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GREAT BARRIER REEF MARINE MONITORING PROGRAM

Inshore coral reefs monitoring Annual Report 2022–23

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Front cover photo: A diver conducts coral demography surveys over a dense thicket of branching *Acropora*. © Australian Institute of Marine Science, Photographer: C. Thompson.

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

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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.

Inshore reefs have remained in an overall 'poor' condition since 2019 (Figure 1). Over this period Coral cover scores have improved, reflecting increased coral cover in most regions. In contrast, the cover of macroalgae, which compete with coral for space, remains high on several reefs in each region with scores for this indicator continuing to decline (Figure 1). There have been no consistent trends in Cover change, Composition or Juvenile coral indicator scores, with each indicator returning an overall score at the upper end of the 'poor' range in 2023.

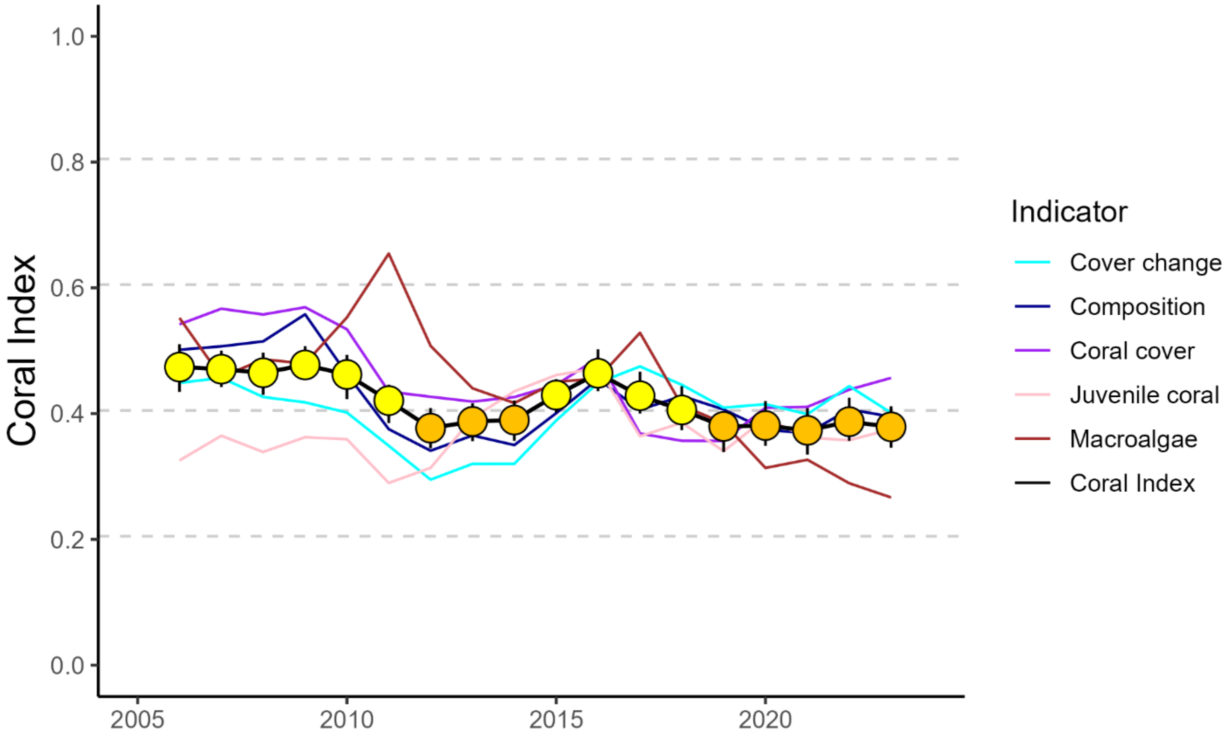

Figure 1 Trends in the Coral Index and contributing indicator scores for the inshore Reef. Coral Index scores are coloured according to Reef Water Quality Report Card categories: orange = 'poor', yellow = 'moderate'.

Overall, negative relationships between change 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, higher prevalence of macroalgae and lower cover of corals along gradients of declining water quality highlight 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. An expected increase in disturbance frequency due to 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.

Coral communities are naturally dynamic, going through periods of recovery following mortality after acute disturbances, such as cyclones. Improvement of coral community condition scores from a low point in 2011 through to 2016 demonstrated the capacity of inshore coral communities to recover. However, between 2016 and 2021, the cumulative pressures imposed by cyclones, high seawater temperatures, flooding and high crown-of-thorns starfish densities contributed to a period of decline.

There were no acute climatic pressures imposed on inshore reefs in 2022–2023. Sea water temperatures in early 2023 were below those likely to cause coral bleaching, and no cyclones passed through the inshore Reef. Corallivorous crown-of-thorns starfish were again present on reefs in the Johnstone Russell–Mulgrave sub-region. While their numbers have declined since a peak in 2020, in 2023 juvenile crown-of-thorns starfish were observed at ‘outbreak’ density at Franklands East. The impact of these starfish on corals was reduced by culling undertaken by the Crown-of-thorns Starfish Control Program.

Although Coral cover has been improving since 2019, this contrasts with the continued decline in Macroalgae scores that reached new lows in 2023. In combination, poor scores for the Juvenile coral and Cover change indicators, and the decline in Macroalgae scores in 2023, are of concern as they suggest reduced resilience of coral communities during the current recovery period.

Coral community condition, expressed as the Coral Index, is a composite of five indicators combined for all reefs in a region. 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,
- **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.

The following sections summarise the condition of coral communities in mid-2023 in each Natural Resource Management region in which inshore reefs are monitored.

Wet Tropics region coral community condition

Inshore coral communities remain in ‘moderate’ condition. However, the stability of the Coral Index observed since 2016 masks differing trends within the three sub-regions (Barron–Daintree, Johnstone Russell–Mulgrave and Herbert–Tully).

- In the Barron–Daintree sub-region, the Coral Index score continued to improve but remained within the ‘moderate’ condition range. Coral cover has recovered to levels observed in 2011, prior losses were caused by disease, crown-of-thorns starfish and then cyclone Ita in 2014. The Cover change indicator demonstrates this recovery has occurred at or above expected rates. In contrast, ongoing high cover of macroalgae and correspondingly low densities of juvenile corals at shallow sites at Snapper North continue to put downward pressure on the Coral Index.

- In the Johnstone Russell–Mulgrave sub-region, the Coral Index score has fluctuated between ‘moderate’ and ‘good’ condition since 2016. In 2022, Coral cover reached the highest levels recorded since the program began in 2005 and remained at similar levels in 2023. This improvement in coral cover occurred despite ongoing impacts of crown-of-thorns starfish. Ongoing removals of these starfish from Fitzroy Island and the Frankland Group by the Crown-of-thorns Starfish Control Program will have reduced their impact on corals. As for the Barron–Daintree sub-region, low Macroalgae scores corresponded to low Juvenile scores at several reefs, and this combination limits the Coral Index score in 2023.
- In the Herbert–Tully sub-region, the Coral Index score has gradually declined since 2020 but remains ‘moderate’. High water temperatures causing coral bleaching in early 2022 will have contributed to reduced scores for the Cover change indicator. Macroalgae remains ‘poor’ as high levels of macroalgae persist at Dunk Island and Bedarra Island.

Burdekin region coral community condition

Coral Index score remains ‘moderate’ having declined from a high point in 2020, which was reached after a period of recovery following the impact of cyclone Yasi in 2011. Thermal stress in early 2020 and to a lesser extent in 2022 was sufficient to cause coral bleaching at most reefs. Despite these bleaching events Coral cover in 2021 and 2022 continued to increase with the regional average in 2023 higher than observed since the beginning of the program in 2005.

The two indicators most influential in the post-2020 decline of the Coral Index were Juvenile coral and Macroalgae. Scores for Macroalgae remain ‘poor’ or ‘very poor’ at all but the two most offshore reefs monitored. The Juvenile coral score remains ‘poor’, with declines since 2020 most evident at 5 m depths at reefs where the density of juvenile *Turbinaria* has declined.

Mackay–Whitsunday region coral community condition

The Coral Index score was unimproved in 2023 and has remained ‘poor’. Recovery from the severe impact of cyclone Debbie in 2017 remains slow and most indicators remained in the ‘poor’ category. Coral cover has begun to improve at some reefs with improvement more consistent at shallower sites. Signs of recovery are most evident in 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.

Fitzroy region coral community condition

The Coral Index score remained ‘poor’ with no improvement in 2023. Scores for Coral cover have continued in the ‘moderate’ range and while increasing slightly in recent years, the rate of this recovery has been slow. Very high cover of macroalgae and low densities of juvenile corals at most reefs are reflected in ‘very poor’ and ‘poor’ indicator scores, respectively.

The state of reefs varied markedly across the region. Coral cover was highest at Barren Island, the reef furthest from the coast. In contrast, persistent cover of large, brown macroalgae continue to suppress coral community recovery at most other reefs.

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 2019). 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.

Macroalgal cover 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). High macroalgal abundance may suppress reef resilience (e.g., Hughes *et al.* 2007, Foster *et al.* 2008, Cheal *et al.* 2010, but see Bruno *et al.* 2009) 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 energy in two ways; by feeding on ingested particles and planktonic organisms (heterotrophic feeding) and from the photosynthesis of their symbiotic algae. 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).

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 2023 Reef Water Quality Report Card. This report includes results from coral reefs monitored by AIMS as part of the MMP until July 2023 with inclusion of data from inshore reefs monitored by the AIMS Long-Term Monitoring Program (LTMP) from 2005 to 2023. 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*, physical damage 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 (Gruber *et al*. 2024, 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 (Gruber *et al.* 2024). 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. For the Fitzroy region coral bleaching was also observed in 2006, and that year is also excluded from the baseline climatology. 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

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

The first, $Obs.DHD,$ is derived from the logger time-series and presents the summer (December to March) exposure to temperatures greater than the (sub-)region's seasonal climatology as:

$$Obs.DHD = \sum T_i - Tci$$

Where, T_i is the mean temperature recorded by all loggers in a (sub-)region on a particular day (i), and Tci is (sub-)region's climatological monthly mean temperature for that day of the year. Only positive anomalies are summed.

The second, degree heating days (DHD), was derived from ~4 km^2 pixels adjacent to each coral monitoring location downloaded from the Australian Bureau of Meteorology (BoM) satellite-based interactive website ReefTemp Next Generation[1]. DHD values were calculated as the sum of daily positive deviations from 14-day IMOS climatology – a one-degree exceedance for one day equates to one DHD, a two-degree exceedance for one day equates to two DHD. DHD anomalies are summed over the period 1 December to 31 March each summer.

Finally, 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 ~16 km^2 pixel in a calendar year. DHW estimates differ from DHD not only on the summation scale of weeks of exposure (rather than days) but also on the baseline temperature stress. 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).

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[1] ReefTemp Next Generation was developed through the Centre for Australian Weather and Climate Research – a partnership between the Commonwealth Scientific and Industrial Research Organization (CSIRO) and BoM (Garde *et al.* 2014).

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

	Data range	Method	Usage	Data source
Climate				
Riverine discharge	1980 – 2023	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 Gruber *et al.* (2024)
Riverine DIN, sediment and PN loads	2006 – 2023		covariate in analysis of temporal change in Coral Index	MMP Water Quality (Gruber *et al*. 2024)
Sea temperature	2005 – 2023	*in situ* sensor at coral sites	regional plots, thermal bleaching disturbance categorisation, *in situ* DHD estimates	MMP Inshore Coral monitoring
DHD	2006 – 2023	remote sensing, ~4 km^2 pixels adjacent to coral sites	informing attribution of thermal stress, regional plots, thermal bleaching disturbance categorisation, thermal stress maps	BoM
DHW	2006 – 2023	remote sensing	informing attribution of thermal stress, thermal stress maps	National Oceanographic and Atmospheric Administration
Cyclone tracks	2005– 2023		informing attribution of storms as cause of observed coral loss, cyclone track maps	BoM
Environment at coral monitoring sites				
Chlorophyll *a* (Chl *a*) and total suspended solids (TSS)	2003 – 2023	remote sensing and coupled niskin samples	Chl *a* exposure, mapping. Chl *a* and TSS concentrations covariates in analysis of variability in Coral Index score changes	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
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, Gruber *et al*. 2024). 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 2019–2023 was estimated as the mean exposure to each water-type over that period.

Wet-season concentrations of Chl *a* and TSS within each colour class were estimated based on distributions Chl *a* and TSS measured from near-surface water samples, following the sampling methods outlined in Gruber *et al.* (2024). 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, Midshelf 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 median extracted as the median of this combined distribution. The five yearly median for each reef was estimated as the mean of the annual wet-season medians.

Table 2 Water types estimated from Sentinel imagery. Table 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 two 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. 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.

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

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
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	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	●	●		●		●	+	●		●	+		●		●				

Figure 2 Coral sampling locations 2023.

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. A total of 32 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. The number of juvenile coral colonies were counted along the permanently marked transects. 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.* 2009). 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 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 scale ranges from 0+ when individual colonies were bleached or damaged, and through the categories 1 to 5 when 1%–10%, 11%–30%, 31%–50%, 50%–75% and 75%–100% of colonies were affected. The categories 1 to 5 are further refined by inclusion of a -ve or +ve symbol when affected proportions are estimated as being in the lower or upper portion of the category. The physical damage category may include anchor as well as storm damage. The LTMP include these surveys over the full 50 m length of transects used in that program.

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).

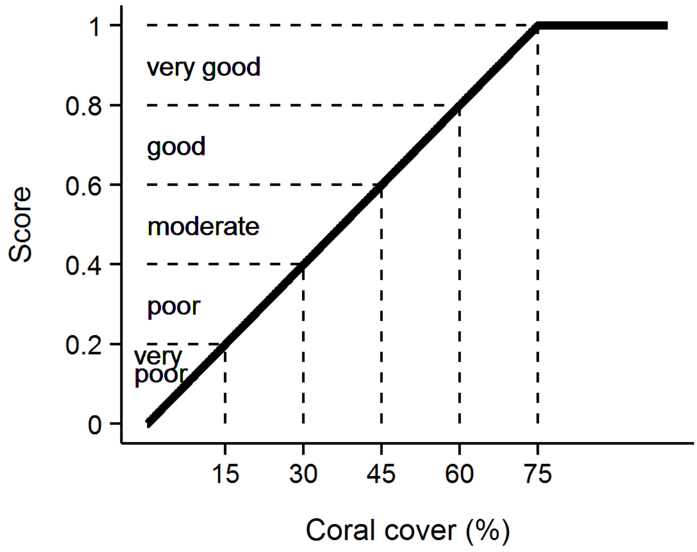

Figure 3 Scoring diagram for the Coral cover indicator metric. Numeric scores and associated condition classifications based on observed coral cover are presented.

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.

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).

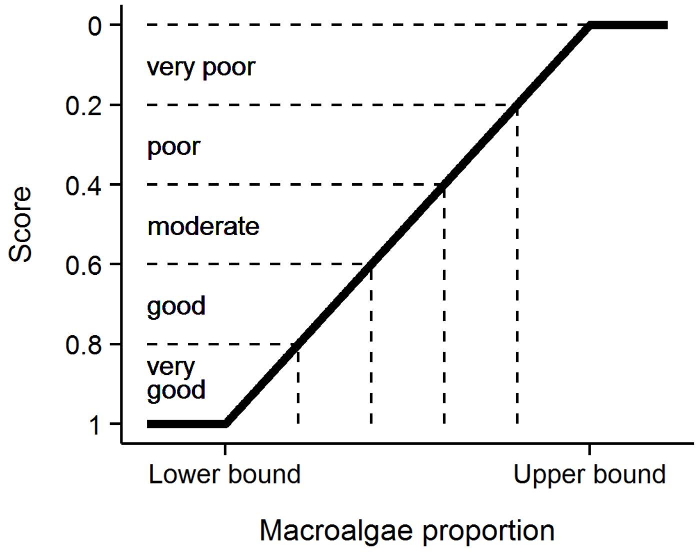

Figure 4 Scoring diagram for the Macroalgae indictor metric. Upper and lower threshold values are reef and depth specific. Numeric scores and associated condition classifications are presented. Note that for this metric the y-axis is inverted as high values reflect poor condition.

2.4.3 *Juvenile coral indictor 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).

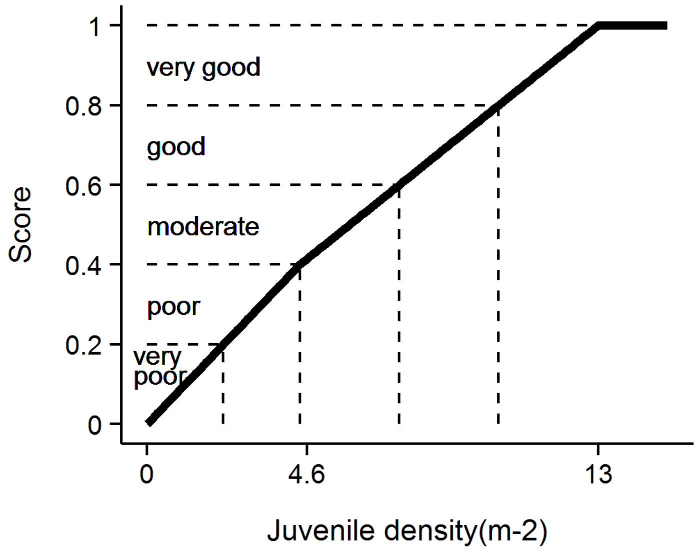

Figure 5 Scoring diagram for the Juvenile coral indicator metric. Numeric scores and associated condition classifications are presented.

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.

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

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

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

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

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

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

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

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

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 $v\bar{A}cr_i$.

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 rhat (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.

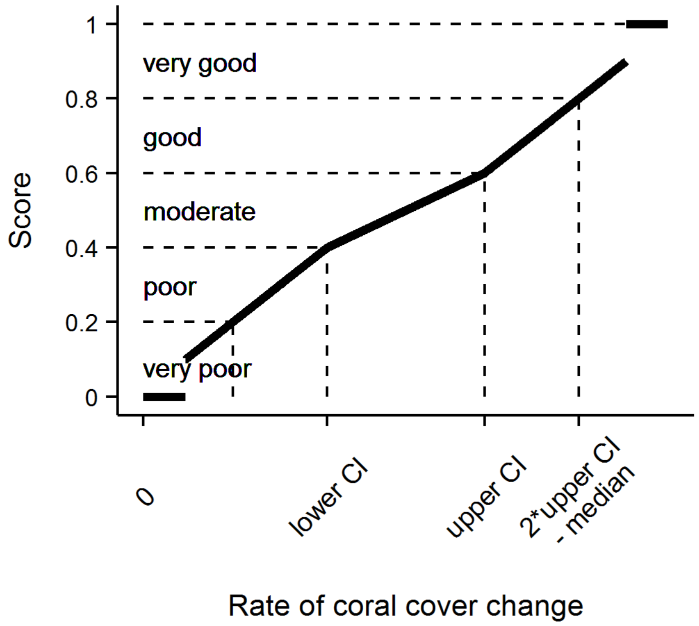

Figure 6 Scoring diagram for Cover change indicator metric.

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).

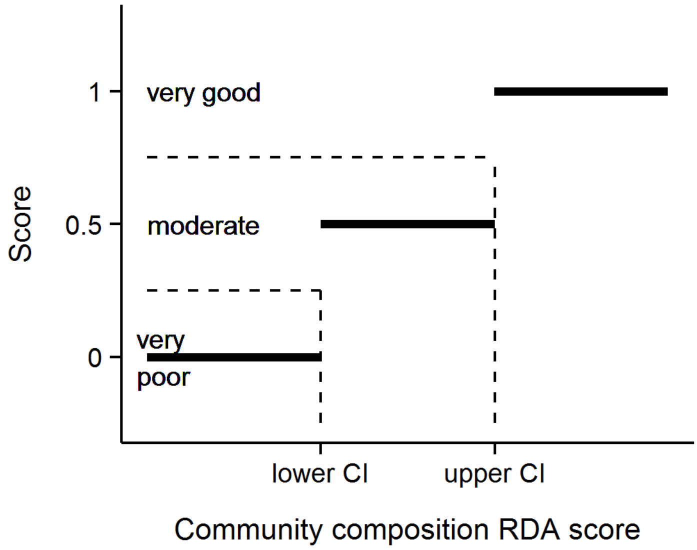

Figure 7 Scoring diagram for the Composition indicator metric

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 5. 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 5 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 6).

Table 6 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 2023	Reef and region	Chl *a*, TSS	Generalised linear mixed models, predicted responses
4.7.2	Temporal variability in Coral Index in relation to water quality	region	Regional riverine: discharge, Total N and Total P loads.	Generalised additive models, predicted responses
Appendix 1: Additional Information	Trends in benthic community composition.	reef/Depth		Plots
	Summaries of 2023 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 via generalised linear mixed models. Each combination of Coral Index or indicator score and depth were fit separately to two water quality proxies: mean Chl *a* and TSS concentrations. General Reef-wide trends were identified on the basis that Akaike information criterion (AICc) values for models fitting the Coral Index or indicator responses to the water quality proxy, and including random intercepts for each region, were at least two units lower than the simpler model that did not include the water quality proxy. 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. The exception was the Composition indicator scores that were modelled using a probit regression due to their categorical response. Indicator values for the Macroalgae and Composition indicators (proportion of algal cover categorised as macroalgae, and product of genus cover and water quality eigenvector weightings) were also examined, as the scores for these indicators are based on thresholds that account for variability along water quality gradients. Macroalgal proportion was also fit using a beta distribution, and a gaussian distribution was used for genus composition values.

Where relationships between Coral Index or indicator scores or indicator values were implied based on AICc comparisons, the generality of the response was further explored by plotting predicted responses from more complex models that also allowed for varied slopes among regions by inclusion of an interaction between water quality proxy and region to the models described above. The results of these models are plotted and confidence intervals for slopes within each region estimated to identify the regions contributing most to the general Reef-wide trends. Generalised linear mixed models were fit via the mgcv package (Wood 2019) while the probit model for community composition was fit with the polr function in the MASS package within the R Statistical and Graphical Environment (R Core Team 2023).

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 and PN, DIN and sediment loads from the adjacent catchments (section 2.1.2),
- pollutant exposure (section 2.1.6).

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 covariates in each model were selected to represent the maximum exposure of the two water years ending in the survey year (y+2). 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 7) 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 7 Information considered for disturbance categorisation.

Disturbance	Description
Thermal bleaching	Consideration of *in situ* DHD 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 2022–23 reporting period, no cyclones were likely to have produced damaging waves to 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, Table A6). Signs of recovery of coral cover in the wake of this cyclone are variable (section 4.5).

Numerous smaller cyclones have crossed the inshore Reef over the last decade (Figure 8) causing more moderate and localised damage (Table A6, see also ((sub-)regional summaries section).

3.2 Sea temperature

Sea temperatures over the 2023 summer were above long-term averages, but below published thresholds of 60 to100 DHD (Garde *et al.* 2014) or 4 DHW (NOAA 2018) that are likely to lead to significant coral bleaching (Figure 9 and Figure 10). Higher temperature anomalies were recorded in 2022 and centred on the Burdekin region; however, minimal bleaching or loss of coral was observed during our surveys in July (Figure 9 and Figure 10, Figure 25e). However, it is likely that if corals did bleach over the summer, they would have recovered their pigment by the time we surveyed. In 2020, the highest deviations occurred in inshore areas south of Hinchinbrook Island (Figure 9 and Figure 10). 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 and Figure 10).

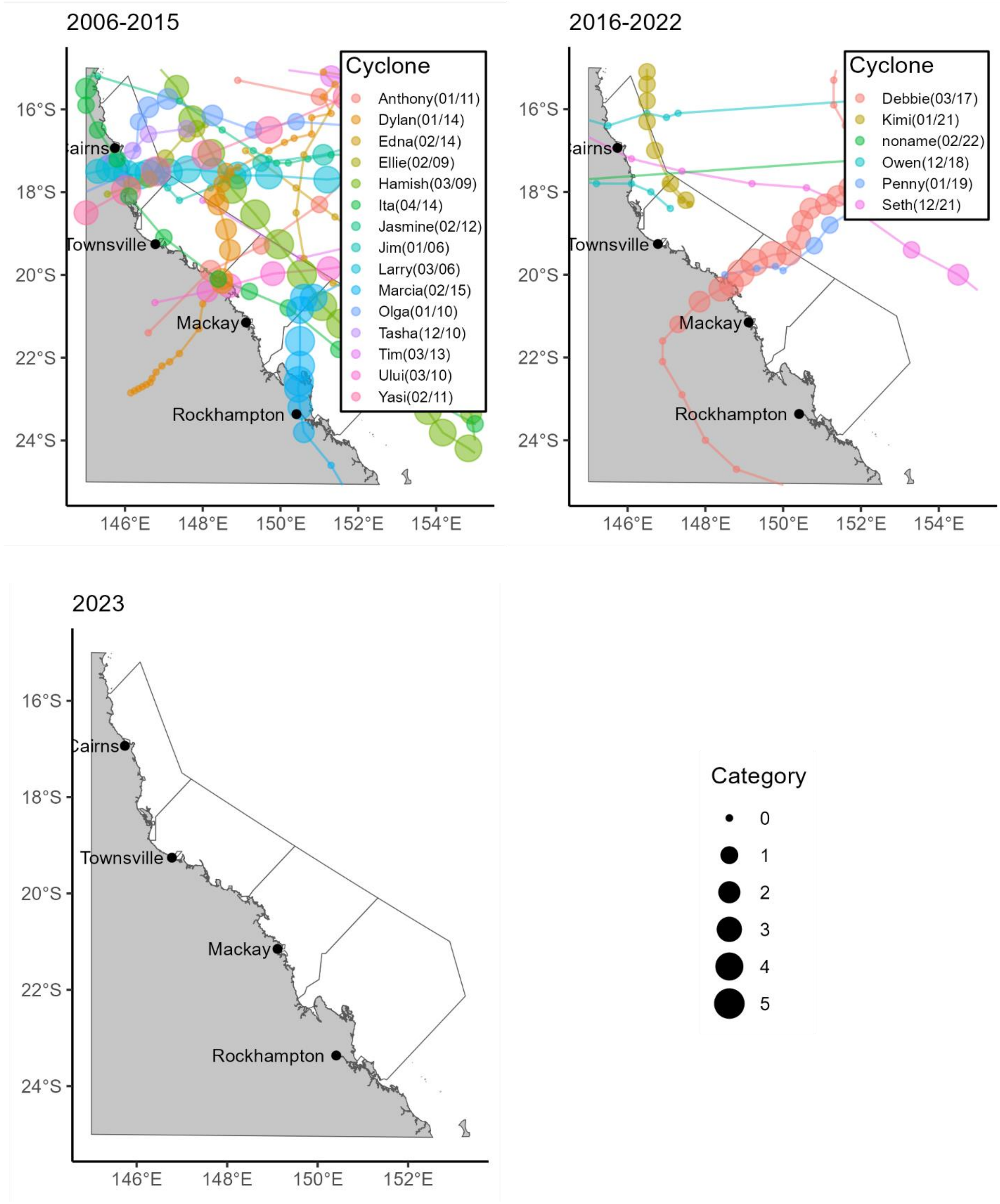

Figure 8 Cyclone tracks for systems crossing the inshore Reef since 2006. Tracks sourced from the BoM

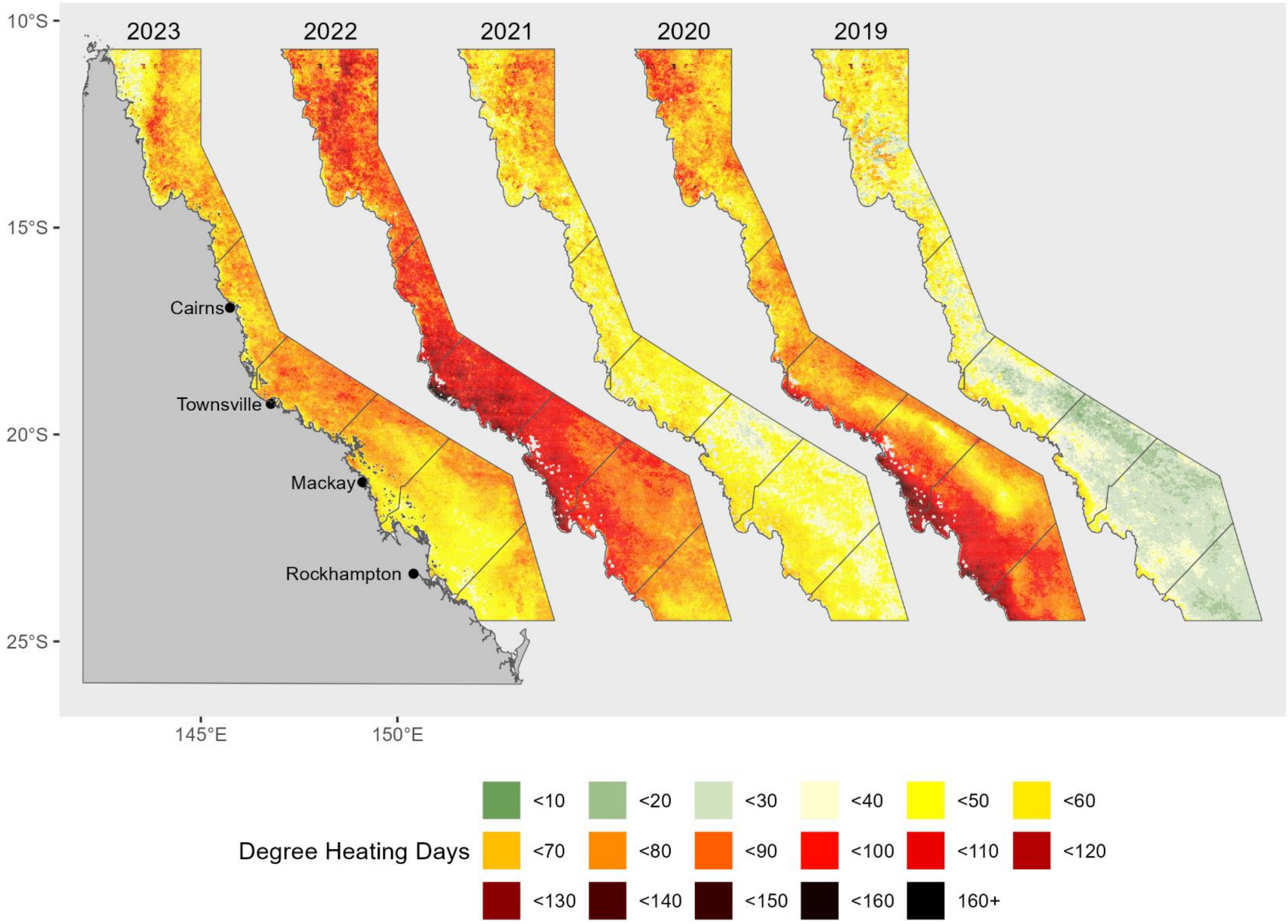

Figure 9 Annual DHD estimates for the Reef. Data are the annual DHD accumulations over the summer period (1 December to 31 March) for ~4 km^2 pixels based on temperatures exceeding 14 Day IMOS climatology. Data were sourced from the BoM ReefTemp next generation web data service .

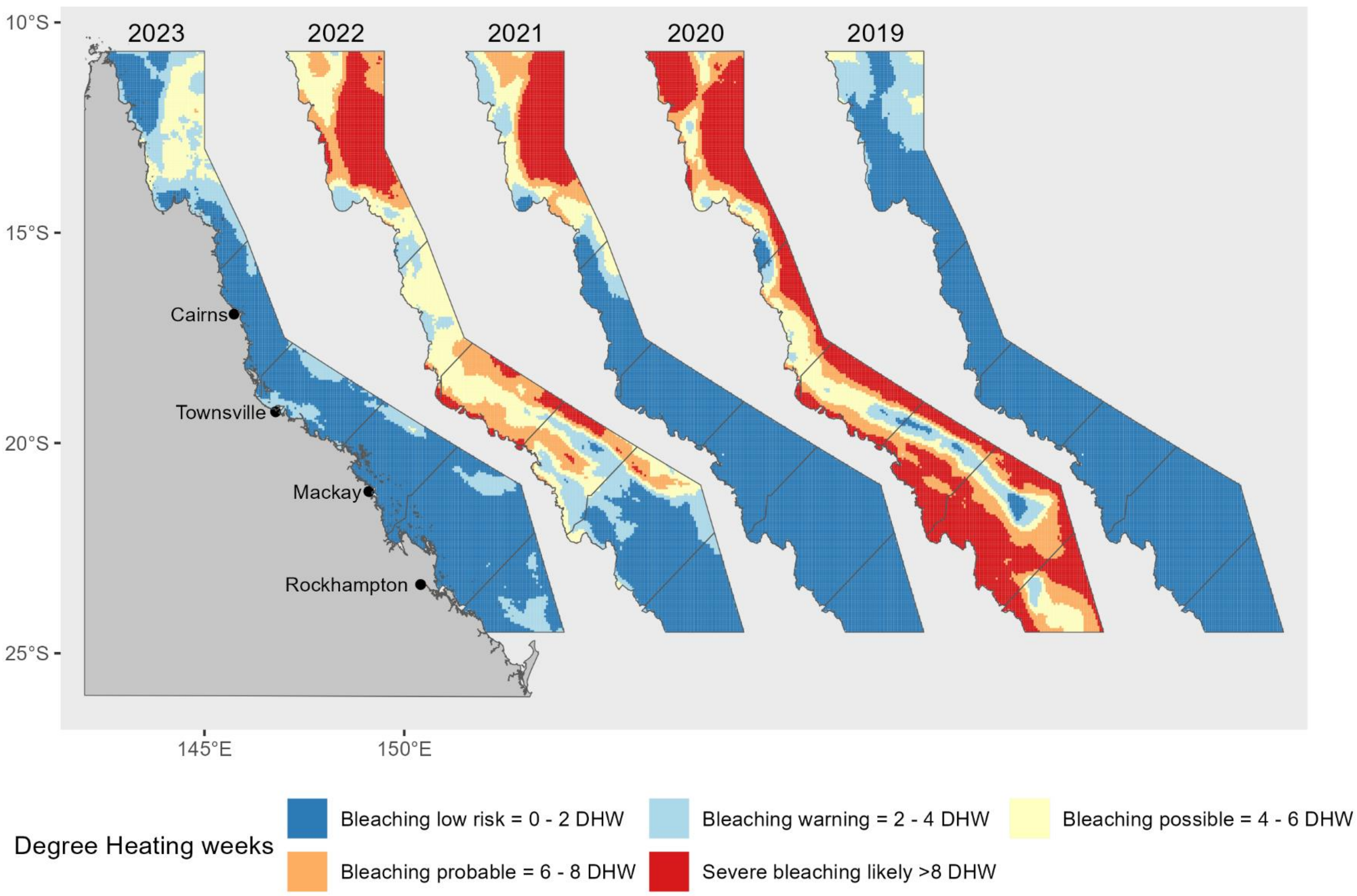

Figure 10 Annual DHW estimates for the Reef. Data are the annual maximum DHW estimates for each ~25 km^2 pixel. Data were sourced from NOAA coral reef watch.

3.3 Crown-of-thorns starfish

In 2023, the density of crown-of-thorns starfish were above outbreak levels at Franklands East (75 ha^{-1}) and a single juvenile was observed at Fitzroy East. Recently dead coral consistent with crown-of-thorns starfish feeding scars was also observed at Franklands West, High West, High East and Fitzroy West but no starfish were observed within the SCUBA search transects. Regionally in the Wet Tropics, numbers have continued to decline from the high levels observed in 2020 (Table 8). In 2023, no crown-of-thorns starfish were observed during MMP surveys in other regions; however, starfish were present in the outer Whitsunday Islands as evidenced by culling data presented in Table 9.

Since 2012 crown-of-thorns starfish have remained present on reefs in the Johnstone Russell–Mulgrave sub-region, with numbers peaking at outbreak levels (> 30 individuals per hectare) at five of the six reefs monitored in 2020 (Figure A8). 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 8). In 2023, juvenile starfish were again present at Franklands East with coral scarring at other reefs in the Johnstone Russell–Mulgrave region suggesting juvenile starfish were present at both High Island and Fitzroy Island.

Table 8 Size class distribution of crown-of-thorns starfish on inshore reefs in the Wet Tropics. Included are the percentages culled, as listed in Table 9, 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	32	**77**	25	**66**	**57(4)**	43(3)	

Table 9 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.

Year	Snapper Island	Low Isles	Green Island	Fitzroy Island	Frankland Group	Black 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)	**1**(0.01)		**6**(0.06)	**109**(0.21)	**4**(0.01)

3.4 River discharge

Discharge in 2023 was marginally above median levels. At the scale of the Reef, interannual variability in discharge highlights potential for increased risk to corals over the period 2007–08 to 2012–13 and then in 2018–19 (Figure 11).

In 2018–19, record flooding of the Daintree River in combination with minor storm damage attributed to pre-cyclone Owen resulted in the loss of 38% of hard coral cover at 2 m depth at Snapper Island South (Figure A1). This was the only acute disturbance to have directly impacted inshore coral communities over the 2018–19 summer.

Heavy rainfall in February 2019 resulted in major flooding of rivers in the Burdekin region and above median discharges from rivers in the Mackay–Whitsunday region and Herbert–Tully and Johnstone Russell–Mulgrave sub-regions. There was no evidence that these floods had any direct impacts on coral communities at reefs monitored in 2019. Species of *Acropora,* known to be sensitive to exposure to low salinities (Berkelmans *et al.* 2012), were surviving at the shallow sites on reefs most proximal to the flooding rivers. However, it is likely that the level of discharge contributed to chronic pressures on coral communities as evidenced by increased levels of disease in these regions (Figure A7). Closer to the coast, the authors' personal observations were that corals at Virago Shoal off the coast of Townsville were killed by floods of the Ross River, while corals along the eastern face of Cape Cleveland were killed by the plume of the Burdekin and / or Haughton rivers.

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 (Table A6, Figure A6). As observed in 2023 recovery from this event was occurring at Keppels South but was limited, at best, at Pelican Island.

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.

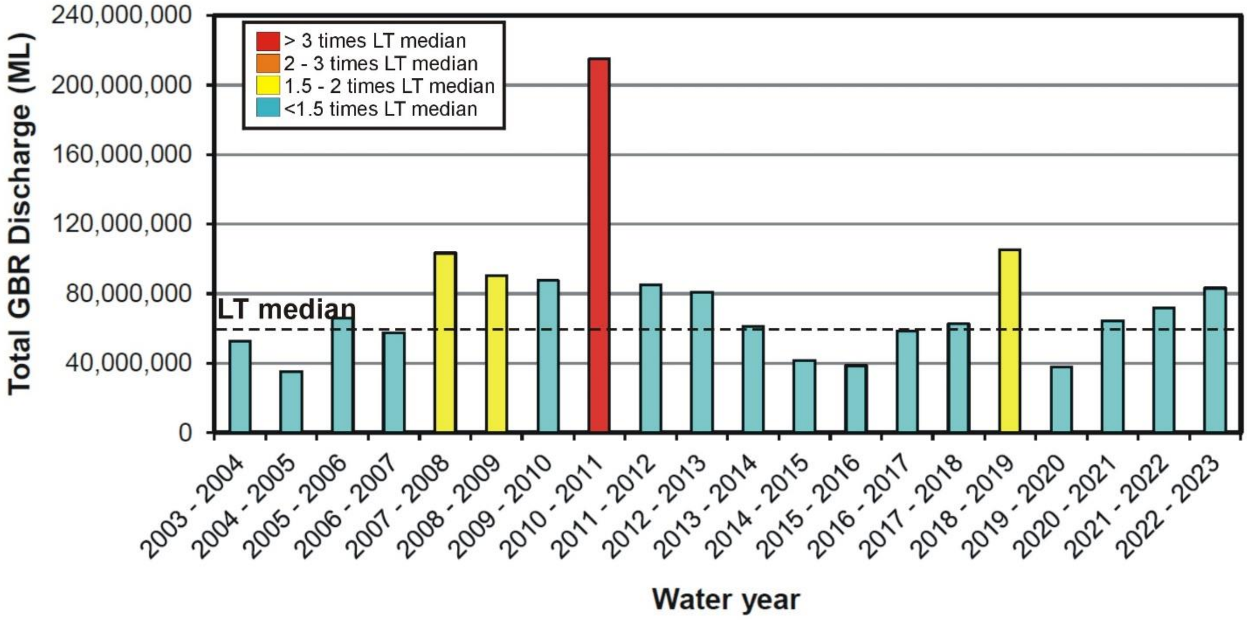

Figure 11 Annual total river discharge to the Reef. Annual estimates aggregate over the water year: 1 October to 30 September, for the 35 main Reef basins. Values are colour coded relative to proportion of long-term (LT) median (1986–2016) discharge. Figure source: Gruber *et al*. 2024, data source: 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 A10 to Figure A15). These plots are sourced from the complimentrary annual MMP Inshore Water Quality annual report (Gruber *et al*. 2024). For full details of the methods used to create these plots the reader should refer to that report.

Salient points to note are:

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

Within section 4 of this report, reference to trends in indicators or deviations from guidelines follow the convention applied by Gruber *et al*. (2024). 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 ANDS TREND

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 available online at http://apps.aims.gov.au/reef-monitoring/.

4.1 Reef-wide coral community condition and trend

At the whole of Reef-scale, the Coral Index score remained largely unchanged from that observed since 2019 and remains ‘poor’ (Figure 12). The decline from ‘moderate’ in 2016 represents the combined pressures associated with cyclone Debbie in 2017, high sea temperatures causing coral bleaching, predation of corals by crown-of-thorns starfish and flooding of the Daintree River (Figure 8, Figure 10, Table 8, Table A5).

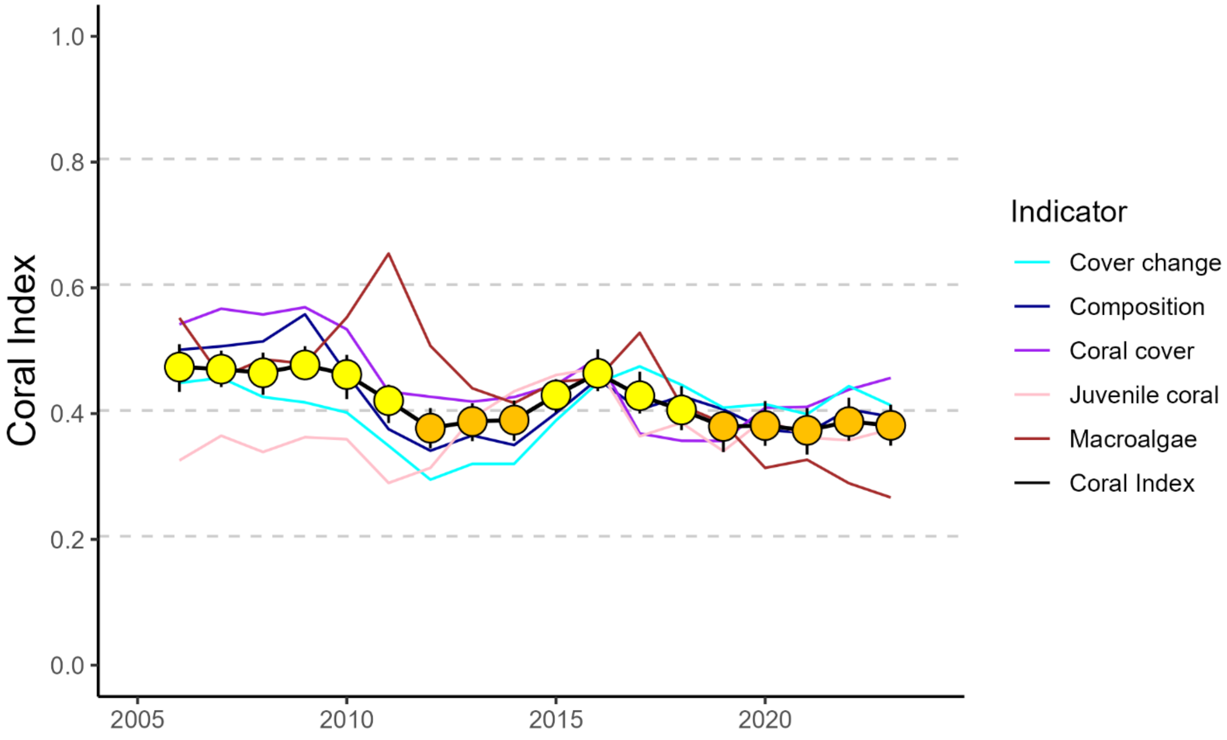

Figure 12 The Reef level trend in Coral Index and indicator scores. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = ‘poor’, yellow = ’moderate’. Error in Coral Index scores were derived from bootstrapped distribution of regional indicator scores weighted by the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

The recovery of coral communities between 2013 and 2016 demonstrated the inherent resilience of the inshore coral communities following a period punctuated by impacts of cyclones and high discharge from the Reef's catchments. Yet, it is unsurprising that the current condition has returned to being 'poor' given the level of pressure imposed marine heat waves in recent years (see Figure 13). In 2023, there has been a slight increase in scores for Coral cover, with this indicator and Cover change now in the 'moderate' score range (Figure 12). In contrast, the Macroalgae indicator score has declined with increased cover of macroalgae continuing to put a downward pressure on coral community recovery (Figure 12).

Ultimately, the Reef level coral community condition reflects large-scale averages and overall responses of coral communities exposed to varied past and ongoing pressures. The following sections explore results at finer spatial resolution. However, what is clear from the Reef-level disturbance time-series is that, since 2005 inshore reefs have been exposed to multiple disturbance events, the impacts of which 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 almost half (44%) of all coral cover losses on inshore reefs (Figure 13, 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 been documented to have caused the greatest losses.

When interpreting Figure 13 it is important to note that the past biennial sampling designs of both the MMP and LTMP 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 13). 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.

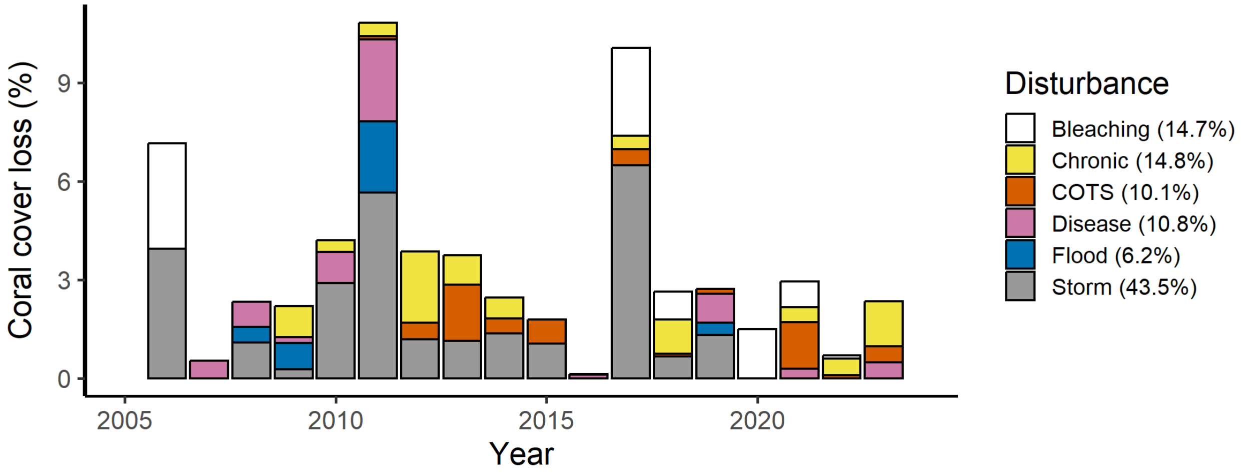

Figure 13 Hard coral cover loss by disturbance type across the inshore Reef. Length of bars represents the mean loss of cover across all reefs in each year. Colours represent the identified cause of cover loss. COTS = crown-of-thorns starfish

Thermal bleaching events have contributed to 14.7% of the coral cover losses since 2005. High water temperatures causing bleaching and subsequent loss of coral cover occurred in 2006, 2017, 2020 and, to a lesser extent, 2022 (Figure 13, Table A6). At many of the reefs exposed to marine heatwave conditions in 2020 corals were bleached at the time of survey in 2020, and the loss of coral cover observed in 2021 has been attributed to the longer-term impacts that killed or reduced corals growth after surveys in 2020. 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 13, Table A6), their potential impact has been reduced by the removal of starfish by the Reef Authority's Crown-of-thorns Starfish Control Program (Table 9). 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.

Loss of corals from direct exposure to low salinity flood waters has been limited to 2 m depths on reefs closest to rivers during major flood events (Table A6). 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.

The losses of coral cover attributed to disease and chronic pressures (25.6%, Figure 13) are considered to reflect the impacts of poor water quality. 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. Elevated levels of nutrients and fine organic sediments may 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), and potentially magnify the effects of heat stress events (Wiedenmann *et al.* 2013, Fisher *et al.* 2019, Cantin *et al.* 2021, Brunner *et al.* 2021).

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).

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 slight decrease in the Coral Index scores between 2022 and 2023 driven by declines in Macroalgae and Cover change scores. The relatively stable condition observed since 2016 (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, as detailed in the following sections. Even though the Cover change score has decreased to moderate in 2023, the Coral cover score remains in the ‘good’ range. At the regional level, no indicator scores have fallen below moderate levels since 2014.

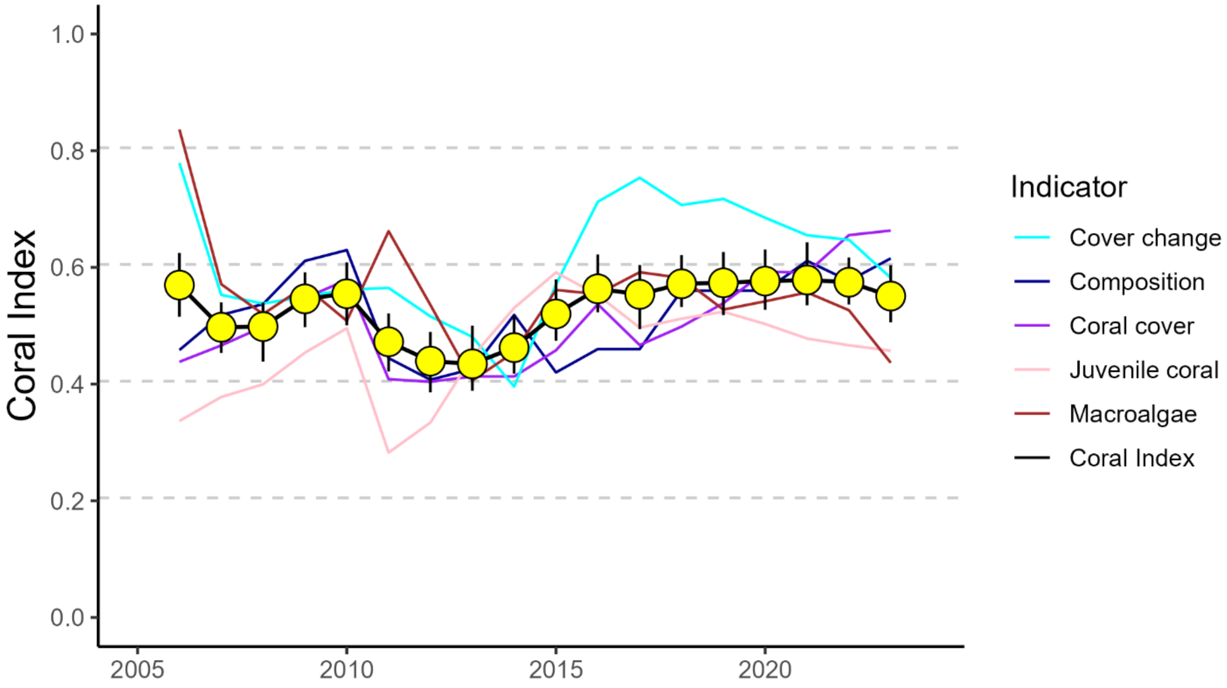

Figure 14 Trends in Coral Index and indicator scores for the Wet Tropics region. Coral Index scores are coloured by report card category: yellow = ’moderate’. Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at individual reefs.

4.3.2 Barron–Daintree sub-region

Coral communities remain in ‘moderate’ condition but have steadily improved since 2019 (Figure 15). A low point in Coral Index scores was recorded in 2014 following: an outbreak of coral disease in 2011, predation by crown-of-thorns starfish in 2012 and 2013 and then damage attributed to cyclone Ita in April 2014 (Figure 16). Since then, recovery of coral communities was interrupted by high water temperatures causing coral bleaching in 2017 (Figure 16c) and, at 2 m depth at Snapper South, exposure to floodwaters and cyclone Owen in 2019 (Figure 16, Figure 17, Table A6).

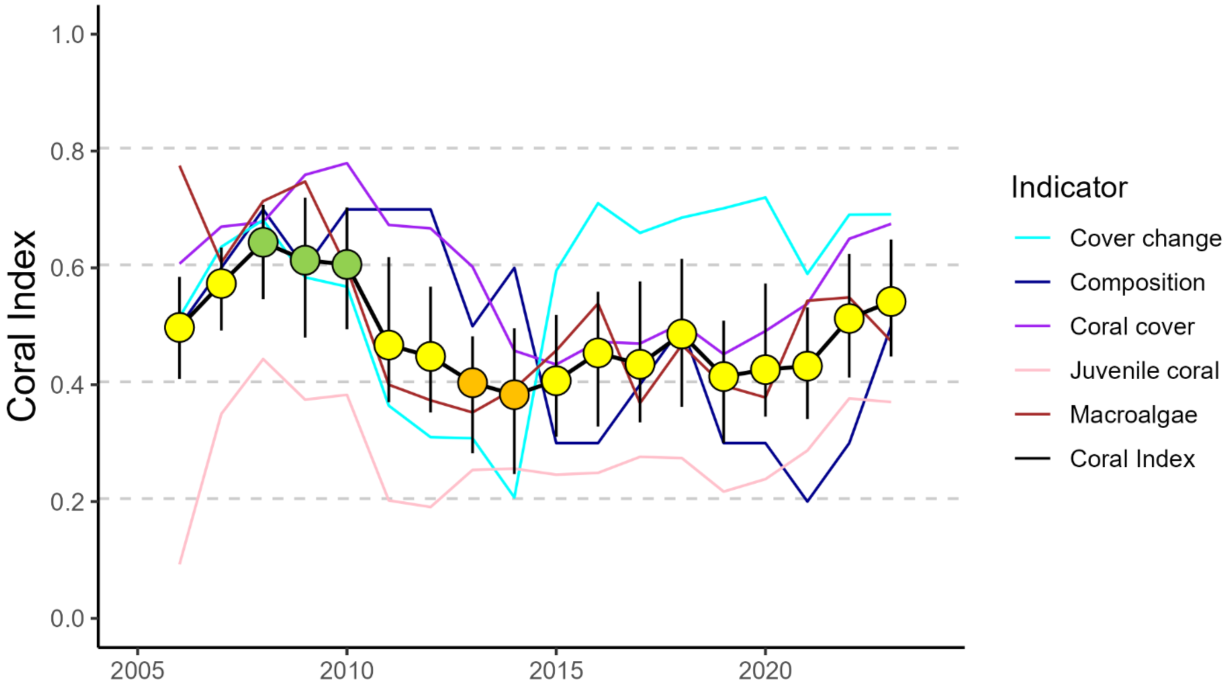

Figure 15 Trends in Coral Index and indicator scores for the Barron–Daintree sub-region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = ‘poor’, yellow = ’moderate’ and green = ’good’. Error in Coral Index scores were derived from bootstrapped distributions of indicator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

Most indicators have markedly improved since 2019 (Figure 15, Table 10). The greatest improvement was for Coral cover, with only the Macroalgae score at 2 m depth (Table 10) showing a consistent decline.

Table 10 Coral Index and indicator score comparisons in the Barren–Daintree sub-region. 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	-0.29	0.88	-0.13	0.61	-0.42	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	0.96	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	0.09	0.68	0.19	0.97	0.18	0.83	-0.06	0.64	-0.03	0.53	0.17	0.73

No acute pressures impacted reefs in the Barron–Daintree sub-region in 2023 (Figure 16e).

The Coral cover indicator score was categorised as ‘good’ (0.68, Table A7, Figure 15), having increased from a low in 2014 (Figure 17a). The recovery of coral cover was briefly interrupted in 2019 by floodwaters that reduced coral cover at Snapper South (2 m) and crown-of-thorns starfish that reduced coral cover at Low Isles (Figure 16, Figure 17a, Figure A1). Consistent among reefs in 2023 was higher cover of the hard coral *Acropora* compared to 2019 levels (Figure A1).

The Cover change indicator remained ‘good’ (0.69, Table A7) after transitioning from ‘moderate’ in 2022, with recent recovery of hard coral cover exceeding modelled predictions at all reefs in the region (Table A7).

The Composition indicator increased to ‘moderate’ in 2023 (0.5, Table A7, Figure 15). This result largely reflected the reinstatement of *Acropora* within the communities at each reef in the region.

The Macroalgae indicator remains ‘moderate’, although scores for this indicator had declined relative to 2019 levels at 2 m depth (Table 10, Figure 15). The score for this indicator varied greatly between sites and depths with Low Isles and Snapper South (2 m) categorised as ‘very good’, countered by Snapper North (2 m) and Snapper South (5 m) categorised as ‘very poor’ (Table A7). Macroalgae cover at Snapper North (2 m) has remained extremely high since 2014 (Figure A1).

The Juvenile coral indicator remains in the ‘poor’ category (0.37, Table A7, Figure 15), although this has been improving since 2019 at 2 m depth (Table 10). Between 2022 and 2023 there was a substantial increase in *Acropora* juveniles in 2 m depth at Snapper South, in contrast, numbers of juveniles at Low Isles show a general decline across most genera (Figure A1).

In 2023, the concentration of NOx was the only sampled water quality parameter that exceeded guideline values (Gruber *et al.* 2024). The concentration of NOx has shown no long-term trend since the redesign of the sampling program for water quality in 2015 with values in 2023 similar to those observed in 2015 (Figure A10, Gruber *et al*. 2024). The short-term water quality index remained ‘good’, and similar to values seen since 2019 (Figure A10a). Over the period 2019–2023, 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).

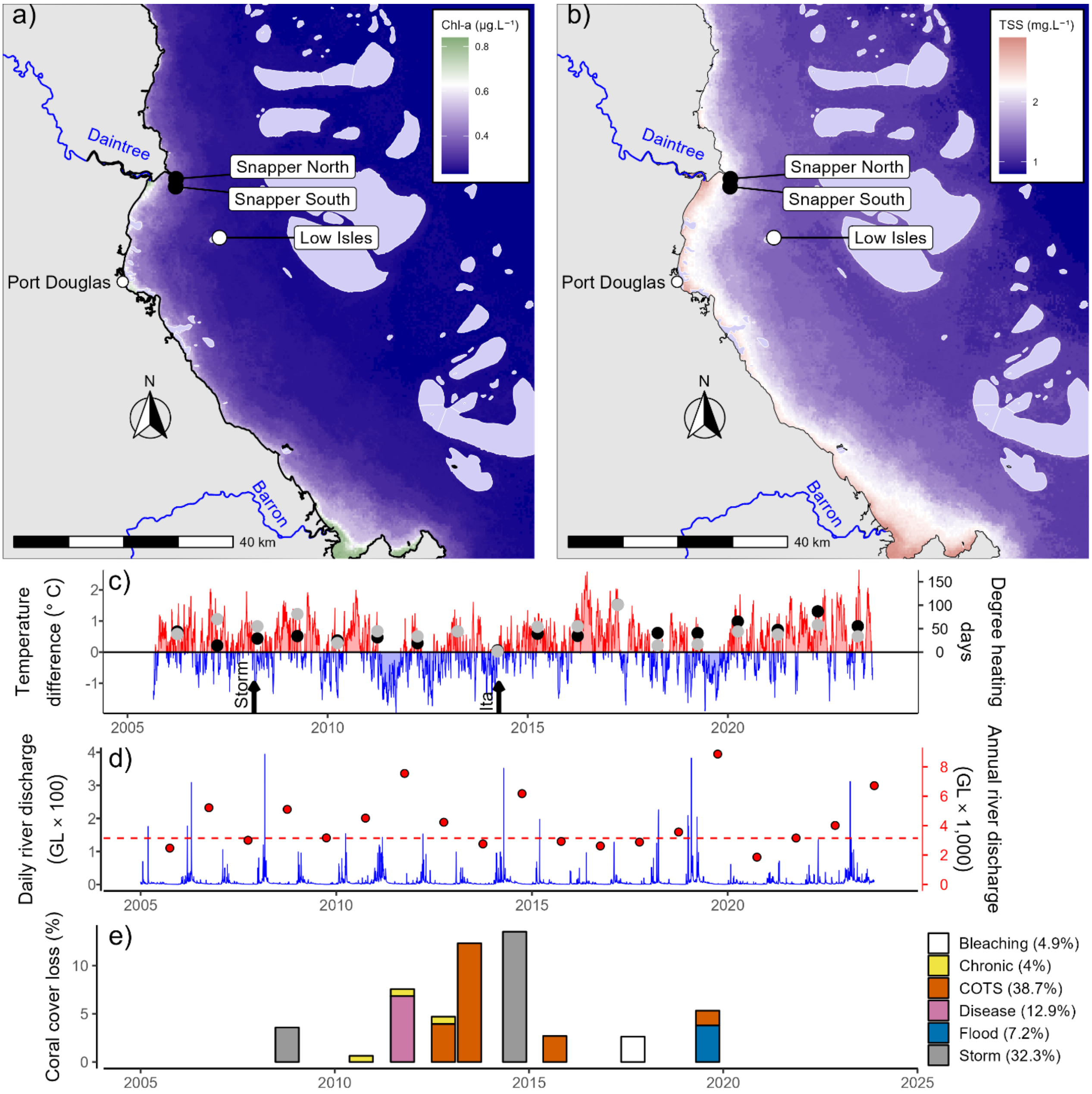

Figure 16 Barron–Daintree sub-region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) median wet season Chl *a* and b) median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2019–2023, 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 DHD over the summer period (1 December – 31 March) as reported by BoM (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.

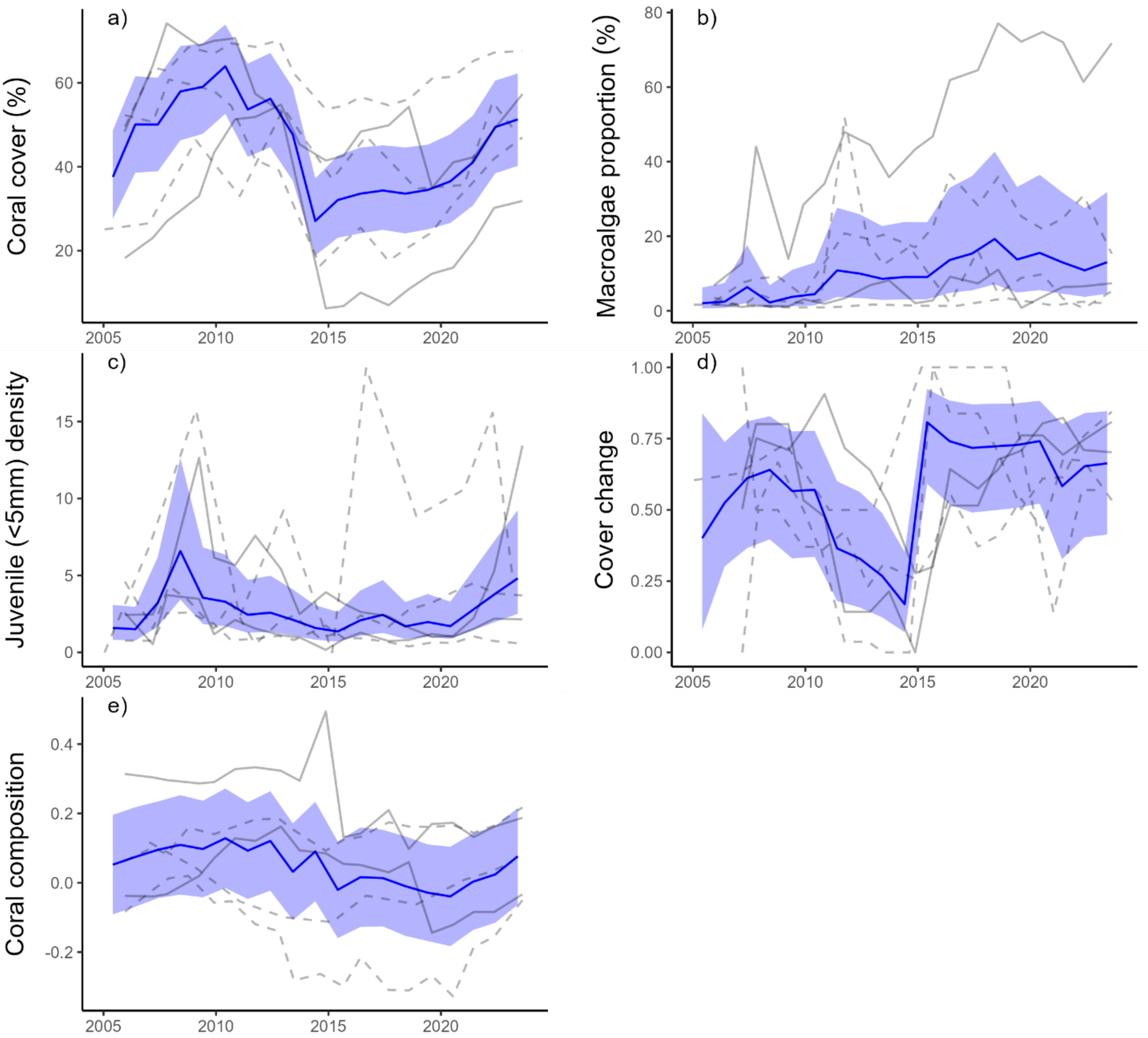

Figure 17 Barron–Daintree sub-region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.3.3 Mulgrave sub-region

The 2023 Coral Index score was categorised as ‘moderate’, having declined slightly since 2021 (Figure 18). A low point in the Coral Index occurred in 2012 following severe damage to coral communities caused by cyclone Yasi, and high levels of coral disease, with recovery occurring through to 2016 (Figure 18, Figure 19e). Since 2016 the Coral Index has fluctuated around the threshold between ‘moderate’ and ‘good’ scores (Figure 18). The most consistent declines contributing to the lower score in 2023 have been for the Macroalgae indicator and to a lesser extent the Cover change indicator (Table 11).

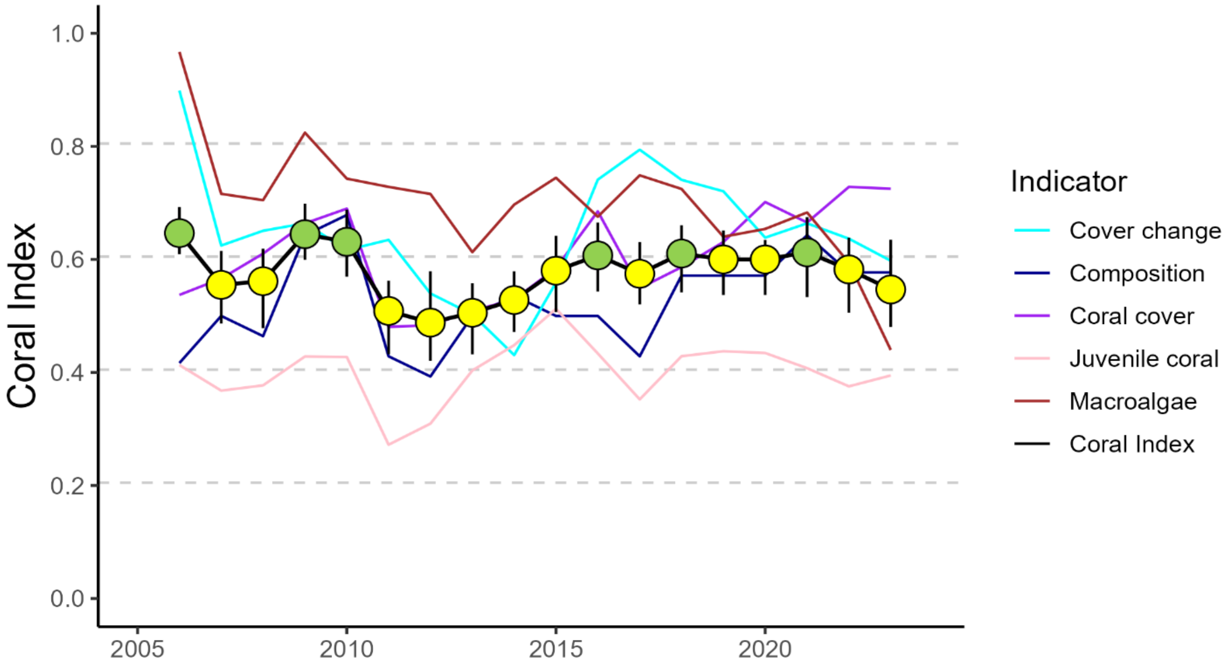

Figure 18 Trends in Coral Index and indicator scores for the Johnstone Russell–Mulgrave sub-region. Coral Index scores are coloured by Reef Water Quality Report Card categories: yellow = ’moderate’ and green = ’good’. Error in Coral Index scores were derived from bootstrapped distributions of indicator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

Table 11 Coral Index and indicator score comparisons in the Johnstone Russell–Mulgrave sub-region. 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	0.67	0.14	0.77	-0.10	0.73	0.16	0.82	0.22	0.71	-0.06	0.54
2016 to 2023	2	-0.08	0.66	0.04	0.57	-0.30	0.71	0.05	0.64	-0.21	0.69	0	0.50
	5	-0.06	0.66	-0.03	0.58	-0.18	0.69	-0.04	0.56	-0.15	0.68	0.07	0.59

The trend in the Coral Index in this sub-region reflects the impact, and subsequent recovery, of coral communities following cyclones Tasha and Yasi in 2011 (Figure 19c, e). These cyclones caused substantial damage to coral communities not only through the physical action of these large storms, but also the associated flooding that exposed 2 m sites at High West to sufficiently low salinity waters that killed some corals (Figure A2, Table A6). The effects of cyclones were further compounded by the increased prevalence of disease in 2011, which coincided with high discharge from local rivers (Figure 19e,d, Figure A7). Fitzroy Island, which had escaped serious damage from the cyclones, lost a substantial proportion of hard coral cover to disease (Figure 19). Fitzroy East lost between 60% (2 m) and 42% (5 m) of hard corals, predominantly *Acropora* (Table A6, Figure A2).

The plateau in recovery of the coral communities in recent years has been influenced by ongoing predation of corals by crown-of-thorns starfish (Figure 19e, Figure A8) and thermal bleaching in 2017. In 2023, crown-of-thorns starfish were at outbreak levels at Franklands East. Although no crown-of-thorns starfish were observed at High Island, the presence of feeding scars and outbreak densities in 2022 were considered in attributing observed coral losses in 2023 to these starfish (Figure A8,Table A6).

Discharges from local rivers were slightly above median level over the 2022–2023 water year (Table A5); however, peak flows were relatively low (Figure 19d).

The Coral cover indicator score was categorised as ‘good’ after remaining relatively stable between 2022 and 2023 (Figure 18, Table A7). Coral cover increased at Fitzroy Island and Franklands East due primarily to increased cover of *Acropora* (Figure A2). Coral cover also increased at Frankland West (2 m) where there was a modest increase in cover of both hard and soft corals (Figure A2). Declines at High Island were mostly due to reduced cover of *Acropora* spp., attributed to crown-of-thorns starfish predation. At Franklands West (5 m) the coral community is dominated by large stands of branching and sub-massive *Porites*, with cover variable among years as dense mats of red macroalgae expand and contract in the spaces between the coral’s branches and nubbins, or sections of colonies cleave off (Figure A2).

The Cover change indicator score was categorised as ‘moderate’ (0.6, Table A7) in 2023 after declining slightly from ‘good’ in 2022 (Figure 18). This is the first time the score has dipped below the ‘good’ range since 2015; however, change in this indicator has been inconsistent among reefs and overall hard coral cover regionally is continuing to recover at expected rates (Figure 18, Figure 20d).

The Composition indicator has remained ‘moderate’ in 2023 with little change from 2022 (Figure 18).

The Macroalgae indicator score has declined in recent years but remains ‘moderate’ in 2023 (Figure 18, Table A7). Across the region, the cover of the persistent brown macroalgal 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 2023 at High West, Franklands East (5 m) 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’ having varied around the boundary between ‘moderate’ and ‘poor’ since 2016 (Figure 18).

In 2023, the concentrations of NOx and turbidity were the only sampled water quality parameters that exceeded guideline values (Gruber *et al.* 2024). The concentration of NOx has however shown a declining trend since the redesign of the sampling program for water quality in 2015 (Gruber *et al.* 2024). The short-term water quality index remained ‘moderate’ having remained close to the boundary of ‘good’ since 2019 (Figure A11a). Over the period 2019–2023, 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).

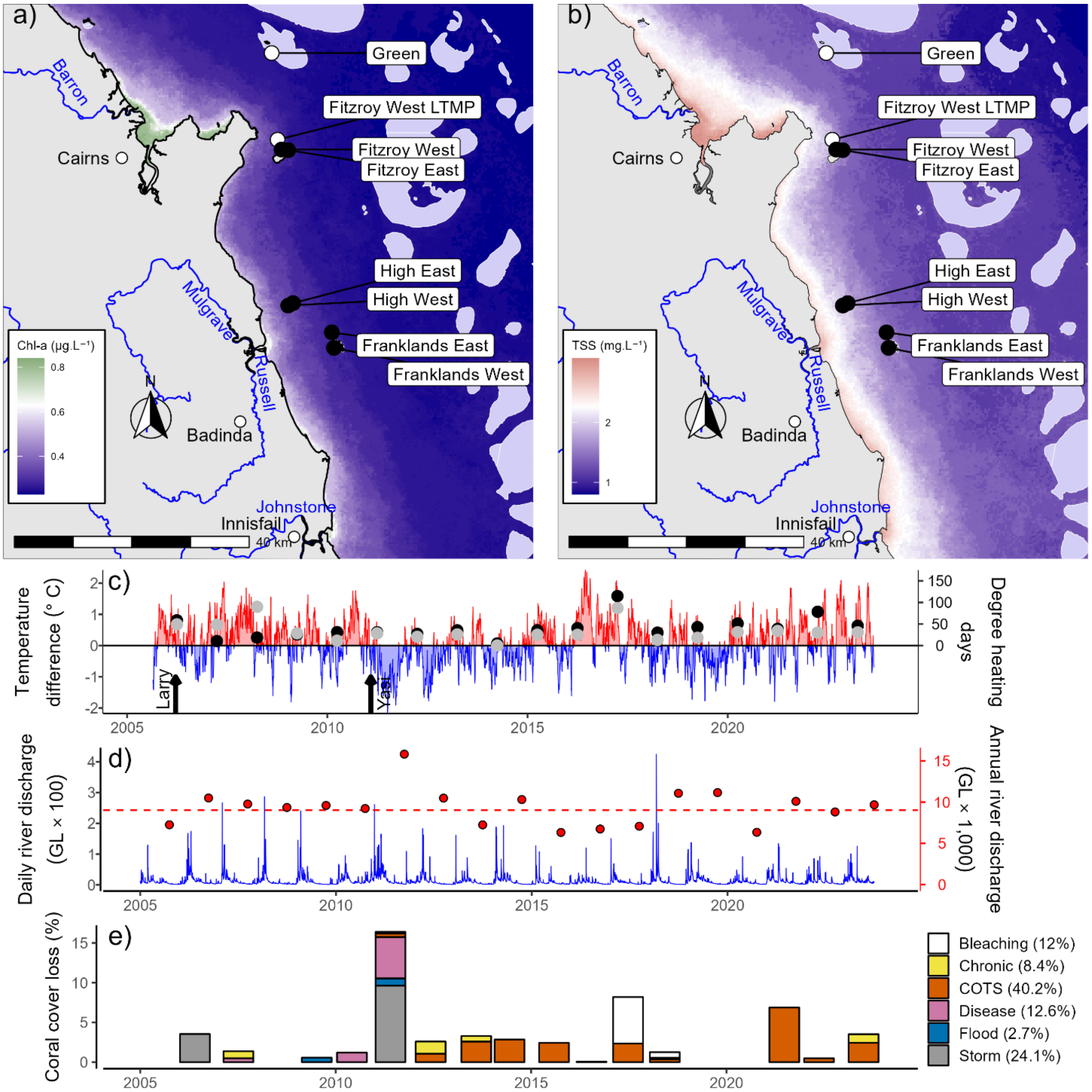

Figure 19 Johnstone Russell–Mulgrave sub-region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) median wet season Chl *a* and b) median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2019–2023, 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 DHD over the summer period (1 December – 31 March) as reported by BoM (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.

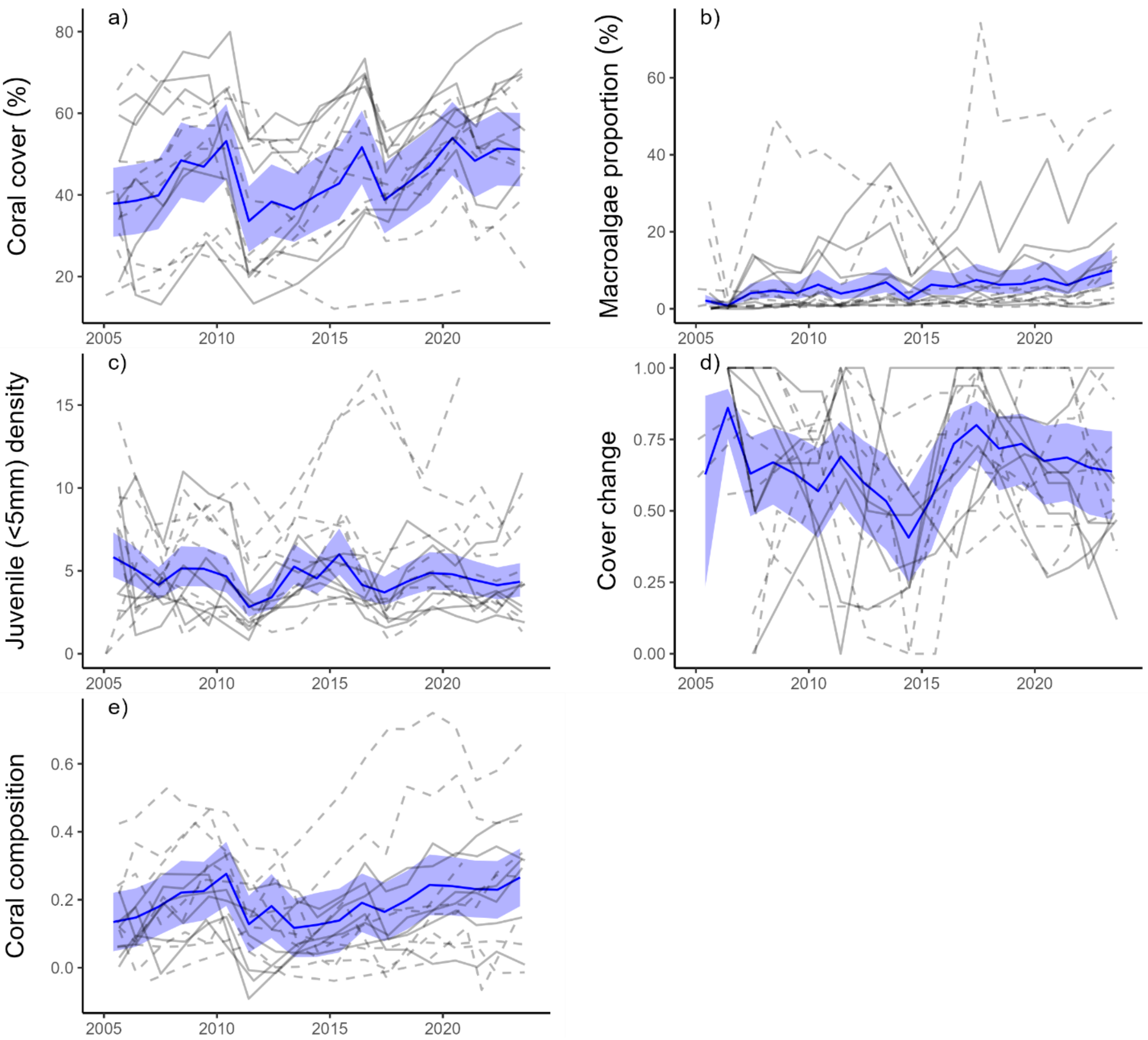

Figure 20 Johnstone Russell–Mulgrave sub-region indicator trends. a– e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.3.4 Herbert–Tully sub-region

The Coral Index was ‘moderate’ in 2023 (Figure 21). Although the Coral Index has declined from a ‘high’ in 2020, Coral cover has continued to increase since 2012 (Figure 21). The current downward trend is driven by moderation of both Juvenile coral and Cover change scores that remain in the ‘good’ and ‘moderate’ range, respectively (Table 12, Figure 21).

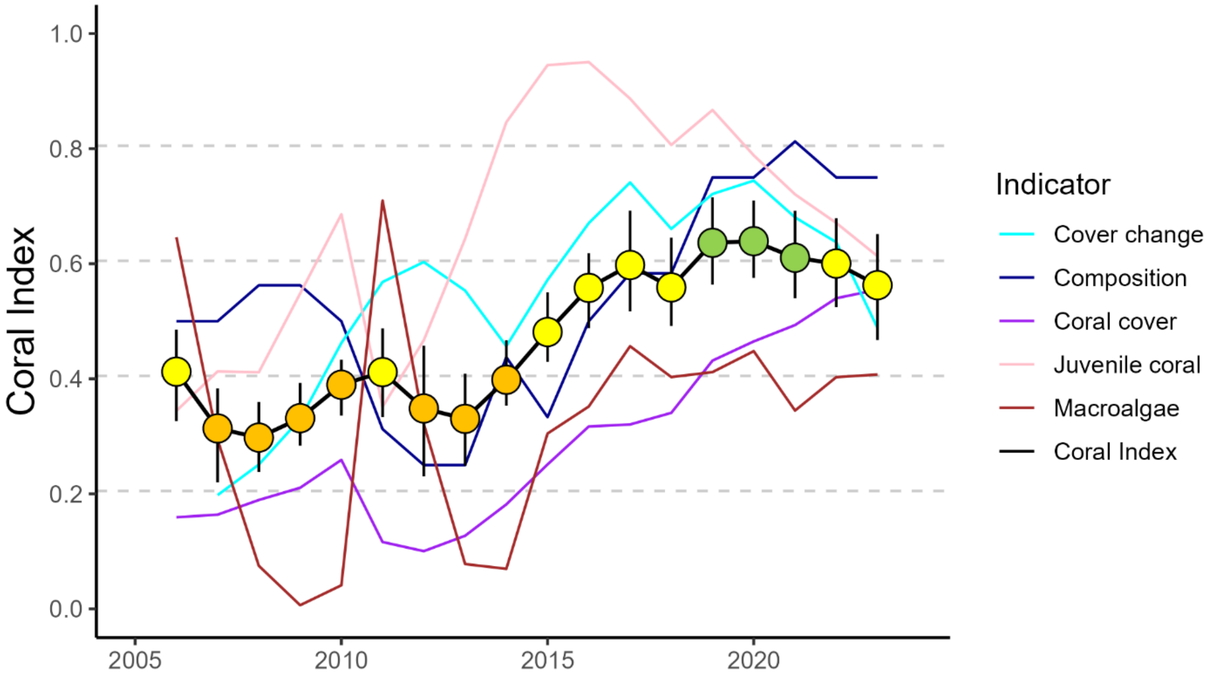

Figure 21 Trends in Coral Index and indicator scores for the Herbert–Tully sub-region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = ‘poor’, yellow = ’moderate’ and green = ’good’. Error in Coral Index scores were derived from bootstrapped distributions of indicator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

Table 12 Coral Index and indicator score comparisons in the Herbert–Tully sub-region. 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	0.33	0.94	-0.38	0.93
	5	0.13	0.80	-0.07	0.66	0.60	0.89	-0.07	0.56	0.30	0.74	-0.13	0.70
2011 to 2014	2	0.02	0.65	0.06	0.89	-0.67	0.92	0.52	0.93	-0.05	0.61	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 2023	2	0.0	0.50	0.11	0.95	0.03	0.70	-0.20	0.97	-0.14	0.78	0	NA
	5	-0.13	0.84	0.07	0.81	-0.12	0.75	-0.15	0.79	-0.37	0.90	0	NA

Since 2020, minor losses in hard coral cover at some reefs have been attributed to high water temperatures that caused coral bleaching in 2020 (Figure 22c, e) and above median levels of coral disease (Figure 22e, Figure A7).

The Coral cover indicator score has been steadily increasing since 2011 and remained categorised as ‘moderate’ in 2023 (Figure 21, Table 12). In 2023, there was an increase in Coral cover at all sites except Dunk South (5 m) (Figure 23a). Of the three reefs monitored since 2005 it is only at Dunk South (5 m) that coral cover in 2023 remains below that observed prior to the impacts of cyclone Larry in 2006 (Figure A3). Recovery of coral cover mostly reflects increased cover of the genera *Acropora* and *Montipora*, although *Turbinaria* cover has also increased at Dunk North (5 m) in particular (Figure A3).

The Cover change score changed from ‘good’ in 2022 to ‘moderate’ in 2023 after a period of decline from 2020 to 2023 that was most pronounced at the 5 m depth (Figure 21, Figure 23d, Table 12, Table A7). In 2023, scores were ‘poor’ for Barnards (5 m), Dunk South (5 m) and Bedarra (2 m) (Table A7). During the period of 2020 to 2023, and especially in 2023, levels of disease were above median levels (Figure A7). Although disease was not categorised as an acute disturbance, 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 been ‘good’ since 2019 except for a brief spike to ‘very good’ in 2021 (Figure 21).

The Macroalgae indicator has slightly increased to become 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 maximum values of one at both depths at Barnards (Table A7). At reefs with a value of zero, the macroalgae community is dominated by brown algae of the family Sargassaceae (Table A11).

The Juvenile coral indicator remains categorised as ‘good’ although it has been declining since 2014 and is now 0.61, right on the threshold of ‘moderate’ (Table 12, Table A7, Figure 23c). Bolstering scores for Juvenile coral were strong cohorts of *Turbinaria* (Family: Dendrophylliidae), which recruited in the years following cyclone Yasi, as these have either died or grown beyond the juvenile size classes contributing to the reduced densities of juvenile corals (Figure 23c, Figure A3).

In 2023, the concentrations of NOx, PN and turbidity exceeded guideline values (Gruber *et al.* 2024). The concentration of NOx has however shown a declining trend since the redesign of the sampling program for water quality in 2015, in contrast PN has increased over this period (Figure A12, Gruber *et al.* 2024). The short-term water quality index remained ‘moderate’ (Figure A12a). Over the period 2019–2023, 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 22a, b, Table A8).

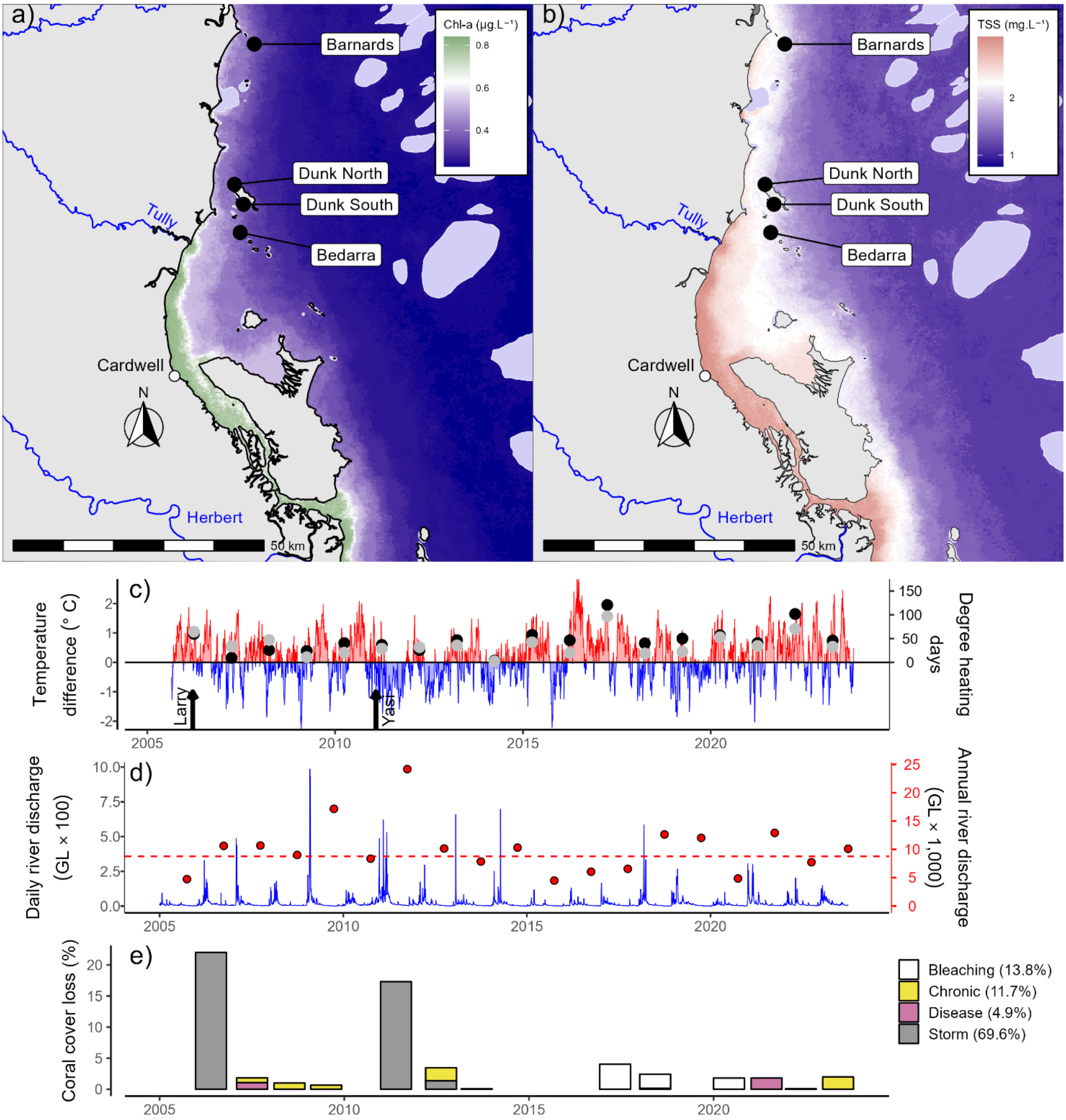

Figure 22 Herbert–Tully sub-region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) median wet season Chl *a* and b) median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2019–2023, white 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 DHD over the summer period (1 December – 31 March) as reported by BoM (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.

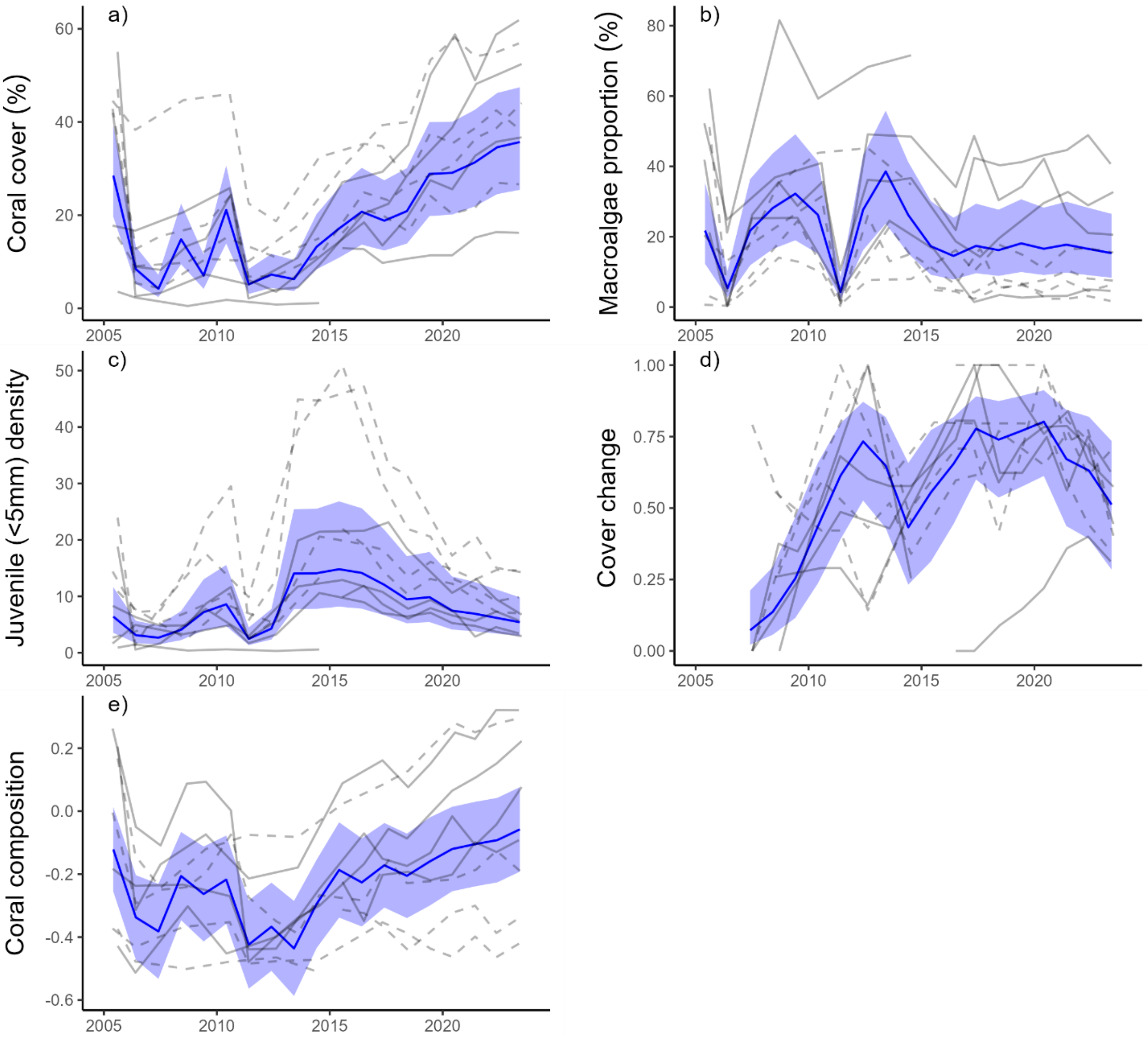

Figure 23 Herbert–Tully sub-region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.4 Burdekin region

The Coral Index remained below a high point observed in 2020 but was still within the ‘moderate’ condition range (Figure 24). Only the Coral cover indicator has improved since 2020 and was at the highest level recorded since 2006 (Table 13, Figure 24). Scores for the remaining indicators were all lower than observed in 2020, with the most consistent declines occurring at 2 m depths for Macroalgae and 5 m depths for Juvenile coral (Figure 24, Table A7). In 2023, Macroalgae and Juvenile coral scores were categorised as ‘poor’ (Figure 24, Table A7).

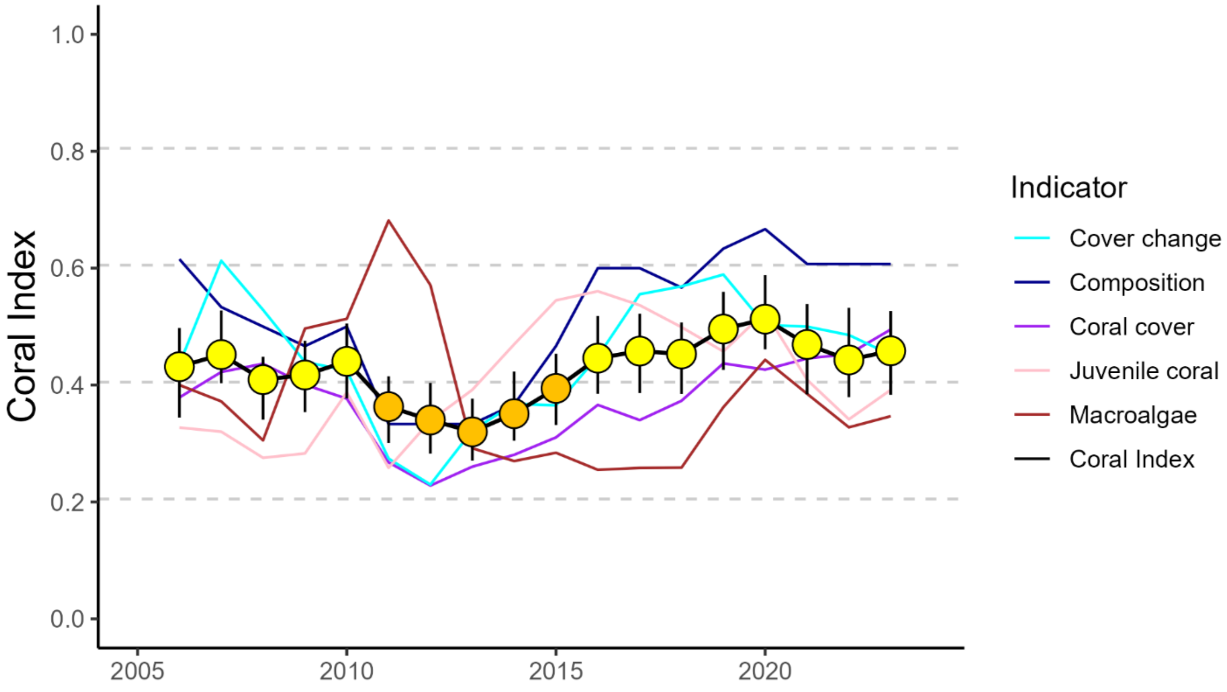

Figure 24 Trends in Coral Index and indicator scores for the Burdekin region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = ‘poor’, yellow = ’moderate’. Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at individual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

Table 13 Index and indicator score comparisons in the Burdekin region. 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.

		Coral Index		Coral cover		Macroalgae		Juvenile coral		Cover change		Composition	
Period	Depth	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 2023	2	-0.07	0.83	0.10	0.99	-0.26	0.73	-0.06	0.70	-0.04	0.54	-0.08	0.67
	5	-0.06	0.75	0.06	0.84	-0.03	0.59	-0.18	0.77	-0.06	0.57	-0.06	0.64

There were no acute disturbances in the region over the summer of 2022–23; however, annual river discharge for 2023 was above the long-term median discharge for the region (Figure 25d). The rate of increase in coral cover in 2022–2023 was low, suggesting there were chronic pressures influencing corals (Figure 25e). The most recent acute pressures to impact coral communities were marine heat wave conditions that caused coral bleaching in 2017, 2020 and to a lesser extent 2022.

The combined impacts of these marine heat waves stalled the recovery that had occurred through to 2020 (Figure 24) after severe impacts caused by cyclone Yasi in 2011 (Figure 10, Figure 25c,e, Table A6).

The Coral cover indicator score remained categorised as 'moderate', having continued to increase since 2013 (Figure 24, Table 13). In 2023, hard coral cover had increased at both depths at Palms East, Lady Elliot, Havannah North, Pandora North, and Havannah, and for 5 m depth at Magnetic (Figure A4). Increases were attributed to recovery of *Acropora, Montipora, Goniopora* and *Alveopora*, and Merulinidae (Figure A4).

The regional rate of increase in hard coral cover over the last four years remained within modelled expectations as reflected by the 'moderate' score for the Cover change indicator score (Figure 24, Table A7). However, the rate of hard coral recovery was 'poor' at several reefs, including both depths at Palms East, the 2 m depths at Pandora, Lady Elliot and Magnetic, and the 5 m depths at Havannah and Palms West.

The Composition indicator score is 'good' and has remained relatively stable since 2021 (Figure 24, Table A7).

The Macroalgae indicator has continued to decline and remains 'poor' (Figure 24, Table A7). Very poor scores were recorded at Havannah, Havannah North, Magnetic and Pandora North, and the 2m depths only at Lady Elliot and Pandora (Table A7, 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', although the scores were highly variable among reefs ranging from 0.16 to 0.81 (Figure 24, Figure 26c, Table A7). Juvenile density has increased at Palms West (5 m), and both depths at Palms East, Lady Elliot, Pandora, Havannah and Magnetic (Figure A4). Decreases were observed at Palms West (2 m) and Pandora North and were greatest at Havannah North (Figure A4). Influential in the regional decline in juvenile densities at 5 m depths in recent years has been declines in 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 2023, the concentrations of NOx and turbidity were the only measured water quality parameters to exceeded guideline values (Gruber *et al.* 2024). Both NOx and turbidity have, however, shown a declining trend since the redesign of the sampling program for water quality in 2015 (Figure A13, Gruber *et al.* 2024). The short-term water quality index remained 'moderate' (Figure A13a). Over the period 2019–2023 wet-season Chl *a* concentrations were below guideline values at the coral monitoring sites, while TSS concentrations exceeded guidelines at Magnetic and Lady Elliot (Figure 25 a, b, Table A 8).

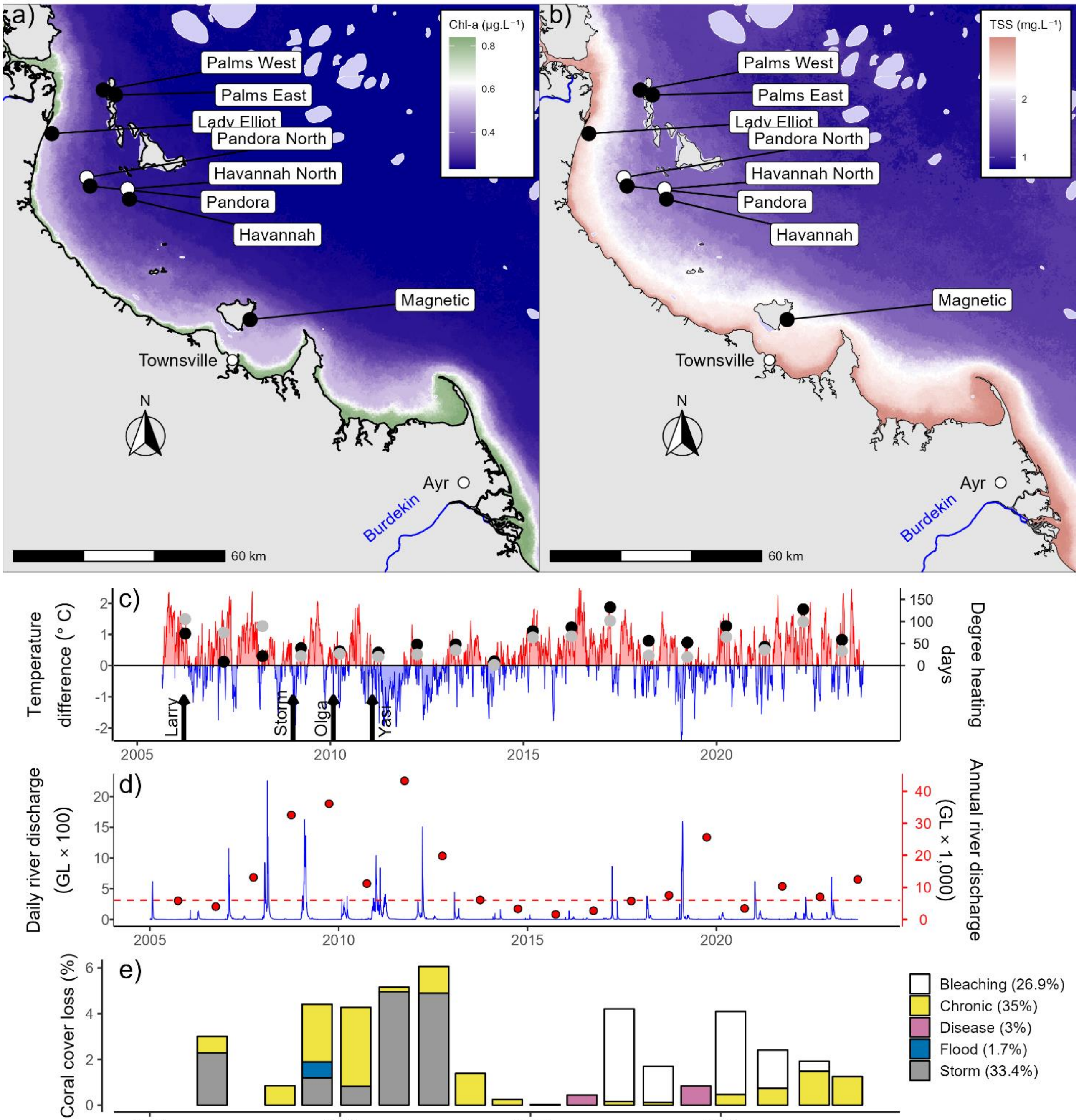

Figure 25 Burdekin region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) median wet season Chl *a* and b) median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2019–2023, white breaks 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 DHD over the summer period (1 December – 31 March) as reported by BoM (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.

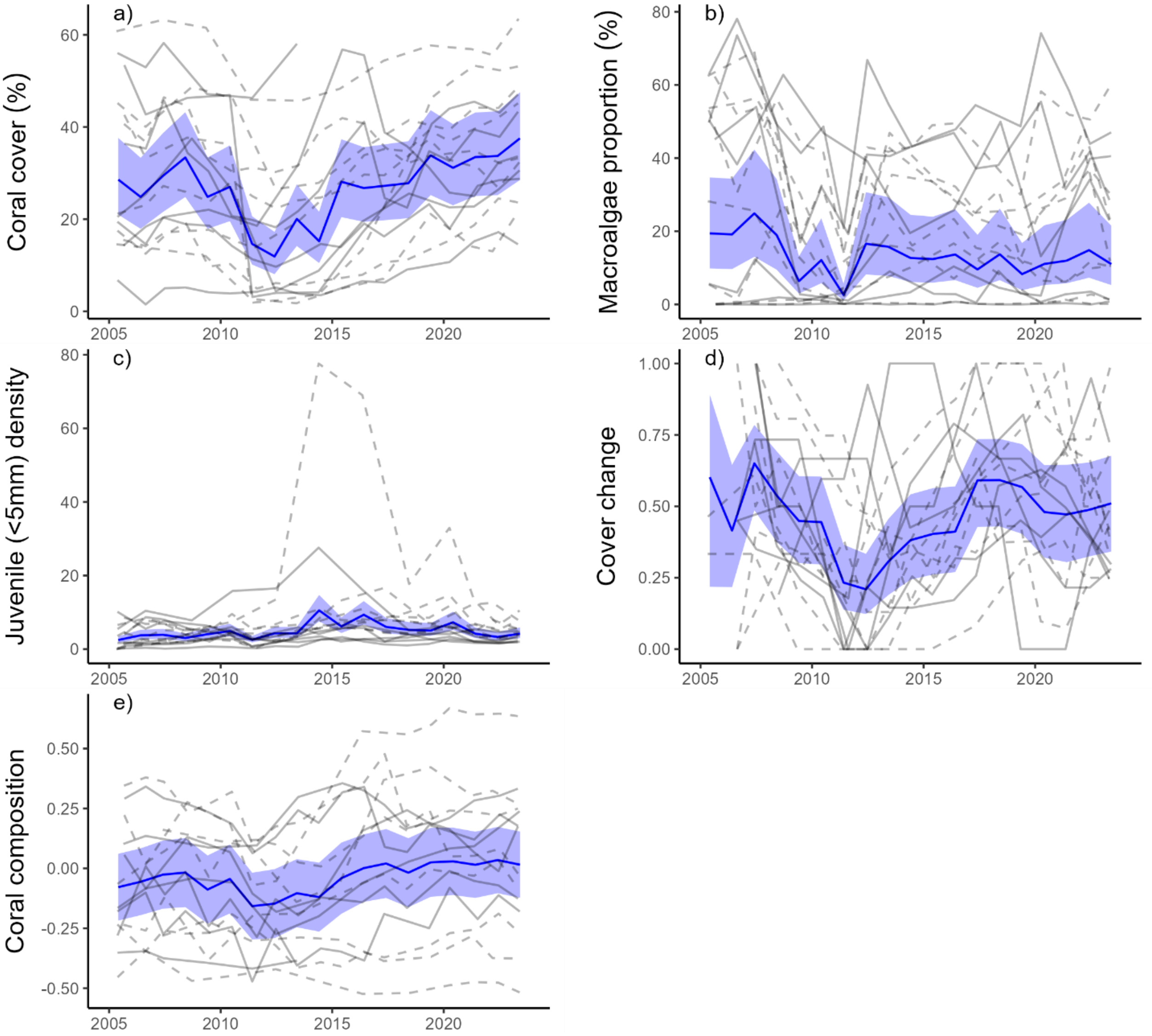

Figure 26 Burdekin region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.5 Mackay–Whitsunday region

In 2023, the Coral Index score remained ‘poor’ (Figure 27). In 2023, the Juvenile coral indicator continues to improve but remains within the ‘moderate’ score range. Coral cover scores have also improved (Figure 27) but only at the 2 m depth (Table 14). All other indicator scores were in ‘poor’ condition (Figure 27). These changes represent the early signs of recovery following the decline in the Coral Index, and the individual indicators, over the period 2016–2020 in response to the severe impact of cyclone Debbie in 2017 (Table 14, Figure 28e).

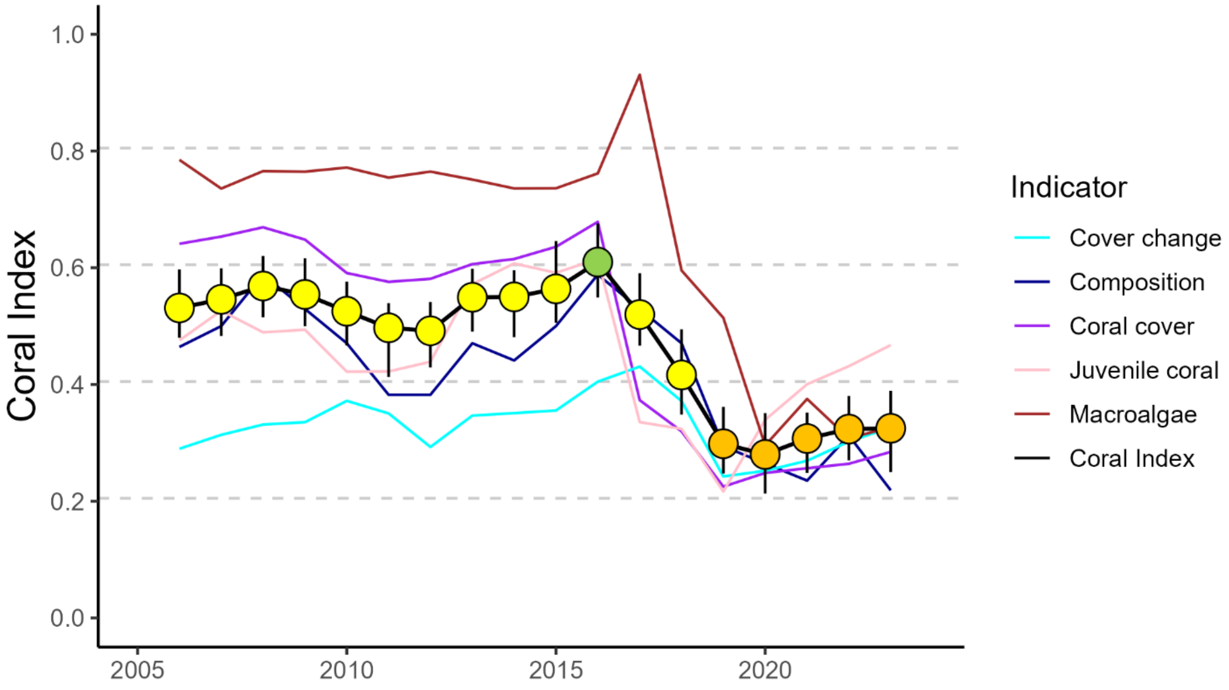

Figure 27 Trends in Coral Index and indicator scores for the Mackay–Whitsunday region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = ‘poor’, yellow = ’moderate’, green = ’good’. Error in Coral Index scores were derived from bootstrapped distributions of indicator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

Table 14 Coral Index and indicator score comparisons in the Mackay–Whitsunday region. 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	-0.43	0.83	-0.28	0.86	-0.06	0.57	-0.25	0.76
2020 to 2023	2	0.01	0.55	0.04	0.76	-0.04	0.53	0.10	0.84	-0.01	0.51	-0.07	0.65
	5	0.07	0.67	0.03	0.66	0.17	0.67	0.14	0.72	0.10	0.64	-0.06	0.55

There were no acute disturbances in the period 2022–2023. Summer temperatures in early 2023 were below those likely to cause severe coral bleaching (Figure 9, Figure 10). The 2023 water year was the first year since 2019 that river discharges exceeded the regional median (Figure 28d), with the O'Connell River recording annual discharge at twice the long-term median level (Table A5). However, there were no signs of flood impact among the reef communities. Disease levels remained below the long-term median level and unchanged from the previous year (Figure A7). There were no crown-of-thorns starfish observed and very few *Drupella* reported in the period 2022–2023 (Figure A8, A9).

The combined cover of hard and soft corals has gradually increased since the impacts of cyclone Debbie (Figure 27, Figure 29a), with recovery at 2 m depths more consistent than at 5 m depths (Table A7). Coral cover remains in the 'poor' or 'very poor' range at most reefs (Table A7). The highest coral cover was at the 2 m depth at Shute Harbour, where the cover of hard corals (predominantly *Acropora*, Figure A5) has steadily increased and entered the 'very good' range in 2023 (Table A7). 'Moderate' levels of coral cover were observed at Border and at the 5 m depths at Dent and Shute Harbour; all reefs that were spared severe losses in coral cover caused by cyclone Debbie. Elsewhere, coral cover remained low.

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 (Figure 27) and they have not consistently improved through to 2023 (Table 14), highlighting the ongoing slow recovery of hard corals across the region. Scores were, however, variable among reefs (Figure 29) with 'moderate' or 'good' scores recorded at Shute Harbour, Daydream and Hayman in 2023 (Table A7).

Reductions in the Composition score following cyclone Debbie reflect the disproportionate loss of Acroporidae corals. In 2023, Composition scores among reefs remain low and variable (Figure 29e, Table A7).

Juvenile coral scores declined steeply following cyclone Debbie but are beginning to rebound (Figure 27, Figure 29c, Table 14). The abundance of juveniles has increased at all reefs since initial post cyclone Debbie observations in 2017 or 2018 (Figure 29). High densities of juvenile corals were observed at Hayman and Daydream where scores were in the 'good' to 'very good' range (Figure A5, Table A7). At Hayman, the genus *Acropora* is strongly represented among juvenile corals while at Daydream a more diverse assemblage of juvenile corals has recruited (Figure A5). Although the density of juvenile corals observed along transects in 2023 is close to, or above, values seen prior to cyclone Debbie (Figure 29c, Figure A5), this is not fully reflected in Juvenile coral scores (Figure 27) as the scores correct for the area of transects available to juvenile corals, and the available area remains relatively high as a result of lower coral cover (Figure 29a).

In 2020, the Macroalgae indicator score declined substantially relative to the levels observed prior to cyclone Debbie (Figure 27). Since 2020, the regional Macroalgae score has remained poor (Figure 27, Table 14). In 2023, Macroalgae scores were at minimum values of zero at both depths of Dent, Double Cone, Pine and Seaforth, and at 2 m depth at Daydream. In stark contrast were 'moderate' to 'very good' scores at other reefs (Table A7). Of note was the improvement of the Macroalgae score at Daydream (5 m) that contributed to a change in Coral Index category from 'poor' to 'moderate' (Table A7). This improvement was due to the almost complete loss of the dominant macroalgae *Lobophora* at that depth (Table A11). Macroalgae cover increased at Double Cone, Pine and Seaforth islands (Figure A5); however, these changes had no influence on indicator scores, which were at minimum values of zero in 2022 (Thompson *et al.* 2023).

In 2023, the concentrations of most water quality parameters exceeded guideline values with only PN and Chl *a* meeting the guidelines (Gruber *et al.* 2024). While the short-term water quality index remained 'moderate' it has trended up since a low point in 2017 (Figure A14a). All water quality parameters have shown an improving trend since 2015 (Gruber *et al.* 2024). Over the period 2019–2023, 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).

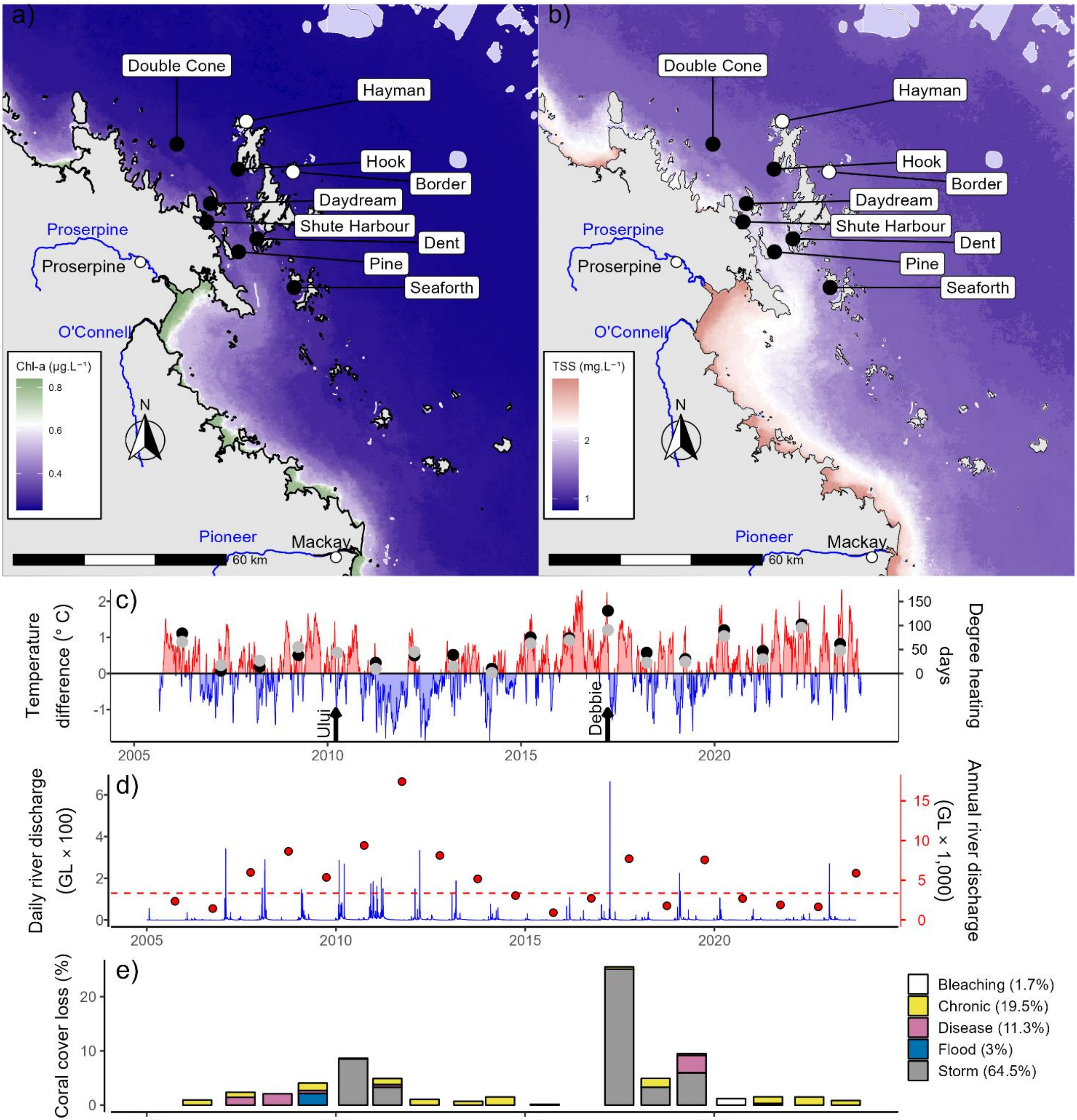

Figure 28 Mackay-Whitsunday region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) median wet season Chl *a* and b) median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2019–2023, white 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 DHD over the summer period (1 December – 31 March) as reported by BoM (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) break-down of hard coral cover loss by disturbance type; length of bars represents the mean loss of cover across all reefs.

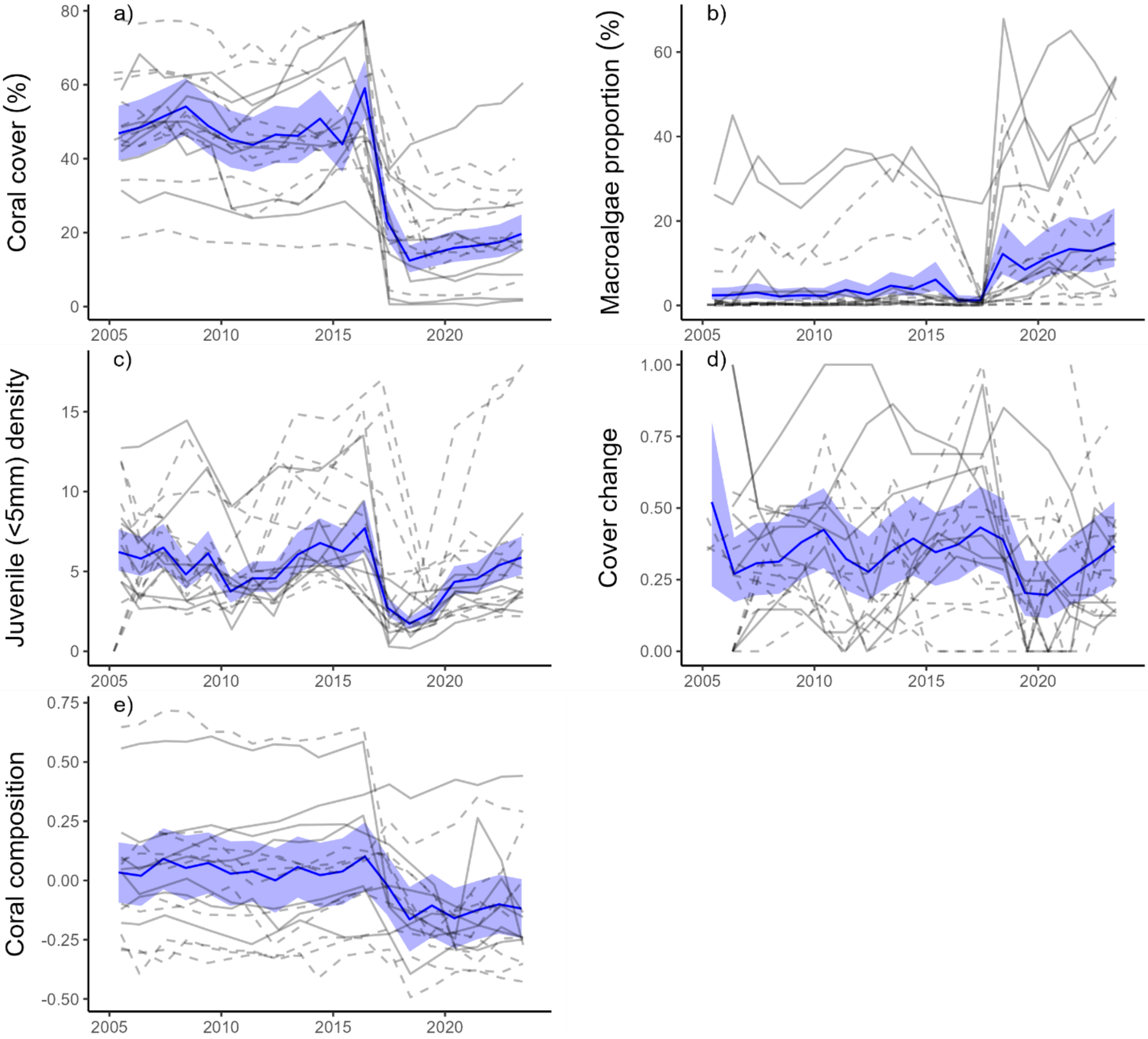

Figure 29 Mackay–Whitsunday region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.6 Fitzroy region

The Coral Index score remained ‘poor’ in 2023 with no improvement following a period of recovery from 2014 to 2020 (Figure 30, Table 15). Since 2020, scores for Juvenile coral and Macroalgae at 5 m depths have declined. Conversely, the Coral cover score has slowly improved since 2020 but remains within the ‘moderate’ range and this improvement was not consistent among reefs (Figure 30, Table 15). Since the impacts of floods and storms that drove the Coral Index to a ‘very poor’ low point in 2014, the Macroalgae score has remained ‘very poor’ while Cover change, Coral cover and Composition have all improved (Figure 30, Table 15).

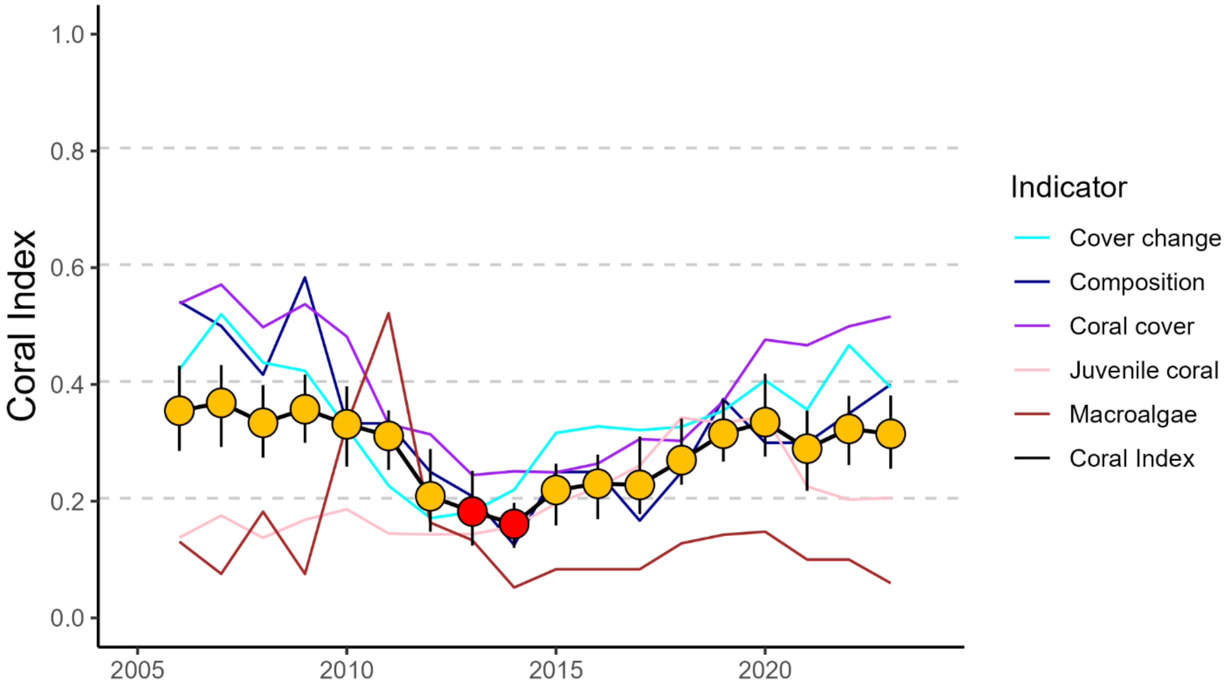

Figure 30 Trends in Coral Index and indicator scores for the Fitzroy region. Coral Index scores are coloured by Reef Water Quality Report Card categories: red =’ very poor’, orange = ‘poor’. Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at individual reefs.

Table 15 Coral Index and indicator score comparisons in the Fitzroy region. 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	0.99	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	0.70	0.0	0.51	-0.1	0.70	-0.16	0.73	-0.02	0.53	0.1	0.69

In the period 2022–2023 no acute disturbances impacted coral communities in the Fitzroy region. Summer temperatures were below the bleaching threshold (Figure 9, Figure 10, Figure 31c). Combined discharges from the Fitzroy River and Water Park Creek were only slightly above long-term median levels (Figure 31d, Table A5). There have been no crown-of-thorns starfish observed on the reefs in this region and very few *Drupella* were reported in the period 2022–2023 (Figure A8, Figure A9).

Increases in the combined cover of hard and soft corals occurred between 2014 and 2020 elevated the score for Coral cover into the ‘moderate’ range (Figure 30). While the score in 2023 was slightly higher than observed in 2020, changes in coral cover have been inconsistent among reefs (Figure 32a, Table 15). At 2 m depths, Coral cover scores have increased at most reefs since 2020, due primarily to increasing cover of *Acropora*, with two exceptions: Pelican, where coral cover remains very low and Middle where the cover of *Acropora* has declined (Figure A6). In contrast, at 5 m depth coral cover increased only at North Keppel (Figure A6)*.* Declines in *Acropora* cover at Barren (5 m) and Middle (2 m) in 2023 were due to disease (Figure A6, Table A6).

From 2014–2020, scores for Cover change also improved and have ocellated around the boundary between ‘poor’ and ‘moderate’ levels thereafter (Figure 30, Table 15). Cover change in 2023 was at the upper boundary of the ‘poor’ range, indicating slower than expected recovery of hard coral cover in recent years. (Figure 32d, Table A7). Cover change scores at Middle (‘very poor’), North Keppel and 5 m depth at Keppels South (‘poor), contrast with the ‘moderate’ to ‘good’ scores at other reefs and depths in the region (Table A7).

The regional Composition score remained in the ‘poor’ category in 2023 (Figure 30, Table 15) despite substantial variability among reefs (Table A7). Contributing to the low regional score were scores of zero at both depths at Middle and the 2 m depths of Pelican and Keppels South (Table A7). At each of these reefs there is a lower proportion of *Acropora* in the coral communities than was observed during the first five years of observation that form the reference state for this indicator (Table A7, Figure A6).

The Macroalgae indicator score has remained ‘very poor’, other than for a brief improvement due to the loss of macroalgal cover caused by Fitzroy River floodwaters in 2011 (Figure 30, Table 15). In 2023, the proportion of macroalgal cover increased at most reefs (Figure 32b). However, these increases had minimal influence on the Macroalgae score as the macroalgal proportional cover was already above thresholds at which negative influences on coral resilience are expected and scores of zero returned (Table A7). At reefs with high macroalgal cover, the algal community is dominated by brown macroalgae of the genus *Lobophora* or a combination of *Lobophora* and species in the family Sargassaceae (Table A11).

Despite declining since 2020, the Juvenile coral score remains ‘poor’ (Figure 30, Table 15). In 2023, only at the 2m depth at Barren was the density of juvenile corals within the ‘moderate’ range (Table A7), a score reflecting moderate numbers of juvenile *Leptastrea* at this location (Figure A6). Between 2014 and 2020 scores for this indicator had increased, moving from ‘very poor’ to ‘poor’ by 2016 (Table 15, Figure 30). The increased density of juveniles through to 2020 reflect increased densities of a variety of coral genera with little consistency in the relative increases of these genera among reefs (Figure A6).

In situ water quality monitoring was reinstated in 2021 after being discontinued in 2015. In 2015, the long-term water quality index was assessed as improving and scored as ‘good’ (Gruber *et al*. 2024). Conditions from 2021 to 2023 were also categorised as ‘good’ (Figure A15). In 2023, the short-term water quality index score (Gruber *et al*. 2024) was also ‘good‘, with most water quality parameters being below guideline values (Figure A15). Parameters that exceeded guideline values included dissolved inorganic forms of nitrogen (NOx) and phosphate (PO_4) (Figure A15). Over the period 2019–2023 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 31a,b, Table A8).

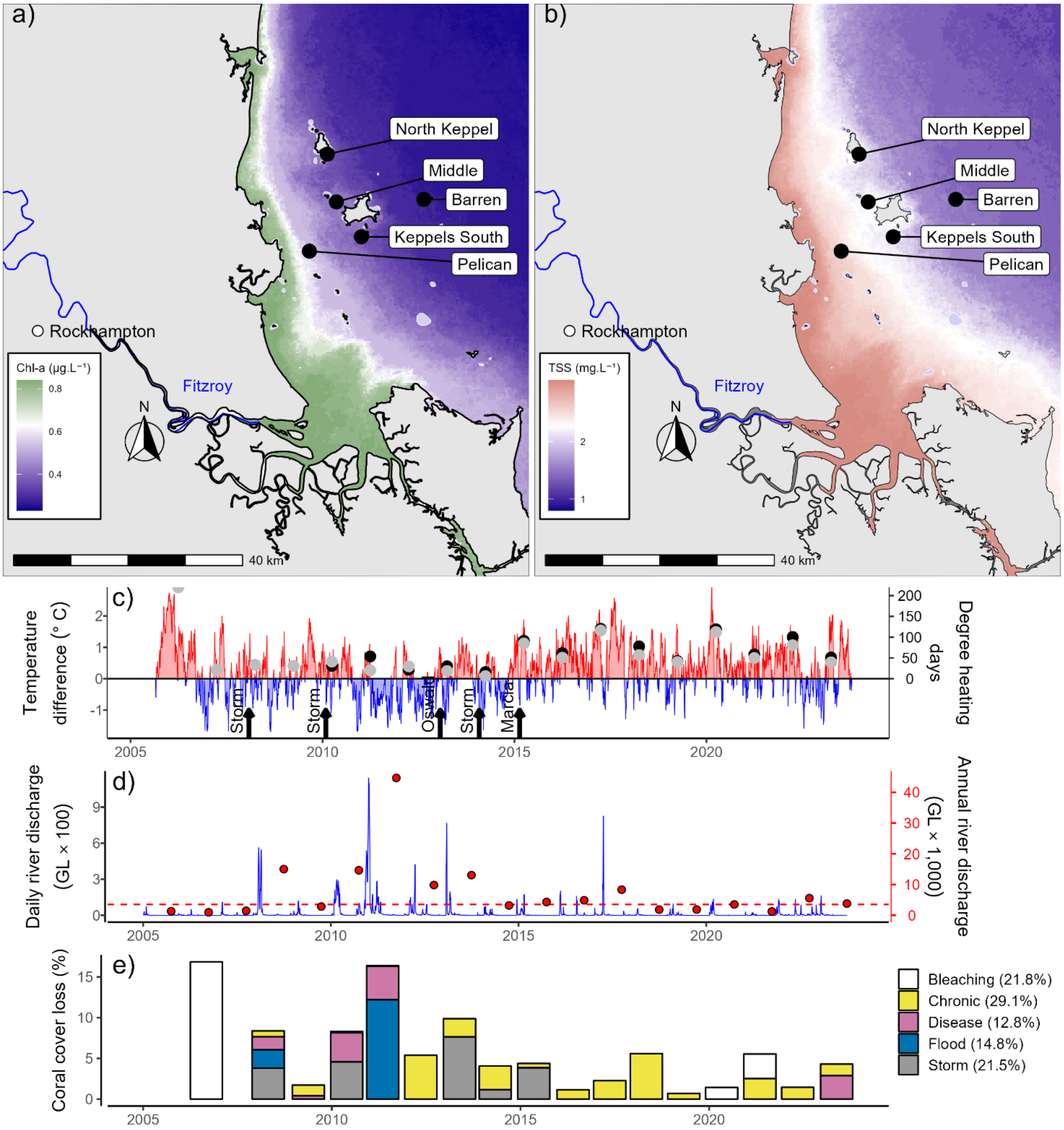

Figure 31 Fitzroy region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) median wet season Chl *a* and b) median wet season TSS concentrations. Water quality data are the mean of median levels over the period 2019–2023, white breaks 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 DHD over the summer period (1 December – 31 March) as reported by BoM (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.

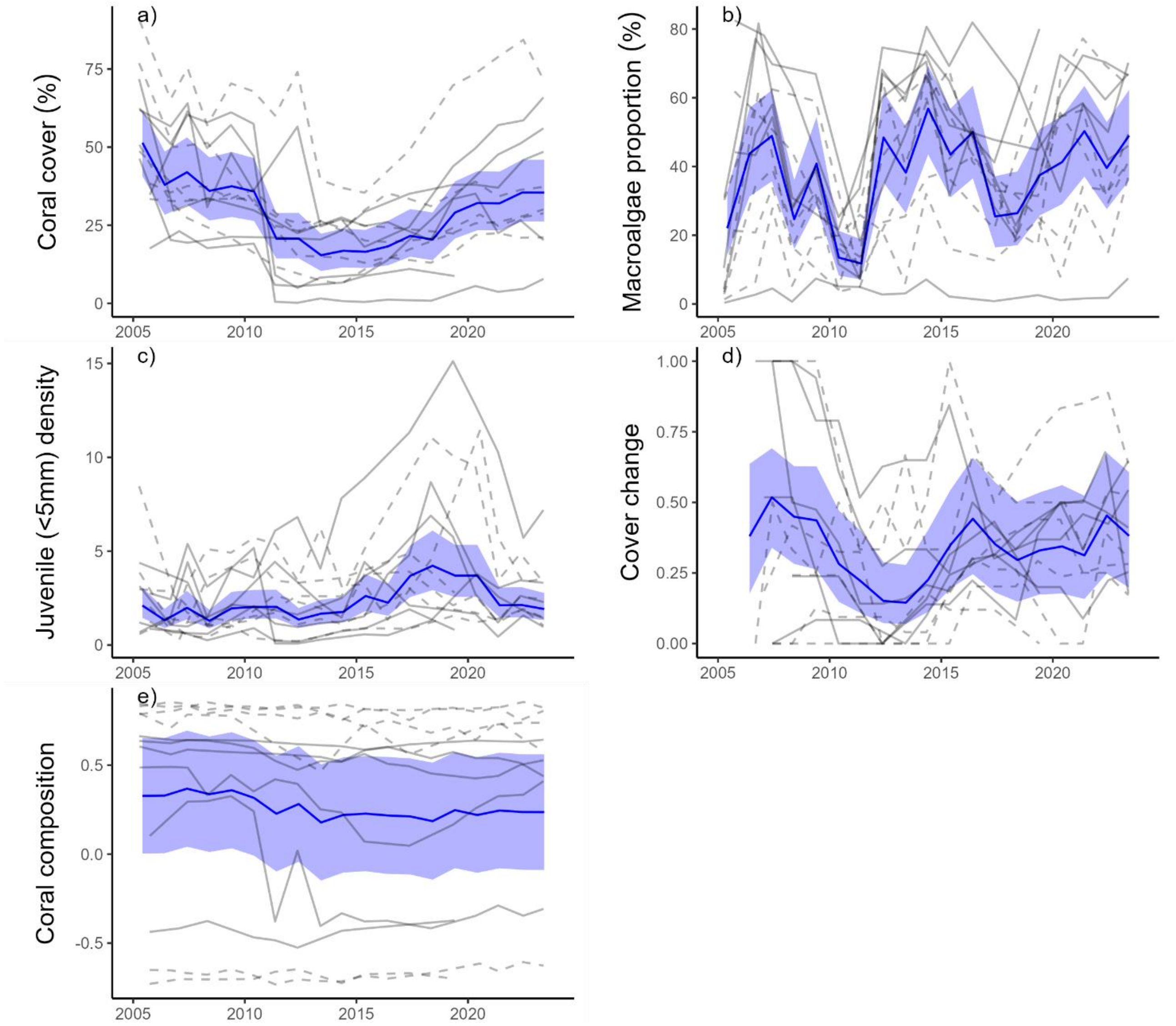

Figure 32 Fitzroy region indicator trends. a– e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.7 Response of coral communities to environmental conditions

4.7.1 Location along water quality gradients

The Reef-wide Coral Index scores in 2023 did not show consistent trends along wet-season water quality gradients estimated from sentinel data between 2019 and 2023 (Table A8).

Of the individual indicators:

- scores for Coral cover were negatively related to increasing concentration of Chl *a* and TSS at 2 m depths. These relationships were statistically apparent in the Wet Tropics and Fitzroy regions and followed the same general trend in the Burdekin region (Table 16, Figure 33).
- Cover change, Composition, Juvenile coral or Macroalgae indicator scores did not vary predictably along water quality gradients in 2023.

Table 16 Relationship between indicator scores and gradients in water quality. Tabulated values are upper (u) and lower (l) confidence intervals of the trend in scores for each combination of indicator, and depth, for which Reef-wide relationships between scores in 2023 and water quality proxies (mean of wet season median for Chl *a* and total TSS between 2019 and 2023) were observed (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 score 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											
Coral cover score	2	-12.9	-1.2	-27.6	-5.5	-10.7	5.6	-10.0	30.3	-23.6	-0.02
Total suspended solids concentration											
Coral cover score	2	-1.9	-0.1	-3.6	-0.6	-1.7	0.8	-1.2	3.9	-3.8	0.1

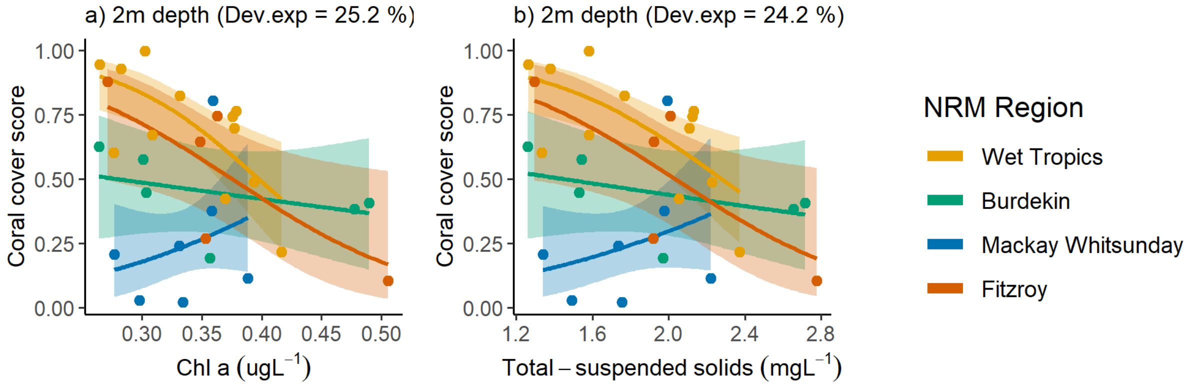

Figure 33 Coral cover indicator score relationships to water quality. Plots present predicted relationship within each region. Confidence intervals in predicted slopes are provided in Table 16.

To ensure scores are sensitive to change at each reef, the Macroalgae and Composition indicators are scored against thresholds that vary along water quality gradients. As such, the spatial analysis of scores masks underlying differences in the values underpinning these scores.

Community composition values were derived from the product of genus-level coral cover estimates and eigenvalues for the distribution of genera along a water quality gradient (Table A4). That community composition is negatively related to Chl *a* and TSS concentration (Figure 34) is entirely to be expected given the derivation and intent of this indicator. Steeper relationships are evident at 5 m reflecting the cumulative pressure of reduced light and higher rates of sedimentation at this depth. There is no relationship between community composition and water quality in the Mackay–Whitsunday region where conditions are more similar among reefs than in other regions (Figure 34).

Reef-wide, the proportion of algal cover comprised of macroalgae (Macroalgae proportion) shows a positive relationship to both Chl *a* and TSS at 2 m depth but not at 5 m depth (Table 17). These relationships are most evident in the Burdekin region (Table 17, Figure 35).

Table 17 Relationship between Macroalgae and Composition indicator values and water quality gradients. Tabulated values are upper and lower confidence intervals of the trend in values for each combination of indicator 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											
Macroalgae proportion	2	3.6	14.1	-3.1	18.3	1.74	18.5	-12.7	24.6	-1.4	20.8
Community composition	2	-3.0	-0.4	-4.7	-0.01	-2.9	0.9	-2.1	6.5	-6.0	-1.2
	5	-5.3	-2.3	-7.2	-1.6	-5.5	-1.0	-4.0	3.8	-9.6	-3.8
Total suspended solids concentration											
Macroalgae proportion	2	2.2	8.5	-0.1	1.1	0.1	1.2	-1.2	2.2	-0.2	1.0
Community composition	2	-0.4	-0.04	-0.6	0.02	-0.5	0.1	-0.3	0.9	-0.9	-0.1
	5	-0.8	-0.3	-1.0	-0.2	-0.9	-0.2	-0.5	0.5	-1.5	-0.5

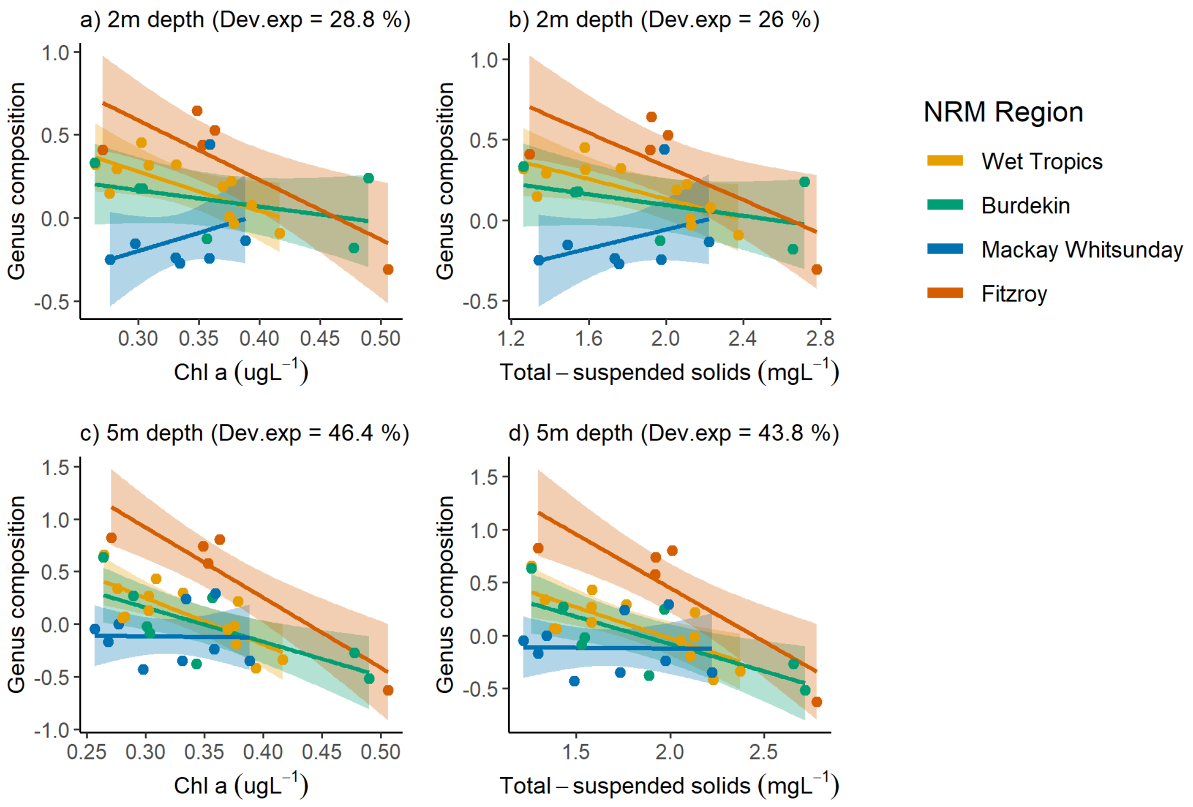

Figure 34 Relationship between coral community composition and water quality. Plots present predicted relationship within each region. Confidence intervals in predicted slopes are provided in Table 17.

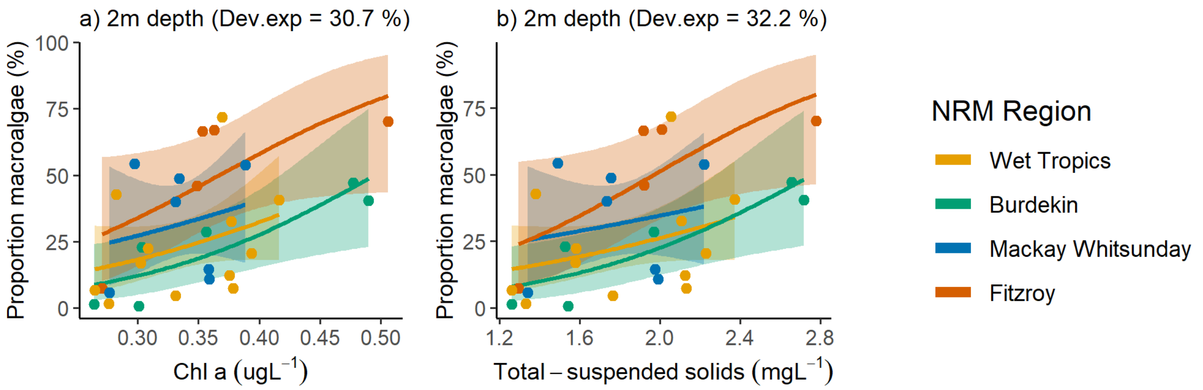

Figure 35 Relationship between Macroalgae proportion and water quality. Plots present predicted relationship within each region. Confidence intervals in predicted slopes are provided in Table 17.

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 (Table 18, Figure 36). 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. Relationships between loads of particulate and dissolved nitrogen, suspended sediment and Coral Index change generally mirror those described for discharge (Table 18). This is not surprising as nutrient loads within rivers are correlated with river discharge.

Table 18 Relationship between changes in the Coral Index scores and environmental conditions. Tabulated are the proportion of deviance explained by models fit to relationships between the time-series of Coral Index score changes during non-disturbance periods and summaries of environmental condition during those periods. Shading indicates the relationship was monotonic with higher increase in Coral Index scores at lower exposures to the environmental pressure. An (*) marks relationships that were not monotonic although either the most negative Coral Index score changes were observed at high exposures or most positive changes occurred at lower exposures. Blank cells indicate no relationship was observed with AICc values within 2 units of null models.

Region	Freshwater Discharge	PN (JCU extrapolated load)	DIN (JCU extrapolated load)	Sediment (JCU extrapolated
Wet Tropics	9.7%	9.5%	9.2%	9.5%
Burdekin	8.1%	6.1%*	7.4%	
Mackay–Whitsunday				
Fitzroy	14.8%*	11.4%*	8.1%	12.3%

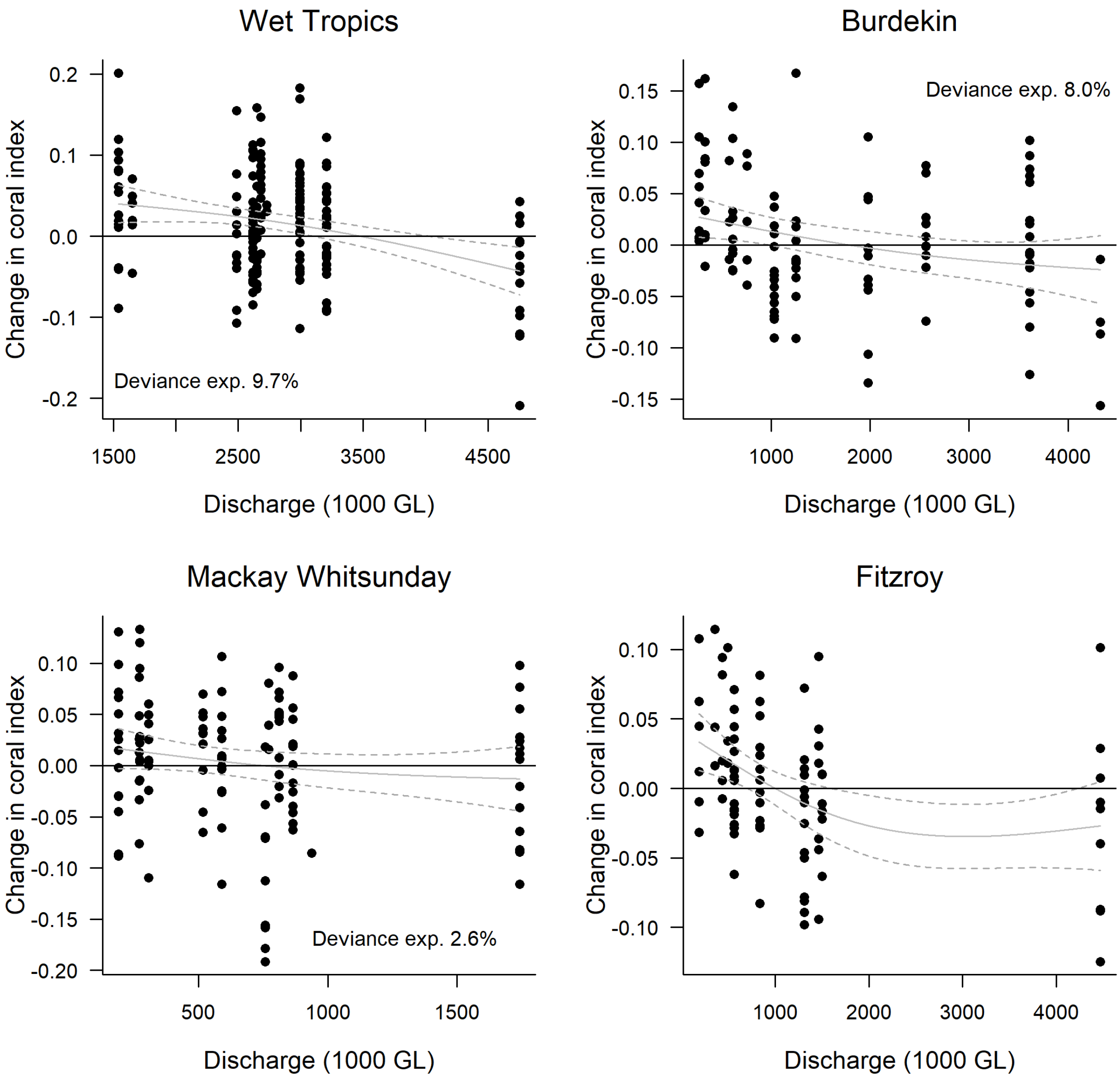

Figure 36 Relationship between the Coral Index and freshwater discharge from local catchments. Plotted points represent observed change in the Coral Index score at each reef and depth over a two-year period. Observations following years for which acute disturbances impacted communities in the period 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 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.

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 have caused almost half (43%) of all coral cover losses on inshore reefs 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. Since 2017 there has been no impact from severe cyclones, signifying a period during which coral recovery should occur.

However, high water temperatures over the 2017, 2020 and, to a lesser degree, 2022 summers have interrupted the recovery of coral communities. These marine heat waves elevated the relative impact of coral bleaching to account for 15% of coral cover loss on inshore reefs since 2005. In 2020, although bleaching was severe at several reefs in the Burdekin and Keppel regions, loss of coral cover was relatively minor. However, corals at some reefs were severely bleached at the time of surveys in 2020 and the further loss of coral cover through to 2021 was attributed to the subsequent mortality of these heat stressed corals. Similarly, while no reefs in the Burdekin region were categorised as having lost coral cover due to bleaching in 2022, partially bleached corals were observed during surveys in that year and as discussed below in section 5.2.3, 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, will add some uncertainty to apportioning losses to specific pressures.

Notable from the 2020 event was that on all except one of the 15 MMP reefs at which a bleaching impact was recorded, the proportion of coral lost was greater at the 2 m depth than at the adjacent 5 m depths. This observation was consistent with previous reports of reduced severity of bleaching with depth (e.g., Muir *et al*. 2017, Cantin *et al.* 2021). While not within the scope of this report, temperature profiles from the two depths at each reef could be compared to ascertain whether this observation reflects:

- higher thermal stress at 2 m due to stratification of the water column,
- differences in the susceptibility of corals based on taxonomic differences (Marshall & Baird 2000) between depths,

If neither of the above hold, then a degree of protection offered by reduced light intensity with increased depth and/or self-shading due to increased symbiont loads would be plausible (Anthony *et al*. 2007).

In general, 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 have, to date, been spared the magnitude of thermal stress that resulted in widespread mortality of corals elsewhere (Hughes *et al.* 2018). However, the level of bleaching observed on inshore reefs in the Burdekin and Fitzroy regions in 2020 suggest that this event was very close to the threshold that would result in widespread mortality. Worryingly, it is becoming increasingly 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, Heron *et al.* 2018, Oliver *et al.* 2019).

Since 2005, outbreak densities of crown-of-thorns starfish have only been observed on inshore reefs in the Wet Tropics and their impact is discussed in section 5.3.1.

Loss of coral cover due to exposure to low salinity flood waters has 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.

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 Coral cover as a coral community state. 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 the categorisation of both acute and chronic pressures helps to focus on reef recovery processes, it is inevitable that acute and chronic pressure interact. In short, quantification of 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). 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 Gruber *et al*. 2024) 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 Gruber *et al.* 2024) 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 inshore coral reefs.

It is evident from the MMP marine water quality time-series that there were gradual declines in water quality through to 2012, a period 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 (Gruber *et al.* 2024). 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 2023, 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. In general, 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.

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). As expected, with close to long-term median inputs from most catchments in recent years, and no major cyclones contributing to resuspension of TSS concentrations since cyclone Debbie in 2017, suspended sediment concentrations met guideline values in most regions (Gruber *et al.* 2024).

5.2 Ecosystem state

5.2.1 Coral community condition based on the Coral Index

Spatial and temporal trends in Coral Index scores reflect the cumulative influence of multiple acute disturbances and the moderation of recovery by chronic environmental pressures. In all regions, scores reached a low point between 2012 and 2014 following multiple acute disturbances, and high discharge of freshwater, nutrients and sediment from adjacent catchments. In all regions, recovery was observed and the condition in 2023 reflects both the strength of this recovery but also the influence of more recent pressures. Of concern is that improvements in the Coral Index in most regions have stalled in recent years.

In 2023:

- The Barron–Daintree sub-region score remained ‘moderate’. Improvement relative to a low point in 2014 has occurred more at 5 m than at 2 m depths. Currently, low Composition scores reflect the low cover of *Acropora* at Snapper Island relative to that observed prior to a series of losses caused by coral disease, crown-of-thorns starfish, floodwaters and storms. Low scores for the Juvenile coral indicator coincide with low scores for Macroalgae and suggest high cover of macroalgae is likely to be limiting coral recruitment processes at some reefs.
- The Johnstone Russell–Mulgrave sub-region score has varied about the threshold between ‘moderate’ and ‘good’ since 2015. The ongoing presence of crown-of-thorns starfish has limited the Coral Index score. While coral cover has tended to recover well when numbers of these starfish are low, their feeding over recent years will have reduced the amount of coral currently observed. Increased levels of macroalgae at several reefs in 2023 and continued low densities of juvenile corals contribute to slightly reduced scores.
- The Herbert–Tully sub-region score has declined from a high point in 2020. Most influential in this decline have been declines in Coral change and Juvenile coral scores and at 5 m depths Macroalgae scores; however, scores for all indicators remain classified as either ‘moderate’ or ‘good’ in 2023.
- Burdekin region score remains ‘moderate’. A slight decline since 2020 resulted due to reduced juvenile densities and increased prevalence of macroalgae. Although Coral Index scores have substantially improved since 2013, high levels of macroalgae on many reefs continue to limit the recovery of coral communities.
- The score for the Mackay–Whitsunday region remains poor but has increased marginally since 2020. This increase was influenced by a significant improvement in the Juvenile coral indicator, which is now ‘moderate’ and represents an important stage in the recovery of coral communities following their being severely impacted by cyclone Debbie in 2017.
- Slow recovery of reefs in the Fitzroy region has paused in recent years. The Coral Index score remains ‘poor’. Although Coral cover scores have increased slightly, the current ‘poor’ score for Cover change shows the recovery of hard coral cover remains slow. Persistently high cover of macroalgae and low densities of juvenile corals at most reefs continue to limit the recovery of coral communities.

Variability in the condition of coral communities along water quality gradients highlights the pressure imposed by poor water quality. Reef-wide inshore Coral cover scores decline with increasing Chl *a* and TSS concentrations in surrounding waters. This relationship is most evident at 2 m depths, where a statistically significant relationship was observed in the Wet Tropics and Fitzroy regions. It should be acknowledged that within-region statistical estimates will have low power due to the small number of reefs sampled and will be highly sensitive to the state of individual reefs. Further, differential exposure to recent acute disturbance events will confound the interpretation of the relationship between coral cover and the chronic pressure associated with water quality. For example, the monitoring sites at Shute Harbour were protected from wave damage during cyclone Debbie and retain very high coral cover while the nearby sites at Daydream Island were severely impacted. This variable exposure to a recent acute event results in high variability in current coral cover estimates at reefs sharing similar water quality.

In addition to the confounding influence of variable loss of coral cover caused by cyclone Debbie, the relatively low variability in water quality conditions among reefs in the Mackay–Whitsunday region reduces the scope for strong differentiation of each indicator of coral community condition. Compounding this lack of differentiation among sites is that satellite derived estimates of water quality are derived from open waters adjacent to the sampled reefs, assimilating estimates from waters ~1–3 km from the coral sites. This spatial mismatch means that fine-scale (<1 km) hydrodynamic processes that influence the conditions experienced by the corals will not be resolved by satellite derived estimates of water quality.

The lack of relationship between Macroalgae or Composition scores and environmental gradients is influenced by the underlying metrics for these indicators. 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 these 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).

In previous years, scores for the Macroalgae indicator at 2 m depth were negatively related to Chl *a* concentration. The sensitivity of this relationship was limited by reefs returning scores of zero due to having levels of macroalgae in excess of their reef specific reference points, and hence returning low scores across much of the water quality gradient. This issue continued in 2023 with Reef-wide scores for Macroalgae declining as more reefs exceed their reef specific thresholds.

Relating the data underpinning the Macroalgae indicator to reef-level water quality demonstrates there is a higher proportion of macroalgae in algal communities at 2 m depth on reefs exposed to relatively high concentrations of Chl *a* and TSS. This relationship is most evident in the Burdekin Region. While the relationship between the proportion of macroalgae and water quality in the Fitzroy region was not statistically significant, this result is influenced by high levels of macroalgae across much of the water quality gradient with only Barren Island, the reef with the lowest concentrations of Chl *a* and TSS, not having very high levels of macroalgae in 2023.

Similarly, 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 2023. 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 2023 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.

The Cover change score is also standardised for community composition, with expectations for increase in hard coral cover to be higher for the family Acroporidae than for other hard corals. This means that the score for a reef with low cover of Acroporidae will be higher than for a reef with less Acroporidae despite equal increases in hard coral cover. This feature of the indicator was designed to allow realistic expectations for hard coral cover increase based on the coral communities present but does bias the scores within a reef when changes in community composition occur. In short, the lack of relationship between Cover change scores and location of a reef along the water quality gradient does not mean that the rate of change in hard coral cover is the same.

Acute disturbance events are primarily responsible for the loss of coral cover at most reefs (Lam *et al.* 2018). The impact of poor water quality is evident in the rate that coral communities recover from these events. In the Wet Tropics, Burdekin and Fitzroy regions, coral community resilience, estimated as the change in Coral Index scores during periods that reefs were free from acute disturbances, was reduced when discharge from the adjacent catchments, and the associated loads of nutrients and sediments, were high. However, while this has been a consistent feature observed within the MMP for several years, the strength of this relationship has declined, reflecting the relatively stable Coral Index scores in recent years despite near median discharges in most regions.

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).

Also limiting the detection of a relationship between regional discharge and change in the Coral Index scores for the Mackay–Whitsunday region was declines in the Coral Index that occurred in 2006 when discharge was low. While the 2006 declines remain unexplained, they are best explained by other acute impacts such as temperature stress, as indicated by *in situ* temperature loggers.

In general, the spatial and temporal variability in Coral Index scores presented in this report 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. 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 selected for communities tolerant of those conditions (De Vantier *et al.* 2006). This means that communities in different locations will have different susceptibilities to water quality pressures (e.g., Morgan *et al.* 2016). It is precisely the inability to accurately measure, or predict, 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).

5.2.2 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. As would be expected given a lack of severe disturbances, coral cover has increased in recent years in all regions. Increased cover was recorded at most reefs in the Barron–Daintree and Herbert–Tully sub-regions of the Wet Tropics, the Burdekin region and at 2 m depths in the Mackay–Whitsunday region. In the Johnstone Russell–Mulgrave sub-region, changes in coral cover have been more variable, largely due to the ongoing impact of crown-of-thorns starfish. In the Fitzroy region and at deeper sites in the Mackay–Whitsunday region improvement in coral cover has been less consistent.

In 2023, coral cover was generally higher at reefs exposed to lower concentrations of TSS and Chl *a*; however, poor water quality does not necessarily preclude high cover of corals on inshore reefs. 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.

5.2.3 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 a 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.

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 was 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 since 2017. The time-series of Cover change scores in the Mackay–Whitsunday region suggest pressures imposed on coral growth by the ambient environmental conditions were for several years following cyclone Debbie when the long-term water quality index declined into the ‘poor’ range (Gruber *et al.* 2024). 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 (Gruber *et al.* 2024).

However, recent declines or continued poor scores for the Cover change indicator have occurred during a period of relatively low catchment input and raise the emerging issue of 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 2019, meaning any low-level or protracted impacts of the 2020 and 2022 marine heat waves may have contributed to the declining but ‘moderate’ Cover change scores in the Herbert–Tully sub-region and Burdekin region, and ongoing poor scores in the Fitzroy region.

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 preform 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).

5.2.4 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.

In 2023, the Composition indicator score remained ‘poor’; however, there was substantial variability among the sub-regions. The ‘good’ (Herbert–Tully and Burdekin) and ‘moderate’ (Johnstone Russell–Mulgrave and Barron–Daintree) scores contrast the ‘poor’ scores in the Mackay–Whitsunday and Fitzroy 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 group 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 this genus within the coral communities is essential for rapid recovery of coral cover following disturbances.

In most regions, the scores for this indicator have tended to track those for Coral cover. Influencing this relationship is the disproportional loss of *Acropora* in response to acute pressures and their subsequent recovery. The current ‘poor’ scores for this indicator in the Mackay–Whitsunday region is largely due to the loss of *Acropora* cover following cyclone Debbie. Although early signs of *Acropora* recovery were observed in 2022 and 2023 with low but increasing numbers of juveniles observed on some reefs, these have yet to translate to a recovery of the proportional cover of *Acropora* within the coral communities at most reefs.

In the Fitzroy region, most reefs were dominated by branching *Acropora* in the early years of the MMP and the current poor scores demonstrate that this, typically very rapidly growing, group has yet to recover at several locations. However, the Composition scores in 2023 have improved as some reefs slowly recover.

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 ‘good’ 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. The early years are used as a 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); 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.

5.2.5 Macroalgae

Coral reef macroalgae generally benefit from increased nutrient availability due to run-off (e.g., Schaffelke *et al.* 2005, Adam *et al.* 2021). As coral competitors, macroalgae suppress both coral growth and juvenile settlement or survival (e.g., Tanner 1995, McCook *et al.* 2001, Birrell *et al.* 2005, 2008a, b, Clements *et al.* 2018, Doropoulos *et al*. 2022) 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 on several reefs within each region offers strong support for the presence of such feedbacks, with observed relationships between Chl *a* and TSS concentrations, both likely proxies for nutrient availability, and the proportion of macroalgae at 2 m depths, linking nutrient availability to reduced coral community resilience caused by high levels of macroalgae.

Unlike the coral indicators that are plausibly responding to water quality extremes (e.g., following flood events), the persistence of macroalgae suggest that ambient water quality levels are important for the maintenance of high macroalgal cover. While reef-specific thresholds for scoring macroalgae allow for increased levels of macroalgae in response to naturally occurring gradients of water quality, their cover in 2023 regularly exceeded thresholds likely to have detrimental influences on coral recruitment and growth resulting in scores of zero at many reefs.

It is important to note that the relationship between the water quality parameters of Chl *a* and TSS does not necessarily indicate a direct causal relationship between these parameters and pressures imposed by macroalgae. Chl *a* is a measure of phytoplankton biomass and these microalgae are likely to respond to nutrient availability in a similar way to benthic macroalgae. Work by Schaffelke and Klump (1998) demonstrate nutrient limited growth for a species of *Sargassum* 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).

Irrespective of the underlying mechanisms that control macroalgae on reefs, the environmental conditions at sites where Chl *a* concentration frequently exceeds the summer guideline value support macroalgal cover at a level detrimental to coral community resilience. The distribution of large brown macroalgae shows a strong relationship to environmental conditions of high nutrient availability at 2 m depths. At 5 m depths the relationship is not as strong, likely due to light becoming limiting for macroalgal growth in deeper, turbid, nutrient-rich inshore waters (Jones *et al*. 2021). Additionally reduced wave driven resuspension with depth allows the build-up of fine sediments on the substrate (Wolanski *et al*. 2008, Thompson *et al.* 2017) likely further limiting macroalgal proliferation.

5.2.6 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 2023, declines in Juvenile coral scores were evident in the Herbert–Tully sub-region, where the score remained ‘good’, and the Fitzroy region and at 5 m depth in the Burdekin region, which both remained poor. In contrast, juvenile coral densities have increased in the Mackay–Whitsunday region elevating the score to ‘moderate’. Scores remain ‘poor’ and unchanged in the Barron–Daintree and Johnstone Russell–Mulgrave sub-regions.

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 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*.

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 high densities of juveniles from genera such as *Turbinaria*, *Goniastrea* and *Favites* that tend to occur in poor water quality environments (Table A8).

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[2]) 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.

5.3 Regional summaries

5.3.1 Wet Tropics

At the regional level, the Coral Index scores have remained relatively stable at ‘moderate’ since 2016, although between 2022 and 2023 there was a slight decline driven by a decline in Cover change and Macroalgae scores. In 2023, the Cover change indicator declined to ‘moderate’, while the Composition indicator increased to ‘good’. Macroalgae and Juvenile coral remained classed as ‘moderate’ and the Coral cover indicator remained classed as ‘good’. While there were no severe disturbances over this period, scores within sub-regions have varied as communities have been impacted by, and recovered from, localised pressures.

Coral Index scores for the Barron–Daintree sub-region have improved in recent years as the coral community continues to recover from the impacts of coral bleaching in 2017, and the combined influence of flooding of the Daintree River and cyclone Owen prior to 2019 surveys. Reefs in this region escaped exposure to high levels of thermal stress in 2020 and 2022, with negligible impact observed. The lack of disturbances in the Barron–Daintree sub-region between 2019 and 2023 has allowed recovery to occur with most indicators increasing for at least one depth with the other staying stable. Only Macroalgae at 2 m has seen scores declining as a dense mat of macroalgae persists at Snapper North 2 m depth. Red macroalgae are also common at Snapper South 5 m. The high levels of macroalgae at these sites correspond to low densities of juvenile corals and in combination are the primary factor limiting the sub-regional Coral Index score to the ‘moderate’ range.

In the Johnstone Russel–Mulgrave sub-region the Coral Index has remained relatively stable, oscillating about the boundary between ‘moderate’ and ‘good’ scores since entering the ‘good’ range in 2016. The lack of further recovery in this sub-region has been influenced by the ongoing presence of crown-of-thorns starfish. Predation by crown-of-thorns starfish has been identified as the primary cause of coral cover loss in the sub-region for most years since 2013. Only in 2020 were no impacts of crown-of-thorns starfish recorded. Since the beginning of the MMP in 2005 predation by crown-of-thorns starfish has been attributed to 40% of the interannual hard coral losses. In 2023, the largest loss was at High West where, although no crown-of-thorns starfish were observed on the transects during the 2023 survey, large feeding scars were evident and large crown-of-thorns starfish had been observed during the 2022 survey. In contrast, juvenile and sub-adult crown-of-thorns starfish were observed at Franklands East; however, no loss of coral was recorded as rapid growth of *Acropora* colonies led to a net increase in coral cover.

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

The Wet Tropics is the only region in which crown-of-thorns starfish have been common on inshore reefs, although they have been recorded in the Burdekin at Palms East (2016) and Palms West (2019), and culling has occurred in the outer Whitsunday Islands in 2022/23. In recent years, the Crown-of-thorns Starfish Control Program has helped to mitigate the impact of crown-of-thorns starfish[3] with 24,519 individuals removed from the monitoring reefs since 2013, 10,895 of these from Fitzroy Island and the Frankland Group in the three years preceding the 2023 surveys, although only 149 of which were removed in the last year. Although numbers of crown-of-thorns starfish observed by the MMP and removed by culling were lower in 2023, both programs record relatively high proportion of juveniles in the population signifying their ongoing recruitment and potential for future impacts.

Coral lost to crown-of-thorns starfish and any bias in the Cover change estimates caused by low levels of predation (see section 5.2.3) will have contributed to keeping the Coral Index score below the ‘good’ range in recent years. However, the current ‘poor’ scores for the Juvenile coral indictor and decline in the Macroalgae indicator are also noteworthy. The lowest density of juvenile hard corals occurred at High East and Franklands West and, as occurred in the Barren–Daintree sub-region, coincide with relatively high levels of red macroalgae.

The Herbert–Tully sub-region was categorised as ‘good’ from 2019 to 2021 and declined from 2020 to 2023, most prominently at the 5 m depth where Macroalgae, Juvenile coral and Cover change scores all declined significantly. This sub-region was more exposed to marine heatwave conditions in both 2020 and 2022 than the sub-regions to the north and this alone may explain the recent decline in Cover change score (see section 5.2.3). However, the combination of low juvenile densities and high levels of macroalgae exists here, as seen elsewhere in the region.

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. However, several reefs received a ‘poor’ grade in 2023 (Snapper North and Bedarra 2 m, Franklands West and Dunk South 5 m, and both depths at High East). Common to all these ‘poor’ reefs were ‘very poor’ scores for Macroalgae and this, along with the regional negative relationship between coral cover and water quality suggest that water quality continues to limit the resilience of some reefs.

5.3.2 Burdekin

The Coral Index score for the Burdekin region declined from a peak reached in 2020 and remains ‘moderate’ in 2023. The decline from 2020 is due primarily to declines in Juvenile coral scores at 5 m depth and Macroalgae scores at 2 m depth. In contrast, the mean cover of corals across the region has continued to increase significantly and in 2023 reached its highest level since the inception of the MMP in 2005. While attaining the highest level of coral cover observed over 19 years of monitoring is a positive indication of the resilience of coral reefs in the region, variability in recovery trajectories and individual indicator scores suggest ongoing environmental pressures are limiting the condition of some reefs.

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 43.8% in 2015 to a low of 13.2% in

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

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). *Acropora* cover is again increasing at this reef and was at 21.4% in 2023, but *A. pulchra,* a species common prior to 2017, is no longer present on the transects (*pers. obs*. Author).

Coral bleaching, in response to marine heatwaves, has accounted for 27% of interannual coral losses since 2005, with all these losses occurring since the 2017 mass bleaching event, during which time it has been the primary cause of coral loss in the region. Marine heatwaves in 2020 and 2022 will have contributed to reductions in the Coral Index, because although coral cover has increased, the score for this indicator is lower than it would have been had some losses not occurred. While the Cover change score has remained ‘moderate’ this is variable among reefs. In 2023, out of the six reefs with *Acropora* cover at > 10%, the Cover change score was ‘poor’ at five, indicating it is the slower growing corals that are performing to expectations rather than the fast-growing *Acropora*.

The indicator scores that have declined most since 2020 were Macroalgae and Juvenile coral, both of which remain categorised as ‘poor’ in 2023, although both do show a slight uptick from lows observed in 2022. For Macroalgae, there is a clear demarcation in scores between Palms East and Palms West, where scores were ‘very good’, and all other sites deeper into Halifax Bay and Cleveland Bay, where scores were either ‘poor’ or ‘very poor’. There was a significant positive relationship between the proportional representation of macroalgae in the algae community and concentrations of Chl *a* and TSS at the 2 m sites in this region that demonstrates the role of poor water quality in this result. Although water quality over the short term appears to be improving, possibly reflecting reduced inputs from the catchments in recent years, concentrations of NOx remain well above guideline values in this region (Gruber *et al*. 2024).

The densities of juvenile corals have always been variable among reefs and depths, but the consistent decline in the Burdekin region since the 2020 and 2022 bleaching events 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.

5.3.3 Mackay–Whitsunday

The Coral Index in the Mackay–Whitsunday region declined dramatically from 2016 through to 2019 due to the impacts of cyclone Debbie. In 2023, the Coral Index has remained ‘poor’. However, coral communities were showing some signs of recovery on the back of increasing densities of juvenile corals and modest increases in coral cover at some reefs.

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’. It was only at the Shute Harbour sites, where corals where not severely impacted by cyclone Debbie, and at Hayman and Daydream that hard coral cover has increased in line with modelled expectations. Of note is that the loss of coral cover in recent years at Dent was caused by coral disease that killed large areas of branching *Acropora* between 2017 and 2020. It is not known if this disease was associated with moderate levels of damage caused by cyclone Debbie, the ensuing poor water quality or some other chronic pressure.

With the severe loss of coral cover at many sites, successful recovery will rely heavily on the recruitment and survival of juvenile corals. Although the Juvenile score continued to increase in 2023, the score remained ‘poor’ at most reefs. It was only at Border and those sites listed above that had at least ‘moderate’ scores for Cover change that juvenile densities were sufficient to return ‘moderate’ or higher scores for Juvenile coral.

A final indication of the lack of resilience of these reefs is the ongoing high levels of macroalgae that tend to coincide with poor scores for the Cover change and Juvenile indicators. Initial increase 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 2023, very poor scores for Macroalgae were also recorded at Dent and Double Cone and the 2 m depth at Daydream. These levels of macroalgae will almost certainly be contributing to the low densities of juvenile corals at nearly all of these locations. Among those reefs with relatively high macroalgae cover, the ubiquitous brown macroalgae family Sargassaceae is mixed with other abundant macroalgae according to the local habitat; *Dictyota* at Double Cone, a range of red macroalgae at Daydream (2 m depth), and red macroalgae and *Lobophora* at Seaforth and Pine (2 m depth). The presence of Sargassaceae within macroalgae communities 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 cyclone Debbie on water quality within the region, with a marked decline in the short-term index in 2017 (Gruber *et al.* 2023). Encouragingly, both the short and long-term water quality index are gradually improving, with the long-term index returning to ‘moderate’ in 2021 and continuing to improve within the ‘moderate’ rating through to 2022. In 2023, the index was almost back to the same level as observed in 2010, prior to cyclone Ului, and consistent with levels in which prior, albeit slow, recovery of coral communities has been observed.

In 2023, the condition of reefs was highly variable. Shute Harbour was minimally impacted by cyclone Debbie and was in good condition with ‘moderate’ or better scores for all indicators. Hayman, Border, Daydream (5 m depth) and, to some degree, Hook were showing signs of recovery. All other reefs had high levels of macroalgae, low densities of juvenile hard corals and lower than expected rates of increase in coral cover, demonstrating ongoing limited capacity for recovery.

5.3.4 Fitzroy

The Coral Index score for the Fitzroy region has remained ‘poor’ and not improved since 2020; the highest point since recovering from a series of pressures that variably impacted coral communities across the region between 2008 and 2015.

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 2015, discharge from the Fitzroy River has been at, or below, median levels and there have been no severe weather events causing damaging waves. Under these conditions some recovery of coral cover has 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 variable among reefs with 'poor' or 'very poor' scores for Cover change within the branching *Acropora*-dominated communities at North Keppel, Middle and the 5 m depth at Keppels South, contrasting the 'moderate' to 'good' scores elsewhere.

The poor rate increase in hard coral cover coincides with persistently high levels of macroalgae. The cover of macroalgae across the region increased dramatically following the loss of corals due to bleaching in 2006 (Diaz-Pulido *et al.* 2009, Ceccarelli *et al.* 2022). 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, where, when first visited in 2005, *Acropora* cover was 70% and there was almost no macroalgae. The macroalgae cover at Middle includes a high proportion of large brown algae of the Sargassaceae family and the genus *Lobophora.* The persistence of these macroalgae at Middle, where macroalgae cover was 50% in 2023, has almost certainly limited the recovery of coral cover. The timeseries of coral and macroalgae covers at Middle, 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 2023 had declined within that range. In 2023, the only reef with juvenile coral densities above the 'poor' or 'very poor' range was Barren (2 m depth), notably the only reef at which the Macroalgae score was above the minimum value of zero. However, this Juvenile score was buoyed by high abundance of *Leptastrea* that have not contributed to meaningful increase in the cover of this genus, despite high numbers of juveniles recorded in previous years. 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)

5.4 Management response

Coral reefs in general are subjected to cumulative impacts of acute disturbances and environmental pressures (Bozec *et al*. 2022). Put simply, successful management should promote a balance between coral losses and subsequent recovery. The identification of 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 2023 noted a decline in densities of crown-of-thorns starfish at High Island but recorded moderate densities of crown-of-thorns starfish in juvenile or sub-adult size classes demonstrating an ongoing pressure, and potential source of replenishment of these starfish to reefs in this region.

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 suggest 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 maintain high cover of macroalgae. As such, removal of algae such as *Lobophora* and *Sargassum* 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. The rate of this recovery is, however, influenced by the loads of nutrients and/or sediments entering inshore waters, particularly during flood events. To maintain the balance between disturbance and recovery of the inshore Reef it is essential that management actions provide corals with optimum conditions to cope with ever increasing global stressors of climate change and ocean acidification (Bellwood *et al.* 2004, Marshall & Johnson 2007, Carpenter *et al.* 2008, Mora 2008, Hughes *et al*. 2010, Claar *et al*. 2020).

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

6 CONCLUSIONS

The cumulative impacts of acute disturbances, including cyclones, crown-of-thorns starfish, thermal stress and low salinity flood plumes, have clearly impacted the condition of inshore reefs (Lam *et al.*

2018, Ceccarelli *et al.* 2020, Thompson *et al.* 2020). Results from 2023 confirm that chronic pressures attributed to poor water quality continue to suppress the recovery of coral communities following these acute events.

The persistence of inshore coral communities will depend on the long-term balance between the frequency and severity of acute pressures and the ability of corals to recover. Central to this balance 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. 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 2019, Abelson 2020).

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, effects of interactions between water quality stressors and with other acute disturbances have only been quantified for a limited combination of pressures and few coral species (e.g., Uthicke *et al.* 2016). In combination, these knowledge gaps limit the ability to quantify thresholds for water quality that are 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 variation in water quality (Thompson *et al.* 2020).

Spatially, results from this project substantiate that macroalgal abundance is enhanced, to the detriment of corals, in areas exposed to chronic high nutrient availability (Fabricius *et al.* 2005). 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 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 supressed during periods of increased loads of sediments and/or nutrients from the adjacent catchments. On balance, Coral Index scores have returned to those observed at the beginning of the project. However, in 2006 when the Coral Index was first estimated, some reefs in these regions had been previously impacted by acute disturbances and as such the 2006 condition may not be an appropriate aspirational reference point.
- On reefs with high cover of macroalgae, the recovery of coral communities has been stalled. Acute disturbance to coral communities, in combination with high nutrient concentrations, are likely to have promoted the initial high cover of macroalgae. Once established, macroalgae are often highly persistent as density-dependent feedbacks, which bolsters their competitive advantage relative to that of corals. As a result, the strength of the relationship between changes in Coral Index scores and environmental variability may be underestimated.
- Since 2017 marine heat wave conditions have impacted reefs in all regions. These added pressures have occurred during a period of relatively low rainfall and minimum cyclone activity during which coral communities should be in a state of recovery. These additional pressures

increase the need for mitigation of pressures associated with poor water quality that continue to limit the recovery potential of some reefs.

- Crown-of-thorns starfish continue to be present on reefs in the Johnstone Russell–Mulgrave sub-region. Ongoing control of these starfish continues to limit their impact on coral community condition in this region.
- In the Mackay–Whitsunday region, high turbidity coupled with high rates of sedimentation can result in unsuitable conditions for the recruitment of some corals at deeper sites. Despite the persistence of water quality conditions considered unfavourable for many corals, large colonies of turbidity-tolerant species remain on many surveyed reefs. The magnitude of impact from cyclone Debbie in 2017 is unprecedented in the monitoring time-series from this region. It will be informative to observe how quickly these communities recover. While still low on most reefs, improved coral recruitment in 2022 and 2023 has been observed at some the severely damaged reefs signalling that recovery is underway. Of ongoing concern is the persistence of macroalgae that have colonised some severely impacted reefs as these will further limit the recovery potential at these locations.

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 expected increase in disturbance frequency, 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	
	Upper	Lower	Upper	Lower
Barnards	23.0	4.8	20.8	1.7
Barren	13.0	3.7	12.6	1.6
Bedarra	23.0	5.3	15.6	1.9
Border			8.2	1.4
Daydream	13.5	3.5	10.4	1.5
Dent	11.6	3.5	10.2	1.5
Double Cone	8.9	3.4	7.6	1.4
Dunk North	23.0	4.6	13.5	1.7
Dunk South	23.0	5.3	15.6	1.9
Fitzroy East	11.7	3.5	10.0	1.5
Fitzroy West	12.5	3.3	13.3	1.5
Franklands East	12.2	3.4	10.5	1.5
Franklands West	11.4	3.4	15.8	1.5
Havannah North			21.7	1.5
Havannah	18.2	3.4	25.0	1.6
Hayman			9.4	1.4
High East	11.2	3.4	13.0	1.4
High West	22.4	4.4	12.1	1.6
Hook	9.3	3.4	8.1	1.4
Keppels South	23.0	3.9	24.0	1.7
Lady Elliot	23.0	6.1	15.3	1.9
Langford			7.9	1.4
Low Isles			8.9	1.4
Magnetic	23.0	6.4	19.0	2.0
Middle	23.0	5.2	23.0	1.8
North Keppel	23.0	5.1	22.6	1.8
Palms East	12.2	3.6	10.5	1.5
Palms West	12.8	3.4	17.5	1.5
Pandora North			13.1	1.6
Pandora	23.0	4.7	16.2	1.6
Pelican	23.0	6.4	18.8	2.0
Pine	18.3	4.4	11.2	1.6
Seaforth	11.8	3.4	10.2	1.4
Shute Harbour	17.6	4.2	11.7	1.6
Snapper North	18.7	4.4	11.3	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 Gruber *et al.* (2024). 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**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**	**2019**	**2020**	**2021**	**2022**	**2023**
Wet Tropics	Daintree River	1,918,174	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
	Mossman River	604,711	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
	Barron River	622,447	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
	Russell–Mulgrave River	4,222,711	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
	Johnstone River	4,797,163	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
	Tully River	3,393,025	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
	Murray River	1,484,246	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
	Herbert River	3,879,683	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
Burdekin	Black River	293,525	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
	Haughton River	558,735	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
	Burdekin River	4,406,780	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
	Don River	496,485	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
Mackay Whitsunday	O'Connell River	835,478	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
	Pioneer River	616,216	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
	Plane Creek	1,058,985	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
Fitzroy	Waterpark Creek	392,614	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
	Fitzroy River	2,875,792	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

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	
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
	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)
	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)*	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), 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)	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), 2013 (32% at 2 m,36% at 5 m), 2014(5% at 2 m)
	Fitzroy West LTMP	12%			COTS and continued bleaching 2000 (80%), COTS: 2013 (6%), 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)
	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)
	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				Other recorded disturbances
		1998	2002	2017	2020	
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)		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	
Burdekin	Palms East	0.93	0.80			cyclone Larry 2006 (23% at 2 m, 39% at 5 m), cyclone Yasi 2011 (83% at 2 m and at 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)
	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%)
	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%)
	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)	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				Other recorded disturbances
		1998	2002	2017	2020	
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				Other recorded disturbances
		1998	2002	2006	2020	
Fitzroy	Barren	1	1	25% (2 m) 30% (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)	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)	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 (41% at 2m)
	Keppels South	1 (6%)	1 (26%)	27% (2 m) 28% (5 m)	1% (2 m) 2% (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)		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 2023. 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	Macroalgae	Cover change	Composition	Coral Index
Barron–Daintree	Low Isles	5	0.66	0.40	0.91	0.57	0.50	0.61
	Snapper North	2	0.42	0.08	0	0.70	0	0.24
		5	0.63	0.31	0.62	0.84	0.50	0.58
	Snapper South	2	0.76	1.00	0.84	0.81	0.50	0.78
		5	0.90	0.05	0	0.53	1.00	0.50
Moderate			**0.68**	**0.37**	**0.47**	**0.70**	**0.50**	**0.54**
Johnstone Russell–Mulgrave	Fitzroy East	2	0.60	0.38	1.00	0.49	0.50	0.59
		5	0.76	0.62	0.87	0.61	0	0.57
	Fitzroy West	2	1.00	0.85	0	1.00	1.00	0.77
		5	0.92	0.74	0.66	0.89	0.50	0.74
	Fitzroy West LTMP	5	0.63	0.51	1.00	1.00	1.00	0.83
	Franklands East	2	0.94	0.25	0.62	0.46	1.00	0.65
		5	0.52	0.43	0	0.50	1.00	0.49
	Franklands West	2	0.93	0.20	0	0.60	1.00	0.55
		5	0.79	0.11	0	0.53	0.50	0.39
	High East	2	0.67	0.16	0	0.12	0.50	0.29
		5	0.62	0.18	0	0.36	0.50	0.33
	High West	2	0.74	0.34	0.57	0.47	0	0.42
		5	0.29	0.36	1.00	0.74	0	0.48
Moderate			**0.73**	**0.39**	**0.44**	**0.60**	**0.58**	**0.55**
Herbert–Tully	Barnards	2	0.82	0.29	1.00	0.63	1.00	0.75
		5	0.76	0.76	1.00	0.35	1.00	0.77
	Dunk North	2	0.70	0.55	0	0.57	0.50	0.47
		5	0.59	1.00	0.51	0.71	0.50	0.66
	Dunk South	2	0.49	0.25	0.14	0.44	1.00	0.46
		5	0.51	0.56	0.01	0.40	0.50	0.40
	Bedarra	2	0.22	0.48	0	0.31	1.00	0.40
		5	0.36	1.00	0.61	0.48	0.50	0.59
Moderate			**0.55**	**0.61**	**0.41**	**0.49**	**0.75**	**0.56**
Burdekin	Palms East	2	0.63	0.20	1.00	0.25	1.00	0.62
		5	0.71	0.33	1.00	0.34	1.00	0.68
	Palms West	2	0.58	0.42	1.00	0.72	0	0.54
		5	0.44	0.62	1.00	0.25	0	0.46
	Havannah North	5	0.29	0.62	0	0.5	1.00	0.48
	Havannah	2	0.45	0.26	0	0.49	1.00	0.44
		5	0.60	0.34	0	0.39	1.00	0.47
	Pandora	2	0.19	0.16	0	0.27	0.50	0.22
		5	0.32	0.46	0.39	0.42	1.00	0.52
	Pandora North	5	0.85	0.40	0.11	0.44	0	0.36
	Lady Elliot	2	0.41	0.28	0	0.28	1.00	0.39
		5	0.65	0.81	0.35	0.69	0	0.50
	Magnetic	2	0.38	0.17	0	0.30	0.50	0.27
		5	0.43	0.41	0	1.00	0.50	0.47
Moderate			**0.50**	**0.39**	**0.35**	**0.45**	**0.61**	**0.46**

Table A7 continued

(sub-) region	Reef	Depth	Coral cover	Juvenile	Macroalgae	Cover change	Composition	Coral Index
Mackay–Whitsunday	Hayman	5	0.27	1.00	0.84	0.79	0	0.58
	Border	5	0.53	0.57	1.00	0.18	0	0.46
	Hook	2	0.21	0.44	0.59	0.34	0	0.32
		5	0.36	0.29	0.44	0.38	1.00	0.49
	Double Cone	2	0.03	0.21	0	0.13	0	0.08
		5	0.23	0.19	0	0.28	0	0.14
	Daydream	2	0.02	0.69	0	0.43	0	0.23
		5	0.09	1.00	0.91	0.46	0	0.49
	Dent	2	0.38	0.33	0	0.12	0	0.17
		5	0.42	0.30	0	0.25	0	0.19
	Shute Harbour	2	0.81	0.51	0.50	0.41	1.00	0.65
		5	0.43	0.59	0.95	0.47	1.00	0.69
	Pine	2	0.12	0.30	0	0.17	0	0.12
		5	0.22	0.34	0	0.40	0	0.19
	Seaforth	2	0.24	0.32	0	0.15	0.50	0.24
		5	0.20	0.39	0	0.25	0	0.17
Poor			**0.28**	**0.47**	**0.33**	**0.33**	**0.22**	**0.32**
Fitzroy	Barren	2	0.88	0.59	0.60	0.54	0.50	0.62
		5	0.95	0.09	0	0.64	0.50	0.44
	North Keppel	2	0.65	0.06	0	0.25	1.00	0.39
		5	0.40	0.09	0	0.28	0.50	0.26
	Middle	2	0.27	0.17	0	0.18	0	0.12
		5	0.28	0.22	0	0.17	0	0.14
	Keppels South	2	0.75	0.21	0	0.41	0	0.27
		5	0.50	0.20	0	0.30	0.50	0.30
	Pelican	2	0.10	0.12	0	0.65	0	0.18
		5	0.38	0.31	0	0.52	1.00	0.44
Poor			**0.52**	**0.21**	**0.06**	**0.40**	**0.40**	**0.32**

Table A8 Environmental covariates for coral locations. For chlorophyll a (Chl *a*), total suspended solids (TSS) estimated from a square of nine 1km square pixels adjacent to each reef location. Mean of wet season median concentrations over the 2019-2023 wet seasons. Medians for each year were estimated based on the distributions of measured concentrations in samples taken within water-types. These distributions were resampled in proportion to the time each location was classified into one of four water-types based on water colour extracted from Sentinel satellite imagery (Gruber *et al.* 2024). Values exceeding Reef wide wet-season guidelines of 0.63 µgL^{-1} Chl *a*, and 2.4 mgL^{-1} for TSS, based on season adjustments to annual Guideline values (GBRMPA 2010, Gruber *et al.* 2024)

(sub-)region	**Reef**	Wet season Chl *a* (µgL^{-1})	Wet season TSS (mgL^{-1})
Barron–Daintree	Low Isles	0.30	1.46
	Snapper North	0.41	2.10
	Snapper South	0.42	2.15
Johnstone Russell–Mulgrave	Fitzroy East	0.29	1.40
	Fitzroy West LTMP	0.34	1.68
	Franklands East	0.28	1.31
	Franklands West	0.31	1.49
	High East	0.34	1.67
	High West	0.42	2.14
Herbert–Tully	Barnards	0.37	1.84
	Dunk North	0.41	2.09
	Dunk South	0.42	2.18
	Bedarra	0.45	2.34
Burdekin	Palms East	0.27	1.29
	Palms West	0.33	1.64
	Lady Elliot	0.51	2.65
	Pandora North	0.38	1.93
	Pandora	0.39	2.00
	Havannah North	0.31	1.52
	Havannah	0.33	1.66
	Magnetic	0.50	2.59
Mackay–Whitsunday	Hayman	0.26	1.22
	Border	0.28	1.35
	Hook	0.30	1.44
	Double Cone	0.32	1.57
	Daydream	0.36	1.84
	Shute Harbour	0.39	2.00
	Dent	0.39	1.99
	Pine	0.42	2.17
	Seaforth	0.36	1.81
Fitzroy	Barren	0.28	1.34
	North Keppel	0.38	1.95
	Middle	0.39	1.96
	Keppels South	0.40	2.02
	Pelican	0.53	2.77

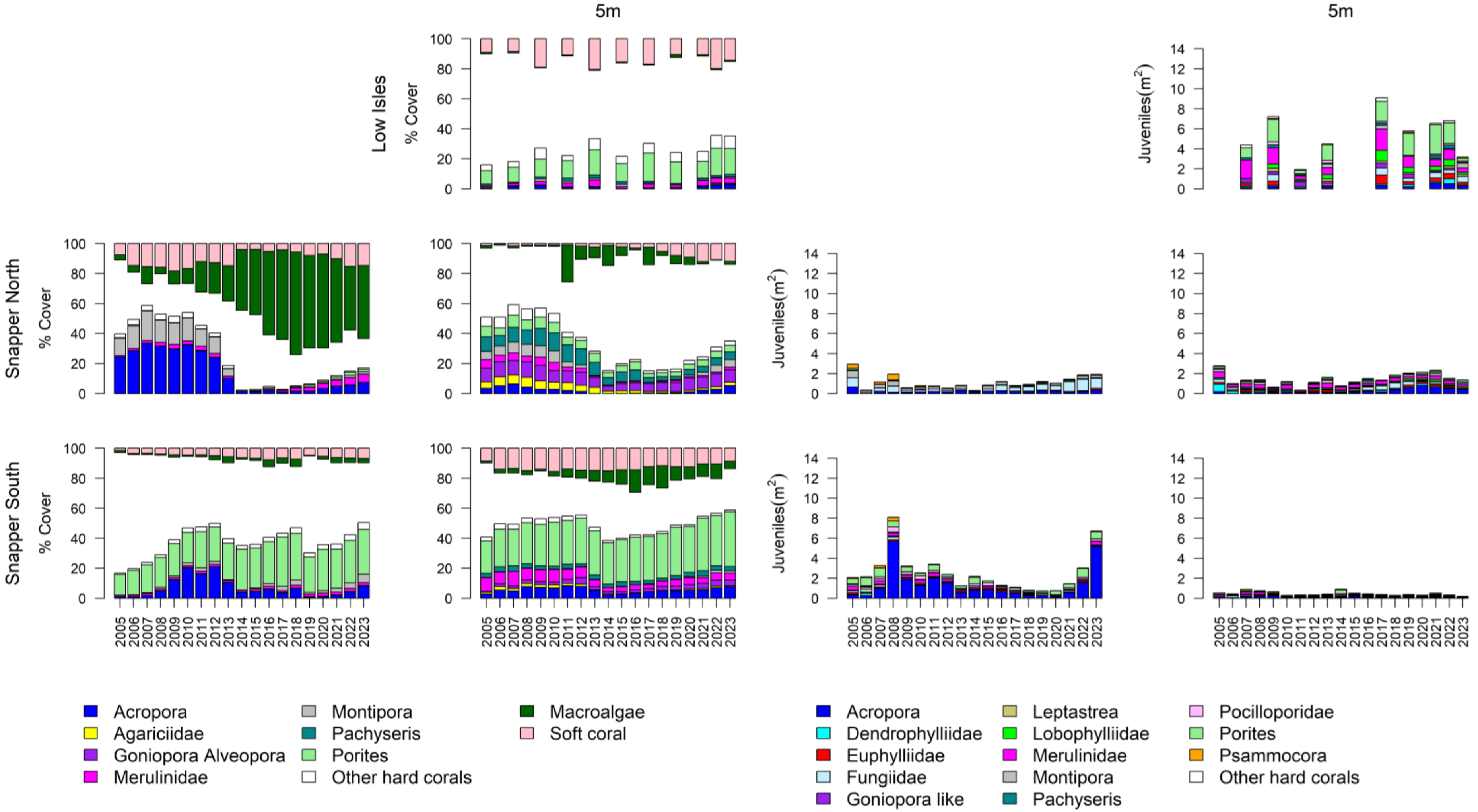

Figure A1 Barron–Daintree sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends relevant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

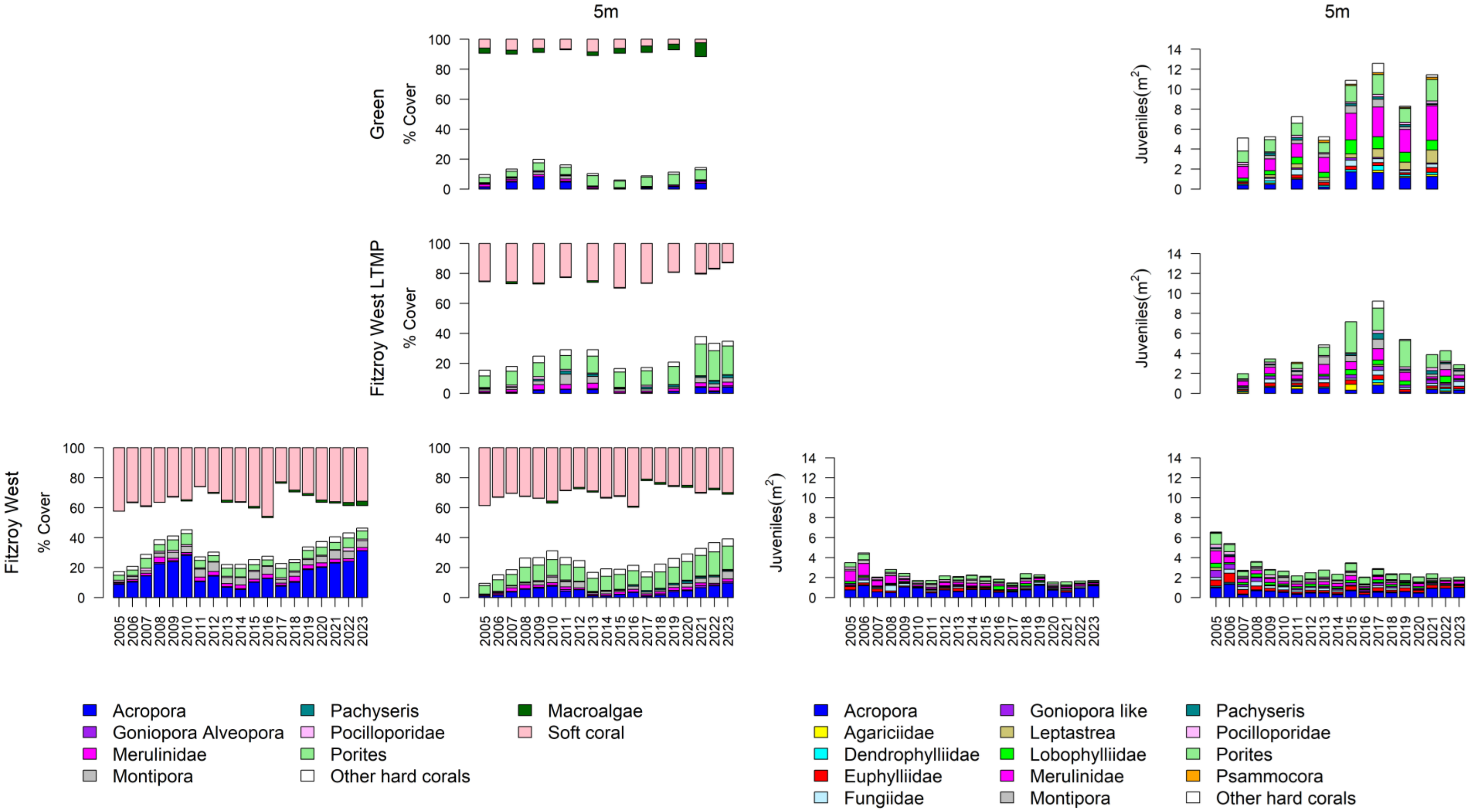

Figure A2 Johnstone Russell–Mulgrave sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate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

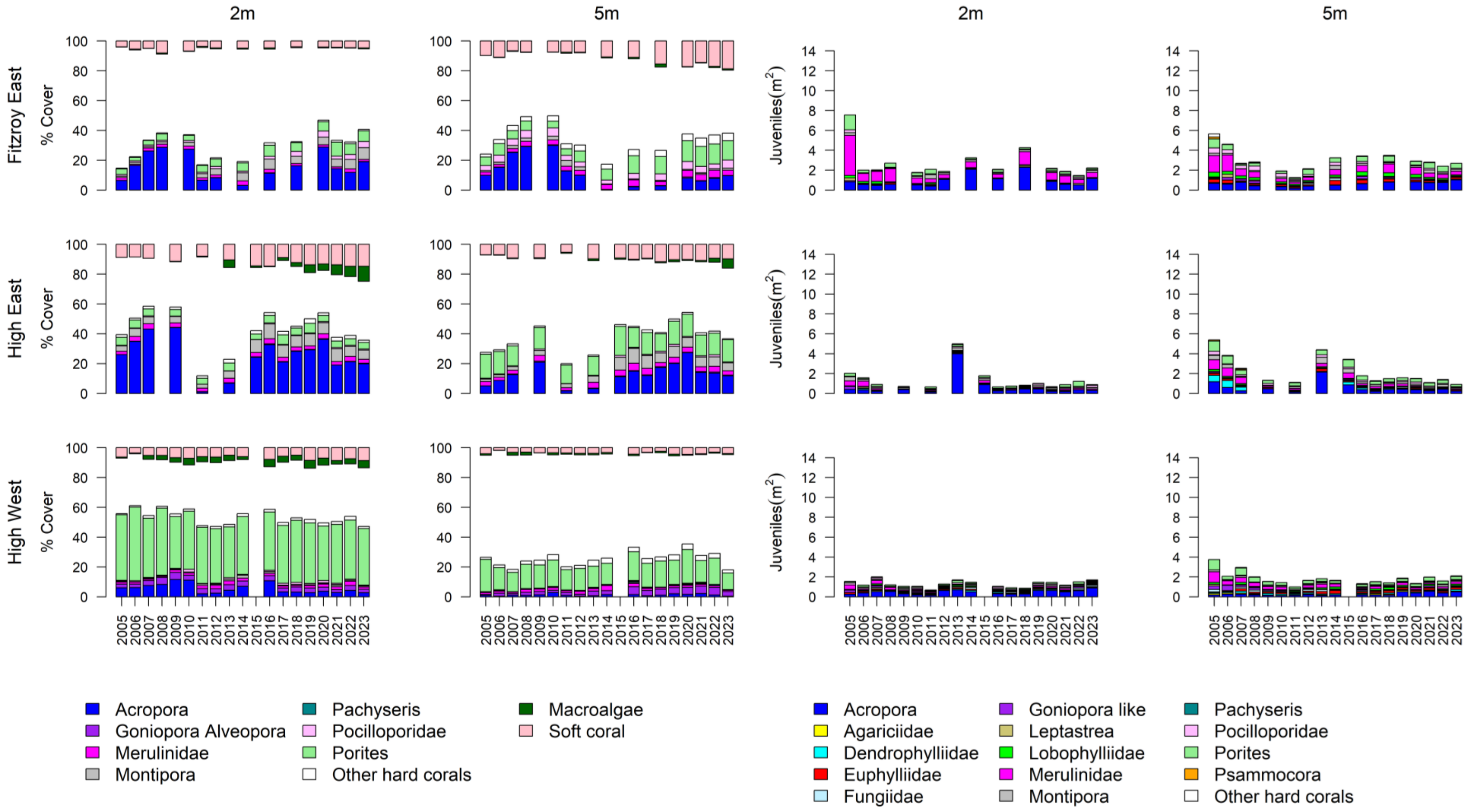

Figure A2 continued

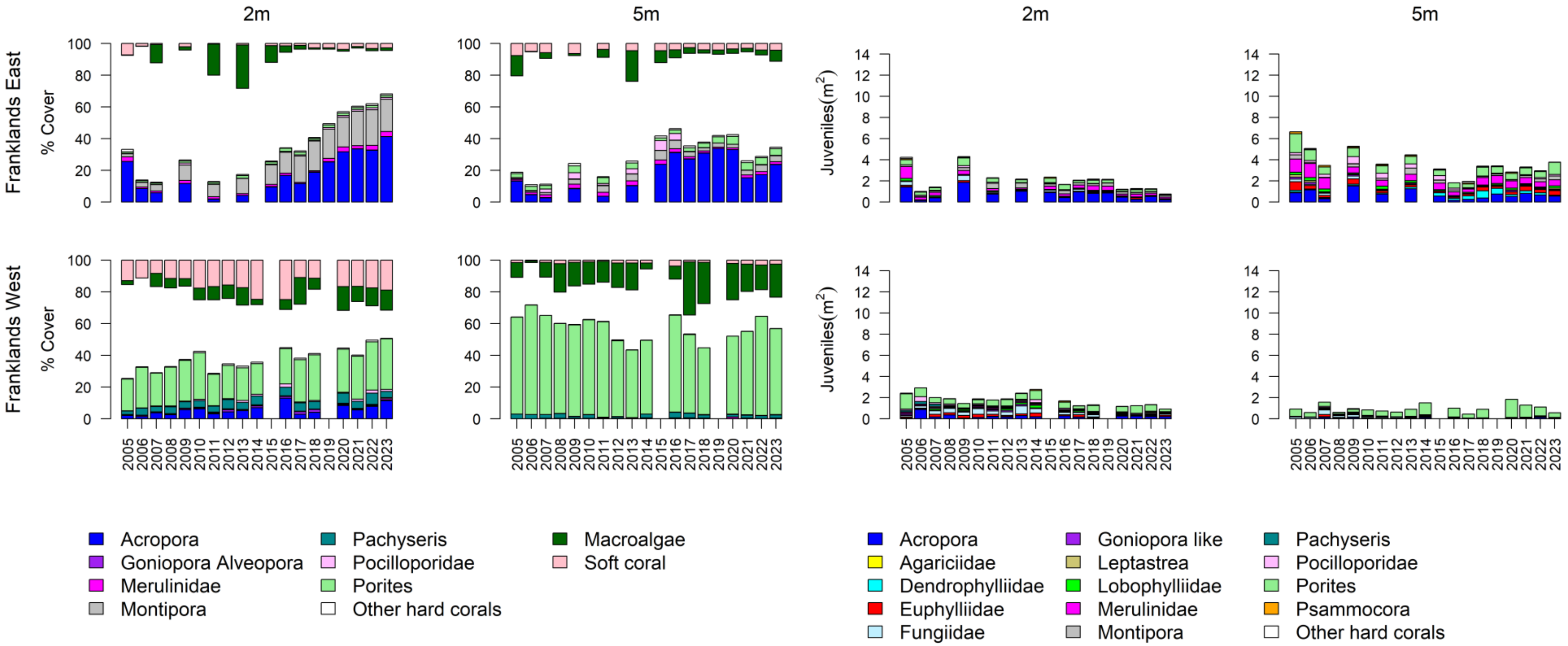

Figure A2 continued

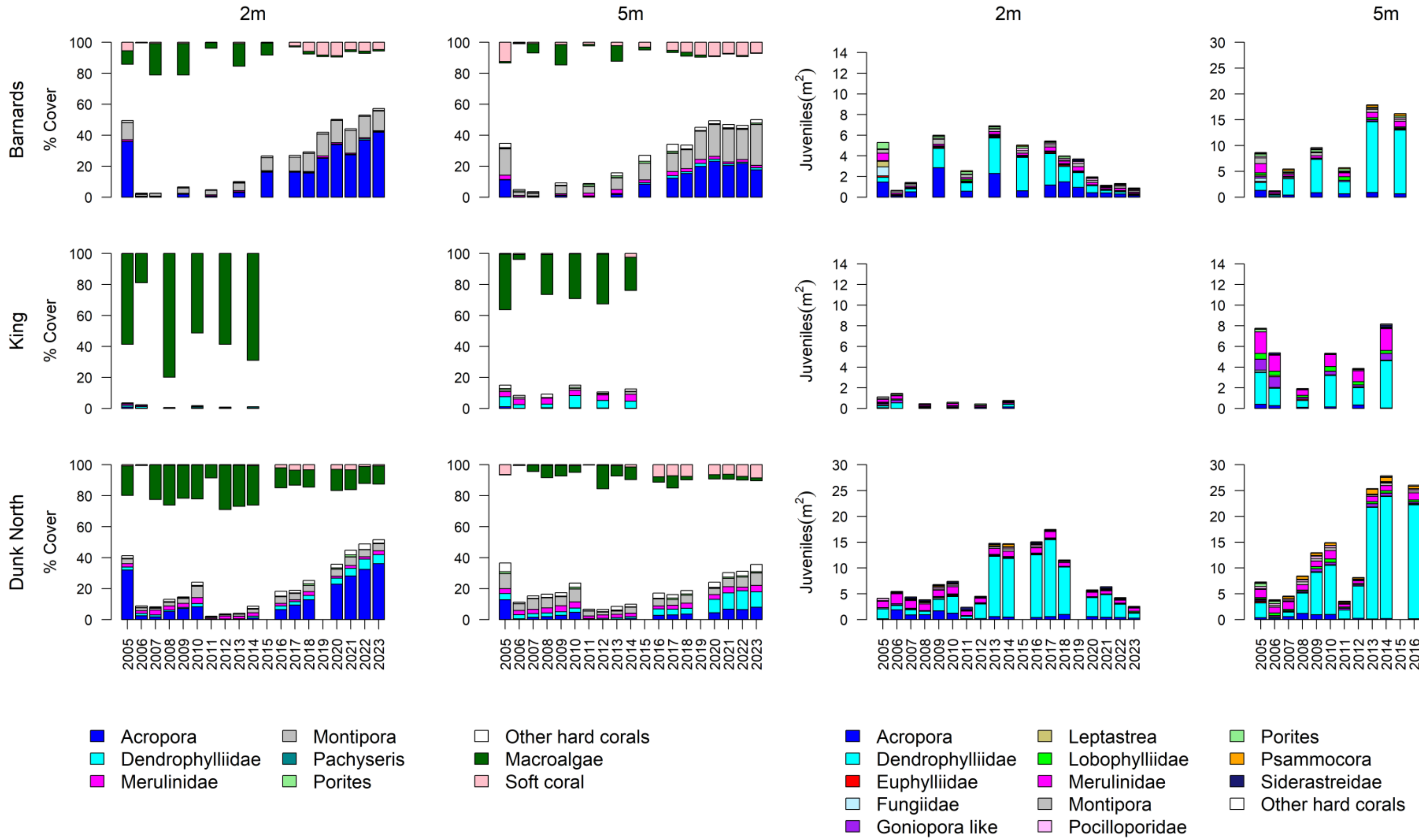

Figure A3 Herbert–Tully sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends with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

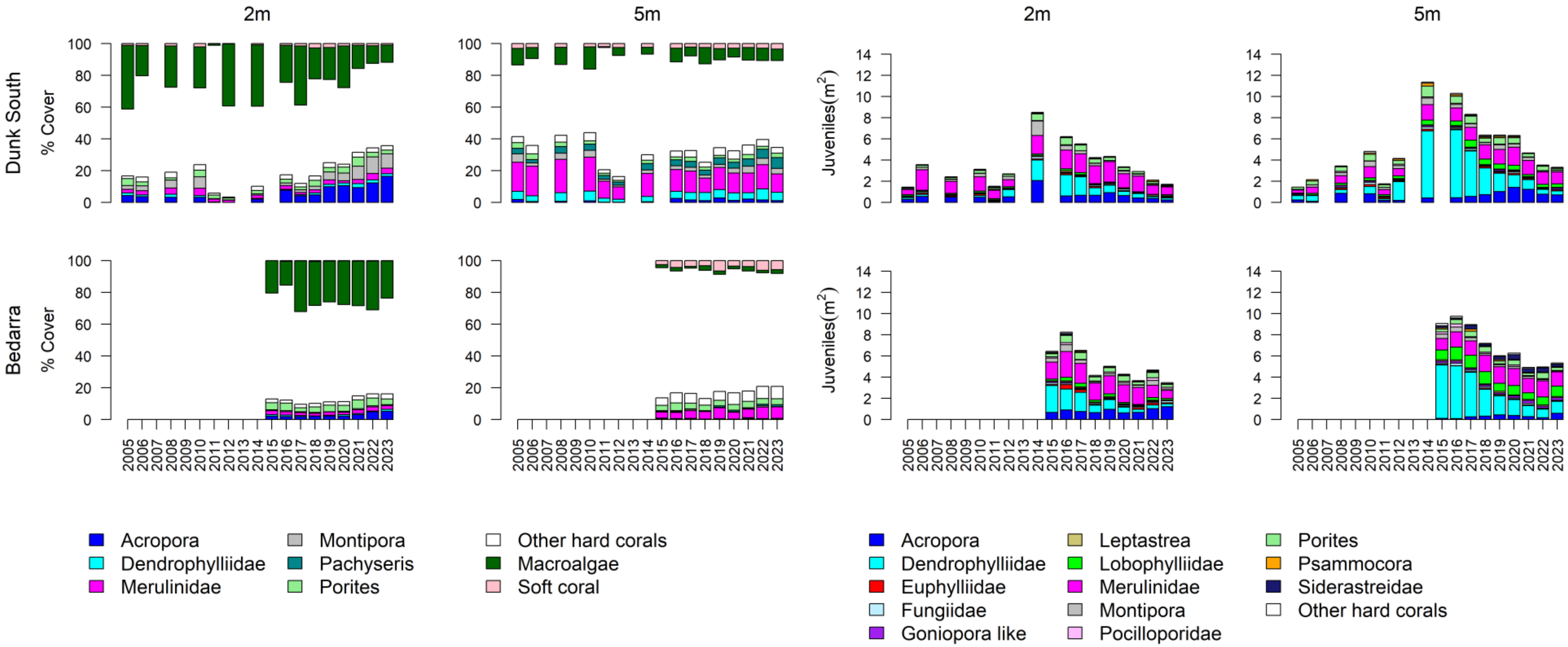

Figure A3 continued

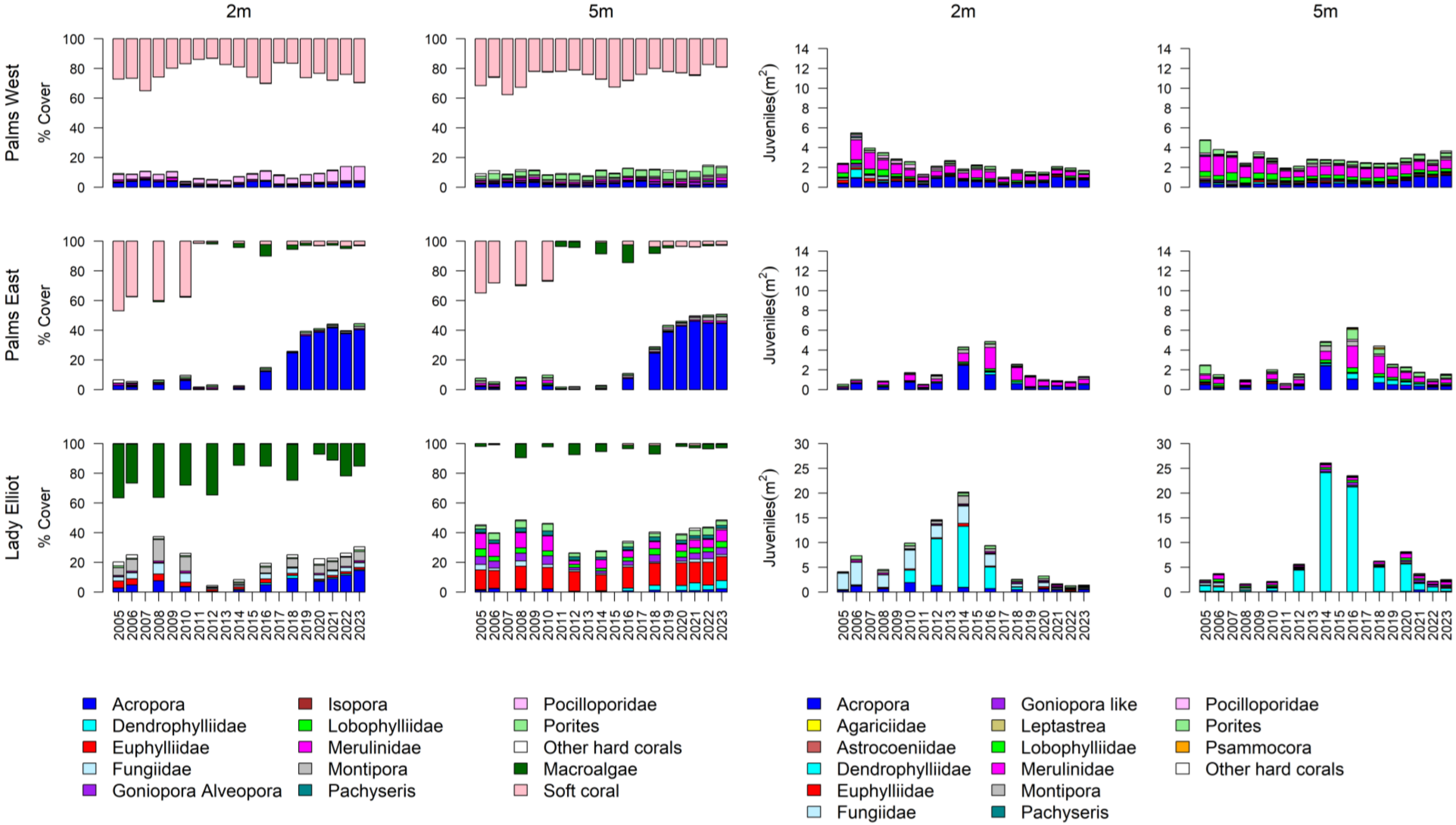

Figure A4 Burdekin region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends with relevant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

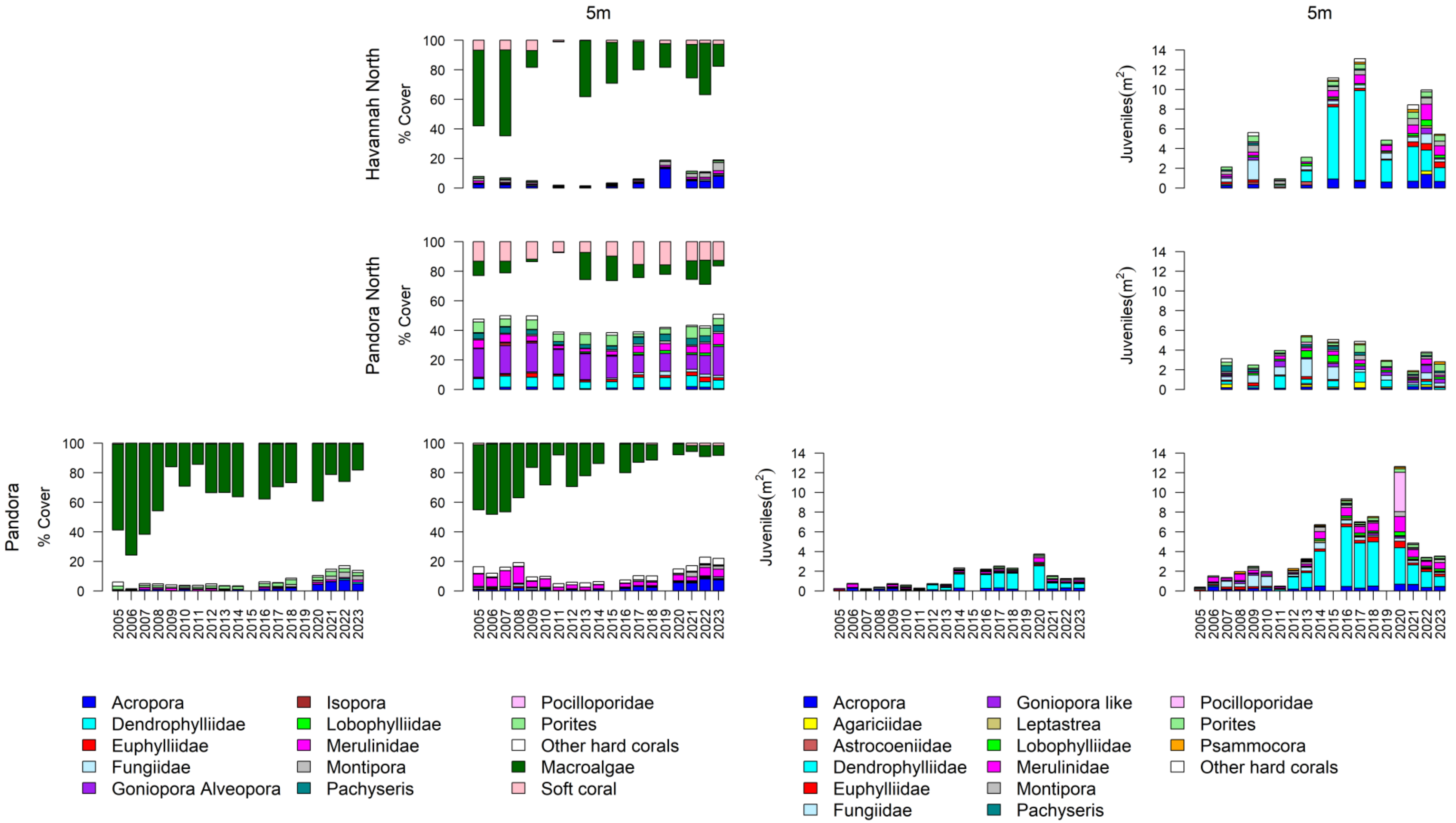

Figure A4 continued

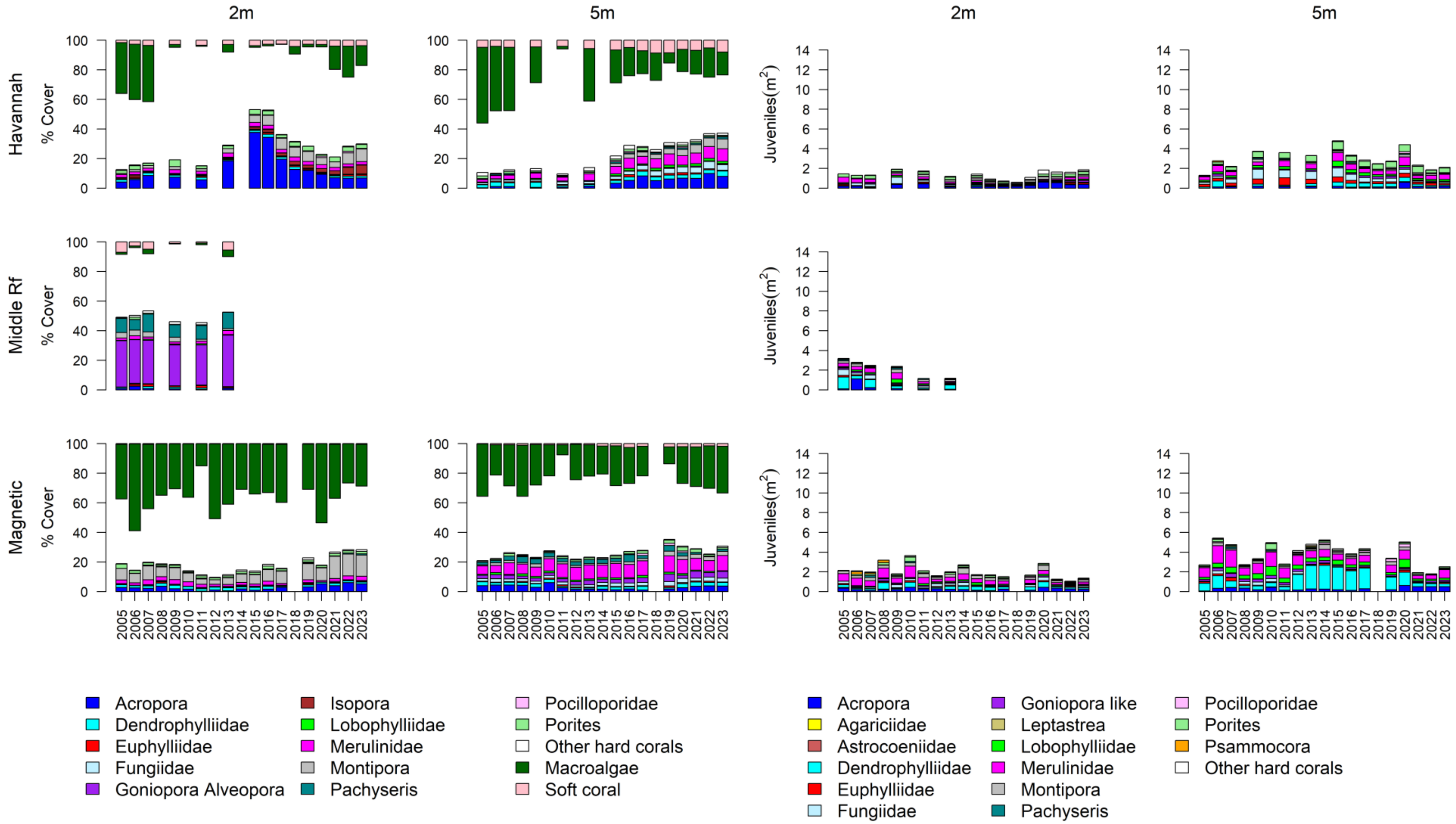

Figure A4 continued

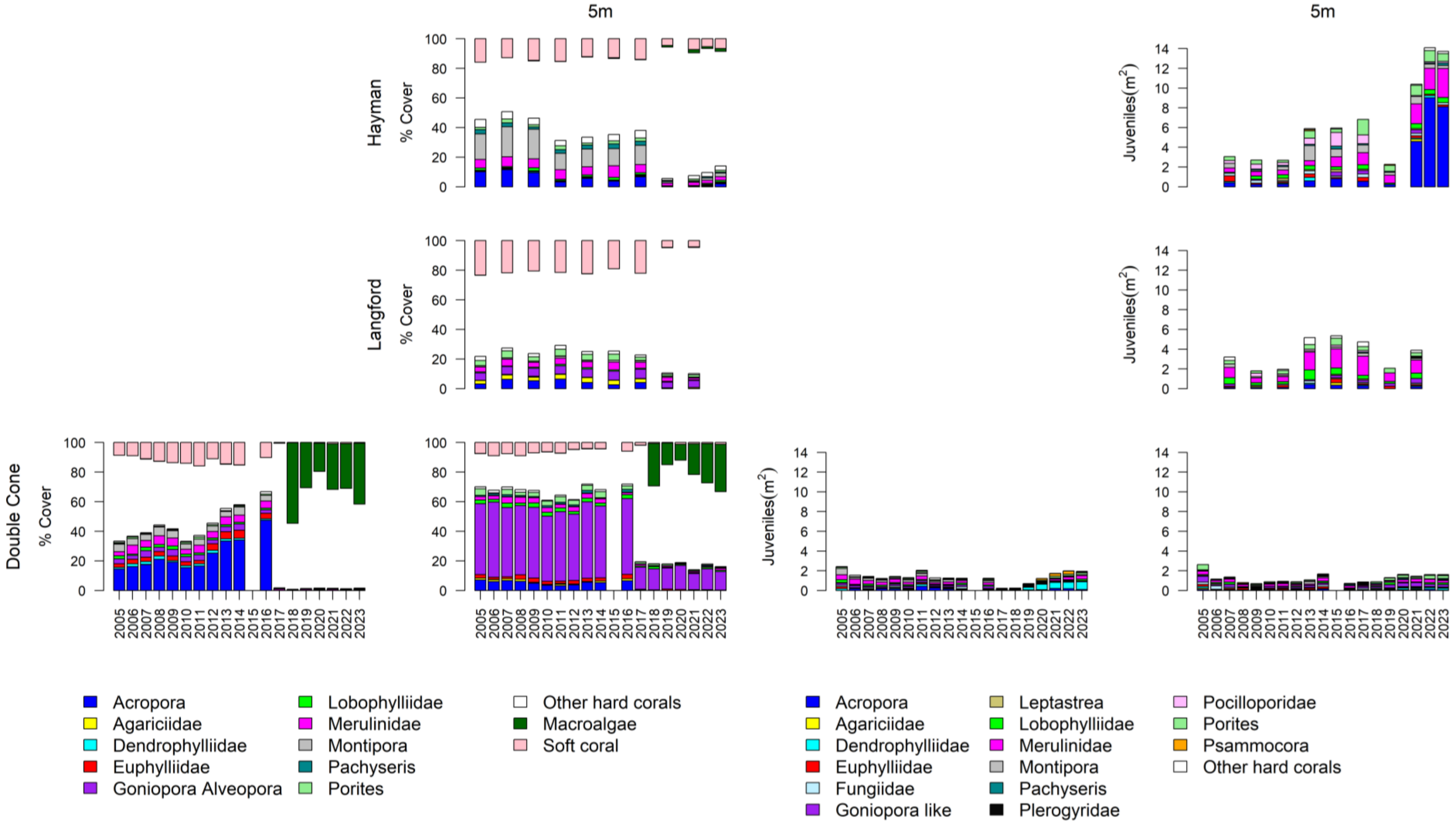

Figure A5 Mackay–Whitsunday region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends with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

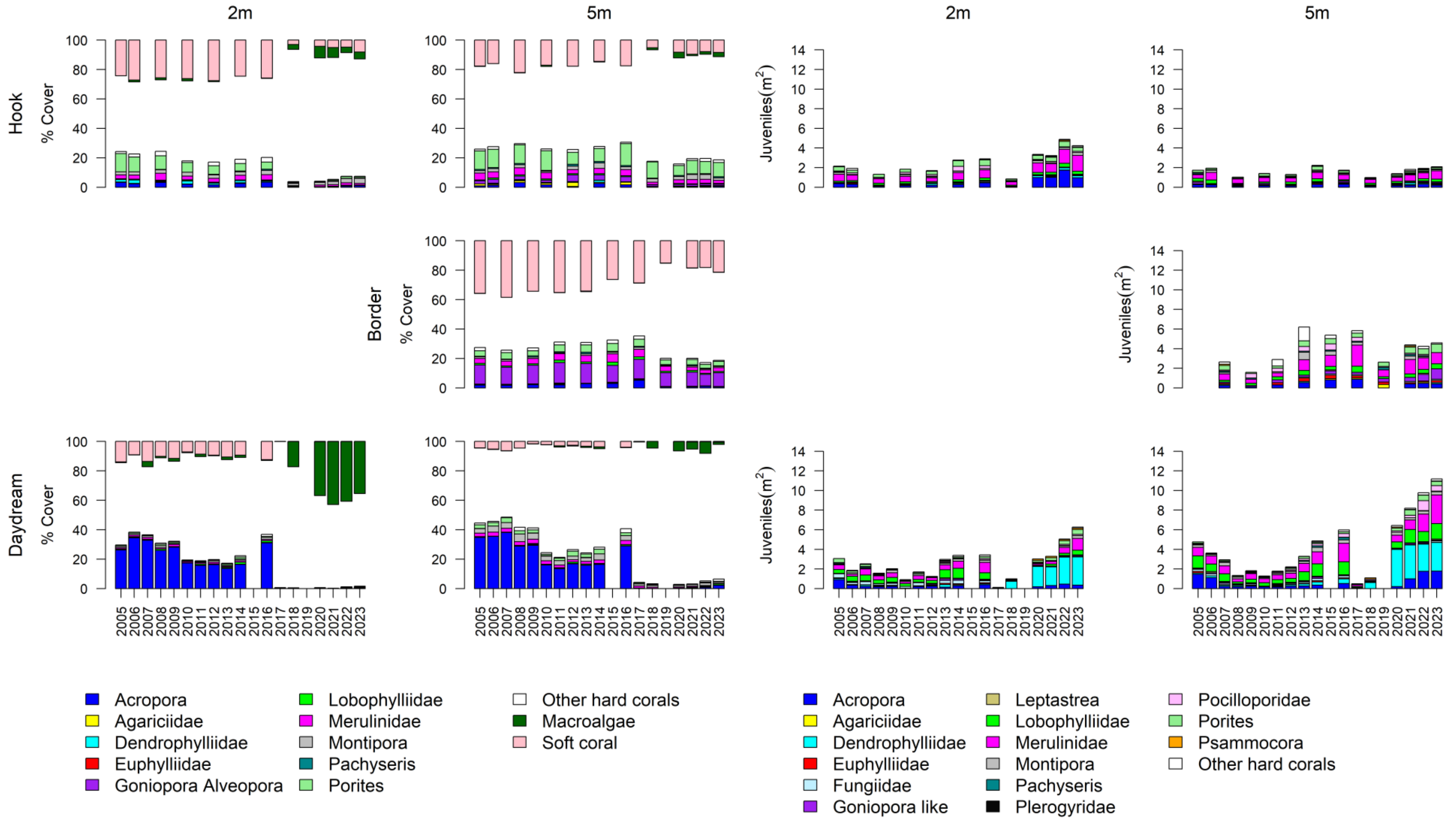

Figure A5 continued

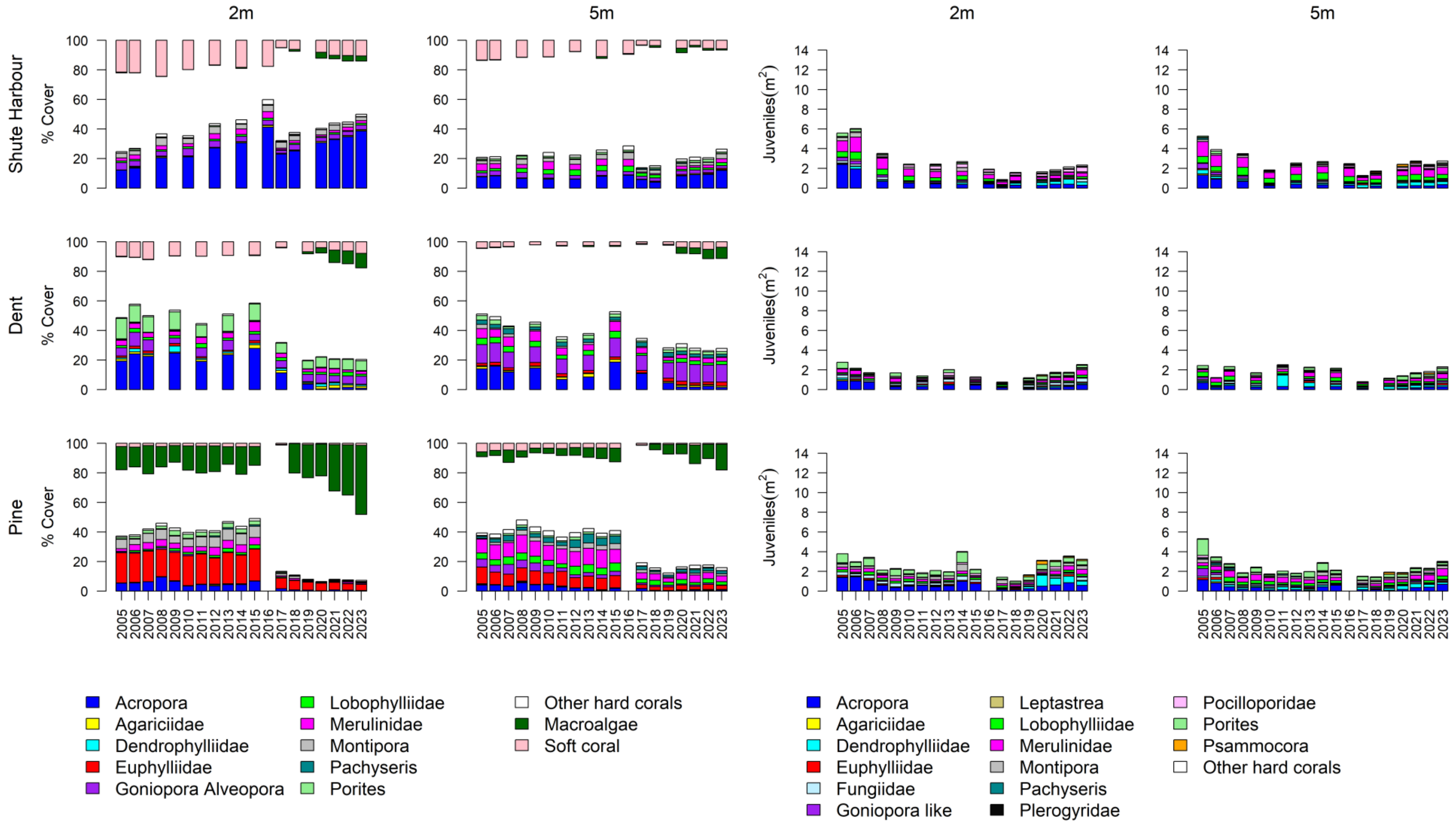

Figure A5 continued

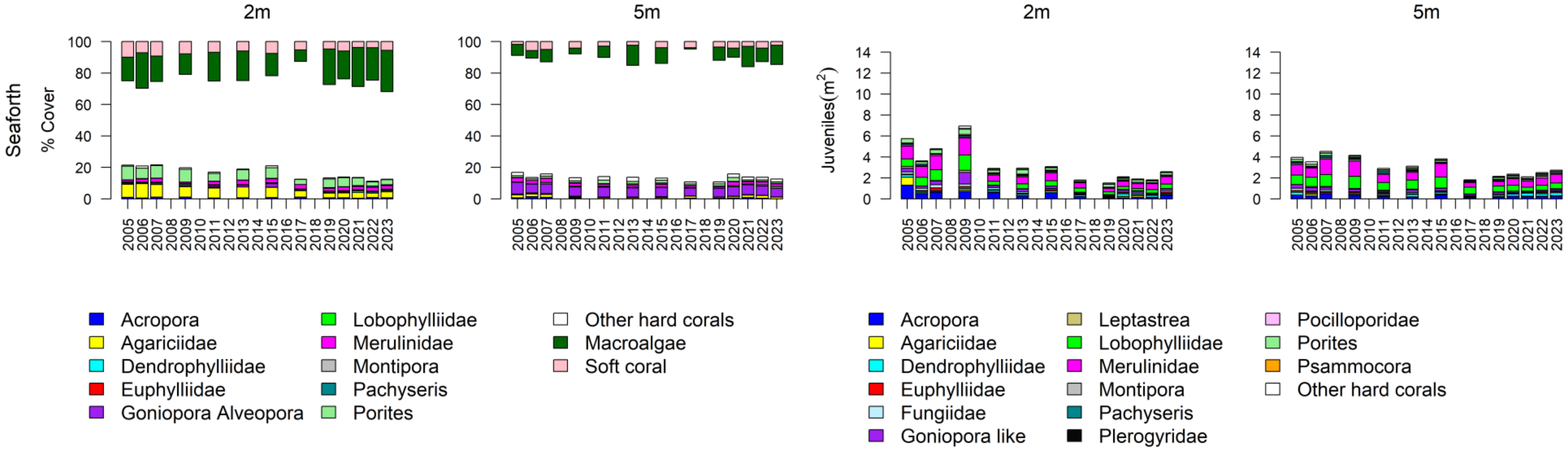

Figure A5 continued

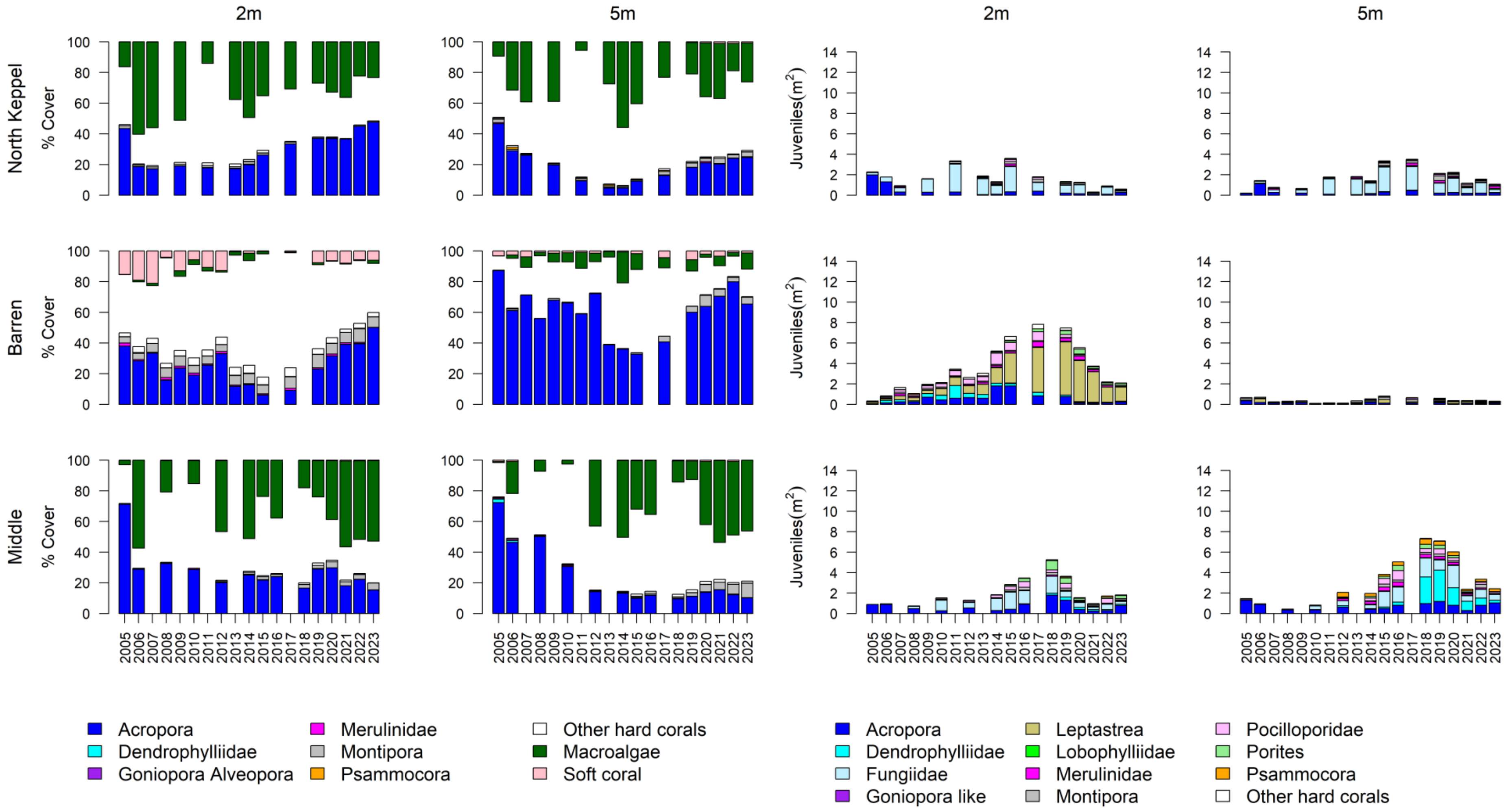

Figure A6 Fitzroy region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends with relevant groupings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

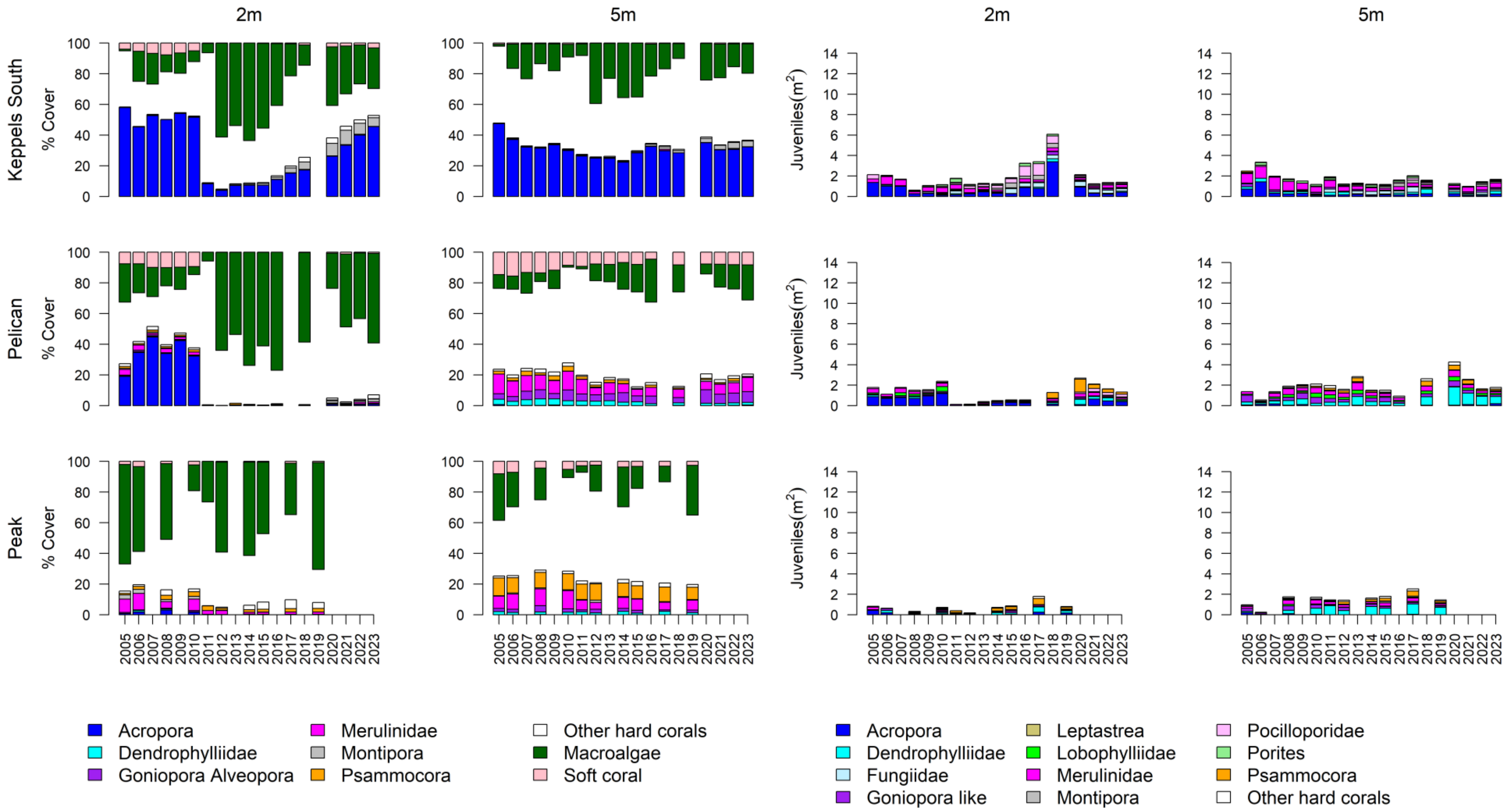

Figure A6 continued

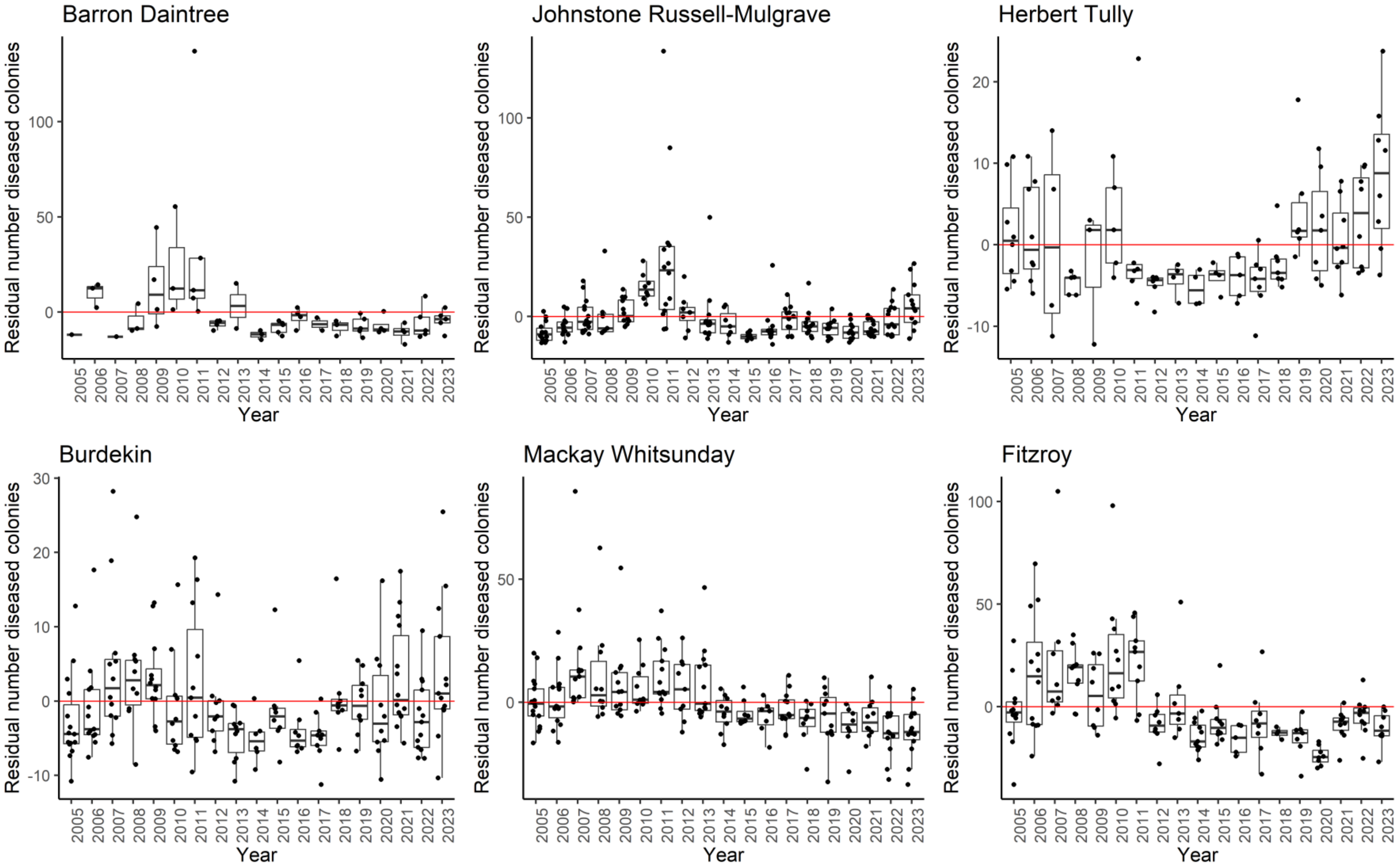

Figure A7 Coral disease by year in each region. Boxplots include the number of coral colonies suffering ongoing mortality attributed to either disease, sedimentation or ‘unknown causes’ for each reef, depth and year. Data are standardised to the reef and depth mean across years (see section 2.3.3).

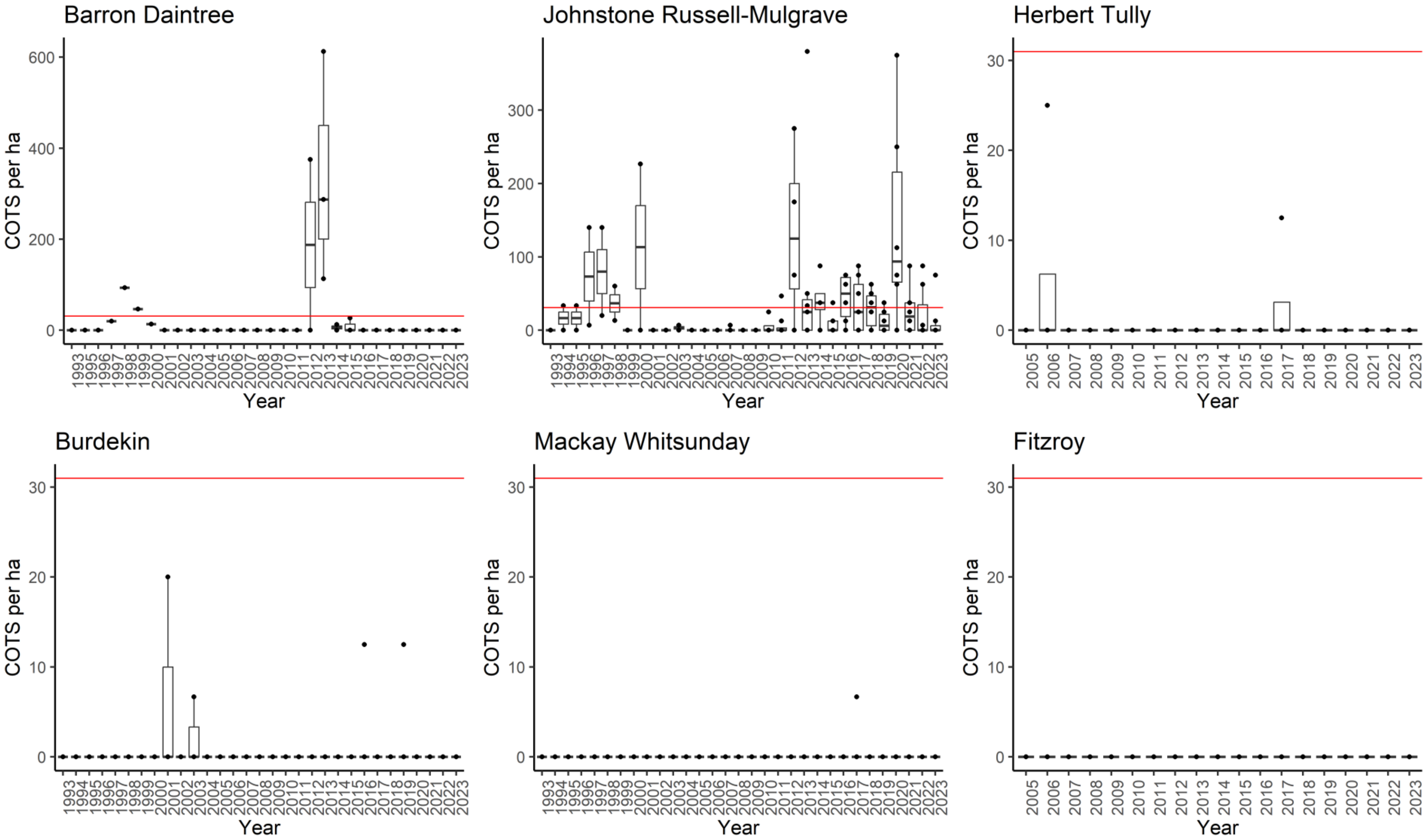

Figure A8 Crown -of-thorn-starfish mean density (individuals/ha) by year in each region. Red line indicates outbreak densities of 31 individuals per hectare (see section 2.3.3 for derivation).

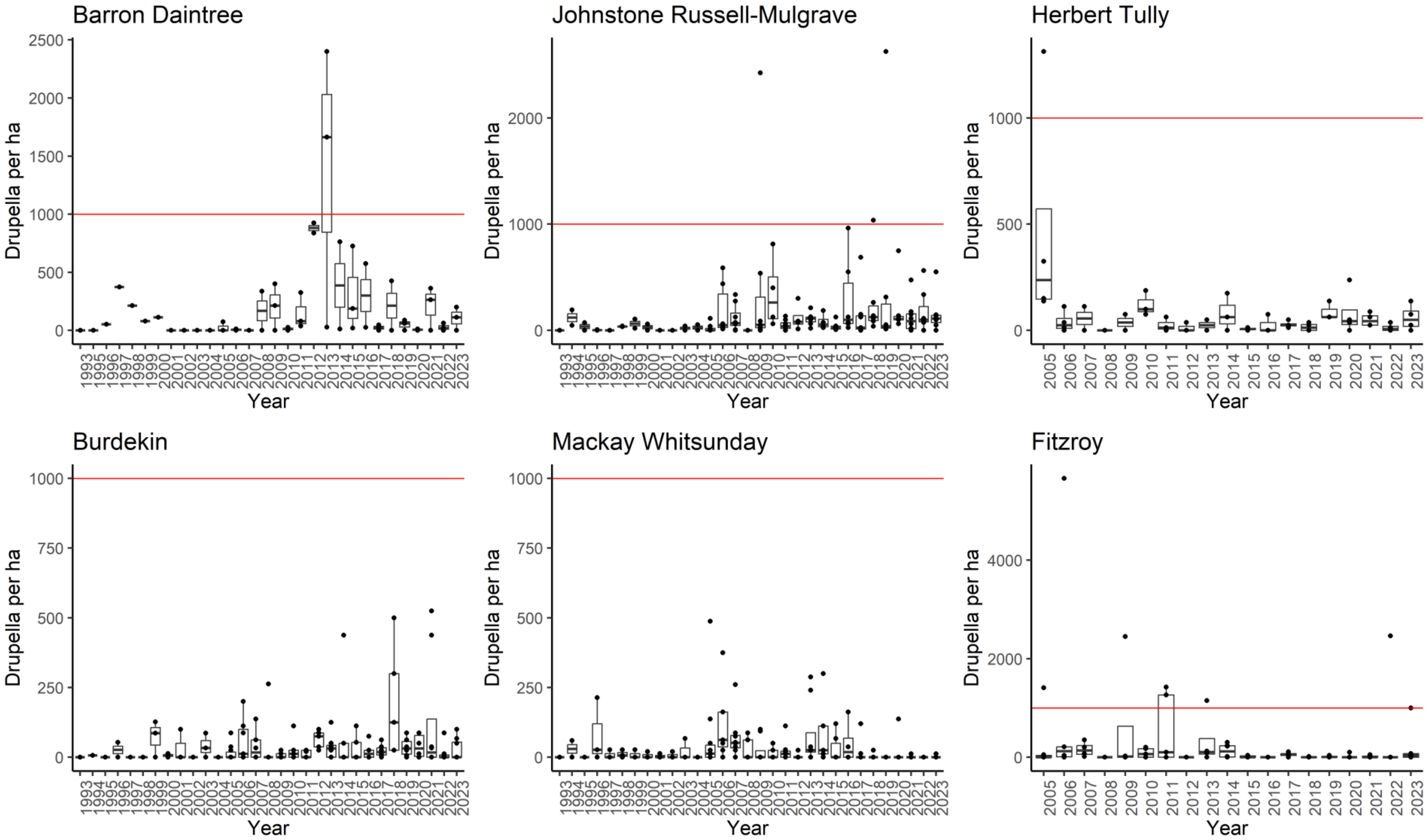

Figure A9 Mean density of *Drupella* by year in each (sub-)region. Red line indicates densities of *Drupella* which have detrimental impact on coral communities (see section 2.3.3 for derivation).

Table A9 Percent cover of hard coral genera 2023. Genera for which cover did not exceed 1% on at least one reef-depth or were unidentified to genus level are grouped as "Other".

(sub-)region	Reef	Depth	*Acropora*	*Alveopora*	*Caulastrea*	*Cyphastrea*	*Diploastrea*	*Dipsastraea*	*Echinopora*	*Favites*	*Galaxea*	*Goniastrea*	*Goniopora*	*Hydnophora*	*Isopora*	*Leptoseris*	*Lobophyllia*	*Merulina*	*Montipora*	*Mycedium*	*Oxypora*	*Pachyseris*	*Paragoniastrea*	*Pavona*	*Pectinia*	*Platygyra*	*Pocillopora*	*Podabacia*	*Porites*	*Seriatopora*	*Turbinaria*	Other
Baron Daintree	Low Isles	5	2.87	0	0.10	0.17	0.17	0.37	1.40	0	3.92	0.17	0.63	0	0	0	1.53	0.50	0.80	0.17	0.57	1.47	0.10	0.33	0.33	0.17	0.07	0.07	17.38	0.10	0.10	1.69
	Snapper North	2	7.38	0	0	0	0	0.04	5.25	0	0.17	0.04	0	0	0	0	0.04	0.08	1.63	0	0	0	0	0.08	0	0.04	0.08	0	1.58	0	0	0.71
		5	5.32	0	0.06	0	0	0.25	0	0.12	0.75	0	8.28	0.13	0.69	0.38	0.12	0.19	5.00	0.50	0.06	5.19	0	1.82	0.25	0.25	0	0	4.38	0	0	1.31
	Snapper South	2	8.13	0	0	0.13	0	0.17	0	0.33	2.54	1.13	0.21	0	0	0	0.12	0	5.51	0	0	0	0.04	0.25	0	0.04	0.71	0	29.89	0	0.12	1.21
		5	7.75	0	4.31	0	0	0	0.31	0	0	0	3.69	0	0	0	0.12	0	1.50	0	0	2.81	0	0.69	0	0	0	0	36.44	0	0.12	1.06
Johnstone Russell-Mulgrave	Fitzroy East	2	19.00	0	0	0.19	0	0.12	0.12	0.25	0.12	0.50	0.06	0	0	0	0.44	0	7.88	0	0	0	0	0	0	0.25	4.06	0	7.19	0	0	0.42
		5	9.56	0	0	0	0.94	0	1.69	0.44	0.94	0.25	0.19	0.06	0	0	1.75	0.06	1.31	0	0.12	0.06	0	0.12	0	0.50	5.31	0	13.06	0	0	1.80
	Fitzroy West	2	31.19	0	0	0	0.38	0.25	1.38	0.06	0.19	0	0.19	0.06	0	0	1.07	0	4.58	0	0	0	0	0.06	0	0.12	1.13	0	5.33	0	0	0.43
		5	9.69	0	0.38	0.06	0.75	0.31	0.38	0	0.69	0.06	0.88	0.06	0	0.12	1.62	0.25	5.10	0.19	0.50	0.75	0.06	0	0	0.06	0.38	0	15.59	0	0	1.25
	Fitzroy West LTMP	5	4.12	0	0.49	0.17	0	0.16	0.56	0.06	0.79	0.06	0.76	0.07	0	0	0.76	0.13	2.89	0.35	0.40	1.89	0	0.07	0.16	0.20	0.10	0.10	19.10	0.03	0	1.31
	Franklands East	2	41.25	0	0	0.12	0	0.06	2.12	0.12	0.19	0	0.06	0.06	0.31	0	0	0.69	20.44	0	0	0	0	0.12	0	0	1.12	0	1.25	0	0	0.25
		5	23.62	0	0	0	0	0.13	0.31	0.12	0.69	0	0.19	0.06	0	0	0.12	0.44	3.70	0	0	0.06	0	0.13	0.12	0.44	0.25	0	4.07	0	0	0.18
	Franklands West	2	11.45	0	0	0	0	0	0.56	0	0	0	0.38	0.12	0	0	0	0	0.69	0	0	3.83	0	0	0	0.19	0.75	0	32.01	0.50	0	0.25
		5	0	0	0	0	0	0	0.38	0	0.06	0	0	0	0	0	0.06	0	0	0	0	1.94	0	0	0	0	0	0	54.19	0.31	0	0
	High East	2	19.84	0	0	0.06	0	0.12	0.88	0.38	0.12	0.44	0.38	0.12	0	0	0.50	0	6.21	0	0.06	0	0	0.12	0	0.44	0.63	0	4.51	0	0.25	0.68
		5	11.82	0	0.06	0.06	0	0.06	1.63	0.38	0.19	0.06	0.38	0.06	0	0	0.06	0.12	5.44	0	0	0	0.06	0.06	0	0.25	0.25	0.19	14.78	0	0	0.75
	High West	2	2.75	0	0	0.06	0.06	0.44	0.25	0.19	0.69	0.06	2.39	0.12	0	0	0	0	0.63	0	0	0	0	0.12	0	0.25	0.56	0	38.00	0	0	0.63
		5	0.38	0	0	0.06	0	0.38	0	0.12	0.44	0.12	3.12	0	0	0	0.06	0	0.06	0	0	0.19	0	0.62	0.06	0.06	0.06	0	11.25	0	0	1.11
Herbert Tully	Barnards	2	41.97	0	0	0.19	0	0.12	0	0.06	0.31	0	0	0	0	0	0	0	12.70	0	0	0	0	0	0	0.12	0.69	0	0.31	0	0.38	0.37
		5	17.63	0.19	0	0.06	0	0.19	0.31	0.19	0	0	0.19	0	0	0	0.13	0	26.42	0.25	1.12	0.19	0	0	0	0.44	0.25	0	0.75	0	1.25	0.44
	Dunk North	2	36.19	0	0	0.81	0	0.50	0.06	0.38	0	0.12	0.06	0.38	0	0	0	0	4.56	0.06	0	0	0	0	0	0	1.75	0	0.38	0	5.19	1.16
		5	7.94	0	0	0.56	0	0.81	0	1.25	0.25	0.12	0.12	0.50	0	0	0.50	0.19	8.06	0.31	0.06	0.12	0.06	0.06	0	0.12	1.75	0.19	0.69	0	9.31	2.56
	Dunk South	2	16.39	0	0	1.69	0	0.38	0.06	0.63	1.06	0	0.19	0.06	0	0	0.12	0	9.01	0	0	0	0.06	0.50	0	0.44	0.06	0	2.32	0	1.75	1.06
		5	1.13	0	0.06	0.94	0.06	2.38	0.38	1.38	0.06	1.57	0.44	0	0	0	0.56	1.32	3.38	1.82	0.82	6.88	0.19	0.25	0.50	1.01	0.19	0.38	2.70	0	5.26	1.00
	Bedarra	2	5.19	0	0	0.56	0	0.38	0.06	0.63	0.19	0	0.06	0.31	0	0	0.12	0.19	0.44	0	0.25	0	0	0.31	0	0.25	0.44	0.06	3.63	0	1.19	1.82
		5	0.44	0	0.44	0.06	0.12	4.31	0	0.56	0	0	3.31	0.06	0	0	1.81	0.81	0.19	0.38	0.19	1.25	0	0.50	0.31	0.19	0.25	0.56	3.75	0	0.19	1.18

Region	Reef	Depth	*Acropora*	*Alveopora*	*Caulastrea*	*Cyphastrea*	*Diploastrea*	*Dipsastraea*	*Echinopora*	*Favites*	*Galaxea*	*Goniastrea*	*Goniopora*	*Hydnophora*	*Isopora*	*Leptoseris*	*Lobophyllia*	*Merulina*	*Montipora*	*Mycedium*	*Oxypora*	*Pachyseris*	*Paragoniastrea*	*Pavona*	*Pectinia*	*Platygyra*	*Pocillopora*	*Podabacia*	*Porites*	*Seriatopora*	*Turbinaria*	Other
Burdekin	Palms East	2	40.32	0	0	0	0	0.38	0	0.12	0	0.06	0	0	0	0	0.06	0	1.56	0	0	0	0	0	0	0.06	0.12	0	1.50	0	0	0.25
		5	44.56	0	0	0.06	0.06	0.19	0.12	0.31	0.19	0.19	0.12	0	0	0	0.06	0.06	2.88	0	0	0	0	0	0	0.25	0.38	0	1.06	0	0	0.30
	Palms West	2	3.00	0	0	0.06	0	0.31	0.12	0	0.06	0	0.19	0	0	0	0	0	0.31	0	0	0.19	0.06	0	0	0	9.45	0	0	0	0	0.12
		5	1.88	0	0	0.31	0.19	0.38	0	0.31	0.06	0.38	1.44	0	0	0	0.06	0	1.00	0.12	0.06	0.06	0	0.12	0.06	0.19	0.94	0	4.50	0.06	0	2.00
	Havannah North	5	7.93	0	0.03	0.20	0	0.30	0.17	0.07	0.27	0.17	0.07	0.33	0	0	0	0.20	5.53	0.10	0.23	0.23	0	0.10	0.07	0	0	0.03	1.00	0	0.49	1.45
	Havannah	2	6.88	0	0.06	0	0	0.38	0.94	0.06	1.00	0.12	0.25	0.12	5.94	0	0.19	0	8.63	0.06	0	0	0	0.44	0.13	0.25	0.38	0	2.62	0	0.50	1.00
		5	7.88	0	0.06	0.38	0	0.06	0.75	0.31	0.50	0.06	0.31	0.31	0.44	0	0.88	4.82	6.64	0.06	0.75	1.38	0.06	0.63	1.38	0.12	0.13	0.31	0.75	0	3.70	4.52
	Pandora	2	4.76	0	0	0.62	0	0.19	0.06	0.06	0	0	0.06	0.25	0	0	0	0.19	2.70	0	0	0	0	0	0	0.12	0.19	0	2.06	0	1.06	1.62
		5	7.38	0	0	0.75	3.25	0.94	0.25	0.88	0.56	0.12	0	1.06	0	0	0.12	0.19	2.12	0.06	0.06	0.12	0	0	0	0.38	0.38	0.25	0.25	0	0.56	2.31
	Pandora North	5	0.57	0	0.19	0.13	0	0.26	3.31	0	1.56	0.03	19.56	0.16	0	1.28	0.40	0.98	1.28	1.03	0.75	4.27	0.03	0.50	1.09	0.23	0.10	0.30	4.45	0	5.76	2.61
	Lady Elliot	2	14.47	0	0	0	0	0	0.25	0.12	1.38	0	0.06	0	0	0	0.31	0	6.32	0	0	0	0	2.14	0.06	0.50	0	0	1.06	0	0.69	3.19
		5	2.12	0.94	0	0.31	0	1.06	0	1.19	16.00	0	3.69	0.88	0	0	2.00	0.44	1.06	2.06	2.06	1.94	0.06	0	1.06	0.44	0	1.12	3.00	0	5.19	2.00
	Magnetic	2	5.19	0	0	1.00	0	0.50	0.06	0.75	0.50	0	0.12	0.38	0	0	0.06	0.12	14.31	0	0	0.88	0	0.50	0	0.06	0	0.06	1.62	0	1.19	0.92
		5	3.71	0	0	0.75	0	2.44	0	1.19	0.31	0	4.25	0.50	0	0	0.19	2.94	2.76	0.25	0.06	1.50	0	0	0.38	1.88	0	2.00	1.13	0	2.20	2.25
Mackay–Whitsunday	Hayman	5	1.97	0.03	0	0.20	1.85	0.57	0.13	0.23	0.30	0.43	0.37	0.37	0.07	0	0.80	0.07	2.46	0.10	0.17	0.67	0	0.16	0	0.10	0.27	0.03	1.23	0.10	0.17	1.14
	Border	5	0.66	0.13	0	0	0.20	0.70	0.10	0.17	0.03	0.36	9.33	0	0	0	0.66	0.03	0.79	0.30	0	0.43	0	0.17	0.40	0.39	0.13	0	2.96	0.10	0	0.66
	Hook	2	1.06	0	0	0.31	0	0.31	0	0.12	0	0.12	0	0.06	0	0	0.06	0	3.44	0	0.06	0.25	0	0.12	0	0.19	0.06	0	0.81	0	0.06	0.37
		5	0.69	0	0.06	0.06	0.82	1.19	0.25	0.19	0	0.06	1.06	0	0	0	0.31	0	1.63	0.06	0.06	0.88	0	0.25	0	0.06	0.31	0	9.69	0	0	0.86
	Double Cone	2	0.06	0	0	0	0	0.12	0.19	0	0	0	0	0	0	0	0.12	0.06	0.19	0	0	0.06	0	0	0	0.06	0.12	0	0.44	0	0.06	0.06
		5	0	0	0.06	0.06	0	0.19	0	0.06	0.19	0.19	12.50	0	0	0	0.81	0.19	0.06	0	0	0.38	0.06	0	0.12	0	0	0.38	0.31	0	0	0.49
	Daydream	2	0.44	0	0	0.06	0	0	0	0	0	0	0.06	0	0	0	0	0	0.12	0	0	0	0	0	0	0	0.06	0	0.50	0.06	0.19	0.12
		5	2.25	0	0	0.06	0	0	0	0.12	0	0.06	0	0	0	0	0.06	0.06	0.94	0	0.50	0.06	0	0	0	0	0.19	0.12	0.31	1.12	0	0.56
	Dent	2	0.94	0	0.06	0.06	0	0.19	0.50	0	0.31	0	5.50	0	0	0.06	1.25	0.57	0.25	0	0	0.12	0	1.31	0.19	0.12	0.06	0.06	6.63	0	0.88	1.24
		5	1.44	0	0	0	0	0.25	0.31	0.13	2.19	0.12	11.88	0	0	0.25	1.07	0.25	0.44	0.12	1.07	1.81	0	0.75	0.88	0.44	0.38	0.19	1.81	0	0.06	2.01
	Shute Harbour	2	38.64	0	0.44	0	0	0.06	0.25	0.06	0.06	0.06	3.07	0	0.13	0	1.19	0.19	2.57	0.06	0.12	0.06	0.06	0.69	0	0.19	0.38	0.06	0.13	0	0.06	1.37
		5	11.94	0	0.06	0	0	0	0.25	0.06	0.88	0.12	1.88	0.50	0.12	0	0.94	0	3.31	0.62	1.00	0.25	0.06	0.50	0.69	0.19	0.75	0.25	0.94	0	0	0.92
	Pine	2	0.38	0	0.06	0.06	0	0.06	0	0.06	4.06	0.06	0.06	0.06	0	0	0.06	0	0.57	0	0.06	0.06	0	0	0.31	0	0.12	0	0.44	0	0.06	0.69
		5	0.63	0	0	0	0	0.38	0.31	0.12	2.63	0	0.31	0	0	0.19	1.19	0.25	1.44	0.69	0.75	2.07	0	0	1.94	0.25	0	0.94	0.19	0	0.25	1.38
	Seaforth	2	0.62	0	0.06	0.06	0	0.94	0.44	0.75	0	0.06	0.44	0	0	0.06	0.38	0	0.12	0	0	0.31	0	3.75	0	0	0.12	0	3.32	0	0	0.99
		5	0	0.19	0.25	0.12	0.75	0.69	0	0.12	0	0.25	5.31	0.06	0	0.19	0.56	0.06	0.31	0	0.31	0	0.12	0.88	0	0	0	0	1.00	0	0	1.37

Region	Reef	Depth	*Acropora*	*Alveopora*	*Caulastrea*	*Cyphastrea*	*Diploastrea*	*Dipsastraea*	*Echinopora*	*Favites*	*Galaxea*	*Goniastrea*	*Goniopora*	*Hydnophora*	*Isopora*	*Leptoseris*	*Lobophyllia*	*Merulina*	*Montipora*	*Mycedium*	*Oxypora*	*Pachyseris*	*Paragoniastrea*	*Pavona*	*Pectinia*	*Platygyra*	*Pocillopora*	*Podabacia*	*Porites*	*Seriatopora*	*Turbinaria*	Other
Fitzroy	Barren	2	50.08	0.06	0	0	0	0	0	0	0	0	0	0	0.94	0	0	0	6.76	0	0	0	0	0	0	0.06	0.56	0	0.25	0	0	1.19
		5	65.33	0	0	0	0	0	0	0	0	0	0	0.12	0	0	0	0	4.32	0	0	0	0	0	0	0	0.38	0	0	0	0	0.06
	North Keppel	2	47.62	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.56	0	0	0	0	0	0	0	0.06	0	0	0	0	0.19
		5	24.50	0	0	0	0	0	0	0	0	0	0	0	0	0	0.31	0	2.94	0.25	0	0	0.12	0	0	0.25	0.44	0	0.06	0	0	0.31
	Middle	2	15.31	0	0	0.06	0	0	0	0	0	0	0	0	0	0	0	0	4.38	0	0	0	0	0	0	0	0	0	0.06	0	0.06	0.06
		5	10.25	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	9.38	0	0	0	0	0	0	0	1.12	0	0.12	0	0.12	0.12
	Keppels South	2	45.44	0.06	0	0.06	0	0	0	0	0	0	0.06	0	0	0	0	0	5.69	0	0	0	0	0	0	0	1.56	0	0	0	0	0
		5	32.18	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3.75	0	0	0	0	0	0	0	0.19	0	0.25	0	0.13	0.12
	Pelican	2	0.94	0	0	0.56	0	0	0	0.06	0	0	0.19	0.12	0	0	0	0	1.75	0	0	0	0.19	0	0	0	1.00	0	0	0	0.12	2.06
		5	0.31	5.75	0	0.69	0	0.19	0	4.94	0	0.06	1.06	0.38	0	0	0	0	0.19	0	0	0	2.12	0	0	0.81	0.06	0	0.31	0	1.12	2.49

Table A10 Percent cover of soft coral families 2023. 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'.

(sub-)region	Reef	Depth	Briareidae	Cladiellidae	Clavulariidae	Erythropodiidae	Helioporidae	Lemnaliadae	Nephtheidae	Sarcophytidae	Xeniidae	Other
Baron–Daintree	Low Isles	5	11.45	0.07	0.07	0	0.03	0	0	2.47	0.03	0.20
	Snapper North	2	5.21	0.04	8.64	0	0	0	0	0.83	0	0
		5	1.19	0.06	0.06	0	0	0	0	0	10.63	0
	Snapper South	2	0.38	0.08	0.46	0	4.21	0	0	1.63	0	0.04
		5	3.25	0	0	0	5.06	0	0	0.06	0	0.44
Johnstone Russell–Mulgrave	Fitzroy East	2	0.44	0.25	1.19	0	0	1.25	0	1.50	0	0
		5	6.81	0.50	0.62	0	0	0.38	0	10.38	0	0
	Fitzroy West	2	0.38	0	0	0	0	0.25	0	35.02	0.06	0
		5	0	0.06	0.12	0	0	1.39	0	28.28	0	0
	Fitzroy West LTMP	5	0.56	0.07	0	0	0	0.26	0	11.61	0	0.07
	Franklands East	2	0	0.25	0.50	0	0.56	0	0	1.38	0	0
		5	0.38	0	1.56	0	0	0	0	2.19	0.12	0
	Franklands West	2	0	0.06	8.70	0	0	0	0	10.14	0	0
		5	0	0	1.56	0	0	0	0	0.94	0	0
	High East	2	6.38	2.19	0.06	0	0	0	0	6.07	0	0
		5	9.27	0	0.06	0	0	0	0	0.38	0	0
	High West	2	0.12	0	0	0	4.63	0	0	3.82	0	0
		5	0.88	0	0	0	2.25	0	0	0.81	0	0
Herbert–Tully	Barnards	2	1.62	1.62	0.06	0	0	0.50	0	0.06	0.75	0
		5	2.88	0.63	0	0	0	0.12	0	0.56	2.70	0
	Dunk North	2	0.06	0.19	0.25	0	0	0	0	0.19	0.12	0
		5	0.25	1.44	0	0	0	0	0	0.38	5.31	1.12
	Dunk South	2	0.44	0.12	0.12	0	0	0	0	0.25	0	0
		5	2.76	0.19	0	0	0	0	0	0.44	0	0
	Bedarra	2	0.06	0.06	0	0	0	0	0	0	0	0
		5	4.12	0.19	0.06	1.00	0	0	0	0.19	0.12	0.06

Region	Reef	Depth	Briareidae	Cladiellidae	Clavulariidae	Erythropodiidae	Helioporidae	Lemnaliadae	Nephtheidae	Sarcophytidae	Xeniidae	Other
Burdekin	Palms East	2	0	0	0	0	0	0	0	2.63	0	0
		5	0	0.25	0.12	0	0	0	0	1.94	0	0
	Palms West	2	0.81	0	0.81	0	0	0	9.51	18.28	0	0
		5	3.75	0.19	1.44	0	0	0.06	3.14	10.01	0	0.50
	Havannah North	5	1.29	0.03	0.83	0	0	0	0	0.17	0.30	0.17
	Havannah	2	3.25	0.06	0	0	0	0	0	0.38	0	0
		5	7.89	0	0	0	0	0	0	0.06	0	0
	Pandora	2	0	0.44	0.06	0	0	0	0	0.06	0	0
		5	0	0.62	0.19	0	0	0	0	0.81	0	0
	Pandora North	5	5.12	0.17	4.16	0	0.03	0	0.03	3.04	0.07	0.03
	Lady Elliot	2	0	0	0	0	0	0	0	0	0	0.06
		5	0.38	0	0	0	0	0	0	0.06	0	0
	Magnetic	2	0.06	0	0	0	0	0	0	0.44	0	0
		5	0.06	0	0	0	0	0	0	1.57	0	0.12

(sub-)region	Reef	Depth	Briareidae	Cladiellidae	Clavulariidae	Erythropodiidae	Helioporidae	Lemnaliadae	Nephtheidae	Sarcophytidae	Xeniidae	Other
Mackay-Whitsunday	Hayman	5	0.23	0.63	0	0	0	0.10	0.23	5.38	0.07	0
	Border	5	0.27	1.59	0	0	0	0.03	0.30	18.67	0.33	0.17
	Hook	2	0.12	2.25	0	0	0	0	0	5.69	0	0
		5	3.07	0.56	0	0	0	0	0	4.83	0	0
	Double Cone	2	0.06	0.19	0	0	0	0	0	0.25	0	0
		5	0.44	0	0	0	0	0	0	0.81	0	0
	Daydream	2	0	0	0	0	0	0	0	0	0	0
		5	0	0.25	0	0.12	0	0	0	0.19	0	0
	Dent	2	4.88	0.63	0	0	0	0	0.31	2.06	0	0
		5	0.81	0.56	0	0	0	0.06	0	2.13	0	0.06
	Shute Harbour	2	0.44	2.90	0	0	0	0	1.01	5.72	0.50	0
		5	0.06	1.06	0	0.06	0	0.25	0.25	3.75	0.38	0
	Pine	2	0	0.38	0	0	0	0	0	0.69	0.31	0
		5	0.06	0.06	0	0	0	0	0	0.44	0.06	0
	Seaforth	2	1.63	0.56	0	0.19	0	0	0	3.19	0	0
		5	0	0	0	1.62	0	0	0	0.50	0.12	0.06

(sub-)region	Reef	Depth	Briareidae	Cladiellidae	Clavulariidae	Erythropodiidae	Helioporidae	Lemnaliadae	Nephtheidae	Sarcophytidae	Xeniidae	Other
Fitzroy	Barren	2	0.12	4.82	0	0	0	0	0	0.19	0.88	0
		5	0	0.50	0	0	0	0	0	0	0.88	0
	North Keppel	2	0	0.06	0	0	0	0	0	0	0	0
		5	0	0.56	0	0	0	0	0	0.31	0	0
	Middle	2	0	0.06	0	0	0	0	0	0	0.19	0.06
		5	0	0	0	0	0	0	0	0	0	0
	Keppels South	2	0	0.94	0	0	0	0	0	0	2.19	0
		5	0	0	0	0	0	0	0	0.06	0.50	0
	Pelican	2	0	0.44	0	0	0	0	0	0.31	0	0.06
		5	0	0.06	0	0.12	0	0	0	6.81	0.06	1.25

Table A11 Percent cover of macroalgae groups 2023. 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
			Amansia	*Asparagopsis*	Crustose coralline	*Hypnea*	*Laurencia*	*Peyssonnelia*	Undefined	*Halimeda*	Undefined	*Dictyopteris*	*Dictyota*	*Lobophora*	*Padina*	*Rosenvingea*	Sargassaceae	*Spatoglossum*	*Stypopodium*	Undefined		
Baron–Daintree	Low Isles	5	0	0	3.03	0	0	0.07	0.27	0.17	0.20	0	0	0	0	0	0	0	0	0.20	0	39.22
	Snapper North	2	0.08	1.42	5.84	0.46	0	0.71	40.22	3.75	0.58	0	1.25	0	0	0	0	0	0	0.13	0	13.30
		5	0	0	3.70	0	0	0.31	0.31	0.06	0	0	1.00	0.12	0	0	0	0	0	0.12	0	31.23
	Snapper South	2	0.21	0	3.63	0.58	0	0.25	1.54	0	0.33	0	0.04	0	0	0	0	0	0	0.04	0	34.05
		5	0.31	0	6.69	0.12	0	0.44	3.75	0	0.06	0	0	0.12	0	0	0	0	0	0	0	20
Johnstone Russell–Mulgrave	Fitzroy East	2	0	0	1.00	0	0	0	0.75	0	0	0	0	0	0	0	0	0	0	0	0	46.06
		5	0	0	2.19	0	0	0.12	0.81	0	0	0	0	0	0	0	0	0	0	0	0	33.25
	Fitzroy West	2	0.50	0	1.19	1.57	0	0.19	0.63	0	0	0	0	0	0	0	0	0	0	0	0	12.90
		5	0	0	2.56	0.38	0	0.12	0.63	0	0	0	0	0	0	0	0	0	0	0	0	16.73
	Fitzroy West LTMP	5	0	0	2.28	0	0	0.03	0.37	0.03	0	0	0	0	0	0	0	0	0	0	0	31.37
	Franklands East	2	0	0	0.75	1.00	0	0.06	0.56	0	0.12	0	0	0	0	0	0	0	0	0	0	23.31
		5	0	0	3.26	0	0	0.31	6.66	0	0	0	0.06	0	0	0	0	0	0	0	0	49.50
	Franklands West	2	0	0	1.75	0.75	0.44	0	10.90	0.31	0	0	0.25	0	0	0	0	0	0	0	0	15.21
		5	0	0	3.38	0.12	1.94	0.12	17.31	0.75	0.56	0	0	0	0	0	0	0	0	0.06	0	15.88
	High East	2	1.00	0	3.01	5.72	0	0.06	3.38	0	0	0	0	0	0	0	0	0	0	0	0	32.24
		5	2.06	0	7.88	0.38	0	0.56	3.13	0	0.19	0	0	0	0	0	0	0	0	0	0	32.34
	High West	2	0	0	4.95	0.50	0	0.19	3.95	0	0.06	0	0.25	0	0	0	0	0	0	0	0	30.62
		5	0	0	3.44	0.12	0	0	0.62	0	0	0	0	0	0	0	0	0	0	0	0	44.00

(sub-)region	Reef	Depth	Rhodophyta (red algae)							Chlorophyta (green algae)		Phaeophyta (brown algae)									Undefined	Turf Algae
			Amansia	*Asparagopsis*	Crustose coralline	*Hypnea*	*Laurencia*	*Peyssonnelia*	Undefined	*Halimeda*	Undefined	*Dictyopteris*	*Dictyota*	*Lobophora*	*Padina*	*Rosenvingea*	Sargassaceae	*Spatoglossum*	*Stypopodium*	Undefined		
Herbert–Tully	Barnards	2	0	0	0.88	0.63	0	0	0.44	0	0	0	0	0	0	0	0	0	0	0	0	21.13
		5	0	0	1.25	0	0	0	0.31	0	0	0	0	0	0	0	0	0	0	0	0	16.55
	Dunk North	2	0.06	0	0.56	0	0	0.06	2.00	0	0	0	0.19	0.06	0.06	0	8.81	0	0.06	0.44	0	23.62
		5	0	0	0.81	0	0	0.25	0.50	0	0.44	0	0	0.12	0	0	0.50	0	0	0	0	21.44
	Dunk South	2	0	0	0.88	0	0	0.13	1.62	0	0	0	0.31	2.00	0.12	0	5.31	0	1.25	0.06	0	40.90
		5	0	0	2.44	0	0	0.31	0.69	0	0.06	0	0	6.20	0	0	0	0	0	0	0	37.10
	Bedarra	2	0	0	1.38	1.56	0	0.25	5.01	0	0.31	0	5.12	0.94	0.31	0	8.59	0	0	1.38	0	32.79
		5	0	0	0.44	0	0	0.12	0.25	0	0.06	0	0.81	0.56	0	0	0.38	0	0	0.06	0	33.19
Burdekin	Palms East	2	0	0	0.50	0	0	0	0.19	0	0.31	0	0	0	0	0	0	0	0	0.06	0	41.16
		5	0	0	0.75	0	0	0.12	0.12	0	0.12	0	0	0.06	0	0	0	0	0	0.06	0	34.69
	Palms West	2	0	0	0.12	0	0	0.06	0.06	0	0	0	0	0.12	0	0	0	0	0	0	0	31.97
		5	0	0	0.94	0	0	0	0	0	0.06	0	0	0	0	0	0	0	0	0	0	44.09
	Havannah North	5	0	0.43	2.98	0	0	0.79	0.46	0	0.10	0	0.03	8.97	0	0	3.51	0	0	0.56	0.03	46.29
	Havannah	2	0	0	1.75	0.06	0	0.94	2.26	0	0	0	0.50	5.26	0	0	1.07	0	2.44	0.94	0	43.28
		5	0	0	1.19	0	0	0.06	1.19	0	0.13	0	0.25	9.76	0	0	3.94	0	0.06	0.12	0	34.24
	Pandora	2	0	0	2.38	0	0	0.69	1.06	0	0	0	0.12	4.98	0	0	10.41	0	0	0.38	0	41.75
		5	0	0	3.50	0	0	0.38	1.50	0	0	0	1.50	2.94	0	0	0.06	0	0	0.19	0	52.88
	Pandora North	5	0	0.33	2.84	0	0	0.17	0.20	0	0	0	0	2.81	0	0	0.13	0	0	0.03	0	24.52
	Lady Elliot	2	0	0	1.62	5.82	0	0.75	5.50	0	0	0	1.06	0.88	0	0	0.94	0	0	0.19	0	20.57
		5	0	0	0.56	0.31	0	0.88	1.00	0	0	0	0.06	0.06	0	0	0	0	0	0.06	0	19.50
	Magnetic	2	0	0	1.62	0.50	0	0.31	2.12	0	0.06	0	5.00	7.00	0.06	0	11.62	0	0	1.44	0	30.06
		5	0	0	1.07	0.13	0	2.01	6.20	0	0.06	0	6.33	3.89	0	0	11.86	0	0	1.13	0	19.86

(sub-)region	Reef	Depth	Rhodophyta (red algae)							Chlorophyta (green algae)		Phaeophyta (brown algae)									Undefined	Turf Algae
			Amansia	*Asparagopsis*	Crustose coralline	*Hypnea*	*Laurencia*	*Peyssonnelia*	Undefined	*Halimeda*	Undefined	*Dictyopteris*	*Dictyota*	*Lobophora*	*Padina*	*Rosenvingea*	Sargassaceae	*Spatoglossum*	*Stypopodium*	Undefined		
Mackay–Whitsunday	Hayman	5	0	0	1.06	0	0	0.03	0.77	0	0	0	0	0.13	0	0	0	0	0	1.03	0	70.80
	Border	5	0	0	0.23	0	0	0	0.13	0	0	0	0	0.03	0	0	0	0	0	0.03	0	50.89
	Hook	2	0	0	0.88	0	0	0.12	1.25	0	2.94	0	0	0.19	0	0	0	0	0	0.19	0	75.19
		5	0	0	0.44	0	0	0	1.06	0	1.69	0	0	0.12	0	0	0	0	0	0	0	52.62
	Double Cone	2	0	0	0.19	0	0	0	4.12	0	0.06	0	10.81	3.12	0.44	2.31	16.44	0	0.12	3.69	0	34.50
		5	0	0	0.50	0	0	0.19	6.25	0	0.06	0	11.38	0.62	0.12	0.19	9.25	0	0.12	3.75	0	39.38
	Daydream	2	0	0	0.12	0	0	0.06	11.94	0.06	0.06	0.06	1.81	2.44	2.44	0.06	12.00	0	0	4.50	0	37.31
		5	0	0	0.12	0	0	0	0.62	0	0	0	0.19	0.31	0	0	0	0	0	0.31	0	61.38
	Dent	2	0	0	2.00	0	0	4.75	0.88	0	0.25	0	0	2.88	0	0	0.38	0	0	0.56	0	54.78
		5	0	0	2.38	0	0	2.19	0.50	0	0.19	0	0	4.81	0	0	0	0	0	0	0	51.41
	Shute Harbour	2	0	0	0.19	0	0	0.25	0.31	0	0	0	0.25	1.20	0.13	0	1.13	0	0	0.19	0	28.03
		5	0	0	0.19	0	0	0.06	0.19	0	0	0	0.06	0.44	0	0	0	0	0.06	0	0	37.56
	Pine	2	0	0	2.26	0	0	1.82	16.68	0.31	0.19	0	1.25	12.29	0.31	0	12.72	0	0	1.19	0	38.00
		5	0	0	2.01	0	0	1.57	1.94	0.69	0.19	0	0	12.21	0	0	0.56	0	0	0.25	0	56.42
	Seaforth	2	0	0	1.63	0.56	0	0.19	9.44	0.12	0.19	0.06	0.50	3.88	0.81	0.06	6.75	0	0	3.75	0	37.84
		5	0	0	0.50	0	0	0.06	4.94	0	0.12	0	1.50	2.94	0.44	0	1.00	0	0	1.31	0	45.50

(sub-)region	Reef	Depth	Rhodophyta (red algae)							Chlorophyta (green algae)		Phaeophyta (brown algae)									Undefined	Turf Algae
			Amansia	*Asparagopsis*	Crustose coralline	*Hypnea*	*Laurencia*	*Peyssonnelia*	Undefined	*Halimeda*	Undefined	*Dictyopteris*	*Dictyota*	*Lobophora*	*Padina*	*Rosenvingea*	Sargassaceae	*Spatoglossum*	*Stypopodium*	Undefined		
Fitzroy	Barren	2	0	0	1.31	0	0	0	1.44	0	0.06	0	0	0.56	0	0	0	0	0	0	0	24.39
		5	0	0	5.69	0	0	0.38	1.44	0	0	0	0	8.58	0	0	0	0	0	0	0	12.01
	North Keppel	2	0	0	4.00	0	0	0.06	0.88	0	0	0	0	22.38	0	0	0	0	0	0	0	23.38
		5	0	0	3.81	0	0	0.88	1.12	0	0	0	0	23.31	0	0	0	0	0	0	0	32.06
	Middle	2	0	0	3.94	0.06	0	0.62	3.69	0	0	0	0.06	20.75	0	0	27.00	0.06	0.25	0.06	0	22.56
		5	0	0	2.88	0	0	0.25	3.62	0	0	0	0.62	15.00	0	0	26.50	0	0	0.12	0	23.62
	Keppels South	2	0	0	1.06	0.56	0	0.19	2.56	0	0	0	1.38	8.56	0	0	13.12	0	0	0.12	0	12.00
		5	0	0.19	2.75	0.06	0	1.81	1.63	0	0	0	1.88	13.52	0	0	0.12	0	0	0	0	24.67
	Pelican	2	0	0.38	2.50	0	0	0.38	16.56	0	1.38	1.12	2.25	12.50	0.19	0	21.38	2.00	0.12	0.12	0	22.31
		5	0	0	0.62	0	0	0.25	9.25	0	0.56	0	1.25	9.25	0.06	0	2.12	0	0	0.06	0	22.69

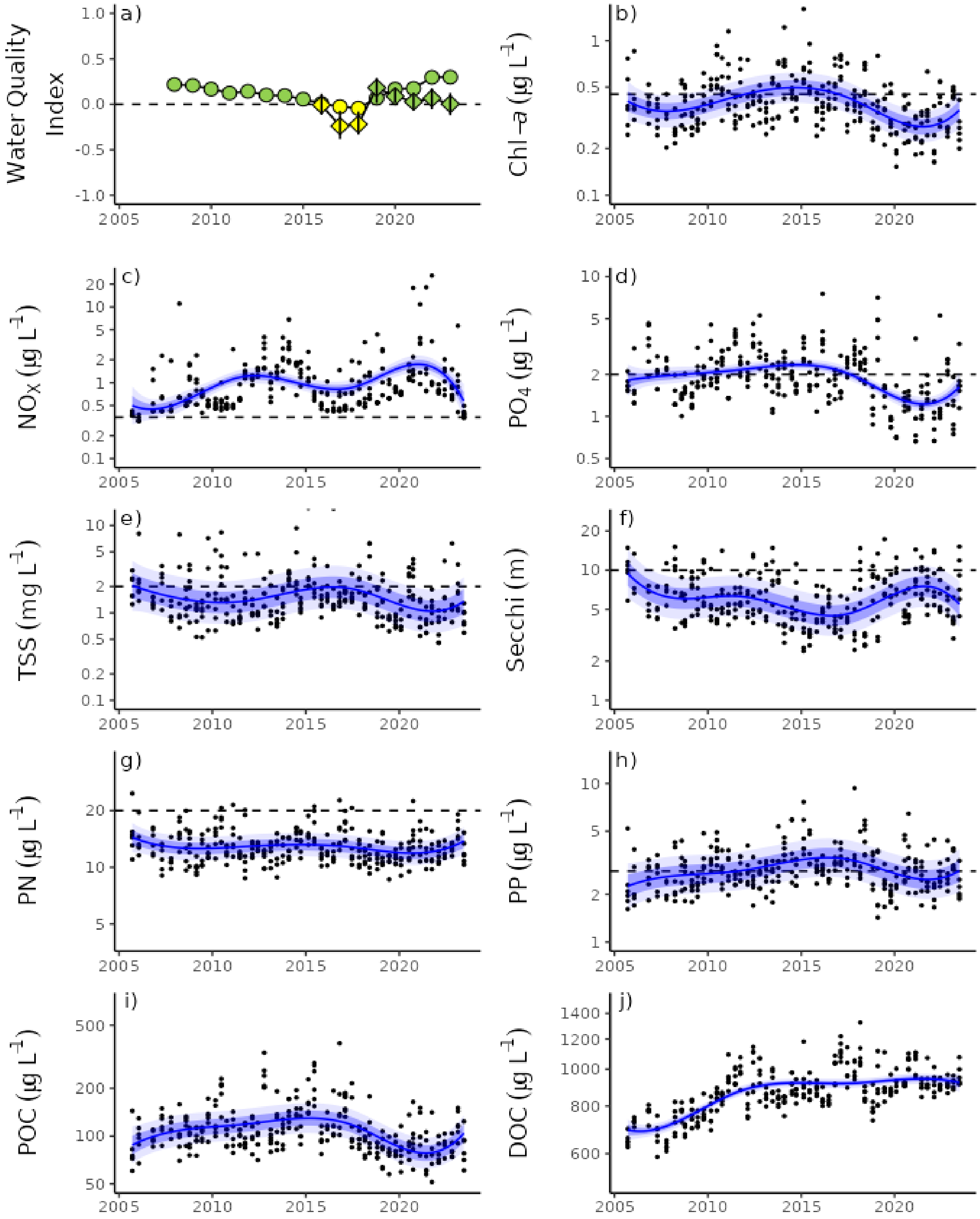

Figure A10 Temporal trends in water quality: Barron–Daintree sub-region. a) water quality index, b) chlorophyll a, c) nitrate/nitrite, d) Phosphate, e) total suspended solids, f) secchi depth, g) particulate nitrogen, h) particulate phosphorus, i) particulate organic carbon and j) dissolved organic carbon. 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, while the annual condition 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); threshold levels have yet to be established. Trends in manually sampled water quality variables are represented by blue lines with blue shaded areas defining 95% confidence intervals of those trends, black dots represent observed data. Dashed reference lines indicate guideline values (GBRMPA 2010). Extract from Gruber *et al*. (2024).

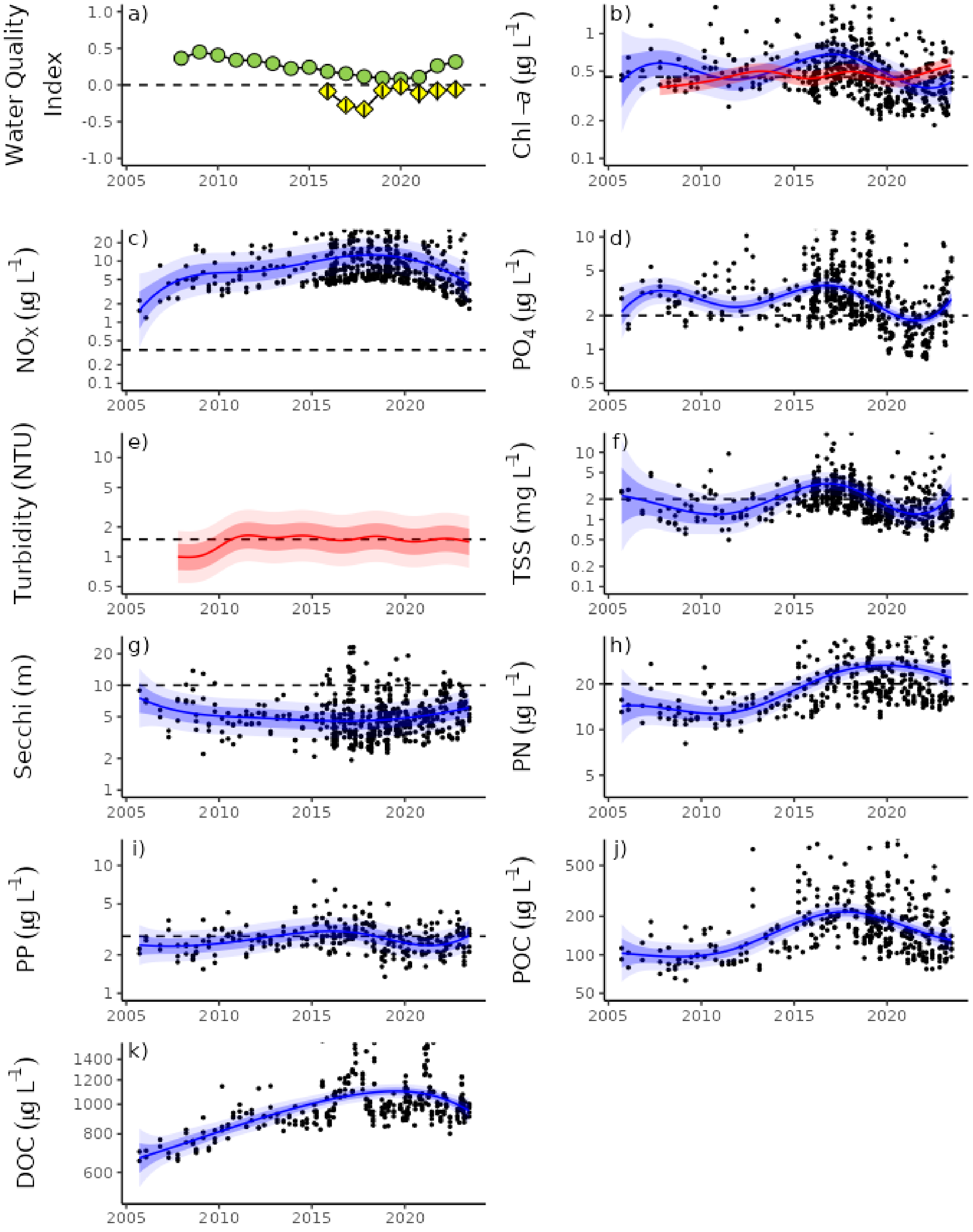

Figure A11 Temporal trends in water quality: Johnstone Russell-Mulgrave sub-region. a) water quality index, b) chlorophyll a, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus j), particulate organic carbon and 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, while the annual condition 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); threshold levels have yet to be established. Trends in manually sampled water quality variables are represented by blue lines with blue shaded areas defining 95% confidence intervals of those trends, black dots represent observed data. Trends of records from ECO FLNTUSB instruments (b, e) are represented in red, individual records are not displayed. Dashed reference lines indicate guideline values (GBRMPA 2010). Extract from Gruber *et al.* (2024).

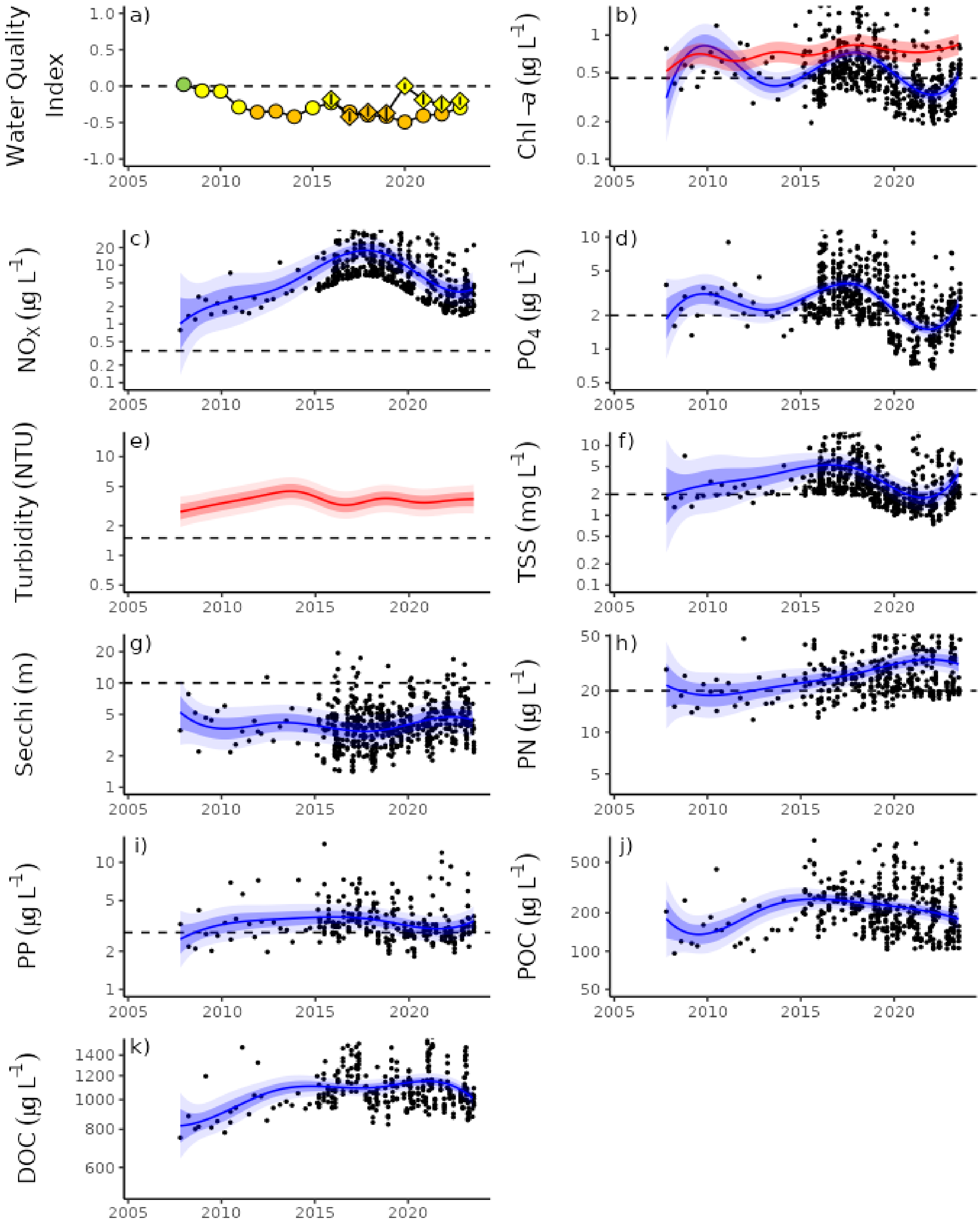

Figure A12 Temporal trends in water quality: Herbert–Tully sub-region. a) water quality index, b) chlorophyll a, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved 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, while the annual condition 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); threshold levels have yet to be established. Trends in manually sampled water quality variables are represented by blue lines with blue shaded areas defining 95% confidence intervals of those trends, black dots represent observed data. Trends of records from ECO FLNTUSB instruments (b, e) are represented in red, individual records are not displayed. Dashed reference lines indicate guideline values (GBRMPA 2010). Extract from Gruber *et al*. (2024).

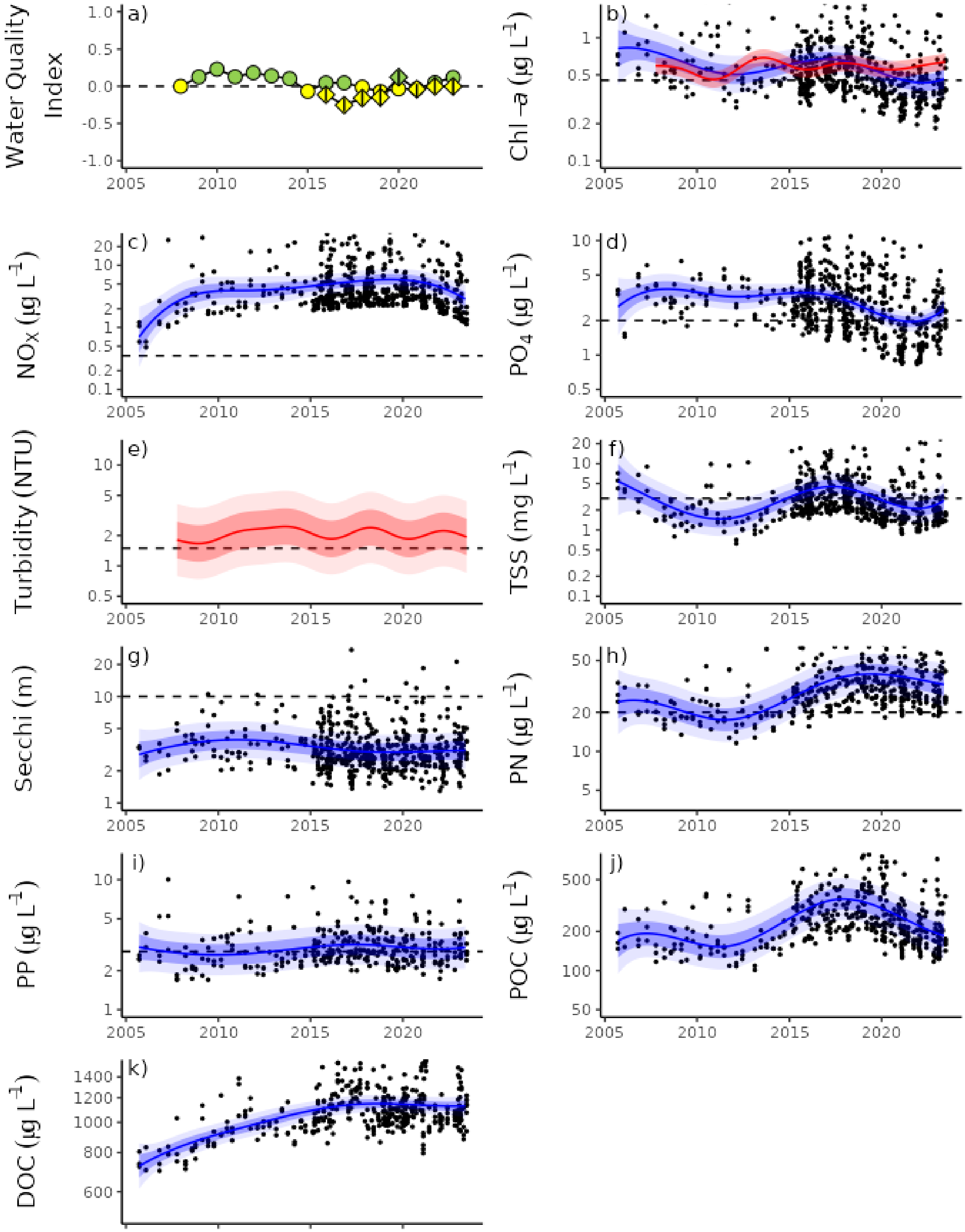

Figure A13 Temporal trends in water quality: Burdekin region. a) water quality index, b) chlorophyll a, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved organic 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, while the annual condition 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); threshold levels have yet to be established. Trends in manually sampled water quality variables are represented by blue lines with blue shaded areas defining 95% confidence intervals of those trends, black dots represent observed data. Trends of records from ECO FLNTUSB instruments (b, e) are represented in red, individual records are not displayed. Dashed reference lines indicate guideline values (GBRMPA 2010). Extract from Gruber *et al.* (2024).

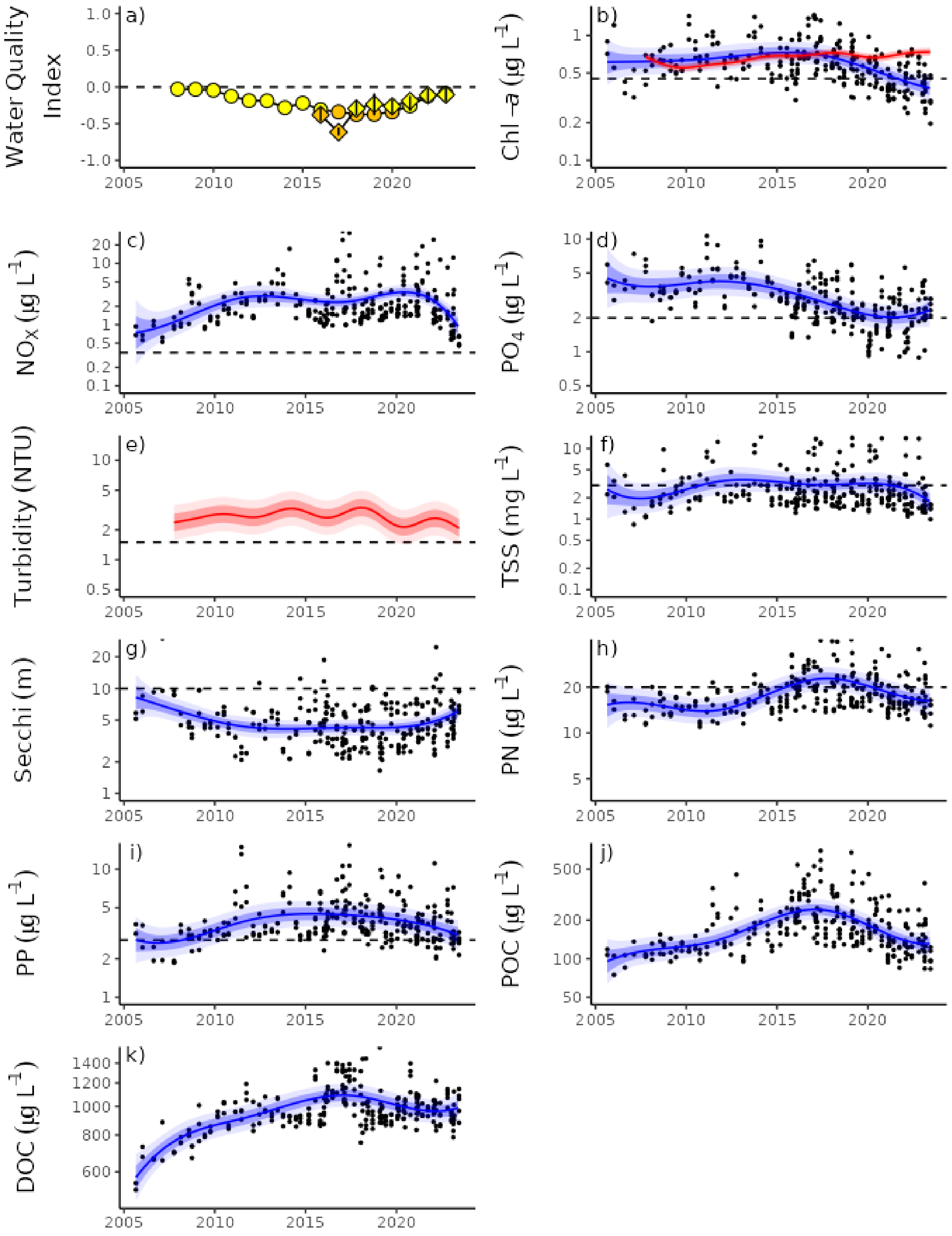

Figure A14 Temporal trends in water quality: Mackay–Whitsunday region. a) water quality index, b) chlorophyll a, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved 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, while the annual condition 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); threshold levels have yet to be established. Trends in manually sampled water quality variables are represented by blue lines with blue shaded areas defining 95% confidence intervals of those trends, black dots represent observed data. Trends of records from ECO FLNTUSB instruments (b, e) are represented in red, individual records are not displayed. Dashed reference lines indicate guideline values (GBRMPA 2010). Extract from Gruber *et al.* (2024).

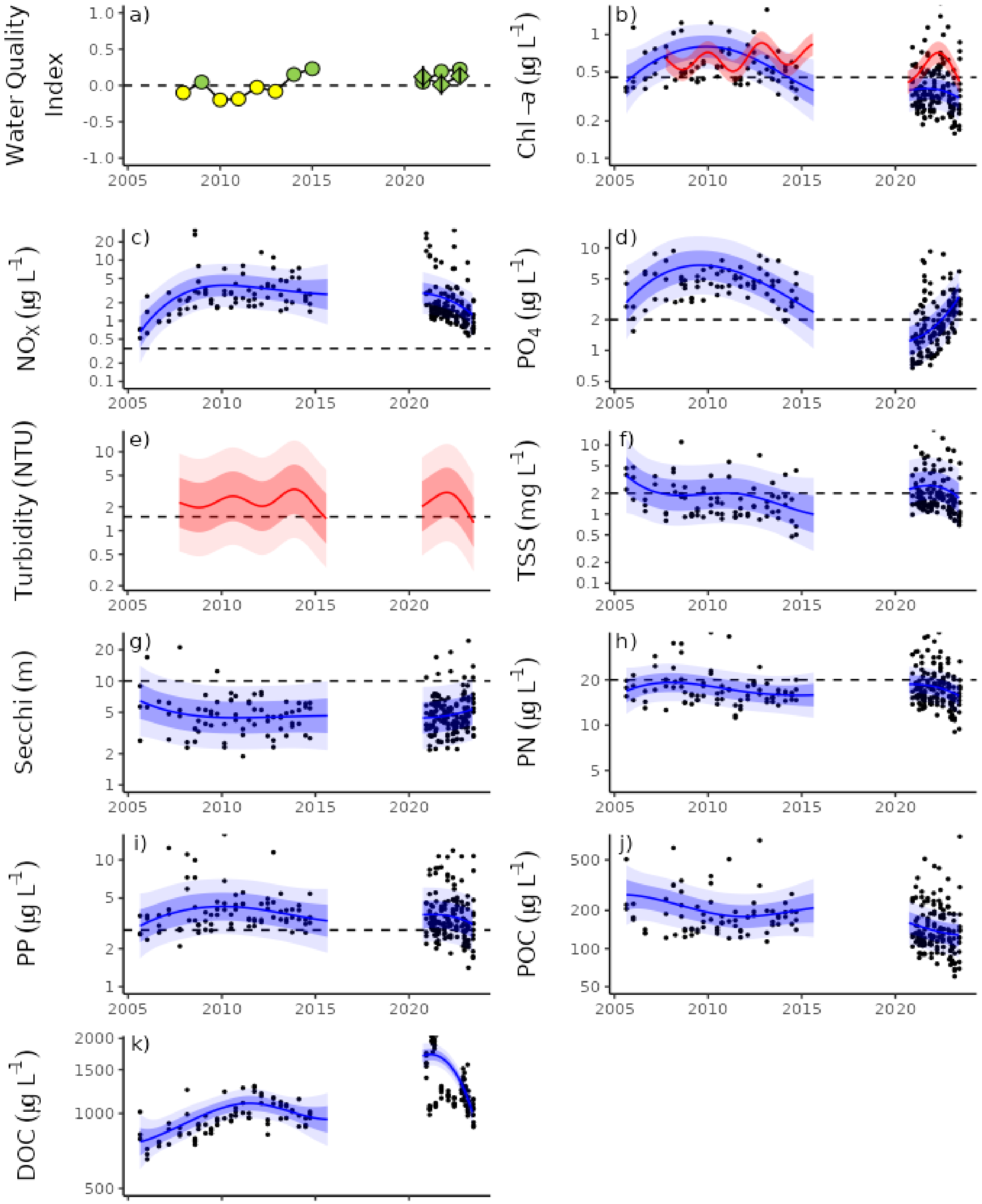

Figure A15 Temporal trends in water quality: Fitzroy region. a) water quality index, b) chlorophyll a, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved organic 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, while the annual condition 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); threshold levels have yet to be established. Trends in manually sampled water quality variables are represented by blue lines with blue shaded areas defining 95% confidence intervals of those trends, black dots represent observed data. Trends of records from ECO FLNTUSB instruments (b, e) are represented in red, individual records are not displayed. Dashed reference lines indicate guideline values (GBRMPA 2010). Water quality monitoring ceased in 2015 and resumed in 2021. Extract from Gruber *et al.* (2024).

Appendix 2: Publications and presentations 2022–2023

Publications

Fabricius, K. E., Crossman, K., Jonker, M., Mongin, M., & Thompson, A. 2023, Macroalgal cover on coral reefs: Spatial and environmental predictors, and decadal trends in the Great Barrier Reef. *PLoS One*, *18*(1), e0279699.

Gonzalez-Rivero, M., Thompson A., Johns K., Ortiz J., Kim S., Fabricius K., Emslie M., Hoey A., Hoogenboom M., Barrios-Novak K., McClure E., Pandolfi J, Mumby P. J., Murray L., Schaffelke B., & Staples T. 2023a, Introduction: Indicator Framework for the evaluation of the condition of coral reef habitats in the Great Barrier Reef. *Report prepared for the Great Barrier Reef Foundation. Australian Institute of Marine Science, Townsville. (23 pp)* available here

Gonzalez-Rivero, M., Thompson A., Johns K., Ortiz J., Kim S., Fabricius K., Emslie M., Hoey A., Hoogenboom M., Barrios-Novak K., McClure E., Pandolfi J, Mumby P. J., Murray L., Schaffelke B., & Staples T. 2023b, Indicator Framework for the evaluation of the condition of coral reef habitats in the Great Barrier Reef: Methodological Documentation. *Report prepared for the Great Barrier Reef Foundation. Australian Institute of Marine Science, Townsville. (138 pp)* available here

Mackay-Whitsunday-Isaac Healthy Rivers to Reef Partnership 2023, *Mackay-Whitsunday-Isaac 2022 Report Card Results Technical Report*. Mackay-Whitsunday-Isaac Healthy Rivers to Reef Partnership, Proserpine, QLD. available here

Page, C. A., Giuliano, C., Bay, L. K., & Randall, C. J. 2023, High survival following bleaching underscores the resilience of a frequently disturbed region of the Great Barrier Reef. *Ecosphere*, *14*(2), e4280.

Shand, A., Taylor, D., 2023, *Technical Report for the Townsville Dry Tropics annual report cards. Updated 2023*. Dry Tropics Partnership for Healthy Waters, Townsville. available here

Wet Tropics Waterways 2023. *Wet Tropics Report Card 2023 (reporting on data 2021-22). Waterway Environments: Results*. Wet Tropics Waterways and Terrain NRM, Innisfail. available here

Presentations

Marine Monitoring Program - Coral 2022. Presentation at Marine Monitoring Program Science Seminar. Great Barrier Reef Marine Park Authority, 21st Sep 2022

Marine Monitoring Program – Coral 2022. Annual presentation to stakeholders. Townsville Yacht Club 17th November 2022