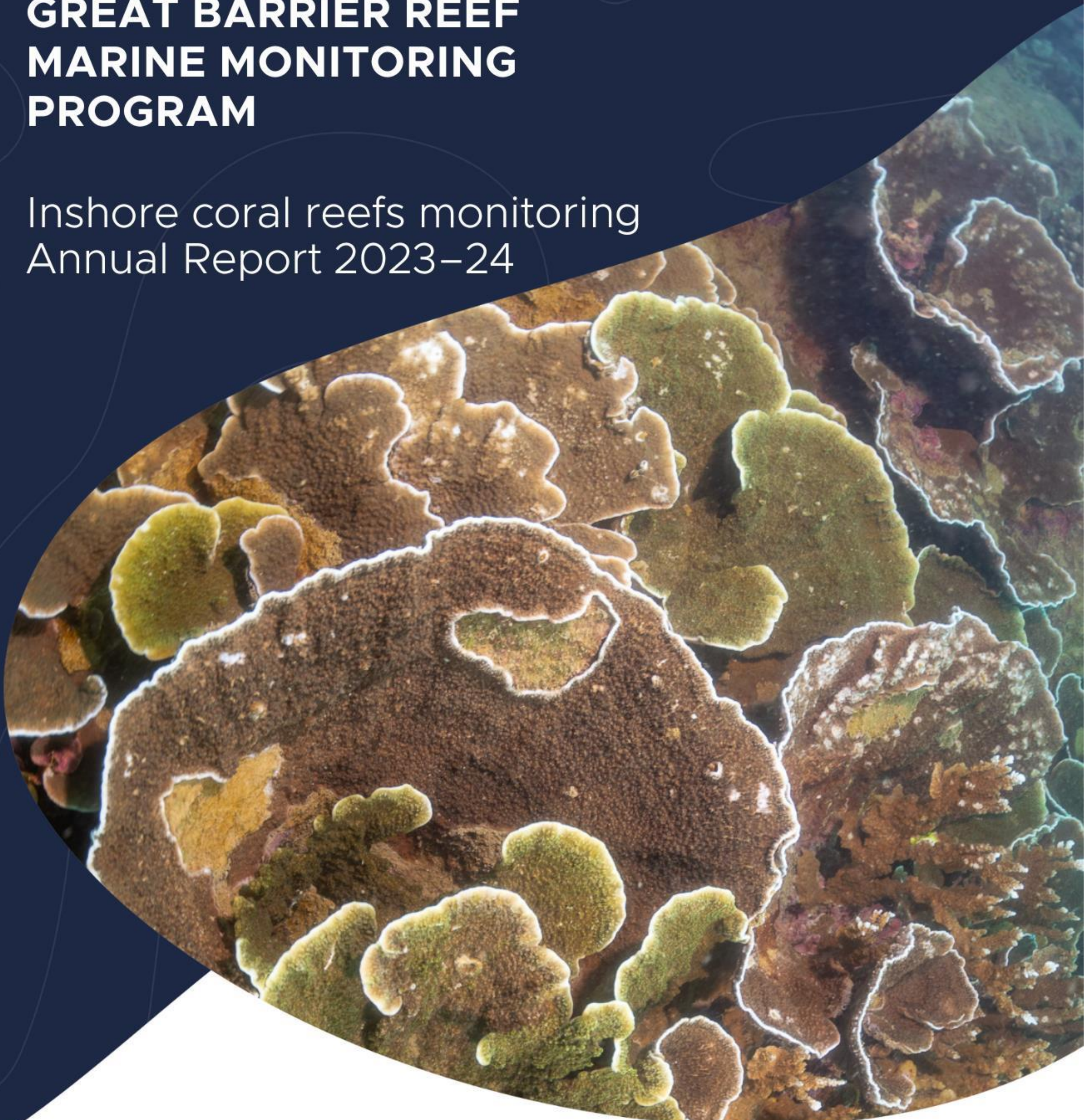
GREAT BARRIER REEF MARINE MONITORING PROGRAM

Inshore coral reefs monitoring Annual Report 2023–24



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Front cover photo: Healthy *Montipora* colonies on the reef slope at Snapper Island North. © Australian Institute of Marine Science, Photographer: T. Ayling.

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 six 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' having declined to the lowest value recorded since the Marine Monitoring Program began in 2005 (Figure 1). Influential in the decline since last year were the impacts of high summer water temperatures that lead to coral bleaching. Bleaching was most severe in the Fitzroy region, where heat stress, measured as degree heating weeks, exceeded any prior observations for inshore areas of the Reef. In the Wet Tropics, the severe impact of Cyclone Jasper added to impacts attributed to coral bleaching and ongoing outbreaks of crown-of-thorns starfish.

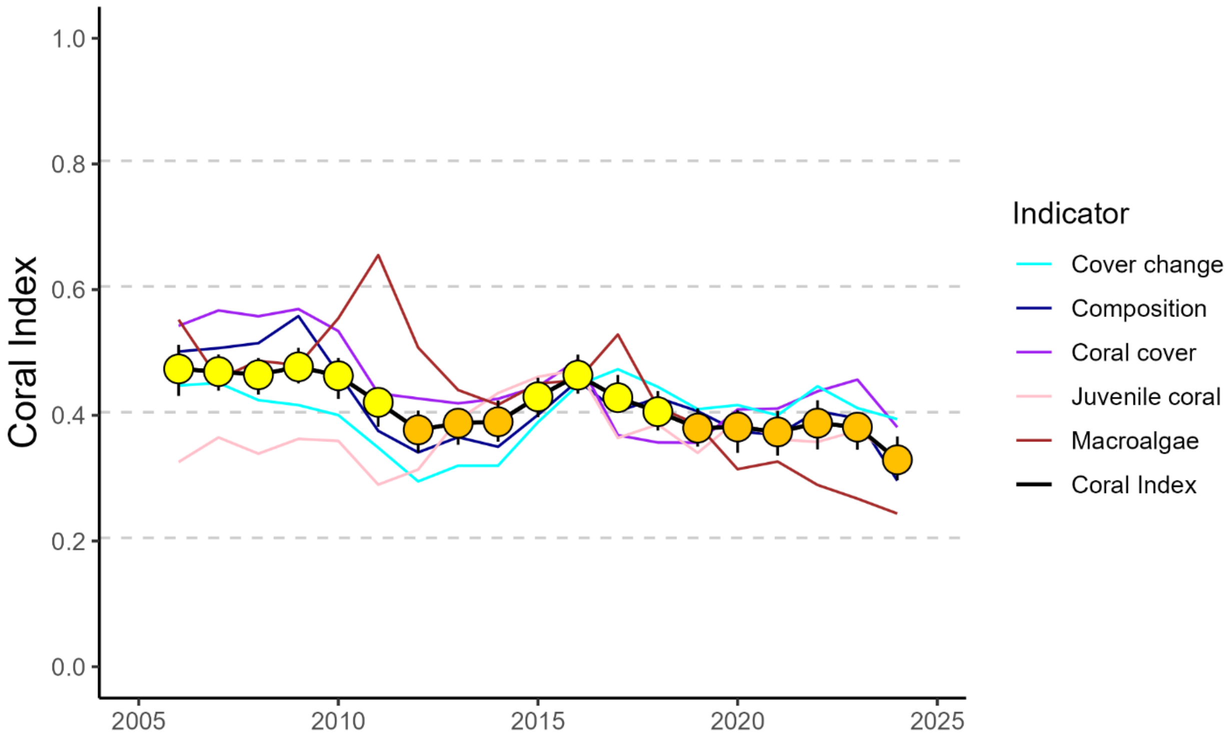

Figure 1 Trends in the Coral Index and contributing indicator scores for the inshore Reef. Coral Index scores are coloured according to Reef Water Quality Report Card categories: orange = 'poor', yellow = 'moderate'.

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 five 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 corals' ability to resist the cumulative environmental pressures to which they have been exposed, but 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 – note that 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.

The Coral Index score is published in the Reef Water Quality Report Card and contributes to the marine condition score. 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. These scores, in combination with additional locally relevant data sources, are also published in regional report cards.

Multiple pressures impacted inshore reefs over the 2023-24.

A severe marine heat wave resulted in coral bleaching and loss of coral cover in the Wet Tropics, Burdekin and Fitzroy regions. The highest heat stress, measured as degree heating weeks, occurred in the Fitzroy Region, where it exceeded levels previously recorded in inshore areas of the Reef.

Cyclone Jasper caused storm and flood damage to reefs in the Northern Wet Tropics region and cyclone Kirrily caused minor storm damage in the Burdekin Region

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

Improvement of coral community condition scores between 2011 and 2016 demonstrated the capacity of inshore coral communities to recover. However, between 2016 and 2024, the cumulative pressures imposed by cyclones and flooding, high seawater temperatures leading to coral bleaching in 2017, 2020, 2022, and again in 2024 and high densities of crown-of-thorns starfish densities 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-2024 in each Natural Resource Management region in which inshore reefs are monitored.

Wet Tropics region coral community condition

Coral communities remain in 'moderate' condition, but condition has declined in each of the three sub-regions monitored.

- In the Barron–Daintree sub-region, reefs 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. However, coral cover was also greatly reduced in the shallow sites at Snapper North. Despite these impacts, the Coral Index score remains in the ‘moderate’ range. Buoying the Coral Index score were ‘good’ scores for the Macroalgae and Cover change indicators, and the timing of surveys for one reef, Low Isles, which was last surveyed prior to the passage of cyclone Jasper. The loss of macroalgae strongly influenced the improvement in Macroalgae score cover from shallow sites at Snapper North. This is almost certainly a short-term response to the impacts of cyclone Jasper, as at other inshore reefs where such removal has occurred, macroalgae have rapidly recolonised. It is likely the resurvey of Low Isles in 2025 will document some impact of cyclone Jasper further reducing the Coral Index score.

- In the Johnstone Russell–Mulgrave sub-region, the Coral Index continued to decline from a high point in 2022. The current decline was 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 observed on most reefs.
- In the Herbert–Tully sub-region, the Coral Index score has also continued to decline but remains ‘moderate’. The Coral cover scores changed little from 2023 at which time it was higher than previously observed. The lack of improvement in Coral cover adds to limited change following high water temperatures in 2022, resulting in a decline in the Cover change score. The Macroalgae score remains ‘poor’ as high levels of macroalgae persist at Dunk Island and Bedarra Island.

Burdekin region coral community condition

High water temperatures causing coral bleaching and cyclone Kirrily combined to cause a slight reduction in Coral cover. The Coral Index score remains within the ‘moderate’ range but lower than the peak value in 2020 reached as coral communities had recovered from the impact of cyclone Yasi in 2011. The recent high summer water temperatures, and cyclone impacts, build on thermal stress events in 2020 and, to a lesser extent, in 2022, that also caused coral bleaching at most reefs. Despite these events, regional-scale coral cover has remained reasonably stable, with these events limiting further recovery rather than decreasing coral cover. The trajectories at individual reefs have been more variable.

Both the Juvenile coral and Macroalgae indicators scores remained ‘poor’ further influencing the Coral index scores and indicative of potential water quality associated pressures on recovery potential. Macroalgae scores were ‘very poor’ on at least one depth at all reefs inshore of Palms East and Palms West.

Mackay–Whitsunday region coral community condition

The Coral Index score remained ‘poor’ in 2024. Recovery from the severe impact of cyclone Debbie in 2017 continues to be slow and most indicators remained in the ‘poor’ category. Coral cover has begun to improve at some reefs, with improvement being more consistent at shallower sites. Signs of recovery are most evident as increasing densities of juvenile corals, but this is primarily at reefs where macroalgae is low. At other reefs persistently high cover of macroalgae continues to limit coral recovery. The early signs of recovery coincide with improving gradually improving water quality in the region.

Fitzroy region coral community condition

The marine heatwave of 2024 has impacted all coral metrics, sending the regional Coral Index to its lowest level in 10 years. Hard coral cover across the region has declined by 38% and post-bleaching mortality is likely to raise this figure as 60% of the surviving corals were still bleached at the time of survey. Those locations with high proportions of *Acropora* corals were impacted most; North Keppel, Barren, Keppels South. The Macroalgae indicator score has improved marginally but remains ‘poor’ as high abundance of brown macroalgae still persists in the region.

1 INTRODUCTION

The proximity of inshore reefs to the coast makes them highly accessible; this elevates their social, economic and cultural importance disproportionately to their small contribution to the area of the Great Barrier Reef World Heritage area's coral estate (GBRMPA 2024). Unfortunately, this proximity also exposes inshore reefs to increased pressures of 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). This ramping-up of pressures facing coral reefs makes it ever more important that the Reef environment 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 coral monitoring program

Disentangling the complexity of interactions between benthic communities and environmental pressures influencing the condition of coral reefs is reliant on accurate, long-term, field-based observations of the response of communities 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 output component 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 is designed to capture key aspects of coral community condition and resilience that is used to track trends in community condition, but also highlights 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 disturbances to inshore reefs include cyclones (often coinciding with 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). 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 existing colonies from remaining tissue fragments (Smith 2008, Diaz-Pulido *et al.* 2009). Elevated concentrations of nutrients, pesticides and turbidity can negatively affect reproduction in corals (reviewed by Fabricius 2005, van Dam *et al.* 2011, Erftemeijer *et al.* 2012). 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) suggest the cumulative pressure that poor water quality will have 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 concentrations, indicating 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), although chemical cues from some species conversely 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 ability to compensate, by heterotrophic feeding, where there is a reduction in energy derived from photosynthesis, e.g., because of light attenuation in turbid waters (Bessell-Browne *et al.* 2017), varies between species (Anthony 1999, Anthony & Fabricius 2000). Similarly, the energy required to shed sediment varies between 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. Reduced energy delivered to corals by their symbionts or competition for space are likely to reduce the rate at which corals grow or increase their susceptibility to disease (Vega Thurber *et al.* 2013). A derivative of coral cover is an indicator based on expected rate of coral cover increase (Thompson *et al.* 2020).

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 2024 with inclusion of data from inshore reefs monitored by the AIMS Long-Term Monitoring Program (LTMP) from 2005 to 2024. 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 reef 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.

2 METHODS

This section provides an overview of the source and manipulation of climate and environment 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 (Moran *et al*. 2025, 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 (Table 3). To match this sampling frequency, the maximum of the total annual discharge from all rivers discharging into a given region for each two-year period between 2006 and 2023 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 sourced from MMP Water Quality (Moran *et al.* 2025). 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 previous description). 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. 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 '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. The pre-development sediment and PN loads were calculated using a combination of the annual mean concentrations from the Source Catchments model and available monitoring data from 'pristine' locations. The corresponding discharge was used as calculated previously to produce a simulation of the pre-development load for the water year (Moran *et al*. 2022)."

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 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. Until 2008 temperature was recorded at 30-minute intervals with the interval reduced to 10 minutes thereafter (Table A2).

Loggers were calibrated against a certified reference thermometer after each deployment and measurements corrected where drift was identified. Temperature records for each logger are generally accurate to ± 0.2°C.

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

Two estimates of seasonal temperature anomalies, as an indication of potential temperature stress to corals, are also presented.

Degree heating weeks (DHW) were downloaded from the National Oceanic and Atmospheric Administration (NOAA) coral reef watch. The product sourced were the maximum DHW estimate for each ~5 km square pixel in a calendar year. DHW estimates accumulate time of exposure of more than 1 degree above the mean of the hottest month from a location's climatology (Liu *et al*. 2014). For each pixel on the globe the seasonal climatology was estimated over the period 1985-2012 allowing the hottest month of the year to be identified. The mean temperature of this month in the years 1985-1990 plus 1993 were estimated and used as the baseline summer maximum temperature. DHW estimates are the accumulation of temperatures that exceed this mean maximum monthly temperature by at least 1 degree Celsius, with the accumulation occurring over a 12-week rolling time period.

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 time series' maximum mean of the hottest month of the year, 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 0.5 degrees above their respective references, a similar approach was promoted by Whitaker & DeCarlo (2024) who applied a 0.4 degree above the mean monthly maximum as a warming threshold.

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

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 – 2024	water gauging stations closest to river mouth, adjusted for ungauged area of catchment	regional discharge plots and table, covariate in analysis of temporal change in Coral Index	DNRME, adjustment as tabulated by Moran *et al.* (2025)
Riverine DIN, sediment and PN loads	2006 – 2023		covariate in analysis of temporal change in Coral Index	MMP Water Quality (Moran *et al*. 2025)
Sea temperature	2005 – 2024	*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 – 2024	remote sensing	informing attribution of thermal stress, thermal stress maps	National Oceanographic and Atmospheric Administration
Cyclone tracks	2005– 2024		informing attribution of storms as cause of observed coral loss, cyclone track maps	BoM
Environment at coral monitoring sites				
Wet season Chlorophyll *a* (Chl *a*) and total suspended solids (TSS)	2003 – 2024	remote sensing and coupled niskin samples	Mapping. Chl *a* and TSS concentrations covariates in analysis of variability in Coral Index score changes and (Chl *a*) analysis of variability in Coral Index and indictor current state	MMP Water Quality
Non-algal particulate	2002 – 2018	remote sensing adjacent to coral sites, resolution ~1 km^2	Macroalgae and Composition metric thresholds, mapping	BoM
K490 light attenuation coefficient	2020-2024	Remote sensing adjacent to coral sites, resolution ~1km^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

2.1.5 Cyclone tracks

Cyclone tracks and intensity were downloaded from the BoM at http://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, Moran *et al*. 2025). In brief, Sentinel satellite data were used to classify waters into 21 Forel-Ule colour classes that were then aggregated into four reef water-types (Table 2). The water-type exposure for each pixel for the period 2020–2024 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 Moran *et al.* (2025). 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, summarised in 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 nine 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, summarised in Table 2, proportionate to the wet-season water-type exposures for each pixel. The resulting nine 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 five 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 in water 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.

For the subset of coral monitoring locations at which there are adjacent MMP water quality monitoring locations (see Table 3) mean concentrations of Chl a, TSS and the ratio of both dissolved and total fractions of N and P from niskin samples were estimated. These estimates were derived from all samples over the period July 2020 to June 2024 and used as explanatory variables for variation in

[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

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 flood waters, 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 waters corresponding to waters with slightly above ambient suspended sediment concentrations and high light 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 four of the six 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 two 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 two 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 determine 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, nothing their 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 2024.

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 two 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 is the exception with sites there at approximately 3 m below LAT.

2.2.4 Site marking

At each reef, two 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 five 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 two 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 five 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 allowed for both a timely estimate of the impact of the acute event and provided 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, productivity gains enabled the return 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. WQ, indicates reefs at which water quality monitoring is undertaken, * indicates WQ was ceased in 2014, and ** indicates WQ was begun in 2015. Blank cells indicate where reefs were not surveyed. Grey fill indicates where reefs were removed from the programs sampling design.

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

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 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 five 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 from those not originally randomly selected. The AIMS LTMP utilised longer 50 m transects sampled at 1 m intervals, from which 40 images were selected.

For most of hard and soft corals, identification to genus level was 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 those small colonies assessed as juveniles resulting from the settlement and subsequent survival and growth of coral larvae, and so does not include small coral colonies considered as resulting 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.

2.3.3 SCUBA search transects

SCUBA search transects document the incidence of disease and other agents of coral mortality and damage. Tracking of these agents of mortality is important as declines in coral community condition due to these agents are potentially 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, 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). Scaring caused by fish bites was not recorded as 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, separately, the proportion of corals that were bleached or had been physically damaged (as indicated by toppled or broken colonies). The category ranges were derived from the six categories 0 to 5 used to score benthic cover from manta tow surveys by the LTMP with addition of + and – to include more differentiation with these categories The physical damage category may include anchor as well as storm damage.

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 five 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 five 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 implies a degree of resistance to any chronic pressures influencing a reef. 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 two 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 reefs within the inshore Reef and allows a natural break point for the categorisation of coral cover into the five 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.

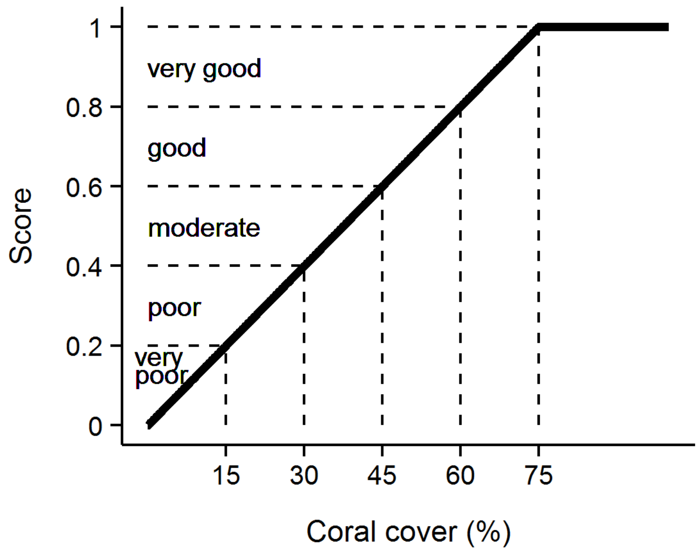

Figure 3 Scoring diagram for the Coral cover indicator metric. Numeric scores and associated condition classifications based on observed coral cover are presented.

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 hig proportion of macroalgae within the limited space available to algae can still be interpreted as imposing a downward pressure on coral resilience.

For the purpose of 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. 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 Chlorophyll *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).

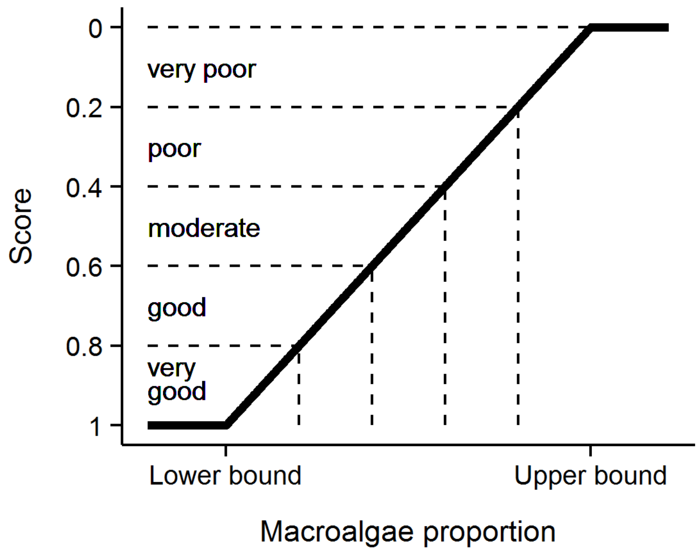

Figure 4 Scoring diagram for the Macroalgae indicator metric. Upper and lower threshold values are reef and depth specific. Numeric scores and associated condition classifications are presented. Note that for this metric the y-axis is inverted as high values reflect poor condition.

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. 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).

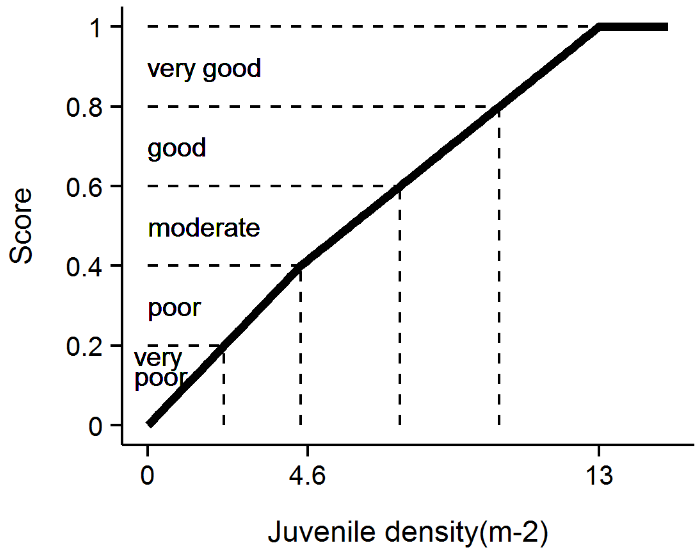

Figure 5 Scoring diagram for the Juvenile coral indicator metric. Numeric scores and associated condition classifications are presented.

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 and indicate a lack of resilience. The Cover change indicator score is derived from the comparison of the observed change in hard coral cover between two 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 the slower growing combined grouping of all other hard corals at each of 2 m and 5 m depths. Initial exploratory analysis provided no justification for a more detailed parameterisation of the coral community, in part due to the increasing imprecise estimates of cover due to declining cover for each group with further sub-setting of the coral community.

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 two 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.

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

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

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

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

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

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

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

$$rAcr = v\overline{A}cr_i$$

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 (β_j and Υ_k, respectively) were also incorporated to account for spatial autocorrelation. Model coefficients associated with the intercept, region and reef (α_i, β_j and Υ_k, respectively) all had weakly informative Gaussian priors, the latter two with model standard deviation. The overall rate of coral growth $rAcr$ constituted the mean of the individual posterior rates of increase for $v\overline{A}cr_i$.

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 three 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.

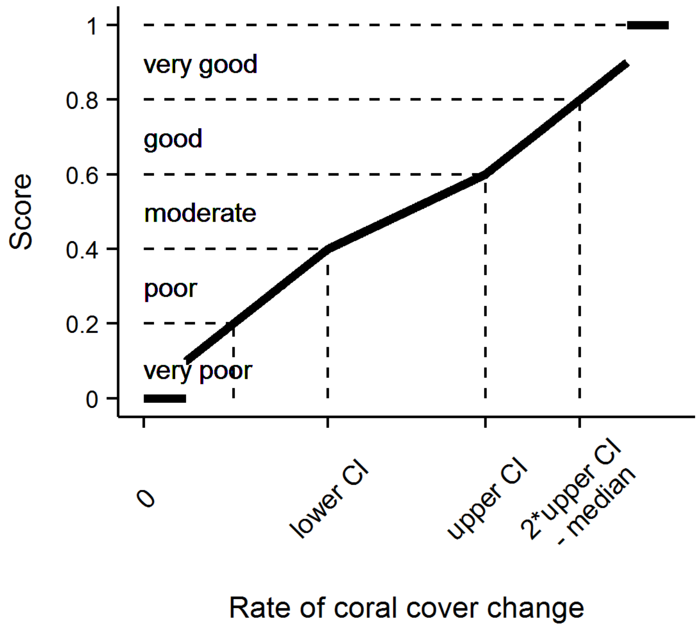

Figure 6 Scoring diagram for Cover change indicator metric.

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 the environmental conditions experienced. 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 life-forms 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 five 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).

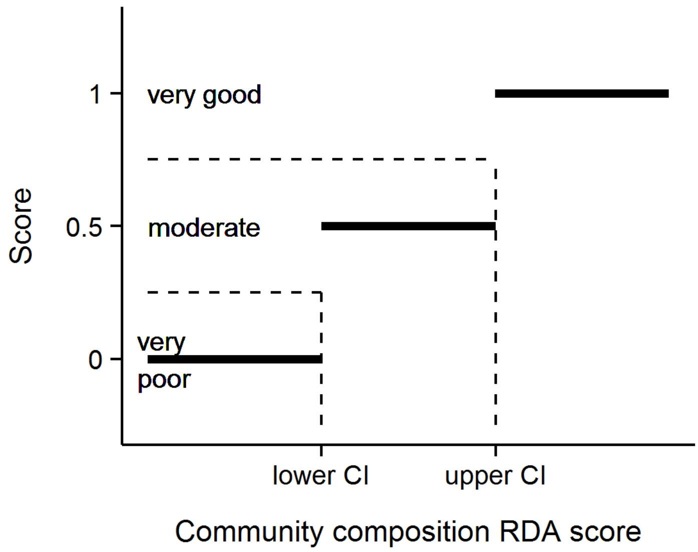

Figure 7 Scoring diagram for the Composition indicator metric

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).

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 five 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 four 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, 4.4, 4.5, 4.6	Trends in Coral Index and individual indicators	(sub-)region		Generalised linear mixed models; pairwise comparisons
4.7.1	Coral Index and indicator scores in 2024	Reef and region	Chl *a*, Light attenuation coefficient, Suspended solids, N to P ratios (Total and dissolved fractions	Generalised linear mixed models, predicted responses
4.7.2	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 2024 observations	reef/Depth		Observed values

2.5.1 Variation in Coral Index and indicator scores to gradients in water quality

The relationships between the most recent Coral Index or indicator scores, at each depth, and the location of reefs along water quality gradients were explored using generalised linear models (GLM). Models were fit separately to each combination of Coral Index or indicator score, and depth, that included either mean Chl *a* concentration or kd490 light attenuation coefficients as explanatory variable crossed with region as a fixed effect. Statistical evidence for water quality influences on the coral community indicators were identified on the basis that Akaike information criterion adjusted for small sample sizes (AICc) values were at least 2 units lower than 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*, Suspended solids and the ratio of N to P (uM) in both the dissolved inorganic fraction and total pools data estimated from MMP water quality niskin samples. These models were focused on overall responses to the environmental variables and included regions as an additive fixed effect.

All water quality data were reef level averages of the period July 2020 to June 2024, noting the Satellite derived Chl *a* estimates are wet season only.

2.5.2 Relationship between Coral Index scores and environmental conditions

The response of coral communities to variation in environmental conditions was assessed by comparing changes in Coral Index scores to annual discharge 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 covariate selected greater of the preceding two 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).

2.5.3 Temporal trends in Coral Index and indicators

A panel of plots provide temporal trends in the Coral Index and the five indicators on which the index is based. The derivation of annual Coral Index scores and associated confidence intervals is detailed in section 2.4.6.

For each of the five indicators that inform the Coral Index, temporal trends and their 95% confidence intervals in their observed values were derived from linear mixed effects models. Models for each indicator included a fixed effect for year and a random effect for each reef and depth combination. The inclusion of random locational effects helps to account for the sampling design that includes a mixture of annual and biennial sampling frequency. To account for missing samples (Table 3) in estimating the trend in Coral Index scores, missing indicator scores were infilled with observations from the preceding year as is done for the estimation of annual Coral Index scores.

Observed trends for individual reef and depth combinations (averaged over sites) are provided as grey lines.

A more detailed summary of proportional benthic cover, derived from photo point intercept transects, and juvenile density at each reef and depth combination is 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.

2.5.4 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.

2.5.5 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:

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

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 will include 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.

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 2023–24 reporting period, two cyclones, Jasper in December 2023 and Kirrily in January 2024, produced damaging waves and associated flooding that affected the regions covered by this report (Figure 8).

Since 2005, three 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 which 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, including cyclone Kirrily in 2024.

3.2 Sea temperature

Sea temperatures over the 2024 summer were above long-term averages and above the threshold 4 DHW (NOAA 2018) that are likely to lead to coral bleaching at most inshore reefs monitored (Figure 9). Extreme temperature anomalies with a mean of 12.5 DHW were recorded at reefs in the Fitzroy region where ongoing bleaching and recent mortality of corals was observed during surveys in May 2024 (Figure 9, Figure 12, Figure 32e). Severe (8-10 DHW) heat stress 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).

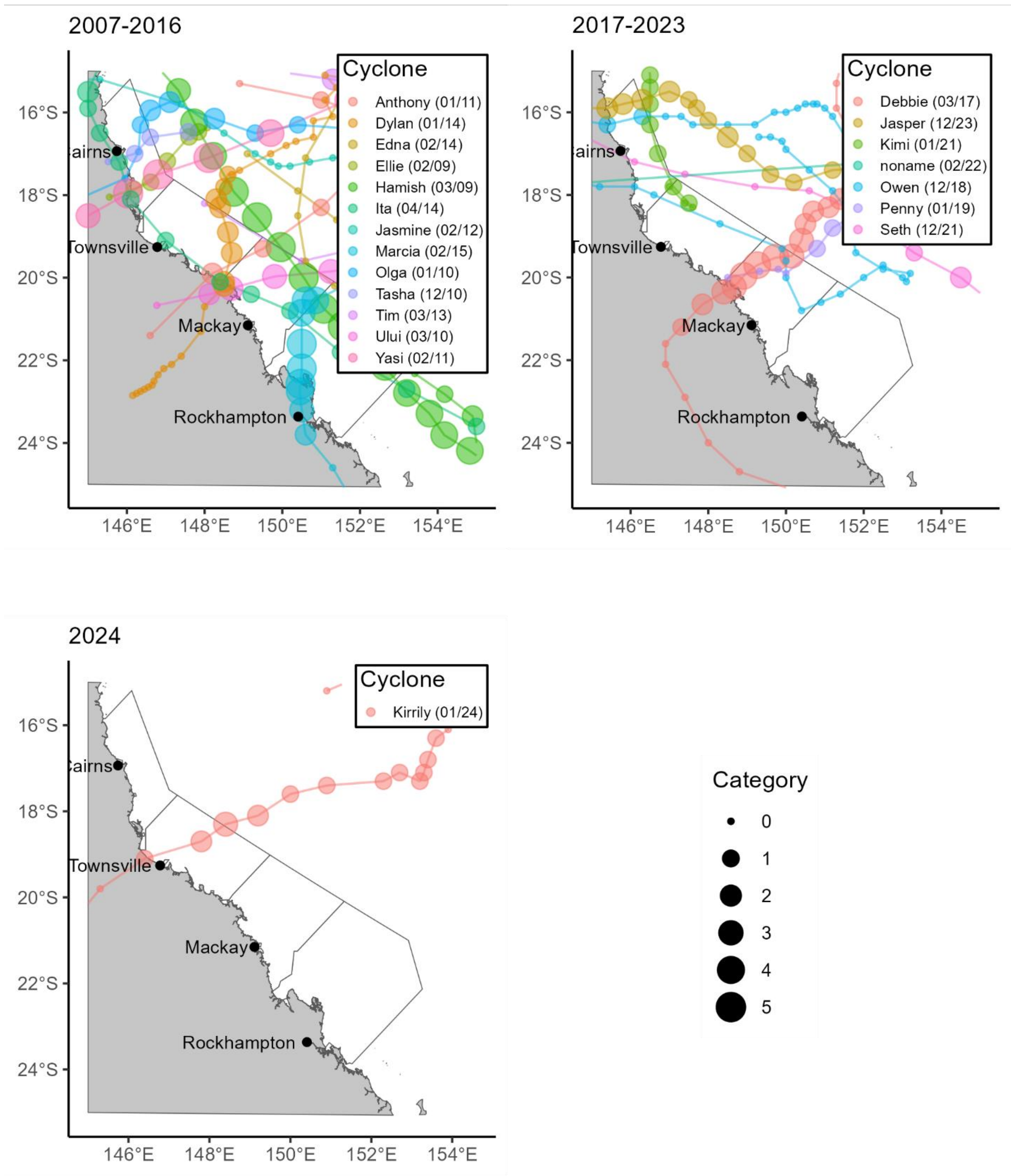

Figure 8 Cyclone tracks for systems crossing the inshore Reef since 2006. Tracks sourced from the BoM

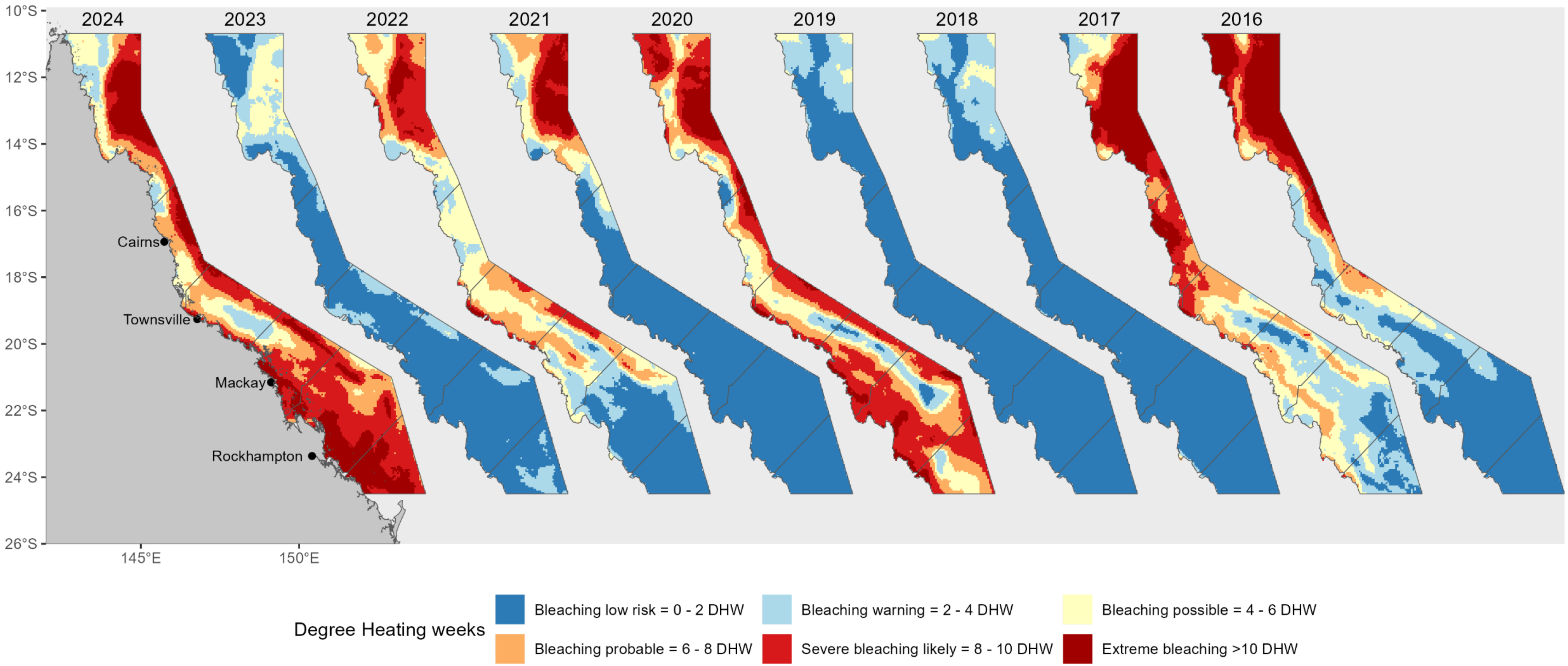

Figure 9 Annual DHW estimates for the Reef. Data are the annual maximum DHW estimates for each ~25 km^2 pixel. Data were sourced from NOAA coral reef watch.

3.3 Crown-of-thorns starfish

In 2024, the density of crown-of-thorns starfish were above outbreak levels at five of the six MMP reefs in the Johnstone Russell-Mulgrave sub-region (Table 9). A single individual was also observed on the LTMP sites at Fitzroy West. Of the inshore reefs reported by the MMP, outbreak densities of crown-of-thorns starfish (30 ha^{-1}) 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 two of the six 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'.

Year	Barron Daintree			Johnstone Russell-Mulgrave							
	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

Within the Johnstone Russell–Mulgrave sub-region, crown-of-thorns densities peaked at outbreak levels (> 30 individuals per hectare) at five of the six 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 2024, juvenile starfish were again present demonstrating their continued recruitment.

Table 10 Number of crown-of-thorns 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 the period between the MMP or LTMP survey in a given year and the previous survey of that reef. The catch rate per diver hour is given in bracket to provide an idea of relative population density. * Denotes reefs no longer surveyed by MMP or LTMP.

Year	Snapper Island	Low Isles	Green Island	Fitzroy Island	Frankland Group	Pelorus Island	Border Island	Hayman Island	Hook Island	Langford and Bird
2013	**135** (4.05)		**3226** (3.63)	**2743** (2.54)						
2014				**1586** (3.36)						
2015		**717** (1.07)	**3320** (2.04)	**348** (0.56)						
2016				**360** (1.12)						
2017		**129** (0.56)	**848** (1.12)	**108** (0.21)	**500** (1.07)					
2018				**4** (0.01)	**343** (0.74)					
2019			**194** (0.37)							
2020										
2021		**4** (0.03)		**2958** (1.10)	**6831** (3.36)					
2022		**2**(0.03)	**233** (1.82)	**122** (0.52)	**498** (1.50)		**11**(0.06)	**17**(0.22)	**116**(0.43)	
2023			**35** (0.05)	**3**(0.01)	**156**(0.26)			**6**(0.06)	**109**(0.21)	**4**(0.01)
2024			*	**4**(0.01)	**1088**(0.58)	**2**(0.02)		**1**(<0.01)		*

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 10, of cohorts 1–4, and percentage followed by number observed in parentheses observed during MMP scuba search surveys.

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					**55 (41)**	39 (29)	6 (4)
2013	24	**35**	31	10	15 (13)	**57 (41)**	28 (21)
2014	12	**42**	36	10	**57 (9)**		43 (6)
2015	**41**	39	16	4	**75 (3)**	25 (1)	
2016	**95**	4	0	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	**84**	14	2	0	29 (2)	**57 (4)**	14 (1)
2020	24	**62**	13	1	27 (19)	**49 (34)**	24 (17)
2021	17	**66**	16	1	6 (1)	25 (4)	**69 (11)**
2022	17	**62**	20	1	15 (2)	23 (3)	**62 (8)**
2023	16	**39**	12	33	**57(4)**	43(3)	
2024	**67**	31	2	0	**58(33)**	42(24)	

3.4 River discharge

Across the Reef, river discharge for the 2023-2024 water year exceeded 1.5 times the long-term median for the first time since 2018–19 (Figure 10). The primary driver of this pattern was heavy rainfall associated with cyclone Jasper that resulted in major flooding of the more northern catchments in the Wet Tropics (Table A5), and the southern catchments of Cape York (Moran *et al.* 2025). Discharge from catchments adjacent to the coral monitoring sites in the Burdekin, Mackay Whitsunday, and Fitzroy regions varied around long-term median values (Table A5).

The impact of these floods was most severe at Snapper Island South where all coral at both 2 m and 5 m deep monitoring sites were killed. Although the exact magnitude of the discharge from the Daintree was not measured, due to the loss of the river gauge during the flood, the complete mortality of corals supports the estimate that this was the highest discharge in recent years for this catchment (Table A5). Previously, major flooding of the Daintree River was recorded in 2018–19 when, in combination with minor storm damage attributed to pre-cyclone Owen, 38% of hard coral cover was killed at 2 m depth at Snapper Island South (Figure A1).

In previous years, the most extensive flood damage to monitored reefs occurred in 2011 in the Fitzroy region when flood waters from the Fitzroy River caused high levels of mortality among corals at 2 m depth on reefs to the south of Great Keppel Island (Figure A6, Table A6, Table A5).

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.

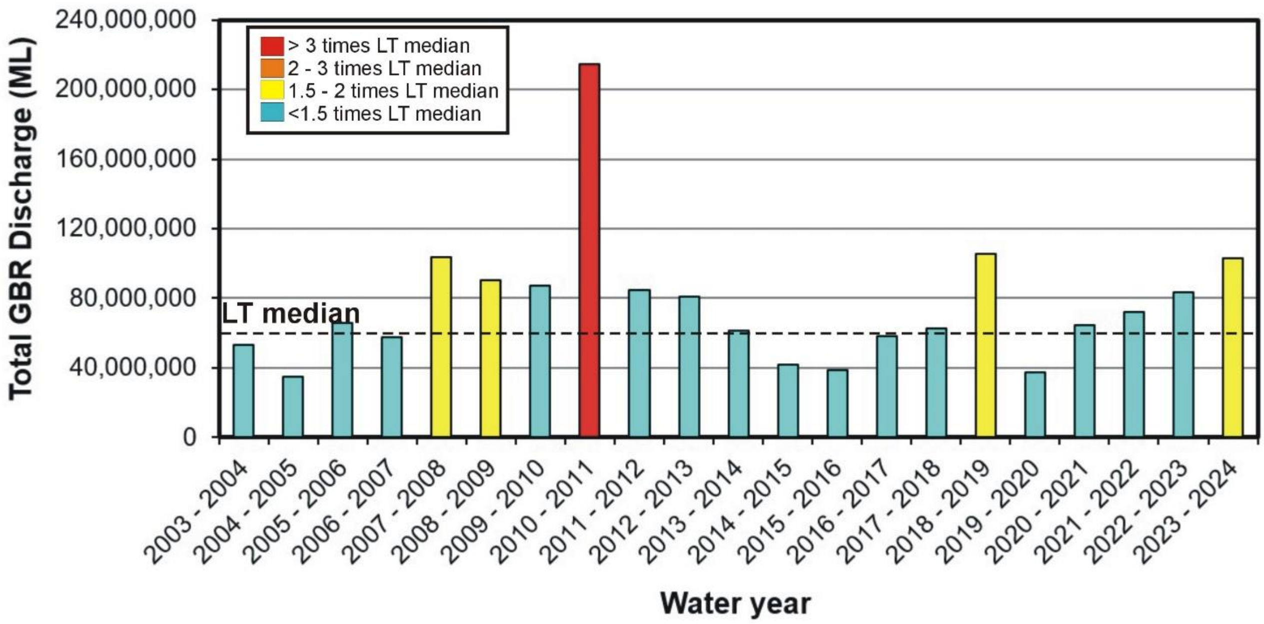

Figure 10 Annual total river discharge to the Reef. Annual estimates aggregate over the water year: 1 October to 30 September, for the 35 main Reef basins. Values are colour coded relative to proportion of long-term (LT) median (1986–2016) discharge. Figure source: Moran *et al*. 2024, data source: DNRM, http://watermonitoring.dnrm.qld.gov.au/host.htm

3.5 Water quality

Summaries of water quality data for each sub-region or region in which coral monitoring occurs are provided in figures (Figure A11 to Figure A16). These plots are sourced from the complimentrary annual MMP Inshore Water Quality annual report (Moran *et al*. 2025). 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 Moran *et al*. (2025). 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. Whereas 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 (4.3–4.6)
- Coral community condition along water quality gradients (4.7.1)
- Influence of discharge, catchment loads and discharge on reef recovery (4.7.2)

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 additional information tables 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 2024 remained ‘poor’, having declined to the lowest value since scoring began in 2006 (Figure 11). The recent decline is strongly influenced by the severe impacts that cyclone Jasper and associated flooding had on reefs in the northern areas of the Wet Tropics, mirrored in the south by high sea temperatures and subsequent coral bleaching that severely reduced coral cover in the Fitzroy region. These recent events compound the decline from high point in condition observed in 2016 as the effects of cyclone Debbie in 2017, high sea temperatures causing coral bleaching, predation of corals by crown-of-thorns starfish, and previous flooding of the Daintree River took their toll (Figure 8, Figure 9, Table 9, Table A5).

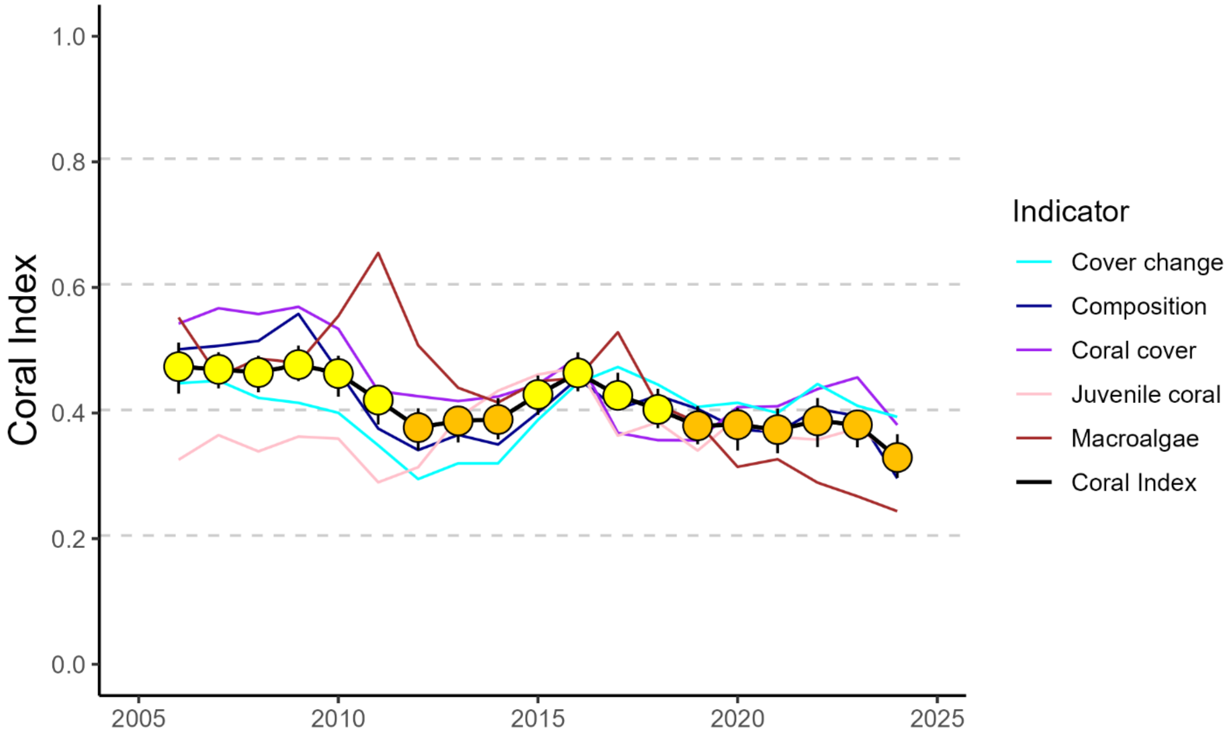

Figure 11 The Reef level trend in Coral Index and indicator scores. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = ‘poor’, yellow = ’moderate’. Error in Coral Index scores were derived from bootstrapped distribution of regional indicator scores weighted by the relative area of inshore coral reefs in each region.

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 the Coral Index scores declined in the face of multiple disturbances (Figure 12). Of particular concern is the ongoing decline in the Macroalgae indicator score as this suggests increasing downward pressure on coral community recovery (Figure 11).

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 are documented to have caused 39% of all coral cover losses on inshore reefs (Figure 12, Table A6). 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.

When interpreting Figure 12 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 six of the seven 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 and 2020 are represented by losses attributed to bleaching in 2018 and 2021 (Figure 12). In these instances, corals were still bleached at the time of surveys in 2017 and 2020, and the subsequent loss of cover was attributed to a delayed response to thermal stress. This point is relevant for the losses attributed to coral bleaching in 2024. At the time of surveys in May 2024 a high proportion of corals in the Fitzroy region were bleached (Figure 31) and it is likely this ongoing stress will result in further losses being attributed next year.

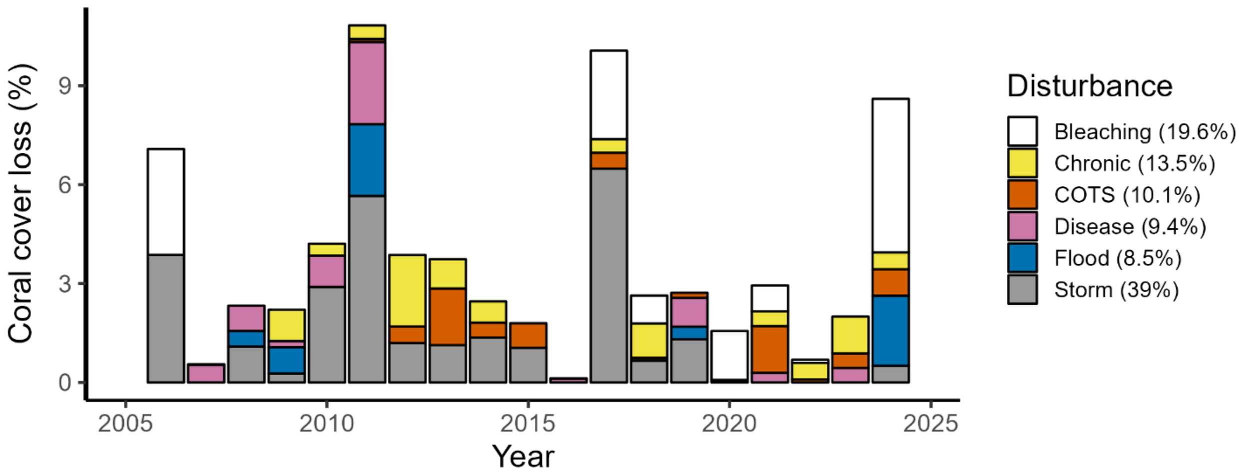

Figure 12 Hard coral cover loss by disturbance type across the inshore Reef. Length of bars represents the mean loss of cover across all reefs in each year. Colours represent the identified cause of cover loss. COTS = crown-of-thorns starfish

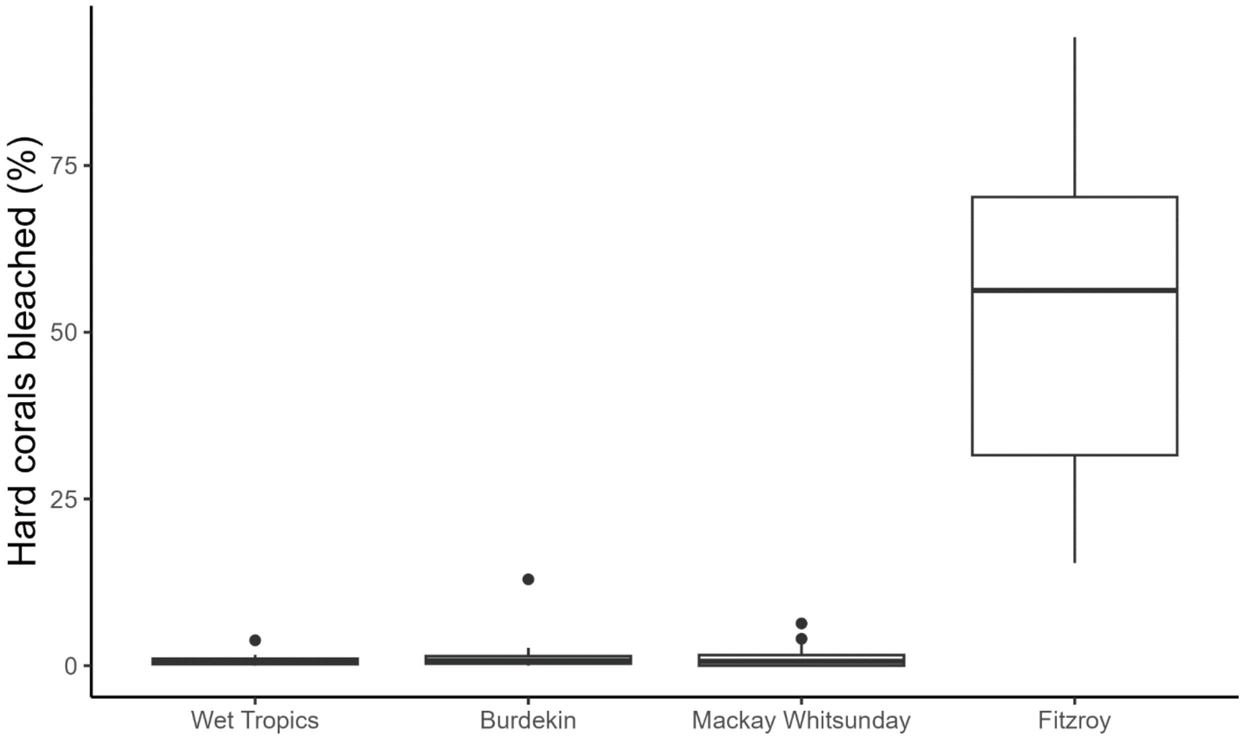

Figure 13 Proportion of hard coral cover bleached at the time of survey in 2024. Data informing each box are the proportion of hard coral points on photo point-intercept transects classified as being fully bleached (white) or partially bleached (obviously pale) at each reef and depth combination within each region.

Thermal bleaching events have contributed to 19.6% of the coral cover losses since 2005. 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 12). It is likely that some losses of cover recorded as disease in 2007 and chronic stressors in 2017, 2018, 2021 and 2022 were also influenced by stress imposed by high water temperatures.

While crown-of-thorns starfish have caused moderate losses (10.1%, Figure 12), 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, 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. Prior to this, loss of corals from direct exposure to low salinity flood waters 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, the 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 the coral cover (Lam *et al.* 2018) and by extension, Coral Index scores in all regions.

4.3 Wet Tropics region

4.3.1 Regional trend

Coral communities within inshore areas of the Wet Tropics remain in 'moderate' condition. However, there has been a decrease in the Coral Index scores between 2023 and 2024 driven by declines in all indicator scores. The relatively stable condition observed from 2016 to 2022 (Figure 14) masks differing trends within 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. At the regional level, in 2024 Juvenile coral is the first indicator to have fallen below moderate levels since 2014 (Figure 14).

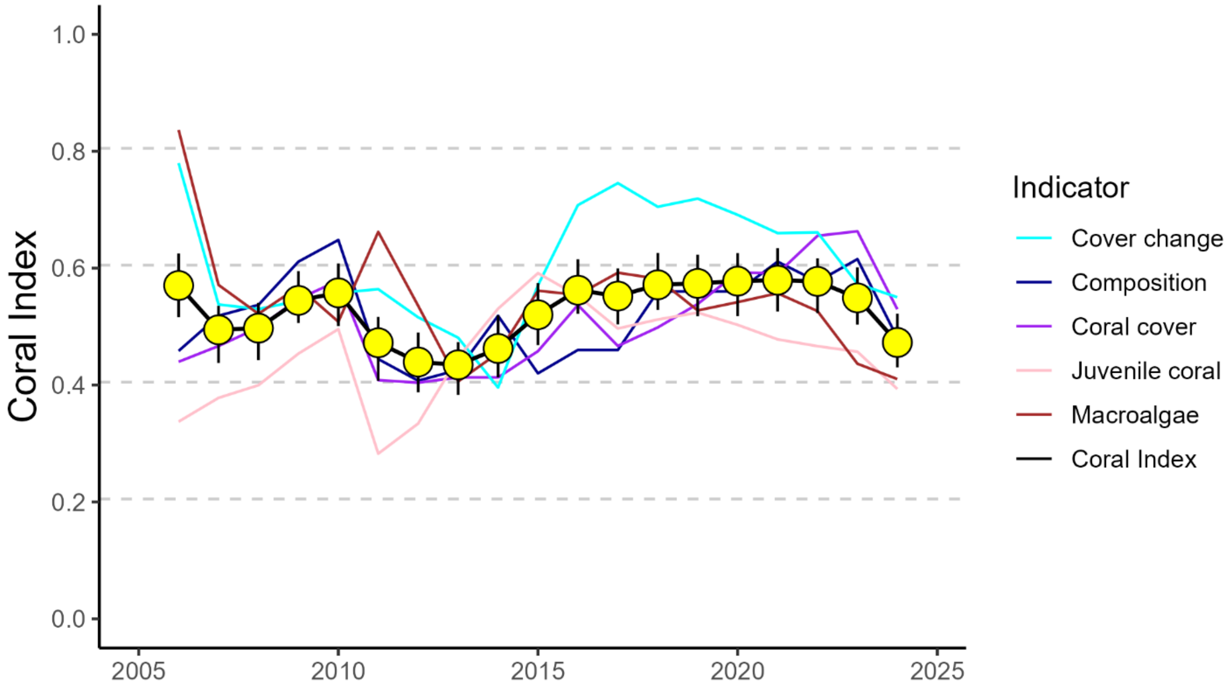

Figure 14 Wet Tropics region Coral Index and indicator trends. Coral Index scores are coloured by report card category: yellow = 'moderate'. Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at individual reefs.

4.3.2 Barron–Daintree sub-region

The condition of coral communities decreased from 2023 to 2024 but remained ‘moderate’ (Figure 15). Coral Index scores had been improving to 2023 since a low point in 2019 following coral bleaching in 2017, and exposure to floodwaters and cyclone Owen in 2019 (Figure 16e, Table 12, Table A6). In December 2023 this region was hit by cyclone Jasper that caused both physical damage to corals and precipitated extreme rainfall that caused major flooding. 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). A more pronounced reduction in the Coral Index was avoided due to improved scores for the Macroalgae and Cover change (at 5 m depth) indicators, while most other indicators declined (Figure 15, Table 12). It should also be noted that surveys at Low Isles preceded the arrival of cyclone Jasper and as such the currently reported ‘good’ Coral Index score at that reef should be considered with caution (Table A7).

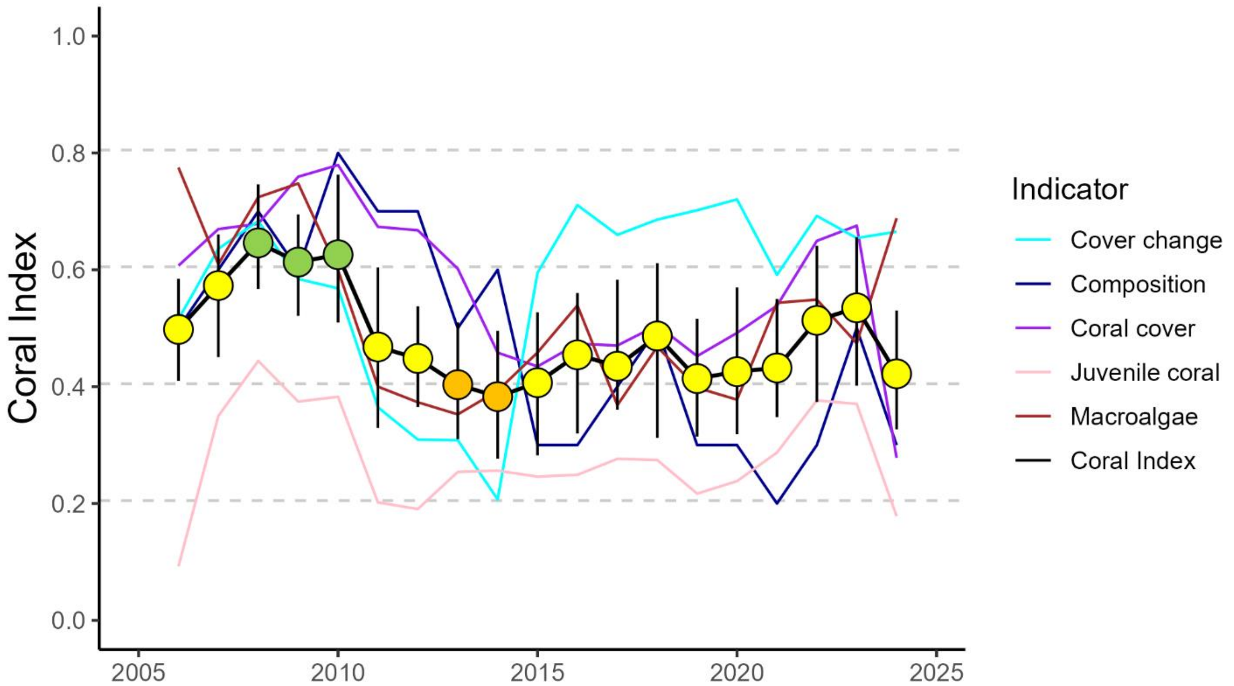

Figure 15 Barron–Daintree sub-region Coral Index and indicator trends. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = ‘poor’, yellow = ’moderate’ and green = ’good’. Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at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

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). The resulting loss of coral cover between 2023 and 2024 was attributed to storm and flood (Figure 16e). 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 at Daintree River (4.8 times) and Baron River (5.8 times) (Table A5).

The Coral cover indicator score was categorised as ‘poor’ (0.28, Table A7, Figure 15), having decreased significantly from ‘good’ in 2023 (Table 12, Figure 17a). From 2019 to 2023 this indicator had steadily improved (Table 12, Figure 16e). In 2024 all corals at both 2m and 5 m depths at Snapper South were killed as the reef was inundated by freshwater. Corals at the 2 m depth at Snapper North were also impacted by cyclone Jasper through a combination of storm damage and freshwater exposure, with some coral bleaching also observed (Figure A1, Figure A7, Table A9, Table A10). 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 51% in 2023 to 21% in 2024, a figure that includes the data from Low Isles obtained before any impact over the 2023-24 summer.

Table 12 Barren–Daintree sub-region Coral Index and indicator score changes. Data compare the changes in scores between local maxima and minima in the Coral Index time-series. 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 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 2024	2	-0.21	0.88	-0.50	0.91	0.08	0.77	-0.53	0.79	-0.08	1.00	0.00	NA
	5	-0.05	0.61	-0.33	0.75	0.30	0.87	0.04	0.56	0.07	0.76	-0.33	0.72

The Cover change indicator remained 'good' (0.67, Table A7) reflecting the rate at which hard coral cover was recovering prior to the recent summer disturbances. Although the score did decline slightly at 2 m depth the scores at both Snapper South and Snapper North remained 'good' based on recovery over the period 2019-2023 (Table 12, Table A7).

The Composition indicator decreased to 'poor' in 2024 (0.3, Table A7, Figure 15). This result reflecting declines at Snapper South 5 m depth where all coral was killed (Figure A1, Table A9, Table A10). The scores did not change at other reefs and remained at 0.5 or, at Snapper North 2 m depth, 0. 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 improved to 'good' (0.69, Table A7, Figure 15). This improvement reflects reduced cover of macroalgae at Snapper Island (Figure A1). It is likely these reductions reflect similar impacts of cyclone Jasper and exposure to floodwaters as described for the coral communities. Although the score for this indicator has improved across the sub-region, the score at Snapper North 2 m depth remained 'very poor' despite a large reduction in macroalgae cover in 2024 (Table A7, Figure A1).

The Juvenile coral indicator has declined to 'very poor' (0.18, Table A7, Figure 15) reflecting the 'very poor' scores at all Snapper Island sites (Table A7). This decline was driven by the total loss of juvenile corals at Snapper South, and large reductions in juvenile densities at Snapper North (Figure A1). In contrast, the survey from Low Isles, prior to the summer impacts, showed an increase in the density of juvenile corals sufficient to return a "very good' score (Figure A1, Table A7).

The was no clear impact of the floods on the regions Water Quality with both long-term and short-term water quality indices remaining 'good' in 2024 (Figure A11a). The concentration of NOx continued to exceeded guideline values but has tended to decline in recent years (Figure A11c). In contrast the concentration of Phosphate increased to breach guideline values for the first time since 2017 (Figure A11d). It should be noted these water quality data include sampling from both before during and after the wet season (Moran *et al*. 2025). Over the period 2020–2024, 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 16a, b, Table A8).

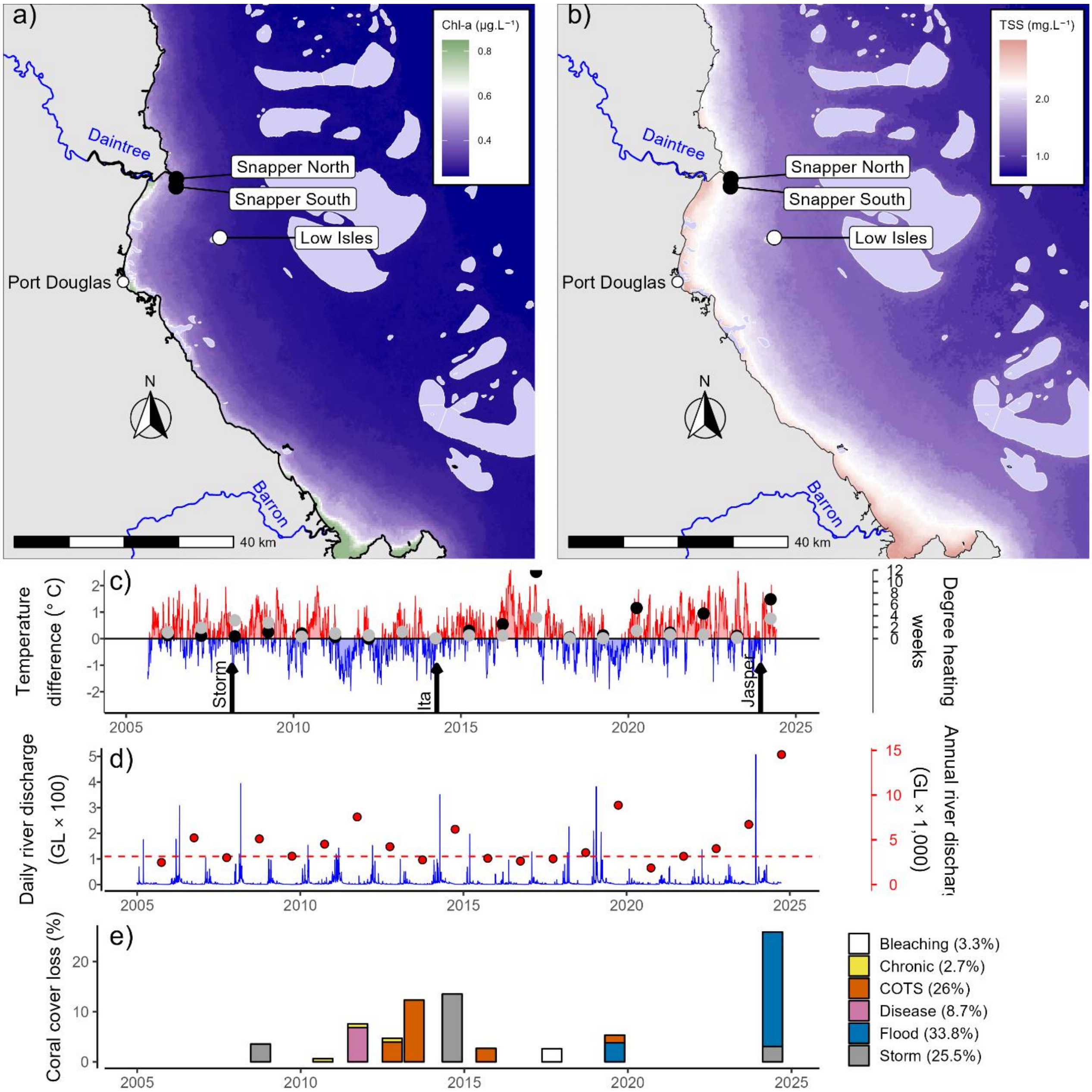

Figure 16 Barron–Daintree sub-region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) median wet season Chl *a* and b) median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2020–2024, 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.

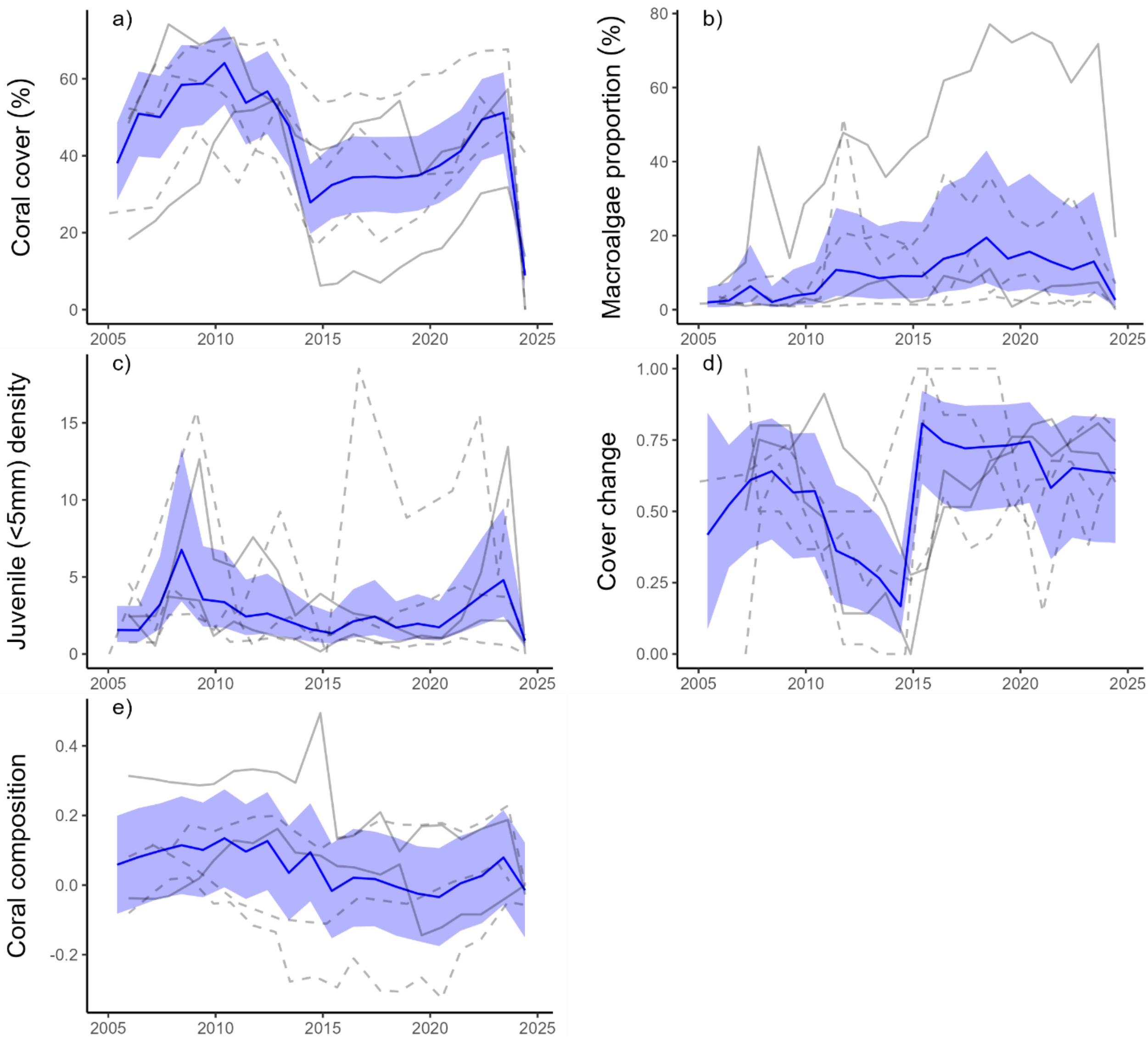

Figure 17 Barron–Daintree sub-region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.3.3 Johnstone Russell-Mulgrave sub-region

The 2024 Coral Index score was categorised as 'moderate', having declined since 2021 (Figure 18,Table 13). Most consistent trends in this recent decline have been declines in the Macroalgae and Composition indicators, and at 2 m depth for Coral cover (Table 13). The decline in Coral cover was most noticeable in 2024 (Figure 18). Prior to recent declines, 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). The Coral Index had stabilised around the threshold between 'moderate' and 'good' scores from 2016 to 2021 (Figure 18).

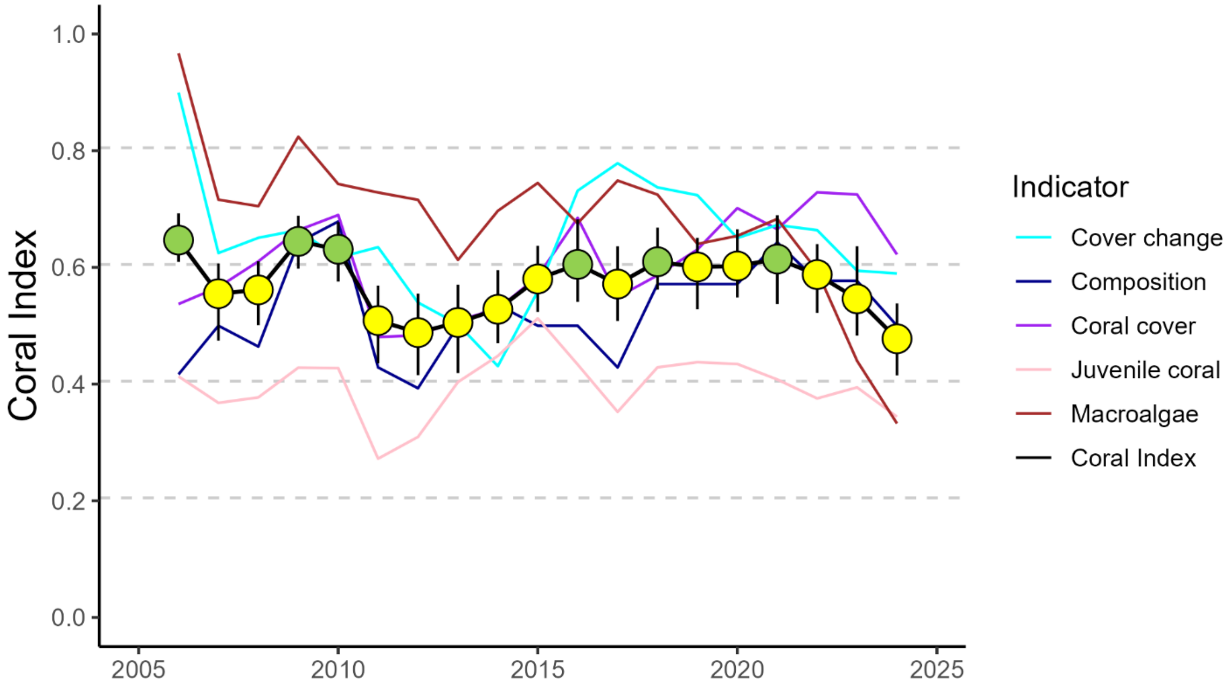

Figure 18 Johnstone Russell–Mulgrave sub-region Coral Index and indicator trends. Coral Index scores are coloured by Reef Water Quality Report Card categories: yellow = 'moderate' and green = 'good'. Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at individual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

Table 13 Johnstone Russell–Mulgrave sub-region Coral Index and indicator score changes. Data compare the changes in scores between local maxima and minima in the Coral Index time-series. 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 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	-0.03	0.56	-0.02	0.53	-0.02	0.51	-0.02	0.66	-0.10	0.61	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 2024	2	-0.15	0.85	-0.15	0.80	-0.31	0.82	-0.03	0.58	-0.09	0.61	-0.17	0.74
	5	-0.15	0.90	-0.02	0.57	-0.48	0.89	-0.01	0.51	-0.10	0.65	-0.14	0.72

The downward trend of the 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 A6). Cyclone Jasper caused substantial damage to coral communities not only via physical action, but by associated flooding, with the Russell-Mulgrave and Johnstone rivers exceeding their median flow by 1.6 and 1.7 times, respectively (Table A5).

In 2024, crown-of-thorns starfish were above outbreak levels at most reefs (Table 9, Figure A9). It was only High West where no crown-of-thorns were observed during the most recent surveys (Table 9). This was an increase upon 2023 where starfish were seen at two reefs, Fitzroy East and Franklands East, and were only above outbreak levels at Franklands East (Table 9). At both Frankland group and Fitzroy Island starfish had been removed by the Crown-of-thorns Starfish Program between 2023 and 2024 surveys (Table 10).

The Coral cover indicator score for 2024 was categorised as ‘good’ (0.62 Table A7), despite a decline from 2023 (Figure 18). Coral cover, particularly of *Acropora* spp., declined at Fitzroy West MMP sites and Fitzroy East 2 m, the primary cause of this decline was attributed to coral bleaching due to elevated sea water temperatures (Figure 9, Figure 19e, Figure A2). Physical damage was also observed at Fitzroy West suggesting waves associated with cyclone Jasper contributed to the reduction in coral cover. Substantial declines in coral cover at High East 2 m were attributed to a combination of wave damage and exposure to fresh floodwaters as corals were both broken and slightly shallower than a clear depth contour demarcating almost complete mortality of Acroporidae in shallow waters (*per obs*. the Authors). At 5 m depth at High East and also at Franklands East coral cover declines reflected reduced cover of *Acropora* spp. in particular, that was attributed to the ongoing presence of crown-of-thorns starfish (Table A6, Figure A2).

In contrast, hard coral cover increased at the 5 m depths at Fitzroy East and Franklands West, and both depths at High West (Figure A2). Coral cover also increased at Fitzroy West LTMP however, it should be noted this site was surveyed prior to elevated sea temperatures and passage of cyclone Jasper over the 2023-24 summer (Figure A2).

The Cover change indicator score remained categorised as ‘moderate’ (0.59, Table A7) in 2024 with very little change from 2023 (Figure 18). The only reef at which the rate of increase in coral cover has fallen below modelled expectations over the four-year window assessed by this indicator was High East (Table A7).

The Composition indicator has remained ‘moderate’ in 2024, although this score has declined across both depths since 2021, reflecting the disproportionate loss of *Acropora* cover (Figure 18, Table 13).

The Macroalgae indicator score has declined to ‘poor’ in 2024 (0.33, Table A7), with significant decline for this indicator at both depths since 2021 (Figure 18, Table 13). Across the region, the cover of the persistent brown macroalgae species typical of many inshore reefs is very low (Table A11). Low Macroalgae scores in this region reflect dense mats of red macroalgae species (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 2024 at High East, Franklands West, and Fitzroy West 2 m 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 varied around the boundary between ‘moderate’ and ‘poor’ since 2016 (Figure 18).

In 2024, the concentrations of dissolved N and P (NOx and PO4) and particulate N exceeded guideline values (Figure A12c, d, h). The concentration of NOx has however shown a declining trend since the redesign of the sampling program for water quality in 2015 (Moran *et al.* 2025). The short-term water quality index remained ‘moderate’ having lingered close to the boundary of ‘good’ since 2019 (Figure A11a). Over the period 2020–2024, 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 19a, b, Table A8).

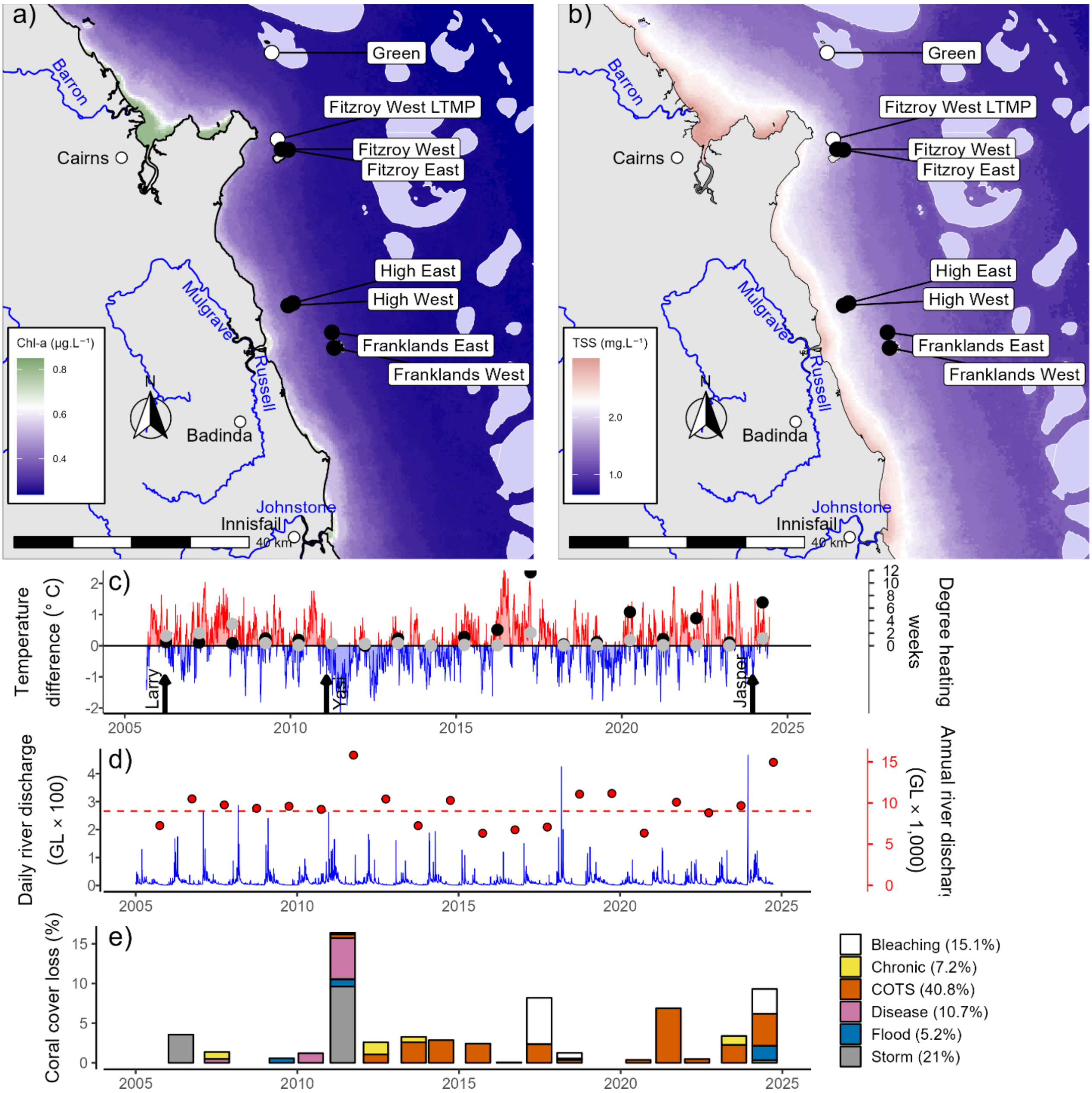

Figure 19 Johnstone Russell–Mulgrave sub-region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) median wet season Chl *a* and b) median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2020–2024, 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.

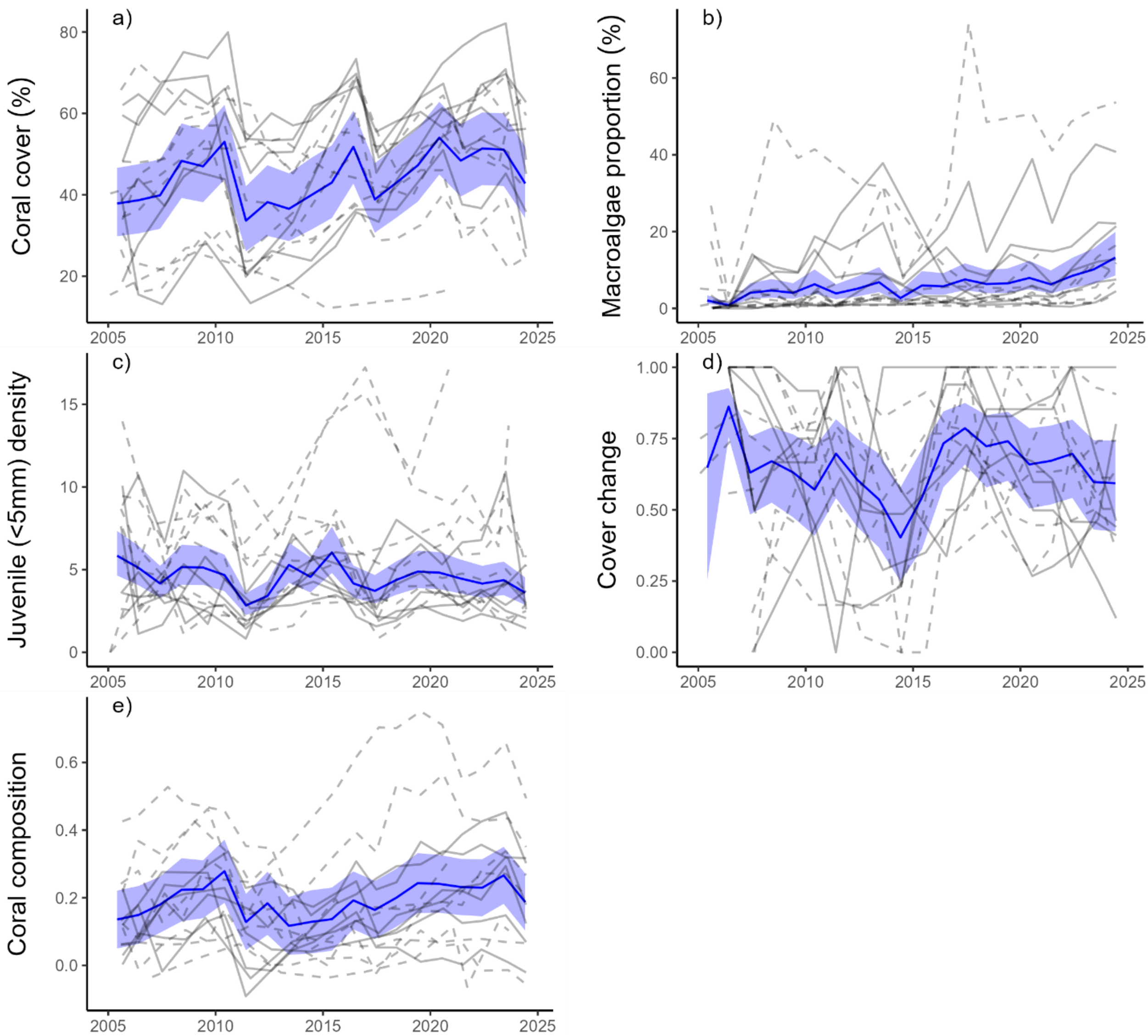

Figure 20 Johnstone Russell–Mulgrave sub-region indicator trends. a– e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.3.4 Herbert–Tully sub-region

The Coral Index was categorised as 'moderate' in 2024 (Figure 21). All indicators have declined between 2023 and 2024, except for Juvenile coral which stayed the same and is the only indicator still categorised as 'good' (Figure 21, Table A7). Since 2020 the Coral Index has steadily declined, despite the general increase in Coral cover scores over this period (Figure 21, Table 14).

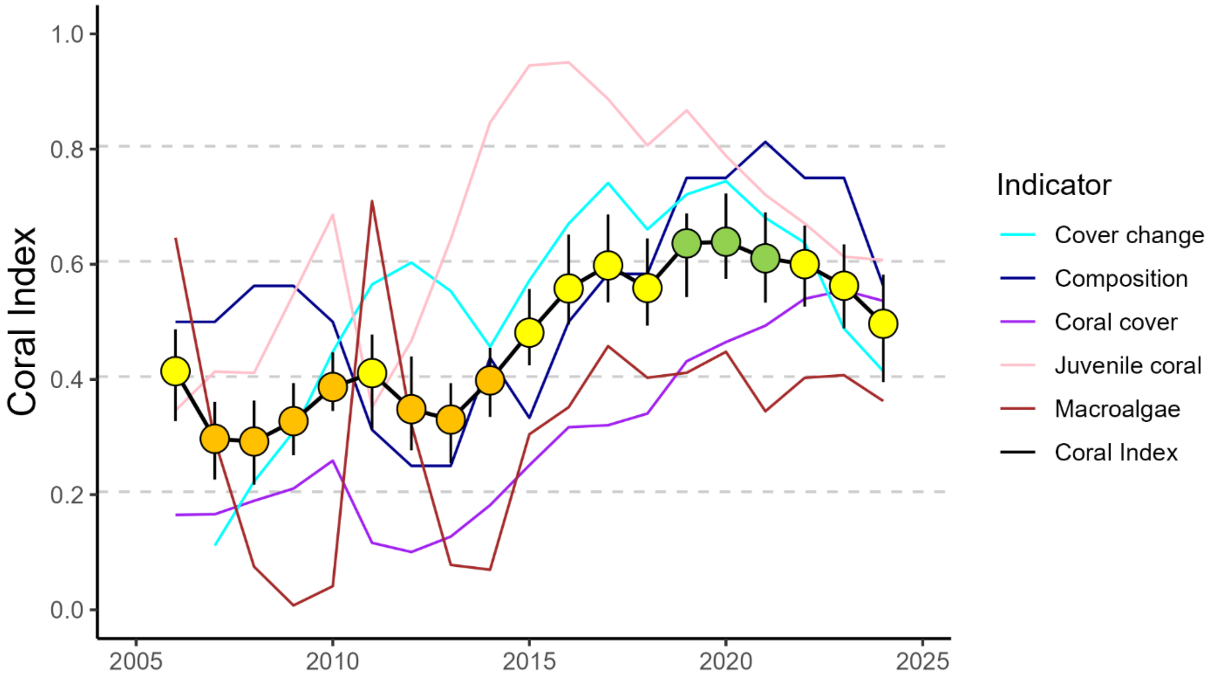

Figure 21 Herbert–Tully sub-region Coral Index and indicator trends. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = 'poor', yellow = 'moderate' and green = 'good'. Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at individual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

Table 14 Herbert–Tully sub-region Coral Index and indicator score changes. Data compare the changes in scores between local maxima and minima in the index time-series. 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 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 2024	2	-0.05	0.63	0.09	0.80	-0.04	0.54	-0.16	0.81	-0.21	0.86	0.00	NA
	5	-0.21	0.93	0.06	0.82	-0.13	0.73	-0.20	0.84	-0.45	0.89	-0.17	0.72

Since 2020, minor losses in hard coral cover at some reefs have been attributed to high water temperatures that caused coral bleaching in 2020 and 2024, and above median levels of coral disease (Figure 9, Figure 22c, e, Figure A8).

The Coral cover indicator score steadily improved from ‘very poor’ in 2012 thought to ‘moderate’ in 2023 as coral cover rebounded from the impact of Cyclone Yasi (Figure 21, Table 14, Figure 23a). In 2024 the score remained ‘moderate’, however recovery stalled due to the slight declines in coral cover observed at Barnards and Dunk North (Figure 21, Figure A3). Both these declines were driven by losses of *Acropora* spp. with relatively high levels of disease observed (Figure A8). The losses at Dunk North were attributed to thermal bleaching (Table A6). The 5 m depth at Bedarra showed a small loss of “other hard corals”, while cover at the 2 m depth increased slightly (Figure A3). Dunk South showed increased cover of *Montipora* spp. and Merulinidae (Figure A3).

The plateau in coral cover in recent years is reflected in the Cover change score that while still categorised as ‘moderate’ (0.41, Table A7) is approaching the border of ‘poor’ (Figure 21, Figure 23d, Table 14). In 2024, this indicator was ‘good’ at Dunk North 5 m, however it was ‘very poor’ for Barnards 5 m, and ‘poor’ for Dunk South 5 m and Bedarra both depths (Table A7). During the period of 2020 to 2024, and especially in 2024, levels of disease were above median levels (Figure A8). Although disease was not categorised as an acute disturbance (except for in 2021), the reduced growth or mortality of infected colonies will have influenced the rate of change in hard coral cover and the losses of coral attributed to chronic pressures (Figure 22e).

The Composition score for this region has fallen to ‘moderate’ (0.56, Table A7) for the first time since 2018 (Figure 21).

The Macroalgae indicator has dropped back to ‘poor’ (0.36, Table A7) in 2024 after being classed as ‘moderate’ in 2023 (Figure 21, Table A7). The scores for this indicator are highly variable between reefs with minimum values of zero at the 2 m depth at both Dunk North and Bedarra and the 5 m depth at Dunk South, and a maximum value of 0.79 at 5 m depth at Barnards (Table A7). At reefs with a value of zero, the macroalgae community is dominated by brown algae of the genus *Lobophora* and family Sargassaceae (Table A11).

The Juvenile coral indicator remains categorised as ‘good’ although it has been declining since 2014 (Table 14, Table A7). In 2024 there was an increase in juvenile corals at Dunk South 2 m and 5 m for the first time since 2019 and 2014, respectively, and at Dunk North for the first time since 2021 (Figure A3). There was also a slight increase at Bedarra, while Barnards declined (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. Moderating scores for this indicator reflect that these corals have either died or grown beyond the juvenile size classes (Figure 23c, Figure A3).

In 2024, most water quality parameters exceeded the guideline values (Figure A13). The concentration of NOx has however shown a declining trend (improving) since the redesign of the sampling program for water quality in 2015 and the concentrations of PN have oscillated over this period (Figure A12, Moran *et al.* 2025). Despite these exceedances, they were not sufficiently large to affect the short-term water quality index which remained ‘moderate’ (Figure A12). Over the period 2020–2024, wet-season concentrations of Chl *a* and TSS, as estimated from satellite imagery, were marginally below wet-season guideline values at all coral monitoring locations (Figure 22a, b, Table A8).

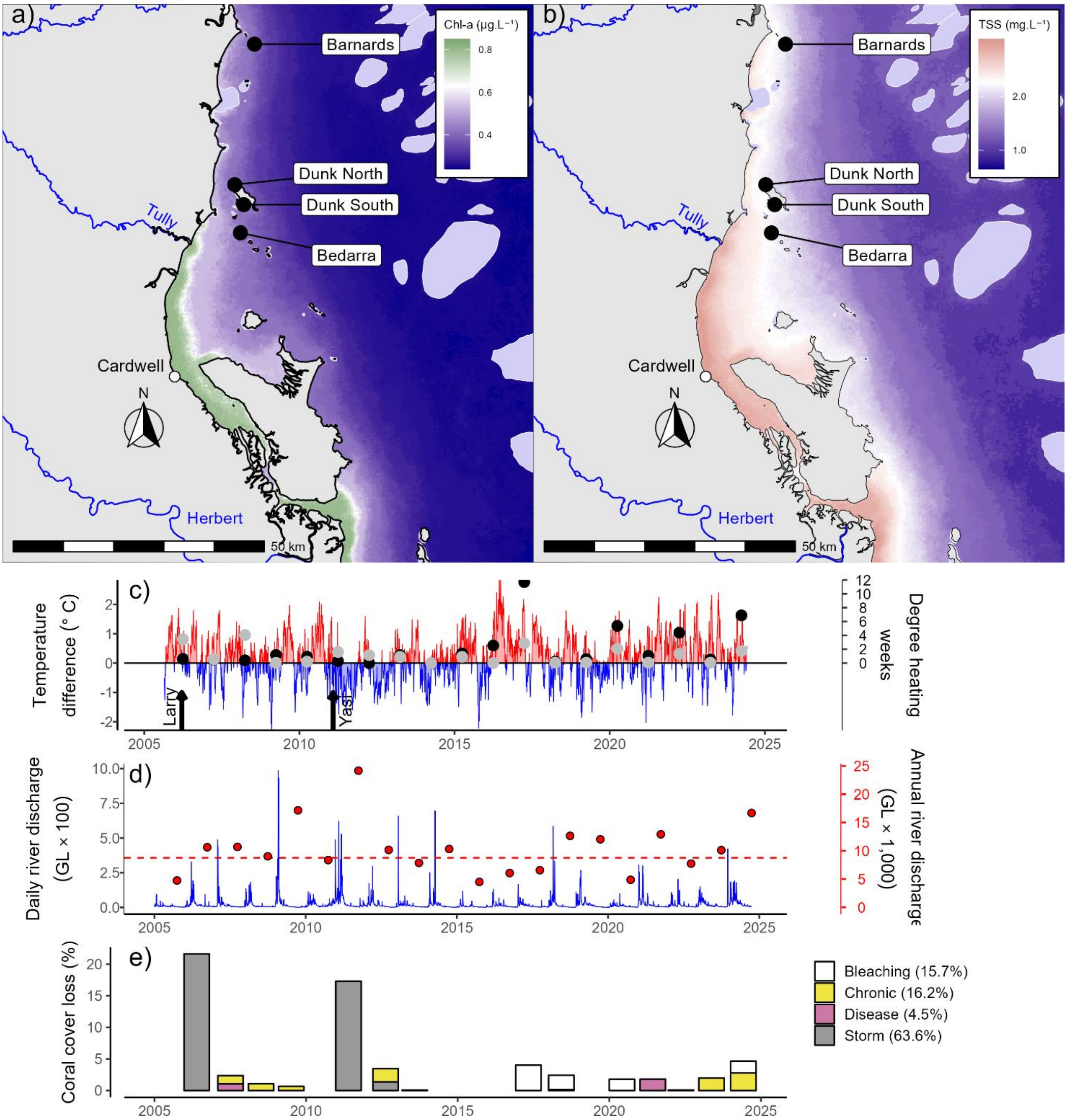

Figure 22 Herbert–Tully sub-region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) median wet season Chl *a* and b) median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2020–2024, white 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.

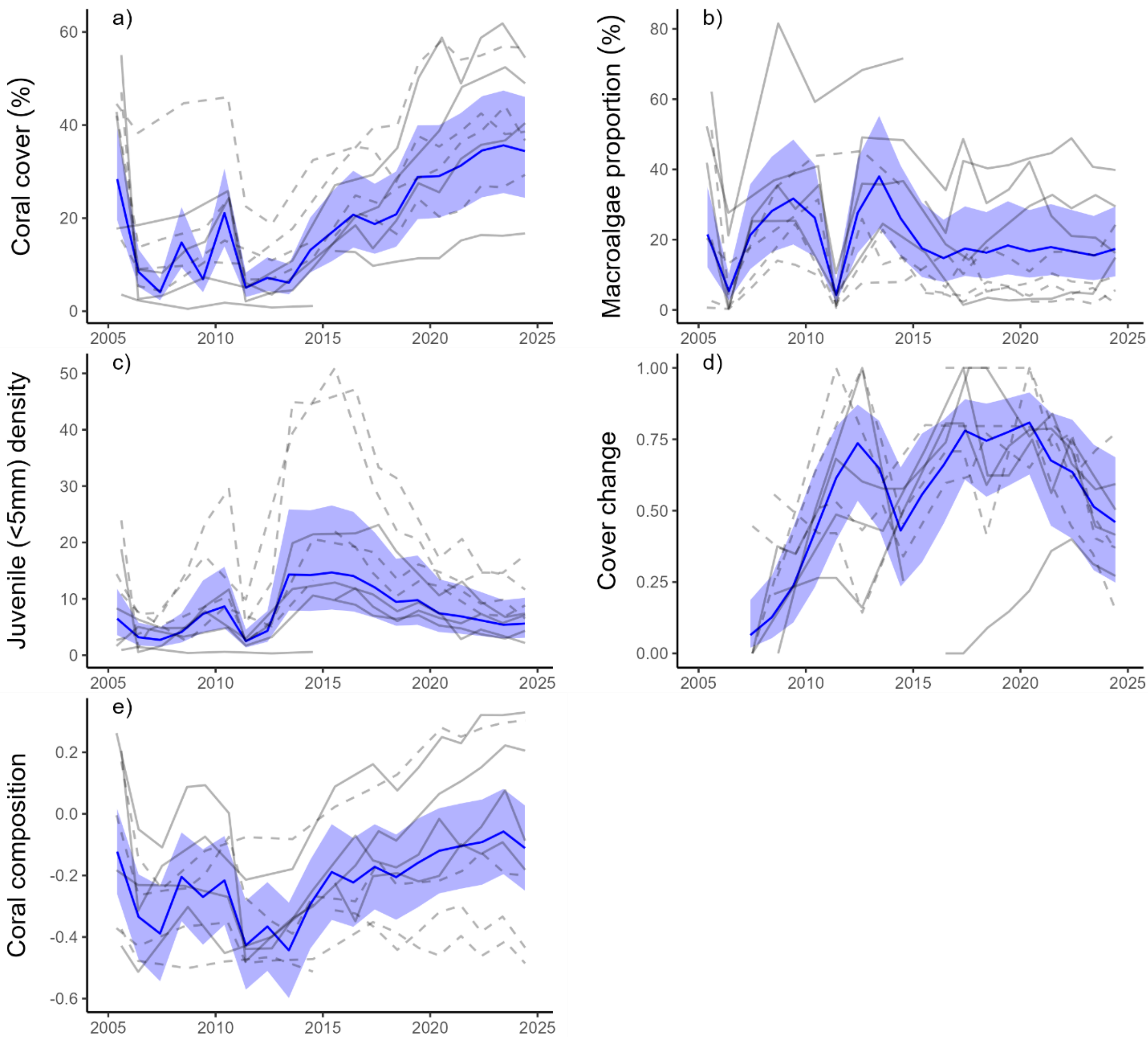

Figure 23 Herbert–Tully sub-region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.4 Burdekin region

The Coral Index remained within the 'moderate' range but has continued to decline from a high point observed in 2020 (Figure 24). All indicators except Coral cover remain below the score for 2020 (Figure 24). The most consistent declines from 2020 occurred at 2 m depths for Macroalgae, 5 m depths for Composition, and both depths for Juvenile coral (Figure 24, Table 15). In 2024, Macroalgae and Juvenile coral scores were categorised as 'poor' despite Macroalgae improving at the 5 m depths since 2020 (Figure 24, Table 15).

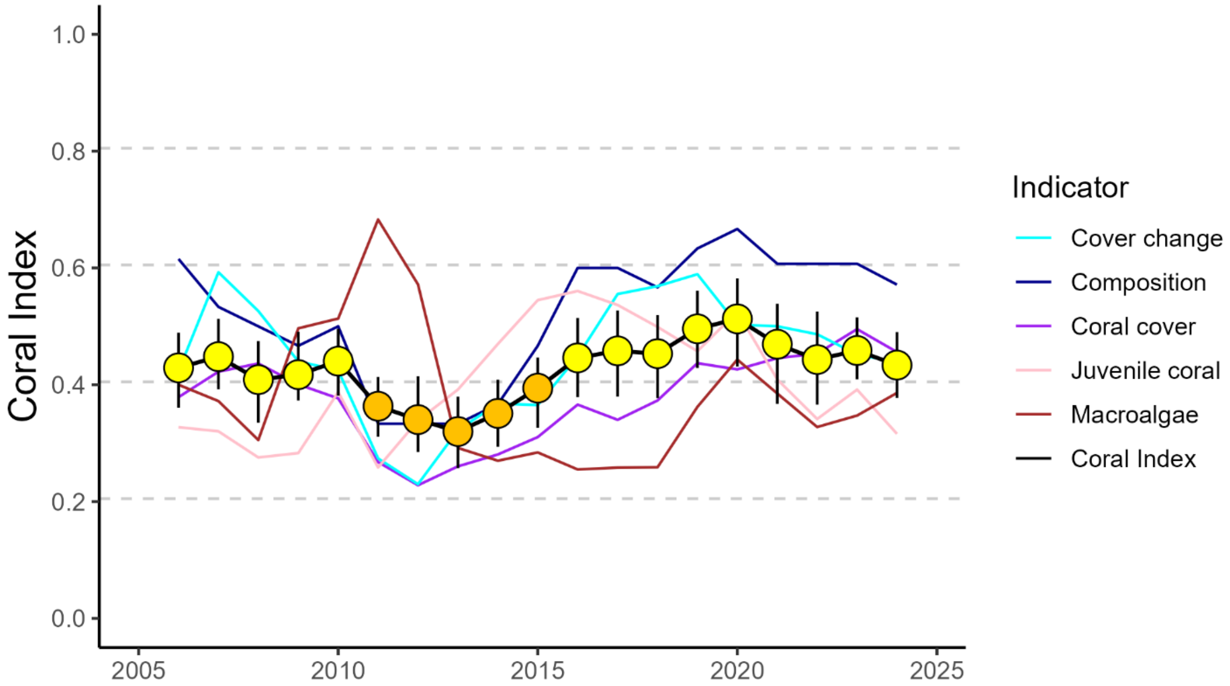

Figure 24 Burdekin region Coral Index and indicator trends. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = 'poor', yellow = 'moderate'. Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at individual reefs.

Table 15 Burdekin region Coral Index and indicator score changes. Data compare the changes in scores between local maxima and minima in the index time-series. 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 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 2020	2	0.14	0.80	0.17	0.80	0.16	0.75	-0.03	0.54	0.0	0.51	0.42	0.75
	5	0.26	0.93	0.22	0.89	0.18	0.77	0.26	0.87	0.33	0.89	0.31	0.76
2020 to 2024	2	-0.09	0.96	0.03	0.61	-0.26	0.73	-0.11	0.90	-0.04	0.57	-0.08	0.67
	5	-0.08	0.81	0.04	0.68	0.04	0.70	-0.28	0.84	-0.08	0.58	-0.13	0.71

Reefs in the Burdekin region where exposed to two acute disturbances over the 2023–24 summer. Most notably, marine heat wave conditions that caused coral bleaching and cyclone Kirrily that crossed the coast on 25th January 2024 caused minor storm damage (Figure 8, Figure 9, Figure 25e,

Table A6). This recent bleaching event adds to impacts of bleaching that occurred as a result of marine heat wave conditions in 2017, 2020 and to a lesser degree 2022 (Figure 9, Figure 25e). Prior to this period of marine heat wave conditions the reefs in this region were severely impacted by cyclone Yasi in 2011 (Figure 24, Figure 25e, Table A6). Note that in Figure 25e the impacts from cyclone Yasi, and the 2017 and 2020 bleaching events span two years due to a combination of some reefs not being resurveyed in the winter immediately following the disturbance, and for bleaching the full impacts are not realised until the following year.

The Coral cover indicator score remained categorised as ‘moderate’ (0.46, Table A7), with year on year changes between 2020 to 2024 reduced, compared to the steady recovery observed from 2013 to 2020 (Figure 24, Table 15). In 2024, coral cover declined at Palms East, Lady Elliot and Magnetic with declines mostly due to reduced cover of *Acropora*, and at Lady Elliot 5 m depth, *Galaxea* (family Euphylliidae) (Figure A4). In contrast coral cover remained stable or increased at other reefs (Figure A4).

The Cover change indicator score has been declining since a peak in 2019 but remains “moderate’ (0.44, Table A7) (Figure 24) as the average rate of increase in hard coral cover over the last four years remains within modelled expectations. However, the rate of hard coral recovery was ‘poor’ at many reefs, including both depths at Palms East and Havannah, the 2 m depths at Lady Elliot and Magnetic, and the 5 m depth at Palms West (Table A7).

The Composition indicator score is ‘moderate’ (0.57, Table A7) and has decreased at the 5 m depths since 2020 (Figure 24, Table 15).

The Macroalgae indicator has continued to decline at the 2 m depths since 2020 and remains categorised as ‘poor’ (0.39, Table A7, Table 15). The scores for this indicator vary drastically between reefs and depths, with five sites above 0.86 ‘very good’, one site at 0.58 ‘moderate’, and the other eight sites below 0.02 ‘very poor’ (Table A7). Very poor scores were recorded for both depths at Havannah North, Havannah, Pandora North, Magnetic, and the 2 m depths at Pandora and Lady Elliot where macroalgae cover is high (Table A7, Figure A4). Although these scores are very poor, there has been a reduction in macroalgae cover since 2023 at 2 m depths for Pandora and Havannah, and for both depths at Magnetic (Figure A4). Where the cover of macroalgae was high, the macroalgal communities were dominated by large brown species of the genus *Lobophora* and/or family Sargassaceae, the exception was Lady Elliot (2 m) where the red macroalgae *Hypnea* was common (Table A11).

The Juvenile coral indicator remained categorised as ‘poor’ (0.32) however, the scores were highly variable among reefs, ranging from 0.11 ‘very poor’ to 0.78 ‘good’ (Table A7). Juvenile density has only increased at Havannah North while most other reefs remained close to densities observed in 2023, except for a decrease at Pandora North (Figure A4). Influential in the regional decline in juvenile densities at 5 m depths since 2020 have been declines in the genus *Turbinaria* (Family: Dendrophylliidae) as strong cohorts that settled on some reefs following cyclone Yasi have died or grown beyond the juvenile size classes (Figure A4).

In 2024, turbidity and the concentrations of NOx and PO_4 quality parameters exceeded guideline values (Figure A14, Moran *et al.* 2025). However, the concentration of NOx continues to decline, and both the short-term and long-term water quality index were classified as ‘good’ (Figure A13a). In waters adjacent to the coral monitoring sites mean wet-season TSS concentrations over the period 2020-2024 exceeded guidelines at Magnetic and Lady Elliot but not at any other location (Figure 25 a, b, Table A 8).

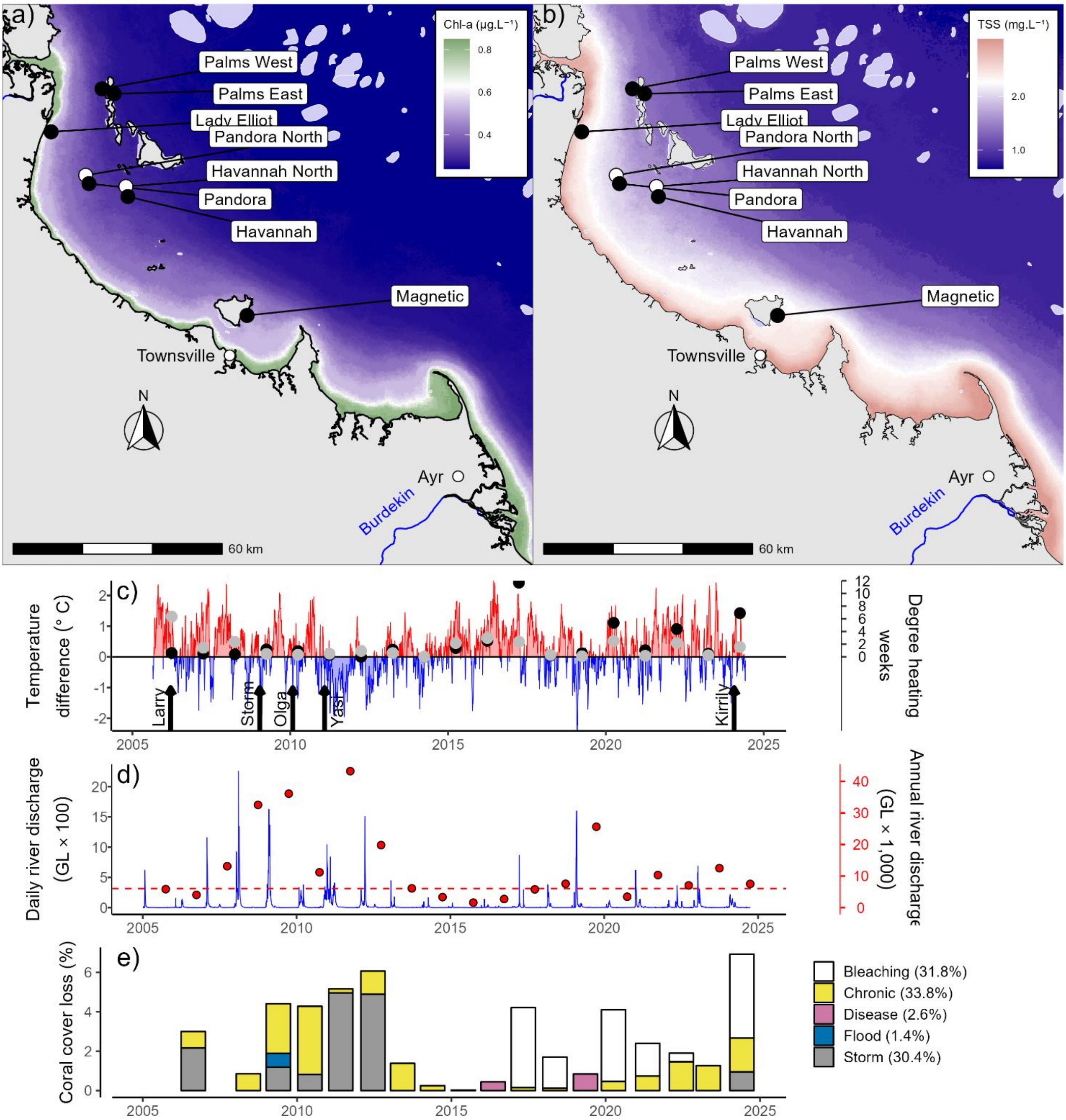

Figure 25 Burdekin region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) median wet season Chl *a* and b) median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2020–2024, white breaks 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.

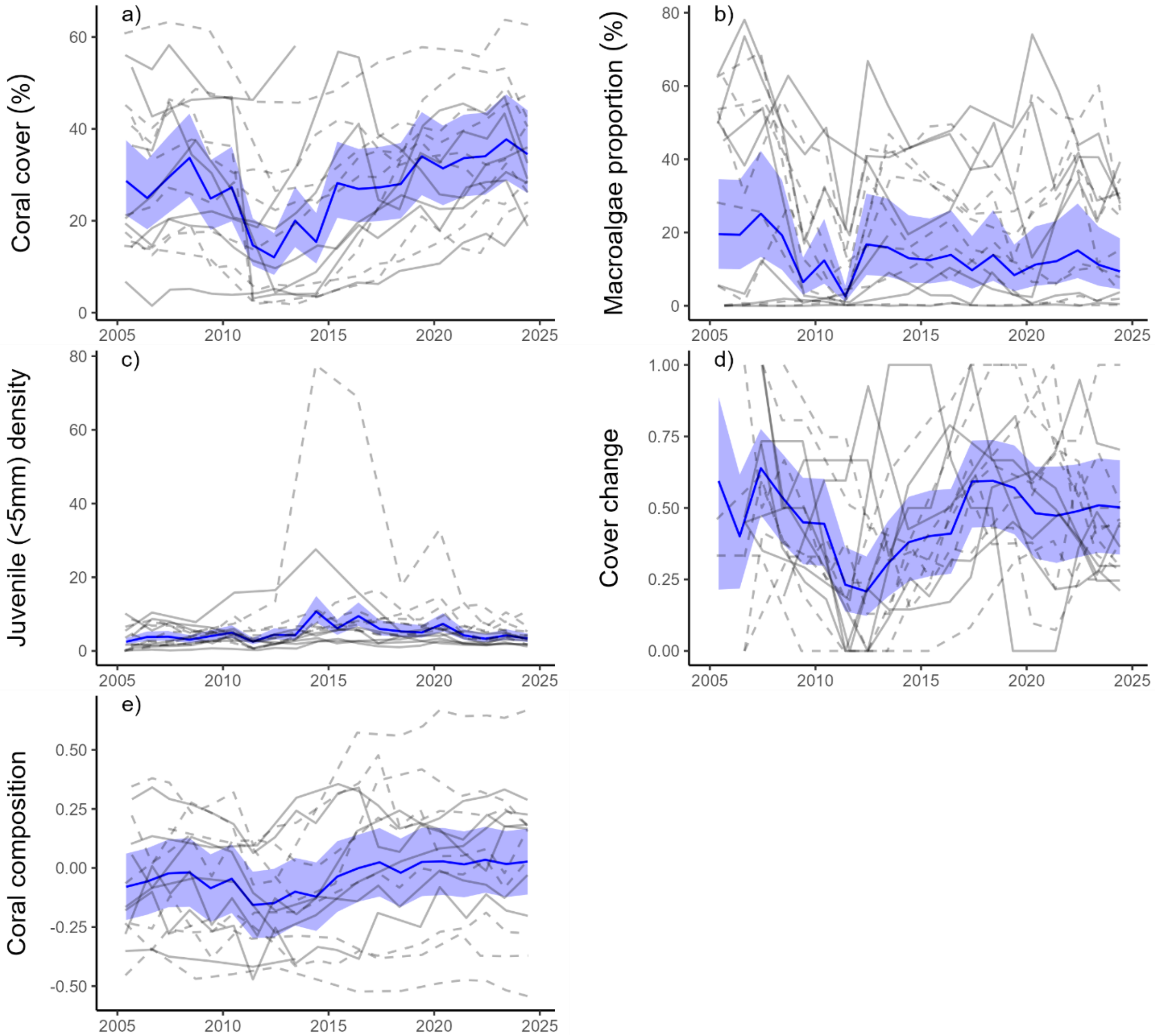

Figure 26 Burdekin region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.5 Mackay–Whitsunday region

In 2024 the Coral Index score remained 'poor' (Figure 27). The Juvenile coral indicator continued to improve but remained within the 'moderate' score range, the only indicator in this range. The Coral cover score has also improved but remains within the 'poor' category along with the other indicators (Figure 27). The improvements in Coral cover and Juvenile coral indicators since 2019 represent gradual recovery following the severe impact of cyclone Debbie in 2017 (Table 16, Figure 28a-e).

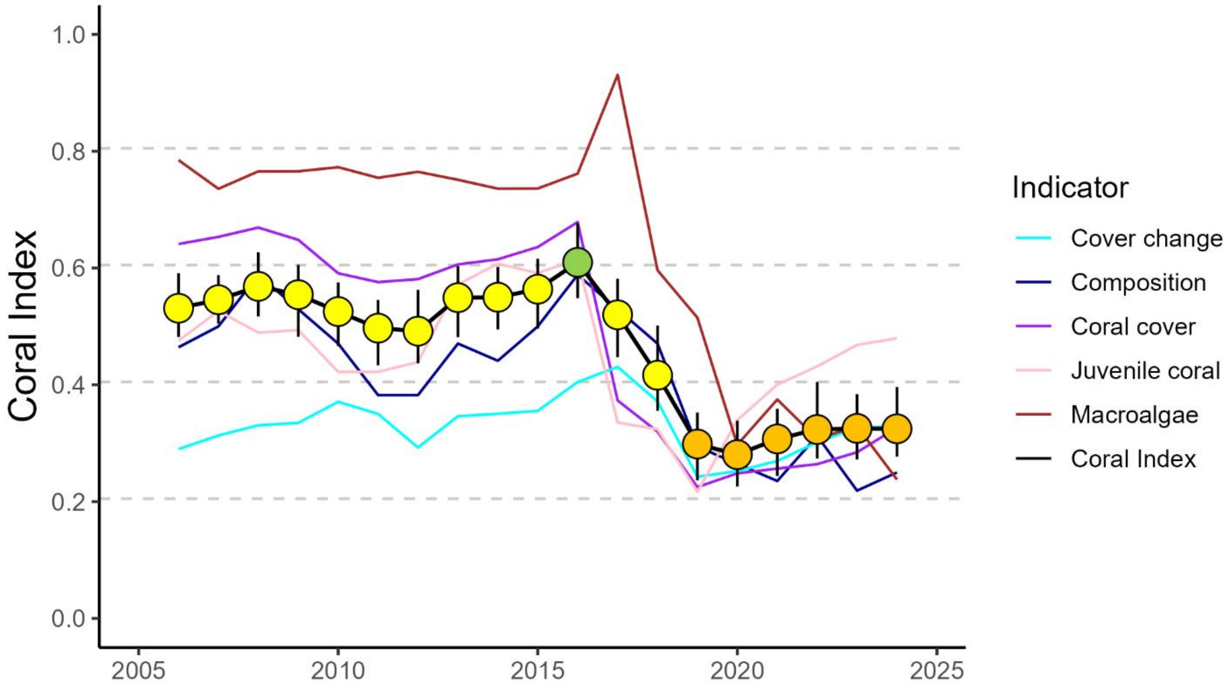

Figure 27 Mackay–Whitsunday region Coral Index and indicator trends. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = 'poor', yellow = 'moderate', green = 'good'. Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at individual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

Table 16 Mackay–Whitsunday region Coral Index and indicator score changes. Data compare the changes in scores between local maxima and minima in the Coral Index time-series. 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 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 2024	2	0.02	0.57	0.08	0.86	-0.05	0.53	0.11	0.88	0.02	0.53	-0.07	0.65
	5	0.07	0.67	0.07	0.81	0.01	0.54	0.16	0.75	0.07	0.58	0.00	0.50

At the coral monitoring locations heat stress estimates derived by NOAA for early 2024 were in 6-10 DHW range, slightly higher than those for 2020 (Figure 9, Figure 28c). However, data from *in situ* temperature loggers suggest the reverse pattern with slightly higher heat stress estimated for 2020

(Figure 28c). During surveys in July there were only a few individual colonies exhibiting signs of bleaching. There were no other acute disturbances in the period 2023-2024. The 2024 water year river discharges were below the long-term median level (Figure 28d). Disease levels remained below the long-term median level and unchanged from 2023 (Figure A8). There were no crown-of-thorns starfish observed and no *Drupella* reported in the period 2023–2024 (Figure A9, Figure A10).

The combined cover of hard and soft corals has maintained a gradual increase since the impacts of cyclone Debbie (Figure 27, Figure 29a, Table 16). However, Coral cover remains in the 'poor' or 'very poor' range at most reefs (Table A7). The highest coral cover continues to be at the 2 m depth at Shute Harbour where the cover of hard corals (predominantly *Acropora*, Figure A5) is 50 %. 'Moderate' scores for Coral cover were maintained at the 5 m depths at Dent and Shute Harbour, and coral cover newly transitioned to 'moderate' at Hook (5 m) and Dent (2 m). Elsewhere, coral cover remained low with scores in the 'poor' or 'very poor range (Table A7).

Scores for Cover change have been in the 'poor' range for most years (Figure 27). The lowest scores for Cover change were observed in 2019 and 2020 and although scores have improved each year this improvement has been inconsistent among reefs (Figure 27, Table 16, Figure 29d,). In 2024 Cover change scores ranged from 'good' at Hayman Island, through 'moderate' at Daydream and Hook (5 m) to 'poor' or 'very poor' at the remaining reefs (Table A7).

The disproportionate loss of Acroporidae corals following cyclone Debbie was reflected by the reductions in the Composition score (Figure 29e, Table 16). In 2024 the Composition score remained 'poor' with no consistent change since 2020 (Figure 29e, 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' although varies markedly among reefs (Figure 26, Table 16, Table A7). In 2024, Macroalgae scores were 'very good' at the more offshore sites at Hayman and Border as well as at Hook (2 m) (Table A7). At the 5 m depths of Shute Harbour and Hook, the Macroalgae scores were 'moderate' and 'poor' respectively (Table A7), in each case the cover of macroalgae was relatively low but above reef-level baselines (Figure A5 ,Table A3). Elsewhere scores were at the minimum value of zero reflecting persistent increase in the cover of macro algae since cyclone Debbie (Table A7, Figure A5).

Juvenile coral scores in 2024 were 'moderate' having steadily improved since a steep decline caused by cyclone Debbie (Figure 27, Figure 29c, Table 16). In 2024, densities of juvenile corals were high at Hayman and Daydream where scores continued in the 'good' to 'very good' range (Figure A5, Table A7). At Hayman, the genus *Acropora* remains strongly represented among juvenile corals, with the family Merulinidae also gaining numbers, raising the total density to 15 m^{-2}, the highest juvenile density among surveyed reefs in 2024 (Figure 29c). Daydream has a more diverse assemblage of juvenile corals with a strong growth in numbers following cyclone Debbie. In 2024 at 5 m depth there was a noticeable decline in the juvenile density of Dendrophylliidae and *Acropora* (Figure A5). Small gains across a range of taxa were enough to transition Pine (5 m) to the 'moderate' category (Table A7). Similarly, a range of minor losses transitioned Shute Harbour (2 m) to the 'poor' category.

In 2024, the concentrations of most water quality parameters exceeded guideline values with only Chl *a* meeting the guidelines, while NOx and TSS were meeting only at some sites (Moran *et al.* 2025). However, concentrations of dissolved inorganic forms of nitrogen (NOx) marginally improved over 2015 to 2024 (Moran *et al*. 2025). While the short-term water quality index remained 'moderate' it has trended up since a low point in 2017 (Figure A14a). From 2019–2024, 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).

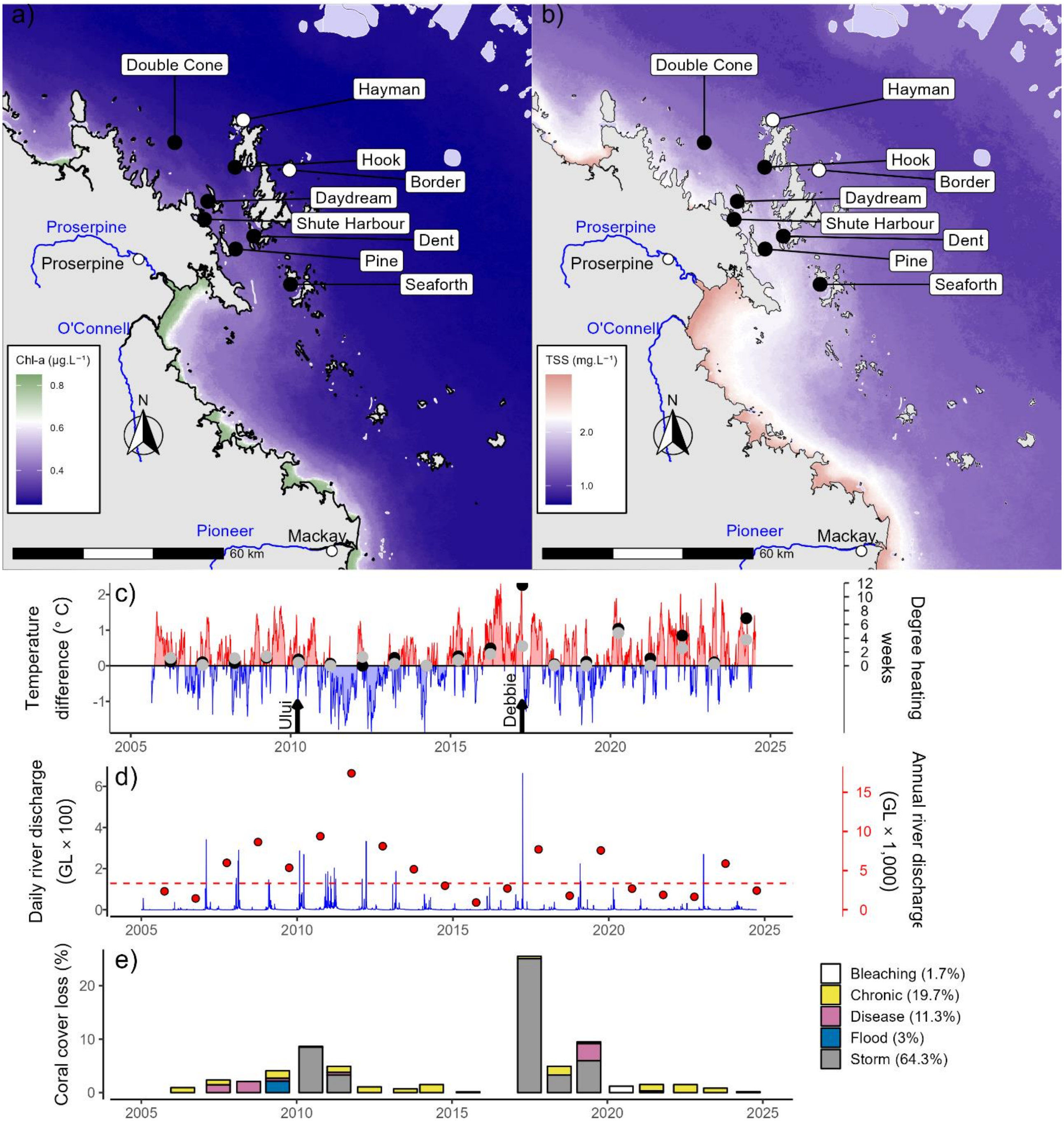

Figure 28 Mackay-Whitsunday region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) median wet season Chl *a* and b) median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2020–2024, white 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.

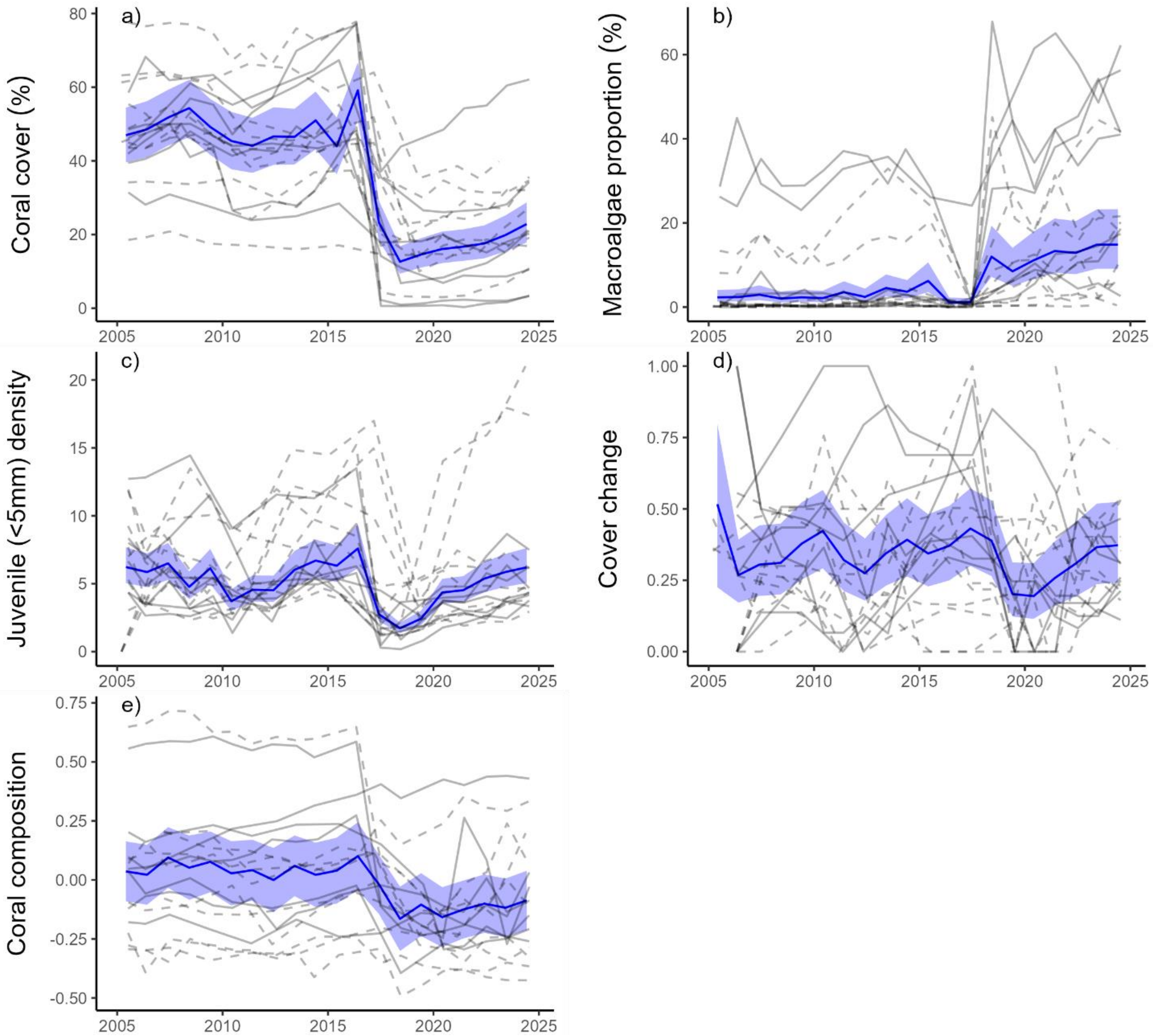

Figure 29 Mackay–Whitsunday region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.6 Fitzroy region

The Coral Index score for the Fitzroy region declined from the 2023 level but remained in the 'poor' category (Figure 30, Table 17). Coral indicator scores declined at both depths: Coral cover declined from 'moderate' to 'poor', Juvenile coral and Composition from 'poor' to 'very poor', and Cover change declined but remained in the 'poor' category (Figure 30, Table 17). The Macroalgae indicator score remains in the 'very poor' category.

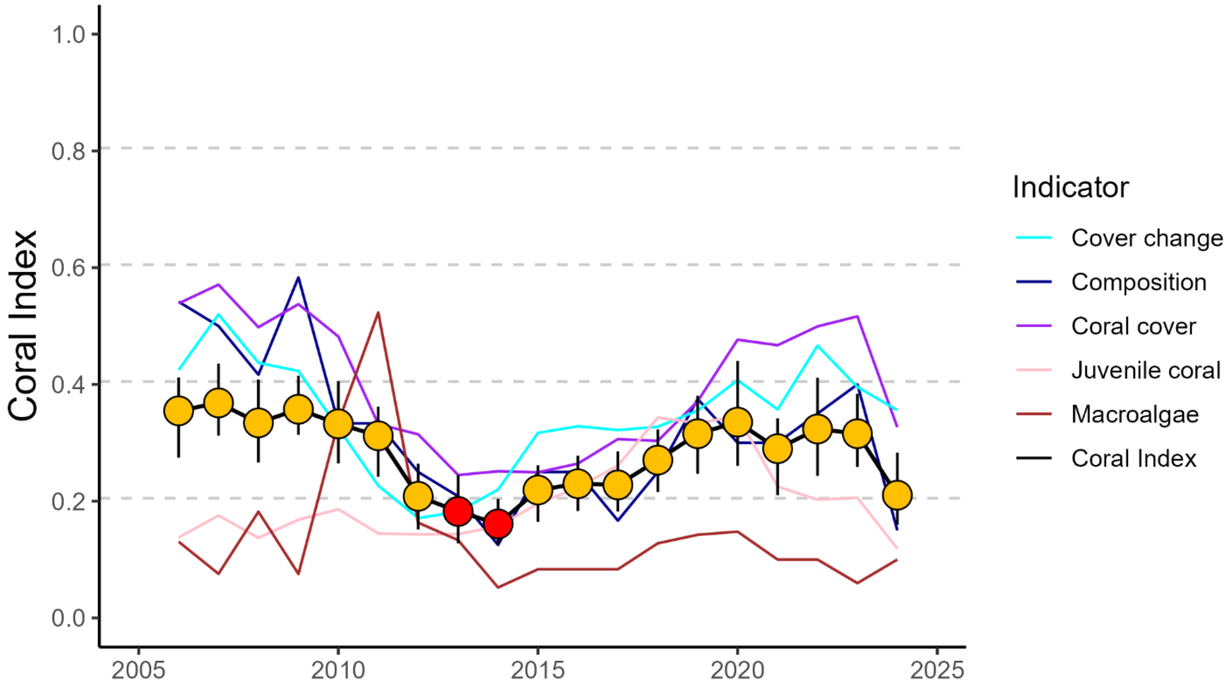

Figure 30 Fitzroy region Coral Index and indicator trends. Coral Index scores are coloured by Reef Water Quality Report Card categories: red =' very poor', orange = 'poor'. Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at individual reefs.

Table 17 Fitzroy region Coral Index and indicator score changes. Data compare the changes in scores between local maxima and minima in the Coral Index time-series. 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 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 2024	2	-0.09	0.93	-0.28	0.94	0.08	0.69	-0.14	0.83	-0.02	0.68	-0.10	0.68
	5	-0.12	0.90	-0.10	0.92	0.00	NA	-0.04	0.87	-0.06	0.77	-0.40	0.83

Reefs in the Fitzroy region were exposed to extreme levels of heat stress and coral bleaching during the heatwave of January 2024 (Figure 9, Figure 32e). Hard coral cover across the region declined from 37% to 23% making the 2024 bleaching the most severe acute disturbance event over the period of the MMP (Figure 32e). Previous events that caused substantial loss of coral cover across the region were bleaching in 2006 and a major flood of the Fitzroy River in 2011 (Figure 32e, Table A6). A series of lesser impacts associated with cyclones and storms between 2008 and 2015 and coral bleaching in 2020 also resulted in reductions in coral cover (Table 17, Figure 33a).

The impact of the 2024 bleaching event reduced Coral cover scores from 'Moderate' in 2023 to 'Poor' (Figure 30, Table 17). Highest losses were recorded amongst dense *Acropora* communities at the shallow 2 m sites with the cover of hard corals more than halved at: Barren (from 60% to 24%), North Keppel (from 48% to 19%), and Middle (from 20% to 10%) and almost halved at Keppels South (53% to 28%) (Figure A6). At 5 m depths at these reefs hard coral cover also declined although losses were proportionally lower (Figure A6, Table A6). At Pelican losses were minor (Figure A6).

Observations during our surveys in May 2024, four months after the peak of the marine heat wave, noted a high proportion of corals were still bleached, particularly the Acroporidae (*Acropora, Montipora*) (Figure 31). This level of ongoing bleaching indicates ongoing stress that will likely lead to further decline in coral cover as occurred in 2021 following observation of a high level of bleached corals during surveys in 2020 (Figure 32e, Figure A7).

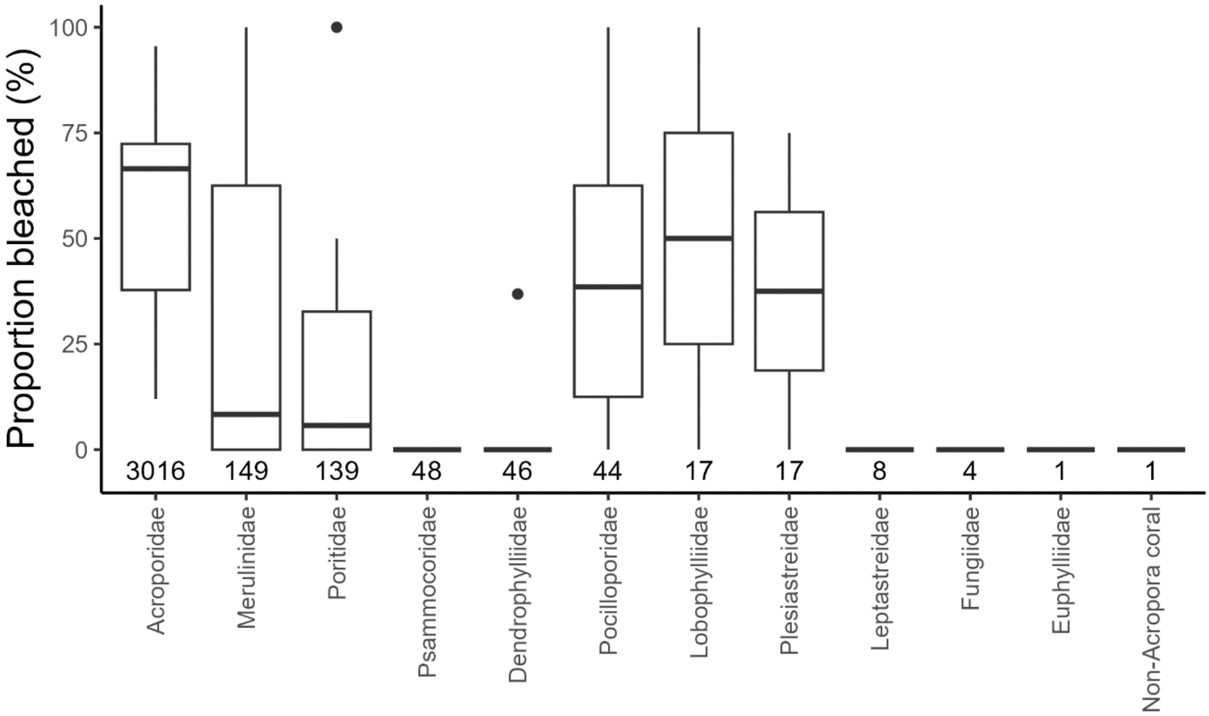

Figure 31 Distribution of bleaching among hard coral families in the Fitzroy region in May 2024. For each family the proportion of total points on photo-intercept transects classified as fully bleached (white) or partially bleached (obviously pale) at each reef and depth combination within the region. Total points are listed against each family.

Soft coral cover in the region is typically low and variable (Figure A6). Changes between 2023 and 2024 have been minor, the most notable being a decline in *Cladiella* at Barren (2 m) (Table A10), likely due to the marine heatwave. At most reefs in the Fitzroy region the contribution of soft coral to the combined coral cover metric is minor. However, at Pelican (5 m) 30% of the coral cover is comprised of soft corals, the majority of which are *Sclerophytum* (Table A10).

In 2024 Cover change score declined further within the 'poor' category (0.36, Table 17), with declines in Cover change scores for individual reefs at Barren (5 m), and both depths at Pelican (Table A7).

The regional Composition indicator category declined from 'poor' to 'very poor' (0.15, Figure 30, Table A7). This result largely reflected the reduced cover of *Acropora* particularly at Barren (2 m, 5 m) and Keppels South (5 m) following the 2024 summer heatwave (Figure A16).

The macroalgae cover at Barren (2 m) declined lifting the Macroalgae indicator score to the maximum of 1 (Figure A6, Table A7). At all other sites the threshold for the proportion of macroalgae continued to be exceeded, maintaining a score of zero (Table A3). Common macroalgae taxa include *Lobophora* that had increased at North Keppel (at 2 m to 37%) and decreased across all other sites (Table A11). Sargassaceae, common at Middle and Pelican, has decreased at 2 m depths at both reefs.

The regional Juvenile coral indicator transitioned from the category ‘poor’ in 2023 to ‘very poor’ (0.12, Table A7) in 2024, continuing a decline from the bleaching event of 2020 (Figure 29). Individual Juvenile coral categories ranged from ‘poor’ to ‘very poor’ (Table A7). Juvenile density declined at all reefs to reach mean levels not seen since 2014 (Figure 33c, Figure 30). The largest decline occurred at Barren (2 m), principally *Leptastrea,* a major contributor to the juvenile coral population since 2011 (Figure A6). It should be noted that the increase in filamentous turf algae that replaces live coral following a bleaching event contributes to the decline in Juvenile density scores. This is particularly evident at the 2 m depths at Barren and Keppels South where widespread loss of *Acropora* contributes to low Juvenile density scores (Table A7, Figure A6).

In situ water quality monitoring was reinstated in 2021 after being discontinued in 2015, due to budget constraint. In 2015, the long-term water quality index was assessed as improving and scored as ‘good’ (Moran *et al*. 2025). Conditions from 2021 to 2024 continued to be categorised as ‘good’ (Figure A15). In 2024, the short-term water quality index score (Moran *et al*. 2025) was also ‘good‘, with most water quality parameters being at or below guideline values (Figure A15). Parameters that exceeded guideline values include dissolved inorganic forms of nitrogen (NOx), particulate phosphorous (PP), and turbidity (Figure A15). Over the period 2019–2024 wet-season Chl *a* concentrations were below guideline values at the coral monitoring sites, and only at Pelican was the TSS guideline value exceeded (Figure 32a,b, Table A8).

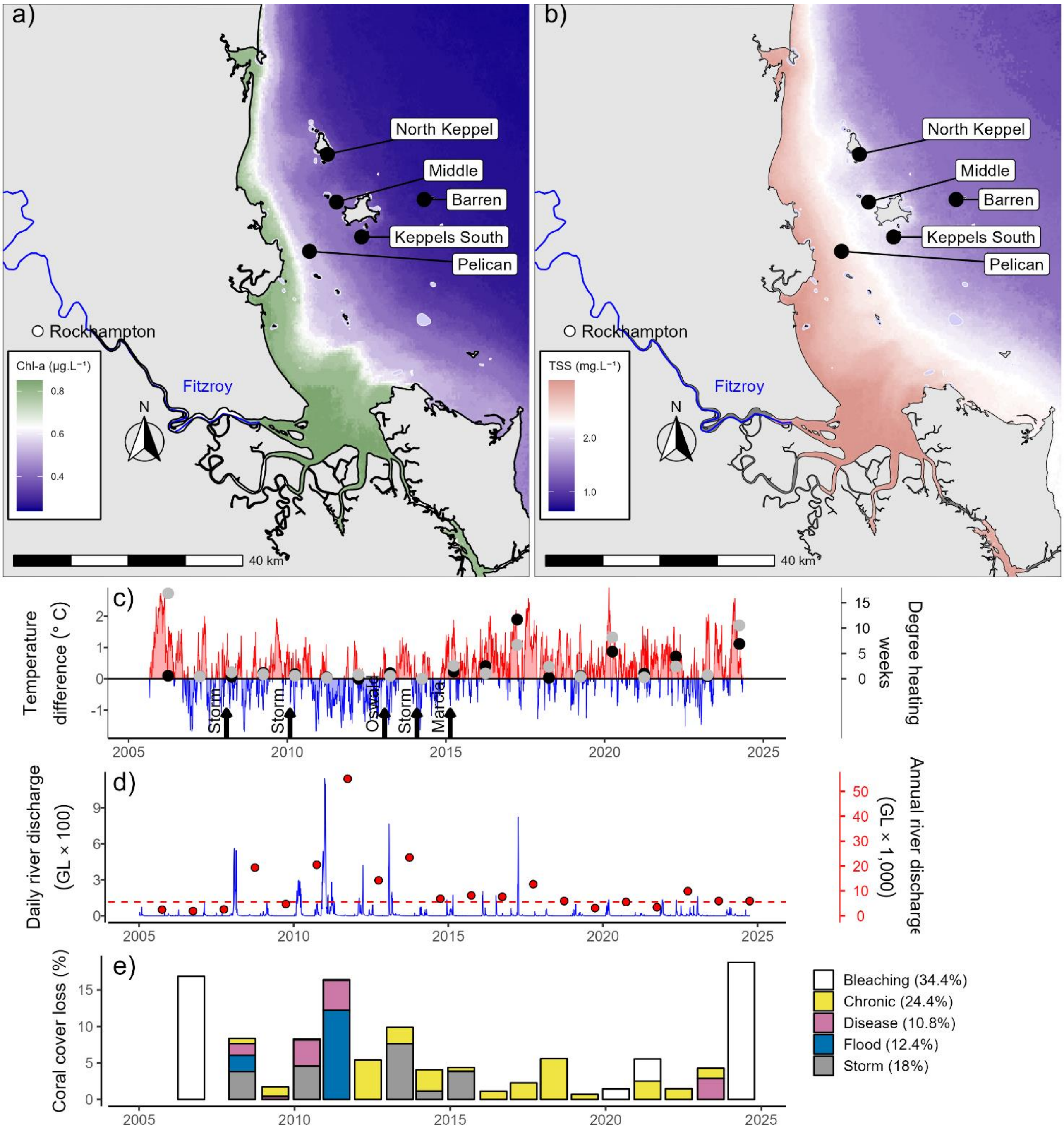

Figure 32 Fitzroy region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) median wet season Chl *a* and b) median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2020–2024, white breaks 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.

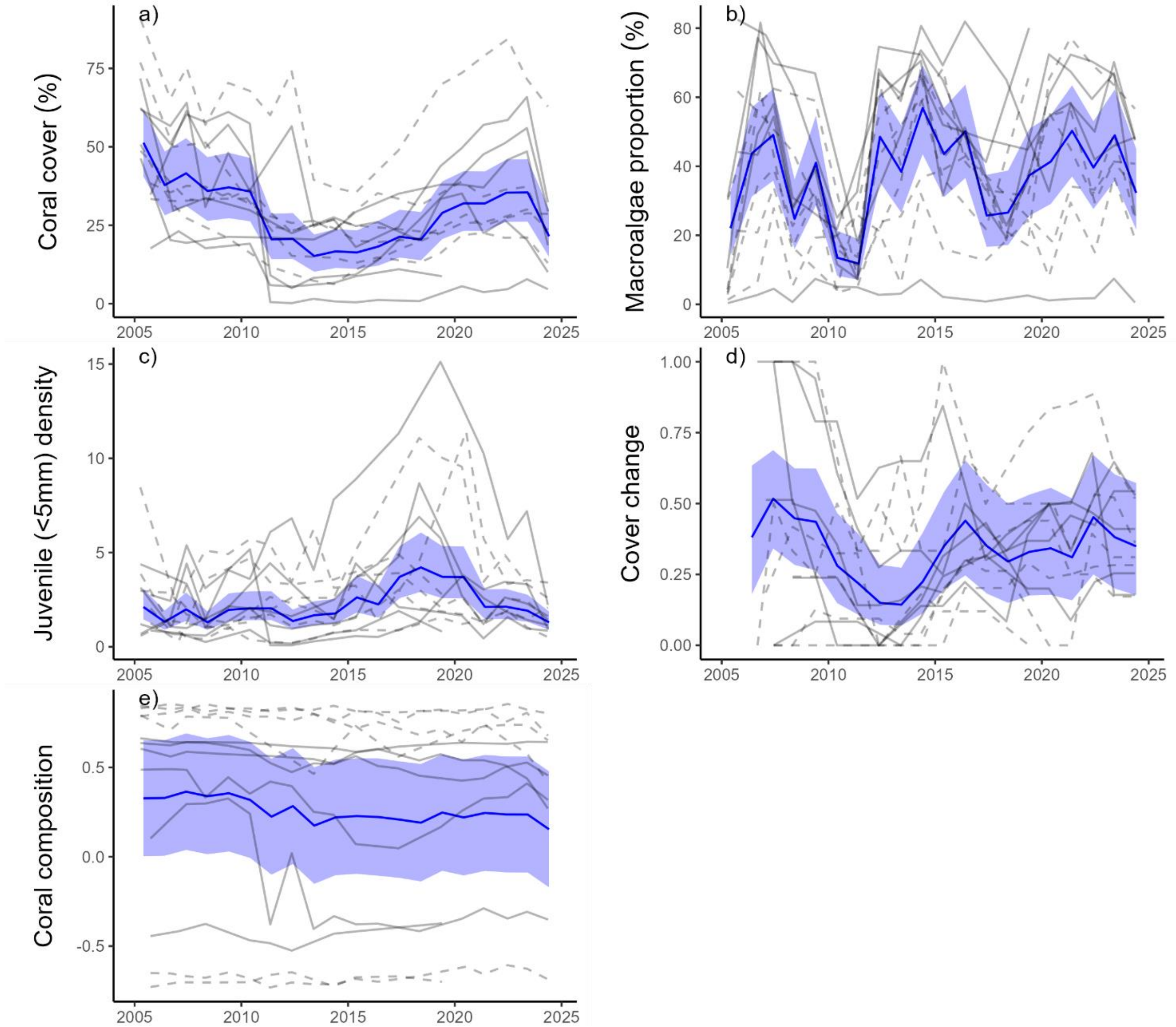

Figure 33 Fitzroy region indicator trends. a– e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.7 Response of coral communities to environmental conditions

4.7.1 Location along water quality gradients

Of the Coral Index and individual indicator scores only the Cover change scores at 5 m depth showed any relationship with gradients of Chlorophyll *a* or the light attenuation coefficient k490, as estimated from satellite data (Table 18, Figure 34, Table A8). These relationships were only evident at the 5 m depth of reefs in the Burdekin region and clearly influenced by the score of 1 at Magnetic (Figure 34, Table A7).

The standardised 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). These water quality vector scores were estimated based on the relationships between genus cover and water quality variables for Turbidity and Chl *a* (see section 2.4.5) as observed during the first 5 years of monitoring at each site. In 2024 community composition values at 5 m depth still related to water quality gradients in all regions other than Mackay Whitsunday (Table 18, Figure 35).

Table 18 Indicator score and value relationships with satellite derived and water quality. Tabulated values are upper and lower confidence intervals of the change in values of each response variable over the range of the water quality variables for each combination of indicator score or value and depth (see section 2.5.1). Slopes for which confidence intervals did not include zero are shaded to highlight the direction of the relationship. Results are presented for each combination of response and environmental variable for which there was statistical support, judged as AICc values at least 2 points lower than the equivalent null model.

Response	Depth	Reef-wide		Wet Tropics		Burdekin		Mackay-Whitsunday		Fitzroy	
		l	u	l	u	l	u	l	u	l	u
Chlorophyll *a* concentration (colour class)											
Coral change score	5	-0.16	0.41	-0.31	0.25	0.38	0.84	-0.45	0.25	-0.55	0.42
Community composition	5	-1.29	-0.62	-1.01	-0.32	-1.2	-0.36	-0.58	0.36	-2.14	-0.95
Light attenuation coefficient kd490											
Coral change score	5	-0.26	0.35	-0.38	0.20	0.41	0.87	-0.53	0.18	-0.56	0.39
Community composition	5	-1.36	-0.66	-0.99	-0.30	-1.28	-0.44	-0.66	0.25	-2.16	-1.04

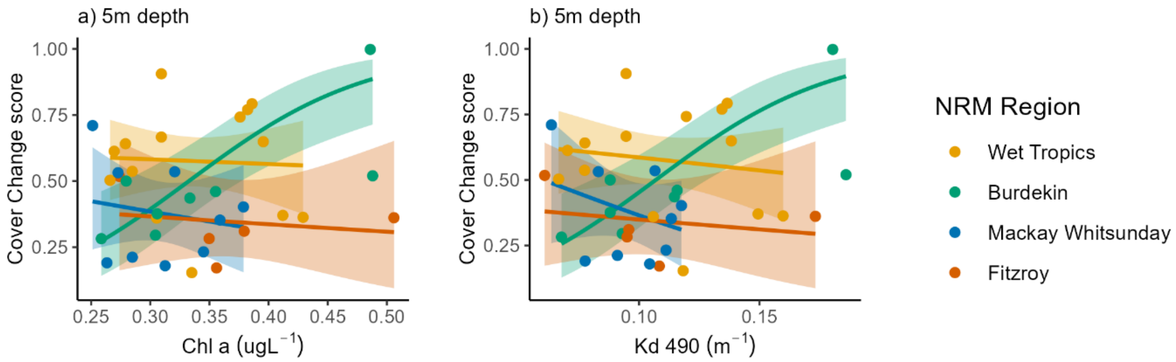

Figure 34 Cover change score relationships.Chl a and light attenuation coefficient (Kd 490)

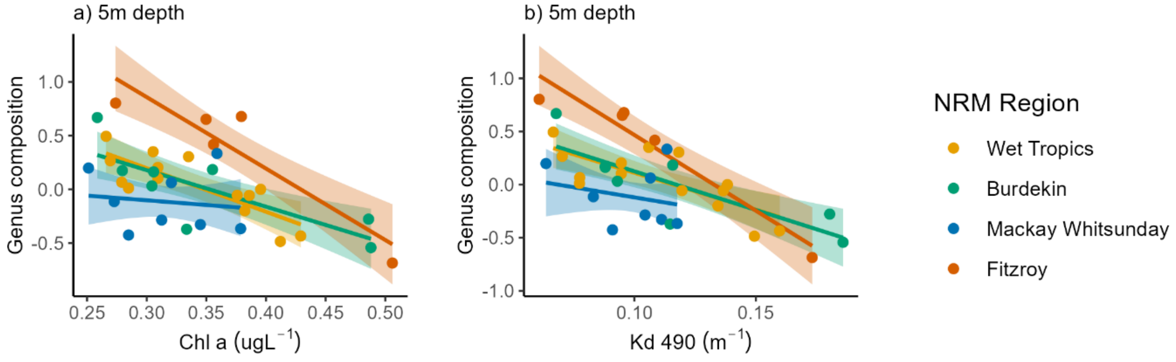

Figure 35 Hard coral community composition relationships to Chl a and light attenuation (Kd 490). Genus composition values are data underlying scores for the Composition indicator.

Relationships between coral reef community attributes at the subset of reefs at which water quality was physically monitored by the MMP were more apparent (Table 19). At 5 m depth Coral cover scores declined with increasing Chl *a* concentration (Figure 36).

Table 19 Relationships between coral reef communities and measured water quality. 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 2020-June2024, variables tested were Chlorophyll a, Total Suspended 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 represents the direction of the change in response with increasing concentrations or values of the water quality summaries.

Response	Depth	Environmental variable	Change across reefs		
			mean	Credible intervals	
				lowest	Highest
Coral cover	5	Chlorophyll a	-0.47	-0.67	-0.27
Macroalgae proportion	2	Chlorophyll a	0.43	0.22	0.65
		Total suspended solids	0.44	0.19	0.69
Macroalgae cover	2	Chlorophyll a	0.36	0.19	0.52
		Total suspended solids	0.38	0.19	0.58
Community composition	2	Total suspended solids	-0.57	-0.86	-0.28
		Chlorophyll a	-0.94	-1.49	-0.40
		Total suspended solids	-1.1	-1.53	-0.7

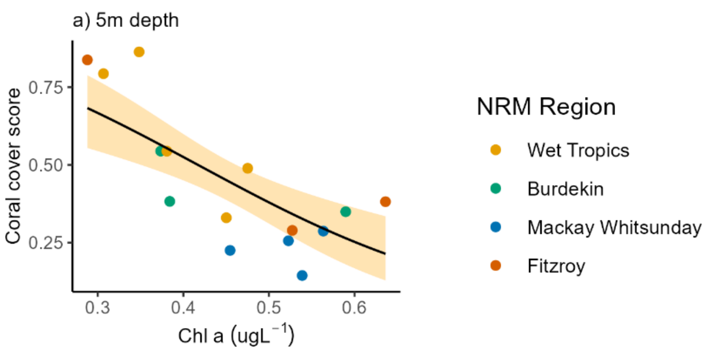

Figure 36 Coral cover indicator score relationships to water quality. Trends represent predicted relationships with 95% credible intervales derived from GLM models. Points represent the observed data.

Relationships between macroalgae and water quality were limited to 2 m depths where both the cover of macroalgae and the proportion of the algal communities comprised of macroalgae increased with increasing Chl *a* and suspended sediment concentrations (Figure 37, Figure 38). Macroalgae cover also declined with increasing concentration of N relative to P (Figure 37).

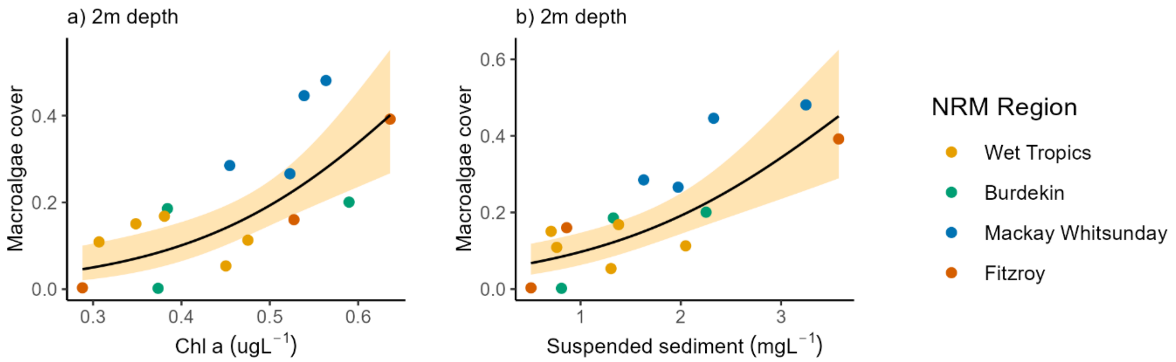

Figure 37 Macroalgae cover relationships to water quality at 2 m depth. Trends represent present predicted relationships with 95% credible intervales derived from GLM models. Points represent the observed data.

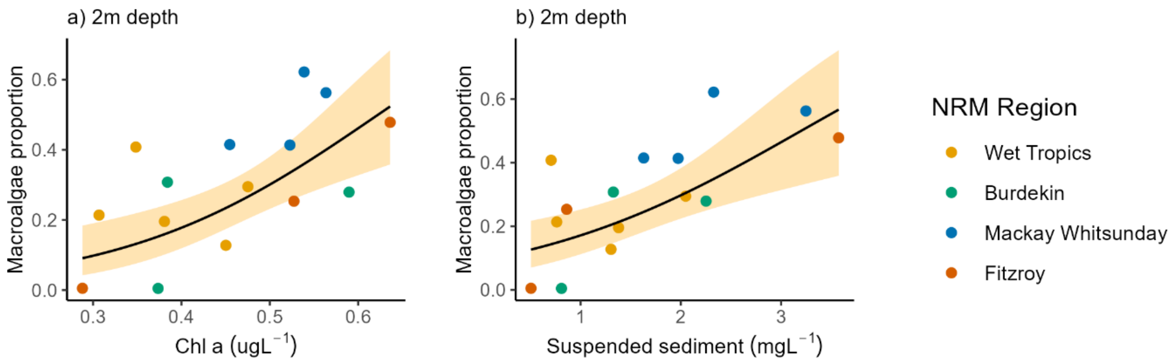

Figure 38 Macroalgae proportion to water quality relationships. Trends represent present predicted relationships with 95% credible intervales derived from GLM models. Points represent the observed data. Macroalgae proportion is the cover of macroalgae divided by the cover of all algae.

As observed in relation to the satellite derived water quality measures, the composition of coral communities varied along gradients (Table 19). At both 2 m and 5 m depths genus composition values varied along suspended sediment and Total N to P ratios (Figure 38, Figure 39). At 5 m depth genus composition also varied along a Chl a concentration gradient (Figure 39).

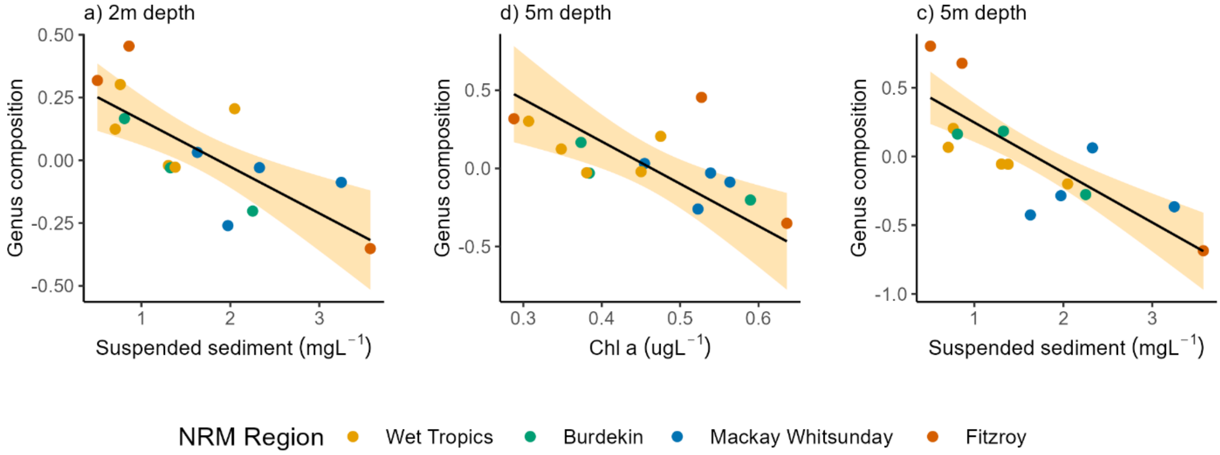

Figure 39 Relationship between coral community composition values and water quality. Trends represent present predicted relationships with 95% credible intervales derived from GLM models. Points represent the observed data. Genus composition values are data underpinning estimates of the Composition score.

4.7.2 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 37). 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, or lagged responses of indicators, will not be included. In the case of lagged influences, such as the initial decrease then post-disturbance increases in macroalgal cover that has been observed on several occasions following cyclones and floods, this will result in the underestimation of the response.

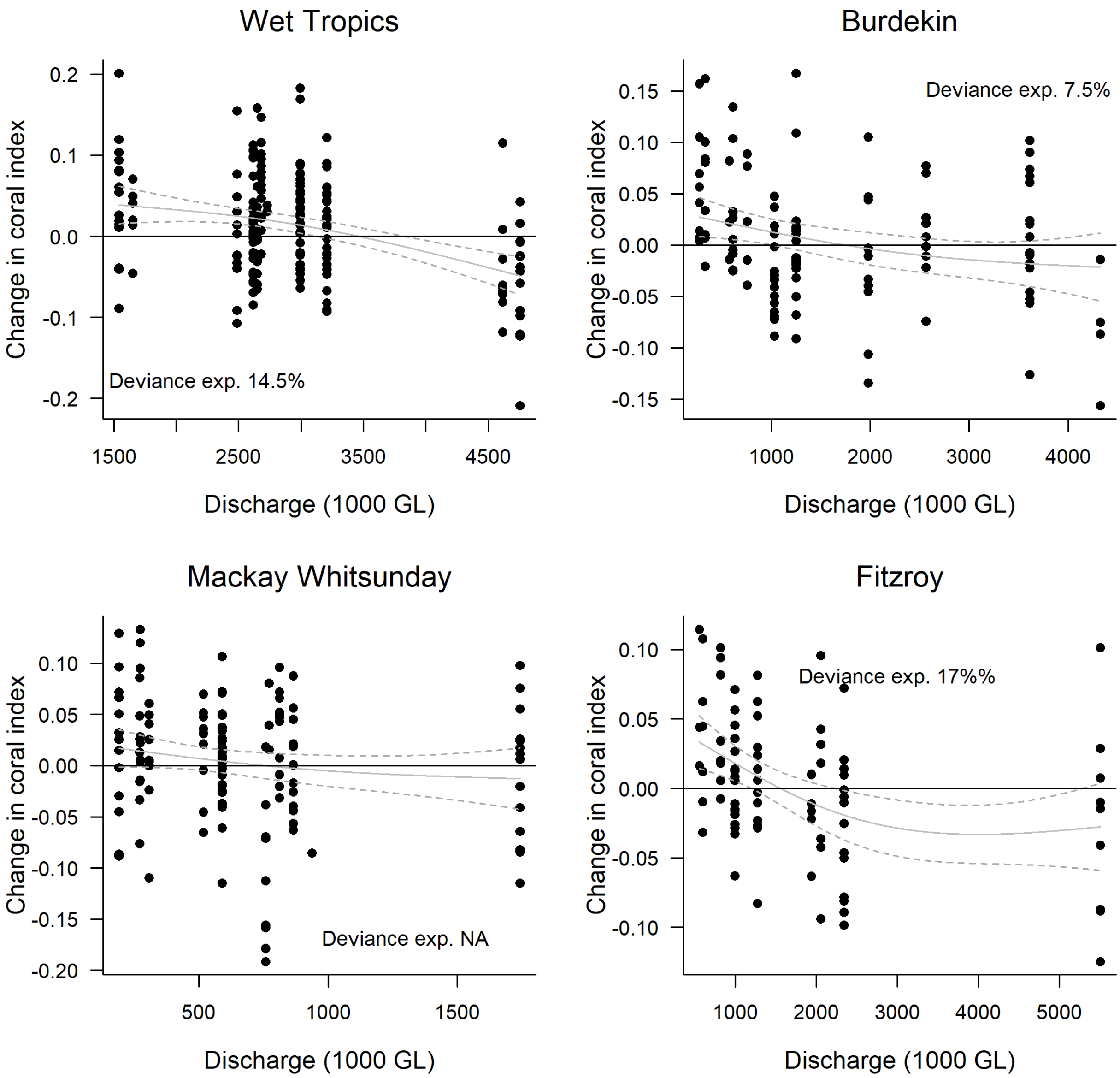

Figure 40 Relationship between the Coral Index and freshwater discharge from local catchments. Plotted points represent observed change in the Coral Index score at each reef and depth over a two-year period. Observations following years for which acute disturbances impacted communities in the period between samples were excluded. Discharge values represent the maximum annual discharge from the region’s major rivers over the two-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 unbalance 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 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. This *state* can then be interpreted in terms of *impact* on desirable ecosystem functioning or services that can be used to inform when and where management action (*response)* is warranted.

It is apparent that the combination of escalating impacts of coral bleaching and slow rates of coral community recovery, especially where environmental conditions promote the proliferation of macroalgae, have resulted in a decline in the condition of inshore reefs.

5.1 Pressures

5.1.1 Acute disturbances

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 40% 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. Between 2017 and 2023, no impact from severe cyclones occurred, signifying a period during which coral recovery should occur. This hiatus in cyclone activity was broken over the 2023-24 summer with two cyclones impacting reefs monitored by the MMP.

Cyclone Kirrily crossed the coast north of Townsville as a category 1 system on the 25th of January. The cyclone tracked to the south of most reefs in the Burdekin region and this, along with its low intensity and rapid movement, ensured minimal impacts occurred. At most, minor storm damage comprised of relatively few broken or overturned corals was observed.

Cyclone Jasper crossed through the Reef on the 13th of December 2023 as a category 2 system before making landfall at Wujal Wujal north of Cape Tribulation. Damage attributed to this storm was observed at Snapper Island with storm damage at 2 m depths also observed at Fitzroy Island West, and High Island East.

In addition to the physical damage caused by waves, Cyclone Jasper precipitated a period of intense rainfall, causing record-breaking flooding in Wet Tropics and southern Cape York rivers (Moran *et al.* 2025). The flood of the Daintree River killed all corals at the Snapper South monitoring sites, marking the most severe impact of any acute event observed during the 20 years of the MMP. Previous losses of coral cover due to exposure to low salinity flood waters have been limited to 2 m depths on reefs: 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.

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. It is likely that further loss of coral will be revealed by surveys in 2025 as some of these bleached corals are likely to die (see for example Byrne *et al.* 2025), as was observed in 2018 and 2021 following the 2017 and 2020 bleaching events under lesser levels of heat stress.

This most recent bleaching event builds on previous events in 2006, 2017, 2020 and, to a lesser degree, 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 a 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.

Notable from the 2020 and 2024 bleaching events was that the proportion of coral lost due to bleaching was greater at the 2 m depth than at the adjacent 5 m depth sites. This observation is consistent with previous reports of reduced severity of bleaching with depth (e.g., Muir *et al*. 2017, Cantin *et al.* 2021) and consistent with the conclusions of Lesser (2024) that oxidative stress increases with increasing irradiance. In turbid water, reduced light intensity with increased depth and/or self-shading due to increased symbiont loads have been previously identified as mechanisms that provide some resistance to bleaching at deeper depths (Anthony *et al*. 2007). 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 and 2020 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).

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 2022/23. In 2024, outbreak densities of crown-of-thorns were observed at Fitzroy Island, The Frankland Group and High Island (on the eastern sites only). In recent years, the Crown-of-thorns Starfish Control Program has helped to mitigate the impact of crown-of-thorns starfish[2] with 17,652 individuals removed from Fitzroy Island and the Frankland Group since 2013, 1088 of these from the Frankland Group in the year preceding the 2024 MMP surveys. Consistent across the cull data and MMP

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

observations have been records of relatively high proportions of juveniles 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 will have 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 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 a natural part of the Reef ecosystem, albeit the contribution 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 Moran *et al*. 2025) is the quantification of the compounding influence of run-off on the naturally occurring gradients, and of any subsequent improvement due to the activities under the Reef 2050 WQIP.

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 Moran *et al.* 2025) 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. 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 and resuspension by wind, currents and tides (Larcombe *et al.* 1995, Bainbridge *et al.* 2018). The trends emerging from the MMP support other studies showing that the additional flux of fine sediment imported by rivers remains 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, Thompson *et al.* 2020). Any increase in turbidity associated with run-off will reduce the level of photosynthetically active radiation reaching the benthos; 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 changes in Coral Index scores and discharge from local rivers are consistent with well documented links between increased run-off and stress to corals (Bruno *et al.* 2003, Kuntz *et al*. 2005, Kline *et al*. 2006, Voss & Richardson 2006, Kaczmarsky & Richardson 2010, Haapkylä *et al.* 2011, 2013, 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 clear relationship between discharge and change in the Coral Index scores in the Mackay–Whitsunday region is likely due to the relatively low discharge and strong currents in this region. Modelling by Baird *et al.* (2019) suggest that “fine catchment-derived sediment that remains suspended near the seabed forms a benthic (or fluffy) layer in the Whitsundays / GBR lagoon that persists for a number of 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. Across the region, strong vertical differentiation in community composition at many Mackay–Whitsunday reefs, where there is a high representation of species tolerant to high turbidity at the 5 m depths, reflects the long-term selective pressure imposed by high turbidity and this may limit sensitivity to any pressures imposed by variable 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 2012, during which high rainfall delivered relatively high loads of sediment and nutrients to the Reef. Water quality has now stabilised or improved in recent years (Moran *et al.* 2025). A feature of the decline following the wet period was a general increase in oxidised forms of dissolved nitrogen (NOx) 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 2024, concentrations for both these water quality parameters remain high, although NOx concentrations appear to be declining 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 recently suggested 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 NOx 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 NOx concentrations observed at MMP sites are low in comparison to P concentrations, suggesting increased NOx concentrations have the potential to increase the growth of *Sargassum* or possibly extend its range along the water quality gradient.

The overall *state* of inshore water quality across the regions monitored in this report is in moderate to good condition (Moran *et al*. 2025). While Chlorophyll *a* and TSS concentrations remained within guideline values across the board, it's worth noting that discharge levels exceeded the long-term median in most regions, with the exception of Mackay-Whitsundays (Moran *et al*. 2025). The Wet Tropics region, in particular, stood out with discharge levels 2.1 times higher than the long-term median—this being the highest since 2011. The spike can be attributed to a tropical rain depression that lingered in the area after cyclone Jasper crossed the region in December 2023. Interestingly, despite a spike in modelled TSS loads during the 23/24 summer, this didn't appear to affect the annual condition index in the same way (Moran *et al*. 2025). However, flood waters from the Barron River were pushed south and intersected with those from the Russell Mulgrave, exposing reefs in the Johnstone Russell–Mulgrave subregion to flood plume conditions (AIMS eReefs Visualisation Portal - gbr4-hydro_temp-wind-salt-current). On the other hand, while Secchi depth and Nitrite + Nitrate (NOx) concentrations exceeded guideline values across all regions, it's worth noting that the NOx levels have shown an overall improving trend in most regions, which could indicate some positive shifts in water quality over time (Moran *et al*. 2025).

In 2024 the losses of coral cover attributed to disease and chronic pressures account for 22.9% 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 magnifying the effects of heat stress events (Wiedenmann *et al.* 2013, Fisher *et al.* 2019, Cantin *et al.* 2021, Brunner *et al.* 2021). A case in point was Fitzroy Island West where the loss of coral cover in 2024 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 flood waters (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).

5.2 Ecosystem state

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

In 2024, the Reef-level Coral Index score declined to the lowest value yet observed. 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 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 2023/24 summer the overall Coral Index had failed to recover since declining into the ‘poor’ range 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 demonstrate indicate reduced recovery potential of coral communities in recent years.

The cycle of disturbance and recovery and resulting coral community condition in 2024 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 declined sharply in 2024 but remained in the ‘moderate’ range. Declines occurred for each of the indicators with scores for Coral cover and Composition declining from ‘good’ to ‘moderate’, Cover change and Macroalgae remaining ‘moderate’ and Juvenile coral dipping into the ‘poor’ range for the first time since 2012. Declines occurred in each of the three sub-regions although the primary causes of these declines varied as outlined 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 Barren–Daintree sub-region

In 2024 the Barron–Daintree sub-region score declined but remained ‘moderate’. This score masks the severe impact to coral communities at Snapper Island where all coral was killed at Snapper South and coral cover greatly reduced at Snapper North 2 m. These losses were caused by the combined impacts of storm damage and subsequent flood waters associated with cyclone Jasper and represent the single most severe disturbance event recorded by the MMP. Any impact from this event at Low Isles is yet to be assessed as the surveys informing the 2024 score were undertaking before the passage of the cyclone. Buoying the Coral Index score were ‘good’ scores for the Cover change and Macroalgae indicators.

Cover change scores are averaged over a four-year window but exclude observations when hard coral cover was impacted by an acute disturbance. Hence the ‘good’ Cover change score reflects the rate coral cover was increasing over the last few years, and while this suggests coral growth was not being substantially impacted by ambient conditions it does not account for the lack of capacity for growth where all corals have been killed. In this instance, the recent observations of cover change will clearly be overestimating the potential for coral cover to increase in the short term. However, this over estimation will be due to the lack of corals rather than the pressure imposed by the ambient environmental conditions, that is the primary focus of this indicator.

The steep improvement in the Macroalgae score resulted as the impacts from the cyclone reduced both the cover of macroalgae and corals with the bare space created initially colonised by algal turfs. It is very likely this improvement in the Macroalgae score will be short-lived as macroalgae have rapidly recolonised after similar losses at other reefs monitored by the MMP.

Of concern for the rapid recovery of coral communities is that the Juvenile coral score has remained in the lower end of the ‘poor’ range for most of the time since 2011. The few instances where scores improved coincide with two strong cohorts of *Acropora* recruits at Snapper South, where the typically low densities of juvenile corals recorded suggest limited larval supply, a situation likely to be exacerbated by local loss of coralsHigh levels of macroalgae at Snapper North are also likely to continue to suppress coral recruitment.

5.2.2.2 Johnstone Russell–Mulgrave sub-region

The decline in the Coral Index in 2024 reflects reduced scores for each indicator. In 2024 the Coral Cover score remained 'good' but had declined at most reefs. Declines were caused by the ongoing presence of crown-of-thorns especially at Frankland East, and the combined impacts of high sea water temperatures causing coral bleaching, storm damage and possibly exposure to low salinity flood waters. The relative contribution of these pressures was difficult to determine and varied among reefs.

Coral bleaching was observed at Fitzroy West and Franklands West during targeted bleaching surveys in February 2024 (Cantin *et al.* 2024). However, temperature stress, as estimated by both NOAA (DHW) and from our *in situ* loggers was relatively low. It is possible that exposure to nutrient enriched flood-plume waters (Moran *et al.* 2025) may have contributed to the susceptibility of corals to thermal stress as poor water quality has been shown to magnify the effects of heat stress (Wiedenmann *et al.* 2013, Fisher *et al.* 2019, Cantin *et al.* 2021, Brunner *et al.* 2021). Physical damage was also observed at Fitzroy West and High East demonstrating the additional exposure to damaging waves at these sites. At High East there was almost complete mortality among *Acropora* and *Montipora* colonies at 2 m depth. Here, while moderate physical damage was evident, the highly stratified nature of this mortality suggested exposure to low salinity flood waters was also a key contributor to observed mortality. At Franklands East, Franklands West and High East, ongoing presence of crown-of-thorns starfish at outbreak densities will have also contributed to losses in coral cover.

Prior to the recent decline, the Coral Index score remained relatively stable and varied around the threshold between 'moderate' and 'good' between 2015 and 2022. Despite the ongoing presence of crown-of-thorns starfish over this period, coral cover increased to the highest levels recorded since the start of the MMP in 2005. It is almost certain that removal of crown-of-thorns by the Reef Authority's Crown-of-thorns Starfish Control Program has mitigated the impact of these starfish. However, it is likely to have been a limiting factor precluding further improvement in the Coral Index over this period.

Contributing to the maintenance of coral cover has been the rate that coral cover has increased when not subject to acute pressures. The Cover change score has remained in the 'good' range for much of the last 19 years, declining in the aftermath of cyclone Yasi and a period of heavy rainfall to 'moderate' levels between 2012 and 2015 and again in 2023 and 2024.

In contrast to the Coral cover and Cover change scores, which demonstrate the ongoing support for increases in cover due to colony growth, are 'poor' scores for the Macroalgae and Juvenile coral indicators. In this sub-region macroalgal communities are dominated by a range of relatively small red algae species that that form dense matts both on dead coral colonies and in the spaces between coral branches and amongst coral rubble. While undertaking juvenile surveys, we have observed that few juvenile hard corals are present where these algae occur.

5.2.2.3 Herbert–Tully sub-region

The Herbert–Tully sub-region score has continued to decline from a high point in 2020. Most influential in this decline have been declines in Cover change and Juvenile coral scores however these remain in 'moderate' and 'good' condition respectively.

At Dunk North there was a sufficient reduction in cover for us to categorise an acute bleaching disturbance. It is highly likely that the poor performance in terms of Cover change at other reefs also reflect the stress associated with high water temperature, possibly exacerbated by the highest discharges from local rivers since 2011. For example, influencing the decline in the Cover change score since 2023 were slight declines in hard coral cover at the Barnards for which no acute disturbance was ascribed. Here, relatively high levels of disease were recorded, an observation consistent with studies that have shown that poor water quality can both increase corals susceptibility to disease and exacerbate the influence of heat stress (see section 5.1.2). Alternatively, the process of bleaching itself has been demonstrated to alter the composition of dissolved organic matter exuded by corals in a way that may promote pathogenic bacterial communities (Sparagon *et al.* 2024).

Although the Juvenile scores have declined substantially over the last five 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, that current juvenile communities at most reefs include a relatively high proportion of the family Merulinidae, a group that is collectively slow growing and so likely to remain in the juvenile size class for longer than fast growing taxa such as *Acropora,* adding a degree of bias to the score for this indicator. Generally, low densities of *Acropora* juveniles coupled with the ongoing ‘poor’ score for the Macroalgae indicator implies a potential pressure on recruitment processes imposed by water quality, a conclusion consistent with the moderate scores for the Water Quality Index with most water quality indicator concentrations encompassing or exceeding guidelines (Moran *et al.* 2025).

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 recovery capacity has been evident, and high turbidity appears to be limiting coral performance.

5.2.3 Burdekin Region

The Coral Index score for the Burdekin region declined from a peak reached in 2020 and remains ‘moderate’ in 2024. The decline since 2020 is due primarily to declines in Juvenile coral scores and, at 2 m depth, Macroalgae scores. Coral cover declined slightly in 2024, down from 2023 when it had reached the highest level since the inception of the MMP in 2005.

Coral cover losses between 2023 and 2024 were mostly attributed to coral bleaching in response to marine heatwave conditions with minor damage also attributed to cyclone Kirrily. The worst impacted reef was Palms East at 2 m depth where hard coral cover declined from 44% to 30% a loss attributed to coral bleaching. In contrast coral cover at Pandora, while still low, had increased to the highest levels observed since the MMP began in 2005 and signals the most tangible evidence of recovery since the demise of corals at this site since ca. 2001 (Done *et al.* 2007).

Since 2005 coral bleaching has accounted for 32% of interannual coral losses, with all these losses occurring since 2017. Despite these repeated impacts, the Coral cover score remained in the ‘moderate’ range in 2024, having increased since 2016. This increase demonstrates that, on balance, the ‘moderate’ rate of hard coral cover increase shown by the Cover change indictor scores has been sufficient to counter the losses incurred in response to the recent series of bleaching events.

Regionally, the 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 was observed between 2013 and 2020 in which the Coral Index increased 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 increasing Coral cover scores. Most other reefs had persistently low cover of fast-growing *Acropora*, therefore increases in coral cover were slower. 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. Since 2016, 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 two, as predicted by Wooldridge (2020). Hard coral cover is again increasing at this reef and was at 22% in 2024, but *A. pulchra,* a species common prior to 2017, is no longer present on the transects (*pers. obs*. Author).

The Macroalgae and Juvenile coral indicator scores remain categorised as ‘poor’. For Macroalgae, there is a clear demarcation in scores between Palms East and Palms West, where scores were ‘very good’, compared to most other sites deeper into Halifax Bay and Cleveland Bay, except at the 5 m sites at Lady Elliot, where macroalgae cover has been consistently low, and Pandora where macroalgae cover has declined in recent years. The lack of macroalgae at Lady Elliot may be explained by the satellite derived water quality estimates that show this reef to be in particularly turbid water that would limit light availability at the 5 m depth.

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.

5.2.4 Mackay–Whitsunday Region

The Coral Index in the Mackay–Whitsunday region declined dramatically from 2016 through to 2019 due to the impacts of cyclone Debbie, and at Dent Island, coral disease. In 2024, the Coral Index has remained 'poor'. While the 'poor' Cover change score demonstrates recovery continues to be slow, increasing densities of juvenile corals and modest increases in coral cover at some reefs show recovery is occurring.

Prior to cyclone Debbie, Coral Index scores had remained relatively stable in the 'moderate' range. During this period, Macroalgae scores remained 'good' as macroalgae cover was very low on most monitored reefs. Equally, Coral cover scores were generally 'good', except for a short decline to 'moderate' levels due to damage imposed by cyclone Ului in 2010. Reductions in the Composition score following cyclones implies additional selective pressures on those species (e.g., genus *Acropora*) sensitive to poor water quality. The primary limitation to Coral Index scores prior to cyclone Debbie was regionally 'poor' scores for the Cover change indicator as rates of coral cover increase were slow despite a lack of acute disturbance events.

Conditions at 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. Unfortunately, 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.

Since cyclone Debbie, the Cover change score has remained 'poor'. The Cover change score averages change in hard coral cover over a four-year period, 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. Noting, that expectations were very low for Daydream due to the almost complete removal of corals by cyclone Debbie, and at Hook where the community is predominantly comprised of slow growing taxa.

With the severe loss of coral cover at many sites, successful recovery will rely heavily on the recruitment and survival of juvenile corals. That the Juvenile score continued to increase within the 'moderate' range in 2024 is certainly a positive sign. However, juvenile densities remain poor at seven of the eleven sites where Macroalgae scores were 'very poor' in 2024.

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). Of concern is that prior to cyclone Debbie, persistently high cover of macroalgae was only present at Seaforth and at 2 m depths at Pine. In 2024, very poor scores for Macroalgae were recorded at Daydream, Dent, Double Cone, Pine, Seaforth and the 2 m depth at Shute. 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 will almost certainly be limiting coral recruitment. Among those reefs with relatively high macroalgae cover, the presence of Sargassaceae and *Lobophora* within macroalgae communities at 2 m depths at Daydream, Double Cone, Pine and Seaforth is worth noting as, once established, these species have proven persistent at other MMP reefs and have the potential to constrain coral recovery, potentially trapping benthic communities in a macroalgal dominated state (Mumby *et al.* 2013, Johns *et al*. 2018).

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 (Moran *et al.* 2025). Encouragingly, both indices have gradually improved within the ‘moderate’ range with the long-term index retuning to the level observed in 2010, prior to cyclone Ului, and consistent with levels in which prior, albeit slow, recovery of coral communities has been observed.

5.2.5 Fitzroy Region

Reefs in the Fitzroy region were severely impacted by marine heatwave conditions in early 2024 driving the Coral Index score to the bottom of the ‘poor’ range. Hard coral cover across the region declined by more than a third pushing the Coral cover score to ‘poor’ for the first time since 2019. As much of the coral killed was in the genus *Acropora* the Composition indicator also declined. Importantly, during surveys in May 2024, a high proportion of surviving corals in the region were still bleached. Given the ongoing stress evidenced by this bleaching it is likely the full impact of this bleaching event is yet to unfold, as further declines in coral cover the year after a bleaching event have been observed in other regions monitored by the MMP.

Contributing to decline in the Coral index was a decline in the Juvenile coral score. This decline reflects a reduction in numbers of juvenile corals compounded by the increase in space available to settling coral because of recently killed corals.

The Cover change score also declined a little in 2024, however as all reefs were classified as having been impacted by coral bleaching this result reflects four-year window over which the indicator is assessed. The current reduction being due to the rapid recovery in coral cover that occurred between 2019 and 2020 no longer being included.

Macroalgae cover remains persistently high, continuing the ‘very poor’ indicator score.

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 two 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. Reduction in light levels over extended periods of time due to increased concentrations of suspended sediments delivered by the floods, as well as dense plankton blooms following the floods, is another plausible explanation for the reduced fitness of corals (Cooper *et al*. 2007) and is supported by the clear relationship between river derived loads and change in Coral Index scores in this region.

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 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 appeared to have occurred as macroalgae were also killed by exposure to low salinity flood waters 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 Sargassaceae family and the genus *Lobophora.* The persistence of these macroalgae at Middle Island, where macroalgae cover was over 40% at both 2m and 5 m depths in 2024, has almost certainly limited the recovery of coral cover. The timeseries of coral and macroalgae covers at Middle Island, in particular, support work that demonstrates high macroalgal cover can lead to positive feedbacks 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 2024 had declined to 'very poor'. Adding to the limitations to coral recruitment imposed by high cover of macroalgae 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 observation that warrants consideration is that much of the algae-covered substrate occurs as the basal sections for live staghorn *Acropora*, or the remnants of these colonies, throughout inshore reefs we have observed 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. With recent losses of coral cover, it will be particularly important to track the Juvenile indicator for evidence of further reductions juvenile densities that would demonstrate broodstock limitation.

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 2024, the overall Coral cover score declined to the lowest value since 2019. The largest decline occurred in the Fitzroy region where coral bleaching, in response to severe marine heat wave conditions, resulted in substantial reductions in coral cover. Bleaching also occurred in Wet Tropics and Burdekin regions however additional pressures were also evident.

In both the Wet Tropics and Burdekin region the attribution of cover loss to bleaching was confounded by additional pressures. At Snapper Island it was clear that the combined exposure to waves and then flooding generated by cyclone Jasper, that killed all coral at Snapper South and most coral at the 2 m depth at Snapper North, negated potential impacts attributable to subsequent thermal stress. Further south in the Johnstone Russell–Mulgrave sub-region, reefs were exposed to thermal stress and severe bleaching was observed, however temperature estimates of DHW from both NOAA and *in situ* loggers suggest that coral losses were higher than expected for the observed anomalies. At 2 m depth at High Island East and Fitzroy West some storm damage was observed, and this will have contributed to coral cover losses. However, there was also a very clear depth gradient to coral mortality at High East consistent with exposure to freshwater. While this did not occur at Fitzroy Island, or at reefs in the Herbert–Tully subregion satellite imagery (Moran et al. 2025) showed the reefs were exposed to nutrient and sediment enriched flood plumes raising the prospect that poor water quality exacerbated the impact of thermal stress (see section 5.1.2). In the Burdekin region minor storm damage attribute to cyclone Kirrily was observed, most notably at Magnetic.

A further impost to coral cover in the Wet Tropics were outbreak densities of crown-of-thorns starfish. The 5 m depths at Franklands East and High East have been most affected by these in recent years.

The only region spared acute impacts in 2024 was Mackay–Whitsunday region where Coral cover scores continued to slowly improve as coral communities recover from the impact of cyclone Debbie.

In 2024, relationships between coral cover and concentrations TSS and Chl *a* were less apparent than they have been in previous years. This is not surprising given the level of recent impacts that will have overwhelmed the more subtle pressures imposed by ambient water quality conditions. It was only at the subset of reefs at which MMP water quality undertakes routine monitoring that Coral cover scores were related to water quality, at these reefs Coral cover scores tended to be lower where Chl *a* concentrations were high. 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 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 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, as 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 having very low coral cover and, at 5 m depth in particular, communities with low representation of fast-growing corals of the genus *Acropora*. Despite this limitation, scores remaining poor in the Mackay–Whitsunday region is of concern, as it highlights the ongoing slow rate of recovery since the severe impacts caused by cyclone Debbie in 2017.

A further issue arose in 2024 at Snapper South where all coral was killed and the Cover change score derived from valid estimates from the previous three years. This is consistent with the intent of the indicator in assessing the likely limitations to coral cover increase given the ambient environmental conditions at the reef. However, in this case the score is initially unintuitive given there is no prospect for an increase in coral cover until new corals settle and grow to a size available to the photo point intercept method used to estimate coral cover. The expected rates of change in coral cover prior to the flood impacts in late 2023 will continue to influence scores for this indicator in coming years. This is something readers should be aware of but, at the same time, we consider these scores remain valid as there is no evidence against which to assess that ambient conditions have changed.

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 subsequent improvement. The 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. The conclusion is that environmental conditions associated with the increased loads of sediments and nutrients delivered by these floods were sufficiently stressful to limit the recovery of coral cover and/or induce disease in susceptible species. This 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 into the ‘poor’ range (Moran *et al.* 2025) suppressed 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 subjectivity surrounding 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, although remaining ‘moderate’ in 2023, the Cover change scores declined in the Johnstone Russell–Mulgrave sub-region. Although crown-of-thorns starfish were active in 2021, 2022 and 2023 at High Island and Frankland Group reefs, acute disturbances due to crown-of-thorns starfish were only attributed to those reefs 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 resulted in an underestimate of the Cover change score in recent years.

Current scores for the Cover change indicator aggregate changes that have occurred since 2020, meaning any low-level or protracted impacts of the 2020 and 2022 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 have discovered that at least some species of corals perform best at, or slightly below, their local average temperature, with performance curves declining once this 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 high levels of bleaching was evident during the initial post event surveys.

In 2024 the Cover change scores were the only indicator scores that showed a relationship to either the wet season Chl a or k490 light attenuation coefficient estimates for each reef. However, a relationship was only evident in the Burdekin region and clearly driven by a single point. On investigation, the high Change score at high Chl a concentration occurred at the 5 m site at Magnetic where only the change between 2022 and 2023 was informative as all other observations over the last 4 years were categorised as bleaching impacts. This result should not be over interpreted.

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. That no relationship was observed in 2024 in the Mackay–Whitsunday region can be explained by low values of community composition at reefs across the entire water quality gradient due to losses of *Acropora* on reefs impacted by cyclone Debbie

In 2024, the Composition indicator score declined in each sub-region of the Wet Tropics and the Burdekin and Fitzroy regions, where declines were largely to proportionate declines in Coral cover scores. Conversely both Coral cover and Composition scores showed slight gains in the Mackay–Whitsunday regions. 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 were improving demonstrating the gradual recovery of this group. The decline to ‘very poor’ in 2024 shows disproportionate loss of *Acropora* during the 2024 bleaching event.

Branching *Acropora* were one group identified by Roff *et al.* (2013) as showing reductions in contemporary communities, with reduced representation since the mid-20th 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 in light of 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 scoring 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

The lack of 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 a higher proportion of macroalgae in algal communities and higher overall cover of macroalgae at 2 m depth on reefs exposed to relatively high concentrations of Chl *a* and TSS. 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.

Of ongoing concern for the resilience of coral communities in 2024 is the increasing presence and persistence of macroalgae. Dense canopies of macroalgae that compete with corals dominate the benthic communities at several reefs in this study. Macroalgae benefit from disturbances that impact coral communities and make available substrate previously occupied by corals (McManus & Polsenberg 2004, Diaz-Pulido *et al.* 2007, Diaz-Pulido *et al.* 2009, Ceccarelli *et al.* 2020). Over the period of the MMP, increased cover of macroalgae was precipitated by the loss of coral cover following widespread disturbances of flooding, cyclones, and bleaching. 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.

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 feedbacks.

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 three *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 NOx values in most regions 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 factors (bottom-up controls such as nutrient levels) (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) link 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 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 of widespread abiotic disturbances such as floods, marine heatwaves, and cyclones is expected to increase in frequency and intensity, 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, increasing the potential for long-term phase shifts on an increasingly larger scale.

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 the juvenile coral indicator scores do not correspond to observed gradients in water quality almost certainly reflects the interaction of a range of additional limiting factors such as acute disturbances, variable connectivity to brood-stock populations and changes in juvenile community composition among sites.

In 2024, Juvenile coral scores declined to 'very poor' in the Barron Daintree sub-region as juvenile corals were killed by the cumulative impacts associated with cyclone Jasper. Similarly, the severe impacts of coral bleaching pushed juvenile densities into the 'very poor' range in the Fitzroy region. While these acute impacts are to be expected, that in both areas Juvenile scores have been 'poor' or 'very poor' for almost the entire 20-year period of the MMP raises serious concern for the recovery of these severely impacted communities.

In the Herbert–Tully sub-region, while the Juvenile score remained 'good' the density of juvenile corals continued to decline, a trend evident for several years. In the Burdekin region and Johnstone Russell-Mulgrave sub-region scores have varied within the poor' range for the last three years.

A recently emerging pattern is that the coral genus *Turbinaria* has recruited 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 pattens of recruitment in taxa necessary for rapid recovery of coral communities, such as *Acropora*.

In contrast to other regions the density of juvenile corals continued to increase in the Mackay–Whitsunday region where this is the only indicator scored as 'moderate'. The 'moderate' score for Juvenile corals in this region is strongly supported by very high recruitment of corals at Hayman, Daydream and Hook (2 m) in recent years. The density of juvenile corals remains low at most other reefs.

Within regions the density of Juveniles is highly variable. At many reefs with persistently very poor scores for Macroalgae, the scores for the Juvenile coral indicator 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 and so potentially adding a positive bias to the indicator score where they are proportionally well represented among the juvenile community.

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.

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.

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. The small size and isolation of many inshore reefs may make such controls particularly feasible. MMP surveys in 2024 noted an increase in densities of crown-of-thorns starfish at most reefs, despite the removal of more than 1000 individuals from the Franklin Group over the preceding year. Elevated populations of crown-of-thorns have been present on these reefs since 2012, with the starfish observed consistently, including individuals across a range of size-class, demonstrating their ongoing recruitment to these reefs. Our data cannot investigate the likely source populations for the juveniles observed but these observations 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 can reduce these pressures have the potential to enhance the resilience of coral communities in inshore areas. It must be noted, however, that the environment occupied by many macroalgae is still suitable for corals, and it may be that density-dependent feedbacks 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), though 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

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

global carbon emissions to avoid irreversible loss of coral reef ecosystems (Van Oppen & Lough 2018, GBRMPA 2019, Hoegh-Guldberg *et al.* 2019).

6 CONCLUSIONS

Results from 2024 reveal the overall condition of inshore reefs has declined to the lowest point since the MMP began in 2005. 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 (GBRMPA 2024, Abelson 2020). Central to maximising recovery potential will be management actions that reduce the influence of chronic pressures 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, coral response thresholds 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 to 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 water quality constituents 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 c.a. 2016-2022 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 cyclones, crown-of-thorns, flooding and high water temperatures resulting in coral bleaching have variously contributed to declines in Coral index scores in each of these regions in 2024.

- 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 the cover of hard corals being reduced by more than a third. High levels of ongoing bleaching at the time of the surveys in 2024 demonstrate ongoing stress and the likelihood of further loss of coral. The added pressures associated with high water temperatures have occurred during relatively low rainfall and minimum cyclone activity during which 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.
- Reefs in the Mackay–Whitsunday region were not exposed to any severe acute pressures over the 2023/24 summer. In 2017 most reefs in the region were severely impacted by cyclone Debbie, from which recovery has been slow. Although the Coral Index remained ‘poor’, the density of juvenile coral has been increasing in recent years, contributing to a gradual improvement in coral cover. While these improvements illustrate ongoing recovery potential, both indicators remain in ‘poor’ condition, as do scores for the Macroalgae and Cover change indicators. This combination strongly points to chronic environmental pressures limiting the recovery potential of coral communities in the region.

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)
‘392’ and ‘Odyssey’ (Dataflow System)	2005 to 2008.	30
‘Sensus Ultra’ (ReefNet)	2008 to 2017	10
‘Vemco Minilog-II-T’ (Vemco)	2015 onward	10
,‘SBE-56‘ (Sea-Bird Scientific) – note: occassional deployments	2018 onward	10
,‘RBR‘ (RBR-Global) – note: increasingly replacing Vemco loggers	2020 onward	10

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	Genus	2 m	5 m
Psammocora	-0.194	-0.366	*Scolymia* *	0.001	0.000
Turbinaria	-0.279	-0.307	*Ctenactis* *	0.016	0.001
Goniopora	-0.320	-0.304	*Anacropora* *		0.001
Goniastrea	-0.115	-0.278	*Physogyra*	0.000	0.001
Pachyseris	-0.077	-0.235	*Cynarina* *	-0.000	0.004
Favites	-0.096	-0.230	*Sandalolitha**	0.003	0.005
Alveopora	-0.076	-0.221	*Montastrea*	0.019	0.005
Hydnophora	-0.047	-0.213	*Fungia*	0.013	0.015
Cyphastrea	-0.386	-0.193	Encrusting *Acropora*	0.048	0.015
Galaxea	-0.081	-0.159	*Acanthastrea* *	-0.014	0.017
Mycedium	-0.017	-0.151	*Symphyllia*	0.034	0.018
Favia	-0.134	-0.136	*Seriatopora*	0.05	0.027
Pectinia	-0.030	-0.126	*Stylophora*	0.035	0.033
Podobacia	-0.025	-0.122	*Oulophyllia*	0.02	0.037
Plesiastrea	-0.125	-0.114	Digitate *Acropora*	0.034	0.039
Echinophyllia	-0.002	-0.11	*Montipora*	-0.131	0.045
Moseleya *	-0.058	-0.091	*Leptastrea* *	0.022	0.048
Oxypora	-0.008	-0.076	*Coeloseris*	0.052	
Merulina	-0.01	-0.073	Bottlebrush *Acropora*	0.153	0.070
Coscinaraea	-0.011	-0.062	*Pocillopora*	0.058	0.074
Duncanopsammia *		-0.042	Branching *Porites*	0.059	0.075
Caulastrea	0.007	-0.041	*Leptoria*	0.054	0.077
Platygyra	0.048	-0.040	*Porites rus*	0.122	0.087
Herpolitha	-0.013	-0.034	*Echinopora*	0.076	0.096
Lobophyllia	0.018	-0.034	Massive *Porites*	-0.054	0.122
Pavona	-0.152	-0.024	*Diploastrea*	0.003	0.173
Astreopora	0.031	-0.023	Tabulate *Acropora*	0.052	0.224
Euphyllia	-0.012	-0.023	Corymbose *Acropora*	0.060	0.240
Leptoseris	-0.011	-0.021	Branching *Acropora*	0.657	0.810
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			

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 Moran *et al.* (2025). 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
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*
	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
	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
	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
	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
	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
	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
	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
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
	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	0.7	0.8	1
	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
	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
	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
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
	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
	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
	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
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
	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
	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

*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 survey 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				Other recorded disturbances
		1998	2002	2017	2024	
Barron–Daintree	Snapper North	0.92 (19%)	0.95 (Nil)	58% (2 m) 38%t (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, 66% at 5 m), cyclone Ita 12th April 2014 (90% at 2 m, 50% at 5 m) – possible flood associated and COTS 2014, cyclone Jasper 13th December 2023 (74% at 2 m, 1% at 5 m)
	Snapper South	0.92 (Nil)	0.95 (Nil)	5% (2 m) 1% (5 m)		Flood 1996 (87%), Flood 2004 (32%), COTS 2013 (26% at 2 m, 17% at 5 m), cyclone Ita April 12th, 2014 (18% at 2 m, 22% at 5 m), Flood 2019 (38% at 2 m, includes probable impact of pre-cyclone Owen), Flood 2024 (100% at 2 m and 5 m, includes impact from cyclone Jasper)
	Low Islets					COTS 1997-1999 (69%), Multiple disturbances (cyclone Rona, COTS) 1999-2000 (61%), Multiple disturbances (cyclone Yasi, bleaching and disease) 2009-2011 (23%), COTS 2013-2015 (38%), COTS + Bleaching 2019 (24%)
Johnstone Russell–Mulgrave	Fitzroy East	0.92	0.95	15% (2 m) 10%(5 m)*	33% (2 m)	cyclone Felicity 1989 (75% manta tow data), 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 (27% at 2 m, 48% at 5 m), Bleaching 2017* assessed in 2018, COTS 2021 (35% 2 m, 12% 5 m)
	Fitzroy West	0.92 (13%)	0.95(15%)	21% (2 m) 24% (5 m)	53% (2 m)	COTS 1999-2000 (78%), cyclone Hamish 2009 (stalled recovery trajectory), 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 13th December 2023 (10% at 5 m)
	Fitzroy West LTMP	12%				COTS and continued bleaching 2000 (80%), COTS 2013 (6%), COTS 2014-15(46%), COTS 2022 (16%)
	Franklands East	0.92 (43%)	0.80 (Nil)	22% (2 m) 30%* (5 m)		Unknown although likely COTS 2000 (68%) cyclone Larry 2006 (64% at 2 m, 50% at 5 m), Disease 2007-2008 (35% at 2 m), cyclone Tasha/Yasi 2011 (61% at 2 m, 41% at 5 m), 2017* COTS likely to have contributed, COTS 2020 (8% at 5 m), COTS 2021 (45% 5 m), COTS 2024 (40% at 2 m, 47% at 5 m)
	Franklands West	0.93 (44%)	0.80 (Nil)	17%* (2 m) 21% (5 m)		Unknown although likely COTS 2000 (35%) cyclone Tasha/Yasi 2011 (35% at 2 m), 2017* COTS likely to have contributed, COTS 2021 (13% 2 m)
	High East	0.93	0.80	27% (2 m) 11%* (5 m)		cyclone Tasha/Yasi 2011 (81% at 2 m, 58% at 5 m), 2017* COTS likely to have contributed, 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)
	High West	0.93	0.80	18% (2 m) 27% (5 m)		cyclone Larry 2006 (25% at 5 m), Flood/Bleaching 2009 (11% at 2 m), Storm 2011 (21% at 2 m, 35% at 5 m), COTS 2021 (26% 5 m), COTS 2023 (15% at 2 m, 42% at 5 m)
	Green			12 %		COTS: 1994 (21%), 1997 (55%), 2011-2013 (44%), 2014-2015 (47%)

Table A6 continued

(sub-)region	Reef	Bleaching 1998	2002	2017	2020	2024	Other recorded disturbances
Herbert–Tully	Barnards	0.93	0.80	17% (2 m)			cyclone Larry 2006 (95% at 2 m 87% at 5 m), cyclone Yasi 2011 (53% at 2 m, 24% at 5 m), Bleaching 2018 (10% at 5 m), Disease 2021 (18% 2 m, 9% 5 m)
	King Reef	0.93	0.85	n/a			cyclone Larry 2006 (56% at 2 m,50% at 5 m), cyclone Yasi 2011 (71% at 2 m, 37% at 5 m)
	Dunk North	0.93	0.80	18% (2 m) 16% (5 m)		15% (2 m) 16% (5 m)	cyclone Larry 2006 (81% at 2 m, 71% at 5 m), Disease 2007 (34% at 2 m), cyclone Yasi 2011 (93% at 2 m, 75% at 5 m)
	Dunk South	0.93	0.85	45% (2 m) 6% (5 m)	20% (2 m) 12% (5 m)		cyclone Larry 2006 (23% at 2 m, 19% at 5 m), cyclone Yasi 2011 (79% at 2 m, 56% at 5 m), Bleaching 2018 (28% at 5 m)
	Bedarra	n/a	n/a	36% (2 m) 10% (5 m)	16% (2 m) 10% (5 m)		Bleaching 2018 ongoing from 2017 (26% at 5 m)

Table A6 continued

Region	Reef	Bleaching					Other recorded disturbances
		1998	2002	2017	2020	2024	
Burdekin	Palms East	0.93	0.80			40% (2 m) 29% (5 m)	cyclone Larry 2006 (23% at 2 m, 39% at 5 m), cyclone Yasi 2011 (83% at 2 m and 5 m)
	Palms West	0.92 (83%)	0.80	30% (2 m) 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 25th January 2024 (28% at 2 m)
	Lady Elliott Reef	0.93	0.85		26% (2 m) 8% (5 m)		cyclone Yasi 2011 (86% at 2 m, 45% at 5 m)
	Pandora Reef	0.93 (21%)	0.85 (2%)	33% (2 m)	18% (2 m)		cyclone Tessie 2000 (9%), cyclone Larry 2006 (80% at 2 m, 34% at 5 m), Storm 2009 (37% at 2 m, 56% at 5 m), cyclone Yasi 2011 (30% at 2 m, 57% at 5 m)
	Pandora North	11%		5 %*	n/a		cyclone Yasi 2011 (25%), cyclone Kirrily 25th January 2024 (11%)
	Havannah	0.93	0.95	37% (2 m) 11% (5 m)	33% (2 m) 8% (5 m)		Combination of cyclone Tessie and COTS 1999-2001 (66%) cyclone Yasi 2011 (35% at 2 m, 34% at 5 m), Disease 2016 (9% at 2 m), Bleaching ongoing impact of 2017 recorded in 2018 (26% at 2 m, 16% at 5 m), Disease 2019 (23% at 2 m), Post 2020 bleaching (2021, 26% 2 m)
	Havannah North	49%	21%		51%		cyclone Tessie 2000 (54%), 2001 COTS (44%) cyclone Yasi 2011 (69%), cyclone Kirrily 25th January 2024 (16%)
	Middle Reef LTMP	(7%)	(12%)	n/a	n/a		Flood 2009 (20%)
	Magnetic	0.93 (24%)	0.95 (37%)	32% (2 m)	36% (2 m) 18% (5 m)	46% (2 m) 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), Post 2020 bleaching (2021, 13% 5 m)

Table A6 continued

Region	Reef	Bleaching 1998	Bleaching 2002	Bleaching 2017	Bleaching 2020	Other recorded disturbances
Mackay–Whitsunday	Hook	0.57	1		27% (2 m) 20% (5 m	Coral 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 (recorded in 2018) (83% at 2 m, 45% at 5 m)
	Dent	0.57 (32%)	0.95	**		Disease 2007(17% at 2 and at 5 m), cyclone Ului 2010 most likely although reef not surveyed in that year (21% at 2 m, 27% at 5 m), cyclone Debbie 2017 (48% at 2 m, 38% 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	0.57	0.95	**	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	0.57	1	**	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	0.31 (44%)	1	**	42% (2 m) 38% (5 m)	Disease 2008 (26% at 2 m, 20% at 5 m), cyclone Ului 2010 (47% at 2 m, 46% at 5 m), cyclone Debbie 2017 (98% at 2 m, 90% at 5 m)
	Shute Harbour	0.57	1	**	10% (2 m)	cyclone Ului 2010 (8% at 2 m), cyclone Debbie 2017 (48% at 2 m, 55% at 5 m)
	Pine	0.31	1	**	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					cyclone Ului 2010 (36%), cyclone Debbie 2017 (recorded 2019) (86%)
	Langford					cyclone Debbie 2017 (recorded 2019) (56%)
	Border		(11%)			cyclone Debbie 2017 (recorded 2019) (45%)

Table A6 continued

Region	Reef	Bleaching 1998	2002	2006	2020	2024	Other recorded disturbances
Fitzroy	Barren	1	1	25% (2 m) 30% (5 m)		58% (2 m) 16% (5 m)	Storm Feb 2008 (43% at 2 m, 24% at 5 m), Storm Feb 2010 plus disease (25% at 2 m,8% 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)
	North Keppel	1 (15%)	0.89 (36%)	61% (2 m) 41% (5 m)	18% (2 m) 7% (5 m)	67% (2 m) 34% (5 m)	Storm Feb 2010 possible although not observed as site was not surveyed in that year. 2011 ongoing disease (26% at 2 m and 54% at 5 m)
	Middle Is	1 (56%)	1 (Nil)	61% (2 m) 38% (5 m)	15% (2 m)	64% (2 m) 53% (5 m)	Storm Feb 2010 plus disease (29% at 2 m, 42% at 5 m) cyclone Marcia 2015 (30% at 2 m, 32% at 5 m), Post 2020 bleaching (2021, 49% 2 m), Disease 2023 (41% at 2m)
	Keppels South	1 (6%)	1 (26%)	27% (2 m) 28% (5 m)	1% (2 m) 2% (5 m)	53% (2 m) 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), Post 2020 bleaching (2021, 22% 5 m)
	Pelican	1	1	17% (5 m)		58% (2 m) 8% (5 m)	Flood /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), Post 2020 bleaching (2021, 66% 2 m)
	Peak	1	1				Flood 2008 (28% at 2 m), Flood 2011 (70% at 2 m, 27% at 5 m)

Note: As direct observations of impact were limited during the widespread bleaching events of 1998 and 2002, tabulated values for these years are the estimated probability that each reef would have experienced a coral bleaching event as calculated using a Bayesian Network model (Wooldridge & Done 2004). The network model allows information about site-specific physical variables (e.g., water quality, mixing strength, thermal history, wave regime) to be combined with satellite-derived estimates of sea surface temperature (SST) to provide a probability (= strength of belief) that a given coral community would have experienced a coral bleaching event. Higher probabilities indicate a greater strength of belief in both the likelihood of a bleaching event and the severity of that event. Where impact was observed the proportional reduction in coral cover is included. For all other disturbances listed the proportional reductions in cover are based on direct observation.

Table A7 Reef-level Coral Index and indicator scores 2024. 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	Depth	Coral cover	Juvenile coral	Macroalgae	Cover change	Composition	Coral Index
Barron Daintree	Low Isles	5	0.66	0.78	0.92	0.54	0.5	0.68
	Snapper North	2	0.18	0.02	0	0.60	0	0.16
	Snapper North	5	0.54	0.10	1	0.79	0.5	0.59
	Snapper South	2	0	0	1	0.74	0.5	0.45
	Snapper South	5	0	0	0.53	0.65	0	0.24
Moderate			**0.28**	**0.18**	**0.69**	**0.67**	**0.3**	**0.42**
Johnstone Russell Mulgrave	Fitzroy East	2	0.48	0.26	0.88	0.44	0.5	0.51
	Fitzroy East	5	0.77	0.48	0.36	0.61	0	0.44
	Fitzroy West	2	0.65	0.22	0	1	0.5	0.47
	Fitzroy West	5	0.79	0.43	0.12	0.91	0.5	0.55
	Fitzroy West LTMP	5	0.78	0.98	0.97	0.67	1	0.88
	Franklands East	2	0.60	0.32	0.54	0.41	1	0.57
	Franklands East	5	0.33	0.30	0.13	0.50	1	0.45
	Franklands West	2	0.84	0.17	0	0.46	0.5	0.39
	Franklands West	5	0.86	0.25	0	0.64	0.5	0.45
	High East	2	0.36	0.12	0	0.12	0.5	0.22
	High East	5	0.55	0.21	0	0.36	0.5	0.32
	High West	2	0.75	0.44	0.54	0.80	0	0.5
	High West	5	0.33	0.29	0.79	0.74	0	0.43
Moderate			**0.62**	**0.34**	**0.33**	**0.59**	**0.5**	**0.48**
Tully Herbert	Barnards	2	0.73	0.19	0.44	0.50	1	0.57
	Barnards	5	0.75	0.52	0.79	0.15	1	0.65
	Dunk North	2	0.65	0.60	0	0.59	0.5	0.47
	Dunk North	5	0.49	0.90	0.32	0.77	0.5	0.6
	Dunk South	2	0.54	0.37	0.39	0.42	1	0.54
	Dunk South	5	0.51	0.70	0	0.37	0	0.32
	Bedarra	2	0.22	0.58	0	0.27	0.5	0.31
	Bedarra	5	0.39	1	0.95	0.36	0	0.54
Moderate			**0.54**	**0.61**	**0.36**	**0.41**	**0.56**	**0.5**
Burdekin	Palms East	2	0.43	0.11	0.99	0.25	1	0.56
	Palms East	5	0.56	0.14	0.96	0.28	1	0.59
	Palms West	2	0.39	0.26	1	0.70	0	0.47
	Palms West	5	0.54	0.44	1	0.38	0	0.47
	Havannah North	5	0.32	0.78	0	0.50	0.5	0.42
	Havannah	2	0.48	0.24	0	0.36	1	0.42
	Havannah	5	0.63	0.34	0.02	0.30	1	0.46
	Pandora	2	0.28	0.16	0	0.45	0.5	0.28
	Pandora	5	0.38	0.44	0.58	0.46	1	0.57
	Pandora North	5	0.84	0.21	0	0.44	0	0.3
	Lady Elliot	2	0.38	0.28	0	0.21	1	0.37
	Lady Elliot	5	0.54	0.54	0.86	0.52	0	0.49
	Magnetic	2	0.25	0.14	0	0.30	0.5	0.24
	Magnetic	5	0.35	0.35	0	1	0.5	0.44
Moderate			**0.46**	**0.32**	**0.39**	**0.44**	**0.57**	**0.43**

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	0.71	0.5	0.71
	Border	5	0.53	0.57	1	0.19	0	0.46
	Hook	2	0.28	0.52	1	0.31	0	0.42
	Hook	5	0.48	0.38	0.36	0.53	0.5	0.45
	Double Cone	2	0.04	0.33	0	0.21	0	0.12
	Double Cone	5	0.22	0.25	0	0.21	0	0.14
	Daydream	2	0.05	0.61	0	0.47	0	0.22
	Daydream	5	0.14	1	0	0.54	0	0.34
	Dent	2	0.46	0.34	0	0.25	0	0.21
	Dent	5	0.46	0.32	0	0.23	0	0.2
	Shute Harbour	2	0.83	0.39	0	0.31	1	0.51
	Shute Harbour	5	0.43	0.5	0.44	0.35	1	0.54
	Pine	2	0.14	0.35	0	0.26	0.5	0.25
	Pine	5	0.29	0.42	0	0.4	0	0.22
	Seaforth	2	0.27	0.28	0	0.11	0	0.13
	Seaforth	5	0.26	0.4	0	0.18	0.5	0.27
Poor			**0.33**	**0.48**	**0.24**	**0.33**	**0.25**	**0.32**
Fitzroy	Barren	2	0.43	0.19	1	0.54	0	0.43
	Barren	5	0.84	0.04	0	0.52	0	0.28
	North Keppel	2	0.25	0.03	0	0.25	1	0.31
	North Keppel	5	0.31	0.09	0	0.28	0.5	0.24
	Middle	2	0.13	0.07	0	0.18	0	0.08
	Middle	5	0.17	0.19	0	0.17	0	0.11
	Keppels South	2	0.39	0.05	0	0.41	0	0.17
	Keppels South	5	0.29	0.12	0	0.31	0	0.14
	Pelican	2	0.06	0.1	0	0.53	0	0.14
	Pelican	5	0.38	0.3	0	0.36	0	0.21
Poor			**0.33**	**0.12**	**0.1**	**0.36**	**0.15**	**0.21**

Table A8 Environmental covariates for coral locations. Wet season Chl *a* and TSS median values over the 2020-2024 wet seasons estimated form the proportion of time Sentinel satellite imagery pixels adjacent to each site were classified as water types I-IV (Moran et al. 2025) and the distribution of niskin samples taken within each water type. Niskin Chl a, Suspended solids and the ration of N to P are mean values from MMP routine water quality monitoring sites between July 2020 and June 2024. Values exceeding Reef wide wet-season (0.63 μgL^{-1} Chl *a*, and 2.4 mgL^{-1} for TSS) or annual (0.45 μgL^{-1} Chl *a*, and 1.6 mgL^{-1} for TSS) guideline values (GBRMPA 2010, Moran *et al.* 2025) are shaded.

(sub-)region	Reef	Wet season Chl *a* (μgL^{-1})	Wet season TSS (mgL^{-1})	Attenuation coefficient k490 (m^{-1})	Niskin Chl a (μgL^{-1})	Niskin Suspended solids (mgL^{-1})	Total N (µM)/ Total P (µM)
Barron–Daintree	Low Isles	0.28	1.46	0.08			
	Snapper North	0.39	2.07	0.14	0.38	1.38	25.89
	Snapper South	0.40	2.14	0.14			
Johnstone Russell–Mulgrave	Franklands East	0.27	1.37	0.07			
	Fitzroy East	0.27	1.38	0.07			
	Franklands West	0.28	1.48	0.08	0.35	0.71	27.52
	High East	0.31	1.65	0.11			
	Fitzroy West	0.31	1.69	0.09	0.31	0.76	25.99
	High West	0.38	2.09	0.12	0.45	1.30	26.03
Herbert–Tully	Barnards	0.34	1.81	0.12			
	Dunk North	0.38	2.07	0.13	0.48	2.05	24.86
	Dunk South	0.41	2.24	0.15			
	Bedarra	0.43	2.36	0.16			
Burdekin	Palms East	0.26	1.27	0.07			
	Havannah North	0.28	1.45	0.09			
	Havannah	0.30	1.62	0.09			
	Palms West	0.31	1.58	0.09	0.37	0.81	26.20
	Pandora North	0.33	1.81	0.11			
	Pandora	0.36	1.93	0.12	0.38	1.33	25.45
	Magnetic	0.49	2.54	0.18	0.59	2.25	23.67
	Lady Elliot	0.49	2.56	0.19			
Mackay–Whitsunday	Hayman	0.25	1.21	0.06			
	Border	0.26	1.32	0.08			
	Hook	0.27	1.39	0.08			
	Double Cone	0.28	1.49	0.09	0.45	1.63	23.87
	Seaforth	0.31	1.68	0.10	0.52	1.97	23.52
	Daydream	0.32	1.70	0.11	0.54	2.33	21.39
	Dent	0.35	1.83	0.11			
	Shute Harbour	0.36	1.92	0.11			
	Pine	0.38	2.05	0.12	0.56	3.25	21.62
Fitzroy	Barren	0.27	1.41	0.06	0.29	0.51	26.68
	North Keppel	0.35	1.93	0.09			
	Middle	0.36	1.92	0.11			
	Keppels South	0.38	2.06	0.10	0.53	0.86	26.60
	Pelican	0.51	2.66	0.17	0.64	3.57	21.12

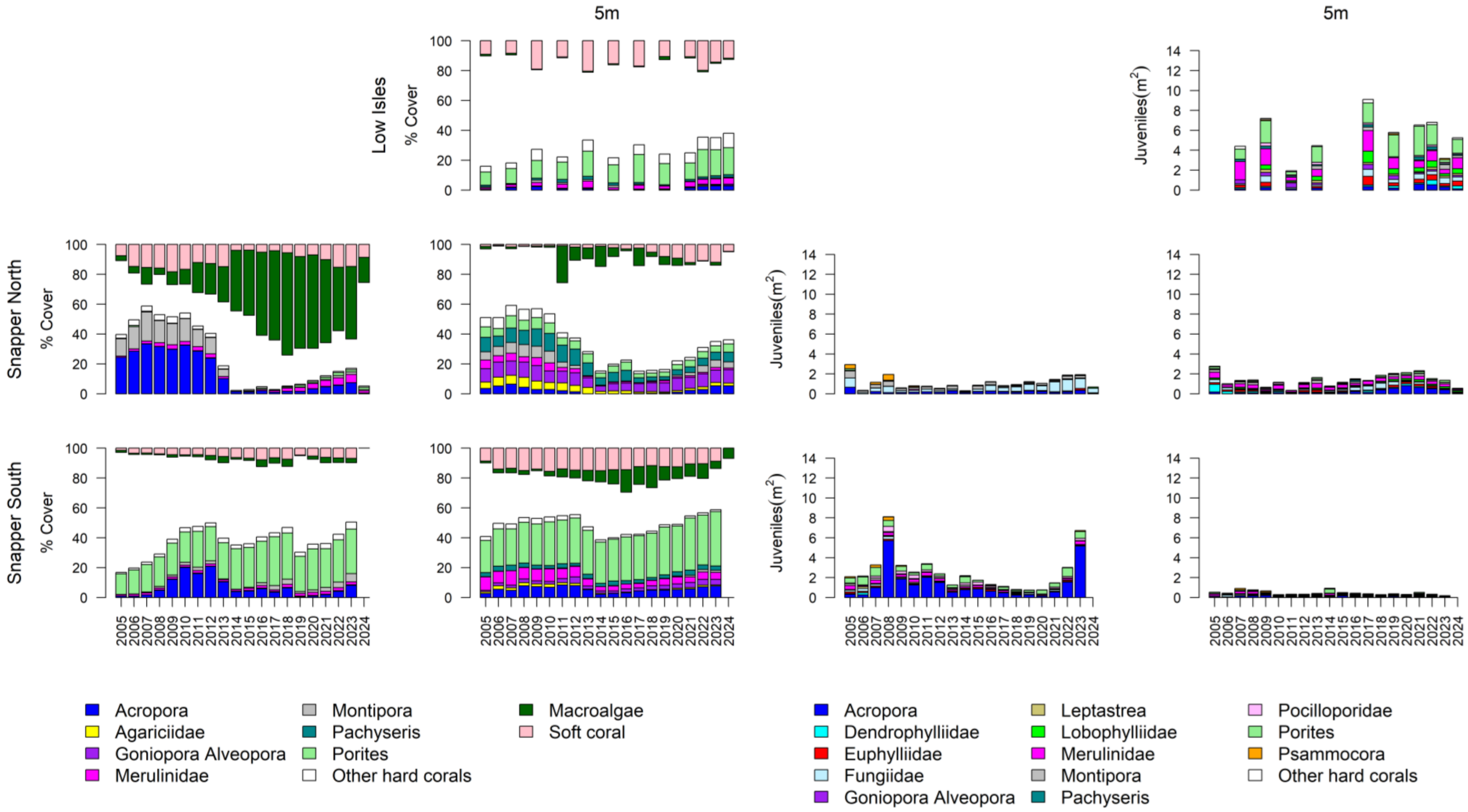

Figure A1 Barron–Daintree sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral groups and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral groups. Separate legends relevant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

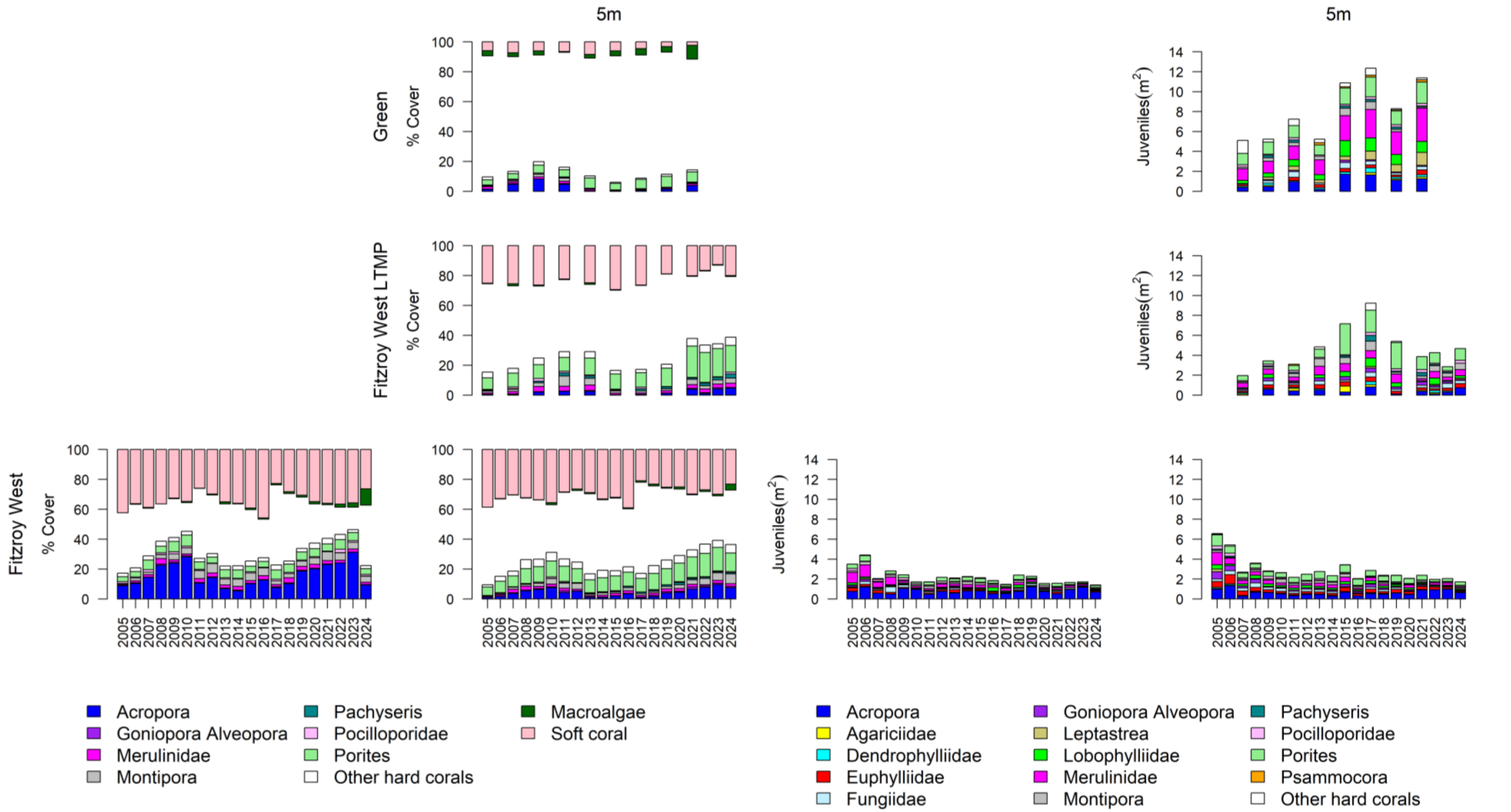

Figure A2 Johnstone Russell–Mulgrave sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral groups and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral groups. Separate legends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

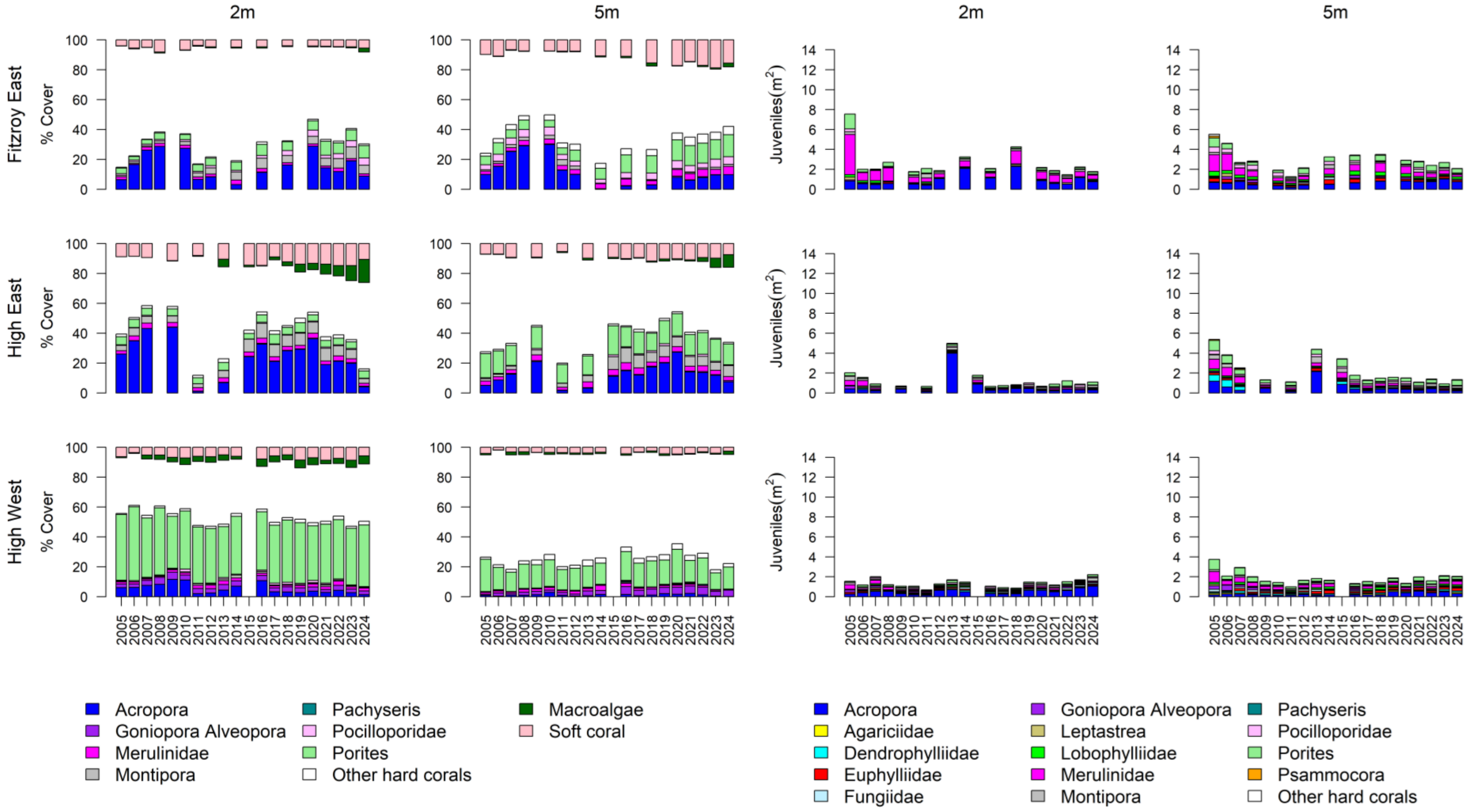

Figure A2 continued

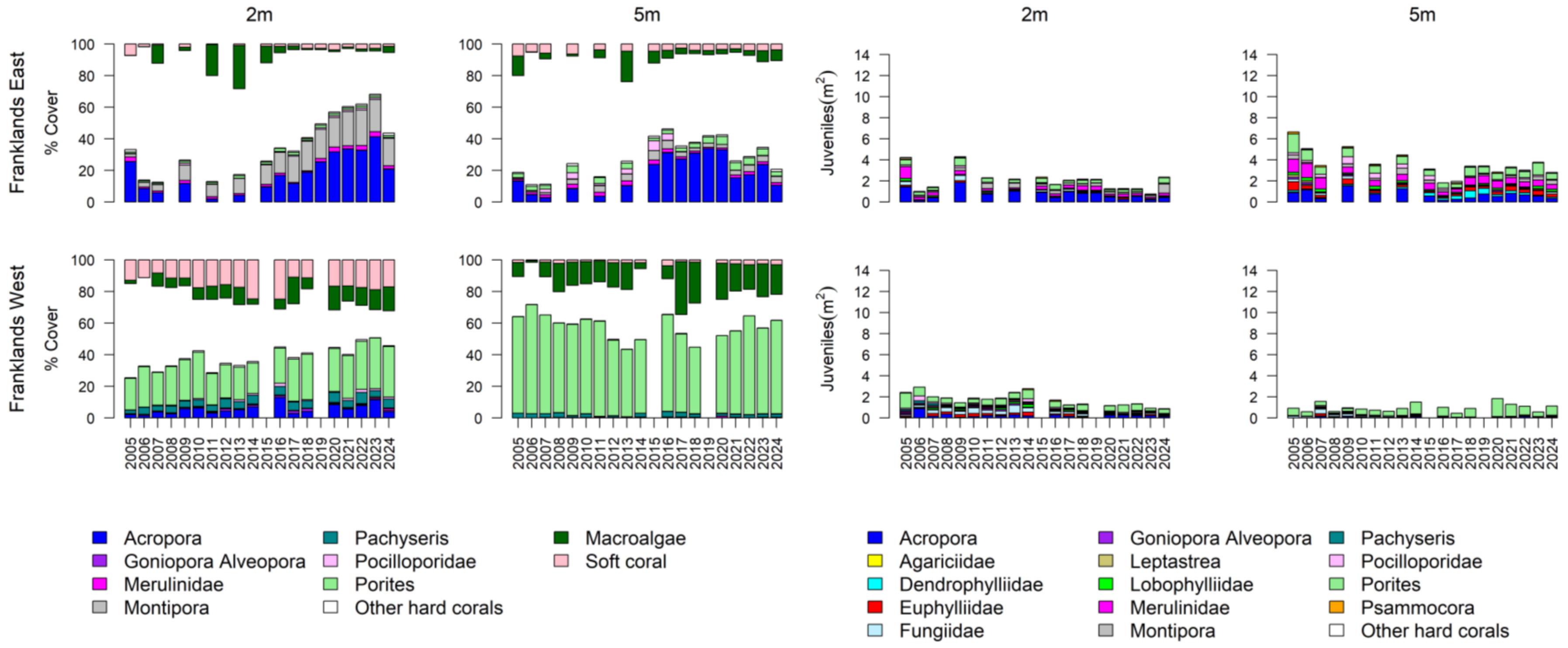

Figure A2 continued

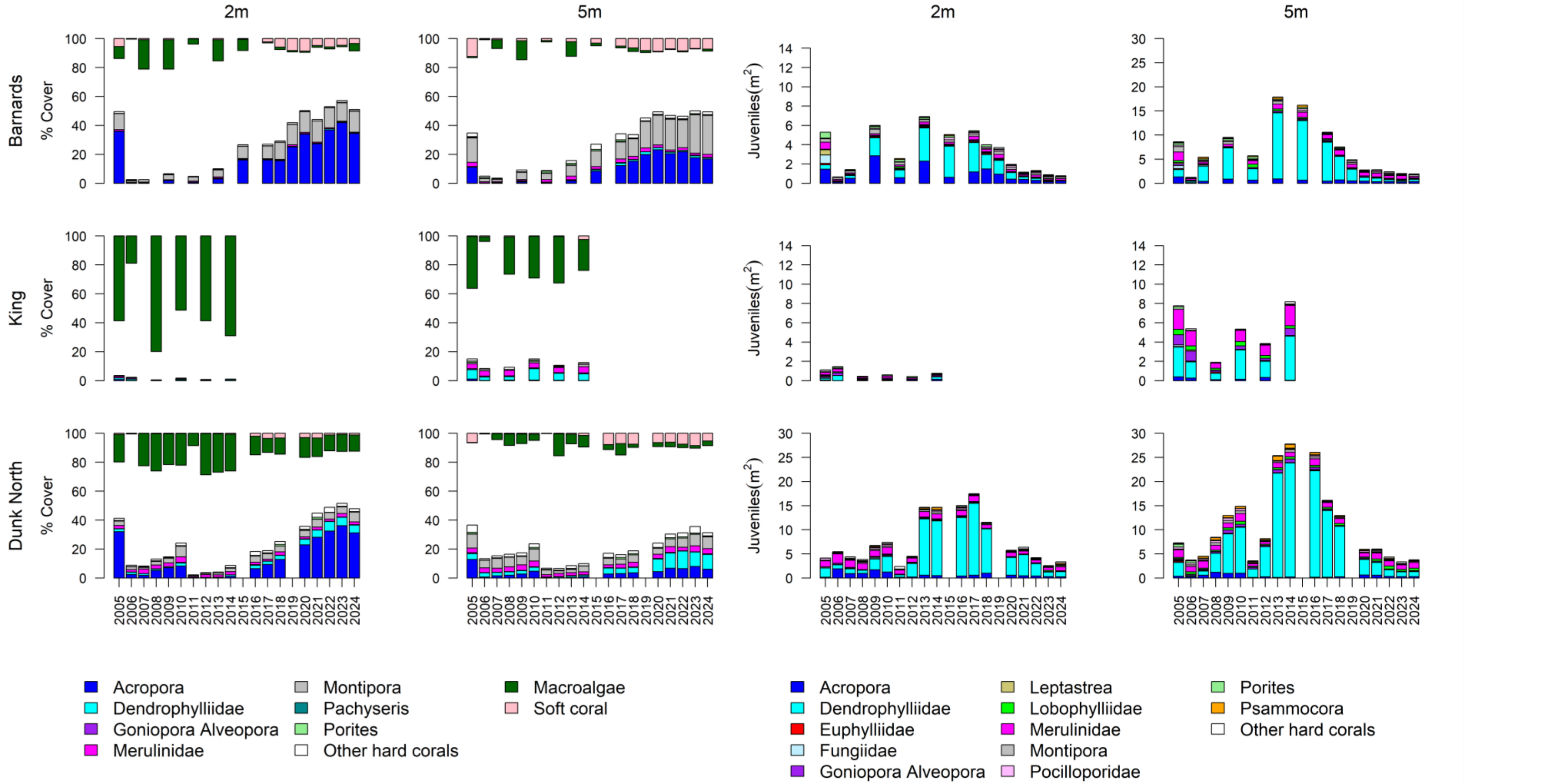

Figure A3 Herbert–Tully sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral groups and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral groups. Separate legends with relevant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

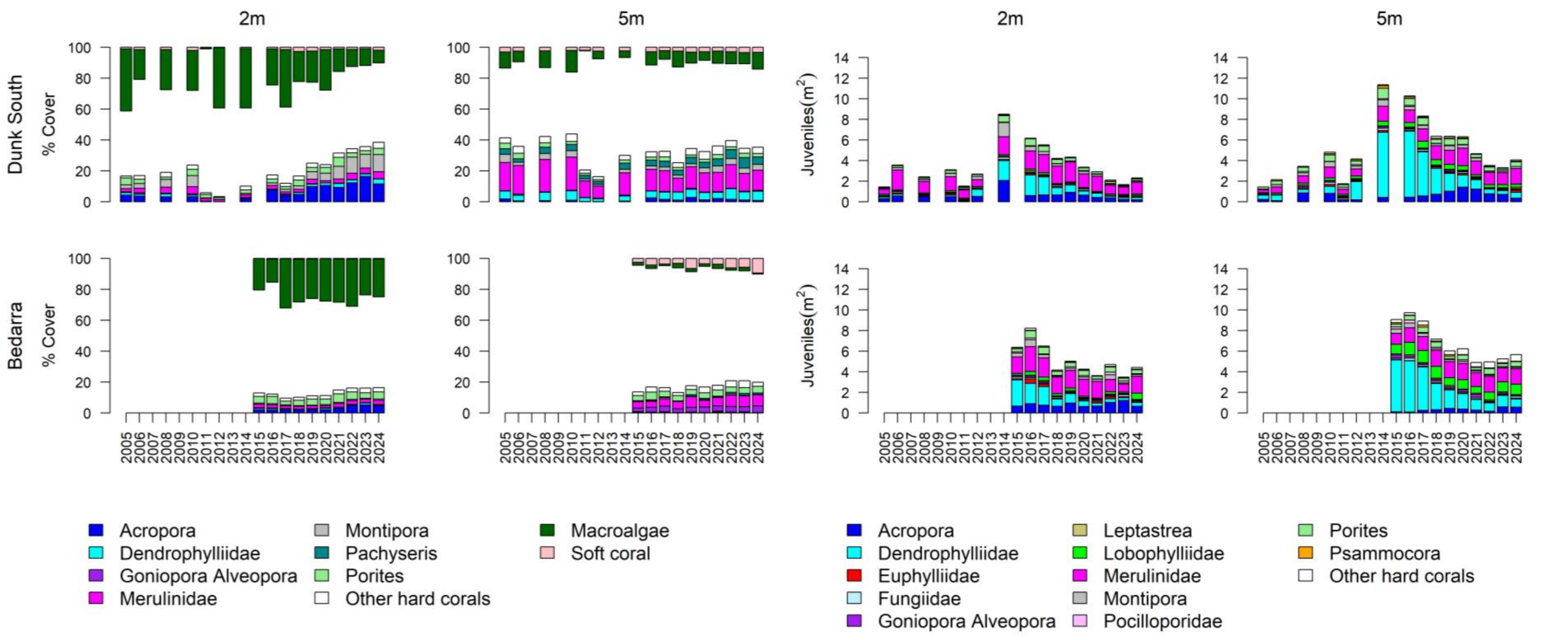

Figure A3 continued

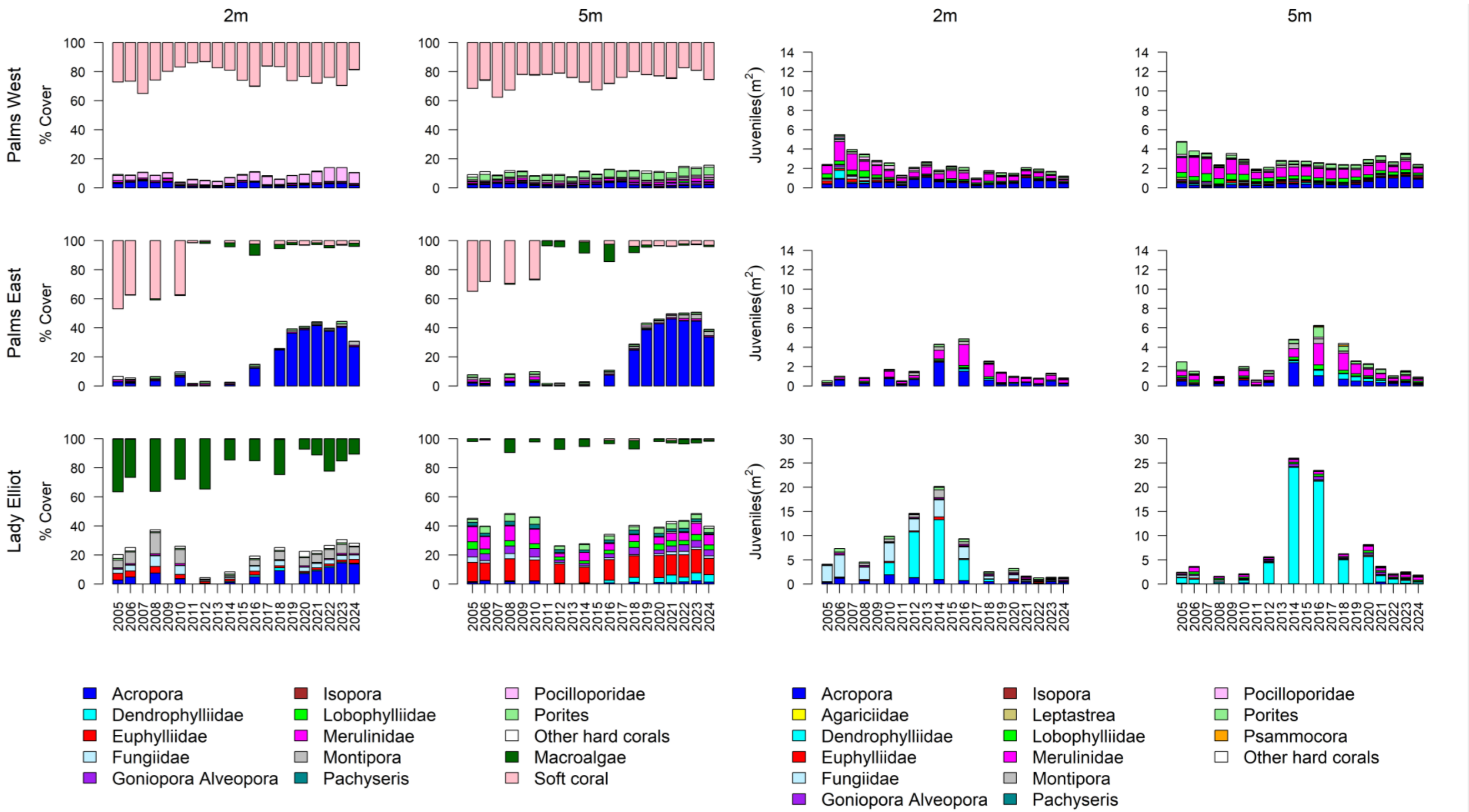

Figure A4 Burdekin region benthic community composition. Cover estimates are separated into regionally abundant hard coral groups and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral groups. Separate legends with relevant groupings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

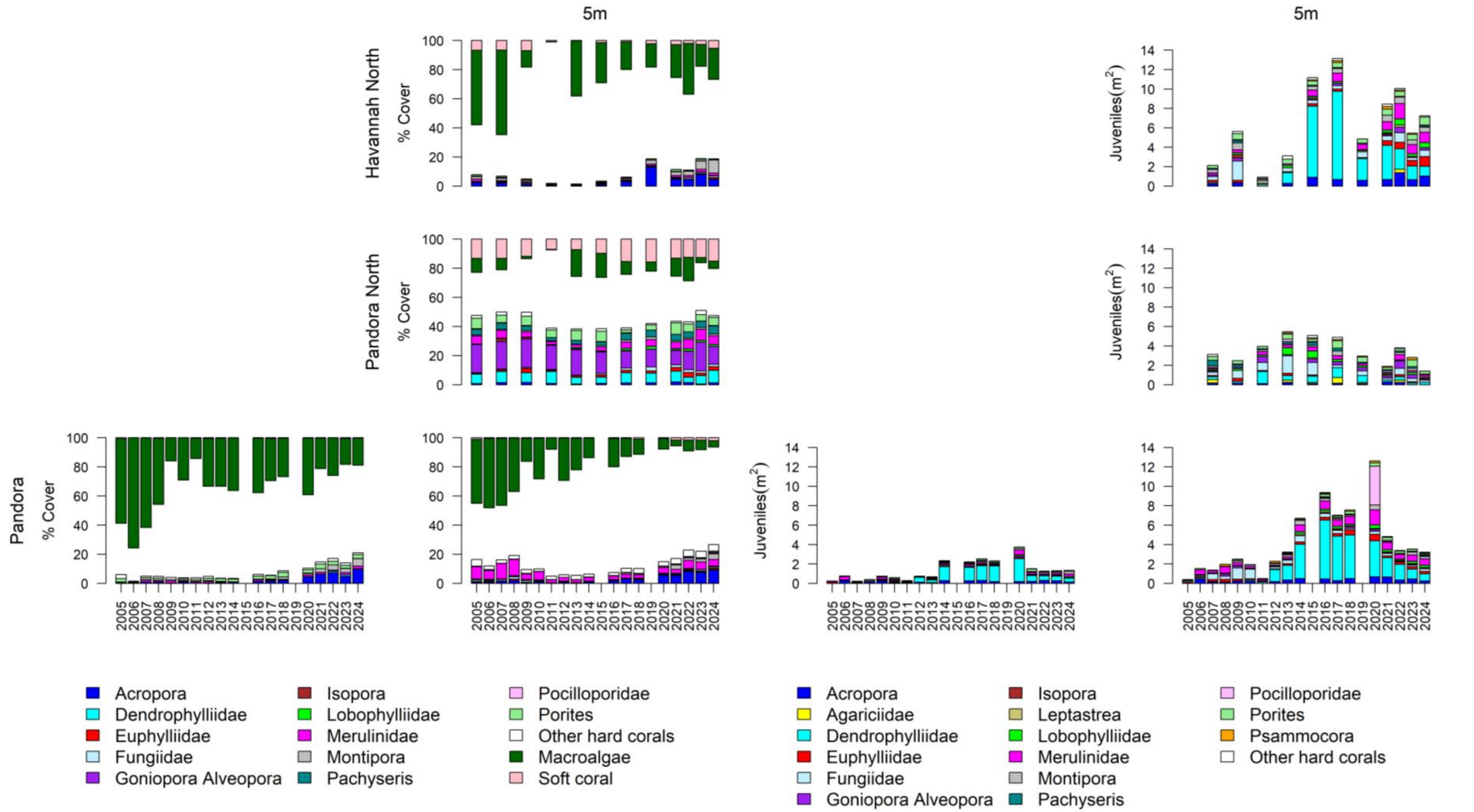

Figure A4 continued

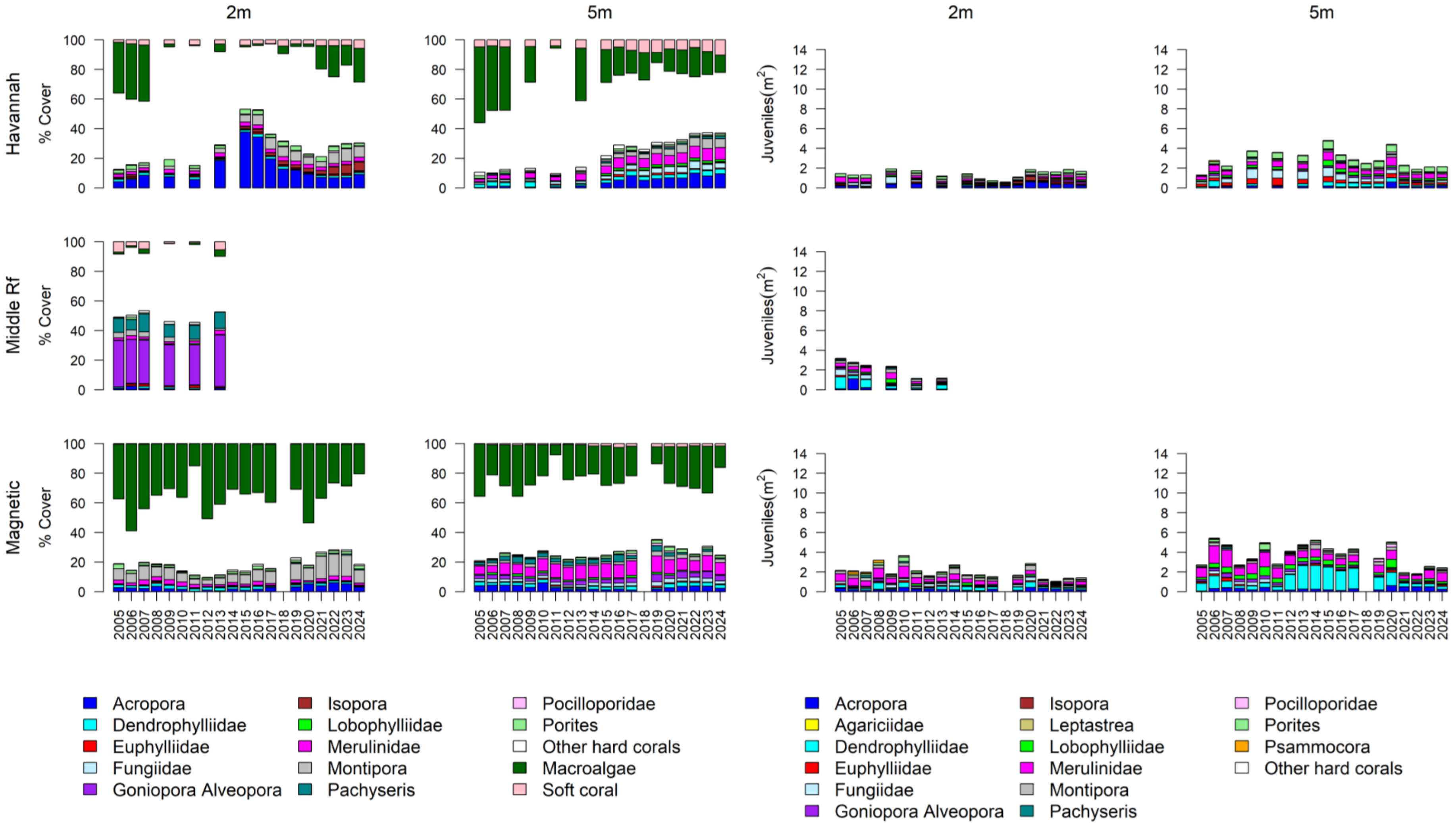

Figure A4 continued

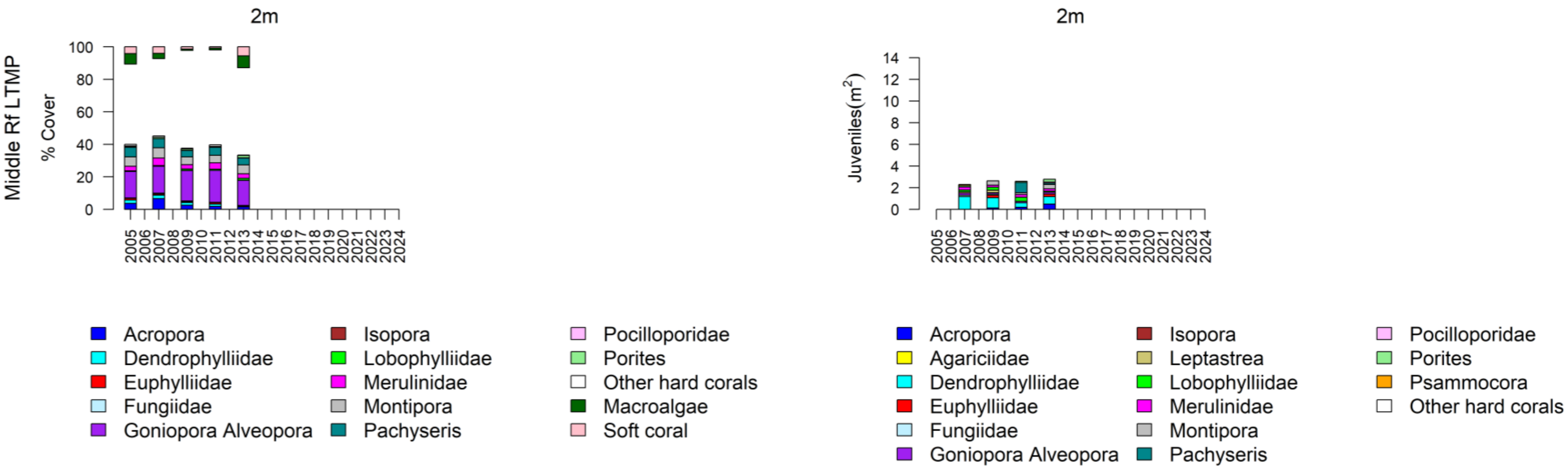

Figure A4 continued

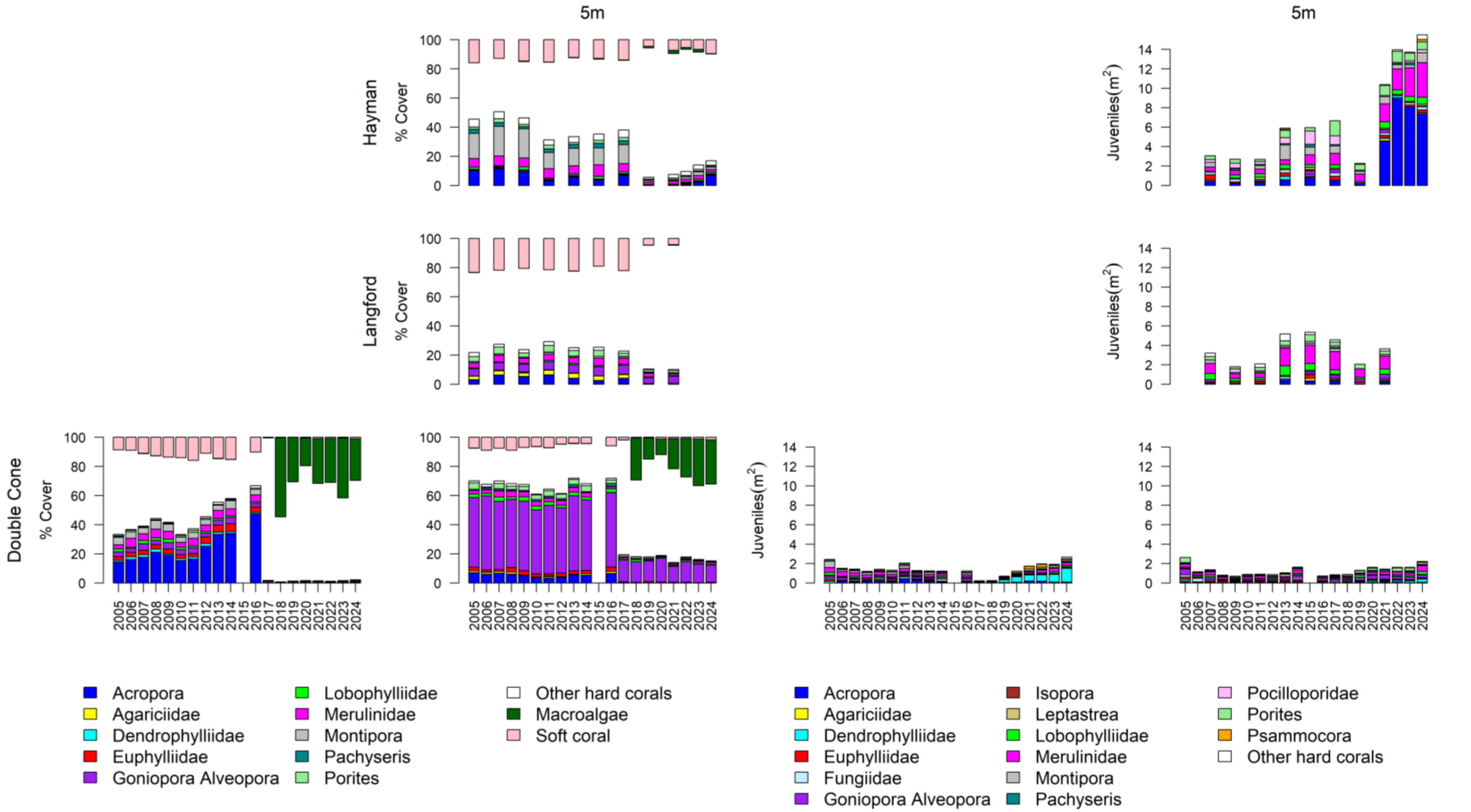

Figure A5 Mackay–Whitsunday region benthic community composition. Cover estimates are separated into regionally abundant hard coral groups and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral groups. Separate legends with relevant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

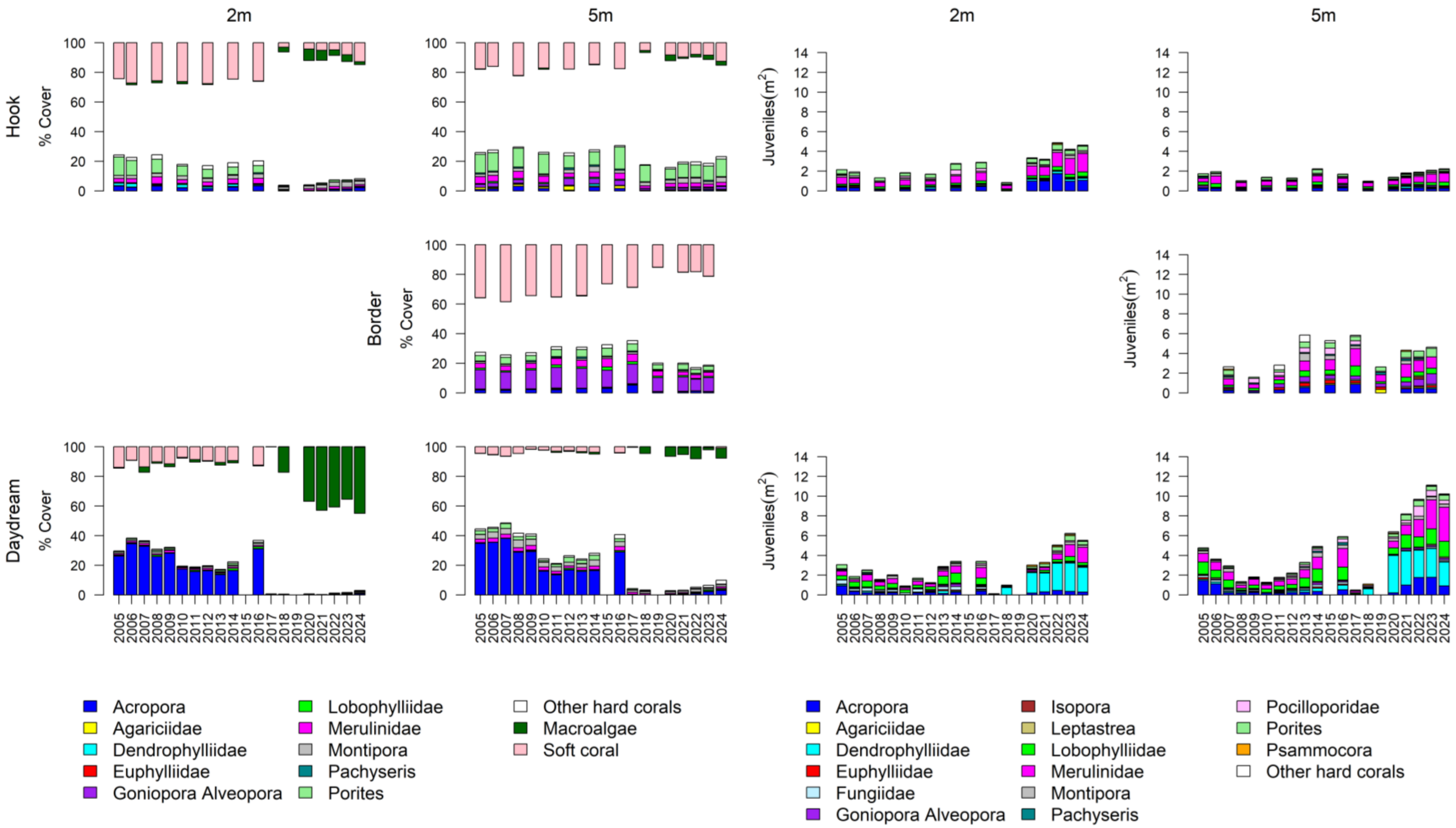

Figure A5 continued

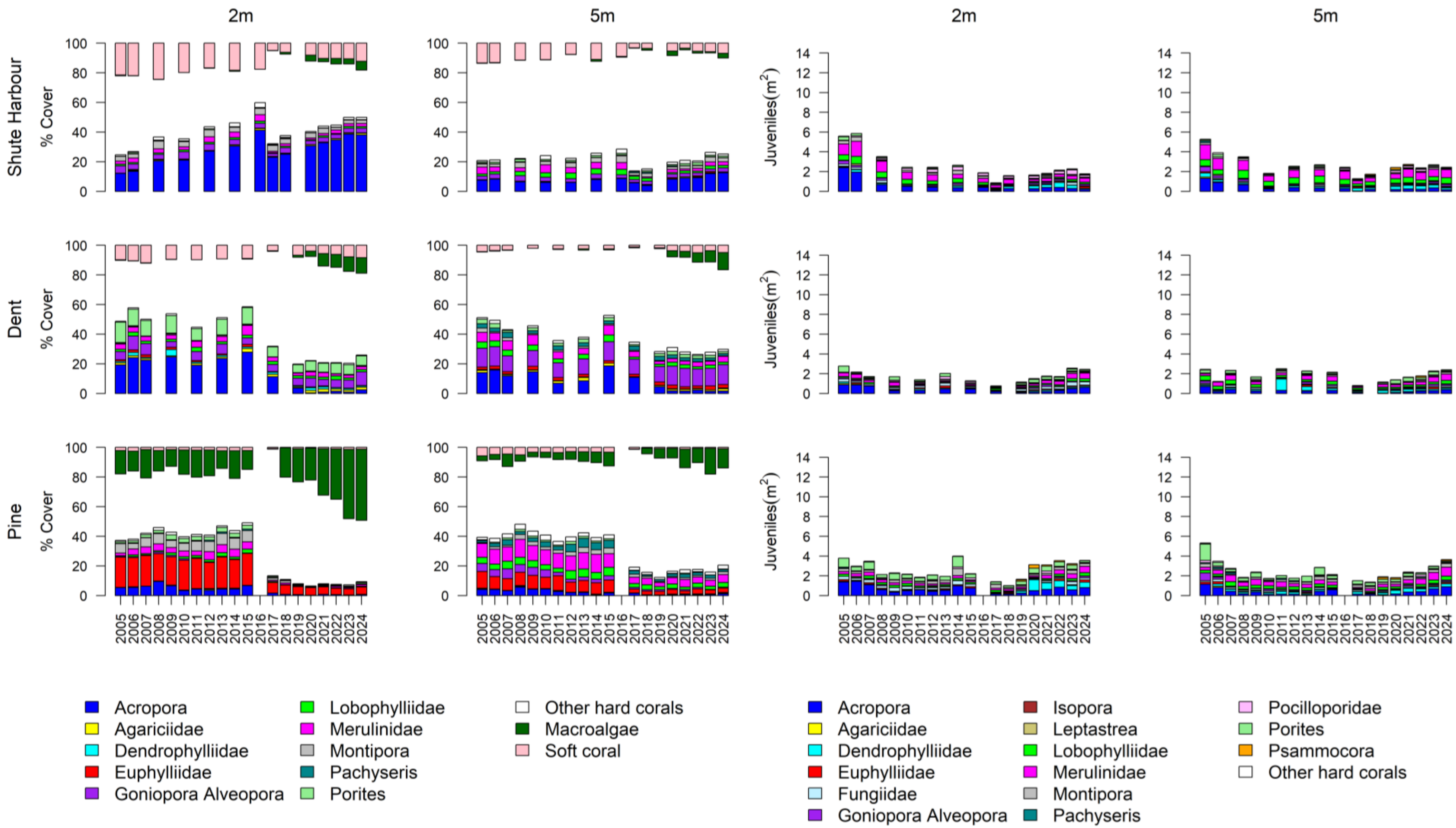

Figure A5 continued

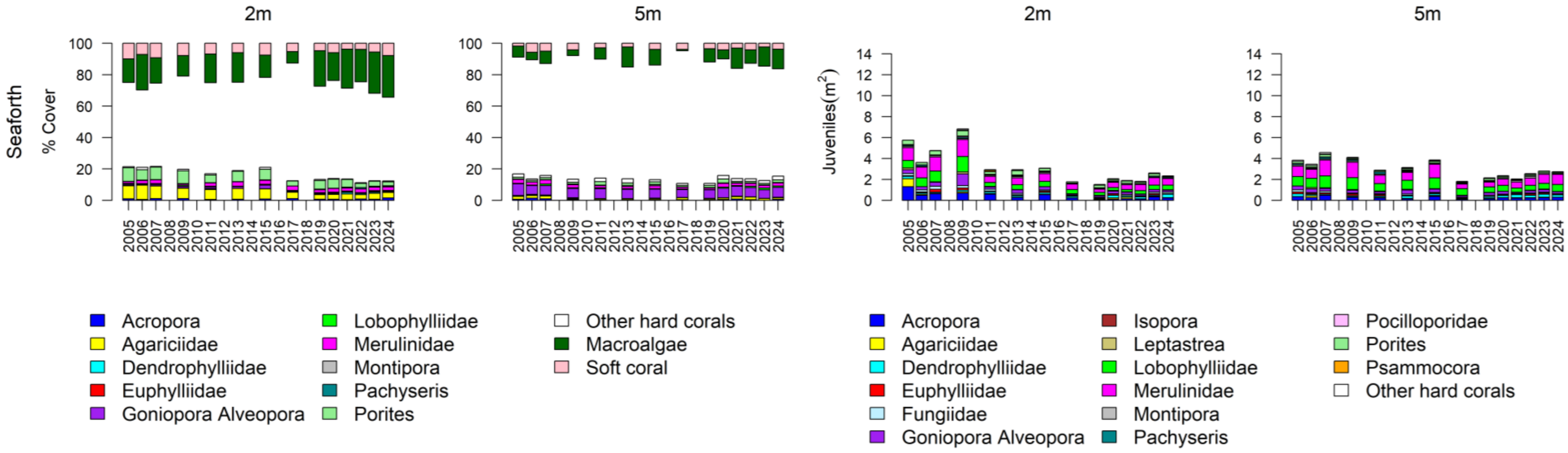

Figure A5 continued

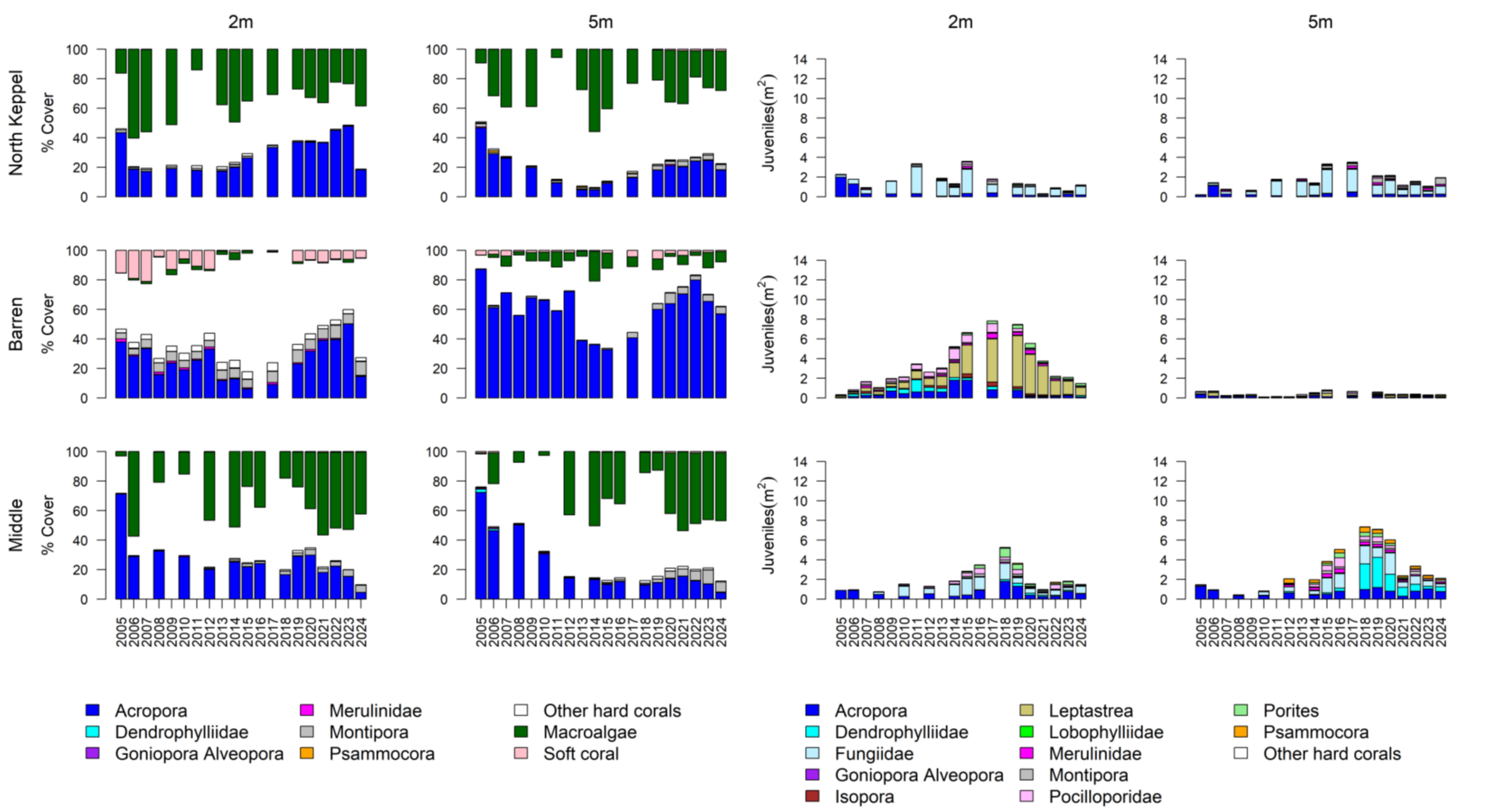

Figure A6 Fitzroy region benthic community composition. Cover estimates are separated into regionally abundant hard coral groups and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral groups. Separate legends with relevant groupings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

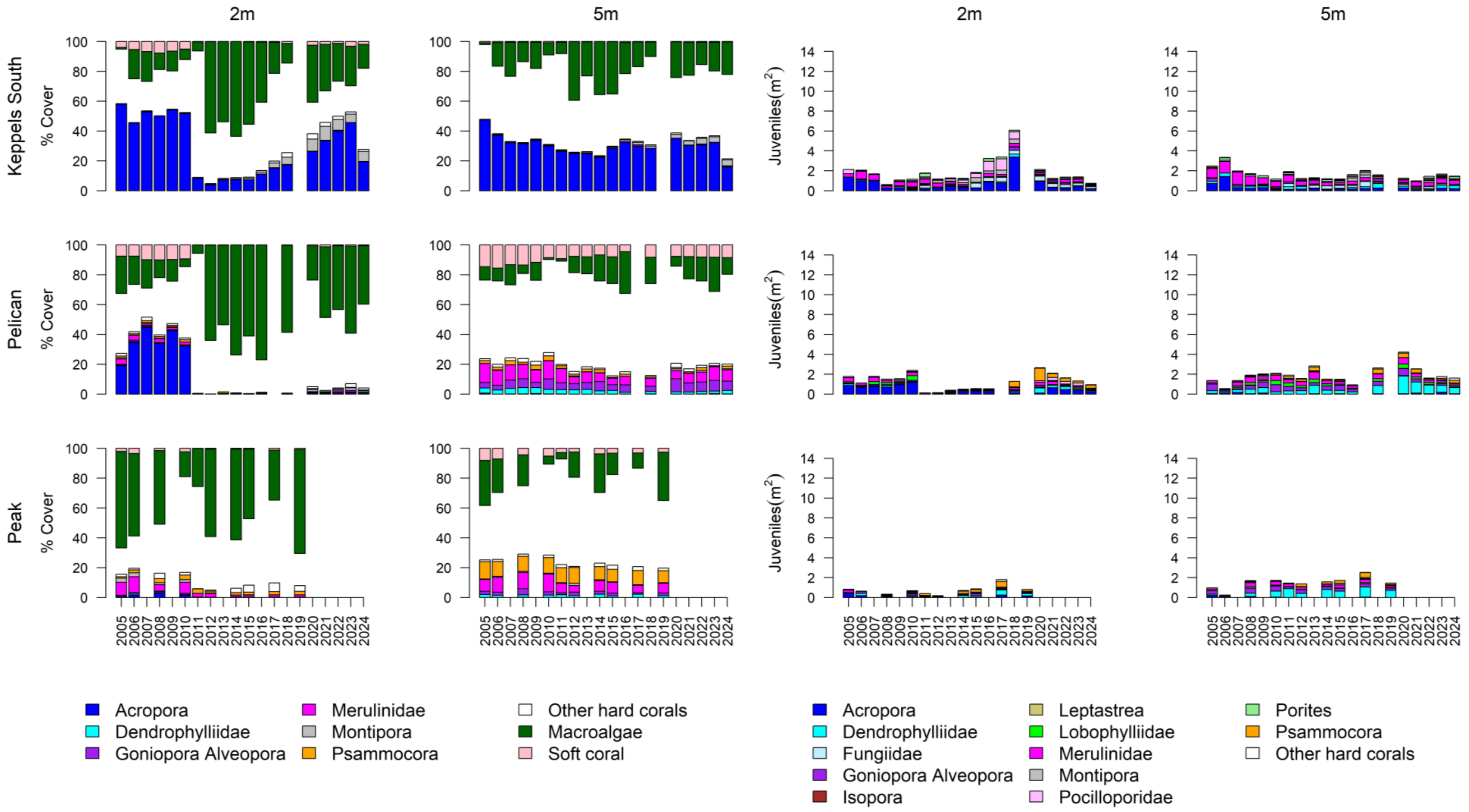

Figure A6 continued

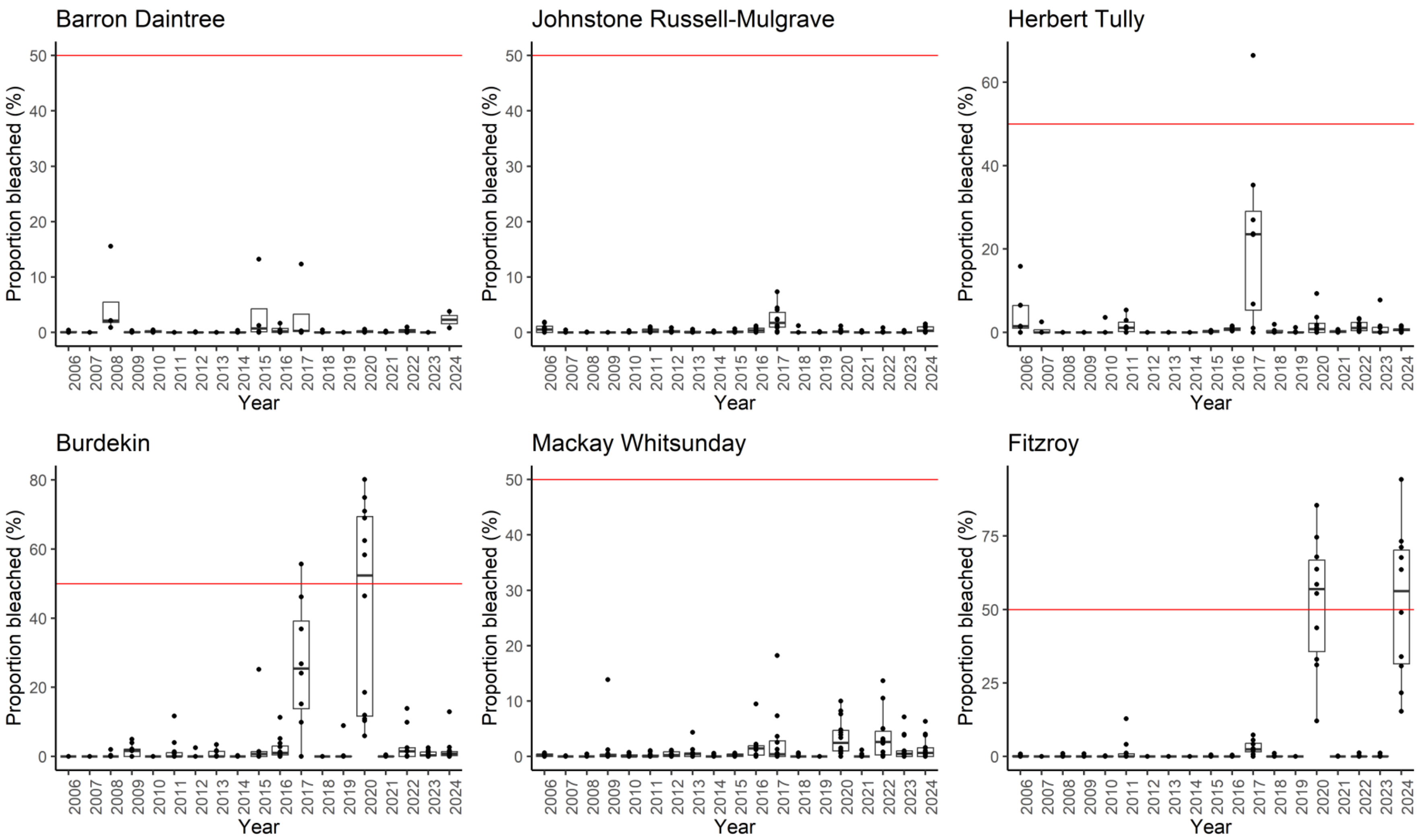

Figure A7 Proportion of hard coral bleached in each sub-region at the time of surveys. Boxplots include the proportion hard coral points from photo transects categorised as being “bleached” or “partially bleached” for each reef, depth within each sub-region ad year.

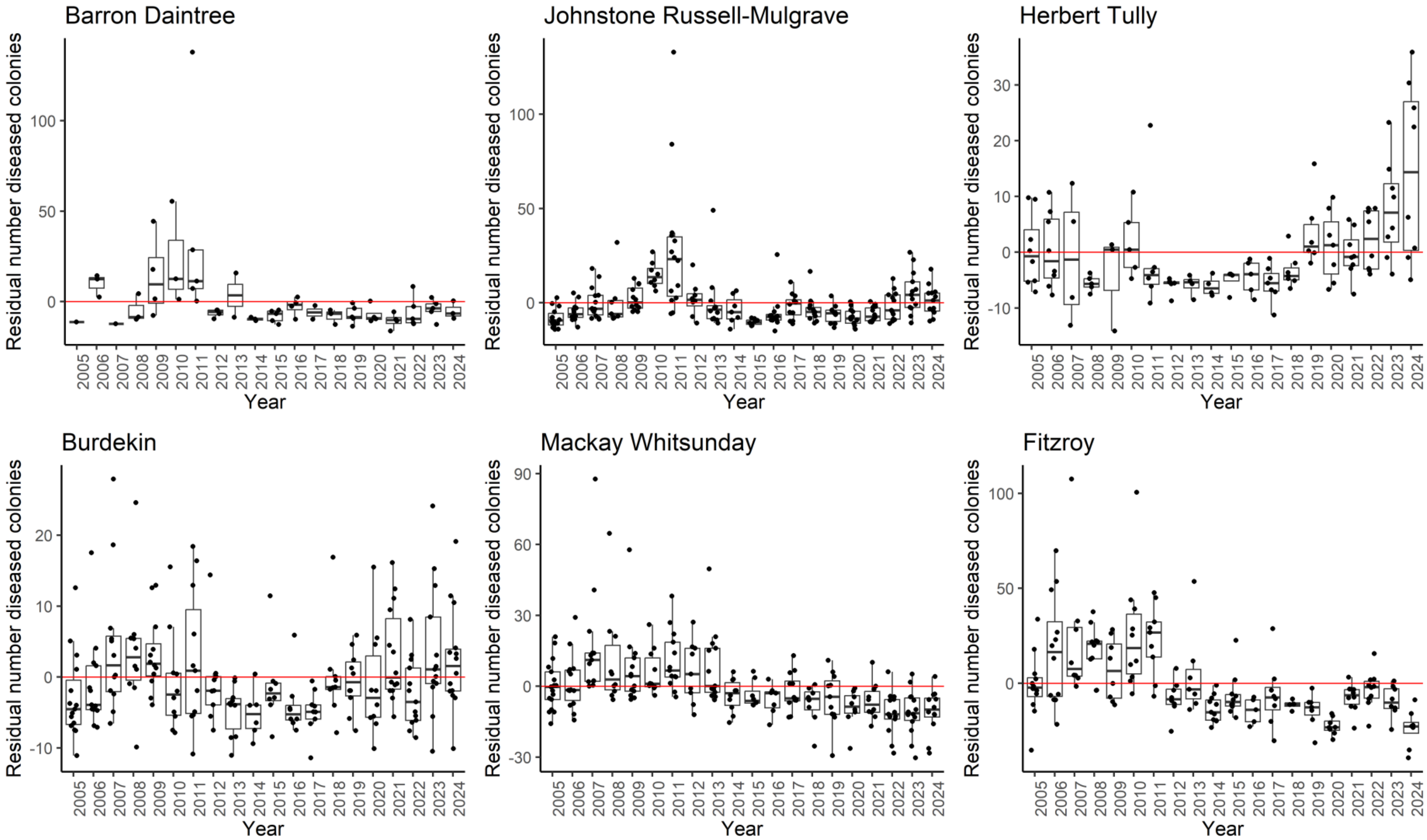

Figure A8 Coral disease by year in each region. Boxplots include the number of coral colonies suffering ongoing mortality attributed to either disease, sedimentation or ‘unknown causes’ for each reef, depth and year. Data are standardised to the reef and depth mean across years (see section 2.3.3).

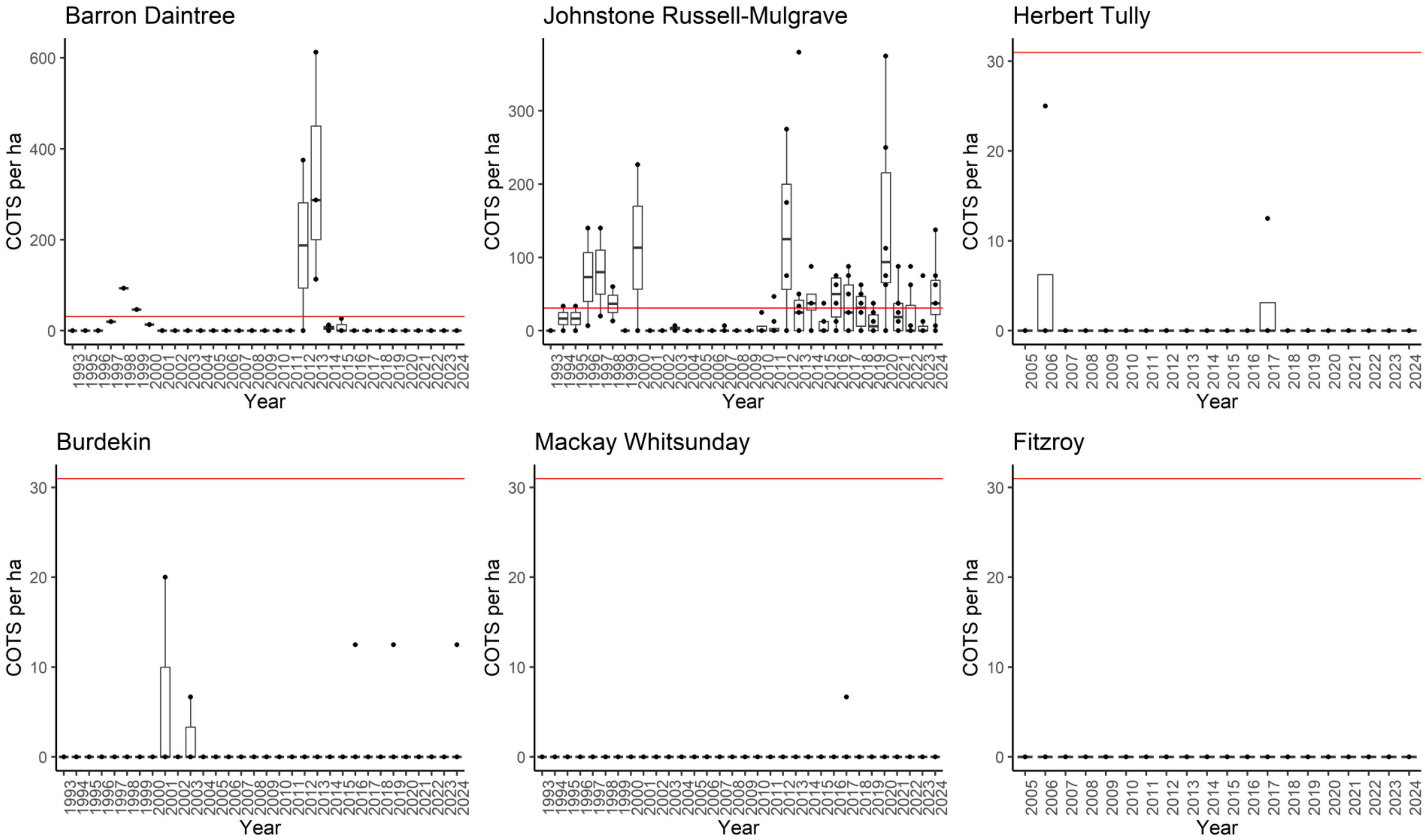

Figure A9 Crown -of-thorn-starfish mean density (individuals/ha) by year in each region. Red line indicates outbreak densities of 31 individuals per hectare (see section 2.3.3 for derivation).

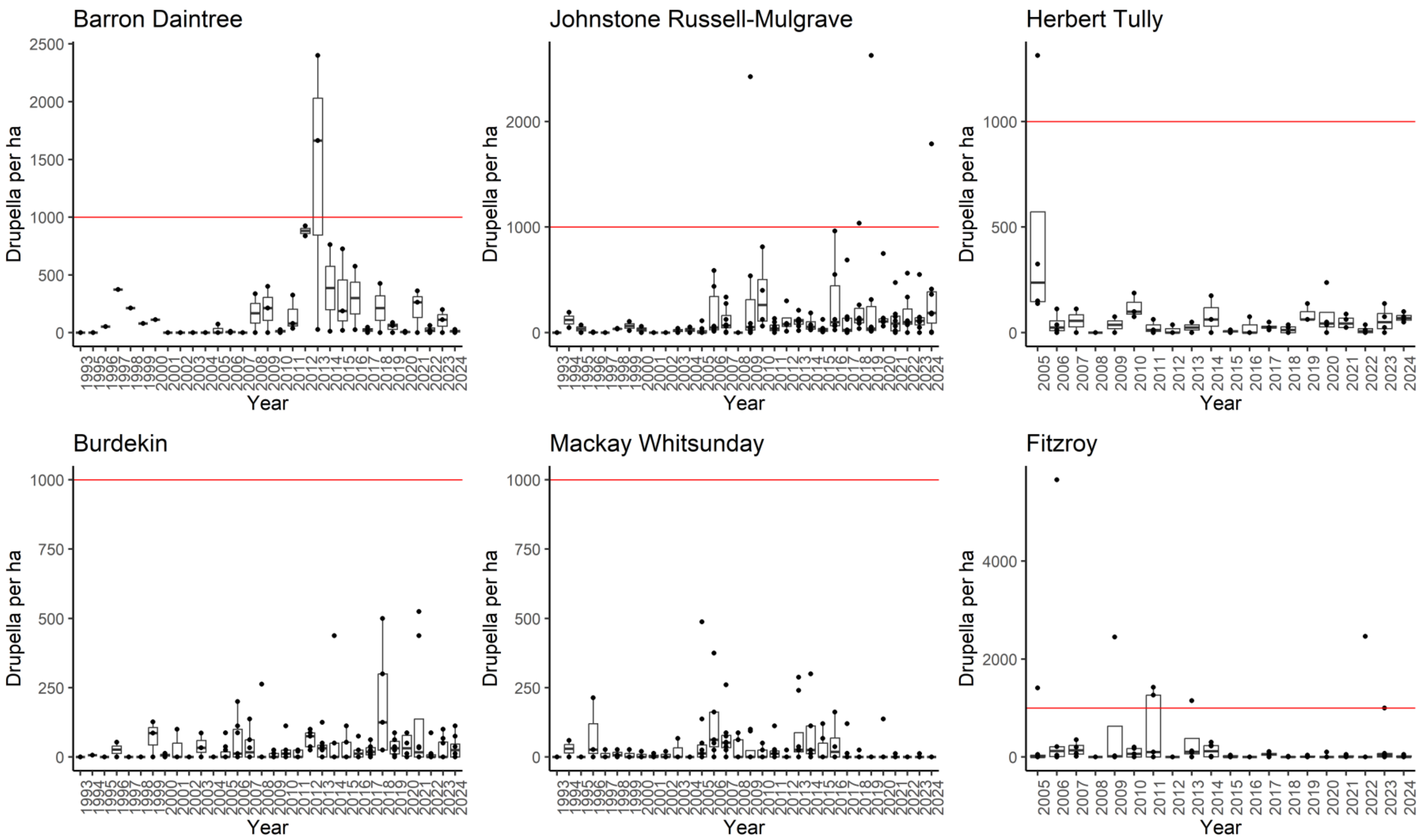

Figure A10 Mean density of *Drupella* by year in each (sub-)region. Red line indicates densities of *Drupella* which have detrimental impact on coral communities (see section 2.3.3 for derivation).

Table A9 Percent cover of hard coral genera 2024. Genera for which cover did not exceed 1% on at least one reef-depth or were unidentified to genus level are grouped as “Other”.

	Reef	Depth	*Acropora*	*Cyphastrea*	*Diploastrea*	*Dipsastraea*	*Echinopora*	*Favites*	*Galaxea*	*Goniastrea*	*Goniopora*	*Lobophyllia*	*Merulina*	*Montipora*	*Mycedium*	*Oxypora*	*Pachyseris*	*Pavona*	*Pocillopora*	*Porites*	*Turbinaria*	Rare genera
Barron Daintree	Low Isles	5	2.8	0	0.8	0.5	2	0.1	3	0.2	0.8	1.4	0.6	0.6	0.1	0.6	1.6	0.2	0.5	18.1	0.2	3.9
	Snapper North	2	0.4	0	0	0	1.3	0	0.3	0	0	0.1	0	1	0	0	0	0.1	0	1.4	0	0.4
		5	5.4	0	0	0.1	0.1	0.1	0.9	0	9	0.2	0.1	4.1	0.2	0	6.5	1.1	0	5.6	0.1	2.7
	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	8.7	0.1	0	0.3	0	0.2	0.1	0.5	0.1	0.6	0	5.8	0	0	0	0.1	4.8	8.3	0	0.8
		5	9.7	0.1	1	1.2	2.4	0.4	1.4	0.4	0.1	1	0.1	1.3	0	0.4	0	0.2	5.2	14.8	0	2.5
	Fitzroy West	2	9.4	0.1	0.4	0.4	0.6	0.1	0.3	0.1	0.2	1	0.1	4	0	0	0.1	0	1.1	3.9	0.1	0.7
		5	7.5	0	1.4	0.1	0.6	0	0.6	0.2	0.9	1.9	0	6.6	0.1	0.1	0.8	0.1	0.6	12.3	0.1	2.4
	Fitzroy West LTMP	5	4.6	0	0.3	0.7	0.1	0	0.7	0.2	0.4	1.7	0.3	3.4	0.4	0.4	2.8	0.2	0.8	17.8	0.1	3.7
	Franklands East	2	20.7	0	0	0.1	1.3	0.1	0.1	0.1	0.3	0.2	0.3	17.4	0	0	0	0	0.6	1.3	0	1.4
		5	10.6	0.2	0	0.2	0.2	0	0.8	0.1	0	0.3	0.2	3.9	0	0	0	0	0.2	2.8	0	1.3
	Franklands West	2	3.8	0	0	0	0.9	0	0.3	0	0.9	0	0	0.5	0	0.1	5.7	0	0.8	31.9	0	0.8
		5	0.1	0	0	0	0.3	0	0.1	0	0.1	0	0	0	0	0	2	0	0	59	0	0.3
	High East	2	4.1	0	0	0.1	0.9	0.2	0.1	0.4	0.3	0.6	0	3.1	0	0	0	0.3	0.4	4.8	0.1	0.8
		5	7.4	0	0	0.1	1.7	0.3	0.1	0.1	0.8	0.1	0.1	7.4	0	0	0.1	0	0.3	14	0	1.4
	High West	2	1.4	0	0.1	0.2	0.3	0.3	0.4	0.1	2.5	0.2	0	0.6	0	0	0	0.6	0.4	41.3	0	2.1
		5	0.1	0	0	0	0.2	0.2	0.2	0.1	4.3	0	0	0.2	0	0	0	0.4	0.1	14.8	0	1.8
Tully Herbert	Barnards	2	34.4	0.1	0	0.1	0	0.2	0.2	0	0	0	0	14.4	0	0	0	0	0.5	0.2	0.4	0.5
		5	16.8	0.1	0	0.4	0.8	0.1	0.2	0.1	0.3	0	0.2	26.9	0.1	0.4	0.1	0	0.4	0.4	0.8	1.1
	Dunk North	2	31.2	0.7	0	0.3	0.5	0.2	0.3	0	0.2	0	0	6.8	0	0	0	0.1	1.2	0.4	4.8	1.2
		5	6	0.4	0	0.6	0.1	1	0.4	0	0.2	0.1	0	8.2	0.1	0.1	0.1	0	0.4	0.6	9.6	3.4
	Dunk South	2	11.6	3	0	0.2	0	0.9	1.1	0	0.1	0.3	0	11	0	0.1	0	1.5	0.1	4.1	3.2	1.4
		5	0.9	0.9	0	2.3	0.1	1	0.1	1.6	0.6	0.3	2.6	3.8	2.3	1.2	4.8	0.5	0.2	2.3	5.9	4
	Bedarra	2	4.6	0.8	0	0.8	0	0.3	0.3	0.1	0.2	0.2	0	0.9	0	0	0.1	0.6	0.5	4.5	0.5	2.2
		5	0.1	0.2	0	4.5	0	0.6	0.1	0.3	4.3	1	0.3	0.2	0.7	0.4	0.9	0	0.1	4.3	0.1	1.7

	Reef	Depth	*Acropora*	*Cyphastrea*	*Diploastrea*	*Dipsastraea*	*Echinopora*	*Favites*	*Galaxea*	*Goniopora*	*Hydnophora*	*Isopora*	*Lobophyllia*	*Merulina*	*Montipora*	*Mycedium*	*Oxypora*	*Pachyseris*	*Pavona*	*Pectinia*	*Platygyra*	*Pocillopora*	*Podabacia*	*Porites*	*Seriatopora*	*Turbinaria*	Rare genera
Burdekin	Palms East	2	26.9	0.1	0	0.2	0	0.2	0	0	0.1	0	0.1	0	2.6	0	0	0	0	0	0	0	0	0.2	0	0	0.5
		5	33.4	0.1	0	0.1	0.1	0	0.2	0	0	0	0.1	0	2.8	0	0	0	0	0	0.2	0.4	0	1	0	0	0.7
	Palms West	2	1.6	0	0	0	0.3	0.1	0	0.1	0	0.1	0	0	0.6	0.1	0	0.1	0	0	0.1	7.2	0	0.3	0	0	0.2
		5	1.9	0	1.2	0.1	0.1	0.3	0	1.1	0.1	0	0.2	0	1.4	0.1	0	0.2	0.1	0.1	0.1	1.5	0	5.4	0	0	1.4
	Havannah North	5	4.6	0.3	0	0.2	0	0.1	1	0.1	0.2	0	0.1	0.3	9	0.1	0.1	0	0	0.1	0.1	0	0	0.5	0	0.2	1.5
	Havannah	2	9	0.1	0	0.3	0.5	0	0.7	0.4	0.8	6	0.3	0.2	7.3	0	0	0	0.3	0.1	0.8	0.4	0	1.6	0	0.4	1.1
		5	9.4	0.1	0	0.7	1.2	0.1	0.8	0.4	0.2	0.4	0.4	3.9	6	0.2	1	1.8	0.5	1.3	0.1	0	0.4	1.1	0	3.2	3.9
	Pandora	2	9.7	1.1	0	0.1	0	0.6	0.1	0.1	0	0	0	0	5.2	0	0	0	0	0	0	0.1	0	2.6	0	0.2	1.2
		5	9.1	0.6	3.6	1.3	0	0.6	0.9	0.2	0.5	0	0.1	0.3	3.9	0	0.2	0.5	0	0	0.4	0.6	0.3	0.8	0	0.4	2.3
	Pandora North	5	1.3	0	0	0.3	2.4	0.1	2.2	12.2	0	0	0.3	1	1.2	1.2	0.6	5.6	0.1	0.6	0.5	0.2	0.4	5.5	0	8.5	3.4
	Lady Elliot	2	13.7	0	0	0	0.3	0.1	2.6	0.1	0.3	0	0.1	0	4.6	0.1	0	0	1.8	0	0.1	0	0	0.6	0	0.6	3.4
		5	1.4	0.1	0	1.1	0.1	1.1	11	3.5	1.1	0	1.6	0.2	0.6	1.6	1.6	1	0	1.4	0.4	0	1.6	2.9	0	4.7	3.3
	Magnetic	2	3.2	0.5	0	0.2	0	0.5	0.2	0.2	0	0	0	0	8.8	0	0	0.8	0	0	0.1	0	0	2.1	0	0.6	1.3
		5	2.3	0.7	0	1.5	0.1	0.9	0.6	3.8	0.1	0	0.2	2.7	2.3	0	0.1	0.7	0	0.3	1.8	0	1.8	1.6	0	2.3	0.9
Mackay Whitsunday	Hayman	5	6.8	0.2	2.1	0.4	0.3	0.2	0.5	0.3	0.2	0.1	0.5	0	1.7	0.1	0.3	0.5	0.1	0	0.2	0.3	0	0.7	0.4	0	1.2
	Hook	2	2.2	0	0.1	0.2	0.1	0.4	0	0.1	0	0	0.1	0	2.4	0.1	0.1	0.1	0.2	0	0.1	0.1	0	0.6	0	0.1	1.4
		5	0.1	0	0.4	0.8	0.2	0.5	0	1.9	0	0	0.2	0	3.5	0.1	0	0.4	0.6	0.1	0	0.1	0	11.8	0	0.4	1.9
	Double Cone	2	0.5	0.2	0	0	0.4	0	0	0	0	0	0.2	0.1	0.3	0	0	0	0	0	0	0.2	0	0.1	0	0.1	0.2
		5	0	0.1	0	0.1	0.2	0.1	0.6	11.6	0	0	0.6	0.1	0	0	0.2	0.3	0	0.1	0.1	0	0	0.5	0	0	0.6
	Daydream	2	0.6	0.1	0	0	0	0	0	0	0	0	0	0	0.7	0	0	0	0	0.1	0	0	0	0.4	0.4	0.2	0.6
		5	3	0.3	0	0.1	0	0.4	0	0.1	0.1	0	0.1	0	1.4	0.1	0.2	0.1	0	0.1	0	0.2	0	0.2	2.1	0.4	1
	Dent	2	2.4	0	0	0.2	0.4	0.1	0.4	9.2	0.1	0	1.5	0.1	0.4	0	0.1	0.2	1.8	0.6	0.1	0.1	0.1	6.5	0	0.8	0.9
		5	1.6	0	0	0.5	0.5	0.1	2.2	13.2	0.2	0.1	0.3	0.8	0.2	0	1.1	1.5	0.9	1.2	0.6	0.1	0.1	1.4	0	0.6	2.5
	Shute Harbour	2	38.1	0.1	0	0.1	0.3	0.1	0.2	3.2	0	0.2	0.9	0.2	2.3	0.1	0.1	0	1.1	0.5	0	0.3	0.1	0.2	0.2	0	1.6
		5	12.6	0.2	0	0.2	0.2	0.1	0.1	2.8	0.2	0.1	0.6	0.1	2.7	0.3	0.6	0.1	0.1	0.6	0.4	0.2	0.1	1.1	0	0.1	1.6
	Pine	2	0.7	0	0	0.1	0	0	5	0.4	0.3	0	0.1	0	0.6	0.1	0.1	0	0	0.6	0	0.1	0	0.8	0	0.1	0.4
		5	1.3	0.1	0	0.3	0.2	0.1	3	0.5	0.3	0	1.9	0.3	1.9	1	1	1.4	0	2.6	0.2	0.1	1.1	0.4	0	0.5	2.3
	Seaforth	2	1.4	0	0	0.6	0.4	0.8	0	0.2	0.1	0	0.3	0	0.2	0	0	0.5	3.5	0.1	0	0	0.2	2.6	0	0.1	1.5
		5	0.6	0	0.9	0.8	0	0.2	0	6.1	0	0	0.6	0	0.2	0.1	0.1	0.2	1.1	0	0.3	0.1	0.1	1.6	0	0.1	2.6

	Reef	Depth	*Acropora*	*Alveopora*	*Favites*	*Goniopora*	*Isopora*	*Montipora*	*Paragoniastrea*	*Pocillopora*	*Psammocora*	*Turbinaria*	*Rare genera*
Fitzroy	Barren	2	14.5	0.3	0.1	0	1.1	9.3	0	0.2	0.3	0	1.6
	Barren	5	56.9	0	0	0	0	4.8	0	0.3	0.1	0	0.1
	North Keppel	2	18.2	0	0	0	0	0.2	0	0.2	0	0	0.1
	North Keppel	5	18	0	0	0	0	3.5	0.1	0	0.4	0	0.5
	Middle	2	4.5	0	0	0	0	4.9	0	0.1	0	0	0.3
	Middle	5	4.4	0	0	0	0	6.7	0	0.5	0	0.1	0.5
	Keppels South	2	19.5	0	0	0	0	6.9	0	1	0	0	0.4
	Keppels South	5	15.9	0.1	0	0	0	4	0.1	0.4	0	0.1	0.9
	Pelican	2	0.6	0.1	0.1	0.1	0	0.9	0	0.4	0.5	0.1	1.2
	Pelican	5	0.1	4.7	3.2	1.1	0	0.8	2.2	0	1.8	2.4	3.9

Table A10 Percent cover of soft coral families 2024. 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'.

	Reef	Depth	*Briareum*	*Clavularia*	*Erythropodium*	*Heliopora*	*Lobophytum*	*Sarcophyton*	*Sclerophytum*	*Xenia*	Other SC
Barron Daintree	Low Isles	5	9.6	0	0	0.1	0.5	0.4	0.4	0	0.6
	Snapper North	2	3.8	4.6	0	0	0.2	0	0	0	0
		5	1.5	0.1	0	0	0	0.1	0	0	2.9
	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	1.3	1.5	0	0.1	0.6	0.2	1.1	0	0.8
		5	6.7	0.5	0	0	0.1	1.3	6.6	0	0.4
	Fitzroy West	2	0.5	0	0	0	12.5	2.5	10.6	0	0.1
		5	0.1	0	0	0	5.4	3.9	13.4	0	0.4
	Fitzroy West LTMP	5	0.7	0	0	0	1.1	3.4	14	0	0.8
	Franklands East	2	0	0.6	0	0.3	0.1	0	0.5	0	0.1
		5	0.9	0.6	0	0	0.2	0.2	1.8	0	0.1
	Franklands West	2	0	8.6	0	0.2	6.3	0.3	1.6	0	0
		5	0	2.3	0	0.1	0.6	0.1	0	0	0
	High East	2	6	0	0	0	0.8	0	3.6	0	0.3
		5	7.2	0	0	0	0.1	0	0.1	0	0
	High West	2	0	0	0	3	1.6	0	1.1	0	0
		5	0.7	0	0	1.2	0	0.1	0.5	0	0
Herbert Tully	Barnards	2	2.4	0	0	0	0.1	0.1	0.6	0	0.3
		5	5.8	0	0	0	0	0	0	0	1.4
	Dunk North	2	0.2	0.3	0	0	0	0	0.4	0	0.2
		5	0.2	0	0	0	0	0.1	0.4	1.3	3.4
	Dunk South	2	0.9	0.4	0	0	0.1	0	0.1	0	0.3
		5	2.6	0	0	0	0	0	0.4	0	0.3
	Bedarra	2	0	0	0	0	0	0.1	0	0	0.2
		5	5.7	0	1.9	0	0	0.5	0.3	0.2	0.9

	Reef	Depth	*Briareum*	*Clavularia*	*Erythropodium*	*Klyxum*	*Lobophytum*	*Nephthea*	*Sarcophyton*	*Scleronephthya*	*Sclerophytum*	Other SC
Burdekin	Palms East	2	0	0	0	0	0.4	0	0.1	0	1.3	0.1
		5	0	0.1	0	0.1	0.4	0	0.2	0	2.1	0.2
	Palms West	2	0.2	1.4	0	0.1	0.8	2.7	1.1	0	12.2	0
		5	3.9	1.3	0	0	0.9	1.3	3.8	1.7	12.2	0.3
	Havannah North	5	1.3	3.3	0	0.1	0	0	0	0	0.1	0.4
	Havannah	2	5	0	0	0	0.4	0.1	0.1	0	0.2	0
		5	10.1	0	0	0	0	0	0.1	0	0.1	0.1
	Pandora	2	0	0	0	0	0	0	0.2	0	0	0.1
		5	0	0.4	0	1.1	0.1	0	0.5	0	0	0.1
	Pandora North	5	8.6	6.2	0	0	0	0	0.3	0	0.1	0
	Lady Elliot	2	0	0	0	0	0	0	0	0	0	0.1
		5	0	0	0	0	0	0	0.2	0	0	0.4
	Magnetic	2	0	0	0	0	0	0	0.2	0	0.2	0
		5	0.1	0	0	0	0.4	0	0.4	0	0.3	0.5
Mackay Whitsunday	Hayman	5	0.4	0	0	1.6	1.4	0	1.2	0	3.6	1.3
	Hook	2	0.8	0	0	5.7	0.9	0	0.1	0	5.2	0.2
		5	2.4	0	0	2.2	0.4	0	1.8	0	5.6	0
	Double Cone	2	0.1	0	0	0.9	0	0	0	0	0	0.1
		5	0.6	0	0	0.3	0.4	0.1	0.1	0	0.3	0
	Daydream	2	0	0	0	0.1	0	0	0.1	0	0.1	0.1
		5	0	0	0	0.1	0	0	0.4	0	0.1	0.4
	Dent	2	5.4	0	0	1	0.2	0.1	0.9	0	0.8	0.1
		5	1.1	0	0	1.1	0.5	0	0.8	0	1.4	0
	Shute Harbour	2	0.9	0	0.1	2.2	0.2	0.4	3.2	0	5.1	0.2
		5	0.1	0	0	0.6	0.2	0.2	2.8	0	2.6	0.4
	Pine	2	0.3	0	0	0.1	0	0	0.3	0	0.4	0
		5	0.3	0	0	0.2	0	0	0.1	0	0.2	0.2
	Seaforth	2	1.9	0.1	0.3	0.7	0	0.1	3.8	0	0.9	0.1
		5	0	0	2.8	0	0.1	0	0.2	0	0.4	0.2

	Reef	Depth	*Cladiella*	*Klyxum*	*Rumphella*	*Sarcophyton*	*Sclerophytum*	*Xenia*	Other SC
Fitzroy	Barren	2	1.8	2	0	0	0.1	1	0.1
		5	0	0.6	0	0	0	0.2	0
	North Keppel	2	0	0	0	0	0	0	0
		5	0	0.6	0	0.6	0	0	0
	Middle	2	0.1	0	0	0	0	0.3	0
		5	0.3	0	0	0.2	0	0.2	0
	Keppels South	2	0.5	0.2	0	0	0	1.2	0
		5	0	0	0	0.1	0.2	0.1	0
	Pelican	2	0	0	0	0	0	0	0.6
		5	0	0.1	1.5	2.8	3	0	1.2

Table A11 Percent cover of macroalgae groups 2024. Genera for which cover exceeded 0.5% on at least one reef are included, rare or unidentified genera are grouped to 'Undefined'.

(sub-)region	Reef	Depth	Rhodophyta (red algae)					Chlorophyta (green algae)			Phaeophyta (brown algae)							Undefined	Turf Algae
			Asparagopsis	Crustose coralline	*Hypnea*	*Peyssonnelia*	Undefined	*Caulerpa*	*Halimeda*	Undefined	*Dictyopteris*	*Dictyota*	*Lobophora*	*Padina*	Sargassaceae	*Stypopodium*	Undefined		
Baron–Daintree	Low Isles	5	0	2.1	0	0.07	0.47	0.1	0.03	0.13	0	0	0.1	0	0	0	0.03	0	42.55
	Snapper North	2	0	4.65	0.17	0.34	14.9	0.04	0.04	0.04	0	0.33	0	0.79	0	0	0.17	0	64.44
		5	0	3.94	0	0	0.25	0	0	0	0	0	0	0	0	0	0	0	38.62
	Snapper South	2	0	0.08	0	0	0	0	0	0	0	0	0	0	0	0	0	0	98.54
		5	0	3.19	0	1.63	5.13	0	0	0	0	0	0.12	0	0	0	0	0	87.43
Johnstone Russell–Mulgrave	Fitzroy East	2	0	0.56	0	0.13	2.26	0	0	0	0	0.13	0	0	0	0	0	0	52.86
		5	0	2.44	0	0.12	2.25	0	0	0.06	0	0	0	0	0	0	0	0	30.07
	Fitzroy West	2	0	1	2.06	0.56	8.01	0	0	0.25	0	0	0	0	0	0	0	0	39.01
		5	0	2.13	0.5	0.44	2.69	0	0.06	0.19	0	0	0.06	0	0	0	0	0	27.09
	Fitzroy West LTMP	5	0	0.83	0	0.23	0.3	0	0	0.03	0	0	0	0	0	0	0	0	30.13
	Franklands East	2	0	1.38	1.13	0.31	2.13	0	0	0	0	0.12	0.06	0	0	0	0.12	0	46.38
		5	0	2.5	0.13	0.44	5.33	0	0	0.12	0	0.38	0.13	0	0	0	0.25	0	63.58
	Franklands West	2	0	1.88	0.75	0.31	12.12	0	0.31	0.06	0	1.38	0.06	0	0	0	0.06	0	19.99
		5	0	4.52	0.13	0.31	16.28	0	0.57	0	0	1.26	0.06	0	0	0	0.25	0	11.74
	High East	2	0	2.19	1.94	1	11.46	0	0	0	0	0	0.25	0	0	0	0.69	0	51.83
		5	0	3.06	0.56	0.94	6.69	0	0	0.12	0	0.06	0	0	0	0	0.06	0	39.81
	High West	2	0	4.73	0.38	0.19	4.72	0	0	0.06	0	0	0	0	0	0	0	0	31.82
		5	0	3.38	0	0.12	2	0	0	0	0	0	0	0	0	0	0.12	0	53.69

(sub-)region	Reef	Depth	Rhodophyta (red algae)					Chlorophyta (green algae)			Phaeophyta (brown algae)						Undefined	Turf Algae
			Asparagopsis	Crustose coralline	*Hypnea*	*Peyssonnelia*	Undefined	*Caulerpa*	*Halimeda*	Undefined	*Dictyopteris*	*Dictyota*	*Lobophora*	*Padina*	Sargassaceae	*Stypopodium*	Undefined		
Herbert–Tully	Barnards	2	0	0.75	2.56	0.69	1.62	0.06	0	0	0	0.06	0	0	0	0	0.12	0	28.5
		5	0	1.19	0.06	0.63	0.75	0	0	0	0	0	0	0	0	0	0	0	22.96
	Dunk North	2	0	1.25	0	0.44	3.2	0	0	0	0	0.06	0.75	0	6.77	0	0.06	0	25.72
		5	0	0.94	0	0.06	1.88	0	0	0	0	0	0.38	0	0.81	0	0.06	0	28.75
	Dunk South	2	0	0.88	0	0.31	1.69	0	0	0	0	0.19	1.63	0	4.19	0	0.25	0	41.89
		5	0	1.25	0	0.75	1.25	0	0	0	0	0	8.88	0	0	0	0	0	32.83
	Bedarra	2	0	0.44	0.63	0.19	3.06	0	0	0.12	0	2.69	0.88	0	16.28	0	0.69	0	36.65
		5	0	0.13	0	0.13	0.19	0	0	0	0	0.06	0.19	0	0.06	0	0	0	32.7
Burdekin	Palms East	2	0	0.88	0	0	0.44	1.56	0	0.12	0	0	0	0	0	0	0	0	54.53
		5	0	0.62	0	0	0.44	0.44	0	0.06	0	0	0	0	0	0	0	0	47.5
	Palms West	2	0	0.12	0	0	0	0	0	0.12	0	0	0	0	0	0	0.06	0	38.5
		5	0	0.56	0	0	0.06	0	0	0.06	0	0	0	0	0	0	0	0	44.34
	Havannah North	5	1.07	1.87	0	0.33	0.37	0.03	0.03	0.07	0	4.91	8.69	0.13	3.14	0	2.61	0.03	38.39
	Havannah	2	0	0.94	0	0.13	0.69	0.12	0	0.06	0	2.76	14.81	0	0.5	2.76	0.88	0.06	34.29
		5	0	0.88	0	0.12	0	0	0	0	0	1.81	9.44	0	0.31	0.06	0.06	0	35.56
	Pandora	2	0	2	0	0.12	0.62	0	0	0	0	5	3.81	0	8.44	0	0.56	0	39.75
		5	0	1.38	0	0	0.19	0	0	0.06	0	2.07	2.07	0	0	0	0	0	50.79
	Pandora North	5	0	1.63	0	0.1	0.8	0	0	0	0	0.4	2.37	0.33	0.77	0	0.23	0	25.95
	Lady Elliot	2	0	1.57	6.03	1.19	1.75	0	0	0	0	0.06	0.75	0	0.25	0	0.38	0	23.73
		5	0	0.69	0	0.69	0.25	0.06	0	0	0	0	0	0	0	0	0.06	0	25.83
	Magnetic	2	0	1.38	0.19	0.19	1.31	0	0	0	0	4.69	4.69	0.06	8.38	0	0.5	0.06	50.38
		5	0	1.38	0	0.56	2.75	0.06	0	0	0	1.38	1.69	0	8	0	0.06	0.06	39.9

(sub-)region	Reef	Depth	Rhodophyta (red algae)					Chlorophyta (green algae)			Phaeophyta (brown algae)							Undefined	Turf Algae
			Asparagopsis	Crustose coralline	*Hypnea*	*Peyssonnelia*	Undefined	*Caulerpa*	*Halimeda*	Undefined	*Dictyopteris*	*Dictyota*	*Lobophora*	*Padina*	Sargassaceae	*Stypopodium*	Undefined		
Mackay–Whitsunday	Hayman	5	0	0.73	0	0.2	0.03	0	0	0	0	0	0.03	0	0	0	0	0	65.4
	Border	5	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	Hook	2	0	0.06	0	0	0.62	0.31	0	0.81	0	0	0.19	0	0	0	0	0	71.62
		5	0	0.88	0	0	0.56	0	0	2.12	0	0	0	0	0	0	0	0	44
	Double Cone	2	0	1	0	0.12	1.38	0	0	0.31	0	8.94	3.19	0.19	13.5	0	0.88	0	39.19
		5	0	1.19	0	0.12	4.88	0	0	0	0	14.69	2.19	0.12	7.19	0	1.19	0	41.19
	Daydream	2	0	0.81	0	0	17.45	0.06	0	0	1.32	11.77	4.13	2.06	4.69	0	3.13	0	26.29
		5	0	0.44	0	0	2.19	0	0	0.06	0	2.44	1.69	0	0	0	0.31	0	53
	Dent	2	0	2.63	0	0.31	4.38	0	0	0	0	0.06	4.88	0	0.31	0	0.56	0	47.32
		5	0	3.06	0	0.38	3.44	0	0	0.06	0	0	7.62	0.06	0	0	0.06	0	42.88
	Shute Harbour	2	0	0.38	0	0.12	0.69	0	0	0	0.06	0.31	2.94	0.12	1.44	0	0.31	0	25.88
		5	0	0.44	0	0.06	0.94	0	0	0.06	0	1.25	0.75	0.06	0	0	0.06	0	40.38
	Pine	2	0	5.27	0.13	2.2	12.33	0.06	0.06	0.25	0	2.14	17.43	0.44	11.85	0	1.19	0	32.09
		5	0	5.57	0	1.63	2.82	0	1.94	0.12	0	0.25	5.7	0	0.31	0	0.13	0	54
	Seaforth	2	0	1.38	0.75	0	7.42	0	0.19	0.19	1.12	2.36	5.68	1.71	4.35	0	2.83	0	36.31
		5	0	1.19	0	0.06	3.51	0	0.19	0.12	0.25	5.06	1.75	0.88	0.44	0	0.19	0	43.97

(sub-)region	Reef	Depth	Rhodophyta (red algae)					Chlorophyta (green algae)			Phaeophyta (brown algae)							Undefined	Turf Algae
			Asparagopsis	Crustose coralline	*Hypnea*	*Peyssonnelia*	Undefined	*Caulerpa*	*Halimeda*	Undefined	*Dictyopteris*	*Dictyota*	*Lobophora*	*Padina*	Sargassaceae	*Stypopodium*	Undefined		
Fitzroy	Barren	2	0	0.81	0	0.12	0.19	0	0	0	0	0	0	0	0	0	0	0	58.91
		5	0	2.39	0	0.5	2.19	0	0	0	0	0	4.27	0	0	0	0	0	27.44
	North Keppel	2	0	1.14	0	0.5	0.38	0	0	0	0	0	37.54	0	0	0	0	0	39.93
		5	0	1.32	0	1.07	0.25	0	0	0	0	0.25	25.26	0	0	0	0	0	37.75
	Middle	2	0	1.56	0	0.75	1.5	0.06	0	0	0	0.06	16.58	0	23.03	0	0.06	0	44.93
		5	0	0.76	0	0.25	2.26	0	0	0	0	0	6.85	0	36.69	0	0.12	0	34.66
	Keppels South	2	0	0.06	2.12	0.19	1.7	0	0	0	0	0.88	2.31	0	8.62	0	0.19	0	47.09
		5	0	0.25	0.44	0.63	1	0.13	0	0	0	7.38	11.99	0.06	0	0	0.12	0	42.55
	Pelican	2	0	2.62	0	0.12	10.25	0	0	1.25	0	0.62	7.69	0	18.88	0	0.38	0	40.12
		5	0	0.94	0	1.06	5.25	0	0	0.5	0	0.12	2.38	0	1.56	0	0.38	0	33.88

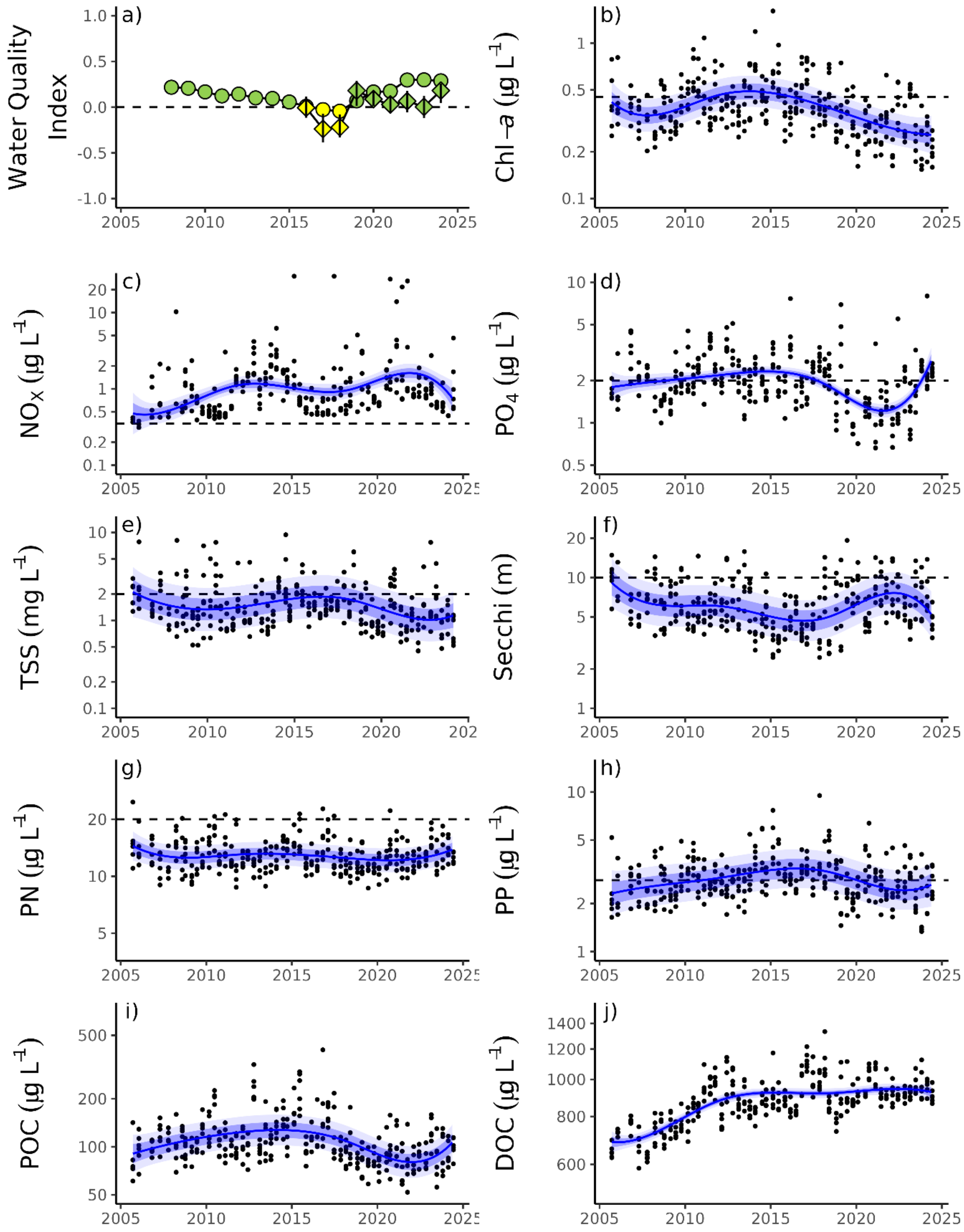

Figure A11 Temporal trends in water quality: Barron–Daintree sub-region. a) water quality index, b) Chlorophyll a, c) nitrate/nitrite, d) Phosphate, e) total suspended solids, f) secchi depth, g) particulate nitrogen, h) particulate phosphorus, i) particulate organic carbon and j) dissolved organic carbon. Water quality index colour coding: dark green- 'very good'; light green – 'good'; yellow – 'moderate; orange – 'poor'; red – 'very poor'. The long-term trend in the WQ index is shown by circles where seasonal and short-term variability are removed, while an updated annual condition Index calculated from 2015 onwards uses diamonds. The water quality index is the aggregate of variables plotted in b, c, e - h and calculated as described in Gruber *et al*. (2020). Trends in PO4, POC and DOC values are plotted here (d, i, j); guideline values for POC and DOC have yet to be established. Generalised additive mixed effect models (trends) are represented by blue lines with blue shaded areas defining 95% confidence intervals of those trends, black dots represent depth weighted averages of observed data. These trends and data are accounting for the effects of wind, waves, tides, and seasons after applying x-z detrending. Dashed reference lines indicate the annual, open coastal Water Quality Guideline values (GBRMPA 2010). Extract from Moran *et al*. (2025).

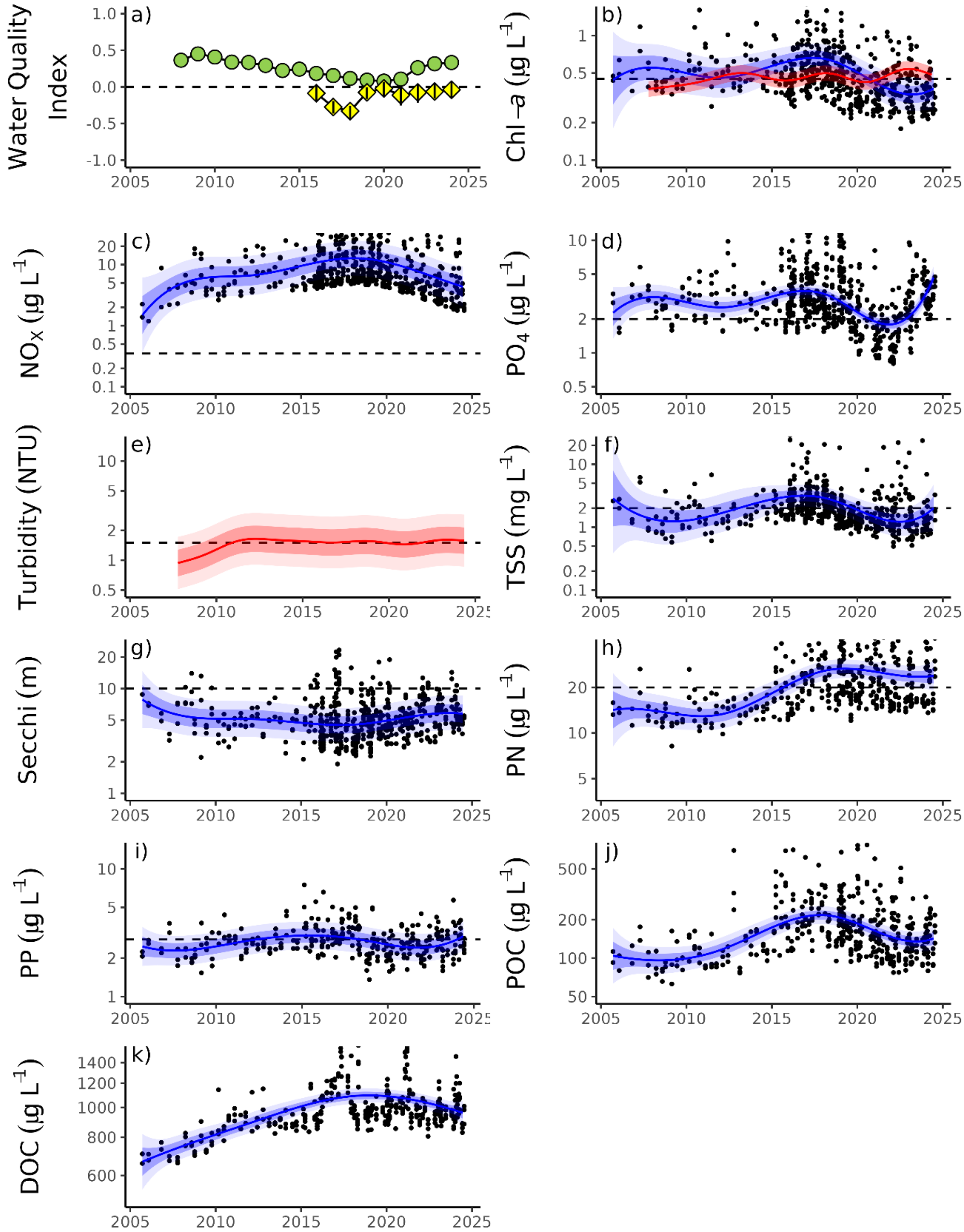

Figure A12 Temporal trends in water quality: Johnstone Russell-Mulgrave sub-region. a) water quality index, b) chlorophyll a, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus j), particulate organic carbon and k) dissolved organic carbon. Water quality index colour coding: dark green- 'very good'; light green – 'good'; yellow – 'moderate; orange – 'poor'; red – 'very poor'. The long-term trend in the WQ index is shown by circles where seasonal and short-term variability are removed, while an updated annual condition Index calculated from 2015 onwards uses diamonds. The water quality index is the aggregate of variables plotted in b, c, f - i and calculated as described in Gruber *et al.* (2020). Trends in PO4, POC and DOC values are plotted here (d, j, k); guideline values for POC and DOC have yet to be established. Generalised additive mixed effect models (trends) are represented by blue lines with blue shaded areas defining 95% confidence intervals of those trends, black dots represent depth weighted averages of observed data. These trends and data are accounting for the effects of wind, waves, tides, and seasons after applying x-z detrending. Trends of records from ECO FLNTUSB instruments (b, e) are represented in red, individual records are not displayed. Dashed reference lines indicate the annual, open coastal Water Quality Guideline values (GBRMPA 2010). Extract from Moran *et al.* (2025).

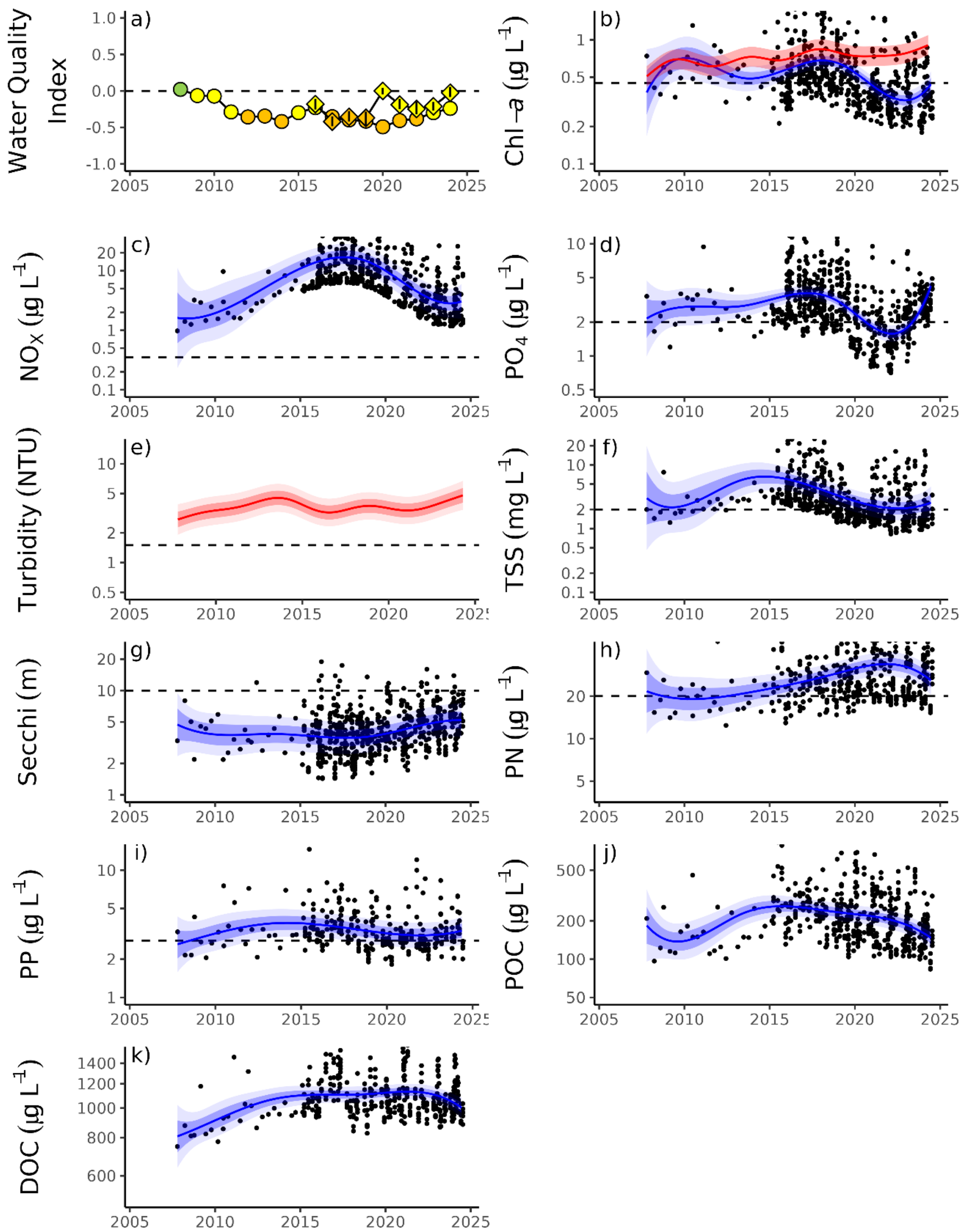

Figure A13 Temporal trends in water quality: Herbert–Tully sub-region. a) water quality index, b) Chlorophyll a, c) Nitrate + Nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved organic carbon. Water quality index colour coding: dark green- 'very good'; light green – 'good'; yellow – 'moderate; orange – 'poor'; red – 'very poor'. The long-term trend in the WQ index is shown by circles where seasonal and short-term variability are removed, while an updated annual condition Index calculated from 2015 onwards uses diamonds. The water quality index is the aggregate of variables plotted in b - i and calculated as described in Gruber *et al*. (2020). Trends in PO4, POC and DOC values are plotted here (d, j, k); guideline values for POC and DOC have yet to be established. Generalised additive mixed effect models (trends) are represented by blue lines with blue shaded areas defining 95% confidence intervals of those trends, black dots represent depth weighted averages of observed data. These trends and data are accounting for the effects of wind, waves, tides, and seasons after applying x-z detrending. Trends of records from ECO FLNTUSB instruments (b, e) are represented in red, individual records are not displayed. Dashed reference lines indicate the annual, open coastal Water Quality Guideline values (GBRMPA 2010). Extract from Moran *et al*. (2025).

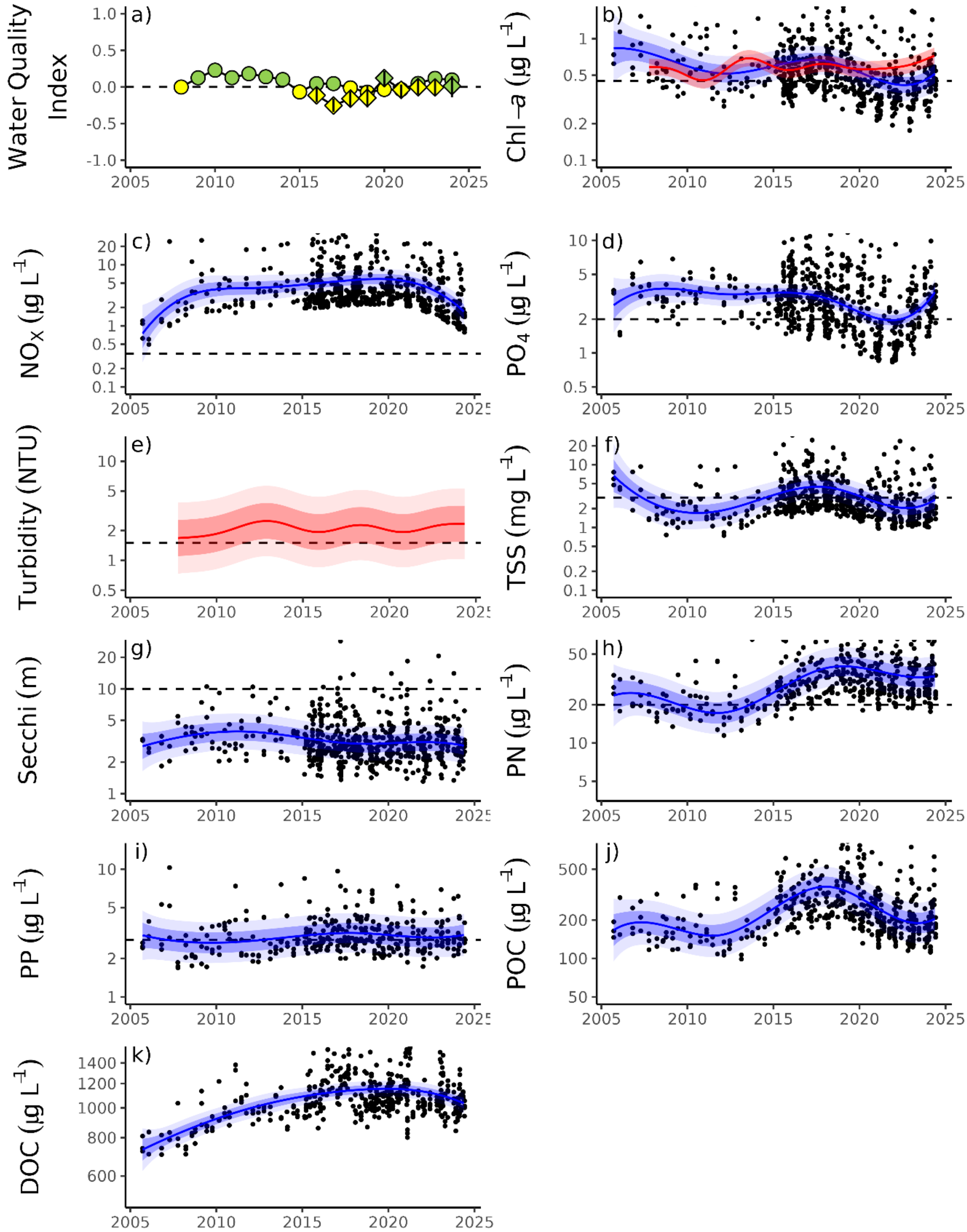

Figure A14 Temporal trends in water quality: Burdekin region. a) water quality index, b) Chlorophyll a, c) Nitrate + Nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved organic carbon. Water quality index colour coding: dark green- 'very good'; light green – 'good'; yellow – 'moderate; orange – 'poor'; red – 'very poor'. The long-term trend in the WQ index is shown by circles where seasonal and short-term variability are removed, while an updated annual condition Index calculated from 2015 onwards uses diamonds. The water quality index is the aggregate of variables plotted in b - i and calculated as described in Gruber *et al*. (2020). Trends in PO4, POC and DOC values are plotted here (d, j, k); guideline values for POC and DOC have yet to be established. Generalised additive mixed effect models (trends) are represented by blue lines with blue shaded areas defining 95% confidence intervals of those trends, black dots represent depth weighted averages of observed data. These trends and data are accounting for the effects of wind, waves, tides, and seasons after applying x-z detrending. Trends of records from ECO FLNTUSB instruments (b, e) are represented in red, individual records are not displayed. Dashed reference lines indicate the annual, open coastal Water Quality Guideline values (GBRMPA 2010). Extract from Moran *et al*. (2025).

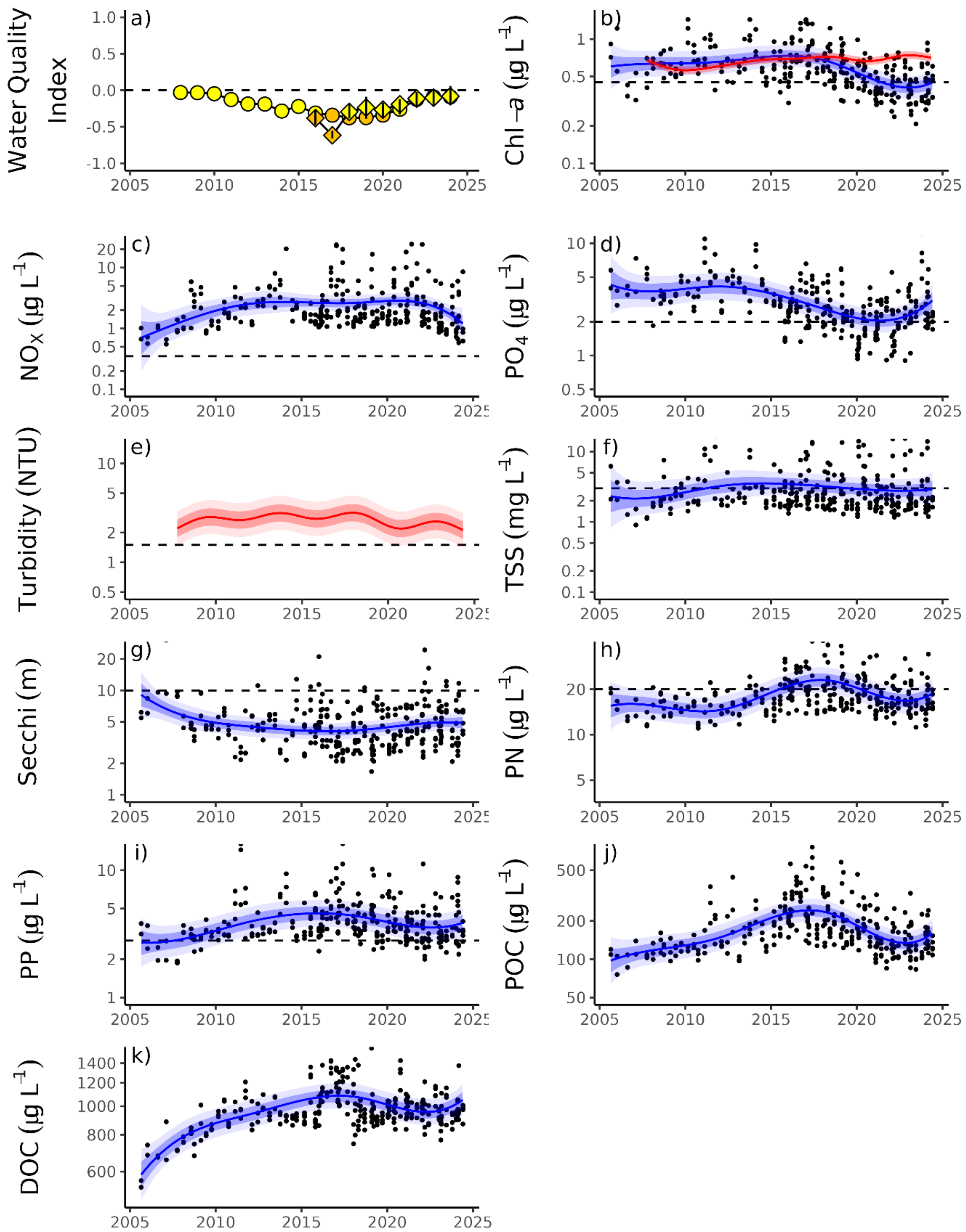

Figure A15 Temporal trends in water quality: Mackay–Whitsunday region. a) water quality index, b) Chlorophyll a, c) Nitrate + Nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved organic carbon. Water quality index colour coding: dark green- 'very good'; light green – 'good'; yellow – 'moderate; orange – 'poor'; red – 'very poor'. The long-term trend in the WQ index is shown by circles where seasonal and short-term variability are removed, while an updated annual condition Index calculated from 2015 onwards uses diamonds. The water quality index is the aggregate of variables plotted in b - i and calculated as described in Gruber *et al*. (2020). Trends in PO4, POC and DOC values are plotted here (d, j, k); guideline values for POC and DOC have yet to be established. Generalised additive mixed effect models (trends) are represented by blue lines with blue shaded areas defining 95% confidence intervals of those trends, black dots represent depth weighted averages of observed data. These trends and data are accounting for the effects of wind, waves, tides, and seasons after applying x-z detrending. Trends of records from ECO FLNTUSB instruments (b, e) are represented in red, individual records are not displayed. Dashed reference lines indicate the annual, open coastal Water Quality Guideline values (GBRMPA 2010). Extract from Moran *et al*. (2025).

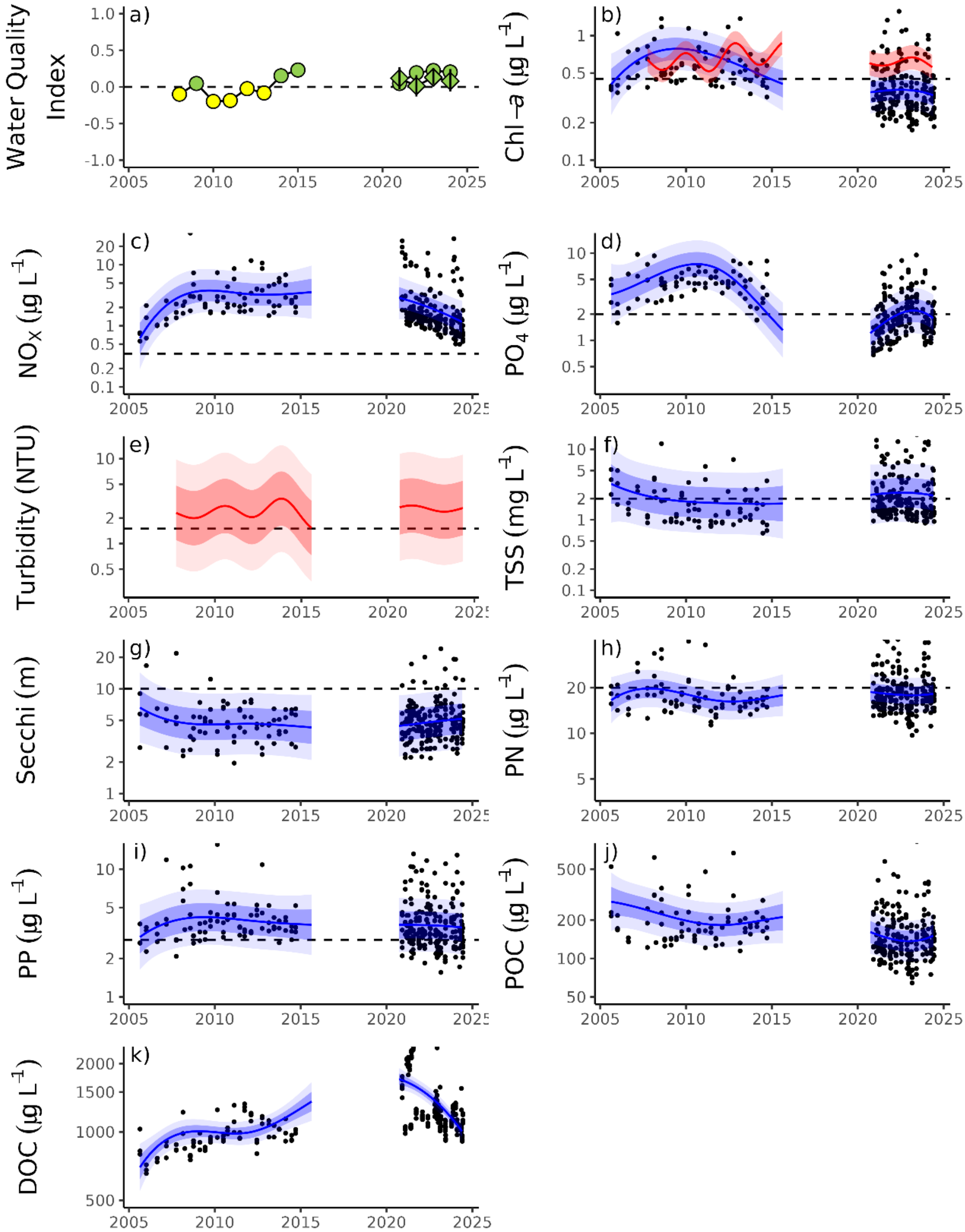

Figure A16 Temporal trends in water quality: Fitzroy region. a) water quality index, b) Cchlorophyll a, c) Nitrate + Nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved organic carbon. Water quality index colour coding: dark green- 'very good'; light green – 'good'; yellow – 'moderate; orange – 'poor'; red – 'very poor'. The long-term trend in the WQ index is shown by circles where seasonal and short-term variability are removed, while an updated annual condition Index calculated from 2015 onwards uses diamonds. The water quality index is the aggregate of variables plotted in b - i and calculated as described in Gruber *et al.* (2020). Trends in PO4, POC and DOC values are plotted here (d, j, k); guideline values for POC and DOC have yet to be established. Generalised additive mixed effect models (trends) are represented by blue lines with blue shaded areas defining 95% confidence intervals of those trends, black dots represent depth weighted averages of observed data. These trends and data are accounting for the effects of wind, waves, tides, and seasons after applying x-z detrending. Trends of records from ECO FLNTUSB instruments (b, e) are represented in red, individual records are not displayed. Dashed reference lines indicate the annual, open coastal Water Quality Guideline values (GBRMPA 2010). Water quality monitoring ceased in 2015 and resumed in 2021. Extract from Moran *et al.* (2025).

Appendix 2: Publications and presentations 2023–2024

Publications

Hock K, Hastings A, Doropoulos C, Babcock RC, Ortiz JC, Thompson A, Mumby PJ (2024) Transient dynamics mask the resilience of coral reefs. Theoretical Ecology 17:1-12. available here

McKenzie L, Pineda M-C, Grech A, Thompson A (2024) Question 1.2/1.3/2.1 What is the extent and condition of Great Barrier Reef ecosystems, and what are the primary threats to their health? In Waterhouse J, Pineda M-C, Sambrook K (Eds) 2022 Scientific Consensus Statement on land-based impacts on Great Barrier Reef water quality and ecosystem condition. Commonwealth of Australia and Queensland Government. available here

Mackay-Whitsunday-Isaac Healthy Rivers to Reef Partnership (2024). Mackay-Whitsunday-Isaac 2024 Report Card Results Technical Report. Mackay-Whitsunday-Isaac Healthy Rivers to Reef Partnership, Mackay, QLD. 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., (2024). Technical Report for the Townsville Dry Tropics Report Card Results 2024 (Reporting on July 2022 – June 2023). Healthy Waters Partnership for the Dry Tropics, Townsville. available here

Wet Tropics Waterways 2024. Wet Tropics Report Card 2024 (reporting on data 2022-23). Waterway Environments: Results. Wet Tropics Waterways and Terrain NRM, Innisfail. available here

Presentations

Marine Monitoring Program - Coral 2024. Presentation at Marine Monitoring Program Science Seminar. Reef Authority, 5th Sep 2024

Marine Monitoring Program – Coral 2024. Annual presentation to stakeholders. Townsville Yacht Club 13th November 2024

Marine Monitoring Program – Coral 2024 Mackay Whitsunday Isaac. Mackay Whitsunday Isaac Paddock to Reef Regional science Forum. Mackay. 28th August 2024

Marine Monitoring Program – Coral and water quality monitoring activities in Manbarra Sea Country. Presentation to Manbarra Elders and Mingga Mingga Rangers. Palm Island 4th July 2024.