationship to water quality alone. In contrast, Hoey and Bellwood (2011) and Roff *et al.* (2015) demonstrate that macroalgae themselves provide positive feedback with grazing pressure reduced under macroalgae canopies. The relative influences of herbivory and nutrients on coral reef macroalgae is undoubtedly complex and likely to depend on 'the species, circumstances and life-history processes under consideration' (Diaz-Pulido & McCook 2003), but also the ratio between grazer population density and the cover of macroalgae (Mumby & Steneck 2008).

The frequency and intensity of widespread abiotic disturbances such as floods, marine heatwaves, and cyclones is expected to increase, with shorter windows of recovery opportunity for corals (Hughes *et al*. 2021, Chand *et al.* 2019, Lough *et al*. 2015). In this context, the correlation between high prevalence of macroalgae and inshore water quality implies that the continued availability of nutrients, punctuated by widespread abiotic disturbances, has the potential to shift the competitive recovery between macroalgae and coral further towards the persistent dominance of macroalgae.

### *5.3.5 Juvenile coral density*

The early life history stages of corals are sensitive to a range of water quality parameters (Fabricius 2011). Direct effects of high concentrations of suspended sediments can reduce fertilisation (Ricardo *et al.* 2016) and the accumulation of sediments on the substrate can preclude larval settlement (Ricardo *et al.* 2017). In contrast, conditions that promote macroalgae are likely to have secondary negative effects on larval settlement and survival (Tanner 1995, McCook *et al.* 2001, Birrell *et al.* 2005, 2008a, b, Johns *et al.* 2018, Doropoulos *et al*. 2022). That Juvenile coral scores in 2025 increase with increasing Turbidity and Chl *a* concentration (for the subset of reefs at which water quality loggers are deployed) almost certainly reflects the interaction of a range of additional limiting factors such as recent acute disturbance history, regional factors such as variable connectivity to brood-stock populations, and differences in composition of juvenile communities among sites. Importantly, at 2 m depth juvenile densities across the range of Turbidity and Chl *a* measured at the logger sites range from 'very poor' to 'poor', demonstrating likely recruitment limitation irrespective of site-specific water quality. Influencing the observed relationships at both depths is the very low density of juvenile corals recorded on the dead, algal colonised parts of branching *Acropora* or branching and sub-massive *Porites* colonies. This substrate makes up much of the space available to coral recruitment at Barren Island, Keppels South and Franklands West–sites with comparatively low concentrations of Chl *a* and Turbidity. High densities of juvenile corals at the 5 m depths at Dunk North, Seaforth and Magnetic– all sharing comparatively high levels of Turbidity and Chl *a*–include high representation of juveniles in the family Merulinidae and either Lobophylliidae or Dendrophylliidae (genus *Turbinaria*), taxa that rarely contribute to high coral cover, or are relatively slow growing, but are an integral component of the diversity of inshore coral communities.

Within regions the density of Juveniles is highly variable. At many reefs with persistently ‘very poor’ Macroalgae scores, Juvenile coral indicator scores were also ‘very poor’. Where this relationship is not evident, higher Juvenile coral scores result from relatively high densities of juveniles from genera such as *Turbinaria*, and the Family Merulinidae, that tend to occur in poor water quality environments (Table A8). The Merulinidae are also likely to be slower growing and so remain within the juvenile size-class for a longer time, potentially adding a positive bias to the indicator score where they are proportionally well represented among the juvenile community.

Across the MMP time-series *Turbinaria* has tended to recruit strongly to reefs following severe disturbance by cyclones. High densities of *Turbinaria* juveniles were observed on reefs in the Herbert–Tully and Burdekin (sub-)regions following cyclone Yasi in 2011, and to a lesser degree following cyclone Larry in 2006, and at Daydream Island following cyclone Debbie in 2017. Declines in juvenile densities in the Herbert–Tully and Burdekin (5 m) regions over the last few years largely reflect the transition of these strong cohorts of *Turbinaria* out of the juvenile size class as individuals have either died or grown. As this genus was not well represented in most adult coral communities prior to the disturbances, it is unclear whether this recruitment pattern is due to natural successional processes or indicates the selection for species more suited to the recent environmental conditions (Sofonia & Anthony 2008). *Turbinaria* juveniles appear tolerant of conditions that limit recruitment of other species, often being observed on loose rubble, silt laden substrate and within dense stands of macroalgae. These strong cohorts of *Turbinaria* can potentially mask patterns of recruitment in taxa necessary for rapid recovery of coral cover, such as *Acropora*. Only at Dunk North, and perhaps Lady Elliot, have the strong cohorts of *Turbinaria* progressed to be representative in coral cover estimates.

The Juvenile indicator scores the ratio between the number of juvenile corals observed and the area of transect occupied by algae. Following acute disturbances, the indicator score can decline due to either, or often both, a reduction in the number of juvenile corals because they were susceptible to the disturbance pressure, or the increase in area of transect occupied by algae, which rapidly colonises areas previously occupied by coral tissue. Recent declines in the Juvenile indicator score in the Barron–Daintree and Herbert–Tully sub-regions and Burdekin region reflect both the reduction in the number of juveniles and increase in area of algae caused by exposure to low salinity floodwaters, and in the Fitzroy region the impact of the 2024 marine heat wave. The influence of increased area of algae can reduce the Juvenile indicator score for several years as the size class of corals considered as juveniles may include at least 3 cohorts of juveniles.

In contrast to other regions, the Juvenile coral score continued to increase in the Mackay–Whitsunday region where strong recruitment at Hayman, Daydream and Hook (2 m) in recent years is contributing to the recovery of coral communities since cyclone Debbie. In contrast, the density of juvenile corals at other reefs remains low with ‘poor’ scores at both depths of Dent, Double Cone and Pine reflecting limited post cyclone recovery.

Monitoring of coral settlement during the early years of the MMP (Davidson *et al.* 2019) indicated sporadic but generally low supply of larvae to reefs in the Burdekin region and a severe reduction in settlement at Pelican Island in the Keppel region following the local loss of corals. These results suggest connectivity to broodstock may also play an important role in the early recovery of reefs. Preliminary hydrodynamic modelling (Luick *et al.* 2007, Connie 2.0[3]) and differences in population genetics of corals (Mackenzie *et al.* 2004) in the Burdekin region both indicate limited connectivity between Halifax Bay and reefs further offshore. Perhaps the most compelling evidence for low larval supply to some inshore reefs has been observed at Snapper South. At the 2 m depths at Snapper South, macroalgae cover is low but juvenile coral densities are also typically low, a situation punctuated by sporadic high recruitment observed in 2008 and again in 2023 (Figure A1) that demonstrates the suitability of the substrate to coral recruitment should larvae be available.

[3] Connie 2.0, CSIRO Connectivity Interface, CSIRO connie3, note that version 2.0 is no longer available.

### 5.4 Management response

Coral reefs, in general, are subjected to cumulative impacts of acute disturbances and environmental pressures (Bozec *et al*. 2022). In the simplest terms, successful management should promote a balance between coral losses and subsequent recovery. Identifying causes of coral loss and relationships between recovery and environmental conditions emerging from the MMP timeseries provide some salient observations that may guide management initiatives.

MMP surveys have noted elevated populations of crown-of-thorns starfish on reefs in the Johnstone Russell–Mulgrave sub-region since 2012, with the starfish observed consistently, including individuals across a range of size-class, demonstrating their ongoing recruitment to these reefs. The Crown-of-thorns Starfish Control Program has helped to mitigate the impact of crown-of-thorns starfish and limit coral loss in the Wet Tropics region where the small size and isolation of these reefs may make such controls particularly feasible. Our data cannot investigate the likely source populations for the juveniles observed but such data could potentially help to focus control efforts should the mitigation of larval supply rather than maintenance of coral cover become a priority.

Within each region, there are reefs where macroalgae cover is persistently high and coral communities fail to recover. That this occurs predominantly in areas with higher Chl *a* and TSS levels, suggests that any actions that improve water quality have the potential to enhance the resilience of coral communities in inshore areas. It must be noted, however, that corals can also attain high cover under these water quality conditions suggesting that once established density-dependent feedback mechanisms likely contribute to maintaining the high cover of macroalgae (Vieira 2020). As such, the removal of algae such as *Lobophora* and Sargassaceae in the early stages of post-disturbance succession may prove a viable and efficient action to avert long-term phase shifts at high-value sites (Ceccarelli *et al.* 2018, Smith *et al.* 2022), noting this may only be feasible at small scales. Grazing by fish and urchins is also an important natural control for macroalgae, and any pressures that are likely to reduce the abundance of grazing organisms should be mitigated.

In most Natural Resource Management regions coral communities retain the ability to recover following impacts from acute disturbances. However, the rate of this recovery is correlated to the loads of nutrients and/or sediments entering inshore waters, particularly during flood events. To maintain the balance between disturbance and recovery of the inshore Reef it is essential that management actions provide corals with optimum conditions to cope with ever-increasing global stressors of climate change and ocean acidification (Bellwood *et al.* 2004, Marshall & Johnson 2007, Carpenter *et al.* 2008, Mora 2008, Hughes *et al*. 2010, Claar *et al*. 2020).

Benthic communities in inshore areas of the Reef show clear responses to gradients in water quality, demonstrating the selective pressure imposed (van Woesik *et al.* 1999, Fabricius & De'ath 2001, Fabricius *et al.* 2005, Wismer *et al.* 2009, Uthicke *et al.* 2010, Fabricius *et al.* 2012). Changes to land management practices should, with time, lead to improved coastal and inshore water quality that in turn supports the health and resilience of the Reef (see Brodie *et al.* 2012 for a discussion of expected time lags in the ecosystem response). It is recognised, however, that the management of locally produced pressures, such as poor water quality, are secondary to the urgent need to reduce global carbon emissions to avoid irreversible loss of coral reef ecosystems (Van Oppen & Lough 2018, Great Barrier Reef Marine Park Authority 2019, Hoegh-Guldberg *et al.* 2019).

## 6 CONCLUSIONS

Results from 2024 revealed the overall condition of inshore reefs had declined to the lowest point since the MMP began in 2005. There was a slight improvement in 2025 as recovery of coral communities in the Mackay–Whitsunday region outweighed ongoing declines elsewhere. It is increasingly clear that the cumulative impacts of acute disturbances, including cyclones, crown-of-thorns starfish, thermal stress, and low salinity flood plumes (Lam *et al.* 2018, Ceccarelli *et al.* 2020, Thompson *et al.* 2020) have outweighed the ability of coral communities to recover.

The persistence of coral communities depends on the long-term balance between the frequency and severity of acute pressures and the ability of corals to recover. It is unequivocal that the unprecedented series of marine heat waves since 2017 (Henley *et al*. 2024) have played a major role in tipping this balance in favour of disturbances. Given projections for increased severity and/or frequency of pressures due to climate change and other human activities (Steffen *et al*. 2013, Halpern *et al.* 2015, Hughes *et al.* 2018), the focus on supporting recovery in a climate of increasing disturbance is ever-sharpening (Great Barrier Reef Marine Park Authority 2024, Abelson 2020, Bozec *et al.* 2025). Central to maximising recovery potential will be management actions that reduce the influence of chronic pressures, such as poor water quality, that either interact with acute events to exacerbate community declines or suppress the recovery process.

Disentangling the influence of run-off on the observed declines in coral community condition, or on the ability of communities to recover, remains difficult for several reasons. Firstly, thresholds for coral response to the cumulative pressures associated with water quality will be spatially variable because of the selection and acclimatisation of corals in response to location-specific conditions. Secondly, extrinsic variability, due to weather, along with low concentrations for many constituents of water quality, limits the ability to quantify pressures resulting from run-off at scales relevant to the communities monitored. Finally, the effects of interactions between water quality stressors and acute disturbances have only been quantified for a limited combination of pressures and a few coral species (e.g., Uthicke *et al.* 2016). In combination, these knowledge gaps limit the ability to quantify water quality thresholds appropriate for the diversity of coral communities found on inshore reefs. However, focusing on the response of the coral communities (as measured by differences in Coral Index and indicator scores) does identify both spatial and temporal patterns in the responses of coral communities to variations in water quality (Thompson *et al.* 2020).

Spatially, results from this project substantiate that macroalgal abundance is enhanced in areas exposed to chronic high nutrient availability (Fabricius *et al.* 2005). In each region, there are reefs with persistently high cover of macroalgae, and coral cover is low or very slow to recover following exposure to acute pressures. Temporally, the recovery of coral communities, assessed as rate of increase in Coral Index scores, shows a negative relationship to river discharge volume and the corresponding loads of sediments and nutrients carried therein. In combination, these results highlight the detrimental influence of poor water quality on the recovery of coral communities following inevitable exposure to acute pressures.

As the time series for the MMP lengthens, some pertinent observations relating to the balance between the impact of disturbances and the recovery of coral communities can be made:

- In the Wet Tropics, Burdekin and Fitzroy regions, coral communities have demonstrated the capacity to recover following severe loss of coral due to acute disturbances. The rate of this recovery has, however, been suppressed during periods of increased loads of sediments and/or nutrients from the adjacent catchments and, more recently, during a period of repeated exposure to high summer water temperatures. While Coral Index scores between 2016 and 2022 had variably returned to those observed at the beginning of the project, it should be noted that in 2006, when the Coral Index was first estimated, some reefs in these regions had been recently impacted by severe acute disturbances, and as such the 2006 condition may not be an appropriate aspirational reference point. The impacts of storms, crown-of-thorns starfish, flooding and high water temperatures resulting in coral bleaching have variously contributed to declines in Coral index scores in each of these regions in 2025.
- On reefs with high macroalgae cover, the recovery of coral communities has been stalled. Acute disturbance to coral communities and high nutrient concentrations are likely to have promoted the initial high cover of macroalgae. Once established, macroalgae are often highly persistent as density-dependent feedback processes bolster their competitive advantage relative to that of corals.
- Since 2017 marine heat wave conditions have impacted reefs in all regions. In 2024 marine heat wave conditions resulted in unprecedented levels of heat stress across the Reef (Henley

*et al.* 2024). In inshore areas, reefs in the Fitzroy region were the most impacted, with surveys in 2025 revealing more than half the cover of hard corals had been killed. Much of the added pressures associated with high water temperatures in recent years has occurred during periods of relatively low rainfall and cyclone activity when coral communities should be in a state of recovery. This increase in disturbance frequency and severity makes it increasingly important to mitigate chronic environmental conditions, such as poor water quality, that limit the recovery potential of coral communities.

- Crown-of-thorns starfish continue to be present on reefs in the Johnstone Russell–Mulgrave sub-region. Ongoing control of these starfish continues to limit their impact on coral community condition in this region.
- In the Mackay–Whitsunday region most reefs were severely impacted by cyclone Debbie in 2017. Recovery from this disturbance had been slow but clearly accelerated in 2025 with the Coral Index improving to 'moderate'. Leading this recovery has been increasing densities of juvenile corals in recent years, contributing to a gradual improvement in coral cover. While these improvements illustrate ongoing recovery potential within the region, not all reefs are performing well and Macroalgae, Cover change, Coral cover and Composition indicator scores remain 'poor'. This combination continues to implicate chronic environmental pressures limiting the recovery potential of coral communities in the region.
- Very intense rainfall in 2024 and 2025 substantially increased the overall proportion of coral cover lost because of exposure to low-salinity floodwaters compared to previous years. These observations serve as a reminder that the development of coral reefs in the inshore zone is ultimately bound by environmental constraints. Any future revision of the sampling design of the program should consider the appropriateness of applying indicator scores to reefs, such as Bedarra, that have limited carbonate reef development.

While the results presented here do not provide clear guidance in terms of load reductions required to improve coral community condition in the inshore Reef, they do support the premise of the Reef 2050 WQIP that the loads entering the Reef, especially during high rainfall periods, are reducing the resilience of inshore coral communities. The potential for phase shifts to algae-dominated states or further delays in the recovery of coral communities because of poor water quality, in combination with the observed high frequency of disturbances, reinforces the importance of managing local pressures to support the long-term maintenance of these communities (Abelson 2020).

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# Appendix 1: Additional Information

Table A1. Source of river discharge data used for daily discharge estimates

| (sub-)region | Rivers – Gauging station |
|---|---|
| Barron–Daintree | Broomfield-108003A, Daintree-108002A, Mossman-109001A, Barron-110001D |
| Johnstone Russell–Mulgrave | Mulgrave River-111007A, Russell River-111101D, North Johnstone-112004A, South Johnstone-112101B |
| Herbert–Tully | Tully River - 113006A, Murray River - 114001A, Herbert River – 116001E then 116001F |
| Burdekin | Bluewater Creek-117003A, Black River-117002A, Haughton River-119003A, Barratta Creek-119101A, Burdekin River-120006B, Don River-121003A, Elliot River-121002A, Euri Creek-121004A |
| Mackay–Whitsunday | O'Connell River-124001B, Andromache River-124003A, St Helens Creek-124002A, Pioneer River-125016A, Sandy Creek-126001A, Carmila Creek-126003A |
| Fitzroy | Waterpark Creek - 129001A, Fitzroy River - 130005A |

Table A2. Temperature loggers used

| Temperature Logger Model (Supplier) | Deployment period | Recording frequency (mins) | Accuracy ± $^{o}$C |
|---|---|---|---|
| ‘392’ and ‘Odyssey’ (Dataflow System) | 2005 to 2008. | 30 | 0.2 |
| ‘Sensus Ultra’ (ReefNet) | 2008 to 2017 | 10 | 0.2 |
| ‘Vemco Minilog-II-T’ (Vemco) | 2015 onward | 10 | 0.2 |
| ,‘SBE-56‘ (Sea-Bird Scientific) – note: occasional | 2018 onward | 5 | 0.002 |
| ,‘RBR‘ (RBR-Global) – note: increasingly | 2020 onward | 5 | 0.002 |

Table A3. Thresholds for the proportion of macroalgae in the algae communities.

| Reef | 2 m Depth | | 5 m Depth | | | Reef | 2 m Depth | | 5 m Depth | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Upper | Lower | Upper | Lower | | | Upper | Lower | Upper | Lower |
| Barnards | 23.0 | 4.8 | 20.8 | 1.7 | | Hook | 9.3 | 3.4 | 8.1 | 1.4 |
| Barren | 13.0 | 3.7 | 12.6 | 1.6 | | Keppels South | 23.0 | 3.9 | 24.0 | 1.7 |
| Bedarra | 23.0 | 5.3 | 15.6 | 1.9 | | Lady Elliot | 23.0 | 6.1 | 15.3 | 1.9 |
| Border | | | 8.2 | 1.4 | | Langford | | | 7.9 | 1.4 |
| Daydream | 13.5 | 3.5 | 10.4 | 1.5 | | Low Isles | | | 8.9 | 1.4 |
| Dent | 11.6 | 3.5 | 10.2 | 1.5 | | Magnetic | 23.0 | 6.4 | 19.0 | 2.0 |
| Double Cone | 8.9 | 3.4 | 7.6 | 1.4 | | Middle | 23.0 | 5.2 | 23.0 | 1.8 |
| Dunk North | 23.0 | 4.6 | 13.5 | 1.7 | | North Keppel | 23.0 | 5.1 | 22.6 | 1.8 |
| Dunk South | 23.0 | 5.3 | 15.6 | 1.9 | | Palms East | 12.2 | 3.6 | 10.5 | 1.5 |
| Fitzroy East | 11.7 | 3.5 | 10.0 | 1.5 | | Palms West | 12.8 | 3.4 | 17.5 | 1.5 |
| Fitzroy West | 12.5 | 3.3 | 13.3 | 1.5 | | Pandora North | | | 13.1 | 1.6 |
| Franklands East | 12.2 | 3.4 | 10.5 | 1.5 | | Pandora | 23.0 | 4.7 | 16.2 | 1.6 |
| Franklands West | 11.4 | 3.4 | 15.8 | 1.5 | | Pelican | 23.0 | 6.4 | 18.8 | 2.0 |
| Havannah North | | | 21.7 | 1.5 | | Pine | 18.3 | 4.4 | 11.2 | 1.6 |
| Havannah | 18.2 | 3.4 | 25.0 | 1.6 | | Seaforth | 11.8 | 3.4 | 10.2 | 1.4 |
| Hayman | | | 9.4 | 1.4 | | Shute Harbour | 17.6 | 4.2 | 11.7 | 1.6 |
| High East | 11.2 | 3.4 | 13.0 | 1.4 | | Snapper North | 18.7 | 4.4 | 11.3 | 1.6 |
| High West | 22.4 | 4.4 | 12.1 | 1.6 | | Snapper South | 23.0 | 4.4 | 13.1 | 1.6 |

Table A4. Eigenvalues for hard coral genera along constrained water quality axis. * Indicates genera with both low cover (maximum < 0.5% on any reef) and limited distribution (present on < 25% of reefs).

| Genus | 2 m | 5 m |
|---|---|---|
| *Psammocora* | -0.194 | -0.366 |
| *Turbinaria* | -0.279 | -0.307 |
| *Goniopora* | -0.320 | -0.304 |
| *Goniastrea* | -0.115 | -0.278 |
| *Pachyseris* | -0.077 | -0.235 |
| *Favites* | -0.096 | -0.230 |
| *Alveopora* | -0.076 | -0.221 |
| *Hydnophora* | -0.047 | -0.213 |
| *Cyphastrea* | -0.386 | -0.193 |
| *Galaxea* | -0.081 | -0.159 |
| *Mycedium* | -0.017 | -0.151 |
| *Favia* | -0.134 | -0.136 |
| *Pectinia* | -0.030 | -0.126 |
| *Podobacia* | -0.025 | -0.122 |
| *Plesiastrea* | -0.125 | -0.114 |
| *Echinophyllia* | -0.002 | -0.11 |
| *Moseleya* * | -0.058 | -0.091 |
| *Oxypora* | -0.008 | -0.076 |
| *Merulina* | -0.01 | -0.073 |
| *Coscinaraea* | -0.011 | -0.062 |
| *Duncanopsammia* * | | -0.042 |
| *Caulastrea* | 0.007 | -0.041 |
| *Platygyra* | 0.048 | -0.040 |
| *Herpolitha* | -0.013 | -0.034 |
| *Lobophyllia* | 0.018 | -0.034 |
| *Pavona* | -0.152 | -0.024 |
| *Astreopora* | 0.031 | -0.023 |
| *Euphyllia* | -0.012 | -0.023 |
| *Leptoseris* | -0.011 | -0.021 |
| *Palauastrea* * | 0.002 | -0.021 |
| *Polyphyllia* * | 0.000 | -0.020 |
| *Heliofungia* | 0.015 | -0.007 |
| *Catalaphyllia* * | -0.002 | -0.006 |
| *Stylocoeniella* * | 0.004 | -0.006 |
| *Pseudosiderastrea* * | -0.001 | -0.006 |
| *Gardineroseris* * | -0.004 | |
| Submassive *Porites* | -0.047 | -0.005 |
| Submassive *Acropora* | 0.043 | -0.004 |
| *Halomitra* * | | -0.002 |
| *Plerogyra* | 0.002 | -0.001 |
| *Lithophyllon** | | -0.001 |
| *Tubastrea** | 0.005 | -0.000 |

| Genus | 2 m | 5 m |
|---|---|---|
| *Scolymia* * | 0.001 | 0.000 |
| *Ctenactis* * | 0.016 | 0.001 |
| *Anacropora* * | | 0.001 |
| *Physogyra* | 0.000 | 0.001 |
| *Cynarina* * | -0.000 | 0.004 |
| *Sandalolitha** | 0.003 | 0.005 |
| *Montastrea* | 0.019 | 0.005 |
| *Fungia* | 0.013 | 0.015 |
| Encrusting *Acropora* | 0.048 | 0.015 |
| *Acanthastrea* * | -0.014 | 0.017 |
| *Symphyllia* | 0.034 | 0.018 |
| *Seriatopora* | 0.05 | 0.027 |
| *Stylophora* | 0.035 | 0.033 |
| *Oulophyllia* | 0.02 | 0.037 |
| Digitate *Acropora* | 0.034 | 0.039 |
| *Montipora* | -0.131 | 0.045 |
| *Leptastrea* * | 0.022 | 0.048 |
| *Coeloseris* | 0.052 | |
| Bottlebrush *Acropora* | 0.153 | 0.070 |
| *Pocillopora* | 0.058 | 0.074 |
| Branching *Porites* | 0.059 | 0.075 |
| *Leptoria* | 0.054 | 0.077 |
| *Porites rus* | 0.122 | 0.087 |
| *Echinopora* | 0.076 | 0.096 |
| Massive *Porites* | -0.054 | 0.122 |
| *Diploastrea* | 0.003 | 0.173 |
| Tabulate *Acropora* | 0.052 | 0.224 |
| Corymbose *Acropora* | 0.060 | 0.240 |
| Branching *Acropora* | 0.657 | 0.810 |

Table A5. Annual freshwater discharge for the major Reef Catchments. Values represented as proportional to the median (1990-2020). Flows corrected for ungauged area of catchments as per Gruber *et al.* (2026). Levels of exceedance of median flow expressed as multiples of median flow: Yellow = 1.5-1.9, Orange = 2.0-2.9, Red = 3.0 and above.

| Region | River | Median | 2003 | 2004 | 2005 | 2006 | 2007 | 2008 | 2009 | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | 2022 | 2023 | 2024 | 2025 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Wet Tropics | Daintree | 1918174 | 0.3 | 1.9 | 0.7 | 1.7 | 1 | 1.2 | 0.8 | 1.6 | 2.1 | 1.3 | 0.9 | 2.3 | 1.1 | 0.9 | 1 | 0.9 | 3 | 0.6 | 1 | 1.3 | 2.4 | 4.8* | 1.4 |
| | Mossman | 604711 | 0.7 | 1.4 | 0.9 | 1.6 | 1 | 1.1 | 0.9 | 1.4 | 1.7 | 1.3 | 1 | 1.6 | 0.7 | 1.1 | 1 | 1.3 | 2.2 | 0.7 | 1.1 | 1.3 | 1.3 | 2.9 | 1.6 |
| | Barron | 622447 | 0.2 | 2 | 0.8 | 1.6 | 0.9 | 3.4 | 1.6 | 1 | 4 | 1.6 | 0.6 | 1.3 | 0.7 | 0.3 | 0.5 | 1.6 | 2.7 | 0.6 | 1.1 | 1.1 | 2 | 5.8 | 2.2 |
| | Russell-Mulgrave | 4222711 | 0.6 | 1.3 | 0.8 | 1.2 | 1.1 | 1.1 | 1 | 1.1 | 1.7 | 1.2 | 0.8 | 1.2 | 0.7 | 0.7 | 0.7 | 1.2 | 1.3 | 0.7 | 1.1 | 1 | 1 | 1.6 | 1.2 |
| | Johnstone | 4797163 | 0.5 | 1 | 0.8 | 1.2 | 1.1 | 1 | 1.1 | 1 | 1.8 | 1.1 | 0.8 | 1.1 | 0.7 | 0.7 | 0.9 | 1.2 | 1.2 | 0.7 | 1.1 | 1 | 1.1 | 1.7 | 1.4 |
| | Tully | 3393025 | 0.5 | 1.1 | 0.7 | 1.2 | 1.3 | 1 | 1.2 | 1 | 2 | 0.9 | 0.9 | 1.2 | 0.7 | 0.8 | 0.8 | 1.1 | 1.2 | 0.6 | 1.2 | 0.9 | 1.1 | 1.6 | 1.4 |
| | Murray | 1484246 | 0.5 | 1 | 0.6 | 1.2 | 1 | 1 | 1.3 | 0.9 | 2.4 | 1.4 | 0.9 | 1.1 | 0.6 | 0.9 | 0.9 | 1.2 | 1.2 | 0.7 | 1.3 | 0.9 | 1 | 1.7 | 1.5 |
| | Herbert | 3879683 | 0.2 | 1 | 0.4 | 1.2 | 1.2 | 1 | 2.9 | 1 | 3.5 | 1.3 | 0.9 | 1.2 | 0.3 | 0.5 | 0.6 | 1.8 | 1.6 | 0.4 | 1.8 | 0.8 | 1.3 | 2.2 | 2.8 |
| Burdekin | Black | 293525 | 0.1 | 0.8 | 0.5 | 1 | 2.2 | 2.5 | 4.6 | 2.2 | 5.5 | 3.2 | 0.8 | 1.8 | 0.1 | 0.5 | 0.3 | 1.9 | 4.6 | 0.5 | 1.5 | 0.9 | 1.2 | 1.8 | 8.0 |
| | Ross | 279376 | 0.8 | 0.6 | 0.7 | 1.5 | 2 | 2.2 | 5 | 2.3 | 5.3 | 2 | 4.4 | 1.1 | 0.7 | 0.7 | 0.7 | 0.9 | 9.1 | 1 | 0.8 | 0.9 | 1.7 | 1.1 | 8.6 |
| | Haughton | 558735 | 0.4 | 0.8 | 1 | 1.1 | 2.2 | 3.3 | 4.4 | 2.1 | 4.7 | 3.2 | 1 | 1 | 0.3 | 0.5 | 0.7 | 1.4 | 5.6 | 0.6 | 1.1 | 1.3 | 2.2 | 1 | 6.4 |
| | Burdekin | 4406780 | 0.5 | 0.3 | 1 | 0.5 | 2.2 | 6.2 | 6.7 | 1.8 | 7.9 | 3.5 | 0.8 | 0.3 | 0.2 | 0.4 | 0.9 | 1.3 | 4 | 0.5 | 1.9 | 1.2 | 2.2 | 1.3 | 6.9 |
| | Don | 496485 | 0.8 | 0.8 | 1.3 | 1 | 1.9 | 3.8 | 3.1 | 1.5 | 5.4 | 2 | 1.4 | 1 | 0.7 | 0.7 | 1.9 | 0.9 | 2.3 | 1 | 1 | 0.8 | 2 | 0.8 | 3.1 |
| Mackay Whitsunday | Proserpine | 859348 | 0.2 | 0.3 | 0.8 | 0.6 | 2.1 | 2.6 | 1.8 | 2.9 | 5.7 | 2.3 | 1.3 | 0.9 | 0.2 | 0.7 | 2.2 | 0.6 | 3 | 0.7 | 0.6 | 0.5 | 2.2 | 0.7 | 2.9 |
| | O'Connell | 835478 | 0.2 | 0.3 | 0.8 | 0.6 | 2.1 | 2.6 | 1.8 | 2.9 | 5.7 | 2.3 | 1.3 | 0.9 | 0.2 | 0.7 | 2.2 | 0.6 | 3 | 0.7 | 0.6 | 0.5 | 2.2 | 0.7 | 2.9 |
| | Pioneer | 616216 | 0.2 | 0.1 | 0.4 | 0.1 | 1.6 | 2.4 | 1.6 | 2.6 | 5.9 | 2.5 | 1.9 | 1 | 0.2 | 1 | 2.3 | 0.4 | 1.9 | 0.6 | 0.4 | 0.5 | 1.2 | 1 | 2.2 |
| | Plane Creek | 1058985 | 0.5 | 0.3 | 0.7 | 0.3 | 1.4 | 2.7 | 1.3 | 2.7 | 3.9 | 2.4 | 1.8 | 0.8 | 0.4 | 0.9 | 2.4 | 0.4 | 1.2 | 1.1 | 0.6 | 0.5 | 1.4 | 0.6 | 2.0 |
| Fitzroy | Water Park Creek | 392614 | 1 | 0.2 | 0.5 | 0.3 | 0.6 | 2.3 | 1 | 2.6 | 4.4 | 1.4 | 4.7 | 2.7 | 1.9 | 1.7 | 2.5 | 1.4 | 0.7 | 1.4 | 1.7 | 2.1 | 1.5 | 2 | 1.0 |
| | Fitzroy | 2875792 | 1 | 0.5 | 0.3 | 0.3 | 0.4 | 4.7 | 0.8 | 4.5 | 14.5 | 3.1 | 3.3 | 0.6 | 1 | 1.4 | 2.4 | 0.4 | 0.5 | 1 | 0.2 | 1.6 | 1.1 | 0.7 | 1.0 |
| | Calliope | 257050 | 2.5 | 1.1 | 0.5 | 0.4 | 0.4 | 1.7 | 1 | 2.6 | 4.7 | 1.9 | 7.1 | 1.6 | 2.5 | 1 | 2.1 | 1 | 0.4 | 0.7 | 0.5 | 1 | 0.5 | 0.7 | 1.1 |

*2024 value for the Daintree estimated based on surrounding catchments due to lose of the gauging station during a flood event.

Table A6. Disturbance records for each surveyed reef. Tabulated losses of coral cover are calculated using the methods described in section (2.5.5) of this report and represent the proportion of hard coral lost compared to projected cover based on previous observations, as opposed to reduction in observed cover that does not account for expected increase in cover because of growth between surveys. * Represent cases where bleaching was the likely primary cause of loss although other factors may have contributed, ** bleaching likely however impact confounded by other severe disturbance. Bleaching events that occurred beyond the span of the available coral monitoring time-series indicated by n/a. COTS refers to population outbreaks of crown-of-thorns starfish

| (sub-)region | Reef | Bleaching 1998 | Bleaching 2017 | Bleaching 2024 | Other recorded disturbances |
|---|---|---|---|---|---|
| Barron–Daintree | Snapper North | 19% | 58% (2 m)<br>38% (5 m) | ** | Flood 1996 (20%), cyclone Rona 1999 (74%), Storm 2008 (14% at 2 m 8% at 5 m), Disease 2011 (21% at 2 m, 27% at 5 m), COTS 2012-2013 (78% at 2 m, 43% at 5 m), cyclone Ita 2014 (90% at 2 m, 50% at 5 m) – possible flood or COTS contribution, cyclone Jasper and flood 2024 (74% at 2 m, 1% at 5 m) |
| | Snapper South | | 5% (2 m)<br>1% (5 m) | | Flood 1996 (87%), Flood 2004 (32%), COTS 2013 (26% at 2 m, 17% at 5 m), cyclone Ita 2014 (18% at 2 m, 22% at 5 m), Flood 2019 (38% at 2 m, probable contribution of storm, Flood 2024 (100% at 2 m and 5 m, includes impact from cyclone Jasper) |
| | Low Isles | ** | | ** | COTS 1997-1999 (69%), Multiple disturbances (cyclone Rona, COTS) 1999-2000 (61%), Multiple disturbances (cyclone Olga and disease) 2011 (23%), COTS 2015 (38%), COTS + Bleaching 2019 (24%), Cyclone Jasper 2025 (22%) |
| Johnstone Russell–Mulgrave | Fitzroy East | na | 15% (2 m)<br>10%(5 m)* | 33% (2 m) | Disease 2010 (15% at 2 m, 5% at 5 m), Disease 2011 (60% at 2 m, 42% at 5 m), COTS 2012 (12% at 5 m), COTS 2014 (29% at 2 m, 48% at 5 m), Bleaching 2018 ongoing from 2017, COTS 2021 (35% 2 m, 12% 5 m), COTS 2025 (14% at 2 m and 5 m) |
| | Fitzroy West | na | 21% (2 m)<br>24% (5 m) | 53% (2 m)<br>** (5 m) | Disease 2011 (42% at 2 m, 17% at 5 m), COTS 2012 (13% at 5 m), COTS 2013 (32% at 2 m,36% at 5 m), COTS 2014 (5% at 2 m), cyclone Jasper 2024 (10% at 5 m) |
| | Fitzroy West LTMP | 12% | | | COTS and continued bleaching 2000 (80%), COTS 2001 (19%), COTS 2013 (9%), COTS 2015(46%), COTS 2022 (16%) |
| | Franklands East | 43% | 22% (2 m)<br>** (5 m) | | Unknown although likely COTS 2000 (68%), cyclone Larry 2006 (64% at 2 m, 50% at 5 m), Disease 2007 (35% at 2 m), cyclones Tasha and/or Yasi 2011 (61% at 2 m, 41% at 5 m), 2017 COTS (30%)–bleaching likely contributed, COTS 2020 (8% at 5 m), COTS 2021 (45% at 5 m), COTS 2024 (40% at 2 m, 47% at 5 m), COTS 2025 (9% at 2 m, 11% at 5 m) |
| | Franklands West | 44% | 17%* (2 m)<br>21% (5 m) | | Unknown although likely COTS 2000 (35%), cyclone Tasha and /or Yasi 2011 (35% at 2 m), 2017* COTS likely to have contributed, COTS 2020 (2% at 2 m), COTS 2021 (13% 2 m), COTS 2025 (18% at 2 m) |
| | High East | na | 27%* (2 m)<br>** (5 m) | | cyclone Tasha and/or Yasi 2011 (81% at 2 m, 58% at 5 m), 2016 COTS (8% 5 m), COTS 2017( 11% 5 m) – COTS likely to have contributed at 2 m, COTS 2018 (10% at 5 m), COTS 2021 (34% 2 m, 29% 5 m), COTS 2023 (18% at 5 m), Flood 2024 (60% at 2 m, includes impact from cyclone Jasper), COTS 2024 (14% at 5 m), COTS 2025 (18% at 5 m) |
| | High West | na | 18% (2 m)<br>27% (5 m) | | cyclone Larry 2006 (25% at 5 m), Flood 2009 (11% at 2 m), Flood and cyclone Tasha and/or Yasi 2011 (21% at 2 m, 34% at 5 m), COTS 2021 (26% 5 m), COTS 2023 (15% at 2 m, 42% at 5 m), COTS 2025 (4% at 2 m, 14% at 5 m) |

Table A6 (continued).

| (sub-)region | Reef | Bleaching 2017 | Bleaching 2020 | Bleaching 2024 | Other recorded disturbances |
|---|---|---|---|---|---|
| Herbert–Tully | Barnards | 17% (2 m) | | | cyclone Larry 2006 (95% at 2 m 87% at 5 m), cyclone Yasi 2011 (53% at 2 m, 27% at 5 m), Bleaching 2018 (10% at 5 m)– ongoing impact of 2017 event, Disease 2021 (18% 2 m, 9% 5 m) |
| | Dunk North | 18% (2 m)<br>16% (5 m) | | 15% (2 m)<br>16% (5 m) | cyclone Larry 2006 (81% at 2 m, 66% at 5 m), Disease 2007 (34% at 2 m), cyclone Yasi 2011 (93% at 2 m, 75% at 5 m), Flood 2025 (69% at 2 m) |
| | Dunk South | 45% (2 m)<br>6% (5 m) | 20% (2 m)<br>12% (5 m) | | cyclone Larry 2006 (18% at 2 m, 19% at 5 m), cyclone Yasi 2011 (79% at 2 m, 56% at 5 m), Bleaching 2018 (28% at 5 m) – ongoing impact of 2017 event, Flood 2025 (68% at 2 m) |
| | Bedarra | 36% (2 m)<br>10% (5 m) | 16% (2 m)<br>10% (5 m) | | Bleaching 2018 (26% at 5 m)– ongoing impact of 2017 event, Flood 2025 (80% at 2 m, 39% at 5 m) |

Table A6 (continued).

| Region | Reef | Bleaching 1998 | 2002 | 2017 | 2020 | 2022 | 2024 | Other recorded disturbances |
|---|---|---|---|---|---|---|---|---|
| Burdekin | Palms East | na | na | | | | 40% (2 m)<br>29% (5 m) | cyclone Larry 2006 (23% at 2 m, 39% at 5 m), cyclone Yasi 2011 (83% at 2 m and 5 m), Storm 2025 (32% at 2 m, 19% at 5 m) |
| | Palms West | 83% | | 30% (2 m)<br>15% (5 m) | | | | Unknown 1995-1997 although possibly cyclone Justin (32%), cyclone Larry 2006 (15% at 2 m), Storm 2010 (68% at 2 m), cyclone Kirrily 2024 (28% at 2 m), Flood 2025 (41% at 2 m, 31% at 5 m) |
| | Lady Elliott Reef | na | na | | 26% (2 m)<br>8% (5 m) | | | cyclone Yasi 2011 (86% at 2 m, 45% at 5 m), Flood 2025 (92% at 2 m, 25% at 5 m) |
| | Pandora Reef | 21% | 2% | 33% (2 m) | 18% (2 m) | | | cyclone Larry 2006 (80% at 2 m, 35% at 5 m), Storm 2009 (37% at 2 m, 56% at 5 m), cyclone Yasi 2011 (30% at 2 m, 57% at 5 m), Storm 2025 (32% at 2 m, 42% at 5 m) |
| | Pandora North | 12% | | 5 %* | n/a | | | bleaching 1996–ongoing impact of 1998 event, cyclone Tessie 2000 (12%), cyclone Yasi 2011 (25%), cyclone Kirrily 2024 (11%), storm 2025 (10%) |
| | Havannah | na | na | 37% (2 m)<br>11% (5 m) | 33% (2 m)<br>8% (5 m) | | | cyclone Yasi 2011 (35% at 2 m, 34% at 5 m), Disease 2016 (9% at 2 m), Bleaching 2018 (26% at 2 m, 16% at 5 m)– ongoing impact of 2017 event, Disease 2019 (23% at 2 m), Bleaching 2021 (26% 2 m)– ongoing impact of 2020 event, Storm 2025 (23% at 2 m, 21% at 5 m) |
| | Havannah North | 39% | 19% | | 51% | | | Bleaching 1999 (27%)–ongoing impact of 1998 event, cyclone Tessie 2000 (54%), 2001 COTS (44%) cyclone Yasi 2011 (69%), cyclone Kirrily 2024 (16%) |
| | Magnetic | 24% | 37% | 31% (2 m) | 36% (2 m)<br>18% (5 m) | 13% (2 m)<br>20% (5 m) | 46% (2 m)<br>26% (5 m) | cyclone Joy 1990 (13%), Bleaching 1993 (10%), cyclone Tessie 2000 (18%), cyclone Larry 2006 (39% at 2 m, 5% at 5 m), cyclone Yasi and Flood/Bleaching 2011 (39% at 2 m, 20% at 5 m), Bleaching 2021 ongoing from 2020 (13% 5 m) |

Table A6 (continued).

| Region | Reef | Bleaching 1998 | Bleaching 2002 | Bleaching 2017 | Bleaching 2020 | Other recorded disturbances |
|---|---|---|---|---|---|---|
| Mackay–Whitsunday | Hook | na | na | | 27% (2 m) 20% (5 m | bleaching Jan 2006, probable although not observed as we did not visit region at time of event. Same for other reefs in region, cyclone Ului 2010 (31% at 2 m,17% at 5 m), cyclone Debbie 2017 (83% at 2 m, 45% at 5 m) |
| | Dent | 32% | na | ** | | disease 2007 (17% at 2 and at 5 m), cyclone Ului 2010 (21% at 2 m, 27% at 5 m), cyclone Debbie 2017 (48% at 2 m, 38% at 5 m), disease 2019 (44% at 2 m, 25% at 5 m), Disease 2021 (16% at 5 m) |
| | Seaforth | na | na | ** | 8% (2 m) | Flood 2009 (16% at 2 m,, 22% at 5 m), cyclone Debbie 2017 (45% at 2 m, 26% at 5 m) |
| | Double Cone | na | na | ** | 15% (2 m) 3% (5 m) | Flood 2009 (13% at 2 m), cyclone Ului 2010 (26% at 2 m, 12% at 5 m), cyclone Debbie 2017 (97% at 2 m, 74% at 5 m) |
| | Daydream | 44% | na | ** | 42% (2 m) 38% (5 m) | Disease 2008 (26% at 2 m, 20% at 5 m), cyclone Ului 2010 (47% at 2 m, 45% at 5 m), cyclone Debbie 2017 (98% at 2 m, 90% at 5 m) |
| | Shute Harbour | | na | ** | 10% (2 m) | cyclone Ului 2010 (8% at 2 m), cyclone Debbie 2017 (48% at 2 m, 55% at 5 m) |
| | Pine | | na | ** | 35% (2 m) | Flood 2009 (14% at 2 and at 5 m), cyclone Ului 2010 (13% at 2 m, 10% at 5 m), disease 2011(15% at 5 m), cyclone Debbie 2017 (74% at 2 m, 56% at 5 m), disease 2019 (40% at 2 m, 29% at 5 m) |
| | Hayman | | 11% | | | Cyclone Celeste 1996 (8%), cyclone Justin 1997 (14%), disease 2009 (13%), cyclone Ului 2010 (36%), cyclone Debbie 2017 (recorded 2019) (86% ) |
| | Border | | 12% | | | cyclone Debbie 2017 (recorded 2019) (45% ) |

Table A6 (continued).

| Region | Reef | Bleaching | | | | | Other recorded disturbances |
|---|---|---|---|---|---|---|---|
| | | 1998 | 2002 | 2006 | 2020 | 2024 | |
| Fitzroy | Barren | na | na | 26% (2 m)<br>29% (5 m) | | 58% (2 m)<br>16% (5 m) | storm Feb 2008 (42% at 2 m, 24% at 5 m), Storm Feb 2010 plus disease (25% at 2 m,8% at 5 m), disease 2011 (16% at 5 m), Storm Feb 2013 (51% at 2 m, 48% at 5 m) ), Storm Feb 2014 (18% at 2 m and at 5 m), cyclone Marcia 2015 (45% at 2 m, 20% at 5 m), clear bleaching mortality in 2020 obscured by rapid growth, disease 2023 (18% at 5 m), bleaching 2025 ongoing from 2024 (32% at 2 m, 27% at 5 m) |
| | North Keppel | 15% | 0.89 (36%) | 62% (2 m)<br>41% (5 m) | 18% (2 m)<br>7% (5 m) | 67% (2 m)<br>34% (5 m) | disease 2011 (28% at 2 m and 55% at 5 m), bleaching 2025 ongoing from 2024 (74% at 2 m, 35% at 5 m) |
| | Middle Is | 56% | | 61% (2 m)<br>38% (5 m) | 15% (2 m) | 64% (2 m)<br>53% (5 m) | storm Feb 2010 plus disease (29% at 2 m, 43% at 5 m) cyclone Marcia 2015 (30% at 2 m, 33% at 5 m), bleaching 2021 ongoing from 2020 (49% 2 m), disease 2023 (41% at 2m), bleaching 2025 ongoing from 2024 (67% at 2 m, 45% at 5 m) |
| | Keppels South | 6% | 26% | 28% (2 m)<br>27% (5 m) | 1% (2 m)<br>2% (5 m) | 53% (2 m)<br>48% (5 m) | flood 2008 and associated disease (14% at 2 m, 15% at 5 m), disease 2010 (12% at 2 m 22% at 5 m), flood 2011 and associated disease (85% at 2 m, 23% at 5 m), bleaching 2021 ongoing from 2020 (22% 5 m), bleaching 2025 ongoing from 2024 (71% at 2 m, 60% at 5 m) |
| | Pelican | na | na | 17% (5 m) | | 58% (2 m)<br>8% (5 m) | flood plus storm 2008 (29% at 2 m, 7% at 5 m), disease 2009 (13% at 5 m), disease 2010 (28% at 2 m), flood 2011 (99%at 2 m, 32% at 5 m), cyclone Marcia 2015 (65% at 2 m, 35% at 5 m), bleaching 2021 ongoing from 2020 (66% 2 m) |

Note: Estimates of impact for the 1998 and 2002 bleaching, that predate the start of the MMP, were estimated from available data sets reported by Sweatman *et al.* 2007. Where impact was observed, the proportional reductions in coral cover are based on observed values of hard coral cover in pre and post bleaching surveys and do not account for the additional loss based on expected growth between samples as reported for the MMP and LTMP sampled reefs.

Table A7. Reef-level Coral Index and indicator scores 2025. Coral Index and (sub-)regional indicator scores are colour coded by Reef Water Quality Report Card categories: red = very poor, orange = poor, yellow = moderate, light green = good and dark green = very good.

| (sub-) region | Reef | Dept | Coral cover | Juvenile coral | Macroalgae | Cover change | Composition | Coral Index |
|---|---|---|---|---|---|---|---|---|
| Barron Daintree | Low Isles | 5 | 0.6 | 1 | 0.92 | 0.67 | 0.5 | 0.74 |
| | Snapper North | 2 | 0.25 | 0.04 | 0 | 0.53 | 0 | 0.16 |
| | Snapper North | 5 | 0.57 | 0.19 | 0.77 | 0.73 | 0.5 | 0.55 |
| | Snapper South | 2 | 0 | 0 | 1 | 0.67 | 0 | 0.33 |
| | Snapper South | 5 | 0 | 0.01 | 0 | 0.32 | 0 | 0.06 |
| Poor | | | **0.28** | **0.25** | **0.54** | **0.58** | **0.2** | **0.37** |
| Johnstone Russell Mulgrave | Fitzroy East | 2 | 0.46 | 0.24 | 0.93 | 0.44 | 0.5 | 0.51 |
| | Fitzroy East | 5 | 0.72 | 0.4 | 0.83 | 0.61 | 0 | 0.51 |
| | Fitzroy West | 2 | 0.74 | 0.23 | 0.35 | 1 | 1 | 0.66 |
| | Fitzroy West | 5 | 0.85 | 0.42 | 0.55 | 0.63 | 0.5 | 0.59 |
| | Fitzroy West LTMP | 5 | 0.74 | 1 | 0.83 | 0.33 | 1 | 0.78 |
| | Franklands East | 2 | 0.65 | 0.25 | 0 | 0.4 | 1 | 0.46 |
| | Franklands East | 5 | 0.35 | 0.28 | 0 | 0.5 | 1 | 0.43 |
| | Franklands West | 2 | 0.74 | 0.07 | 0 | 0.46 | 0.5 | 0.35 |
| | Franklands West | 5 | 0.7 | 0.11 | 0 | 0.5 | 1 | 0.46 |
| | High East | 2 | 0.38 | 0.19 | 0 | 0.14 | 0.5 | 0.24 |
| | High East | 5 | 0.53 | 0.21 | 0 | 0.36 | 0.5 | 0.32 |
| | High West | 2 | 0.75 | 0.18 | 0.42 | 1 | 0 | 0.47 |
| | High West | 5 | 0.32 | 0.16 | 0.75 | 0.74 | 0 | 0.39 |
| Moderate | | | **0.61** | **0.29** | **0.36** | **0.55** | **0.58** | **0.48** |
| Tully Herbert | Barnards | 2 | 0.72 | 0.19 | 0.59 | 0.38 | 1 | 0.57 |
| | Barnards | 5 | 0.74 | 0.46 | 0.84 | 0.27 | 1 | 0.66 |
| | Dunk North | 2 | 0.22 | 0.36 | 0.41 | 0.39 | 0 | 0.28 |
| | Dunk North | 5 | 0.52 | 0.68 | 0.56 | 0.77 | 0.5 | 0.61 |
| | Dunk South | 2 | 0.2 | 0.17 | 0.69 | 0.29 | 0.5 | 0.37 |
| | Dunk South | 5 | 0.53 | 0.58 | 0 | 0.33 | 0 | 0.29 |
| | Bedarra | 2 | 0.05 | 0.22 | 0.04 | 0.15 | 0.5 | 0.19 |
| | Bedarra | 5 | 0.23 | 0.79 | 0.7 | 0.33 | 0.5 | 0.51 |
| Moderate | | | **0.4** | **0.43** | **0.48** | **0.36** | **0.5** | **0.43** |
| Burdekin | Palms East | 2 | 0.37 | 0.1 | 1 | 0.23 | 1 | 0.54 |
| | Palms East | 5 | 0.5 | 0.17 | 0.97 | 0.19 | 1 | 0.57 |
| | Palms West | 2 | 0.21 | 0.22 | 1 | 0.56 | 0 | 0.4 |
| | Palms West | 5 | 0.5 | 0.46 | 1 | 0.5 | 0 | 0.49 |
| | Havannah North | 5 | 0.29 | 0.74 | 0.1 | 0.33 | 0.5 | 0.39 |
| | Havannah | 2 | 0.43 | 0.23 | 0.29 | 0.36 | 0.5 | 0.36 |
| | Havannah | 5 | 0.55 | 0.23 | 0.07 | 0.26 | 1 | 0.42 |
| | Pandora | 2 | 0.25 | 0.1 | 0.61 | 0.43 | 0.5 | 0.38 |
| | Pandora | 5 | 0.25 | 0.19 | 0.93 | 0.47 | 0.5 | 0.47 |
| | Pandora North | 5 | 0.83 | 0.45 | 0.25 | 0.5 | 0 | 0.4 |
| | Lady Elliot | 2 | 0.04 | 0.04 | 1 | 0.25 | 0 | 0.27 |
| | Lady Elliot | 5 | 0.43 | 0.3 | 1 | 0.43 | 0.5 | 0.53 |
| | Magnetic | 2 | 0.26 | 0.35 | 0 | 0.17 | 0.5 | 0.26 |
| | Magnetic | 5 | 0.34 | 0.62 | 0 | 0.55 | 0 | 0.3 |
| Moderate | | | **0.37** | **0.30** | **0.59** | **0.37** | **0.43** | **0.41** |

Table A7 (continued).

| (sub-) region | Reef | Depth | Coral cover | Juvenile coral | Macroalgae | Cover change | Composition | Coral Index |
|---|---|---|---|---|---|---|---|---|
| Mackay Whitsunday | Hayman | 5 | 0.35 | 1 | 1 | 0.59 | 1 | 0.79 |
| | Border | 5 | 0.6 | 0.92 | 1 | 0.19 | 0 | 0.54 |
| | Hook | 2 | 0.39 | 0.54 | 1 | 0.46 | 0.5 | 0.58 |
| | Hook | 5 | 0.58 | 0.34 | 0.77 | 0.53 | 1 | 0.64 |
| | Double Cone | 2 | 0.08 | 0.35 | 0 | 0.46 | 0 | 0.18 |
| | Double Cone | 5 | 0.32 | 0.36 | 0 | 0.46 | 0 | 0.23 |
| | Daydream | 2 | 0.08 | 0.74 | 0 | 0.72 | 0 | 0.31 |
| | Daydream | 5 | 0.23 | 1 | 0.6 | 0.72 | 0 | 0.51 |
| | Dent | 2 | 0.48 | 0.35 | 0 | 0.25 | 0 | 0.22 |
| | Dent | 5 | 0.54 | 0.34 | 0.31 | 0.48 | 0 | 0.34 |
| | Shute Harbour | 2 | 0.85 | 0.41 | 0.62 | 0.2 | 1 | 0.62 |
| | Shute Harbour | 5 | 0.45 | 0.52 | 0.8 | 0.36 | 1 | 0.63 |
| | Pine | 2 | 0.14 | 0.34 | 0 | 0.13 | 0 | 0.12 |
| | Pine | 5 | 0.28 | 0.4 | 0.31 | 0.29 | 0.5 | 0.35 |
| | Seaforth | 2 | 0.26 | 0.35 | 0 | 0.11 | 1 | 0.34 |
| | Seaforth | 5 | 0.35 | 0.5 | 0 | 0.18 | 0.5 | 0.31 |
| Moderate | | | **0.37** | **0.53** | **0.40** | **0.38** | **0.41** | **0.42** |
| Fitzroy | Barren | 2 | 0.39 | 0.13 | 0.46 | 0.58 | 0 | 0.31 |
| | Barren | 5 | 0.65 | 0.03 | 0 | 0.5 | 0 | 0.24 |
| | North Keppel | 2 | 0.09 | 0.01 | 0 | 0.38 | 0.5 | 0.2 |
| | North Keppel | 5 | 0.26 | 0.06 | 0 | 0.37 | 0 | 0.14 |
| | Middle | 2 | 0.08 | 0.06 | 0 | 0.18 | 0 | 0.06 |
| | Middle | 5 | 0.12 | 0.11 | 0 | 0.12 | 0 | 0.07 |
| | Keppels South | 2 | 0.18 | 0.06 | 0 | 0.34 | 0 | 0.12 |
| | Keppels South | 5 | 0.14 | 0.05 | 0 | 0.31 | 0 | 0.1 |
| | Pelican | 2 | 0.09 | 0.11 | 0 | 0.71 | 0 | 0.18 |
| | Pelican | 5 | 0.39 | 0.41 | 0 | 0.36 | 0.5 | 0.33 |
| Very poor | | | **0.24** | **0.1** | **0.05** | **0.38** | **0.1** | **0.17** |

Table A8. Environmental covariates for coral locations. Wet season Chl *a* median values over the 2021-2025 wet seasons estimated form the proportion of time Sentinel satellite imagery pixels adjacent to each site were classified as water types I-IV (Gruber et al. 2026) and the distribution of niskin samples taken within each water type, median k490 light attenuation coefficients from pixels adjacent to reef sites based on MODIS observations over the period July 2021 to June 2025, mean concentrations from MMP routine water quality monitoring Niskin samples between July 2021 and June 2025 samples for oxides of nitrogen ($NO_x$), particulate nitrogen (PN), dissolved inorganic phosphorous ($PO_4$) and particulate phosphorous (PP) and me Chl *a*, Suspended solids, Oxides of Nitrogen, and the ratio of N to P. Values exceeding Reef wide wet-season (0.63 $\mu gL^{-1}$ Chl *a*, and 2.4 $mgL^{-1}$ for TSS) or annual (0.45 $\mu gL^{-1}$ Chl *a*, 1.6 $mgL^{-1}$ TSS and 0.35 $NO_x$ ) guideline values (Great Barrier Reef Marine Park Authority 2010, Moran *et al.* 2025) are shaded.

| (sub-)region | Reef | Satellite data | | Niskin data | | | | Logger data | |
|---|---|---|---|---|---|---|---|---|---|
| | | Chl *a* ($\mu gL^{-1}$) | k490* ($m^{-1}$) | $NO_x$ ($\mu gL^{-1}$) | PN ($\mu gL^{-1}$) | $PO_4$ ($\mu gL^{-1}$) | PP ($\mu gL^{-1}$) | Chl *a* ($\mu gL^{-1}$)l | Turbidity (ntu) |
| Barron–Daintree | Low Isles | 0.287 | 0.077 | | | | | | |
| | Snapper North | 0.398 | 0.137 | | | | | | |
| | Snapper South | 0.400 | 0.138 | | | | | | |
| Johnstone Russell–Mulgrave | Franklands East | 0.274 | 0.067 | | | | | | |
| | Fitzroy East | 0.280 | 0.070 | | | | | | |
| | Franklands | 0.289 | 0.078 | 1.742 | 21.718 | 1.891 | 2.172 | 0.401 | 0.903 |
| | Fitzroy West | 0.312 | 0.095 | 1.578 | 21.164 | 1.863 | 2.109 | 0.445 | 1.195 |
| | High East | 0.324 | 0.106 | | | | | | |
| | High West | 0.407 | 0.120 | 2.501 | 25.713 | 1.978 | 2.803 | 0.478 | 1.156 |
| Herbert–Tully | Barnards | 0.363 | 0.118 | | | | | | |
| | Dunk North | 0.394 | 0.134 | 1.460 | 24.696 | 1.798 | 3.054 | 0.635 | 3.088 |
| | Dunk South | 0.424 | 0.149 | | | | | | |
| | Bedarra | 0.454 | 0.160 | | | | | | |
| Burdekin | Palms East | 0.264 | 0.068 | | | | | | |
| | Havannah North | 0.283 | 0.088 | | | | | | |
| | Havannah | 0.308 | 0.093 | | | | | | |
| | Palms West | 0.324 | 0.088 | 0.935 | 19.846 | 1.796 | 2.159 | 0.566 | 0.820 |
| | Pandora North | 0.348 | 0.115 | | | | | | |
| | Pandora | 0.351 | 0.116 | 0.996 | 21.414 | 1.893 | 2.734 | 0.466 | 1.570 |
| | Lady Elliot | 0.491 | 0.186 | | | | | | |
| | Magnetic | 0.496 | 0.180 | 1.325 | 28.789 | 2.081 | 3.524 | 0.580 | 2.339 |
| Mackay–Whitsunday | Hayman | 0.250 | 0.064 | | | | | | |
| | Border | 0.265 | 0.078 | | | | | | |
| | Hook | 0.274 | 0.083 | | | | | | |
| | Double Cone | 0.283 | 0.091 | 1.356 | 24.608 | 1.857 | 2.805 | 0.624 | 1.415 |
| | Seaforth | 0.320 | 0.104 | 1.798 | 22.959 | 2.232 | 3.040 | 0.615 | 1.698 |
| | Daydream | 0.333 | 0.107 | | | | | | |
| | Dent | 0.357 | 0.111 | | | | | | |
| | Shute Harbour | 0.357 | 0.113 | | | | | | |
| | Pine | 0.387 | 0.118 | 3.291 | 24.276 | 2.567 | 3.195 | 0.704 | 2.315 |
| Fitzroy | Barren | 0.272 | 0.061 | 0.791 | 19.759 | 1.005 | 2.009 | 0.419 | 0.553 |
| | North Keppel | 0.352 | 0.095 | | | | | | |
| | Keppels South | 0.356 | 0.096 | 1.045 | 24.730 | 0.946 | 2.542 | 0.446 | 1.034 |
| | Middle | 0.360 | 0.108 | | | | | | |
| | Pelican | 0.494 | 0.173 | 1.219 | 30.471 | 2.352 | 4.233 | | |

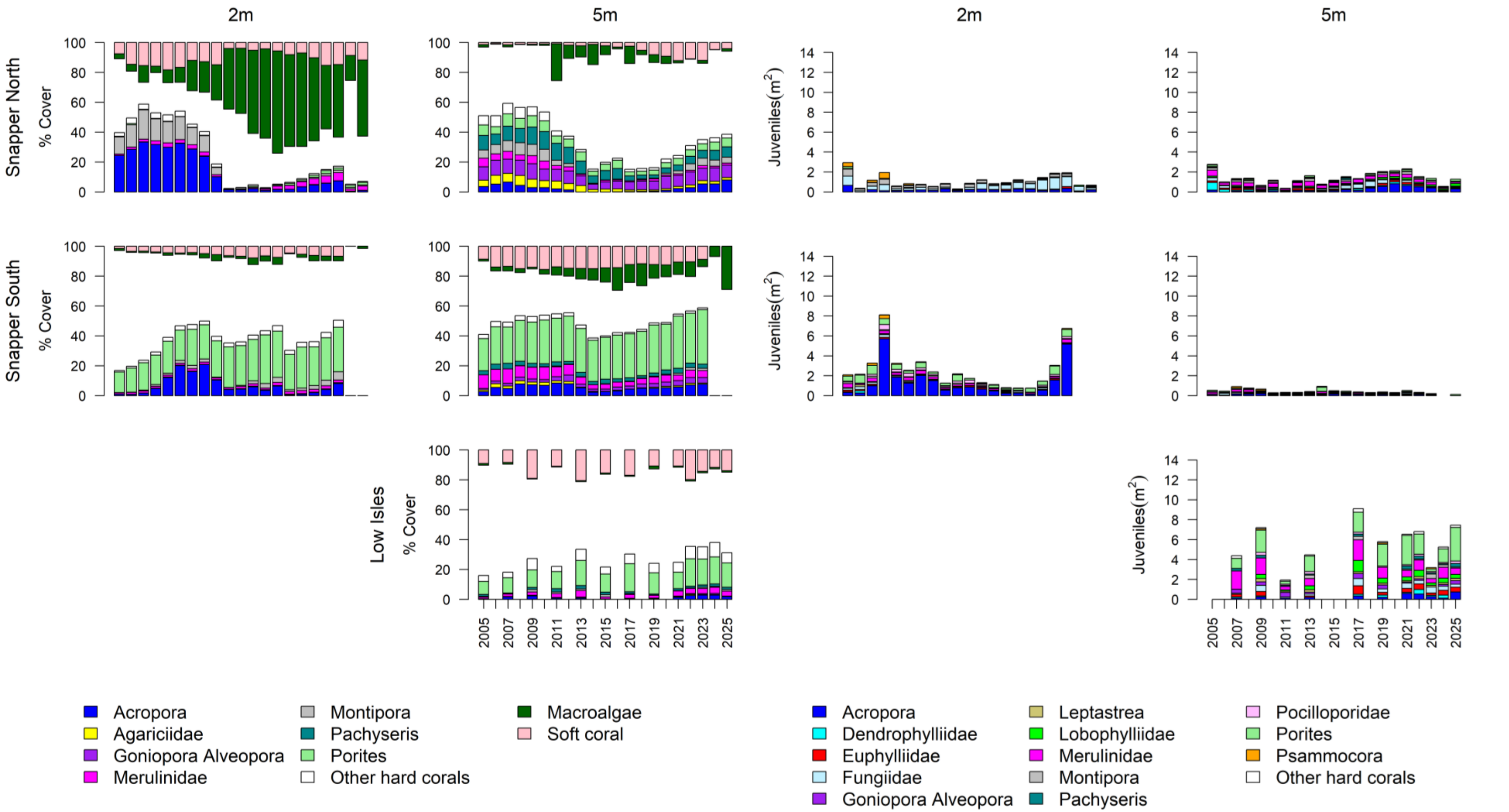



Figure A1. Barron–Daintree sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral groups and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral groups. Separate legends relevant groupings for cover and juvenile density estimates are located beneath the relevant plots. Juvenile density estimates are based on the number of juveniles recorded per square metre of transect.

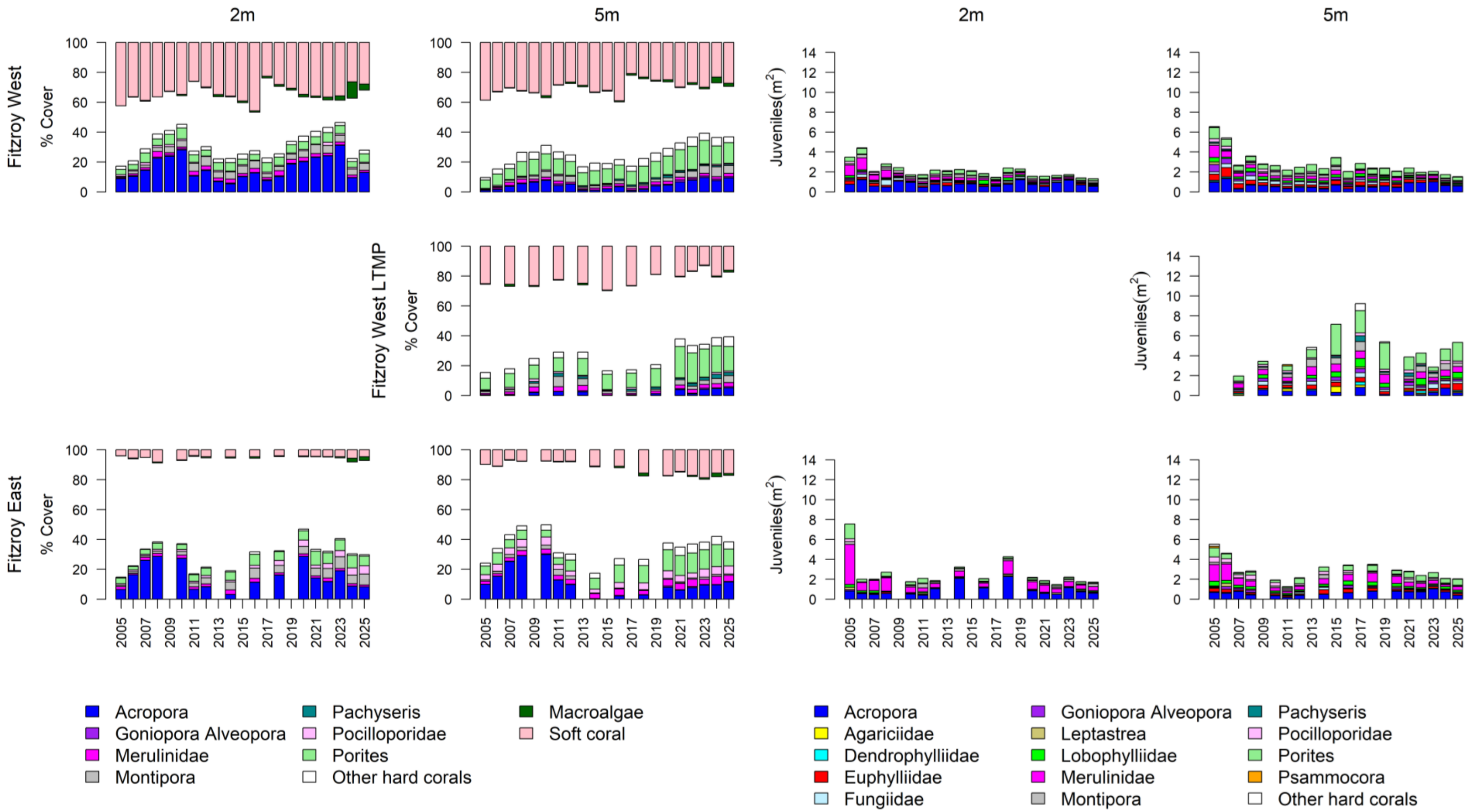
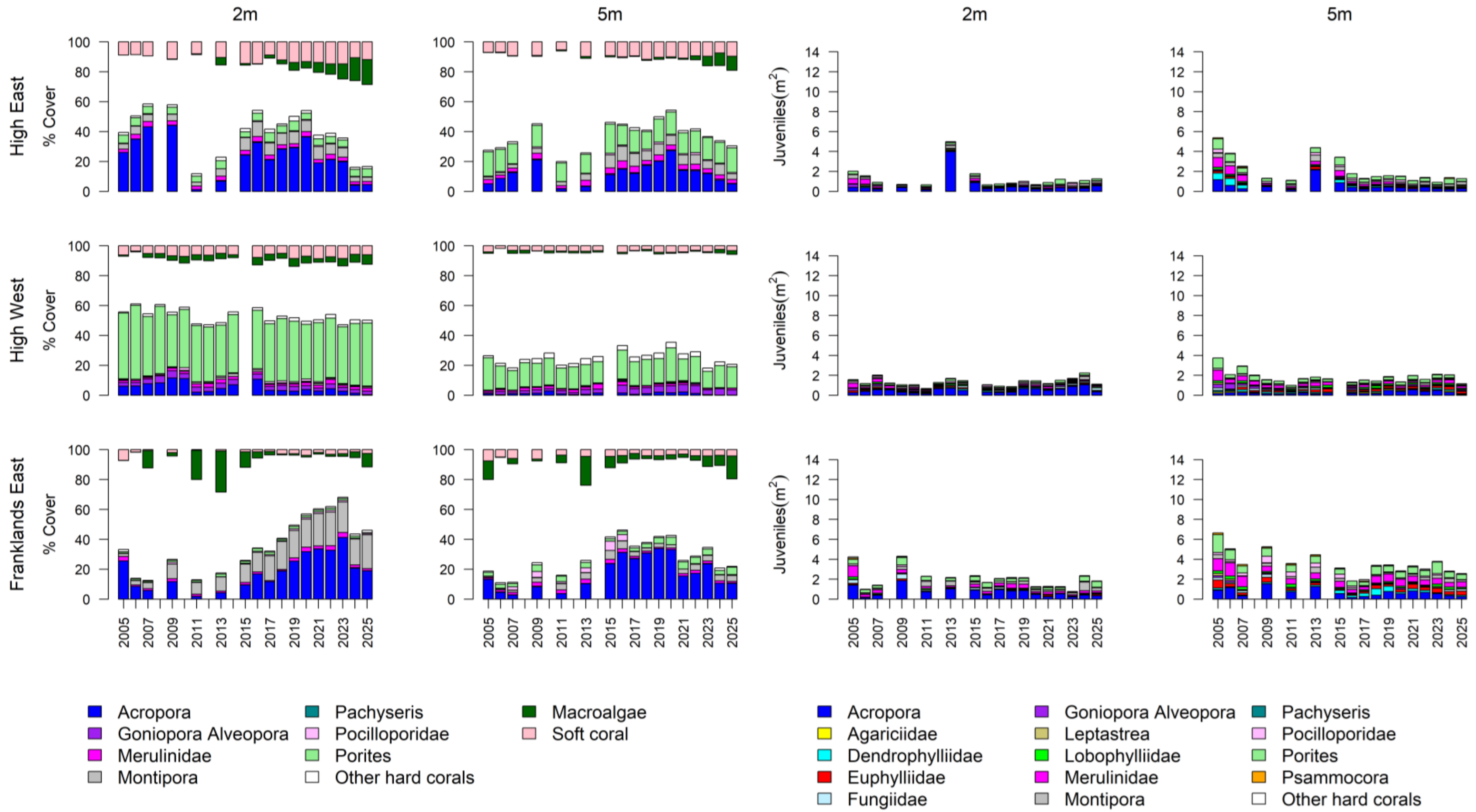



Figure A2. Johnstone Russell–Mulgrave sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral groups and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral groups. Separate legends relevant groupings for cover and juvenile density estimates are located beneath the relevant plots. Juvenile density estimates are based on the number of juveniles recorded per square metre of transect.

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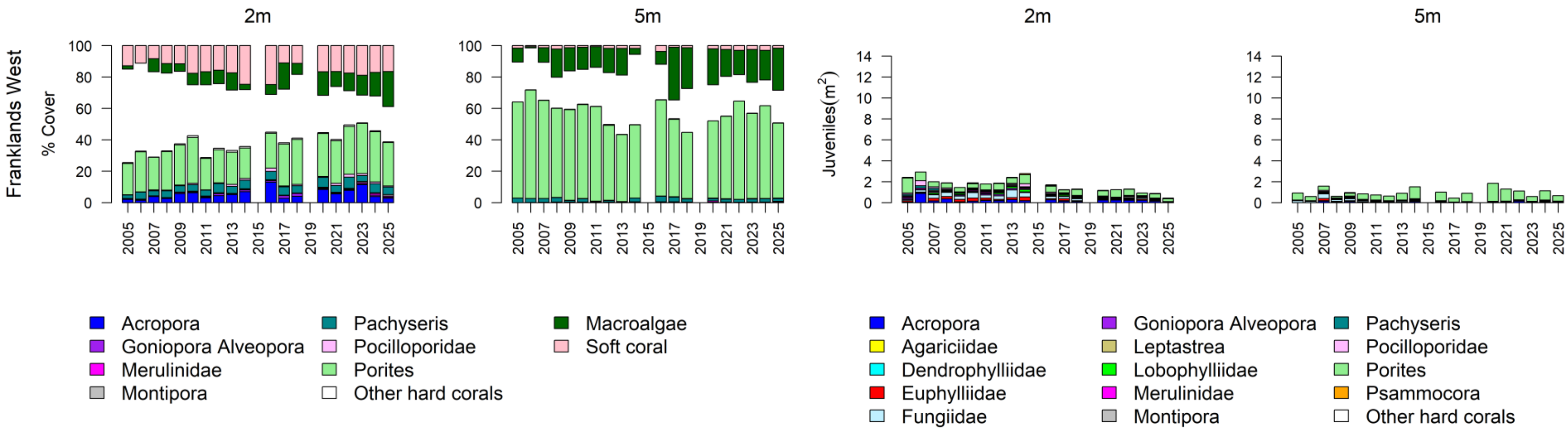



Figure A2 (continued).

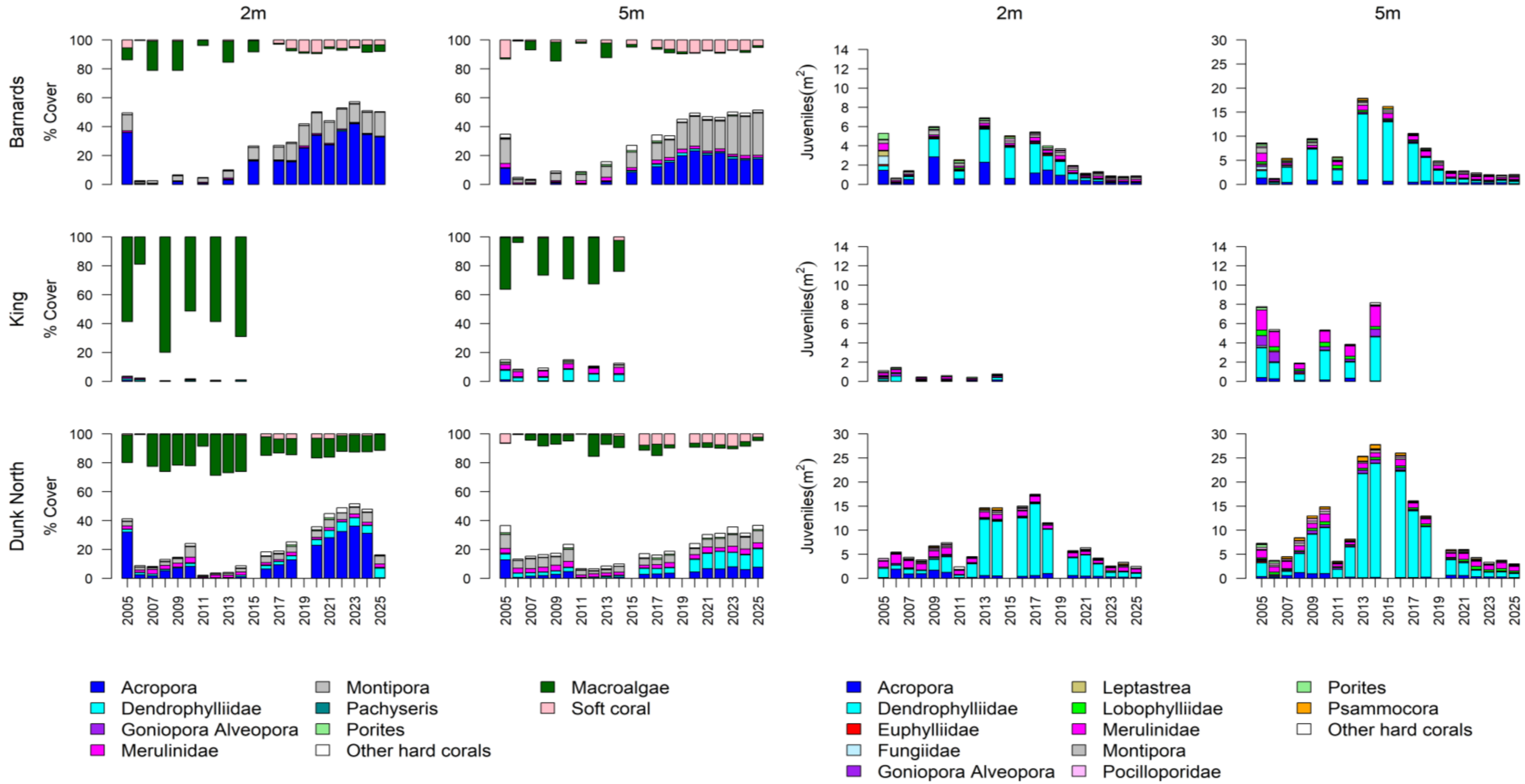
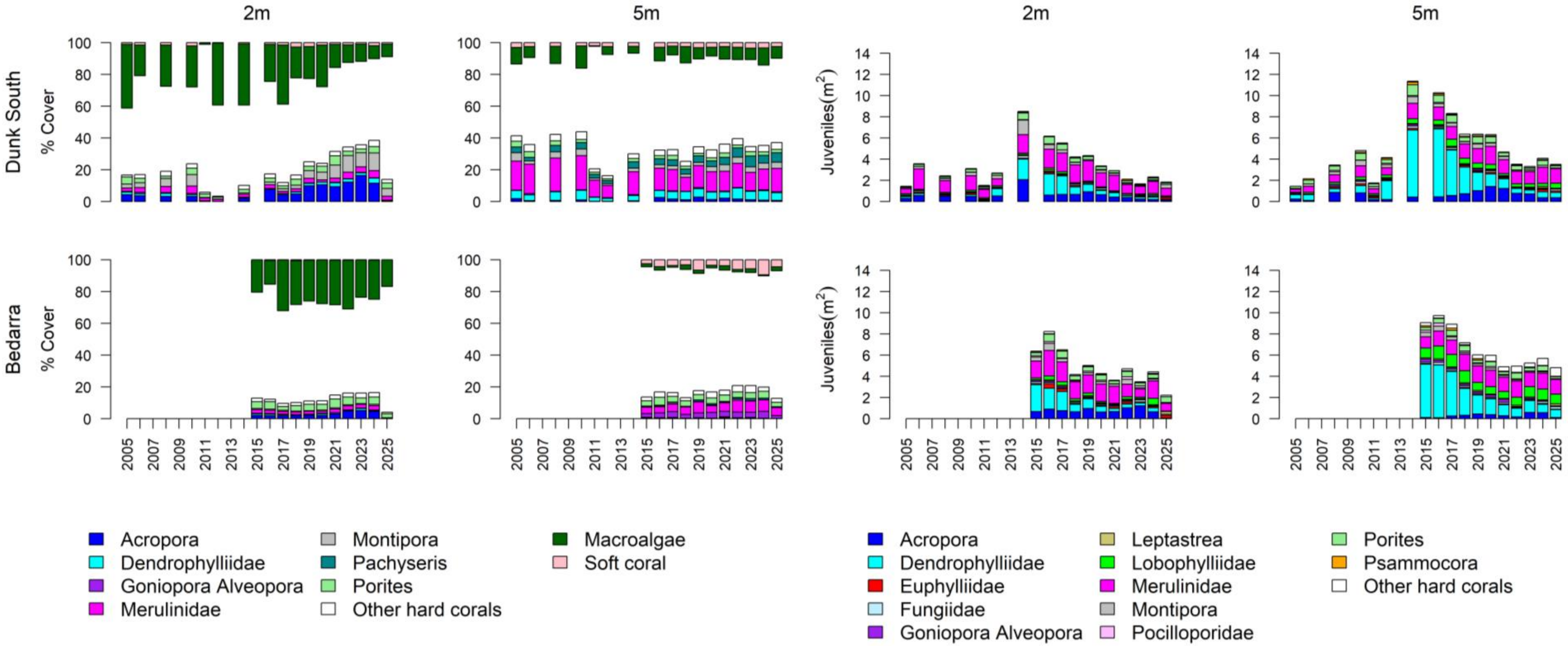



Figure A3. Herbert–Tully sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral groups and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral groups. Separate legends with relevant groupings for cover and juvenile density estimates are located beneath the respective plots. Juvenile density estimates are based on the number of juveniles recorded per square metre of transect.

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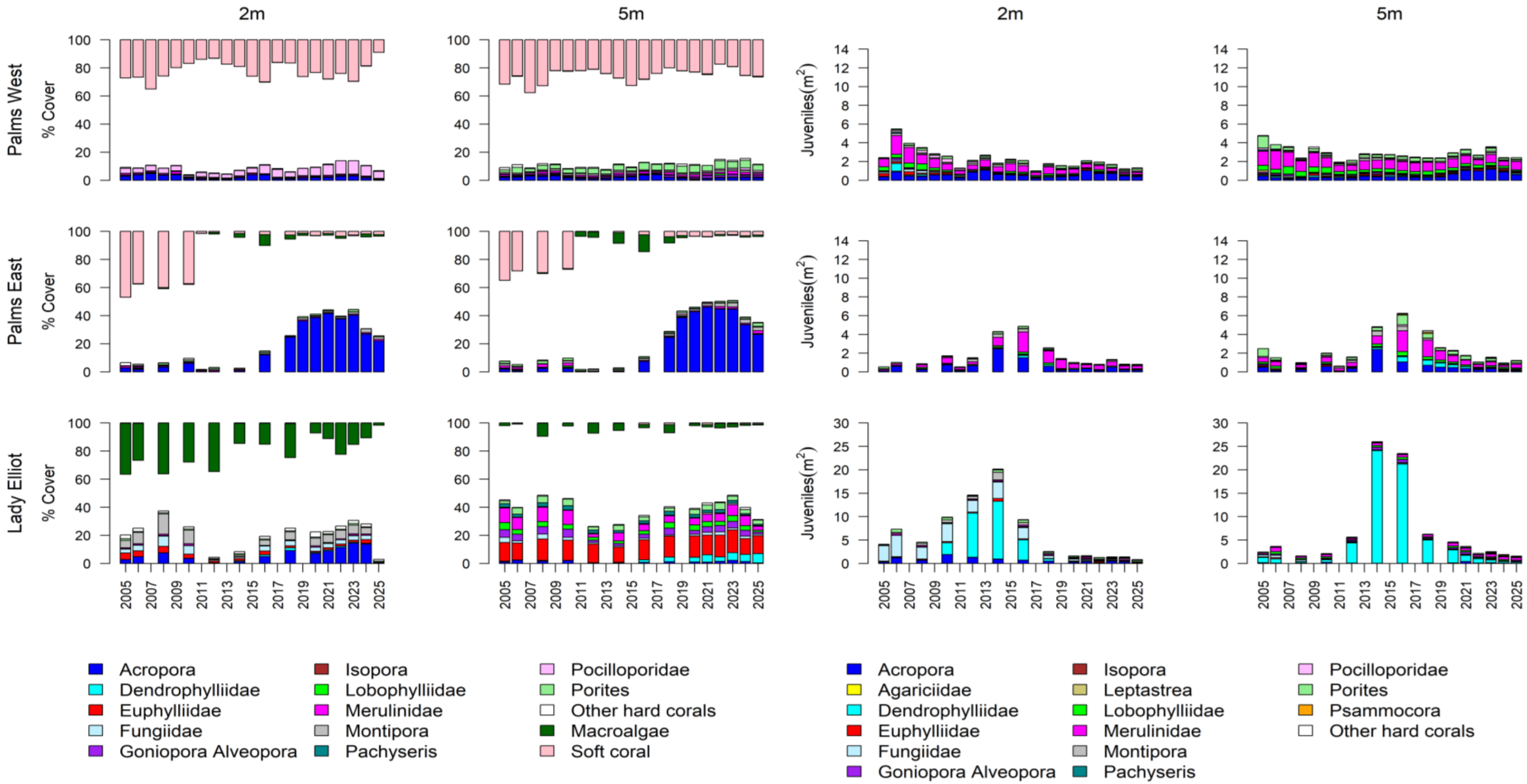
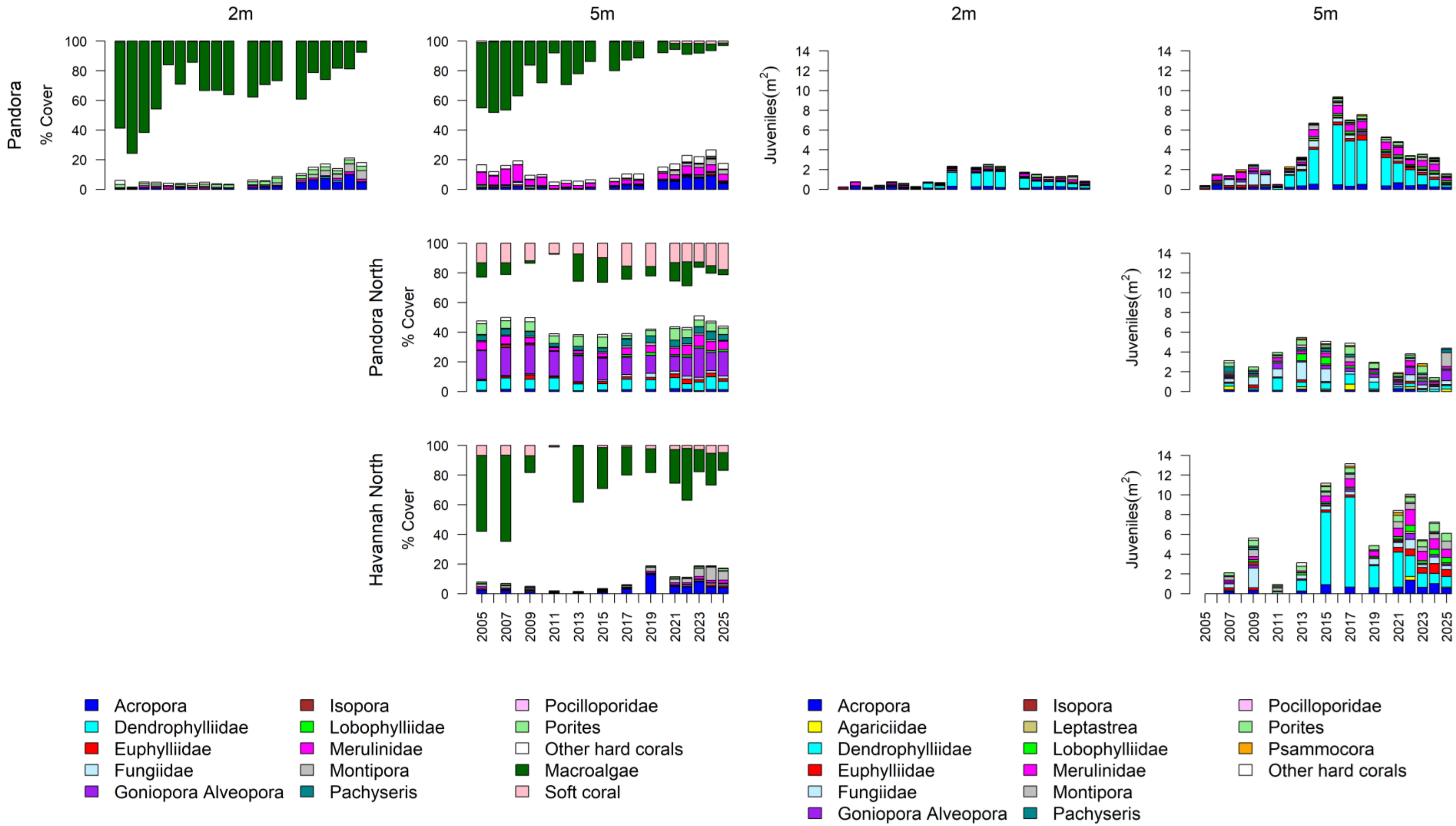



Figure A4. Burdekin region benthic community composition. Cover estimates are separated into regionally abundant hard coral groups and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral groups. Separate legends with relevant groupings for cover and juvenile density estimates are located beneath the respective plots. Juvenile density estimates are based on the number of juveniles recorded per square metre of transect.

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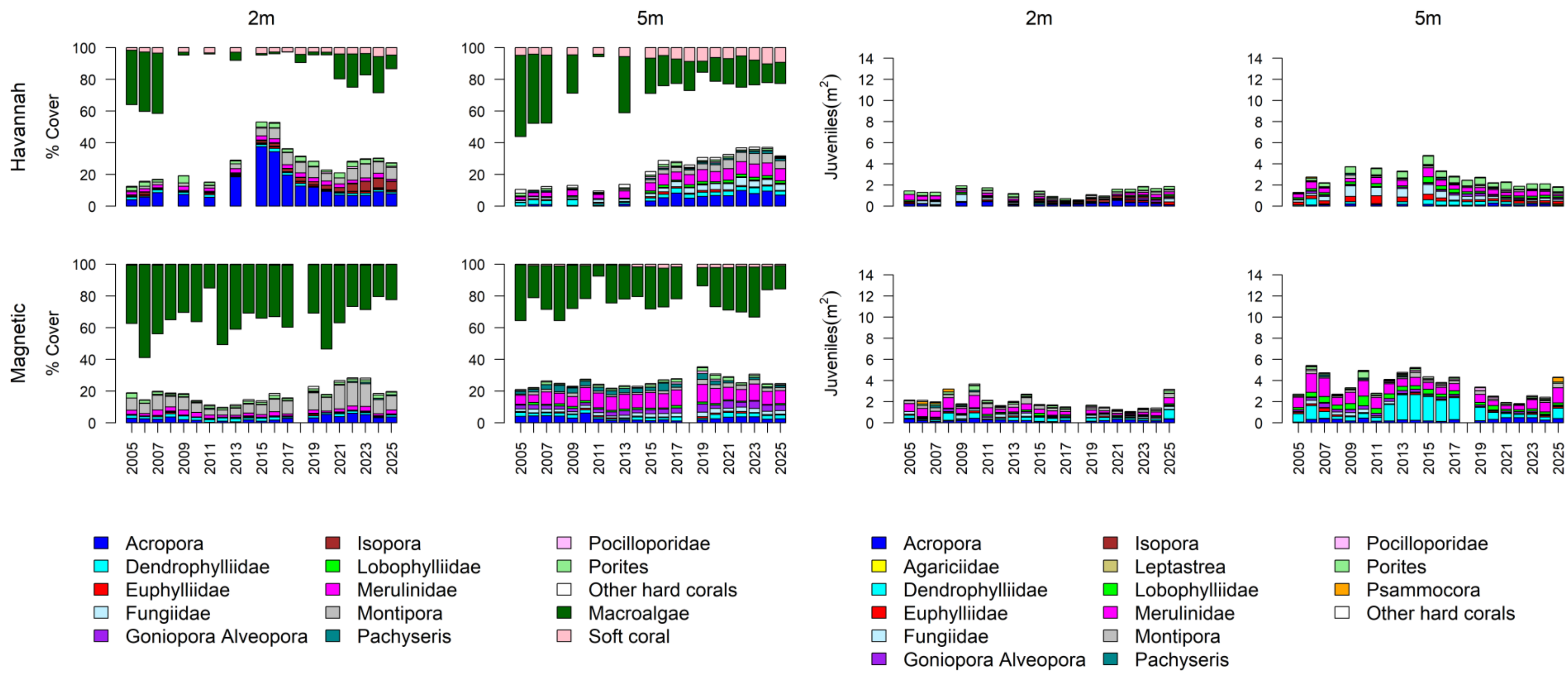



Figure A4 (continued).

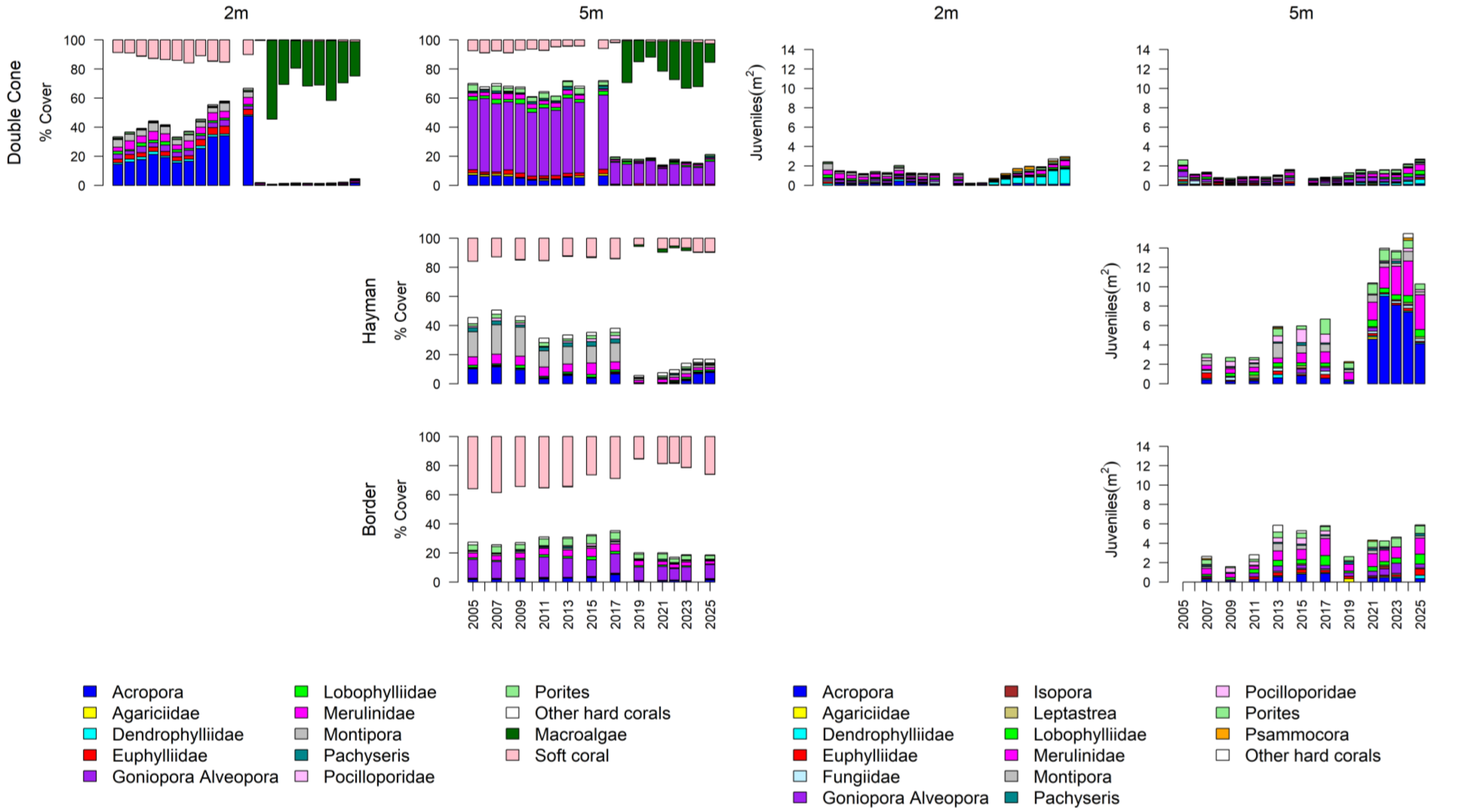
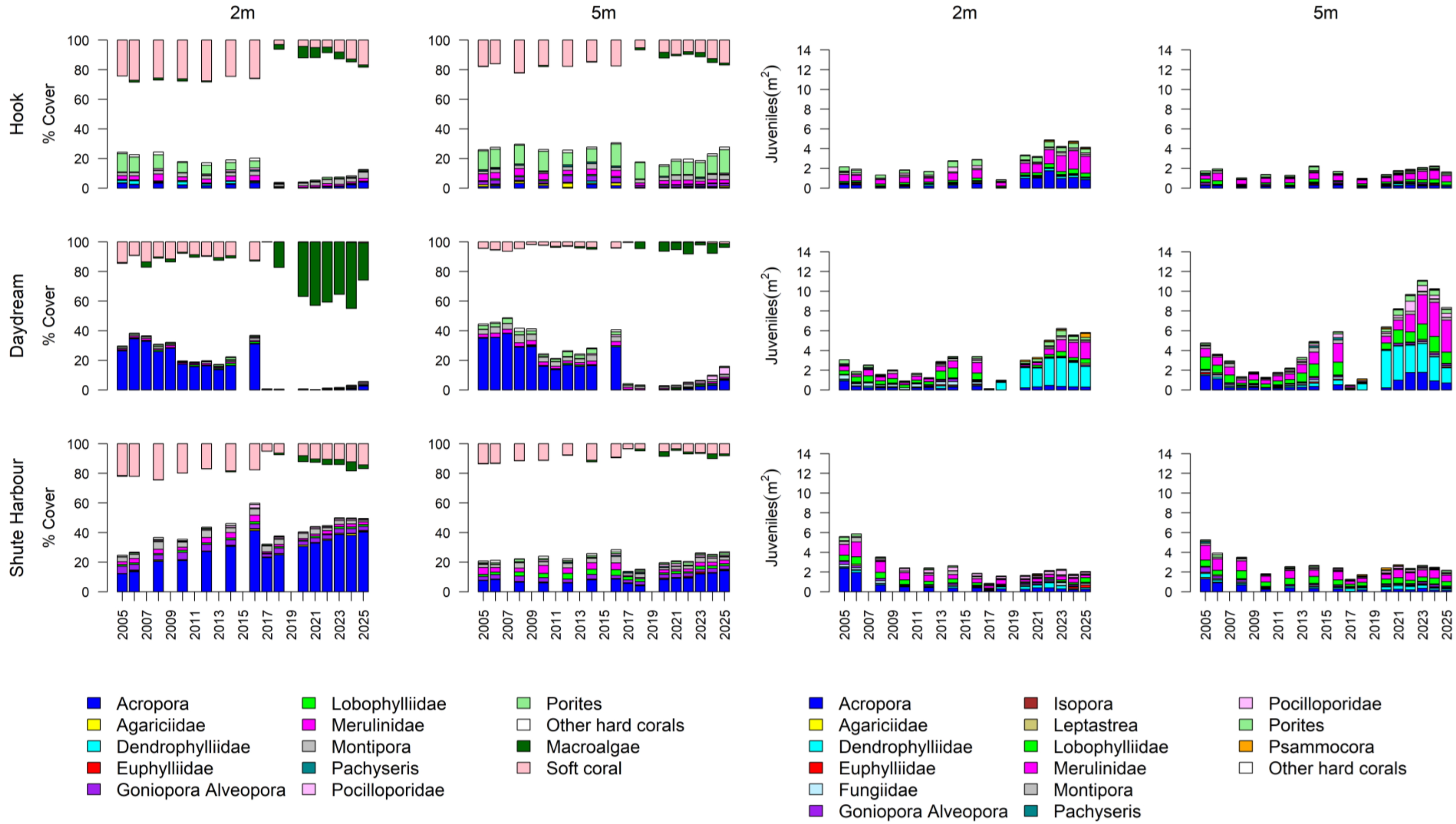



Figure A5. Mackay–Whitsunday region benthic community composition. Cover estimates are separated into regionally abundant hard coral groups and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral groups. Separate legends with relevant groupings for cover and juvenile density estimates are located beneath the respective plots. Juvenile density estimates are based on the number of juveniles recorded per square metre of transect.

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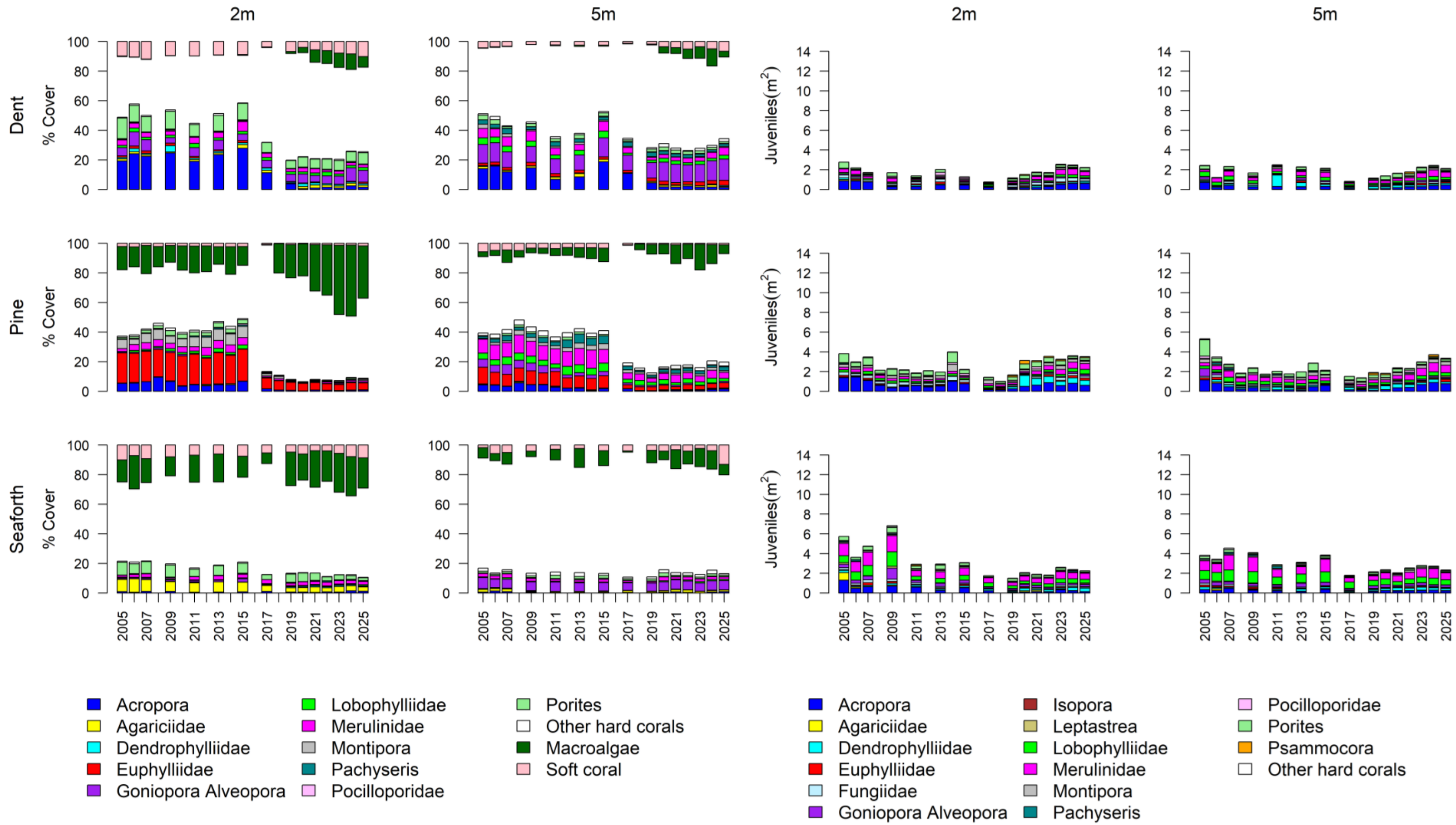


Figure A5 (continued).

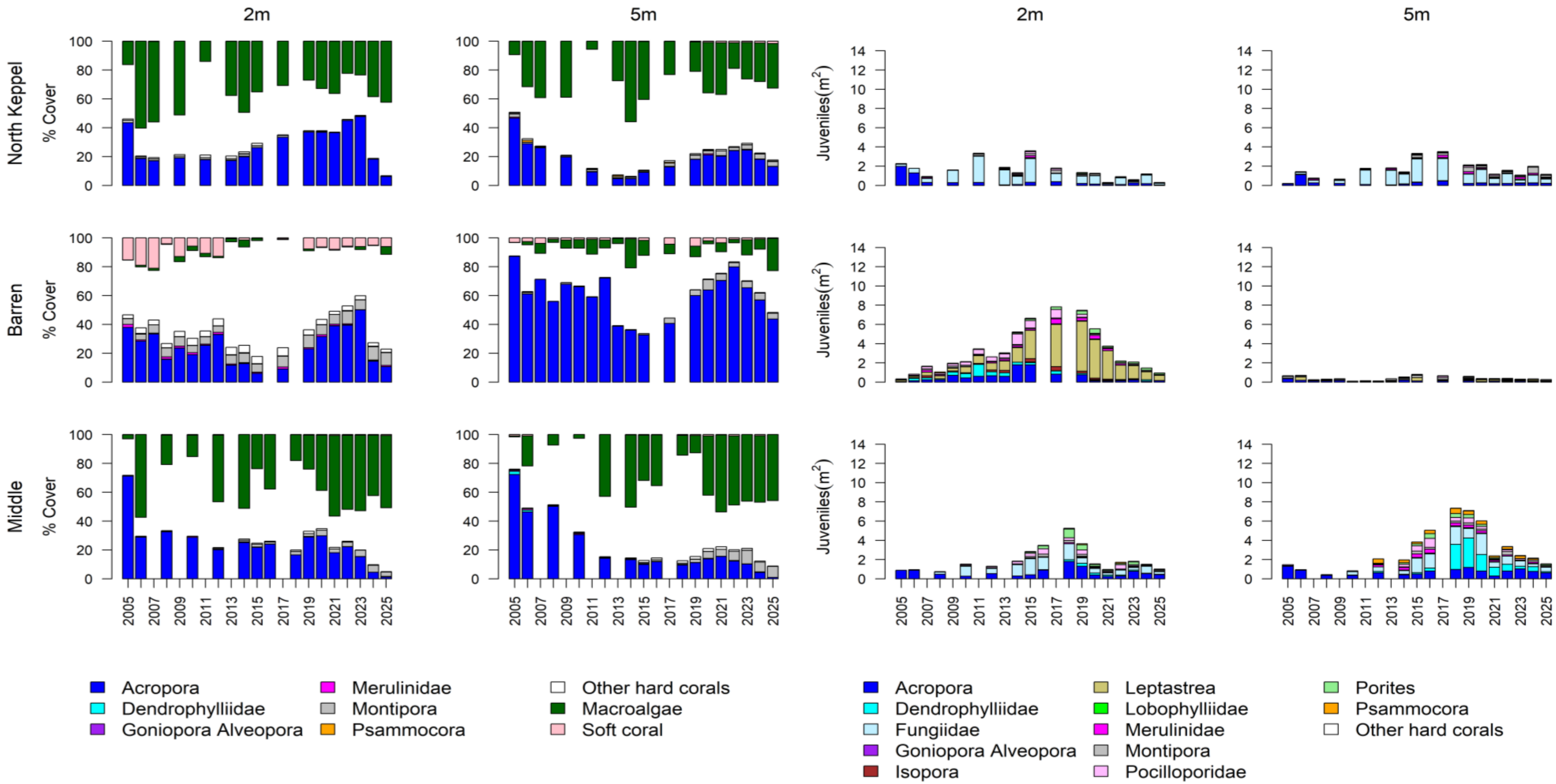


Figure A6. Fitzroy region benthic community composition. Cover estimates are separated into regionally abundant hard coral groups and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral groups. Separate legends with relevant groupings for cover and juvenile density estimates are located beneath the respective plots. Juvenile density estimates are based on the number of juveniles recorded per square metre of transect.

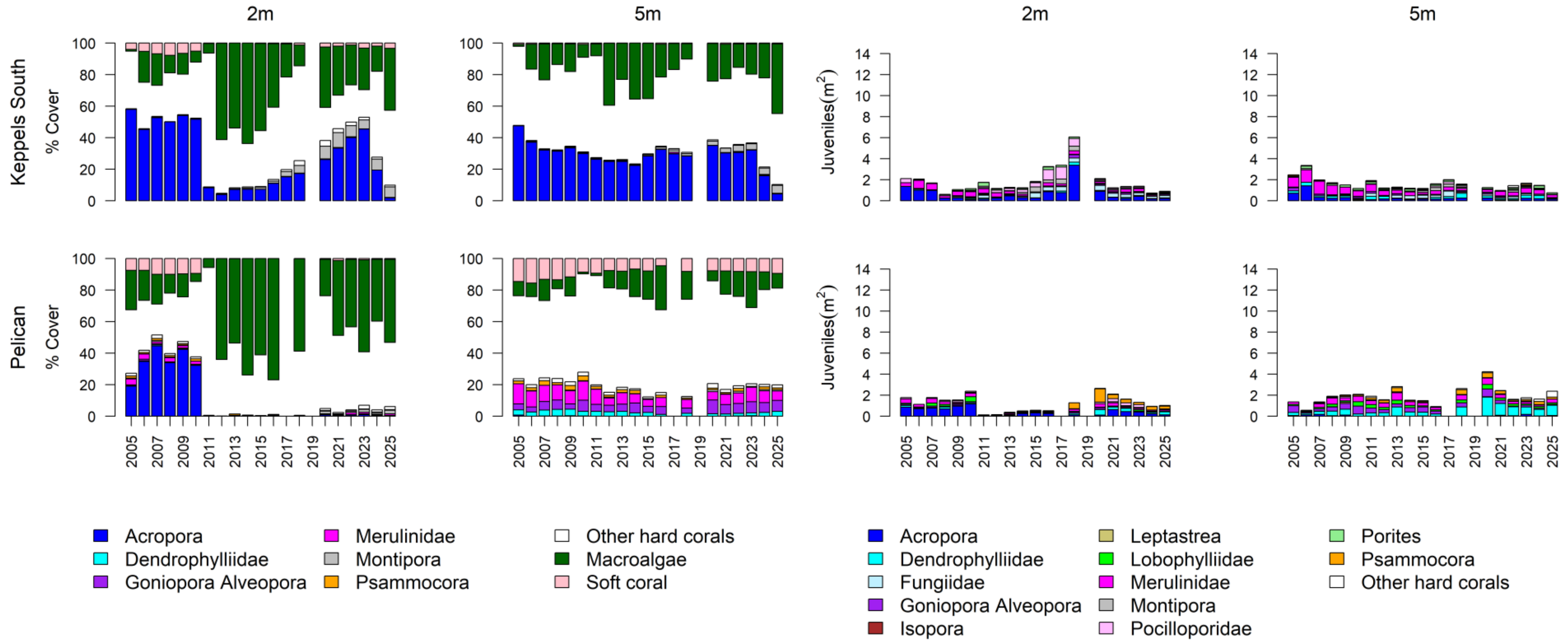


Figure A6 (continued).

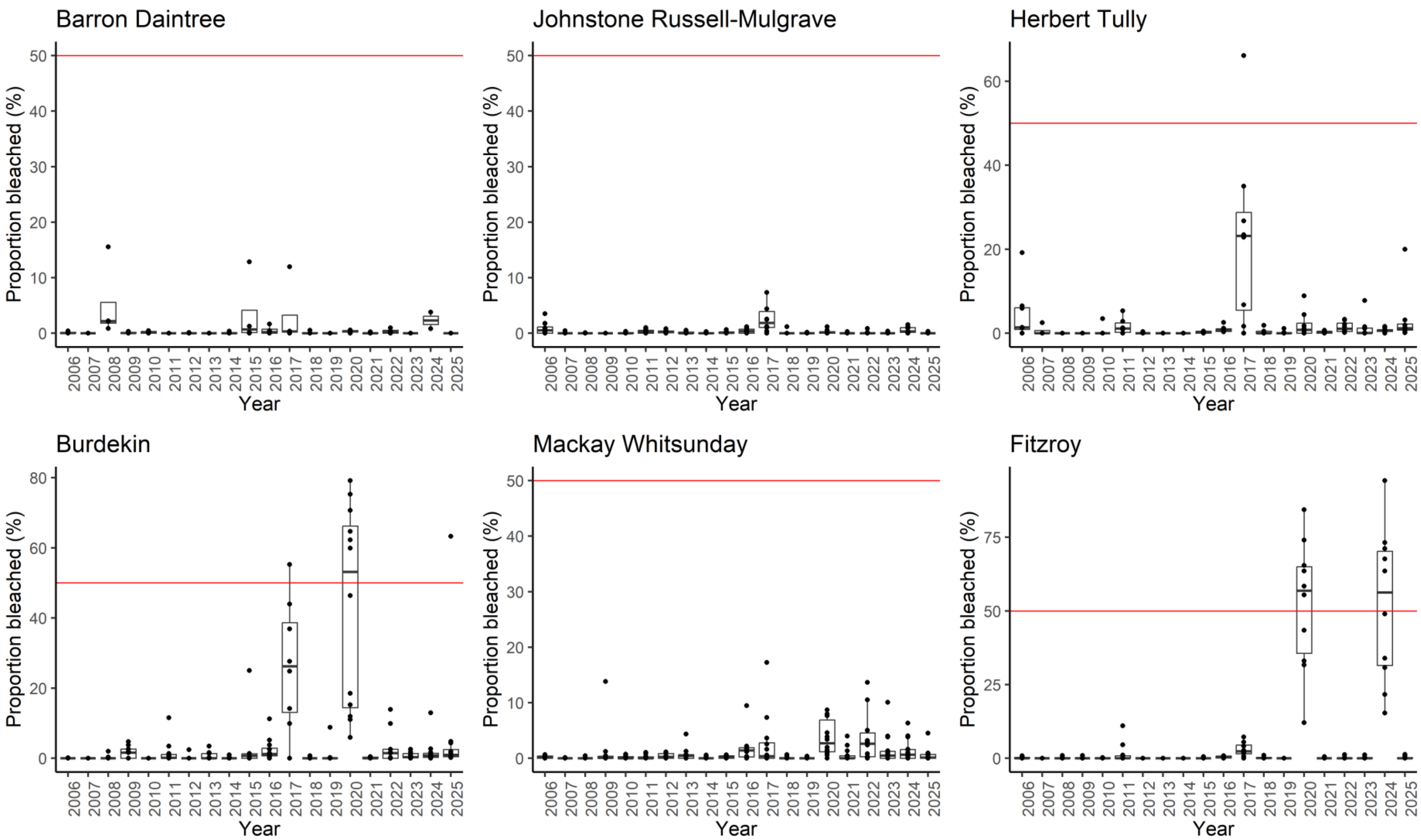


Figure A7. Proportion of hard coral bleached in each sub-region at the time of surveys. Boxplots include the proportion hard coral points from photo transects categorised as being “bleached” or “partially bleached” for each reef, depth within each sub-region ad year. A reference line at 50% bleached is provided to help readers compare among the (sub-)regions as the range of the y-axis varies.

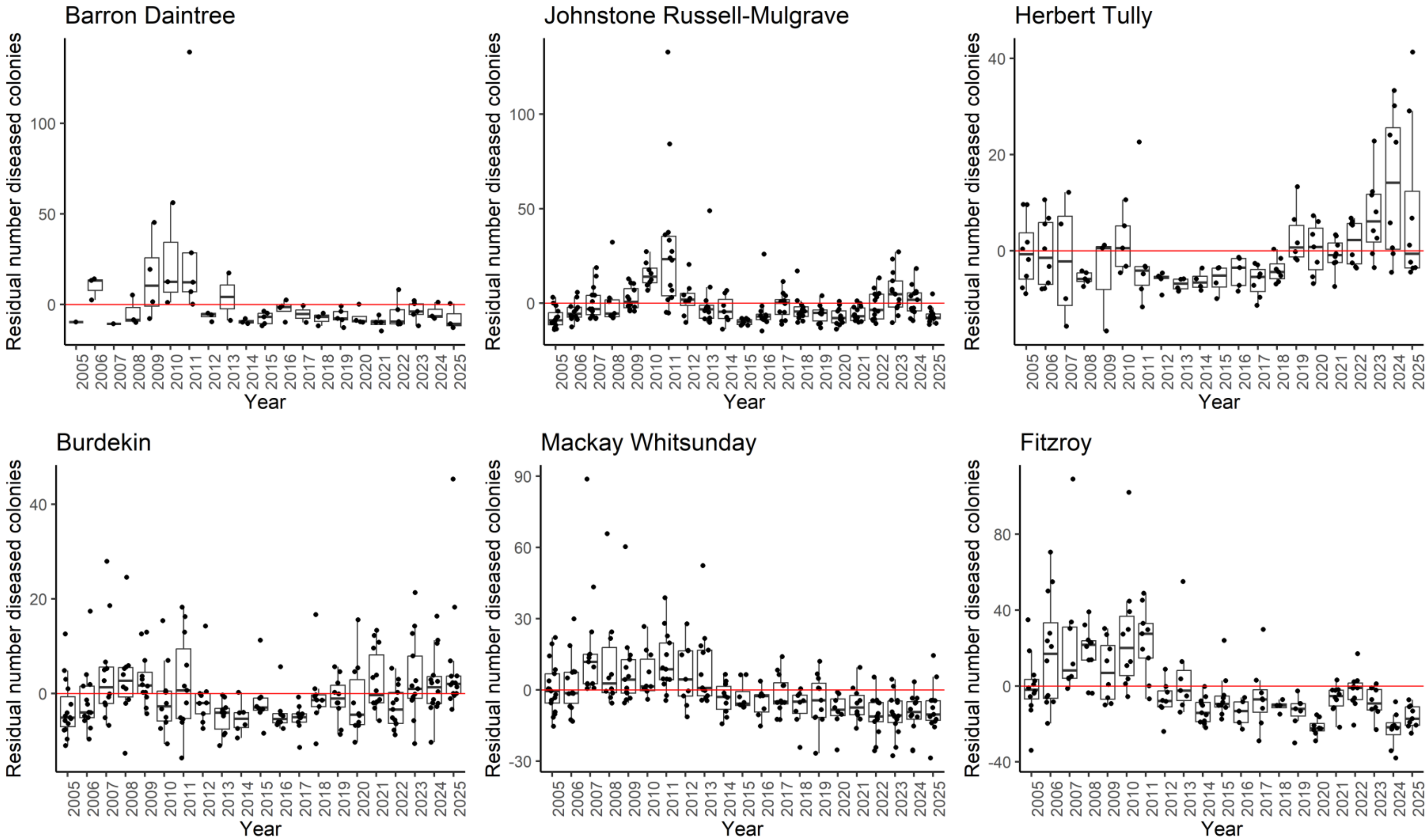


Figure A8. Coral disease by year in each region. Boxplots include the number of coral colonies suffering ongoing mortality attributed to either disease, sedimentation or 'unknown causes' for each reef, depth and year. Data are standardised to the reef and depth mean across years (see section 2.3.3). A red reference line is included to help identify years where disease levels differed from the standardised running mean.

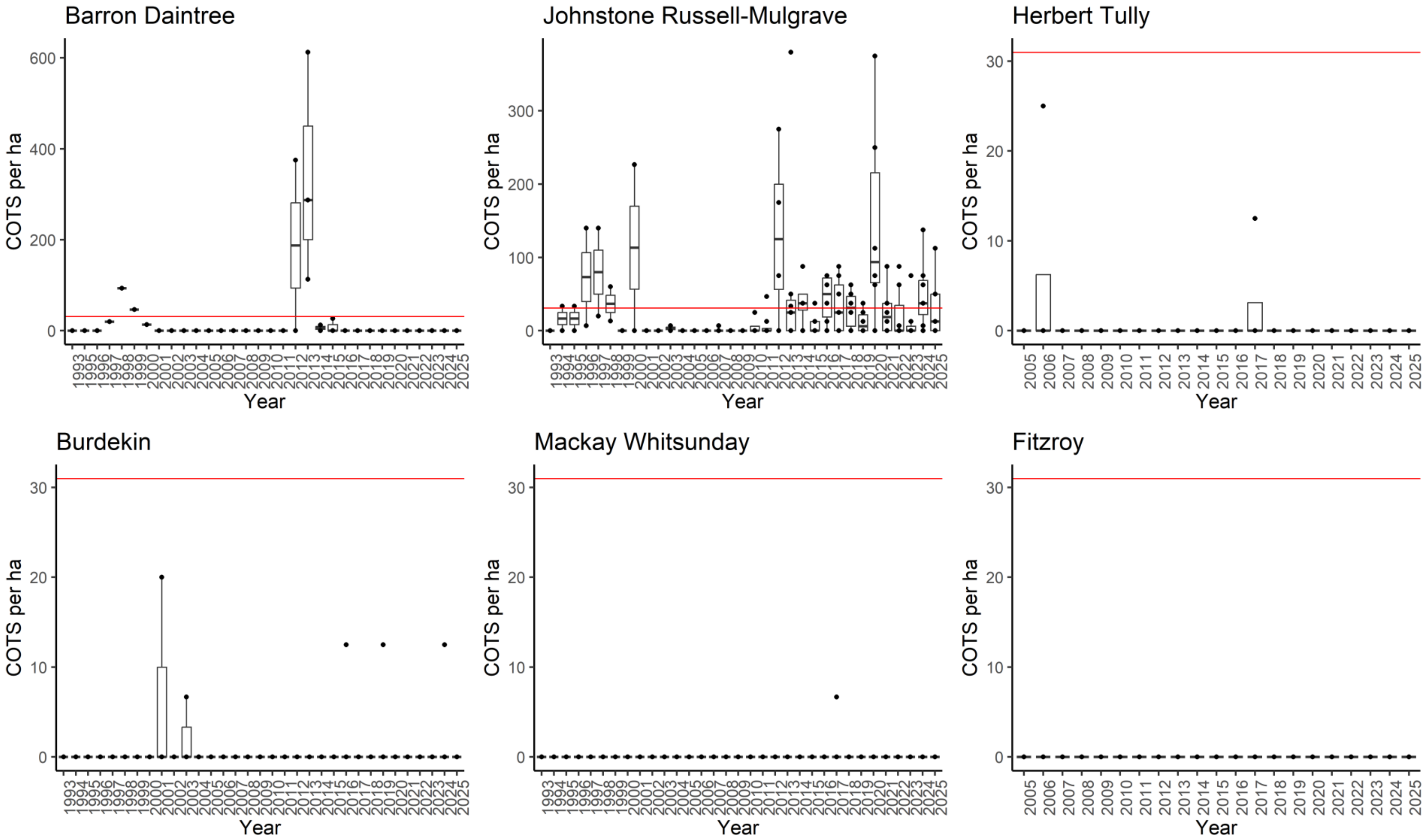


Figure A9. Crown-of-thorn-starfish mean density (individuals/ha) by year in each region. Red line indicates outbreak densities of 31 individuals per hectare (see section 2.3.3 for derivation).

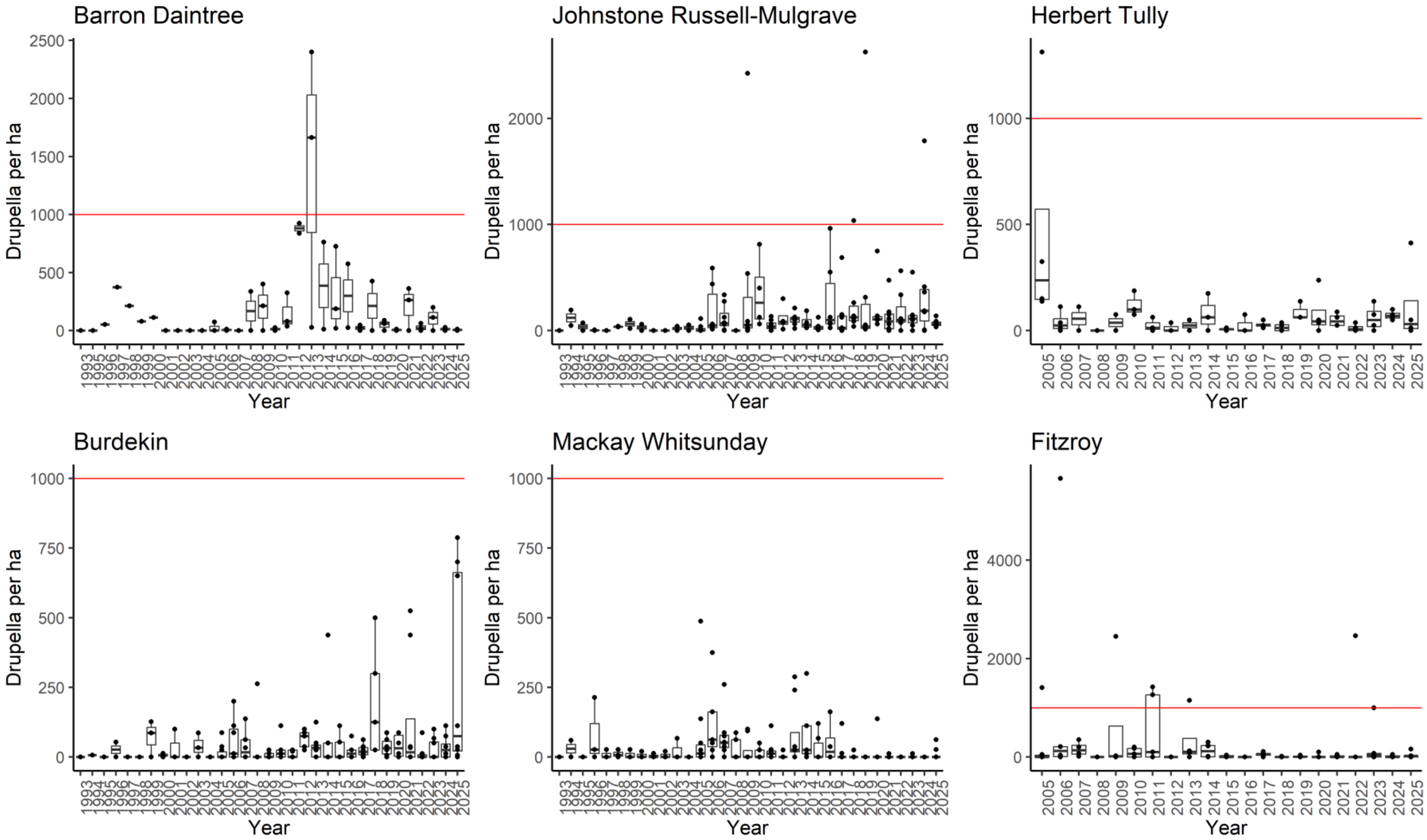


Figure A10. Mean density of *Drupella* by year in each (sub-)region. Red line indicates densities of *Drupella* which have detrimental impact on coral communities (see section 2.3.3 for derivation).

Table A9. Percent cover of hard coral genera 2025. Genera for which cover did not exceed 1% on at least one reef-depth or were unidentified to genus level are grouped as “Rare genera”.

| | Reef | Depth | *Acropora* | *Cyphastrea* | *Dipsastraea* | *Echinopora* | *Favites* | *Galaxea* | *Goniastrea* | *Goniopora* | *Isopora* | *Lobophyllia* | *Merulina* | *Montipora* | *Mycedium* | *Oxypora* | *Pachyseris* | *Pavona* | *Pocillopora* | *Porites* | *Turbinaria* | Rare genera |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Barron Daintree | Low Isles | 5 | 1.9 | 0 | 0.3 | 1 | 0.2 | 2.4 | 0 | 0.2 | 0 | 1.5 | 0.4 | 1.1 | 0.3 | 0.3 | 1.5 | 0 | 0.2 | 16.3 | 0 | 3.4 |
| | Snapper North | 2 | 0.9 | 0 | 0 | 2.8 | 0 | 0 | 0 | 0.1 | 0 | 0 | 0 | 0.7 | 0 | 0 | 0 | 0 | 0 | 1.8 | 0 | 0.6 |
| | | 5 | 7.9 | 0.1 | 0.1 | 0 | 0 | 0.5 | 0 | 8.3 | 0.6 | 0.1 | 0.2 | 3.9 | 0.3 | 0.1 | 6.9 | 1.1 | 0 | 6 | 0 | 2.4 |
| | Snapper South | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Johnstone Russell-Mulgrave | Fitzroy East | 2 | 8.3 | 0.1 | 0.1 | 0 | 0.1 | 0 | 0.6 | 0 | 0 | 0.2 | 0 | 7.3 | 0 | 0 | 0 | 0.2 | 5.5 | 6.5 | 0 | 0.9 |
| | | 5 | 11.6 | 0 | 0.5 | 2.1 | 0.2 | 0.8 | 0.2 | 0.2 | 0 | 1.3 | 0.2 | 0.9 | 0 | 0.2 | 0 | 0.2 | 5.4 | 11.3 | 0 | 3.2 |
| | Fitzroy West | 2 | 12.9 | 0 | 0.2 | 1.1 | 0 | 0.3 | 0.1 | 0.1 | 0 | 0.9 | 0 | 4.7 | 0 | 0 | 0 | 0.1 | 0.7 | 5.7 | 0 | 1.3 |
| | | 5 | 9.4 | 0 | 0 | 0.5 | 0.1 | 0.4 | 0 | 0.8 | 0 | 1.2 | 0.4 | 5.3 | 0.2 | 0.4 | 0.8 | 0 | 0.6 | 13.8 | 0 | 2.9 |
| | Fitzroy West | 5 | 5.1 | 0 | 0.3 | 0.4 | 0.1 | 1 | 0.3 | 0.8 | 0 | 2 | 0.3 | 4.7 | 0.6 | 0.9 | 2 | 0.2 | 0.5 | 16.5 | 0 | 3.6 |
| | Franklands East | 2 | 18.9 | 0.1 | 0 | 1.2 | 0.1 | 0.2 | 0 | 0.1 | 1.1 | 0.1 | 0.1 | 22.5 | 0 | 0 | 0 | 0 | 0.8 | 0.7 | 0.1 | 0.2 |
| | | 5 | 10.5 | 0 | 0.2 | 0 | 0.1 | 0.3 | 0.1 | 0.3 | 0 | 0.2 | 0.1 | 4 | 0 | 0 | 0.2 | 0 | 0.4 | 4.9 | 0 | 0.8 |
| | Franklands West | 2 | 2.8 | 0 | 0 | 0.7 | 0 | 0.1 | 0 | 0.7 | 0 | 0 | 0 | 1 | 0 | 0 | 4.8 | 0.1 | 0.3 | 27.7 | 0 | 0.6 |
| | | 5 | 0.2 | 0 | 0 | 0.4 | 0 | 0 | 0 | 0.2 | 0 | 0 | 0 | 0 | 0 | 0 | 1.9 | 0 | 0.1 | 47.7 | 0 | 0.2 |
| | High East | 2 | 4.4 | 0 | 0.2 | 0.6 | 0.4 | 0.2 | 0.3 | 0 | 0 | 0.9 | 0 | 2.8 | 0 | 0 | 0 | 0.4 | 0.3 | 5.4 | 0.1 | 0.7 |
| | | 5 | 5 | 0 | 0.1 | 1.9 | 0.1 | 0.1 | 0 | 0.3 | 0 | 0.2 | 0 | 3.8 | 0 | 0 | 0.1 | 0.4 | 0.8 | 16.4 | 0.1 | 1.2 |
| | High West | 2 | 0.4 | 0 | 0.5 | 0.5 | 0.2 | 0.4 | 0.1 | 2.3 | 0 | 0.1 | 0.2 | 0.8 | 0 | 0.1 | 0.1 | 0.3 | 0.6 | 42.1 | 0 | 1.5 |
| | | 5 | 0.1 | 0.1 | 0.7 | 0.1 | 0.1 | 0.4 | 0 | 3.5 | 0 | 0 | 0 | 0 | 0 | 0 | 0.1 | 0.3 | 0.1 | 14.1 | 0 | 1.2 |
| Tully Herbert | Barnards | 2 | 32.8 | 0 | 0.1 | 0 | 0.1 | 0 | 0.1 | 0 | 0 | 0 | 0 | 16.5 | 0 | 0 | 0 | 0 | 0.1 | 0.2 | 0.3 | 0.1 |
| | | 5 | 17.6 | 0.2 | 0.2 | 0.3 | 0.1 | 0.1 | 0 | 0.1 | 0 | 0.1 | 0.2 | 29.3 | 0.2 | 0.6 | 0 | 0.1 | 0.4 | 0.4 | 1 | 0.4 |
| | Dunk North | 2 | 0.4 | 1.2 | 0.4 | 0 | 0.3 | 0.1 | 0 | 0.5 | 0 | 0.1 | 0 | 5.5 | 0 | 0 | 0 | 0 | 0.1 | 0.5 | 6.3 | 0.9 |
| | | 5 | 7.6 | 0.8 | 0.6 | 0 | 1.1 | 0 | 0 | 0.3 | 0 | 0.2 | 0.2 | 8.4 | 0.3 | 0.1 | 0 | 0 | 0.8 | 0.9 | 12 | 3.4 |
| | Dunk South | 2 | 0.2 | 1.7 | 0.2 | 0 | 0.4 | 0.8 | 0.1 | 0 | 0 | 0.1 | 0 | 4.8 | 0 | 0 | 0 | 0.6 | 0 | 3.6 | 0.6 | 0.9 |
| | | 5 | 0.7 | 1.1 | 2.1 | 0.4 | 1.4 | 0.1 | 2 | 0.6 | 0 | 0.4 | 2.8 | 3.9 | 3 | 1.1 | 6 | 0.5 | 0.3 | 2 | 4.9 | 3.8 |
| | Bedarra | 2 | 0 | 0.4 | 0.1 | 0 | 0.1 | 0.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.6 | 0 | 2.6 | 0 | 0.2 |
| | | 5 | 0.1 | 0.2 | 3 | 0 | 0.8 | 0.1 | 0.1 | 1.7 | 0 | 0.9 | 0 | 0.1 | 0.4 | 0.1 | 0.4 | 0.1 | 0 | 2.8 | 0.1 | 2.1 |

Table A9 (continued).

| | Reef | Depth | *Acropora* | *Diploastrea* | *Dipsastraea* | *Echinopora* | *Favites* | *Galaxea* | *Goniopora* | *Isopora* | *Leptastrea* | *Lobophyllia* | *Merulina* | *Montipora* | *Mycedium* | *Oxypora* | *Pachyseris* | *Pavona* | *Pectinia* | *Platygyra* | *Pocillopora* | *Podabacia* | *Porites* | *Seriatopora* | *Turbinaria* | Rare genera |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Burdekin | Palms East | 2 | 22 | 0.1 | 0.2 | 0 | 0.5 | 0 | 0 | 0 | 0.1 | 0 | 0 | 1.6 | 0 | 0 | 0 | 0 | 0 | 0.1 | 0.1 | 0 | 0.6 | 0 | 0 | 0.4 |
| | | 5 | 26.6 | 0 | 0.2 | 0.4 | 0.6 | 0.3 | 0.1 | 0 | 0 | 0.2 | 0 | 2.5 | 0 | 0.1 | 0 | 0 | 0 | 0.5 | 0.6 | 0 | 2.3 | 0 | 0 | 0.8 |
| | Palms West | 2 | 0.2 | 0 | 0 | 0.2 | 0.2 | 0 | 0.1 | 0 | 0.1 | 0 | 0 | 0.4 | 0 | 0 | 0 | 0.1 | 0 | 0 | 5.1 | 0 | 0.2 | 0 | 0 | 0.2 |
| | | 5 | 1.7 | 0 | 0.3 | 0.1 | 0.1 | 0.1 | 1.2 | 0 | 0.1 | 0 | 0 | 0.9 | 0 | 0.1 | 0.1 | 0.1 | 0 | 0.1 | 1.3 | 0 | 4 | 0 | 0 | 1.1 |
| | Havannah | 5 | 4 | 0 | 0.2 | 0.2 | 0.1 | 0.9 | 0.1 | 0 | 0.1 | 0.1 | 0.5 | 6.1 | 0 | 0.3 | 0.2 | 0 | 0.1 | 0 | 0 | 0 | 1.5 | 0 | 0.4 | 2.3 |
| | Havannah | 2 | 7.8 | 0 | 0 | 0.6 | 0.1 | 0.7 | 0.4 | 5.4 | 0.1 | 0.2 | 0.1 | 7.4 | 0 | 0 | 0 | 0.1 | 0 | 0.2 | 0.8 | 0 | 1.9 | 0 | 0.5 | 1.1 |
| | | 5 | 7 | 0.2 | 0.4 | 1.3 | 0.2 | 0.5 | 0.2 | 0.5 | 0.1 | 0.4 | 3.8 | 5 | 0.2 | 0.9 | 1.4 | 0.2 | 1.2 | 0 | 0.1 | 0.6 | 0.4 | 0 | 2.4 | 4.4 |
| | Pandora | 2 | 4.7 | 0 | 0 | 0 | 0.5 | 0 | 0.2 | 0 | 1.3 | 0 | 0 | 5.7 | 0 | 0 | 0 | 0.1 | 0 | 0 | 0.1 | 0 | 3 | 0 | 0.2 | 2.4 |
| | | 5 | 4 | 3.3 | 1.4 | 0.2 | 0.6 | 0.3 | 0.4 | 0 | 0.5 | 0 | 0.4 | 3.1 | 0 | 0 | 0.1 | 0 | 0 | 0.1 | 0.1 | 0 | 0.2 | 0 | 0.2 | 2.7 |
| | Pandora | 5 | 1.1 | 0 | 0 | 1.5 | 0.2 | 1.6 | 16.4 | 0 | 0 | 0.5 | 1.4 | 0.5 | 1.1 | 0.8 | 3.9 | 0.3 | 0.6 | 0.1 | 0.1 | 0.4 | 4.4 | 0 | 5.8 | 3.3 |
| | Lady Elliot | 2 | 0 | 0 | 0 | 0 | 0 | 0.5 | 0 | 0 | 0 | 0 | 0 | 0.4 | 0 | 0 | 0 | 1.3 | 0 | 0 | 0 | 0 | 0.6 | 0 | 0.1 | 0 |
| | | 5 | 0.4 | 0 | 1.1 | 0 | 0.2 | 12.6 | 1.3 | 0 | 0 | 0.9 | 0 | 0.6 | 1 | 0.1 | 0.8 | 0 | 0.4 | 0.1 | 0 | 0.9 | 3 | 0 | 6.4 | 1.6 |
| | Magnetic | 2 | 3.6 | 0 | 0.3 | 0.1 | 0.7 | 0.5 | 0.1 | 0.1 | 0 | 0 | 0 | 9 | 0 | 0 | 0.7 | 0.1 | 0 | 0.1 | 0 | 0 | 1.6 | 0 | 1.3 | 1.6 |
| | | 5 | 2.5 | 0 | 2.1 | 0.2 | 1.1 | 0.2 | 3.7 | 0 | 0 | 0.3 | 2.5 | 1.9 | 0.1 | 0.3 | 1.1 | 0.1 | 0.2 | 1.6 | 0 | 1.7 | 0.6 | 0 | 2.1 | 2.4 |

Table A9 (continued).

| | Reef | Depth | *Acropora* | *Diploastrea* | *Dipsastraea* | *Echinopora* | *Favites* | *Galaxea* | *Goniopora* | *Isopora* | *Leptastrea* | *Lobophyllia* | *Merulina* | *Montipora* | *Mycedium* | *Oxypora* | *Pachyseris* | *Pavona* | *Pectinia* | *Platygyra* | *Pocillopora* | *Podabacia* | *Porites* | *Seriatopora* | *Turbinaria* | Rare genera |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Mackay Whitsunday | Hayman | 5 | 7.7 | 2.3 | 0.3 | 0.3 | 0.4 | 0.2 | 0.1 | 0 | 0 | 0.6 | 0 | 1.4 | 0 | 0.1 | 0.4 | 0.1 | 0 | 0.2 | 0.3 | 0 | 0.7 | 0.2 | 0.1 | 1.5 |
| | Hook | 2 | 1.4 | 0.1 | 0.6 | 0.1 | 0.2 | 0.3 | 9.6 | 0 | 0 | 0.2 | 0.1 | 0.6 | 0.1 | 0.2 | 0.5 | 0.3 | 0.4 | 0.2 | 0.2 | 0 | 2.4 | 0 | 0 | 1.2 |
| | | 5 | 3.9 | 0.1 | 0.3 | 0 | 0.6 | 0 | 0.3 | 0 | 0.1 | 0.3 | 0.1 | 4.1 | 0 | 0 | 0.4 | 0.1 | 0.1 | 0.1 | 0.3 | 0 | 0.7 | 0 | 0 | 1 |
| | Double Cone | 2 | 0.5 | 0.5 | 1.2 | 0 | 0.6 | 0.1 | 1.6 | 0 | 0.2 | 0.2 | 0 | 3.2 | 0 | 0 | 0.6 | 0.8 | 0 | 0.1 | 0.6 | 0 | 15.8 | 0 | 0 | 1.9 |
| | | 5 | 1.5 | 0 | 0.1 | 0.3 | 0.1 | 0.1 | 0.1 | 0 | 0 | 0.1 | 0.2 | 0.2 | 0 | 0 | 0.2 | 0 | 0 | 0 | 0.4 | 0 | 0.5 | 0 | 0.3 | 0.4 |
| | Daydream | 2 | 0 | 0 | 0 | 0 | 0.1 | 0.7 | 15.8 | 0 | 0 | 1.1 | 0.3 | 0.4 | 0 | 0.1 | 1 | 0.1 | 0.2 | 0.1 | 0 | 0.1 | 0.9 | 0 | 0 | 0.4 |
| | | 5 | 2.6 | 0 | 0.1 | 0 | 0 | 0 | 0.1 | 0 | 0.1 | 0.1 | 0 | 0.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0.1 | 0 | 0.3 | 0.6 | 0.4 | 0.5 |
| | Dent | 2 | 6.4 | 0 | 0.1 | 0 | 0.3 | 0 | 0 | 0.1 | 0.1 | 0.1 | 0.1 | 1.6 | 0.1 | 0.4 | 0.2 | 0 | 0.2 | 0 | 0 | 0 | 0.8 | 4.3 | 0.2 | 1.1 |
| | | 5 | 1.4 | 0 | 0 | 0.2 | 0.1 | 0.9 | 8 | 0 | 0 | 1.8 | 0.2 | 0.2 | 0 | 0.1 | 0.1 | 1.3 | 0.6 | 0 | 0.1 | 0.1 | 7.6 | 0 | 1.1 | 1.8 |
| | Shute Harbour | 2 | 1.3 | 0 | 0.6 | 0.8 | 0.1 | 3 | 14.2 | 0 | 0 | 1.3 | 0.6 | 0.7 | 0.1 | 1.4 | 1.9 | 0.7 | 1.7 | 0.6 | 0.3 | 0.3 | 1.2 | 0 | 0.7 | 2.9 |
| | | 5 | 40.3 | 0 | 0.1 | 0.3 | 0 | 0.1 | 2.8 | 0.2 | 0.1 | 0.9 | 0.1 | 1.1 | 0.1 | 0.2 | 0.1 | 0.6 | 0.3 | 0.1 | 0.1 | 0 | 0.2 | 0 | 0 | 2 |
| | Pine | 2 | 14.2 | 0 | 0.1 | 0.2 | 0.3 | 0.3 | 2.2 | 0.2 | 0.1 | 1.2 | 0.3 | 2 | 0.6 | 0.1 | 0.6 | 0.4 | 0.5 | 0.1 | 0.2 | 0.3 | 1.4 | 0 | 0.1 | 1.5 |
| | | 5 | 0.8 | 0 | 0 | 0 | 0.1 | 4.6 | 0.1 | 0 | 0 | 0.2 | 0.1 | 0.8 | 0.1 | 0.1 | 0.2 | 0 | 0.6 | 0 | 0.2 | 0.1 | 0.1 | 0 | 0.1 | 0.8 |
| | Seaforth | 2 | 1.5 | 0 | 0.6 | 0.1 | 0.1 | 3.5 | 1 | 0 | 0 | 1.2 | 0.4 | 1.1 | 0.2 | 0.7 | 2.7 | 0 | 2.1 | 0.3 | 0 | 1.2 | 0.6 | 0 | 0.2 | 2.2 |
| | | 5 | 1.2 | 0 | 0.5 | 0.4 | 0 | 0.1 | 0.5 | 0 | 0.1 | 0.7 | 0.2 | 0 | 0 | 0.1 | 0.6 | 2.8 | 0.2 | 0 | 0.1 | 0 | 1.9 | 0 | 0 | 1.4 |

Table A9 (continued).

| | Reef | Depth | *Acropora* | *Alveopora* | *Favites* | *Isopora* | *Montipora* | *Paragoniastrea* | *Pocillopora* | *Psammocora* | *Turbinaria* | *Rare genera* |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Fitzroy | Barren | 2 | 10.7 | 0 | 0 | 1.2 | 8.9 | 0 | 0.1 | 0.3 | 0 | 1.5 |
| | Barren | 5 | 43.7 | 0 | 0 | 0 | 4 | 0 | 0.2 | 0 | 0 | 0.4 |
| | North Keppel | 2 | 6.1 | 0 | 0 | 0 | 0.5 | 0 | 0 | 0.1 | 0 | 0 |
| | North Keppel | 5 | 13.1 | 0 | 0 | 0 | 3.2 | 0 | 0.4 | 0.4 | 0 | 0.5 |
| | Middle | 2 | 1.2 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0.6 |
| | Middle | 5 | 1 | 0 | 0 | 0 | 7.5 | 0 | 0.2 | 0.1 | 0 | 0 |
| | Keppels | 2 | 1.8 | 0 | 0 | 0 | 6.6 | 0 | 1.1 | 0 | 0 | 0.4 |
| | Keppels | 5 | 4.3 | 0 | 0 | 0 | 5 | 0 | 0.2 | 0 | 0.3 | 0.5 |
| | Pelican | 2 | 0.3 | 0.1 | 0.2 | 0 | 2.2 | 0.1 | 0.8 | 0.4 | 0 | 2.1 |
| | Pelican | 5 | 0 | 5.8 | 2.5 | 0 | 0.8 | 1.7 | 0.1 | 1.6 | 2.6 | 4.9 |

Table A10. Percent cover of soft coral families 2025. Families for which cover did not exceed 0.25% on at least one reef or corals not identified to family level are grouped to 'Other SC'.

| (sub-)region | Reef | Depth | Briareidae | Cladiellidae | Clavulariidae | Helioporidae | Isididae | Nephtheidae | Sarcophytidae | Xeniidae | Other SC |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Barron Daintree | Low Isles | 5 | 12.2 | 0.1 | 0.1 | 0 | 0 | 0 | 1.1 | 0 | 0.3 |
| | Snapper North | 2 | 3.5 | 0 | 7.9 | 0 | 0 | 0 | 0.3 | 0 | 0 |
| | | 5 | 0.9 | 0 | 0.1 | 0 | 0 | 0 | 0.1 | 3.1 | 0 |
| | Snapper South | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Johnstone Russell-Mulgrave | Fitzroy East | 2 | 1.4 | 0 | 0.9 | 0 | 0 | 0 | 2.3 | 0 | 0 |
| | | 5 | 8.6 | 0 | 0.6 | 0 | 0 | 0 | 6.6 | 0 | 0.1 |
| | Fitzroy West | 2 | 0.1 | 0 | 0 | 0 | 0 | 0 | 27.6 | 0 | 0.1 |
| | | 5 | 0.1 | 0.1 | 0 | 0 | 0 | 0 | 26.6 | 0 | 0.6 |
| | Fitzroy West LTMP | 5 | 0.3 | 0.3 | 0 | 0 | 0 | 0 | 15.3 | 0 | 0.2 |
| | Franklands East | 2 | 0 | 0.1 | 0.4 | 0.7 | 0 | 0 | 1.4 | 0 | 0 |
| | | 5 | 0.6 | 0.1 | 0.6 | 0 | 0 | 0 | 2.9 | 0 | 0.1 |
| | Franklands West | 2 | 0 | 0 | 8.7 | 0 | 0 | 0 | 7.9 | 0 | 0 |
| | | 5 | 0 | 0 | 1.1 | 0 | 0 | 0 | 0.6 | 0 | 0 |
| | High East | 2 | 5.8 | 0.1 | 0 | 0 | 0 | 0 | 6.1 | 0 | 0 |
| | | 5 | 9.5 | 0 | 0 | 0 | 0 | 0 | 0.1 | 0 | 0 |
| | High West | 2 | 0.1 | 0 | 0 | 2.2 | 0 | 0 | 3.8 | 0 | 0 |
| | | 5 | 1.1 | 0 | 0 | 1.7 | 0 | 0 | 0.4 | 0 | 0 |
| Herbert Tully | Barnards | 2 | 2.6 | 0.4 | 0.4 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 3.4 | 0.2 | 0 | 0 | 0 | 0.1 | 0.1 | 0 | 0.2 |
| | Dunk North | 2 | 0.3 | 0 | 0.1 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0.4 | 0.6 | 0 | 0 | 0 | 0 | 0.2 | 0.1 | 0.9 |
| | Dunk South | 2 | 0.6 | 0 | 0.3 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 2.3 | 0.1 | 0 | 0 | 0 | 0 | 0.1 | 0 | 0 |
| | Bedarra | 2 | 0.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 4.3 | 0 | 0 | 0 | 0 | 0 | 0.2 | 0 | 0 |
| Burdekin | Palms East | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 2.4 | 0 | 0 |
| | | 5 | 0 | 0.5 | 0 | 0 | 0 | 0 | 2.1 | 0 | 0 |
| | Palms West | 2 | 0.9 | 0.1 | 1.1 | 0 | 0 | 0.3 | 6.6 | 0 | 0 |
| | | 5 | 5.6 | 0.2 | 1.1 | 0 | 0 | 2.1 | 16.9 | 0 | 0.1 |
| | Havannah North | 5 | 1.3 | 0.1 | 3.3 | 0 | 0 | 0 | 0 | 0.2 | 0 |
| | Havannah | 2 | 4.4 | 0 | 0.1 | 0 | 0 | 0.2 | 0.1 | 0 | 0 |
| | | 5 | 9.1 | 0 | 0 | 0 | 0 | 0 | 0.1 | 0 | 0 |
| | Pandora | 2 | 0 | 0.2 | 0 | 0 | 0 | 0 | 0.3 | 0 | 0 |
| | | 5 | 0 | 0.2 | 0.4 | 0 | 0 | 0 | 0.8 | 0 | 0 |
| | Pandora North | 5 | 9 | 0 | 7.8 | 0 | 0 | 0 | 0.9 | 0 | 0.1 |
| | Lady Elliot | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0.1 | 0.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0.1 |
| | Magnetic | 2 | 0.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0.1 | 0 | 0 | 0 | 0 | 0.1 | 0.6 | 0 | 0.1 |

Table A10 (continued).

| | Reef | Depth | Briareidae | Cladiellidae | Clavulariidae | Helioporidae | Isididae | Nephtheidae | Sarcophytidae | Xeniidae | Other SC |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Mackay Whitsunday | Hayman | 5 | 0.5 | 2.2 | 0 | 0 | 0 | 0 | 6.3 | 0.2 | 0 |
| | Border | 5 | 0.2 | 3.3 | 0 | 0 | 0.1 | 0.7 | 20.3 | 0.9 | 0.4 |
| | Hook | 2 | 0.2 | 7.4 | 0 | 0 | 0 | 0 | 8.9 | 0 | 0.3 |
| | | 5 | 1.8 | 3.1 | 0 | 0 | 0 | 0.2 | 10.1 | 0 | 0.3 |
| | Double cone | 2 | 0.4 | 0.5 | 0 | 0 | 0 | 0 | 0.3 | 0 | 0 |
| | | 5 | 1.4 | 0.4 | 0 | 0 | 0 | 0 | 0.8 | 0 | 0.1 |
| | Daydream | 2 | 0 | 0.2 | 0 | 0 | 0 | 0 | 0.4 | 0 | 0 |
| | | 5 | 0 | 0.6 | 0 | 0 | 0 | 0 | 0.5 | 0 | 0.1 |
| | Dent | 2 | 5.2 | 1.7 | 0 | 0 | 0 | 0.6 | 2.8 | 0 | 0 |
| | | 5 | 1.1 | 1.9 | 0 | 0 | 0 | 0 | 3.6 | 0 | 0 |
| | Shute Harbour | 2 | 0.9 | 2.6 | 0 | 0 | 0 | 1.6 | 8.8 | 0.3 | 0 |
| | | 5 | 0.7 | 1.1 | 0 | 0 | 0 | 0.3 | 4.1 | 0.4 | 0.2 |
| | Pine | 2 | 0.2 | 0.2 | 0.1 | 0 | 0 | 0 | 1.1 | 0 | 0.1 |
| | | 5 | 0 | 0.3 | 0 | 0 | 0 | 0.2 | 0.4 | 0.1 | 0.1 |
| | Seaforth | 2 | 4.8 | 0.2 | 0 | 0 | 0 | 0 | 3.6 | 0 | 0 |
| | | 5 | 12.2 | 0.1 | 0 | 0 | 0 | 0 | 0.6 | 0.1 | 0 |
| Fitzroy | Barren | 2 | 0.4 | 4.1 | 0 | 0 | 0 | 0 | 0.9 | 0.7 | 0 |
| | | 5 | 0 | 0.3 | 0 | 0 | 0 | 0 | 0.1 | 0 | 0 |
| | North Keppel | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0 | 0.9 | 0 | 0 | 0 | 0 | 0.8 | 0 | 0 |
| | Middle | 2 | 0 | 0.4 | 0 | 0 | 0 | 0 | 0.2 | 0 | 0 |
| | | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Keppels South | 2 | 0 | 0.8 | 0 | 0 | 0 | 0 | 0.1 | 2.4 | 0.1 |
| | | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0.2 | 0.3 | 0 |
| | Pelican | 2 | 0.1 | 0.1 | 0 | 0 | 0 | 0 | 0.3 | 0 | 0.1 |
| | | 5 | 0.2 | 0.9 | 0 | 0 | 1.2 | 0 | 5.9 | 0 | 1.2 |

Table A11. Percent cover of macroalgae groups 2025. Genera for which cover exceeded 0.5% on at least one reef are included, rare or unidentified genera are grouped as “Undefined”

| Region | Reef | Depth | Rhodophyta (red algae) | | | | | | | Chlorophyta (green algae) | Phaeophyta (brown algae) | | | | Turf algae |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | *Amansia* | *Asparagopsis* | Crustose coralline | *Hypnea* | *Laurencia* | *Peyssonnelia* | Undefined | | *Dictyota* | *Lobophora* | Sargassaceae | Undefined | |
| Barron–Daintree | Low Isles | 5 | 0 | 0.03 | 1.31 | 0 | 0 | 0.2 | 0.44 | 0.1 | 0 | 0 | 0 | 0.17 | 44.5 |
| | Snapper North | 2 | 0 | 0.5 | 2.5 | 0.04 | 0 | 0.04 | 35.83 | 0.92 | 12.83 | 0 | 0 | 0.75 | 27.52 |
| | | 5 | 0 | 0.19 | 3.81 | 0 | 0 | 0.06 | 0.31 | 0 | 1.06 | 0 | 0 | 0 | 37.39 |
| | Snapper South | 2 | 0 | 0 | 0.46 | 0.08 | 0 | 0.21 | 0.62 | 0.08 | 0 | 0.08 | 0 | 0.33 | 93.88 |
| | | 5 | 0 | 7.57 | 3.69 | 0.31 | 0 | 0.38 | 17.76 | 0 | 1.19 | 1.31 | 0 | 0.5 | 63.29 |
| Johnstone Russell–Mulgrave | Fitzroy East | 2 | 0 | 0 | 1.25 | 0.19 | 0 | 0 | 1.81 | 0.12 | 0 | 0.19 | 0 | 0.06 | 54.44 |
| | | 5 | 0 | 0 | 2.25 | 0 | 0 | 0 | 1.06 | 0.06 | 0 | 0.06 | 0 | 0 | 37.12 |
| | Fitzroy West | 2 | 0.06 | 0 | 0.63 | 1.69 | 0 | 0.06 | 2.25 | 0 | 0 | 0 | 0 | 0 | 39.03 |
| | | 5 | 0 | 0 | 1.44 | 0.63 | 0 | 0.56 | 0.75 | 0.06 | 0 | 0 | 0 | 0 | 25.73 |
| | Fitzroy West LTMP | 5 | 0 | 0 | 1.37 | 0 | 0 | 0.03 | 0.73 | 0.03 | 0 | 0.03 | 0 | 0.34 | 30.88 |
| | Franklands East | 2 | 0 | 0.69 | 1.88 | 0.69 | 0 | 0 | 6.89 | 0.06 | 0.56 | 0.13 | 0 | 0 | 38.79 |
| | | 5 | 0 | 0.62 | 3.5 | 0.12 | 0 | 0.25 | 11.64 | 0 | 2.38 | 0 | 0 | 0.25 | 53.45 |
| | Franklands West | 2 | 0.38 | 0 | 1.31 | 4.06 | 1.94 | 0 | 12.75 | 0.19 | 2.94 | 0.06 | 0 | 0 | 20.88 |
| | | 5 | 0 | 0 | 4.07 | 0 | 7 | 0 | 17.82 | 1.13 | 0.88 | 0 | 0 | 0 | 16.51 |
| | High East | 2 | 1.19 | 0 | 1.31 | 0.62 | 0 | 0.12 | 3.38 | 0.06 | 11.12 | 0.12 | 0 | 0 | 49.25 |
| | | 5 | 2 | 0 | 3.62 | 1 | 0 | 0.25 | 4.38 | 0.06 | 1.75 | 0 | 0 | 0 | 42.75 |
| | High West | 2 | 0.06 | 0 | 3.81 | 2.06 | 0.25 | 0.12 | 3.7 | 0.06 | 0.06 | 0 | 0 | 0 | 32.22 |
| | | 5 | 0 | 0 | 2.38 | 0.19 | 0.25 | 0 | 2 | 0 | 0 | 0 | 0 | 0.06 | 54,69 |

Table A11 (continued).

| Region | Reef | Depth | Rhodophyta (red algae) | | | | | | | Chlorophyta (green algae) | Phaeophyta (brown algae) | | | | Turf algae |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | *Amansia* | *Asparagopsis* | Crustose coralline | *Hypnea* | *Laurencia* | *Peyssonnelia* | Undefined | | *Dictyota* | *Lobophora* | Sargassaceae | Undefined | |
| Tully-Herbert | Barnards | 2 | 0 | 0 | 1.13 | 3.63 | 0 | 0 | 0.81 | 0 | 0 | 0 | 0 | 0.06 | 31.13 |
| | | 5 | 0 | 0 | 1.62 | 0.19 | 0 | 0.19 | 0.88 | 0 | 0 | 0 | 0 | 0.06 | 24.07 |
| | Dunk North | 2 | 0.06 | 0 | 1.5 | 0 | 0 | 0.38 | 2.44 | 0 | 0 | 0.5 | 7.64 | 0 | 58.58 |
| | | 5 | 0 | 0 | 1.13 | 0 | 0 | 0.06 | 1.38 | 0 | 0 | 0.12 | 0.75 | 0.06 | 31.11 |
| | Dunk South | 2 | 0 | 0 | 2.69 | 0 | 0 | 0.12 | 3.06 | 0 | 0 | 0.94 | 3.56 | 0.25 | 62.81 |
| | | 5 | 0 | 0 | 3.88 | 0 | 0 | 0.06 | 3.31 | 0.06 | 0 | 3.88 | 0 | 0.12 | 35.38 |
| | Bedarra | 2 | 0 | 0 | 1.63 | 0.12 | 0 | 0.06 | 1.44 | 0 | 0.12 | 0.38 | 14.07 | 0.56 | 56.49 |
| | | 5 | 0 | 0 | 0.31 | 0 | 0 | 0.19 | 0.81 | 0 | 0.06 | 0.5 | 0.56 | 0.25 | 43.47 |
| Burdekin | Palms East | 2 | 0 | 0 | 2.19 | 0 | 0 | 0 | 0.25 | 0.12 | 0 | 0 | 0 | 0.62 | 58.07 |
| | | 5 | 0 | 0 | 0.62 | 0 | 0 | 0.19 | 0.69 | 0.06 | 0 | 0 | 0 | 0.06 | 53.31 |
| | Palms West | 2 | 0 | 0 | 0.06 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 49.12 |
| | | 5 | 0 | 0 | 1.06 | 0 | 0 | 0 | 0.38 | 0 | 0 | 0.12 | 0 | 0 | 41.59 |
| | Havannah North | 5 | 0 | 0 | 5.76 | 0 | 0 | 0.37 | 1.71 | 0.03 | 1.03 | 4.49 | 1.95 | 2.32 | 43.18 |
| | Havannah | 2 | 0 | 0 | 0.56 | 0 | 0 | 0.63 | 1.13 | 0.06 | 1.63 | 4.07 | 0.19 | 0.81 | 52.06 |
| | | 5 | 0 | 0 | 0.94 | 0 | 0 | 0.12 | 1 | 0 | 1.69 | 9.45 | 1 | 0.06 | 42.81 |
| | Pandora | 2 | 0 | 0 | 0.38 | 0 | 0 | 0.19 | 0.56 | 0 | 2.19 | 0.25 | 3.75 | 0.06 | 52.12 |
| | | 5 | 0 | 0 | 1.63 | 0 | 0 | 0.31 | 0.19 | 0 | 0.5 | 0.63 | 0 | 0.06 | 60.69 |
| | Pandora North | 5 | 0 | 0 | 4.36 | 0 | 0 | 0.27 | 0.64 | 0 | 0 | 1.94 | 0.53 | 0.07 | 25.88 |
| | Lady Elliot | 2 | 0 | 0 | 3.69 | 0 | 0 | 0.94 | 0.44 | 0 | 0.12 | 0.06 | 0 | 0 | 89.94 |
| | | 5 | 0 | 0 | 0.38 | 0 | 0 | 0.19 | 0.56 | 0 | 0 | 0 | 0 | 0 | 44.32 |
| | Magnetic | 2 | 0 | 0 | 1.88 | 0.38 | 0 | 0.44 | 1.38 | 0 | 1.94 | 4.94 | 12.81 | 0.31 | 43.62 |
| | | 5 | 0 | 0 | 1.94 | 0 | 0 | 0.69 | 2.01 | 0 | 0.88 | 2.76 | 8.19 | 0.19 | 37.74 |

Table A11 (continued).

| Region | Reef | Depth | Rhodophyta (red algae) | | | | | | | Chlorophyta (green algae) | Phaeophyta (brown algae) | | | | Turf algae |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | *Amansia* | *Asparagopsis* | Crustose coralline | *Hypnea* | *Laurencia* | *Peyssonnelia* | Undefined | | *Dictyota* | *Lobophora* | Sargassaceae | Undefined | |
| Mackay–Whitsunday | Hayman | 5 | 0 | 0 | 0.13 | 0 | 0 | 0.07 | 0.17 | 0 | 0 | 0.2 | 0 | 0.1 | 66.89 |
| | Border | 5 | 0 | 0 | 0.1 | 0 | 0 | 0 | 0.07 | 0 | 0 | 0 | 0 | 0.03 | 40.78 |
| | Hook | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0.31 | 1.19 | 0 | 0 | 0 | 0 | 62.12 |
| | | 5 | 0 | 0 | 0.38 | 0 | 0 | 0 | 0.12 | 1 | 0 | 0.06 | 0 | 0 | 38.58 |
| | Double Cone | 2 | 0 | 0 | 0.44 | 0 | 0 | 0 | 1.69 | 0.19 | 5.69 | 0.56 | 15.26 | 0.19 | 46.81 |
| | | 5 | 0 | 0 | 0.31 | 0 | 0 | 0.06 | 1.94 | 0.06 | 2.25 | 2.81 | 5.44 | 0.31 | 50.88 |
| | Daydream | 2 | 0 | 0 | 0.69 | 0 | 0 | 0 | 6.94 | 0.06 | 0.44 | 4.88 | 9.88 | 2.94 | 33.34 |
| | | 5 | 0 | 0 | 0.31 | 0 | 0 | 0 | 0.31 | 0 | 0.06 | 2 | 0 | 0.06 | 45.56 |
| | Dent | 2 | 0 | 0 | 1.81 | 0 | 0 | 0.62 | 1.75 | 0.25 | 0 | 2.19 | 1.88 | 0.44 | 48.44 |
| | | 5 | 0 | 0 | 2.5 | 0 | 0 | 1.06 | 0.81 | 0.12 | 0 | 1.81 | 0 | 0.06 | 45.88 |
| | Shute Harbour | 2 | 0 | 0 | 0.13 | 0 | 0 | 0.19 | 0.31 | 0 | 0.06 | 0.88 | 1 | 0.12 | 24.99 |
| | | 5 | 0 | 0 | 0.38 | 0 | 0 | 0.06 | 0.31 | 0 | 0.06 | 0.56 | 0.06 | 0.19 | 33.5 |
| | Pine | 2 | 0 | 0 | 2.88 | 0.69 | 0 | 0.06 | 3.81 | 0.19 | 0.12 | 8.56 | 20.94 | 1 | 47.81 |
| | | 5 | 0 | 0 | 2.63 | 0 | 0 | 0.62 | 0.81 | 0.25 | 0 | 3.94 | 0.25 | 0 | 62.92 |
| | Seaforth | 2 | 0 | 0 | 1.44 | 0.56 | 0 | 0.19 | 6.94 | 0.19 | 0.38 | 4 | 7.38 | 0.81 | 33.06 |
| | | 5 | 0 | 0 | 0.19 | 0 | 0 | 0 | 2.88 | 0 | 0.12 | 2.38 | 1.31 | 0.44 | 29.81 |

Table A11 (continued).

| Region | Reef | Depth | Rhodophyta (red algae) | | | | | | | Chlorophyta (green algae) | Phaeophyta (brown algae) | | | | Turf algae |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | *Amansia* | *Asparagopsis* | Crustose coralline | *Hypnea* | *Laurencia* | *Peyssonnelia* | Undefined | | *Dictyota* | *Lobophora* | Sargassaceae | Undefined | |
| Fitzroy | Barren | 2 | 0 | 0 | 1.59 | 0 | 0 | 0 | 3.67 | 0.06 | 0 | 1.53 | 0 | 0.05 | 54.04 |
| | | 5 | 0 | 0 | 3.79 | 0 | 0 | 0.05 | 10.41 | 0 | 0 | 11.86 | 0 | 0.05 | 24.65 |
| | North Keppel | 2 | 0 | 0 | 3.69 | 0 | 0 | 0.63 | 3.94 | 0 | 0 | 37.72 | 0 | 0.06 | 46.4 |
| | | 5 | 0 | 0 | 3.06 | 0 | 0 | 0.38 | 4.06 | 0 | 0 | 26.12 | 0.06 | 0.19 | 38.25 |
| | Middle | 2 | 0 | 0 | 1.79 | 0 | 0 | 1.1 | 2.65 | 0 | 0.05 | 22.61 | 23.21 | 0.45 | 41.88 |
| | | 5 | 0 | 0 | 1.2 | | 0 | 0.4 | 1.3 | 0 | 0.7 | 8.7 | 34.22 | 0.4 | 35.65 |
| | Keppels South | 2 | 0 | 0.06 | 0.88 | 2.81 | 0 | 0.88 | 2.75 | 0 | 1.25 | 27.31 | 3.56 | 0.62 | 41.44 |
| | | 5 | 0 | 1.94 | 2.06 | 0.5 | 0 | 2.31 | 5.31 | 0 | 3.12 | 30.62 | 0.06 | 0.31 | 29.5 |
| | Pelican | 2 | 0 | 0 | 2.13 | 0 | 0 | 0.1 | 3.19 | 0.15 | 0.4 | 14.71 | 32.89 | 1.13 | 27.28 |
| | | 5 | 0 | 0 | 1.85 | 0 | 0 | 0.45 | 2.32 | 0 | 0.29 | 5.29 | 0.7 | 0.3 | 37.47 |

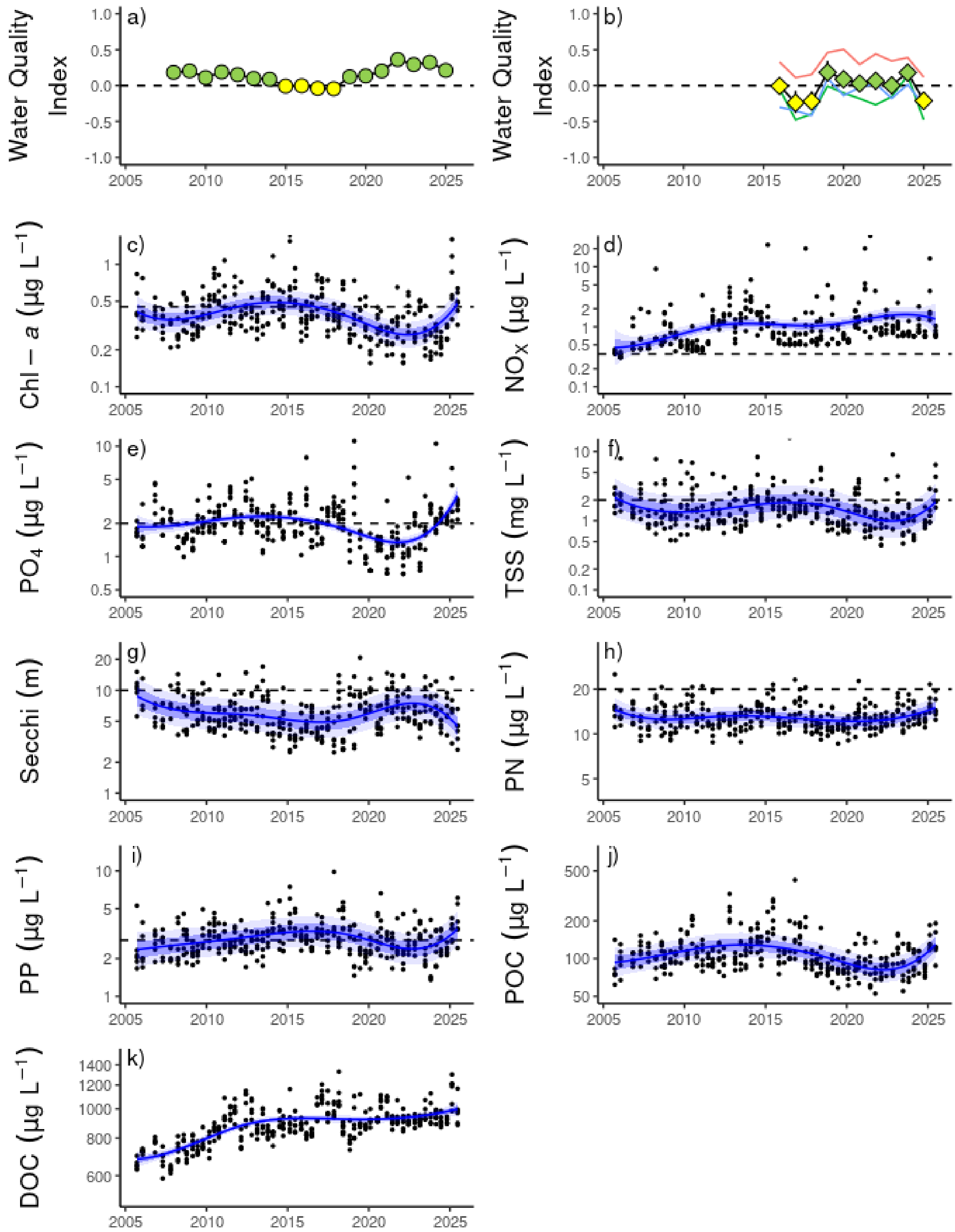


Figure A11. Temporal trends in water quality in the Barron–Daintree sub-region. The WQ condition Index uses 2 formulations to communicate: a), long-term trend (based on pre-2015 sampling design) and b), the annual condition (based on post-2015 sampling design). WQ Index colour coding: 'very good'; 'good'; 'moderate; 'poor'; 'very poor'. Error bars (vertical black lines) represent the 95% quantile intervals. Trends in individual water Quality variables c), Chl *a*, d), nitrate and nitrite , e), phosphate , f), total suspended solids, g), Secchi depth, h), particulate nitrogen, i), particulate phosphorus, j), particulate organic carbon and k), dissolved organic carbon. Generalised additive mixed effect model predictions (trends) are represented by blue lines with shaded areas defining 95% confidence intervals of those trends and black dots represent observed data (depth weighted averages). Dashed horizontal reference lines indicate annual guidelines for open coastal waters, where available. These trends and observations are seasonally detrended. Plots are reproduced from Gruber *et al*. (2026).

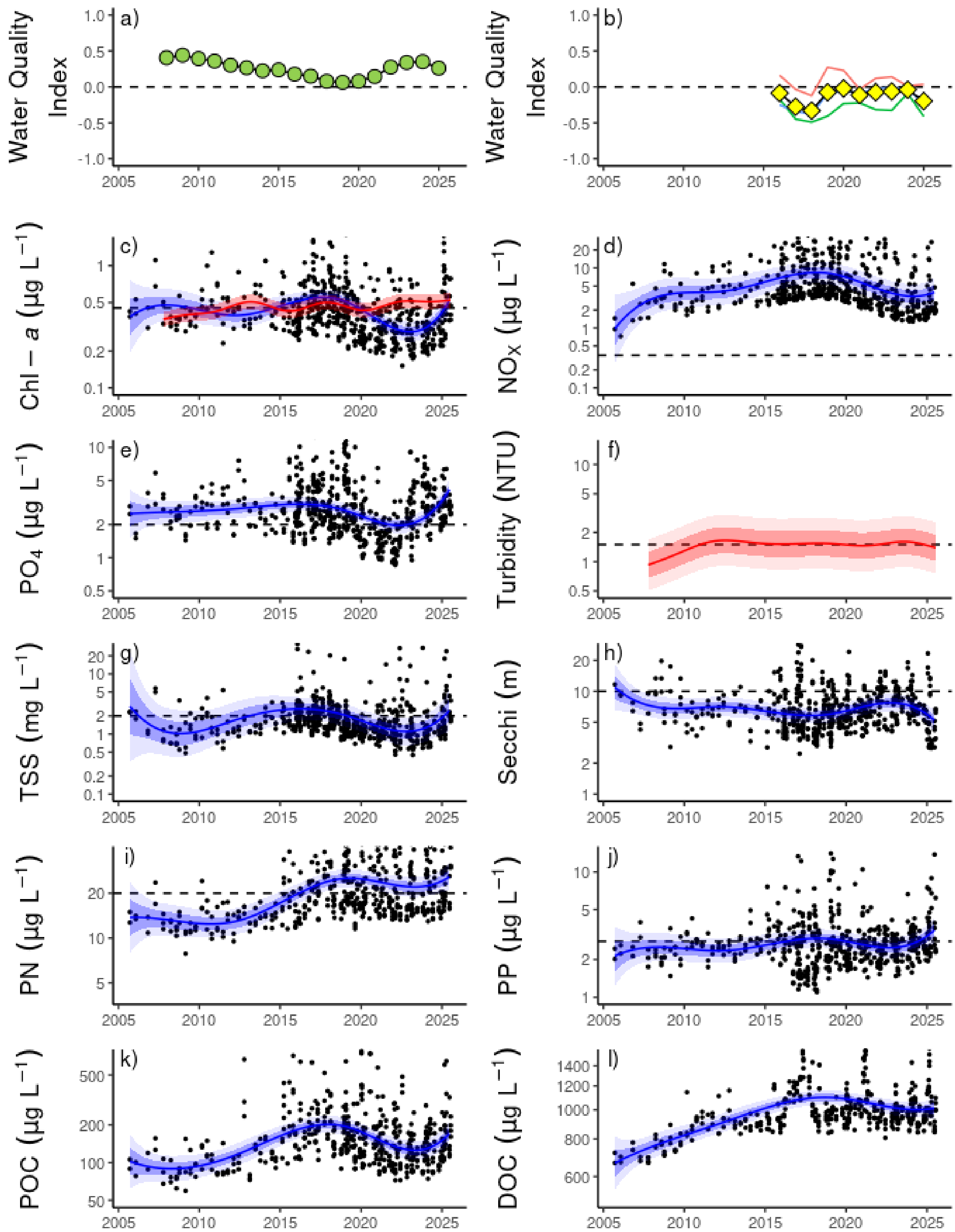


Figure A12. Temporal trends in water quality in the Johnston Russell–Mulgrave sub-region. The WQ condition Index uses 2 formulations to communicate: a), long-term trend (based on pre-2015 sampling design) and b), the annual condition (based on post-2015 sampling design). WQ Index colour coding: 'very good'; 'good'; 'moderate; 'poor'; 'very poor'. Error bars (vertical black lines) represent the 95% quantile intervals. Trends in individual water Quality variables c), Chl *a*, d), nitrate and nitrite, e), phosphate, f) turbidity, g) total suspended solids, h), Secchi depth, i), particulate nitrogen, j), particulate phosphorus, k), particulate organic carbon and l), dissolved organic carbon. Generalised additive mixed effect model predictions (trends) are represented by blue lines with shaded areas defining 95% confidence intervals of those trends and black dots represent observed data (depth weighted averages). Trends of records from ECO FLNTUSB instruments are represented in red. These trends and observations are seasonally detrended. Dashed horizontal reference lines indicate annual guidelines for open coastal waters, where available. Plots are reproduced from Gruber *et al.* (2026).

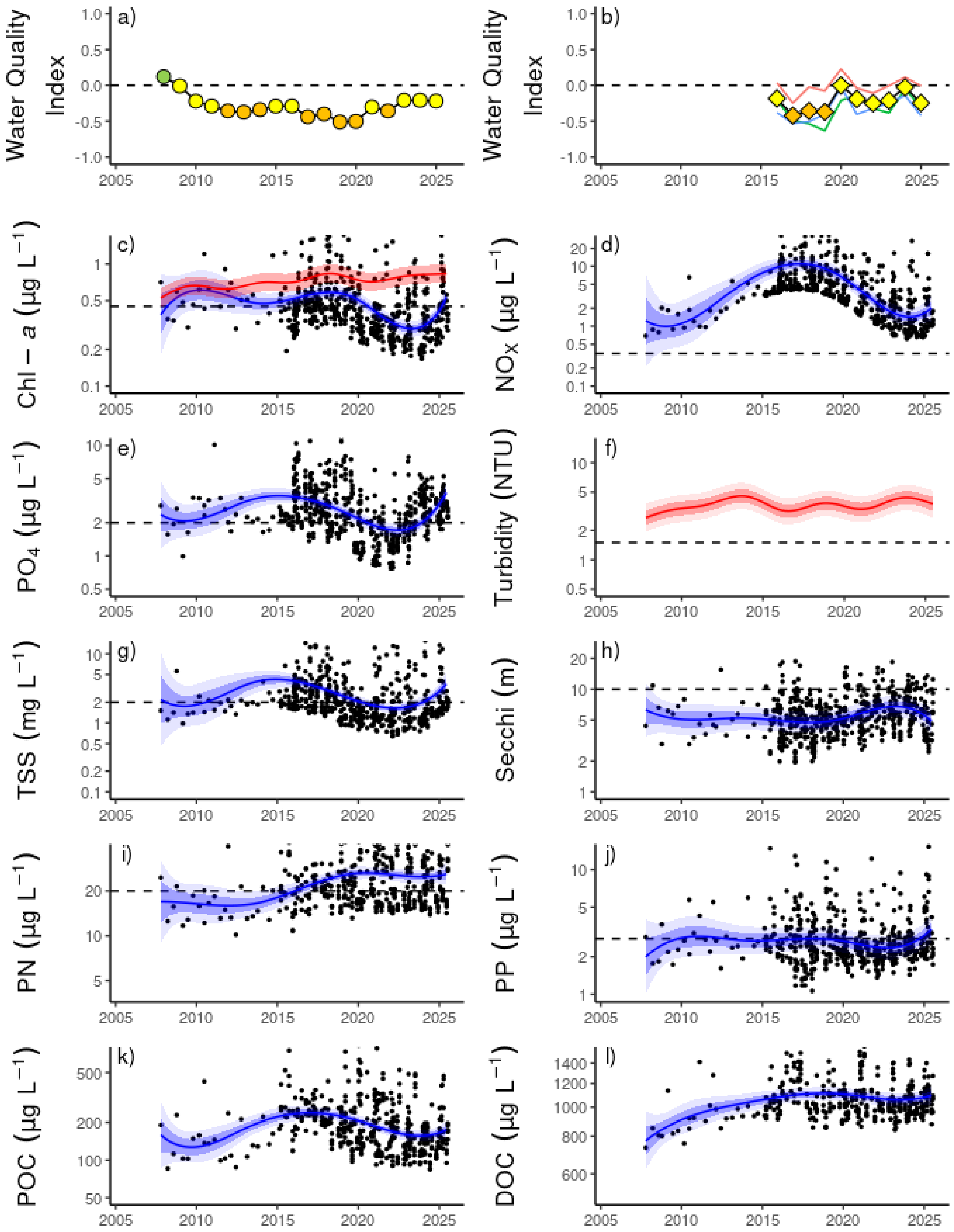


Figure A13. Temporal trends in water quality in the Herbert–Tully sub-region. The WQ condition Index uses 2 formulations to communicate: a), long-term trend (based on pre-2015 sampling design) and b), the annual condition (based on post-2015 sampling design). WQ Index colour coding: 'very good'; 'good'; 'moderate; 'poor'; 'very poor'. Error bars (vertical black lines) represent the 95% quantile intervals. Trends in individual water Quality variables c), Chl *a*, d), nitrate and nitrite, e), phosphate, f), turbidity, g), total suspended solids, h), Secchi depth, i), particulate nitrogen, j), particulate phosphorus, k), particulate organic carbon and l), dissolved organic carbon. Generalised additive mixed effect model predictions (trends) are represented by blue lines with shaded areas defining 95% confidence intervals of those trends and black dots represent observed data (depth weighted averages). Trends of records from ECO FLNTUSB instruments are represented in red. These trends and observations are seasonally detrended. Dashed horizontal reference lines indicate annual guidelines for open coastal waters, where available. Plots are reproduced from Gruber *et al*. (2026).

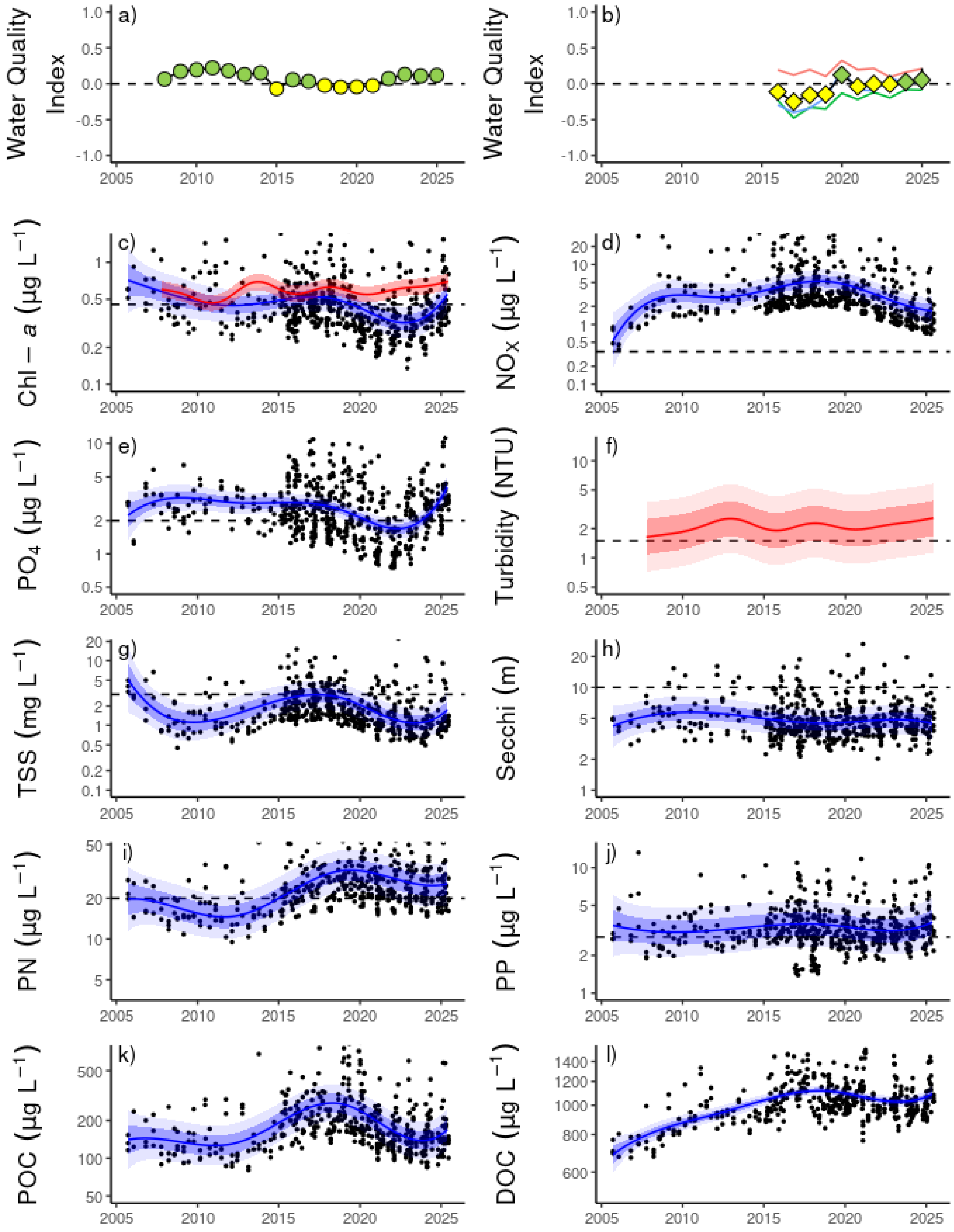


Figure A14. Temporal trends in water quality in the Burdekin region. The WQ condition Index uses 2 formulations to communicate: a), long-term trend (based on pre-2015 sampling design) and b), the annual condition (based on post-2015 sampling design). WQ Index colour coding: 'very good'; 'good'; 'moderate; 'poor'; 'very poor'. Error bars (vertical black lines) represent the 95% quantile intervals. Trends in individual water Quality variables c), Chl *a*, d), nitrate and nitrite, e), phosphate, f), turbidity, g), total suspended solids, h), Secchi depth, i), particulate nitrogen, j), particulate phosphorus, k), particulate organic carbon and l), dissolved organic carbon. Generalised additive mixed effect model predictions (trends) are represented by blue lines with shaded areas defining 95% confidence intervals of those trends and black dots represent observed data (depth weighted averages). Trends of records from ECO FLNTUSB instruments are represented in red. These trends and observations are seasonally detrended. Dashed horizontal reference lines indicate annual guidelines for open coastal waters, where available. Plots are reproduced from Gruber *et al.* (2026).

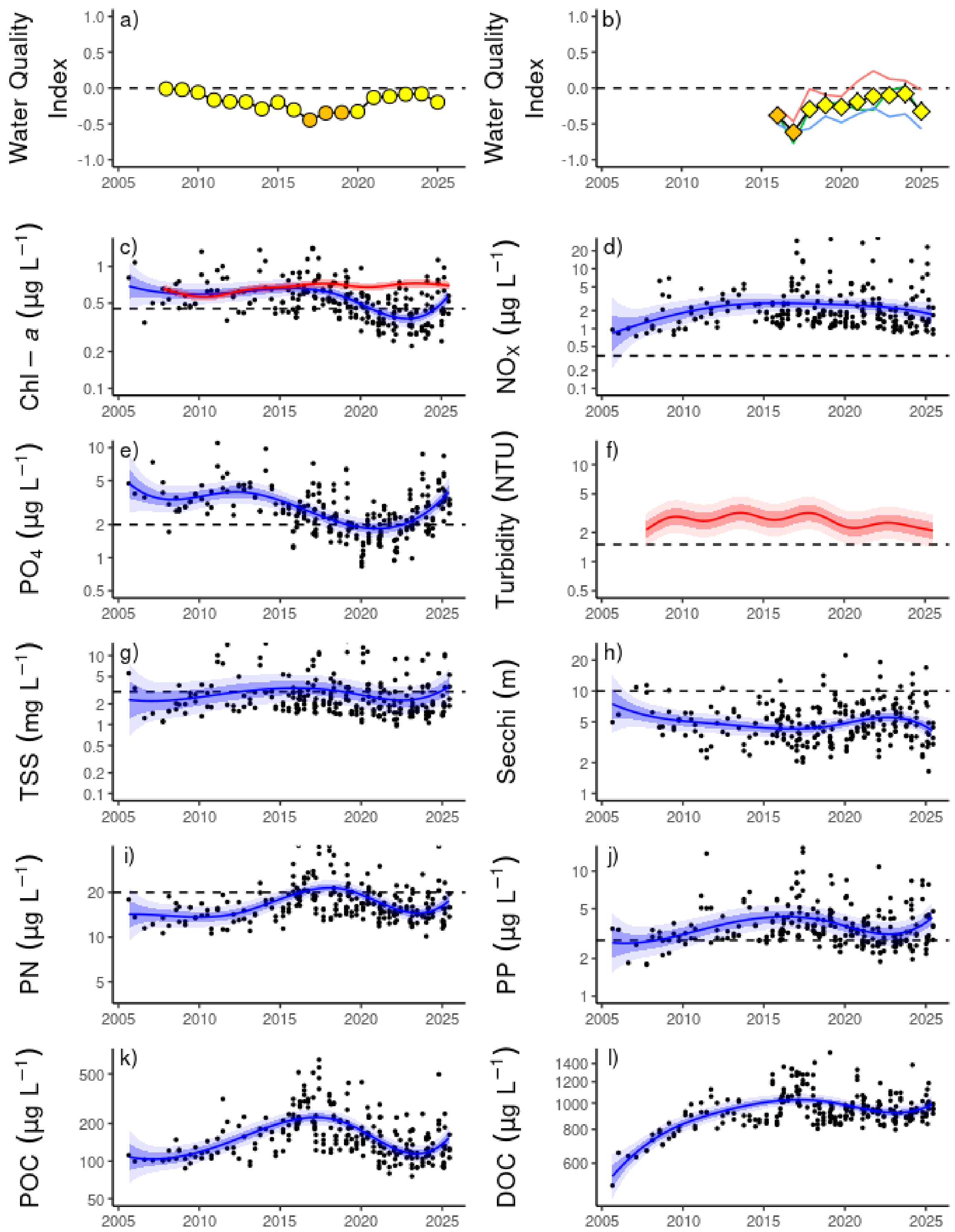


Figure A15. Temporal trends in water quality in the Mackay–Whitsunday region. The WQ condition Index uses 2 formulations to communicate: a), long-term trend (based on pre-2015 sampling design) and b), the annual condition (based on post-2015 sampling design). WQ Index colour coding: ● 'very good'; ● 'good'; ● 'moderate; ● 'poor'; ● 'very poor'. Error bars (vertical black lines) represent the 95% quantile intervals. Trends in individual water Quality variables c), Chl *a*, d), nitrate and nitrite, e), phosphate, f), turbidity, g), total suspended solids, h), Secchi depth, i), particulate nitrogen, j), particulate phosphorus, k), particulate organic carbon and l), dissolved organic carbon. Generalised additive mixed effect model predictions (trends) are represented by blue lines with shaded areas defining 95% confidence intervals of those trends and black dots represent observed data (depth weighted averages). Trends of records from ECO FLNTUSB instruments are represented in red. These trends and observations are seasonally detrended. Dashed horizontal reference lines indicate annual guidelines for open coastal waters, where available. Plots are reproduced from Gruber *et al*. (2026).

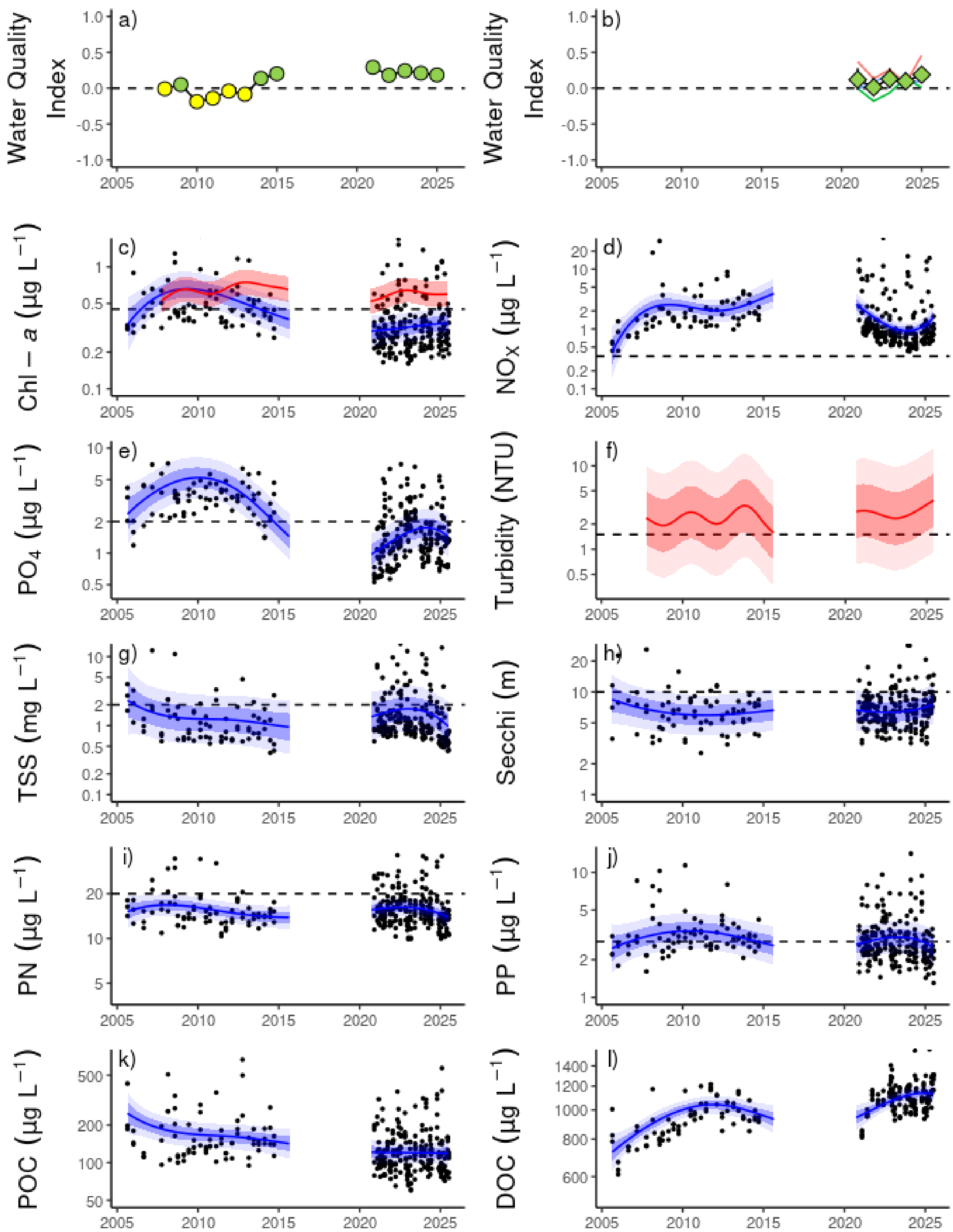


Figure A16. Temporal trends in water quality in the Fitzroy region. The WQ condition Index uses 2 formulations to communicate: a), long-term trend (based on pre-2015 sampling design) and b), the annual condition (based on post-2015 sampling design). WQ Index colour coding: ● 'very good'; ● 'good'; ● 'moderate; ● 'poor'; ● 'very poor'. Error bars (vertical black lines) represent the 95% quantile intervals. Trends in individual water Quality variables c), Chl *a*, d), nitrate and nitrite, e), phosphate, f), turbidity, g), total suspended solids, h), Secchi depth, i), particulate nitrogen, j), particulate phosphorus, k), particulate organic carbon and l), dissolved organic carbon. Generalised additive mixed effect model predictions (trends) are represented by blue lines with shaded areas defining 95% confidence intervals of those trends and black dots represent observed data (depth weighted averages). Trends of records from ECO FLNTUSB instruments are represented in red. These trends and observations are seasonally detrended. Dashed horizontal reference lines indicate annual guidelines for open coastal waters, where available. Plots are reproduced from Gruber *et al.* (2026).

## Appendix 2: Publications and presentations 2024–2025

### Publications

Healthy Rivers to Reef Partnership Mackay-Whitsunday-Isaac (2025). Mackay-Whitsunday-Isaac Report Card Results 2025: Technical Report. Mackay-Whitsunday-Isaac Healthy Rivers to Reef Partnership, Mackay, QLD ISBN: 978-1-7637543-1-7. available here

Prazeres, M., Gruber, R., Howley, C., Lewis, S., McKenzie, L., Thompson, A., Thompson, C., Thompson, K., Waterhouse, J. and Walker, K., (2024). Great Barrier Reef Marine Monitoring Program Synthesis Report 2022–23. available here

Shand, A., Taylor, D., (2025). Technical Report for the Townsville Dry Tropics Report Card Results 2025 (Reporting on July 2023 – June 2024). Healthy Waters Partnership for the Dry Tropics, Townsville.

Wet Tropics Waterways 2025. Wet Tropics Report Card 2024 (reporting on data 2023-24). Waterway Environments: Results. Wet Tropics Waterways and Terrain NRM, Innisfail. available here

Bozec, Y.M., Adam, A.A., Nava, B.A., Cresswell, A.K., Haller-Bull, V., Iwanaga, T., Lachs, L., Matthews, S.A., McWhorter, J.K., Anthony, K.R. and Condie, S.A., 2025. A rapidly closing window for coral persistence under global warming. *bioRxiv*, pp.2025-01.

Great Barrier Reef Marine Park Authority, Australian Institute of Marine Science, and CSIRO, 2025, Reef Snapshot: Summer 2024–25, Reef Authority, Townsville

Warne, D.J., Crossman, K., Heron, G.E., Sharp, J.A., Jin, W., Wu, P.P.Y., Simpson, M.J., Mengersen, K. and Ortiz, J.C., (2025). Mathematical modelling and uncertainty quantification for analysis of biphasic coral reef recovery patterns. *Bulletin of Mathematical Biology*, *87*(9), pp.1-27.

### Presentations

Marine Monitoring Program – Coral 2024 Burdekin. Burdekin Paddock to Reef Regional science Forum. Mackay. 15th May 2025

Marine Monitoring Program - Coral 2025. Presentation at Marine Monitoring Program Science Seminar. Reef Authority, 1st Sep 2025

Marine Monitoring Program – Coral 2025. Annual presentation to stakeholders. Townsville Yacht Club 25th November 2025

Marine Monitoring Program – Coral monitoring activities in Woppaburra Sea Country. Presentation to Woppaburra TUMRA steering committee. Online 2nd Dec 2024.

Marine Monitoring Program – Coral monitoring activities in Manburra Sea Country. Presentation to Manburra Elders and Rangers. AIMS 5th Mar 2025.

Marine Monitoring Program – Coral monitoring activities in Gunggandji Mandingalbay-Yidinji Sea Country. Presentation to Gunggandji Mandingalbay-Yidinji Rangers. AIMS 27th May 2025.

Marine Monitoring Program – Coral monitoring activities in Gunggandji Mandingalbay-Yidinji Sea Country. Presentation to Gunggandji Mandingalbay-Yidinji Rangers. AIMS 11th Sept 2025.

Marine Monitoring Program – Coral monitoring activities in Girringun Sea Country. Presentation to Girringun steering committee. Cardwell, 16th Sept 2025.