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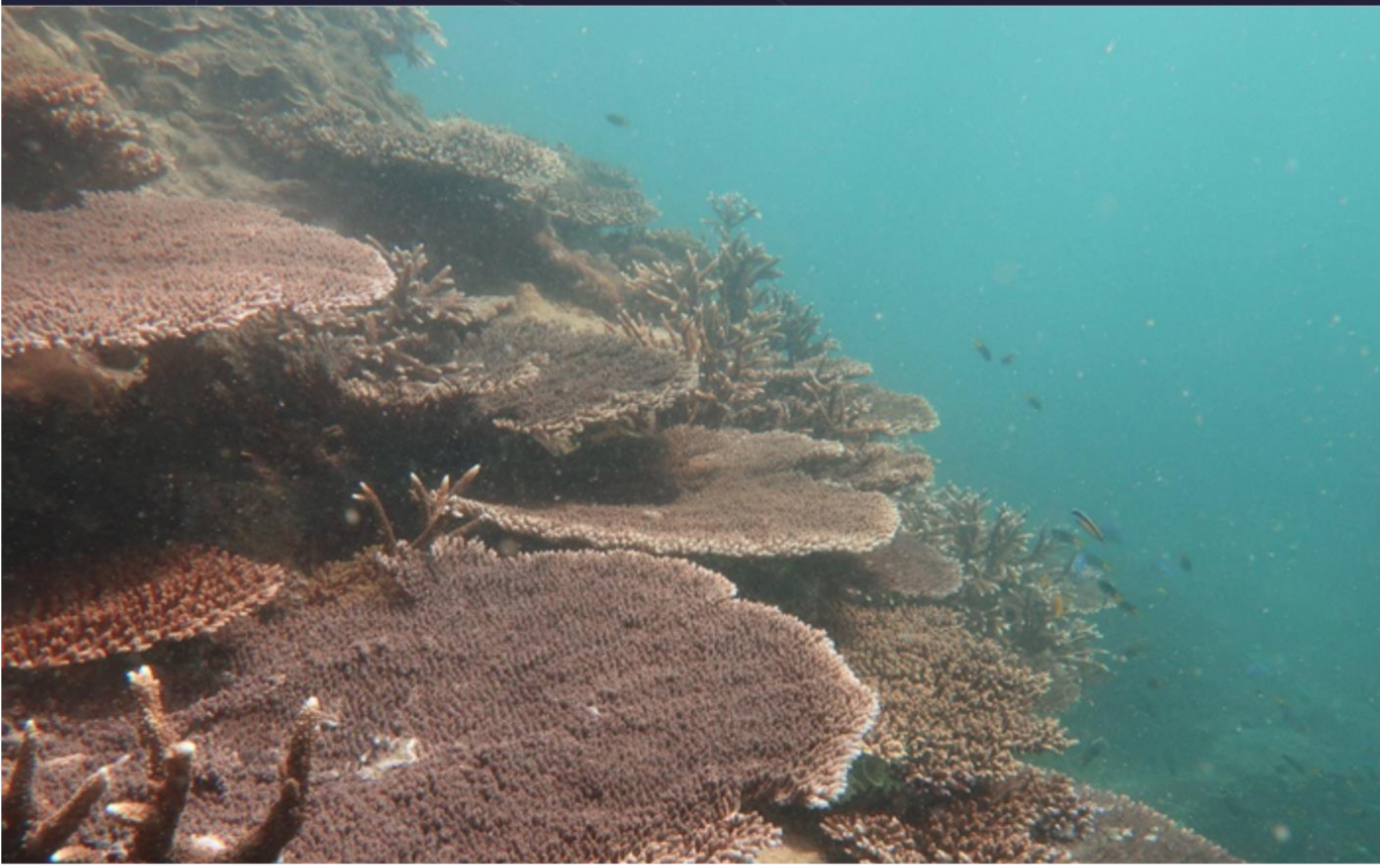
Great Barrier Reef
Marine Park Authority

Great Barrier Reef Marine Monitoring Program

Inshore coral reefs monitoring
Annual Report
2021–22



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Front cover photo: Tabulate and branching *Acropora* dominate shallow reef areas of Barren Island, Fitzroy. © Johnston Davidson, Australian Institute of Marine Science

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
NAP	Non-algal particulate
NOAA	National Oceanic and Atmospheric Administration
Reef 2050 WQIP	Reef 2050 Water Quality Improvement Plan
The Reef	Great Barrier Reef
PAR	Photosynthetically available radiation

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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 respect 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 remained in an overall 'poor' condition in 2022, the Coral Index having increased marginally from a low point in 2021 (Figure 1). This slight improvement has been driven by modest increases in Coral cover in all regions that is also reflected in improvements in Cover change and Composition indicator scores. 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).

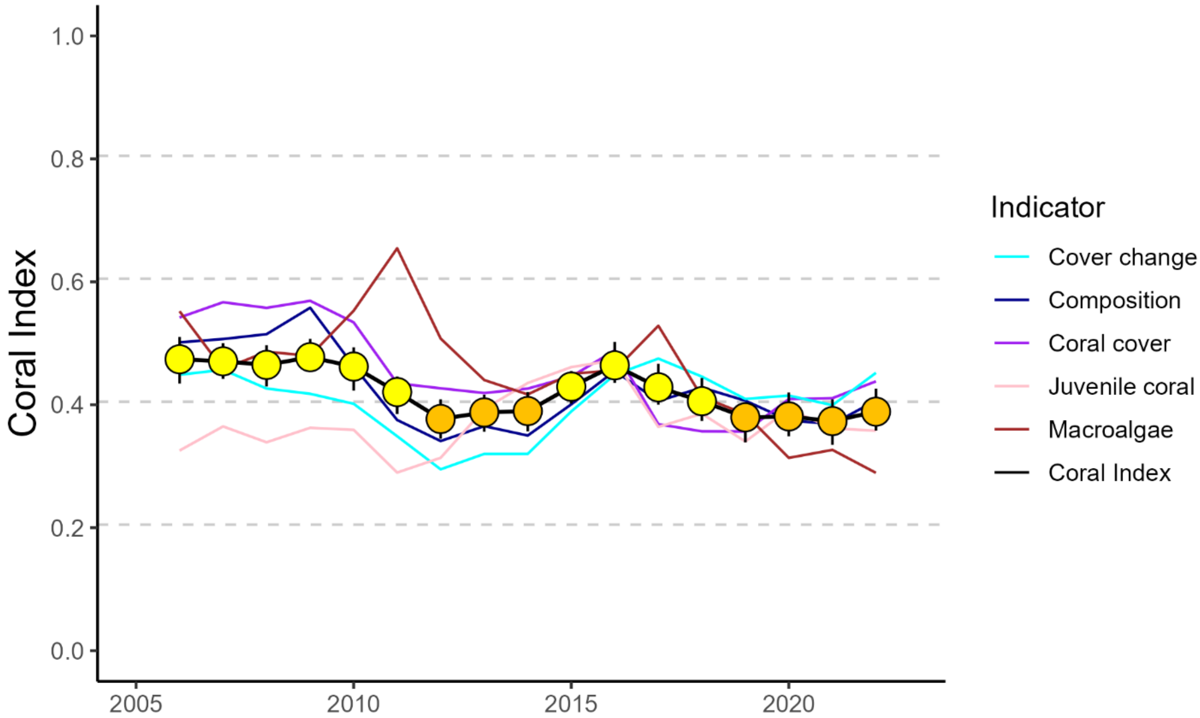

Figure 1 Trends in the Coral Index and contributing indicator scores for the inshore Reef. Coral Index scores are coloured according to Reef Water Quality Report Card categories: orange = 'poor', yellow='moderate'.

Sea water temperatures in early 2022 reached levels sufficient to cause coral bleaching. Temperature anomalies were highest in the Burdekin region, tapering toward the Mackay Whitsunday Isaac and Southern areas of the Wet Tropics regions. Despite these high temperatures and observed coral bleaching, coral cover increased at most reefs. No other severe climate-related pressures affected the inshore Great Barrier Reef (the Reef) over the 2021/2022 summer. Corallivorous crown-of-thorns starfish were again present on reefs in the Johnstone Russell–Mulgrave sub-region. However, their numbers have declined since a peak in 2020. Only at High Island were crown-of-thorns starfish observed at 'outbreak' densities. The impact of these starfish on corals was reduced by culling undertaken by the Crown-of-thorns Starfish Control Program.

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 innate 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. Although Coral cover has been improving since 2019, this contrasts the continued decline in Macroalgae scores that reached new lows in 2022. The dichotomy in scores for Coral cover and Macroalgae highlight that reefs further from the coast have been recovering faster than those further inshore, where Macroalgae scores are generally low.

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
- proportion of **macroalgae** in 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. Regional level coral community condition and trends are summarised below.

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.

- In the Barron Daintree sub-region, the Coral Index score remained 'moderate'. The Coral Index score has recovered to the point observed in 2018, prior to damage caused by flooding of the Daintree River and cyclone Owen in 2019. Ongoing high cover of macroalgae, correspondingly low densities of juvenile corals, and a lack of recovery of sensitive coral species at shallow sites at Snapper Island 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. This stability reflects a balance between ongoing impacts of crown-of-thorns starfish and recovery of coral cover, when and where crown-of-thorns starfish populations were low. Large numbers of crown-of-thorns starfish were removed from Fitzroy Island and the Frankland Group by the Crown-of-thorns Starfish Control Program in the year prior to 2021 surveys and this along with ongoing removals will have reduced their impact on corals. In 2022 all locations monitored were in moderate to good condition and only at High Island was the density of crown-of-thorns starfish above outbreak density.
- In the Herbert Tully sub-region, the Coral Index score declined slightly in 2022, slipping into the 'moderate' range for the first time since 2018. 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 cover persists at Dunk Island and Bedarra Island.

Burdekin region coral community condition

Coral Index scores remain 'moderate' having continued to decline from a high point in 2020 reached following a period of recovery since the impact of cyclone Yasi in 2011. Thermal stress in early 2020 and to a lesser extent 2022 was sufficient to cause severe coral bleaching at most reefs. Despite these bleaching events Coral cover in 2021 and 2022 continued to increase overall with the regional average in 2022 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 density of juvenile corals in 2022 was, 'poor' to 'very poor' at the shallow depths of all reefs.

Mackay–Whitsunday region coral community condition

The Coral Index score has improved marginally since 2020 although remained 'poor'. The recent improvement captures signs of recovery from the severe impact of cyclone Debbie in 2017. Most indicators have improved since 2020, most notably Juvenile coral that is the only indicator to have recovered to 'moderate" level. There were high densities of juvenile corals at some reefs, but not at locations where macroalgae cover remains very high and is likely to be continuing to limit coral recovery.

Fitzroy region coral community condition

The Coral Index score remained 'poor' having improved slightly since 2021. Scores for both Coral cover and Cover change in 2022 were in the moderate range as coral cover on most reefs continued to increase. In contrast, Composition remains poor reflecting an increased proportion of *Montipora* compared to historical observations on several reefs. Continued very high cover of macroalgae and declining or low densities of juvenile corals are reflected in 'very poor' scores for their respective indicators.

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

Role of water quality on inshore reef resilience

While the results presented here do not provide clear guidance in terms of load reductions required to improve Coral Index scores in the inshore Reef, they do support the premise of the *Reef 2050 Water Quality Improvement Plan* that the loads entering the Reef during high rainfall periods are reducing the resilience of these communities. The potential for phase shifts to algae-dominated states, or delayed recovery because of poor water quality, in combination with an expected increase in disturbance frequency, reinforces the importance of managing local pressures to support the long-term resilience of these communities.

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 Hughes *et al.* 2018, Osborne *et al.* 2017). 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 to highlight where and when condition is poor.

The Coral Index is based on the general understanding that healthy and resilient coral communities exist in a dynamic equilibrium, with communities periodically in a state of recovery, punctuated by acute disturbance events. Common 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, Kline *et al*. 2006, Kuntz *et al*. 2005, 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 corals 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, of the order Scleractinia, 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, Cheal *et al.* 2010, Foster *et al.* 2008, 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 (Birrell *et al.* 2008b, Morse *et al.* 1996). 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).

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). 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.* 2017a), 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. that that

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 2022 Reef Water Quality Report Card. This report includes results from coral reefs monitored by AIMS as part of the MMP until July 2022 with inclusion of data from inshore reefs monitored by the AIMS Long-Term Monitoring Program (LTMP) from 2005 to 2022. 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 source and use of these data are summarised in Table 1.

2.1.1 River discharge

Daily records of river discharge (ML) were obtained from Queensland Government Department of Natural Resources and Mines (DNRM) river gauge stations for the major rivers draining to the Reef. For the Reef and each (sub-)region, total annual discharge estimates for each Water-year, 1st October to 30th September, include a correction factor applied to gauged discharges to account for ungauged areas of the catchment (Moran *et al*. 2022, Table A5).

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 2021 was calculated.

2.1.2 River nutrient and sediment loads

Loads of particulate nitrogen (PN), dissolved inorganic nitrogen (DIN) and total suspended sediment (TSS) delivered by rivers were sourced from MMP water quality (Moran *et al.* 2022). 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 TSS 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 TSS 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 TSS 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. A range of logger models have been used, initially recording temperature at 30-minute intervals (until 2008) and then later revised to 10 minute intervals (post-2008) (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 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 Bureau of Meteorology 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 a one-degree heating day, a two-degree exceedance for one day equates to two DHD. DHD anomalies are summed over the period December 1 to March 31 each summer.

Finally, degree heating Weeks (DHW) were downloaded from 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*. 2018).

[1] ReefTemp Next Generation was developed through the Centre for Australian Weather and Climate Research (CAWCR) – a partnership between CSIRO and the Bureau of Meteorology (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 – 2022	water gauging stations closest to river mouth, adjusted for ungauged area of catchment	regional discharge plots and table, covariate in analysis of temporal change in Coral Index	DNRME, adjustment as tabulated by Moran *et al.* (*2022.*)
Riverine DIN, TSS and PN loads	2006 – 2022		covariate in analysis of temporal change in Coral Index	MMP Water Quality (Moran *et al*. *2022*)
Sea temperature	2005 – 2022	*in situ* sensor at coral sites	regional plots, thermal bleaching disturbance categorisation, *in situ* degree heating day estimates	MMP Inshore Coral monitoring
Degree heating days	2006 – 2022	remote sensing, ~4 km^2 pixels adjacent to coral sites	informing attribution of thermal stress, regional plots, thermal bleaching disturbance categorisation, thermal stress maps	Bureau of Meteorology
Degree heating weeks	2006 – 2022	remote sensing	informing attribution of thermal stress, thermal stress maps	National Oceanographic and Atmospheric Administration
Cyclone tracks	2005– 2022		informing attribution of storms as cause of observed coral loss, cyclone track maps	Bureau of Meteorology
Environment at coral monitoring sites				
Chlorophyll *a* and Total suspended solids	2003 – 2022	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 (NAP)	2002 – 2018	remote sensing adjacent to coral sites, resolution ~1 km^2	Macroalgae and Composition metric thresholds, mapping	Bureau of Meteorology
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 Australian Bureau of Meteorology (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

Mean wet-season concentrations of total suspended solids (TSS) and Chlorophyll *a* (Chl *a*) at each reef were estimated based on the methods developed by the water quality component of the MMP (Moran *et al*. 2022, Petus *et al*. 2016). 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). For each pixel the proportion of the wet-season (December through April) that waters were classified into the four water-types was determined. For each reef a set of nine pixels in waters adjacent to the coral monitoring sites were selected and the mean proportional of the wet season that the nine pixel patch was classified as each water-type derived. The reef level concentration of TSS and Chl a was then calculated as the sum of the proportion for each water-type multiplied by the mean measured concentrations within that water-type (Table 2). There were insufficient water samples to provide valid estimates of concentrations in water-type 4 and the concentrations used are almost certainly too high as they were derived from the annual mean concentrations for the three MMP sample locations that were most frequently categorised as being exposed to this water-type. The maximum exposure to water-type 4 was 13% of the wet season in 2019 at Barren Island. It is planned that all available MMP niskin sample data will be used to refine concentration estimates in each water-type in 2022-2023.

Table 2 Water types estimated from Sentinel imagery. Table supplied by Caroline Petus, MMP Water Quality, measured concentrations for water-type 4 adjusted based on observed concentrations at the reef sites most commonly exposed to water-type 4.

Reef water - types	FU Colour classes	Description	Measured concentrations
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 bodies typically contain high sediment and dissolved organic matter concentrations resulting in reduced light levels. It is also enriched in CDOM and phytoplankton concentrations and has elevated nutrient levels.	TSS: 18.3 ± 45.7 mg L^{-1} Chl-a: 1.6 ± 2.4 µg L^{-1}
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 body 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).	TSS 5.9 ± 8.0 mg L^{-1} Chl-a: 0.8 ± 0.8 µg L^{-1}
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.	TSS: 3.9 ± 5.1 mg L^{-1} Chl-a: 0.5 ± 0.5 µg L^{-1}
WT4	No number	Bluish marine waters with high light penetration	TSS: 0.7 mg L^{-1} Chl-a: 0.3 µg L^{-1}

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 logistic 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 datum (LAT). 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
Barron Daintree	Cape Tribulation North	MMP	●	●																
	Cape Tribulation Mid	MMP	●	●																
	Cape Tribulation South	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	●	●	●	●	●	●	●	●	●	●		●	+	●		●	●	●
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 2022.

2.3 Coral community sampling methods

Two 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	Percentage cover of the substratum of major benthic habitat components.	Approximately 34 cm belt along upslope side of transect sampled at 50 cm intervals from which 32 frames are sampled.	Approximately 34 cm 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 belt (dive slate length) along the upslope side of transect. Size classes: 0–2 cm, 2–5 cm	34 cm 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 1m 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 either 0–2 cm, >2–5 cm, or >5–10 cm. In 2019 recording of the 5-10cm 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 either; 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), and unknown (when a cause could not be confidently assumed). Scaring caused by fish bites were not recorded as deemed to be neither indicative of poor coral health or 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 by 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 for 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 5 reporting bands of the Reef Water Quality Report Card. Thus, the scoring for the Coral cover indicator is scaled linearly from zero when cover is 0% through to 1 when cover is at or above the threshold level of 75% (Figure 3).

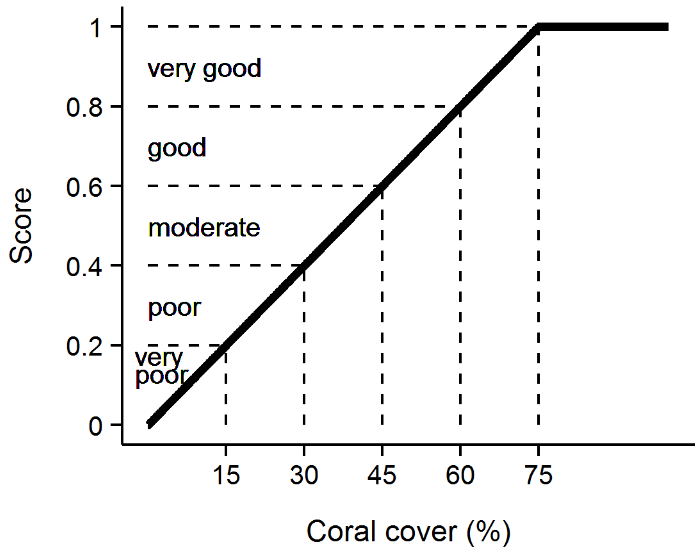

Figure 3 Scoring diagram for the Coral cover indicator metric. Numeric scores and associated condition classifications based on observed coral cover are presented.

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:

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

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 a 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).

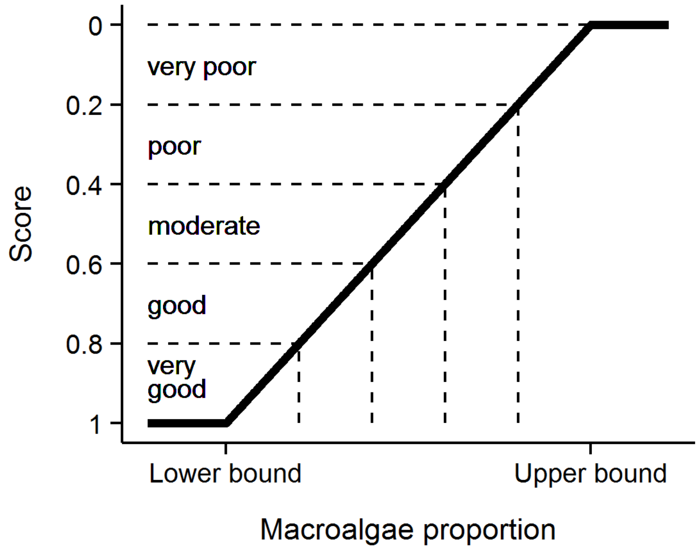

Figure 4 Scoring diagram for the Macroalgae indictor metric. Upper and lower threshold values are reef and depth specific. Numeric scores and associated condition classifications are presented. Note that for this metric the y-axis is inverted as high values reflect poor condition.

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 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 beyond 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 density was scaled linearly from 0 at a density of 0 to 0.4 at a density of 4.6 colonies per m^2, then linearly to a score of 1 when the density was 13 colonies per m^2 or above (Figure 5)

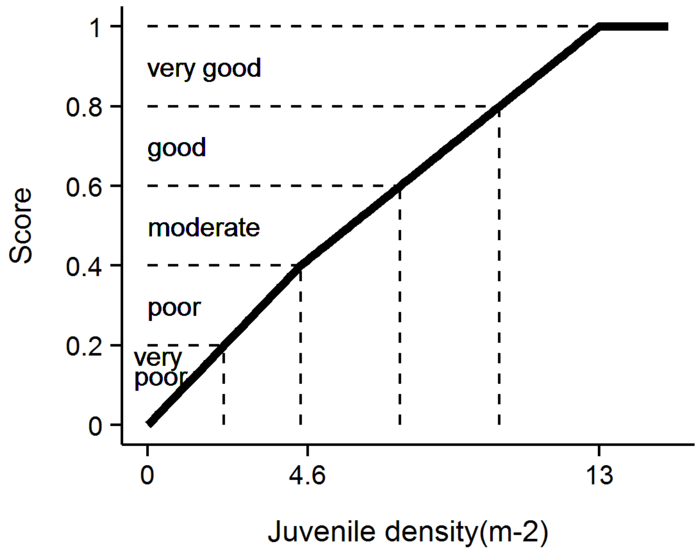

Figure 5 Scoring diagram for the Juvenile coral indicator metric. Numeric scores and associated condition classifications are presented.

2.4.4 Cover change indicator metric

A second avenue for recovery of coral communities is the growth of corals during periods free from acute disturbance (Gilmour *et al*. 2013). Chronic pressures associated with water quality may suppress the rate at which coral cover increases and indicate a lack of resilience. The Cover change indicator score is derived from the comparison of the observed change in hard coral cover between two visits and the change in hard coral cover predicted by Gompertz growth equations (Thompson & Dolman 2010) parameterised from time-series of coral cover available on inshore reefs from 1992 until 2007. Gompertz equations were parameterised separately for the fast-growing corals of the family Acroporidae and the slower growing combined grouping of all other hard corals at each of 2 m and 5 m depths. Initial exploratory analysis provided no justification for a more detailed parameterisation of the coral community, in part due to the increasing imprecise estimates of cover due to declining cover for each group with further sub-setting of the coral community.

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

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

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

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

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

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

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

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

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

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

Where, $Acr_{it}, OthC_{it}$ and Sc_{it} are the cover of Acroporidae coral, other hard coral, and soft coral respectively at a given reef at time (t). $eskK$ is the community size at equilibrium (100) and $rAcr$ is the rate of increase (growth rate) in percent cover of Acroporidae coral. Varying effects of region and reef (β_j and Υ_k respectively) were also incorporated to account for spatial autocorrelation. Model coefficients associated with the intercept, region and reef (α_i, β_j and Υ_k) 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 $vAcr_i$.

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

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

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

Gompertz models were fitted in a Bayesian framework to facilitate combining growth rates and associated uncertainties across models. A total of 20,000 Markov-chain Monte Carlo (MCMC) 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.

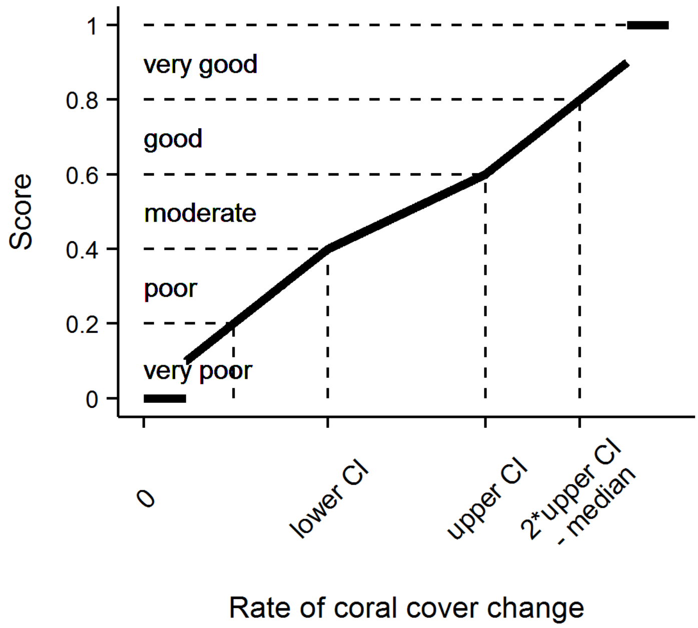

Figure 6 Scoring diagram for Cover change indicator metric.

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 runoff 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.* (2014b). 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, 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).

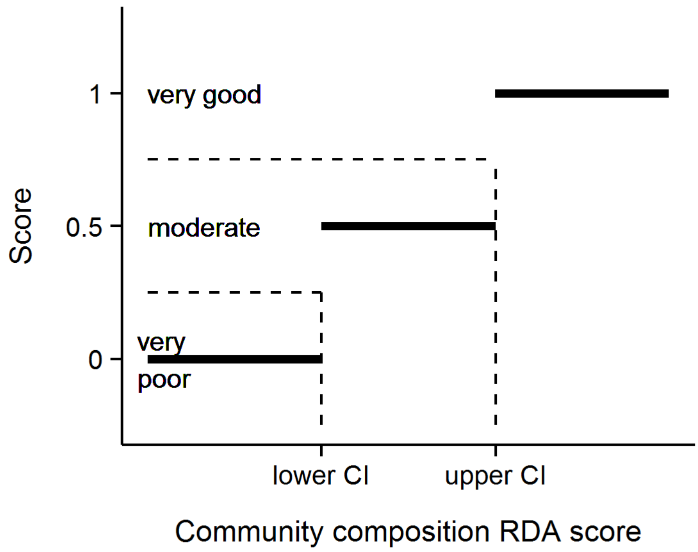

Figure 7 Scoring diagram for the Composition indicator metric

In 2022 AIMS adopted a series of revisions to that 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, but also 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) where 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: from basic observational error, the relevance of thresholds, and then variation in scores for different indicators or communities being assessed.

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

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

Mean Coral Index scores for each (sub-)region were estimated as the mean of observed mean scores for each indicator from all reefs and depths within the (sub-)region. Reef level scores as reported in the Reef Water Quality Report Card were estimated as the weighted mean of regional scores. Weightings applied reflect the relative proportion of inshore coral reef area within the 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 is 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
44	Temporal trend in coral community condition	Reef	Major disturbances	Relative influence of major pressures over the time-series
4.3, 4.4, 4.5, 4.64.3	Trends in Coral Index and individual indicators	(sub-)region		Generalised linear mixed models; pairwise comparisons
4.7.14.7.1	Coral Index and indicator scores in 2020	Reef and region	Chl *a*, PAR	Generalised linear mixed models, predicted responses
4.7.24.7.2	Temporal variability in Coral Index in relation to water quality	region	Regional riverine: discharge, Total N and Total P loads. Chl *a* exposure, PAR	Generalised additive models, predicted responses
Appendix 1: Additional Information	Trends in benthic community composition.	reef/Depth		Plots
	Summaries of 2020 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* concentration and PAR at 8m depth. General Reef-wide trends were identified on the basis that Akaike information criterion (AICc) values for models fitting indicator response to the water quality proxy and including random intercepts for each region were at least 2 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 2018).

2.5.2 Relationship in 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 particulate nitrogen, dissolved inorganic nitrogen and total suspended solids loads from the adjacent catchments (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 GAM models 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 2017).

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 2.4.62.4.

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

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

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

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

Table 7 Information considered for disturbance categorisation

Disturbance	Description
Thermal bleaching	Consideration of *in situ* degree heating day 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 in 2021-22

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 and (sub-)regional level in sections 4.3 to 4.6.

3.1 Cyclones

Tropical cyclones frequently cross the inshore Reef.

Over the 2021-22 reporting period no cyclones were likely to have produced damaging waves to the regions covered by this report. Cyclone Seth formed as a low in the Timor Sea on 23rd December 2021, crossed Cape York Peninsular and progressed into the Coral Sea just south of Cairns before intensifying to a category 1 system well offshore, to the east of Marion Reef on 31st December (Figure 8). Cyclone Tiffany traversed a path to the North of Cooktown as a category 2, no damaging waves occurred within the reporting regions (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). There are yet to be clear signs of recovery of coral cover in the wake of this cyclone (Figure 29a).

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 4.3).

3.2 Sea temperature

Sea temperatures over the 2022 summer were above long-term averages (Figure 9). Temperatures exceeded the published thresholds of 60 to100 degree heating days (Garde *et al.* 2014) or 4 degree heating weeks (NOAA 2018) that are likely to lead to significant coral bleaching. The Burdekin region recorded the highest thermal stress in 2022, however minimal bleaching or loss of coral was observed during our surveys in July (Figure 9, 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, 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, Figure 10).

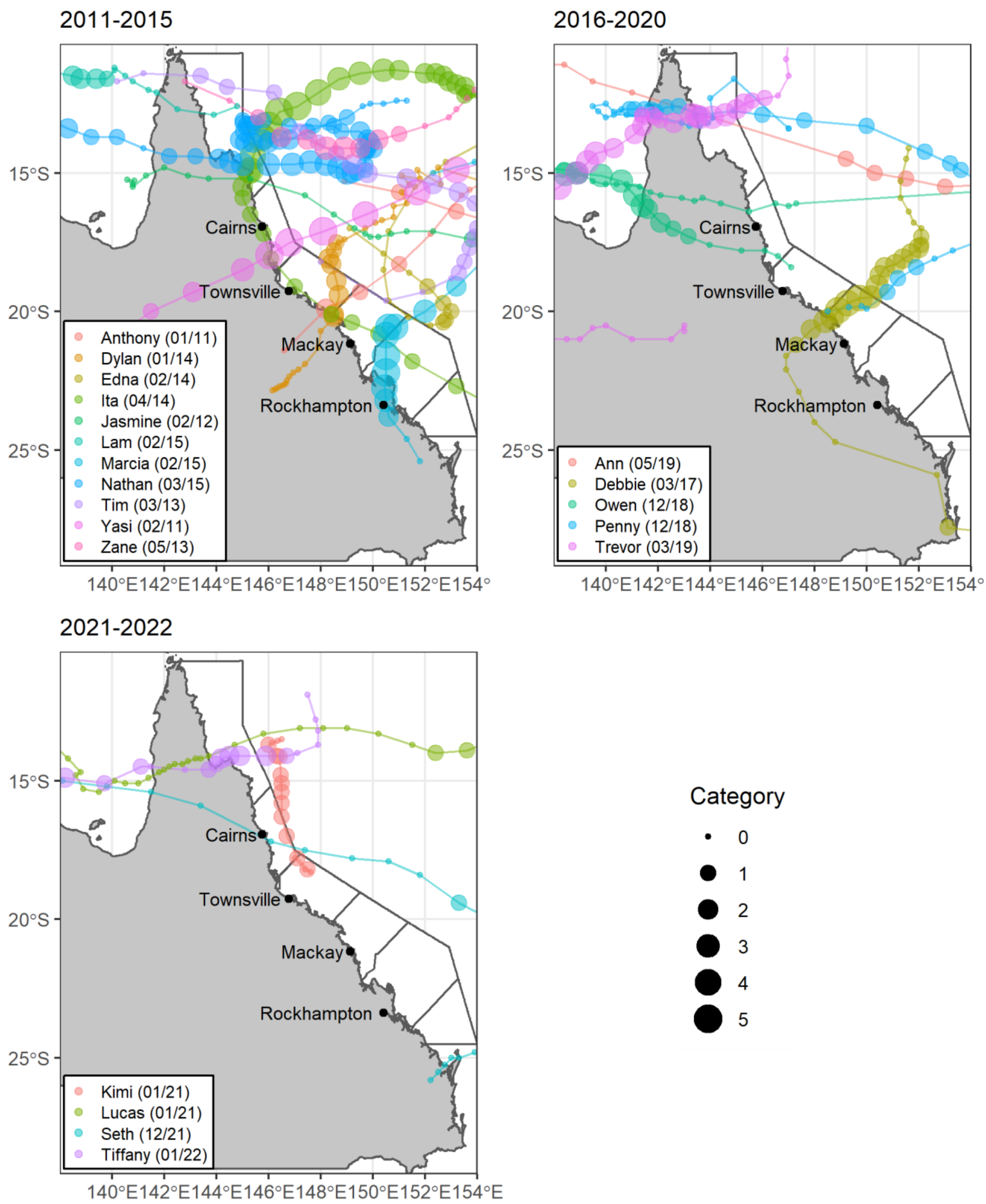

Figure 8 Cyclone tracks for systems crossing the inshore Reef over the last decade. Tracks sourced from the Bureau of Meteorology.

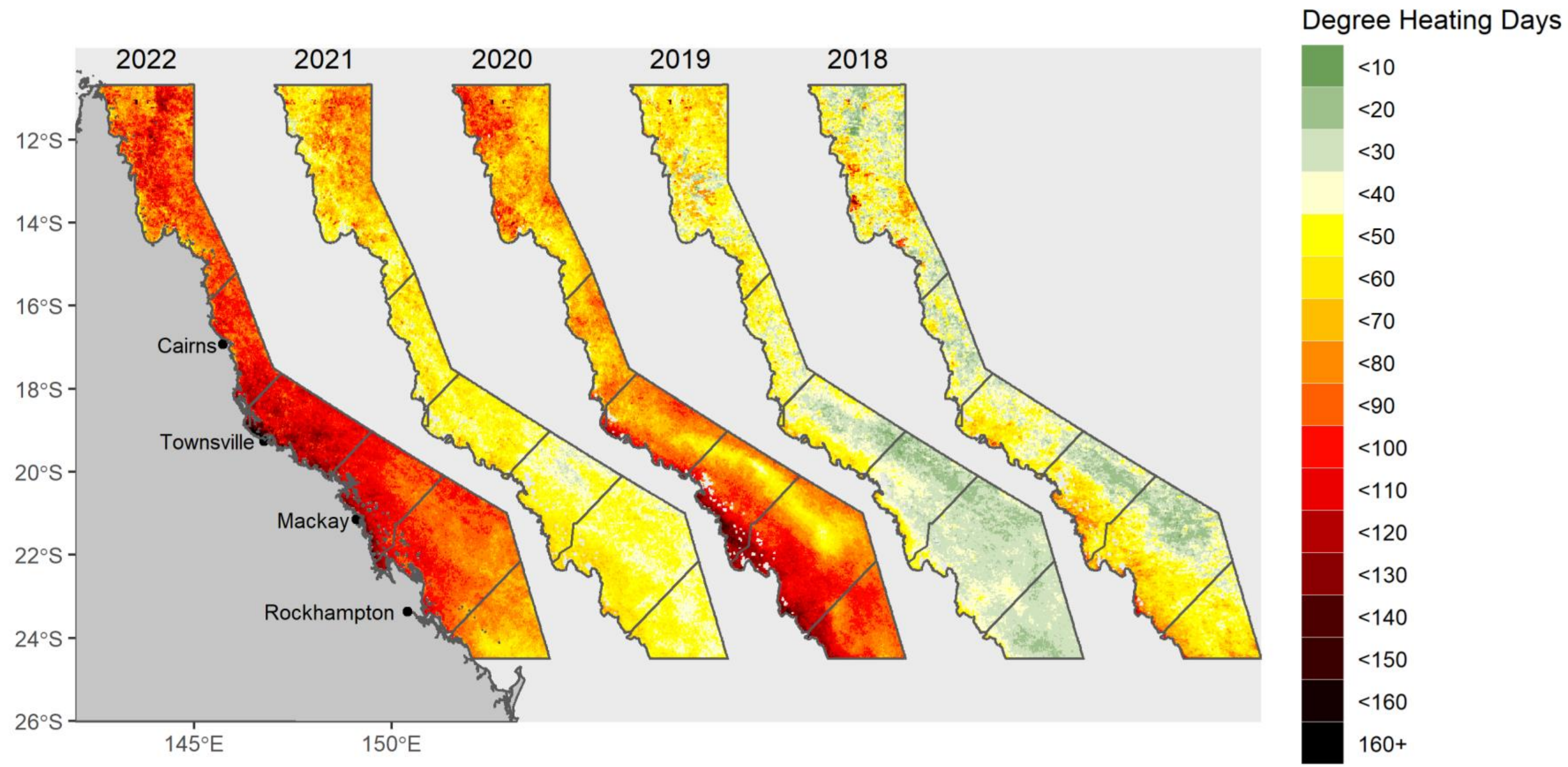

Figure 9 Annual degree heating day estimates for the Reef. Data are the annual degree heating day accumulations over the summer period (1 December to 31 March) for ~4 km^2 pixels based on temperatures exceeding 14 Day IMOS climatology. Data were sourced from [the Australian Bureau of Meteorology ReefTemp next generation web data service](#) .

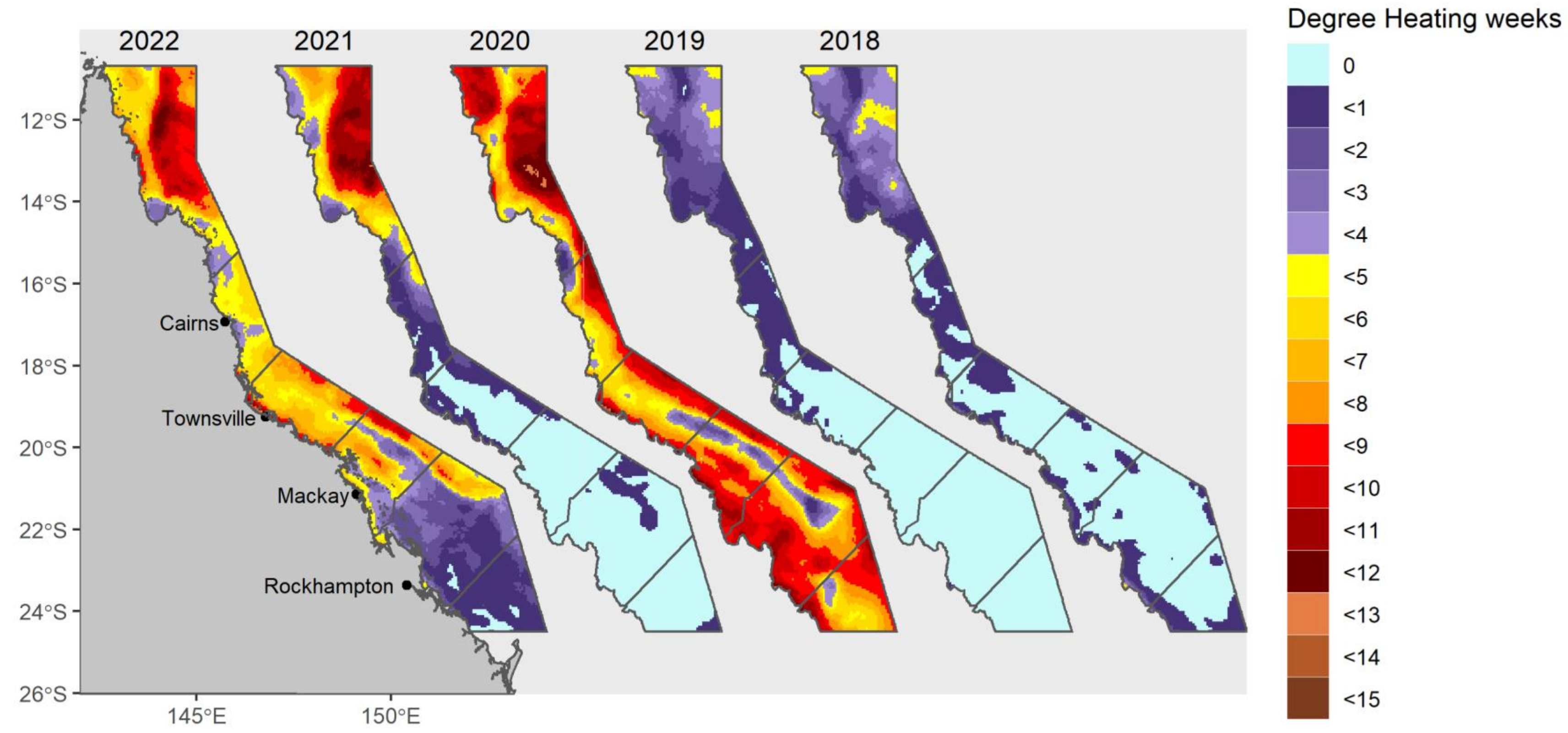

Figure 10 Annual degree heating week estimates for the Reef. Data are the annual maximum degree heating week estimates for each ~25 km^2 pixel. Date were sourced from NOAA coral reef watch.

3.3 Crown-of-thorns starfish

In 2022 the density of crown-of-thorns starfish were above outbreak levels at High East (62 ha^{-1}) and High West (87 ha^{-1}), a single juvenile was observed at Frankland Group West. Regionally in the Wet Tropics, numbers have continued to decline from the high levels observed in 2020 (Table 9, Figure A8). In 2022 no crown-of-thorns starfish were observed during MMP surveys in other regions. Most recent AIMS LTMP results recorded active outbreaks of crown-of-thorns starfish only in the Swains Sector in the south (Swain 2022).

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 9).

Table 8 Number of crown-of-thorns removed. Australian Government Crown-of-thorns Starfish Control Program data supplied by Great Barrier Reef Marine Park Authority, Eye on the Reef. Figures in bold are the number of individuals removed in 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
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			**233** (1.82)	**122** (0.52)	**498** (1.50)

Table 9 Size class distribution of crown-of-thorns starfish on inshore reefs in the Wet Tropics. Included are the percentages culled, as listed in Table 8, of cohorts 1-4, and percentage followed by number observed in parentheses observed by 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)**

3.4 River discharge

Discharge in 2022 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 metre 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 where Fitzroy River flood waters cause 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 2022 recovery from this event was occurring at Keppels South but limited, at best, at Pelican Island.

The influence of high sediment and nutrient loads are not as overtly obvious as the mortality of corals exposured to freshwater and are explored in terms of suppression of coral recovery and variable condition of coral communities along water quality gradients in section 4.74.7.1.

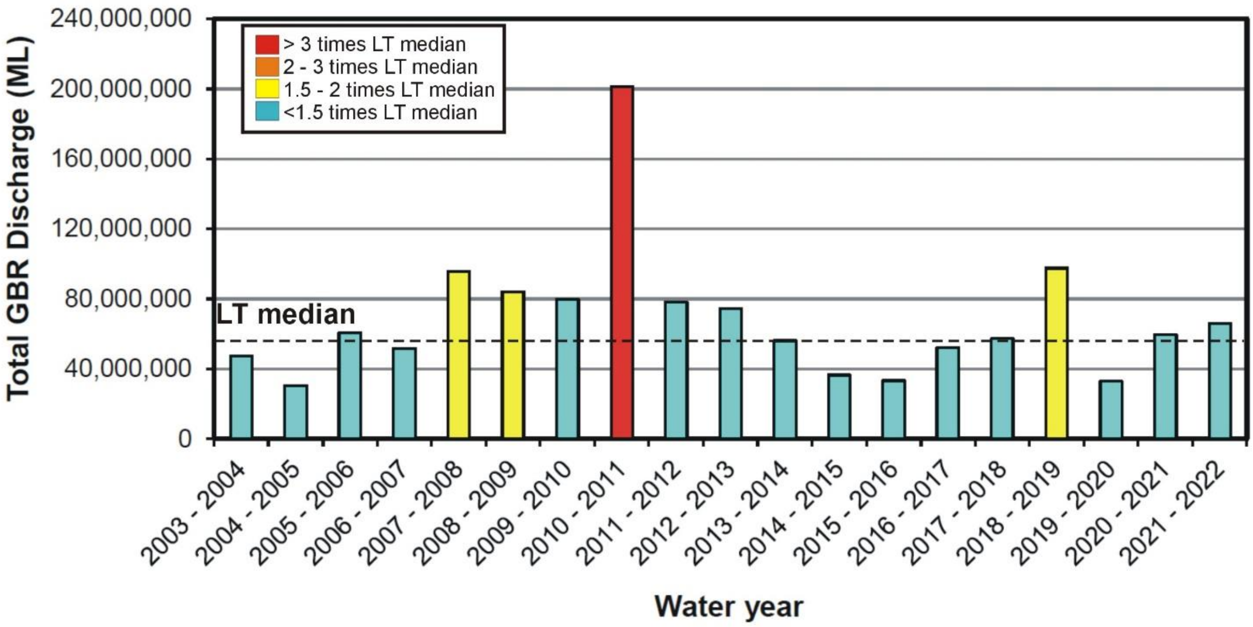

Figure 11 Annual total river discharge to the Reef. Annual estimates aggregate over the water year: 1 October to 30 September, for the 35 main Reef basins. Values are colour coded relative to proportion of long-term (LT) median (1986-2016) discharge. Figure source: Moran *et al*. 2022, data source: DNRM, http://watermonitoring.dnrm.qld.gov.au/host.htm

4 Coral community condition and trend

Results are presented in the following sequence:

- Reef-wide coral community condition (Coral Index scores) and trend (4)
- 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).

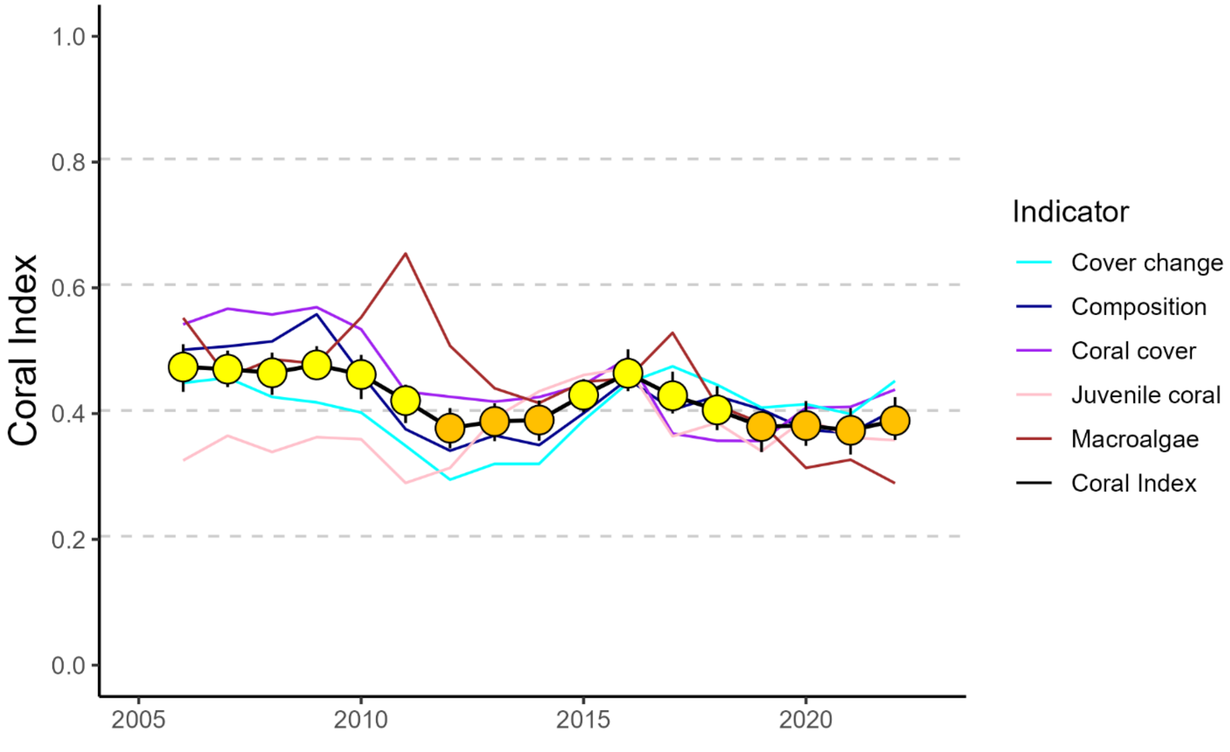

Figure 12 Reef level trend in Coral Index and indicator scores. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = 'poor', yellow='moderate'. Error in Coral Index scores were derived from bootstrapped distribution of regional indicator scores weighted by the relative 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. Yet, it is unsurprising that the current condition has returned to being low given the level of pressure imposed in recent years (see Figure 13). A slight increase in the Coral Index in 2022 reflects small increases in scores for the Coral cover and Cover change indicators, both of which are 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 (45%) 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. Changes in the Composition indicator scores (Figure 12) following acute disturbances indicate that it is species sensitive to poor water quality (primarily *Acropora,* Table A4) that have been disproportionately impacted by these events.

When interpreting Figure 13 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, 2018 when the final MMP reef was resurveyed and 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.

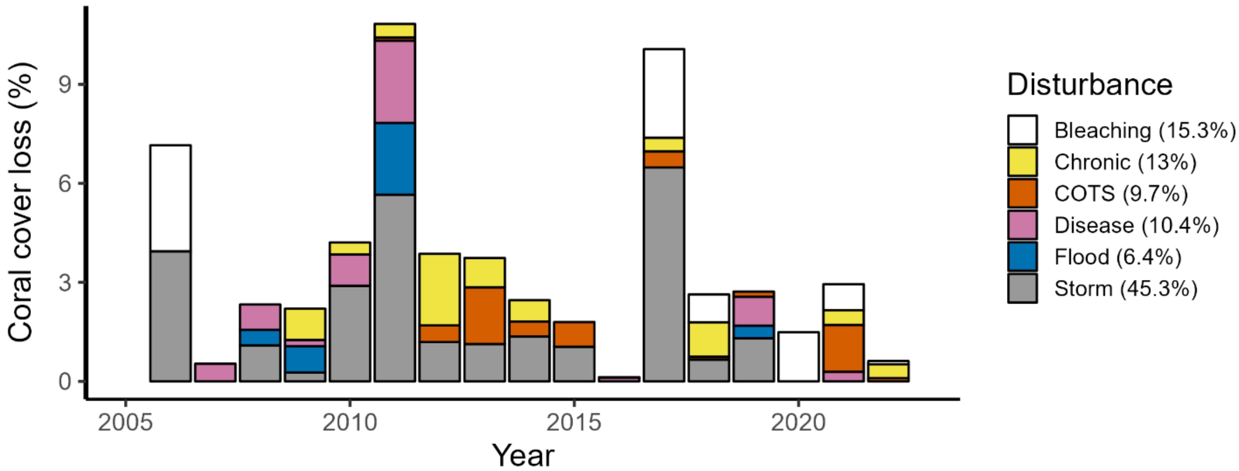

Figure 13 Hard coral cover loss by disturbance type across the inshore Reef. Length of bars represents the mean loss of cover across all reefs in each year. Colours represent the identified cause of cover loss.

Thermal bleaching events have contributed to 15.3% 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, 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 (9.7%, 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 8). 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 (23.4%, 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 runoff in crown-of-thorns starfish outbreak dynamics remains unresolved (Pratchett *et al.* 2017).

4.3 Coral community condition and trends in each (sub-)region

4.3.1 Wet Tropics region

Coral communities in inshore areas of the Wet Tropics remain in 'moderate' condition. The stable condition observed since 2016 (Figure 14) masks differing trends within sub-regions. The stable over-all condition reflects a range of minor disturbances that have variously impacted reefs among the sub-regions, as detailed in the following sections. The high scores for the Cover change indicator in recent years demonstrate the ongoing capacity for coral cover to rebound following these disturbance events. Indeed, the Coral cover score increased into the 'good' range in 2022 for the first time since the start of the monitoring programme. At the regional level, no indicator scores have fallen below moderate levels since 2014.

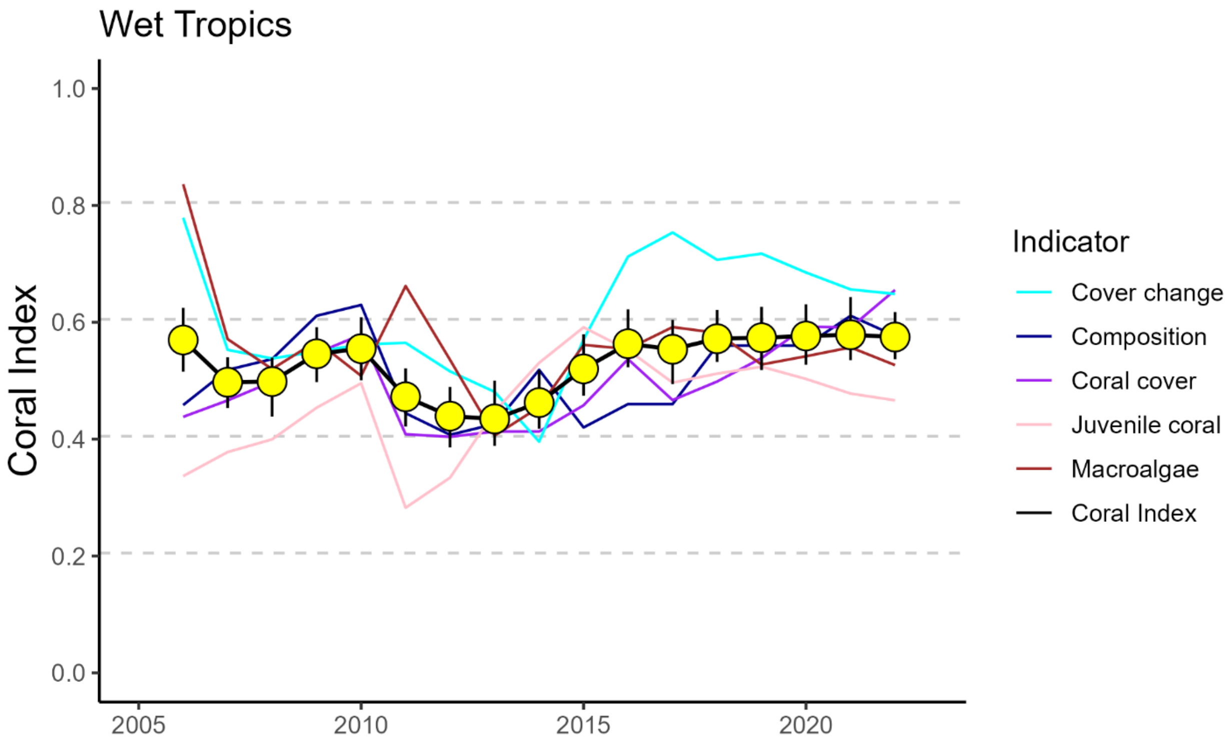

Figure 14 Trends in Coral Index and indicator scores for the Wet Tropics region. Coral Index scores are coloured by report card category: yellow='moderate' Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at individual reefs.

4.3.2 Wet Tropics region: Barron Daintree sub-region

The coral community condition remains within the range of ‘moderate’ but has improved since 2019 (Figure 15). A low point in Coral Index scores was recorded in 2014 following an outbreak of coral disease in 2012, predation by crown-of-thorns starfish since 2012 and then damage attributed to cyclone Ita in April 2014 (Figure 16). Since then, recovery of coral communities has been 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).

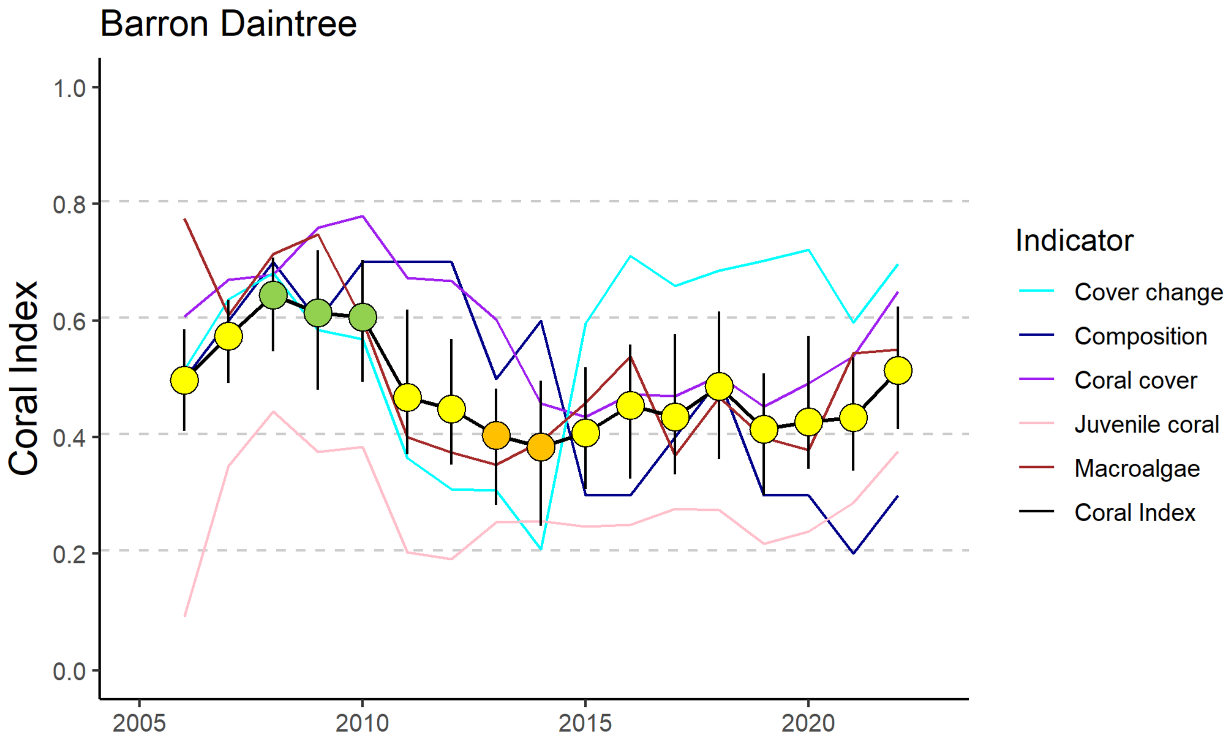

Figure 15 Trends in Coral Index and indicator scores for the Barron Daintree sub-region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = ‘poor’, yellow=’moderate’ and green=’good. Error in Coral Index scores were derived from bootstrapped distributions of indicator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

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	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 2022	2	0.07	1.0	0.20	1.00	-0.06	0.76	0.19	0.80	0.01	0.59	0	NA
	5	0.12	0.83	0.20	0.98	0.29	0.78	0.14	0.80	-0.01	0.51	0	NA

Most indicators have markedly improved since 2014 (Figure 15, Table 10). The Coral cover indicator has seen a gradual rise from a low in 2015, interrupted in 2019 by floodwaters and crown-of-thorns starfish (Figure 16, Figure 17). Between 2021 and 2022 mean coral cover across the region increased from 40% to 50% (Figure 17) with the largest increase occurring at Low Isles where cover jumped from 36% to 55% on the back of increased cover of *Porites*, and Briareidae (Table A8, Figure A1).

The juvenile coral indicator remains in the 'poor' category but has improved to a level approaching that of 2010 (Figure 15) due to an increase in juvenile abundance of *Acropora* and *Porites* spp at Snapper South (2 m), and Merulinidae and Dendrophylliidae families at Low Isles (Figure A1, Table A7).

The Macroalgae indicator remains 'moderate' having paused in a rise from 2020. Very poor scores for this indicator at Snapper North (2 m) and Snapper South (5 m) contrast the 'very good' scores at the other reefs monitored (Table A7). Macroalgae cover at Snapper North (2 m) remains extremely high Figure A1).

The Cover change indicator has transitioned from 'moderate' back to 'good' in 2022, with recent recovery of hard coral cover exceeding modelled predictions at all reefs in the region (Table A7)

The Composition indicator remains 'poor' due primarily to the lower representation of Acropora in coral communities at Snapper North and Snapper South (2 m) compared to that observed during surveys in 2005-2009 (Figure A1, Table A7).

Corresponding to recent improvement in the Coral Index there has been a return to 'good' scores for the Water Quality Index (Figure A10a). This corresponds to a decrease in particulate nutrient and suspended sediment levels and an increase in water clarity; improvements that can be assessed against the GBRMPA guidelines (Figure A10). Not included in the water quality index are concentrations of dissolved organic carbon (DOC) and oxidised nitrogen species (NOx), both of which show substantial increases since 2005 (Figure A10c, j). It remains unclear what has caused these increases (Moran *et al. 2022*) or what the long-term ramifications for corals might be.

The Barron Daintree region experienced above average water temperatures in early 2022 (Figure 16). As with other regions, the effect on the coral communities appears minimal. However, there may have been unobserved impacts as the incidence of disease had increased slightly, though levels remain below the long-term median, and are much less than those of 2009-2011 during a period of sustained increased river discharge (Figure A7, Table A5).

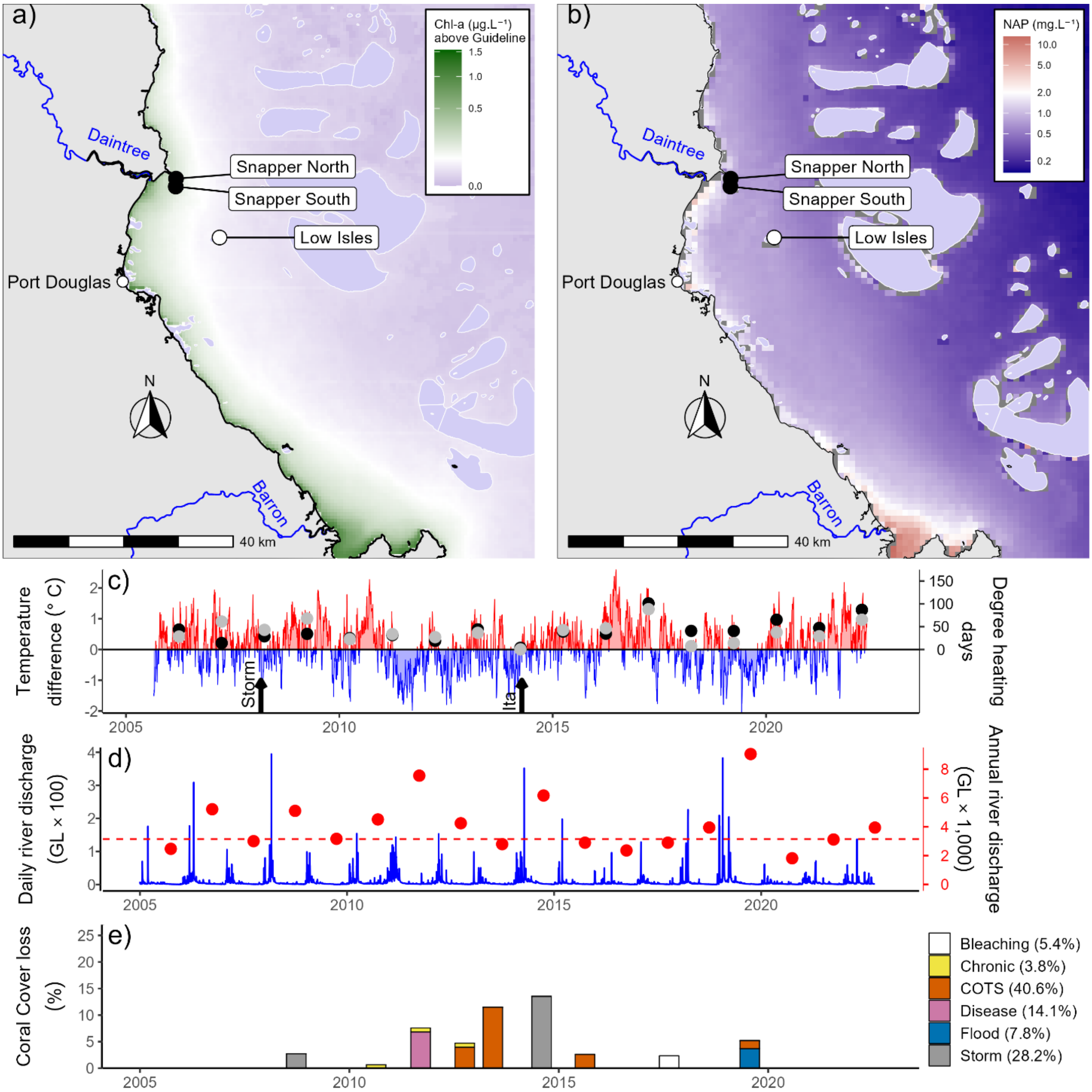

Figure 16 Barron Daintree sub-region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) mean chlorophyll *a* exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are mean levels over the period 2003-2018. c) Seasonally adjusted temperature deviation, timing of cyclones and storms indicated by black arrows, accumulated degree heating days over the summer period (1st of December – 31st 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.

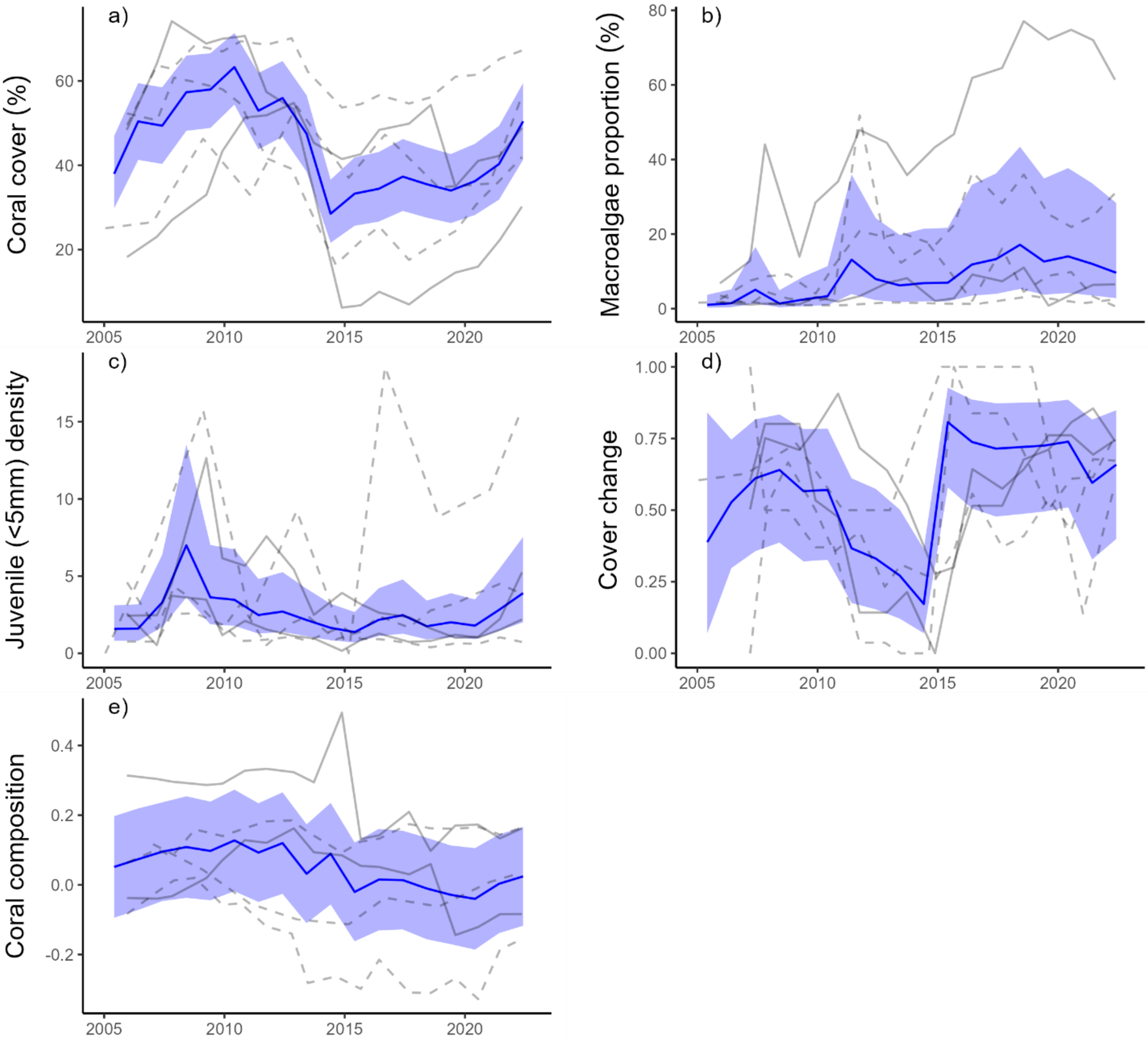

Figure 17 Barron Daintree sub-region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.3.3 Wet Tropics region: Johnstone Russell-Mulgrave sub-region

The 2022 Coral Index score was categorised as 'moderate' having declined slightly since 2021 (Figure 18).

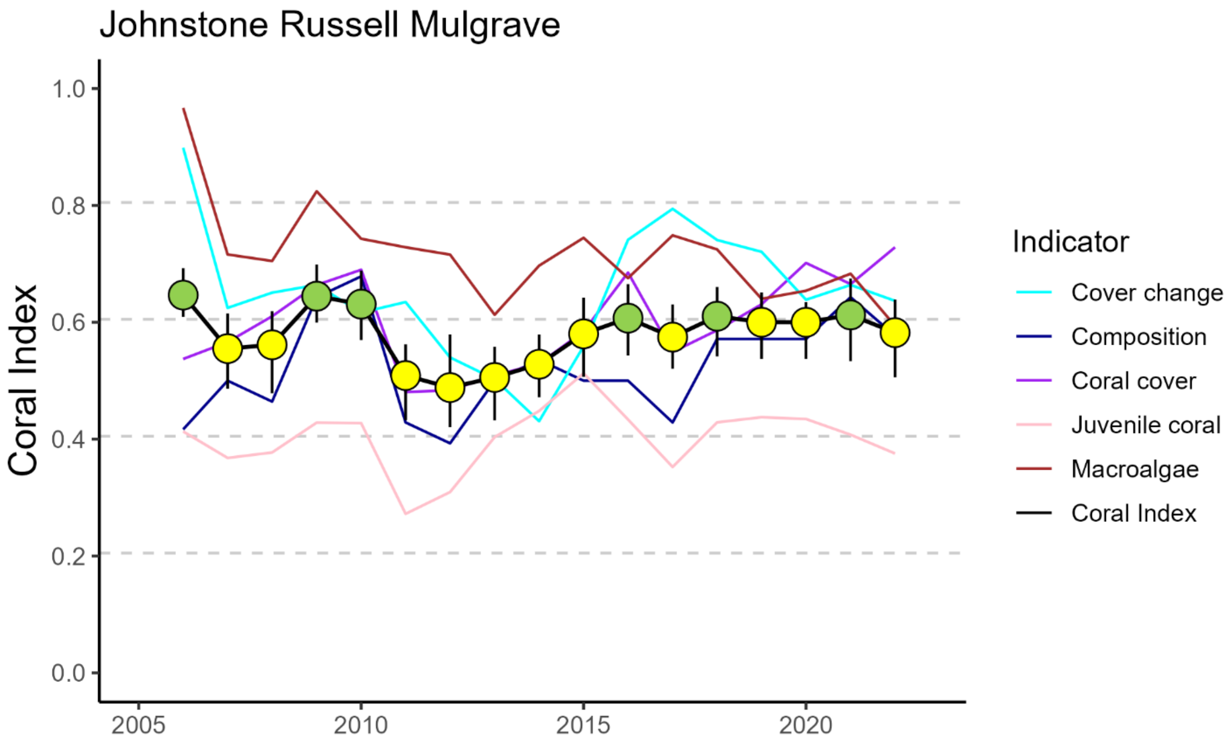

Figure 18 Trends in Coral Index and indicator scores for the Johnstone Russell-Mulgrave sub-region. Coral Index scores are coloured by Reef Water Quality Report Card categories: yellow='moderate' and green='good. Error in Coral Index scores were derived from bootstrapped distributions of indicator scores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

The Coral Index improved between 2012 and 2016 and then fluctuated around the threshold between 'moderate' and 'good' scores through to 2022 (Figure 18). There have been no consistent changes in indicator scores since 2016 (Table 11). The slight decrease in Coral Index score in 2022 results from declines in all indicators except for Coral cover (Figure 18).

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 2022	2	-0.06	0.64	0.02	0.54	-0.20	0.64	-0.00	0.5	-0.19	0.69	0.08	0.57
	5	-0.01	0.53	-0.01	0.52	0.01	0.52	-0.03	0.60	0.04	0.56	0	0.5

Coral cover scores in 2022 reach the highest values observed since the Coral index was introduced in 2006 and remains classified as "good" (Figure 18, Table A7). Coral cover increased at most reefs

(Figure 20a). The largest decline in Coral cover occurred at Fitzroy West (LTMP) where hard coral cover declined by 5% due mostly to a loss of *Acropora* spp. (Figure A11). A decline of 1% coral cover was also observed at Fitzroy East (2 m), again caused by lower cover of *Acropora* (Figure A11). Coral cover at High East (5 m) remained unchanged (Figure A11).The increase in Coral cover scores has been enabled by the rate of increase in hard coral cover during periods that reefs were free from acute disturbances as evidenced by the ongoing 'good' scores for the Cover change indicator (Figure 18).

Macroalgae scores remain 'moderate' in 2022, with an observed very low cover of the persistent brown macroalgal species typical of many inshore reefs (Table A7, Table A11). At Franklands West and, to a lesser degree High East 2 m, red macroalgal species form dense mats among corals (Table A11) leading to low scores for Macroalgae in those locations (Table A7Table A7). The cover of these algae continues to be highly variable among years (Figure 20b).

The juvenile coral indicator score remains relatively low, continuing to be the lowest scoring indicator in the subregion (Figure 18).

In general, the trend in the Coral Index in the 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 at Franklands East, Franklands West and High East. At High West, loss of coral cover at 2 m depth following these cyclones was attributed to low salinity floodwaters (Figure A2, Table A6). The effects of cyclones were further compounded by the increased prevalence of disease in 2011 (Figure 19e, Figure A7). Fitzroy Island, which had escaped serious damage from the cyclones, lost a substantial proportion of hard coral cover to disease; at Fitzroy East between 60% (2 m) and 42% (5 m) of the cover of hard corals, predominantly *Acropora*, was lost (Table A6, Figure A2). This outbreak of disease coincided with high discharge from local rivers (Figure 19d). 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, in addition, by thermal bleaching in 2017.

Crown-of-thorns starfish populations have been at, or near, outbreak levels since 2012 and have been the primary cause of coral cover loss in the region (Figure 19e, Figure A8). In 2022 crown-of-thorns remained at outbreak density at High Island, although only a single juvenile was recorded at Frankland West (Table 9, Figure A8). Limiting the impact of these starfish on coral cover has been the large number removed by the Reef Authority's Crown-of-thorns Starfish Control Program (Table 8).

The marine heat waves that passed through the Reef system in 2020 and 2022 had little influence on the sea temperature in this sub-region (Figure 19c) and there was little evidence of any bleaching-related coral mortality observed during our surveys. The incidence of disease in 2022 increased slightly from that observed in 2021, to around median levels but well below the level associated with high catchment discharge in 2011(Figure A7). Incidences of Brown Band Disease and White Syndrome were observed at Frankland East, Fitzroy East and West and High East.

Discharges from local rivers were slightly above median level over the 2021-2022 water year (Table A5), and peak flows were relatively low (Figure 19d). With the exception of a short flood event in 2018, annual discharges and peak daily flows have been broadly similar since 2012 (Figure 19d) Under these conditions, the coral communities have demonstrated a clear ability to recover when not exposed to disturbances such as thermal stress or crown-of-thorns starfish.

In 2022, most water quality parameters were close to guideline values and the short-term water quality index remained 'moderate', and similar to values seen since 2019 (Figure A11).

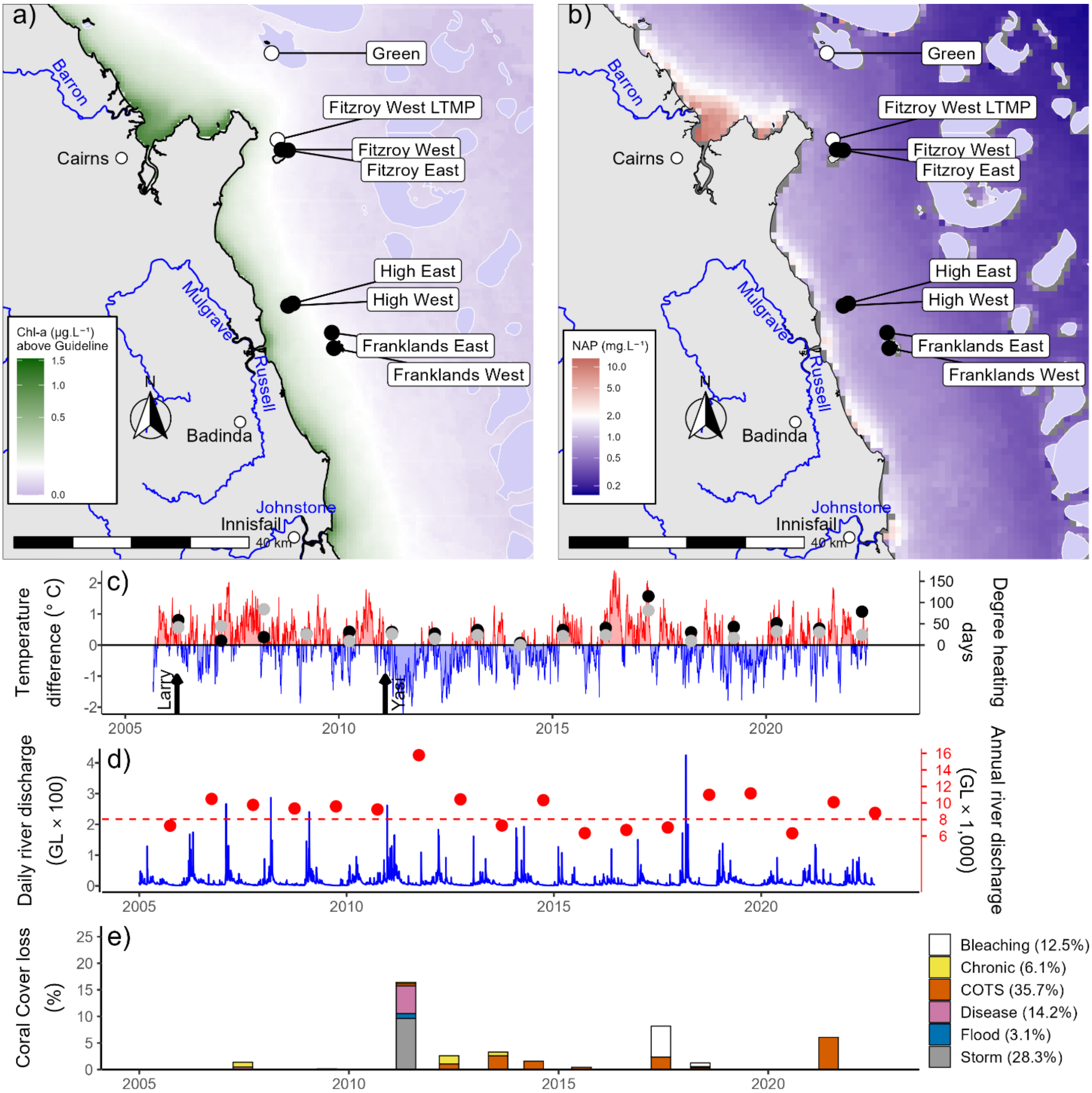

Figure 19 Johnstone Russell-Mulgrave sub-region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) mean chlorophyll *a* exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are mean levels over the period 2003–2018 (Chl) and 2003-2018 (NAP). c) Seasonally adjusted temperature deviation, timing of cyclones and storms indicated by black arrows, accumulated degree heating days over the summer period (1st of December – 31st 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.

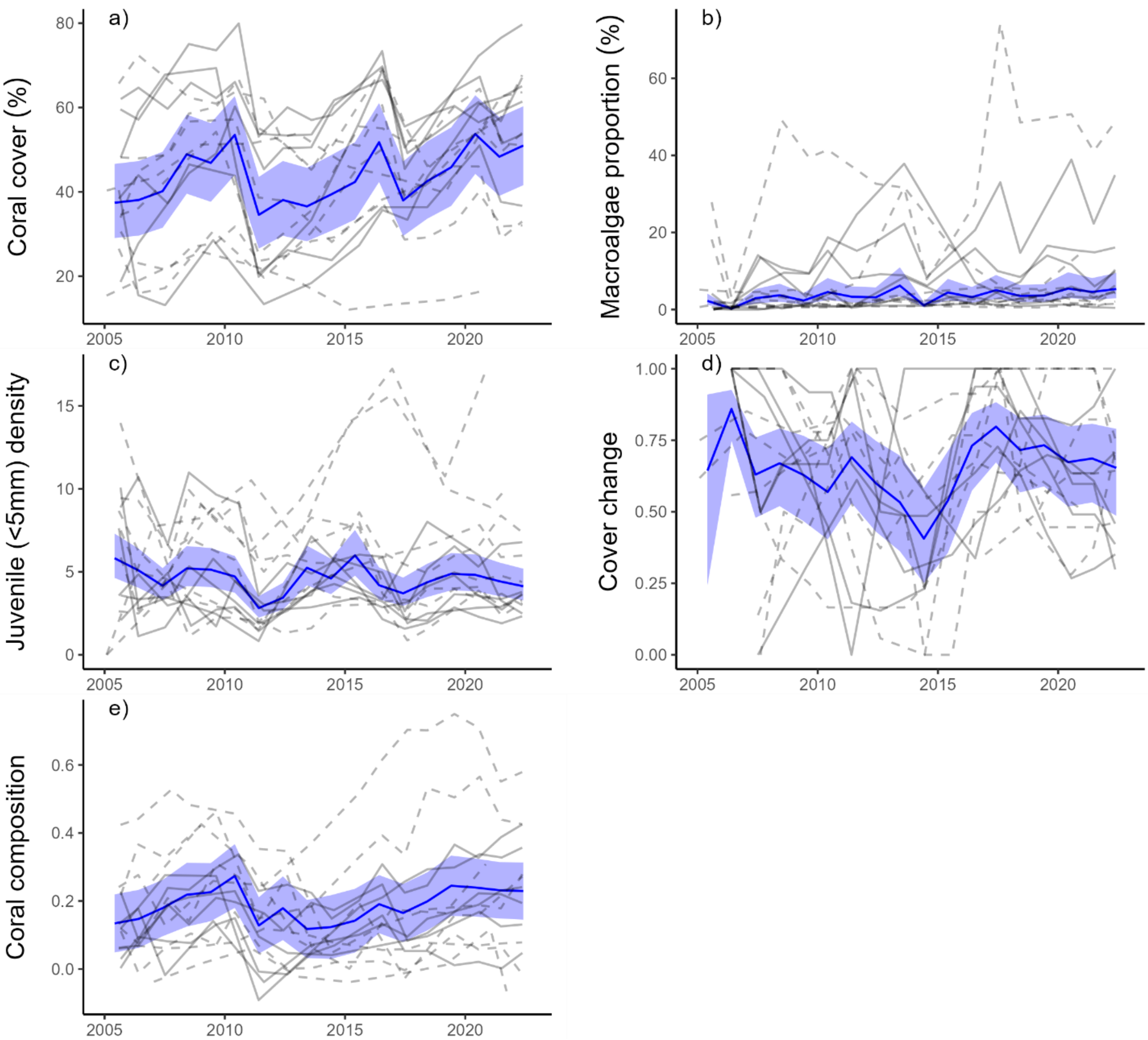

Figure 20 Johnstone Russell-Mulgrave sub-region indicator trends. a– e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.3.4 Wet Tropics region: Herbert Tully sub-region

A slight decline in the Coral Index in 2022 tipped the score into the 'moderate range' for the first time since 2018 (Figure 21).

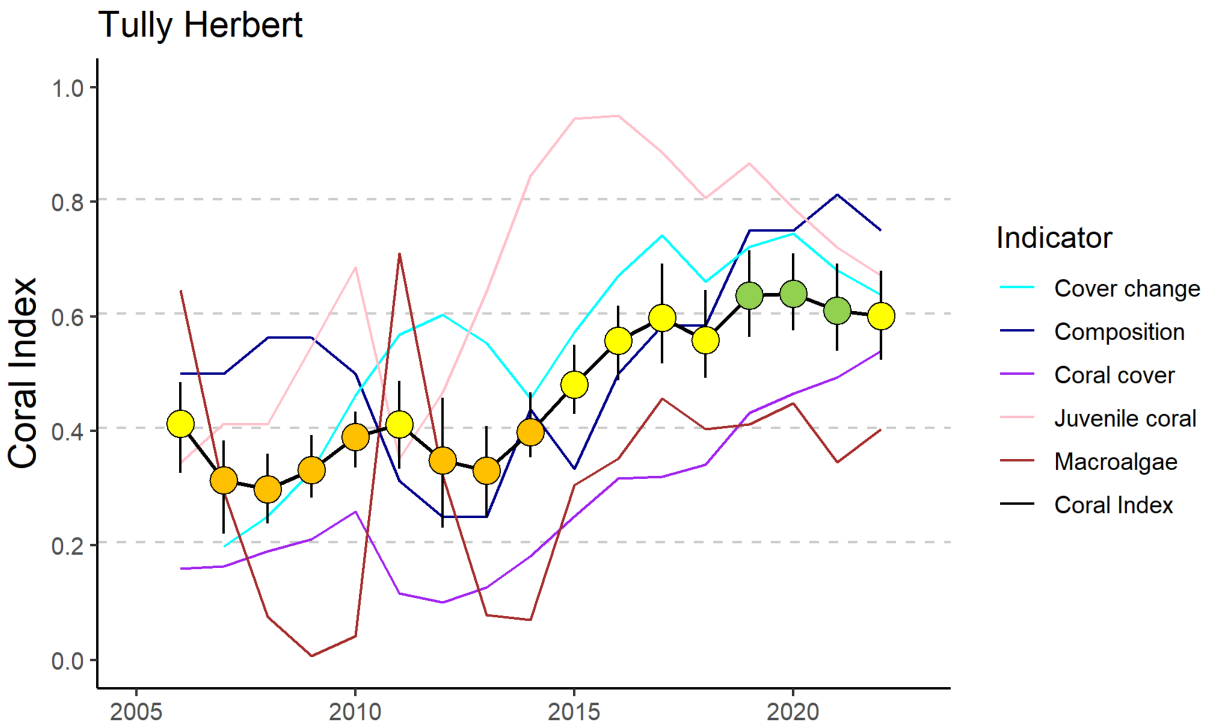

Figure 21 Trends in Coral Index and indicator scores for the Herbert Tully sub-region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = 'poor', yellow='moderate' and green='good. Error in Coral Index scores were derived from bootstrapped distributions of indicator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

The decline in the Coral Index since 2020 is observed to be as a result of a reduction in the Cover change and Macroalgae scores at 5 m depth and overall declining Juvenile coral scores across both depth ranges (Figure 21, Table 12). In contrast Coral cover scores have continued to recover at both depths from the low point reached in 2011 as a result of the severe impact of cyclone Yasi (Figure 21, Figure 22, Figure 23a, Table 12)

Although remaining at 'good' levels, scores for the Juvenile coral indicator have declined since 2019 (Figure 23c) as strong cohorts of *Turbinaria,* which recruited in the years following cyclone Yasi, are growing out of the juvenile size classes (Figure 23c, Figure A3).

Despite the Macroalgae indicator improving to 'moderate' in 2020, in 2021 it returned to 'poor' and while a slight improvement was seen in 2022, the score is still categorised as 'poor'. Scores for this indicator remain at minimum levels of zero at the 2 m depth of Bedarra and Dunk North, and the 5 m depth at Dunk South (Table A7). At these reefs, the macroalgae community is dominated by persistent brown algae of the genus *Sargassum* and *Lobophora* (Table A11).

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 2022	2	0.05	0.67	0.09	0.90	0.02	0.68	-0.08	0.80	0.00	0.52	0	NA
	5	-0.10	0.90	0.06	0.82	-0.11	0.78	-0.15	0.80	-0.22	0.95	0	NA

Since monitoring began in 2005, changes in the Coral Index show a repeat sequence of disturbance and subsequent recovery. Cyclone Larry in 2006 and cyclone Yasi in 2011 severely impacted coral communities, yet rapid recovery occurred in both instances (Figure 22c,d, Figure 23a). The combined impacts of cyclones and storms account for 72% of hard coral cover losses since 2005 (Figure 22e).

Following each cyclone there was an immediate reduction, then protracted decline in the Coral Index scores (Figure 21). This prolonged response primarily reflects an initial improvement, then rapid decline in the Macroalgae indicator scores (Figure 23d, Table 12). During cyclones, macroalgae are stripped from the substrate, temporarily reducing their abundance. Their subsequent recolonisation of the space made available by the cyclone results in an extended decline for the Coral Index scores.

At all sites in 2022 there was an increase in Coral cover (Figure 23a), owing to increases in cover of the genus *Acropora* (predominantly at 2 m depth) and the transition of juvenile *Turbinaria* (Family: Dendrophylliidae) to adult size class (Figure A3). While Cover change score for the region remains 'good' it has declined since 2020, particularly at 5 m depths (Table 12, Figure 21, Figure 23d,). The Cover change indicator is estimated as a rolling mean over scores for the past four years. During this period, levels of disease were above median levels (Figure 22e) suggesting the rate of coral cover increase has been suppressed by the cumulative pressures associated with thermal stress in 2020 and 2022 and above median river discharge in 2021 (Figure 22, Table A5).

The coral sampling sites in this sub-region are primarily influenced by discharge from the Tully and Herbert rivers. All the coral monitoring sites in this sub-region are situated in nutrient rich waters (mean Chl *a* concentration over the wet season exceed the guideline; Figure 22a, Table A8). The combination of high turbidity and high nutrient availability (Figure A12) is consistent with the prevalence of macroalgae observed in the shallow, but not deeper, depths at most reefs (Figure 23b, Figure A3). The long-term water quality index for this sub-region remains poor (Figure A12a). The short-term water quality index remains 'moderate' although it has declined since 2020 (Figure A12).

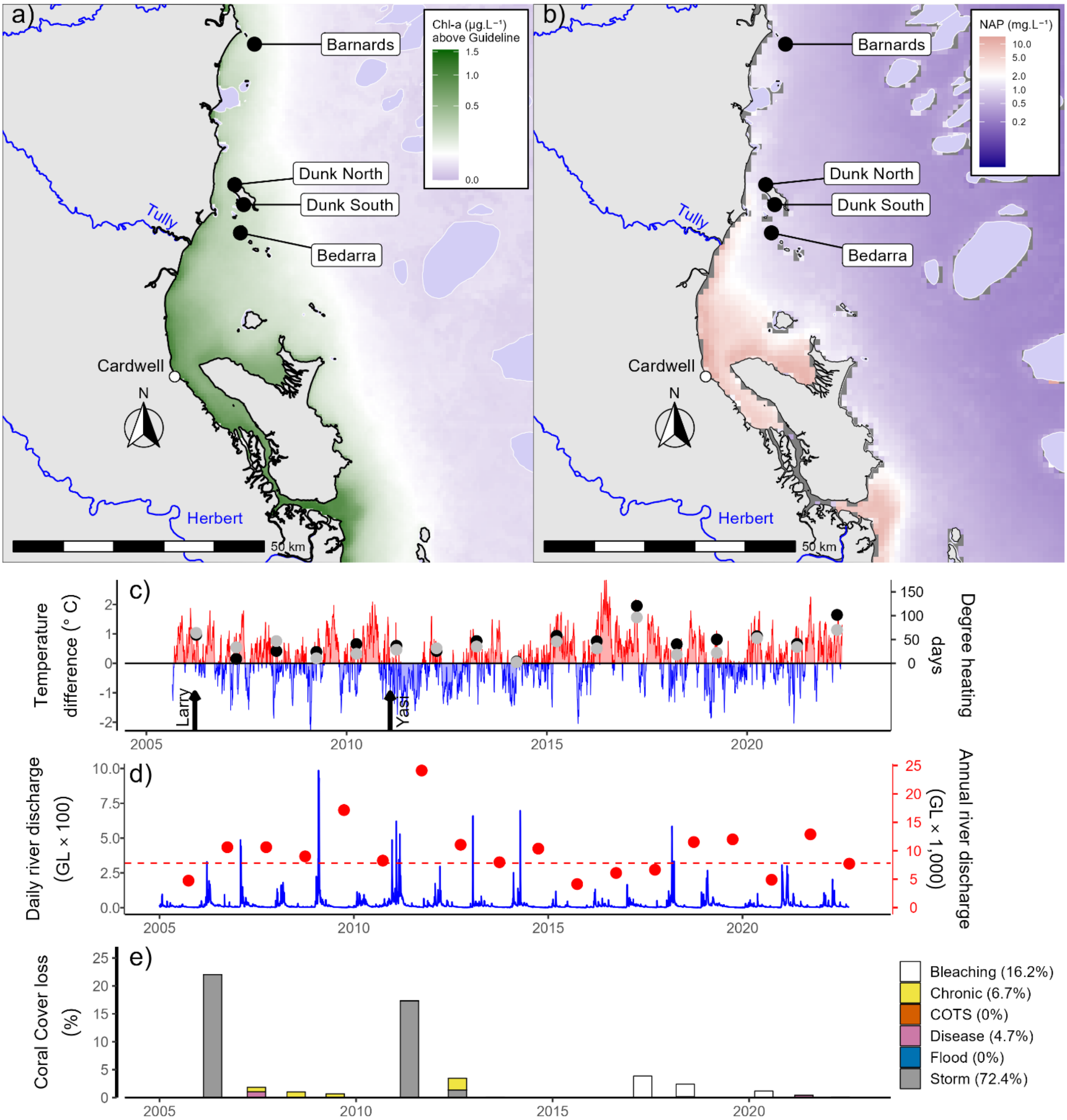

Figure 22 Herbert Tully sub-region environmental pressures. Maps show location of monitoring sites along with, a) mean chlorophyll a exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are mean levels over the period 2003–2018 (Chl *a*) and 2003–2018 (NAP). c) Seasonally adjusted temperature deviation, timing of cyclones and storms indicated by black arrows, accumulated degree heating days over the summer period (1st of December – 31st March) as reported by BoM (black symbols) and derived from in situ loggers (grey symbols) d) Combined daily (blue) and annual (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.

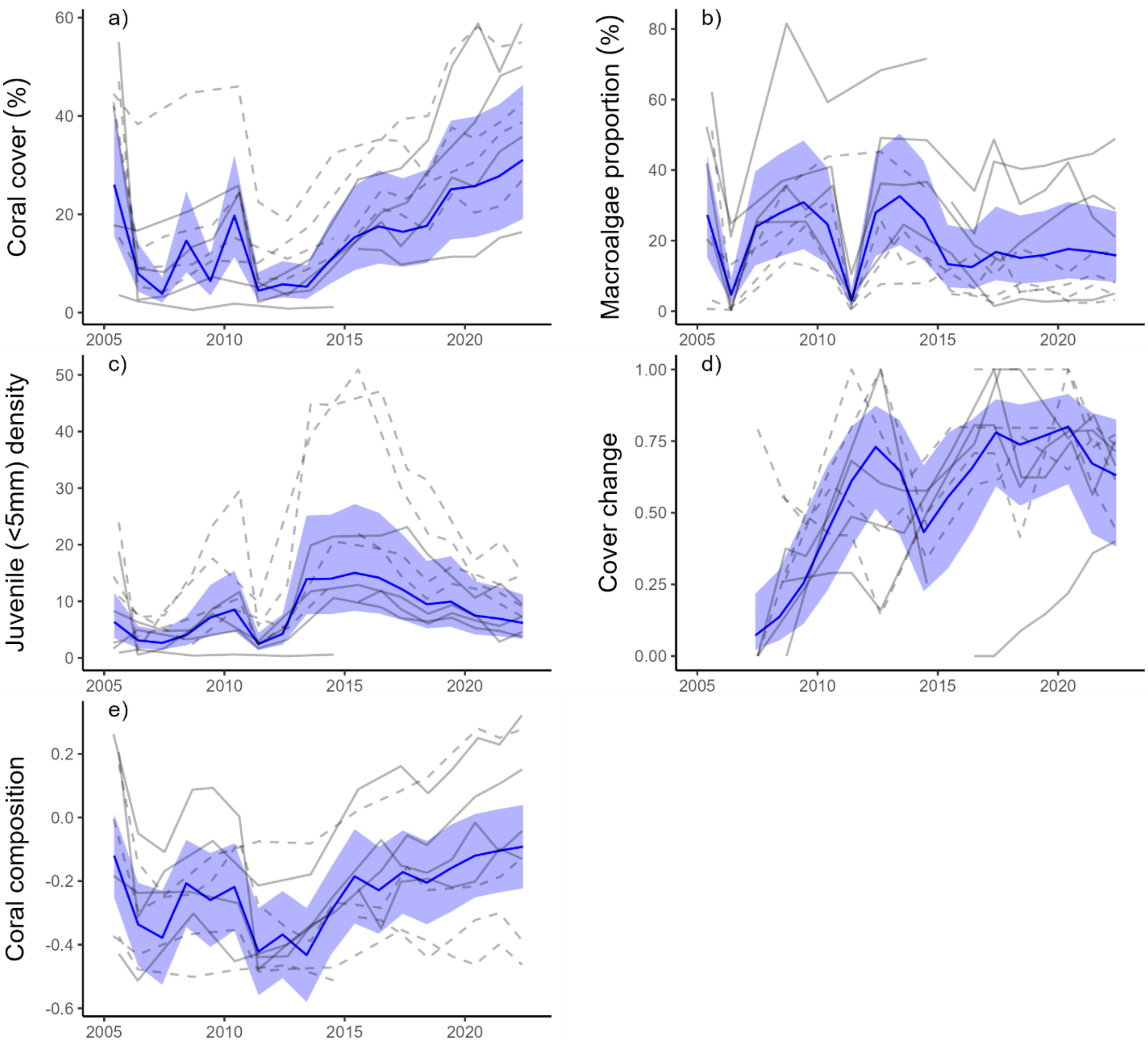

Figure 23 Herbert Tully sub-region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.4 Burdekin region

The Coral Index remained in moderate condition but has continued to decline since 2020 (Figure 24).

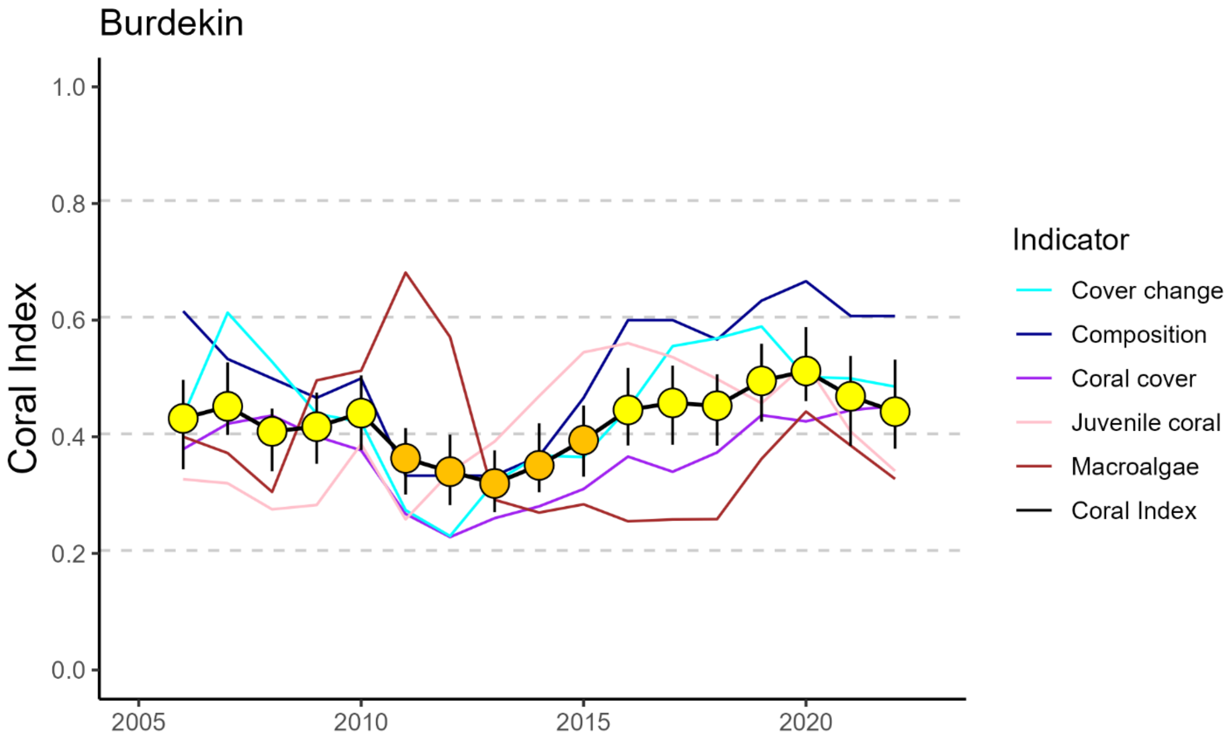

Figure 24 Trends in Coral Index and indicator scores for the Burdekin region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = 'poor', yellow='moderate'. Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at individual reefs.

Declines in the Coral Index since 2020 reflect declines in scores for Macroalgae and Juvenile coral at both depths, and at 2 m depth, Composition (Figure 24, Table 13). In contrast, the Coral cover indicator improved over the last two years at 2 m depth (Table 13). The improvement in the Coral cover captures the ongoing recovery of hard coral communities following a period punctuated by high discharge from the region's catchments and exposure to physical damage from storms and cyclones between 2009-2012 (Figure 25c, d, e).

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.

Period	Depth	Coral Index		Coral cover		Macroalgae		Juvenile coral		Cover change		Composition	
		Score	P	Score	P	Score	P	Score	P	Score	P	Score	P
2010 to 2013	2	-0.08	0.70	-0.09	0.64	-0.17	0.71	-0.04	0.61	-0.05	0.53	-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	0.87	0.18	0.77	0.26	0.87	0.33	0.89	0.31	0.76
2020 to 2022	2	-0.11	0.78	0.07	0.92	-0.26	0.73	-0.12	0.82	0.03	0.56	-0.25	0.73
	5	-0.06	0.81	0.01	0.53	-0.064	0.74	-0.23	0.76	-0.05	0.59	0.06	0.66

There were no severe storms or cyclones that threatened the Burdekin inshore reefs over the summer of 2021-22 (Figure 8) and discharge levels from all river systems were at, or below, long-term medians (Figure 25d, Table A5).

The primary pressure to have influenced coral communities between 2021 and 2022 surveys was a marine heat wave during early 2022 (Figure 9, Figure 10, Figure 25c). Although surveys in 2022 were undertaken over Winter in June and July, bleached or partially-bleached corals were observed at most reefs. The highest levels of pale corals were observed at Palms West 2 m and Magnetic 5 m where 10% and 14 % of coral was observed to be partially bleached. Despite the levels of bleaching observed, there were not large reductions in the cover of hard corals. . In contrast, soft coral cover declined (or remained at zero) at almost all reef and depth combinations surveyed, the one exception being Palms East 2 m where a modest increase was observed in 2022 although cover remains low (Figure A4). There were modest gains in hard coral cover at Havannah, Palms West, Pandora, and Lady Elliot (2 m). Increases were attributed to recovery of *Acropora, Montipora, Isopora* spp, and a suite of low-abundance genera (Table A9, Figure A4). The largest decline in coral cover occurred at Palms East, where cover declined from 45.5% in 2021 to 43.1% in 2022 (Figure A4). This was one reef at which bleached corals were not observed and the cause of this decline appears to have been white syndrome disease amongst *Acropora*.

Cover change indicator score for the region has remained ‘moderate’ with a slight upward trend indicating recovery from the 2020 bleaching event (Figure 26d, Table A7) and an ongoing positive balance between losses and gains in cover in 2022. At the reef level, the Cover change indicator appears more animate as it tracks various impacts and ensuing recovery. For example, at Havannah (2 m), a protracted decline in coral cover was observed through to 2021 followed by improvement in 2022 (Figure A4).

The Macroalgae indicator has continued to decline and remains ‘poor’ (Figure 24, Table A7). Very poor scores were recorded at Havannah North, Havannah, Lady Elliot (2 m), Pandora (2 m), Pandora North and Magnetic where the cover of macroalgae increased or remained at high levels (Table A7, Figure A4). The macroalgal communities are dominated by large brown species of the genus *Lobophora* and/or *Sargassum* at Havannah North, Havannah, Magnetic, Pandora (2 m) and Pandora North, while a mix of red macroalgae species including *Hypnea* and the brown macroalgae, *Dictyota,* are common *at* Lady Elliot (2 m) (Table A11).

Since 2014, Juvenile coral scores had remained ‘moderate’ (Figure 24) though highly variable among reefs (Figure 26c). Yet in 2022, regional juvenile density has declined into the ‘poor’ range (Table A7), with declines in density observed at all reefs (Figure 26c). Juvenile coral indicator scores transitioned to a lower category on reef slopes (5 m) at Palms East, Palms West, Pandora, and for both depths at Lady Elliot (Table A7), with reductions in juvenile genus groups *Acropora*, *Montipora*, and *Porites* spp, and family groups Dendrophylliidae and Merulinidae (Figure A4).

The Composition indicator for the region has declined from a high in 2020 to pause on the boundary of ‘good’ in 2022 (Figure 24 Table A7). This was preceded by a modest but steady rise following a pattern of recovery from the impacts of TC Yasi and subsequent flood plumes of 2011 (Figure 26e). Progress temporarily stalled, principally due to the effect of bleaching events, in 2016, 2017, and 2020. Between 2021 and 2022 the Composition indicator remained static with an equal balance of contributing reef-level scores, (Table A7).

While concentrations for most water-quality parameters declined in 2022, the rise in the concentration of NOx and turbidity levels (Figure A13c, e) has kept the short-term Water Quality Index fluctuating on the border between ‘moderate’ and ‘good’ (Figure A13a). Concentrations of NOx have remained above guideline values for the duration of the program (Figure A13c). Concentrations of dissolved organic carbon have markedly increased over the same period (Figure A13), however this parameter does not contribute to water quality index scores (Gruber *et al.* 2020). At the regional scale, the declining scores for the Macroalgae indicator, demonstrate that while coral communities have retained a degree of resilience the availability of nutrients at many locations are likely to be limiting their overall condition.

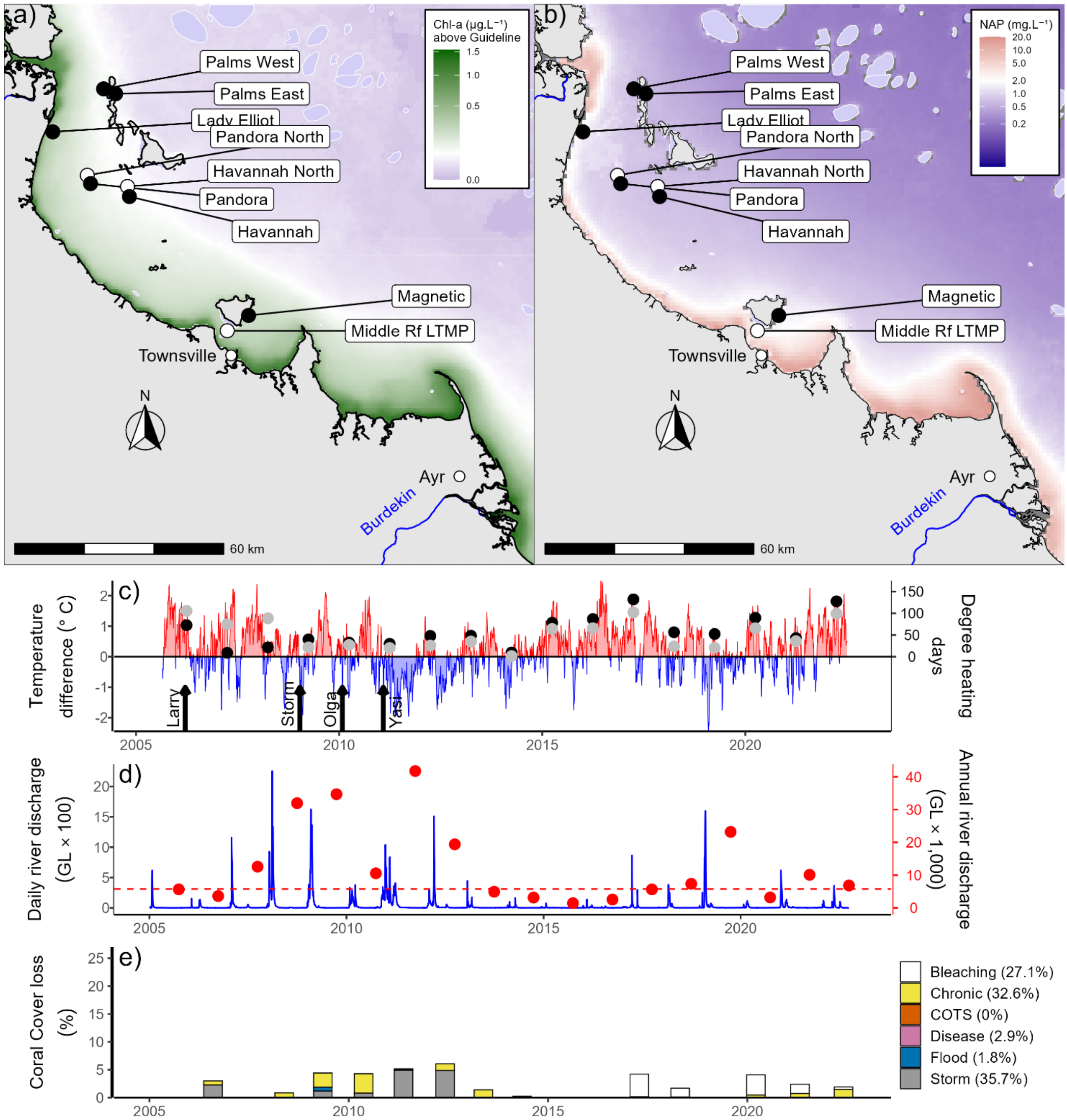

Figure 25 Burdekin region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) mean chlorophyll *a* exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are mean levels over the period 2003–2018 (Chl *a*) and 2003–2018 (NAP). c) Seasonally adjusted temperature deviation, timing of cyclones and storms indicated by black arrows, accumulated degree heating days over the summer period (1st of December – 31st 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.

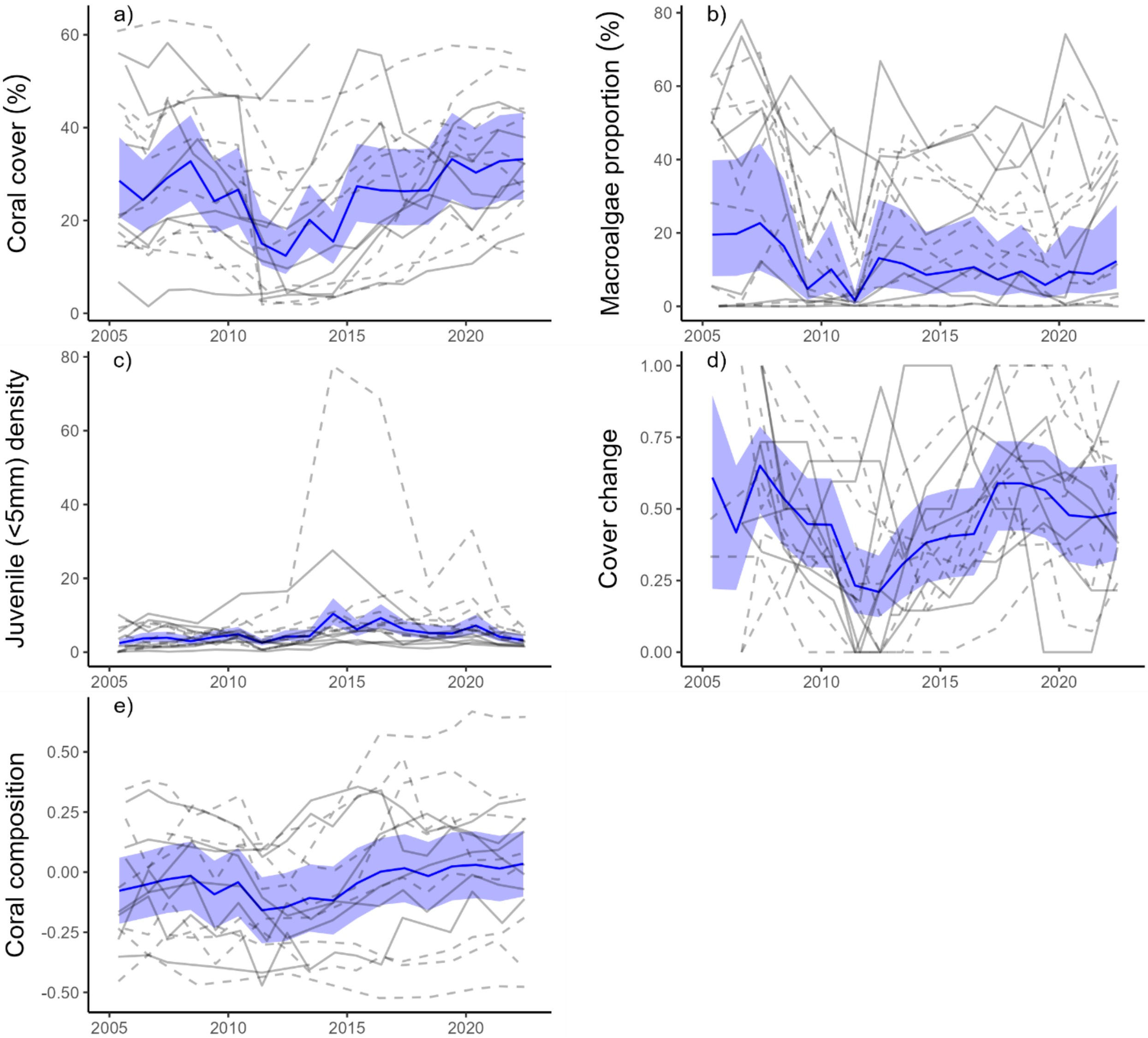

Figure 26 Burdekin region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.5 Mackay-Whitsunday region

The Coral Index score remained 'poor' in 2022 (Figure 27). Over the period 2016 – 2020 there were region-wide reductions for all indicators after the region was impacted by cyclone Debbie in 2017 (Table 14, Figure 28e). Most improved in 2022 was the Juvenile coral indicator which is now within the 'moderate' score range (Table 14, Figure 27). All other indicator scores remain in the 'poor' range (Figure 27).

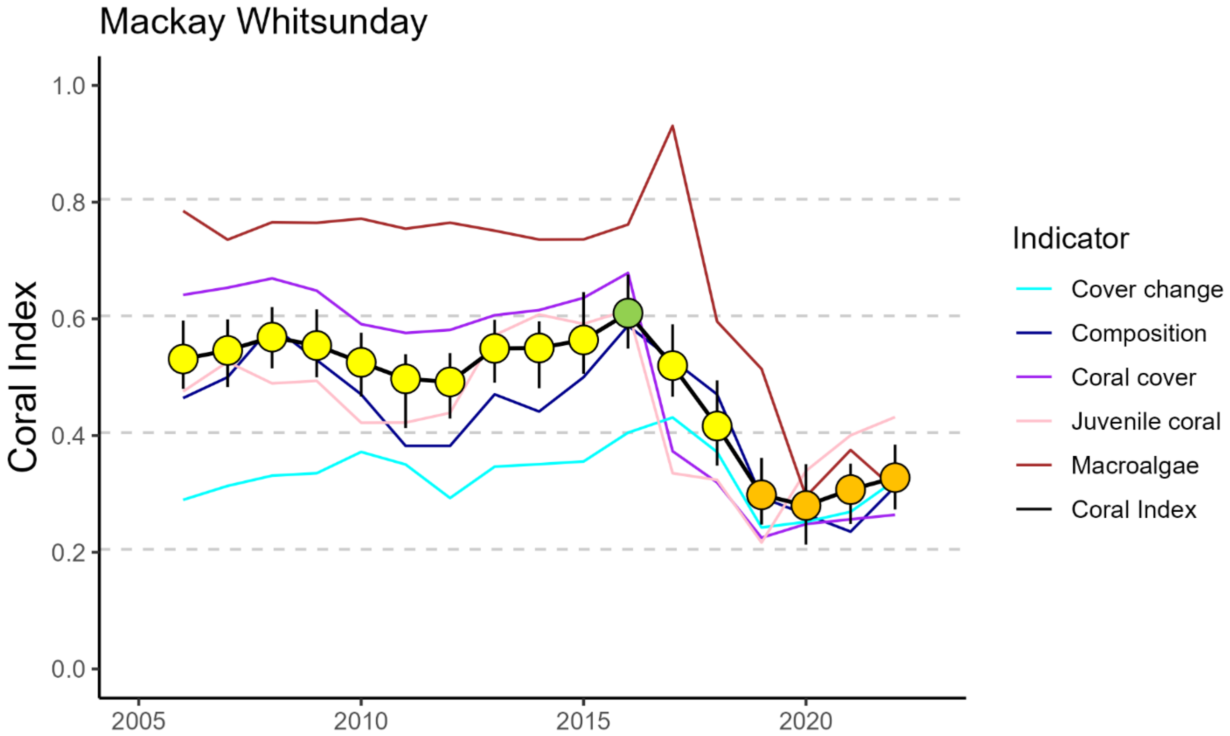

Figure 27 Trends in Coral Index and indicator scores for the Mackay-Whitsunday region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = 'poor', yellow='moderate', green='good'. Error in Coral Index scores were derived from bootstrapped distributions of indicator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

Scores for the Cover change indicator have been consistently low (remaining 'poor' for most years), showing slow rates of hard coral growth relative to other regions (Figure 27). The lowest scores for Cover change were observed between 2019 and 2021 as coral cover showed limited recovery following the severe impact of cyclone Debbie (Table 14). In 2022 the score had improved, although this improvement was variable among reefs and the score remains 'poor' (Figure 27, Table 14, Figure 29). Low scores for this indictor are of particular concern in this region where persistently high turbidity (Figure A14) has selected for relatively slow growing taxa at many of the deeper sites (Figure A5, Table A9Table A9). The slow growth of corals in families other than Acroporidae is implicitly accounted for in the modelled expected changes in coral cover that underpin the Cover change indicator. Further, reductions in the Composition score following cyclone Debbie reflect the disproportionate loss of Acroporidae corals and so further reduce the modelled expectation for increase in hard coral cover. While remaining regionally low in 2022, Cover change indicator scores varied among reefs with values greater than 0.5 estimated at Hayman, and at Hook 5 m, where the cover of Poritidae, genus *Porites,* has increased (Figure A5, Table A9, Table A7).

Juvenile coral indicator scores declined steeply following cyclone Debbie but are beginning to rebound (Figure 27, Figure 29c). The abundance of juveniles has increased at all reefs since initial post cyclone Debbie observations (Figure A5). The considerable rise in density of juvenile corals at Hayman was predominately among the genus *Acropora*, a group previously well represented in the adult population (Figure A5). At Daydream increased density of juvenile corals included a high proportion of *Turbinaria* (Dendrophylliidae), a genus that was not well represented in the adult community prior to cyclone Debbie (Figure A5). The 5 m depth at Daydream is starting to show an increase in juvenile *Acropora* where silt and sediment depositions are decreasing and water quality improving (Figure A14). Although the number of juvenile corals observed along transects is close to or above values seen prior to cyclone Debbie (Figure A5), this is not fully reflected in the Juvenile indicator scores (Figure 29c) as the area of transects available to juvenile corals remained high as a result of lower coral cover (Figure 29a).

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 2022	2	0.04	0.62	0.02	0.65	0.00	0.51	0.05	0.75	0.00	0.51	-0.07	0.66
	5	0.06	0.68	0.01	0.62	0.10	0.65	0.11	0.68	0.05	0.56	0	0.50

Improvement in Macroalgae indicator scores had tracked the regional improvement in Juvenile coral scores, until this year (2022) when the Macroalgae score decreased, although the Juvenile coral score has continued to improve (Figure 27). The Macroalgae scores remain well below those observed prior to cyclone Debbie (Figure 27). In 2022, Macroalgae scores of zero were recorded at both 2 m and 5 m depths at Double Cone, Daydream, Dent, Pine and Seaforth (Table A7Table A7). With the exception of Seaforth, macroalgae cover had increased at one or both depths compared to levels observed in 2021 (Figure A5). However, these increases had very little or no influence on the indicator scores as, apart from Dent 5 m and Daydream 5 m, scores were also zero in 2021 (Thompson *et al.* 2022).

High turbidity across the region (Figure A14) in combination with limited exposure to wave energy among the Whitsunday Islands, results in reduced availability of light and accumulation of fine sediments, particularly at deeper sites as a result of reduced wave energy (Wolanski *et al.* 2008). Over time, coral species tolerant of these conditions (Agariciidae, Euphylliidae, Lobophylliidae, Poritidae (genus *Goniopora*)) have been selected for at the 5 m depths, while Acroporidae and Poritidae (genus *Porites*) are most common at the 2 m depths (Figure A5). The pressure imposed by poor water quality in this region is also expressed by relatively low scores for the Cover change indicator (Figure 27), which in turn contribute to the frequently categorised chronic stresses (Figure 28e). This is particularly a concern for reefs dominated by corals other than Acroporidae, as their growth expectation is low within the model. Although both the long-term and short-term water quality index scores show that water quality is generally below the GBRMPA guidelines in 2022 (

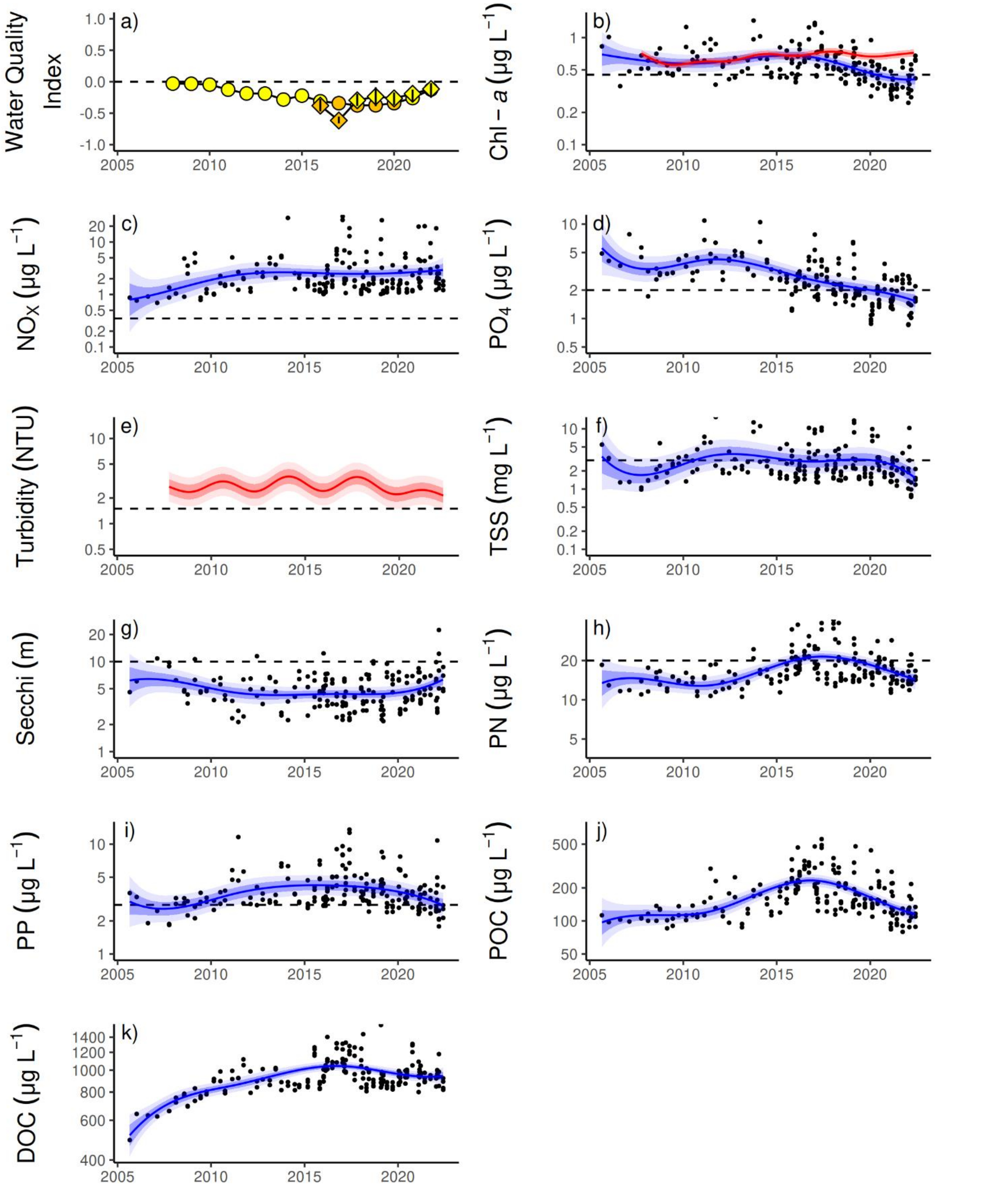

Figure A14), although these scores are on an upward trend.

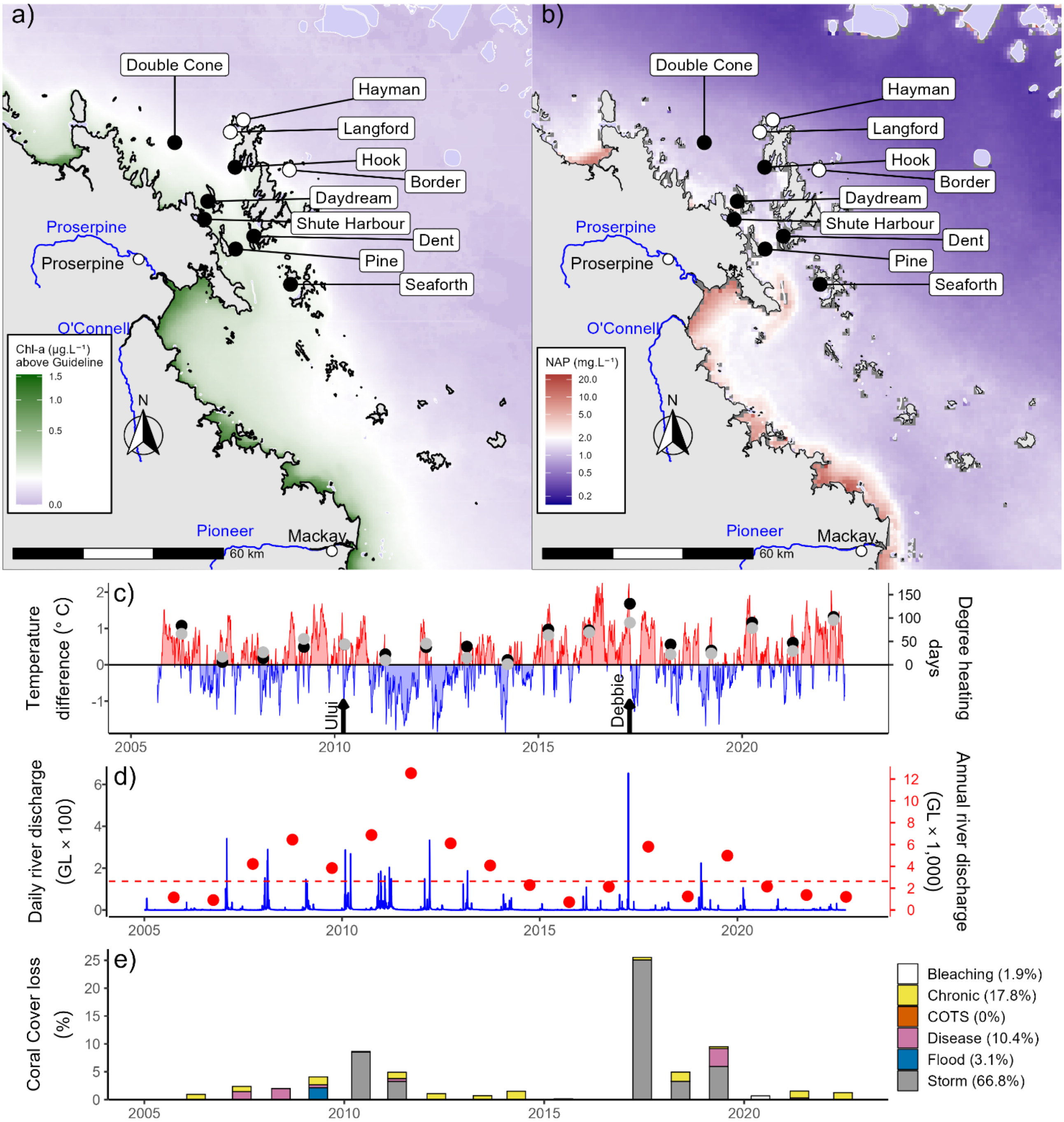

Figure 28 Mackay-Whitsunday region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) mean chlorophyll *a* exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are mean levels over the period 2003–2018 (Chl *a*) and 2003–2018 (NAP). c) Seasonally adjusted temperature deviation, timing of cyclones and storms indicated by black arrows, accumulated degree heating days over the summer period (1st of December – 31st 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.

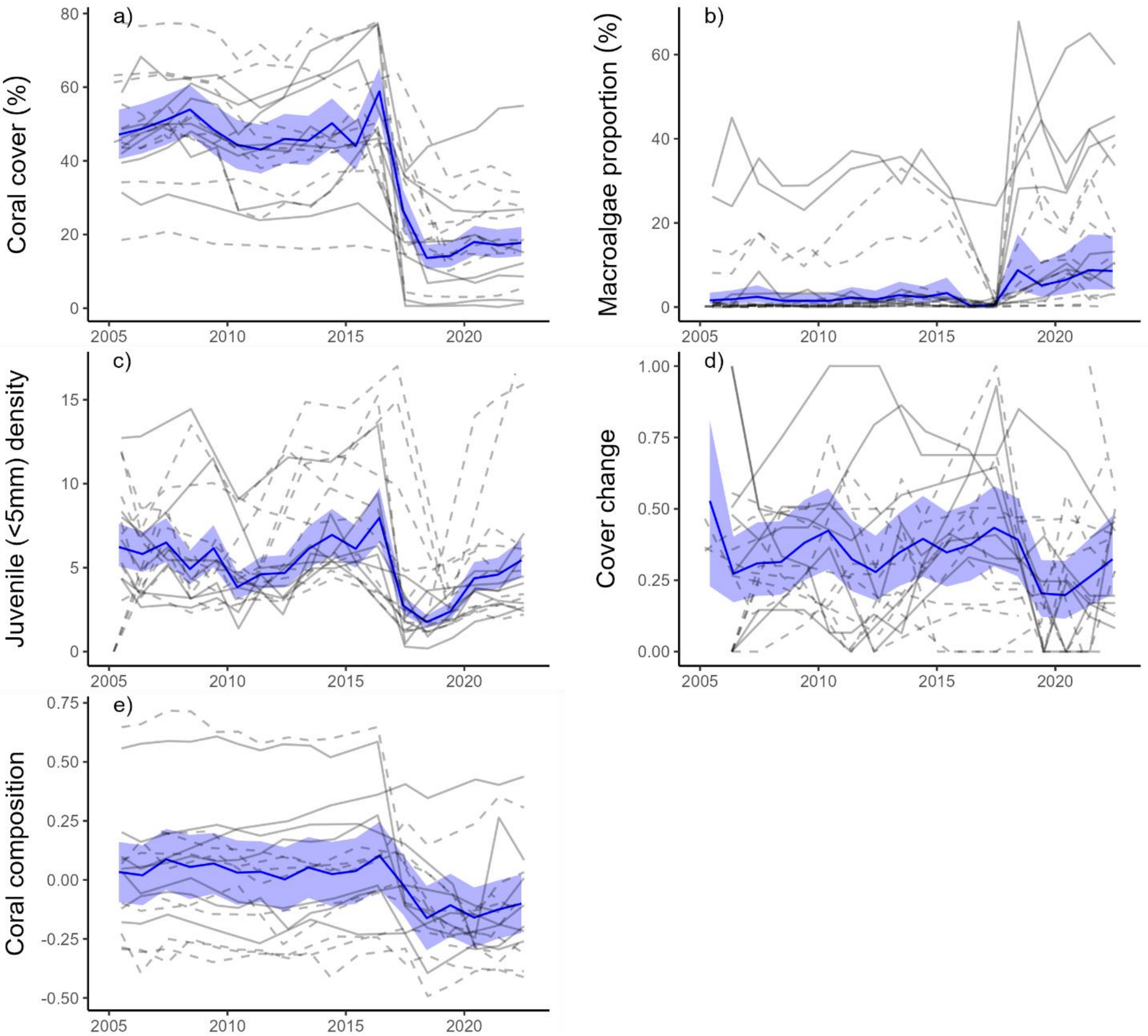

Figure 29 Mackay-Whitsunday region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.6 Fitzroy region

The Coral Index score in the Fitzroy region remains 'poor' despite resuming the general improvement in scores since 2014 (Figure 30,Table 15) following the short-lived reduction presented in 2021.

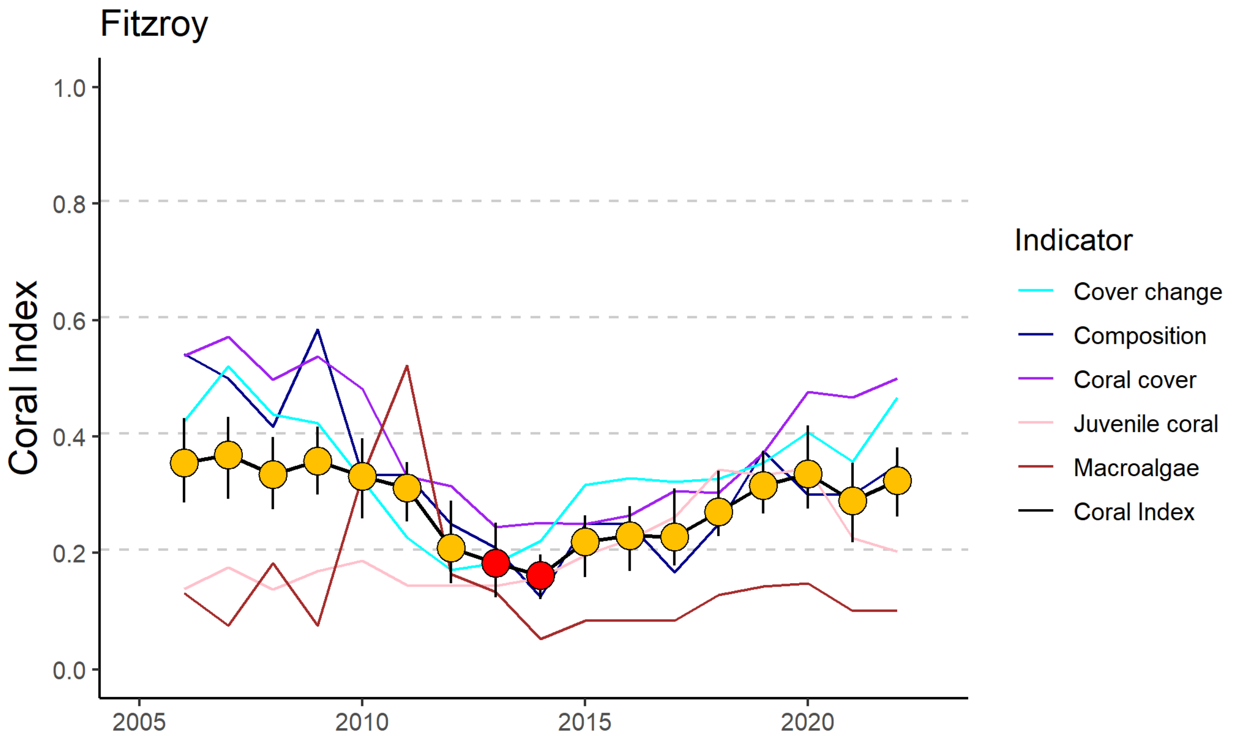

Figure 30 Trends in Coral Index and indicator scores for the Fitzroy region. Coral Index scores are coloured by Reef Water Quality Report Card categories: red='very poor', orange = 'poor'. Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at individual reefs.

Across the region the Coral cover and Cover change indicators were in the 'moderate' range (Figure 30). These two indicators were the most influential in the increase in the Coral Index from 2021 to 2022 (Table 15). The improvement in the Cover change score is noteworthy as this is the first time since 2009 that the rate of coral cover increase has reach modelled expectations over the four-year running mean on which the score is based (Figure 30).

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	0.09	0.71	0.22	0.81	0.23	0.90	0.3	0.71
2021 to 2022	2	0.02	0.64	0.05	0.89	0	NA	-0.06	0.64	0.09	0.79	0	NA
	5	0.05	0.82	0.01	0.74	0	NA	0.01	0.54	0.13	0.80	0.1	0.61

The monitored coral reefs are situated along a distinct environmental gradient within Keppel Bay. Pelican is situated in relatively turbid and nutrient rich waters compared to reefs further offshore (Figure 31a, b). Keppels South, Middle and North Keppel are exposed to concentrations of Chl *a* that exceed guideline values, whereas at Barren the Chl *a* level is lower; all four reefs share reasonably low levels of total suspended solids (Figure 31a, b, Table A8). At Pelican, benthic communities differ markedly between 2 m and 5 m depths (Figure A6) illustrating the substantial attenuation of light due to the high turbidity. The differences in community composition are evident in the baseline conditions for the Composition indicator (Figure 32e). At Pelican, the deeper 5 m sites support slow growing, low-light tolerant corals (*Goniopora*, *Alveopora* spp. and family: Merulinidae) in contrast to fast-growing Acroporidae (*Acropora, Montipora* spp.) that were present in the shallows before being killed and replaced by macroalgae (*Sargassum* spp) following exposure to low salinity flood plumes in 2011 (Figure A6). Closer to the Fitzroy River, Peak is characterised by low cover of corals, low density of juvenile corals and high cover of macroalgae (Figure A6). A lack of carbonate reef development, and limited recovery of coral communities over the period of monitoring suggests that the environmental conditions at Peak are marginal for most corals. On this basis, sampling of this reef was discontinued in 2020. In the less turbid waters surrounding the remaining reefs coral communities are dominated by Acroporidae (Figure A6, Table A9).

Between 2006 and 2015 reefs within this region were exposed to a series of acute disturbances including cyclones and storms, high water temperature leading to coral bleaching, and flooding of the Fitzroy River (Figure 31c-e). These disturbances resulted in a clear reduction in coral cover (Table 15, Figure 32a). The disproportionate loss of *Acropora* (Figure A6) resulted in a reduction in the Composition indicator scores (Table 15, Figure 30). Compounding the impact of the acute disturbances were declines in the Cover change scores between 2007 and 2014 (Table 15). These declines coincided with a period of relatively high discharge from the Fitzroy River and high levels of disease (Figure A7), and are responsible for the “chronic” disturbances in Figure 31e.

In 2022 scores for the Macroalgae indicator remain in the very poor range (Figure 30). The only site that does not have a high proportion of macroalgae is the site at 2 m depth at Barren Island (Figure 32b, Table A7). Initial increases in macroalgae cover occurred as brown algae of the genus *Lobophora* rapidly occupied space made available following the death of corals in 2006 (Figure 32c, Diaz-Pulido *et al.* 2009). Although (Diaz-Pulido *et al.* 2009) report this bloom of *Lobophora* as transient, our ongoing monitoring demonstrates a persistent shift toward high levels of macroalgae on these reefs punctuated by short term reprieves such as in 2011 when flooding killed corals and macroalgae at 2 metres on reefs to the south of Great Keppel Island (Figure 30, Figure 32b). By 2012, macroalgae had recolonised these reefs and cover has since remained high. Of concern is that in recent years cover of macroalgae in the genus *Sargassum* has increased at Middle Island, and Keppels South (Table A11).

Prior to the commencement of the MMP, Queensland Parks and Wildlife Service monitoring of reefs in Keppel Bay from 1993–2003 recorded substantial loss, and subsequent recovery, of coral cover following thermal bleaching events in 1998 and 2002 (Table A6). 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. Elevated water temperatures (2016, 2017; Figure 31c) and exceedance of median discharge levels from the local catchment in 2017 (3.0 times the median flow, Table A5) did not result in substantial loss of coral cover, but are likely causes of observed low scores for the Cover change indicator over this period and coral cover losses attributed to ‘chronic stress’ (Figure 30, Figure 31e). High water temperatures in 2020 resulted in extensive bleaching and observed mortality of corals at Barren Island. In 2022 coral cover resumed a slow increase after a lack of improvement between 2020 and 2021 at the regional scale (Figure 32a, Figure A6).

Water quality monitoring *(in-situ*) was reinstated in 2021 after being discontinued in 2015. In 2015 the water quality index was improving and scored as ‘good’ with similar condition also observed in 2021 and 2022 (Figure A15). Modelling of total suspended solids and dissolved inorganic nitrogen indicate substantially lower concentrations in the region between 2014-2016 and 2018-2021 compared to those associated with higher discharge years (Moran *et al.* 2022).

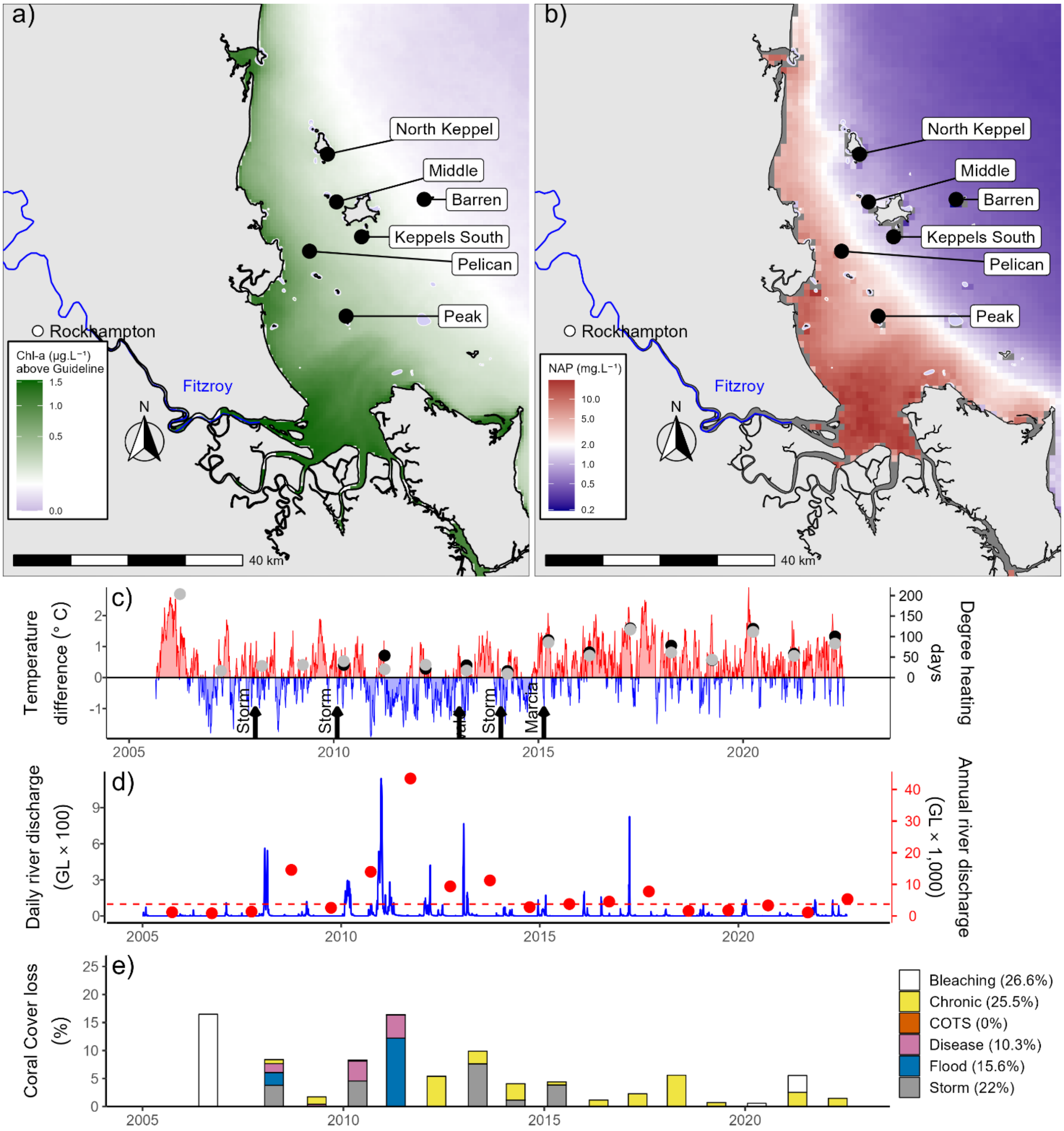

Figure 31 Fitzroy region environmental pressures. Maps show location of monitoring sites along with a) mean chlorophyll *a* exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are mean levels over the period 2003–2018 (Chl *a*) and 2003–2018 (NAP). c) Seasonally adjusted temperature deviation, timing of cyclones and storms indicated by black arrows, accumulated degree heating days over the summer period (1st of December – 31st 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.

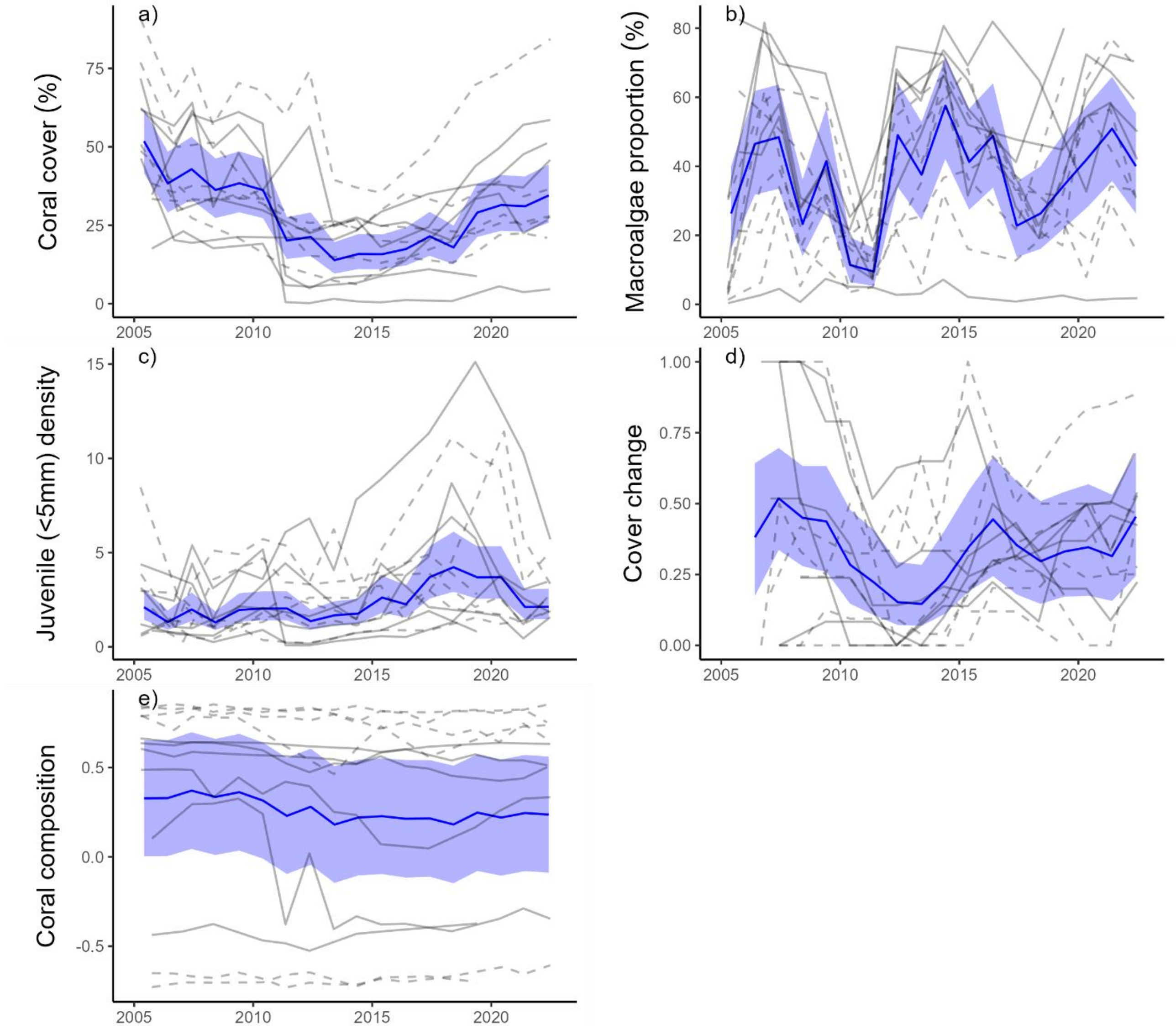

Figure 32 Fitzroy region indicator trends. a– e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.7 Response of coral communities to environmental conditions

4.7.1 Location along water quality gradients

The Reef-wide Coral Index scores in 2022 did not show consistent trends along wet-season water quality gradients estimated from sentinel data between 2019 and 2022 (Table A8).

Of the individual indicators:

- Scores for Coral cover were negatively related to increasing concentration of both Chl *a* and TSS at both at 2 m and 5 m depths. These relationships were strongest at 2 m depth in the Wet Tropics region and at 5 m depth in the Fitzroy Region (Table 16, Figure 33).
- Reef-wide scores for the Macroalgae indicator were negatively related to Chl *a* concentration at 2 m depth only (Table 16). These relationships were not statistically significant in any single region, although showed the same tendency as the reef-wide relationships in each region (Table 16, Figure 34a).

Neither the juvenile coral, Composition, or Cover change indicator scores in 2022 varied predictably along water quality gradients.

Table 16 Relationship between Coral Index and individual 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 Coral Index or indicator, and depth, for which Reef-wide relationships between scores in 2022 and water quality proxies (mean wet season Chlorophyll *a* and Total Suspended Solids) 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	-7.0	-0.7	-12.8	-1.9	-6.2	3.3	-6.2	18.7	-13.3	0.1
	5	-5.5	-0.2	-6.9	1.8	-4.2	3.8	-7.5	6.0	-17.3	-4.3
Macroalgae score	2	-8.9	-0.6	-11.5	4.5	-13.0	1.0	-21.8	12.7	-15.0	1.4
Total Suspended Solids concentration											
Coral cover score	2	-0.7	-0.1	-1.4	-0.2	-0.6	0.4	-1.0	2.3	-1.2	0.0
	5	-0.6	-0.0	-0.8	0.2	-0.4	0.4	-1.0	0.9	-1.4	-0.3

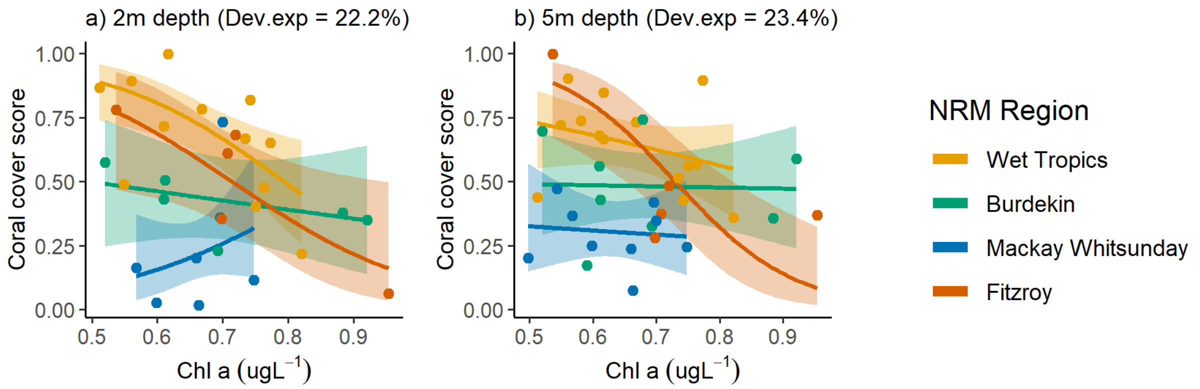

Figure 33 Coral cover indicator score relationships to Chl *a* concentration. Plots present predicted relationship within each region. Confidence intervals in predicted slopes are provided in Table 16.

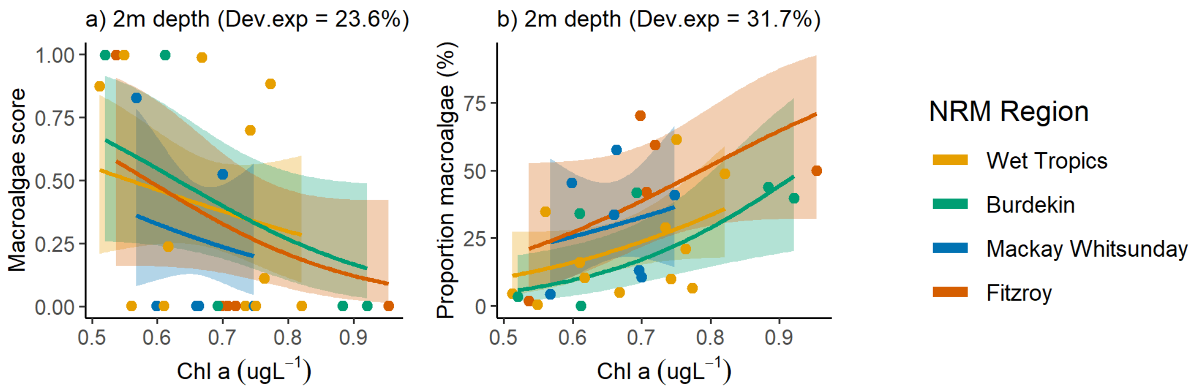

Figure 34 Relationship between Chl a concentration and macroalgae indicator scores and proportions. Plots present predicted relationship within each region. Confidence intervals in predicted slopes are provided in (Table 16).

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. Reef-wide, macroalgae cover 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, but similar tendencies are observed in all regions (Table 17, Figure 34b).

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 35) 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 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	2.2	8.5	-1.0	10.1	1.4	12.1	-9.1	16.1	-1.0	11.8
Community composition	2	-1.6	-0.1	-2.4	0.1	-1.6	0.7	-0.9	4.5	-3.3	-0.4
	5	-3.0	-1.3	-3.5	-0.8	-3.3	-0.8	-2.1	2.0	-5.3	-2.1
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	-1.6	-0.1	-0.3	-0.0	-0.1	0.1	-0.1	0.6	-0.3	-0.1
	5	-3.0	-1.3	-0.4	-0.1	-0.3	-0.1	-0.3	0.3	-0.5	-0.2

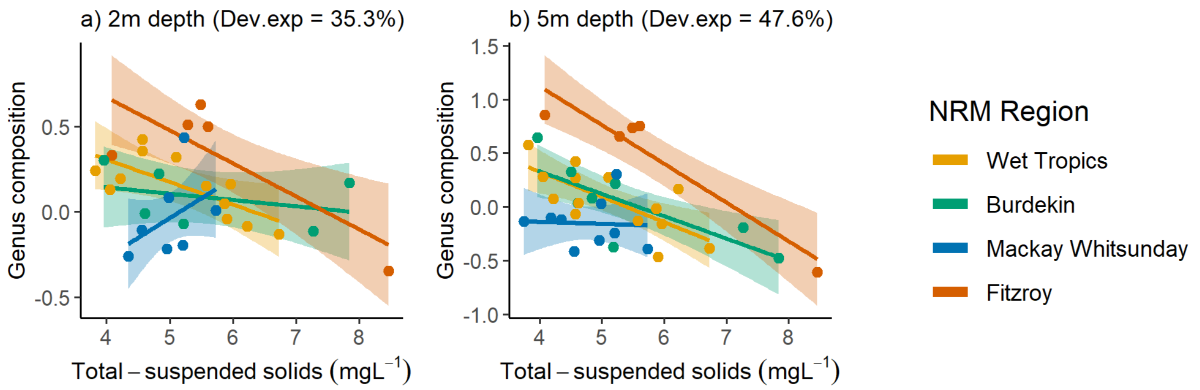

Figure 35 Relationship between coral community composition and water quality. Plots present predicted relationship within each region. Confidence intervals in predicted slopes are provided in Table 16. Colour coding for regional trends are consistent with those in above figures.

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 will result in the underestimation of the response. Relationships between loads of particulate and dissolved nitrogen, total suspended solids 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.

The concentration of Chl a and TSS as estimated by frequency of exposure to different colour classes of water in satellite imagery (Moran *et al.* 2022) was also negatively related to changes in the Coral Index (Table 18). In the Fitzroy and Mackay Whitsunday region these summaries of observed water quality explained a similar proportion of the variability in Coral Index scores as end of catchment load and discharge estimates. In the Mackay Whitsunday region, the relationship to changes in the index were more monotonically declining than those associated with riverine inputs. The relationships in the Wet Tropics was weak and there was no relationship observed in the Burdekin region.

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. A (*) marks relationships that where 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)	TSS (JCU extrapolated load))	Chl a (from water types)	TSS (from water types)
Wet Tropics	13.5%	13.3%	11.8%	13.3%	3.4%	2.8%
Burdekin	7.6%	5.1%	7.4%*		6.8%*	
Mackay-Whitsunday	7.0%*	7.3%*	7.9%*	7.9%*	7.9%	4.2%
Fitzroy	26.8%	24.4%	22.7%	24.5%	23.3%	22.3%

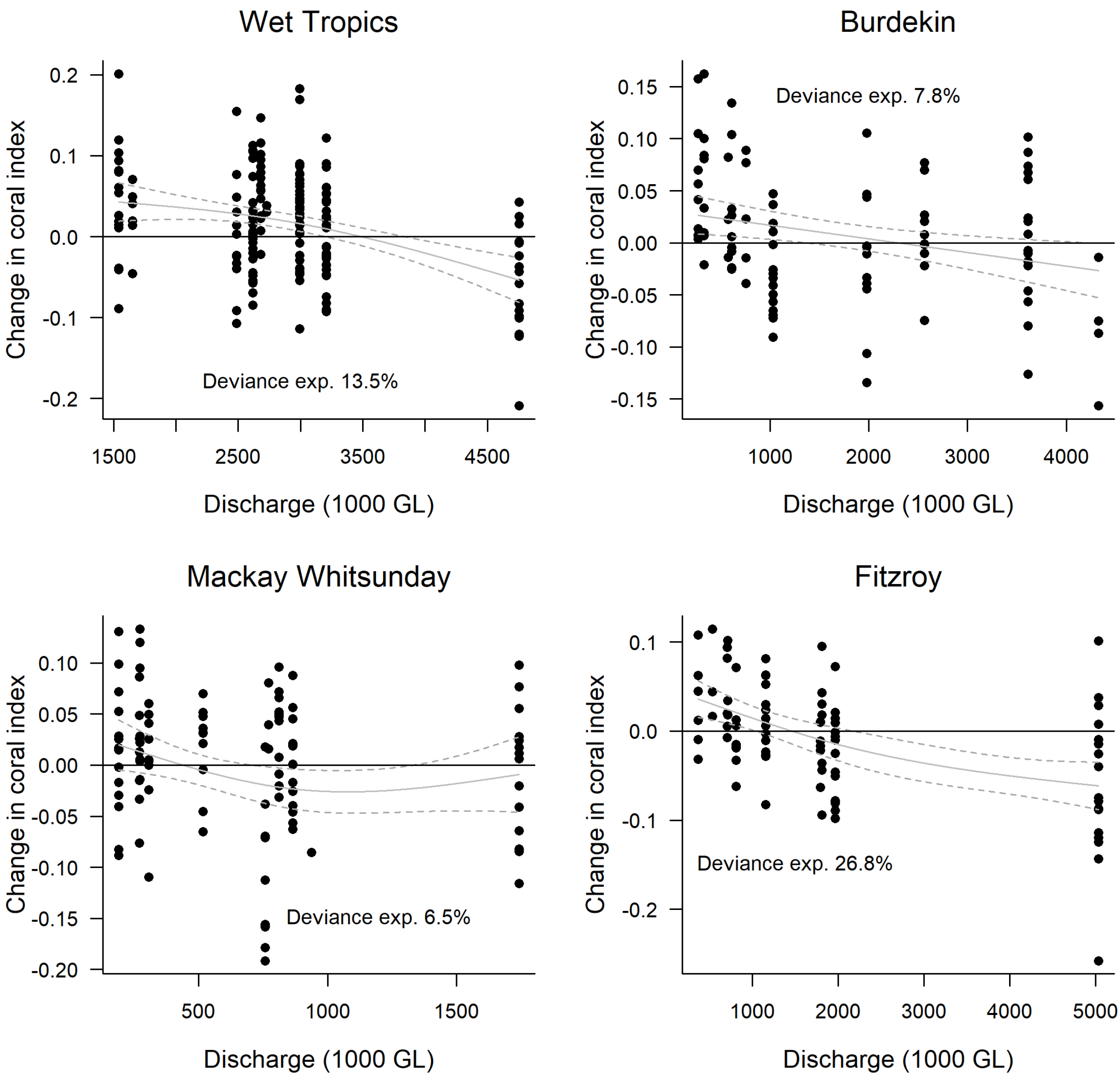

Figure 36 Relationship between the Coral Index and freshwater discharge from local catchments. Plotted points represent observed change in the Coral Index score at each reef and depth over a two-year period. Observations following years for which acute disturbances impacted communities in the period 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 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* (DPSIR) framework (Maxim *et al.* 2009, Rehr *et al.* 2012) allows identification of some of the key drivers and pressures influencing coral community condition and the potential imbalance in the disturbance recovery cycle. These include:

- Social and economic development are two of the *drivers* of human activities; from local, within catchment, through to global scales.
- Human activities result in local scale *pressures* on downstream 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 that are beyond the realm of management under the Reef 2050 Water Quality Improvement Plan, and those related more tangibly to water quality, that may be locally manageable. A primary focus of this component of the MMP is assessing the role of water quality in this balance.
- These *pressures* change the *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 management actions (*response)* that alleviate *impacts*.

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 (46%) of all coral cover losses on inshore reefs since 2005. Unsurprisingly it has been 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) that have caused the greatest losses.

Following high water temperatures over the 2017, 2020 and, to a lesser degree, 2022 summers, the relative impact of coral bleaching has increased to account for 15% of coral cover loss. 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 stressed corals. Such lagged effects of disturbances as well as the potential that the impact of acute events may be tempered 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 fifteen 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 reduced severity of bleaching with depth in models parameterised from bleaching surveys during the 2016 and 2017 with the Wet and Dry tropics regions of the Reef (Cantin *et al.* 2021) and in numerous previous studies (e.g., Muir et al. 2017). 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 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 suffered lower loss of coral cover because of 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, measured as DHW, 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 has 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, 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 two metre 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. Such exposure to low salinity waters add to the list of disturbances faced by coral communities near river outfalls and will likely limit the development of coral reefs where exposure is frequent.

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.

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 (Uthicke *et al.* 2010, Thompson *et al.* 2010, Browne *et al*. 2010).

Such gradients are a natural part of the Reef ecosystem, albeit the contribution of run-off-derived pollutants has increased since European development of the Reef catchment (Belperio & Searle 1988, Waters *et al.* 2014). The premise underpinning the Reef 2050 Water Quality Improvement Plan (WQIP) is that anthropogenic contaminant loads delivered by rivers create conditions that suppress the health or resilience of the Reef's ecosystems. The core focus of the water quality monitoring component of the MMP (see separate report by Moran *et al*. 2023) is the quantification of the compounding influence of run-off on the naturally occurring gradients, and any subsequent improvement due to the activities under the Reef 2050 WQIP.

For corals, the pressures relating to land management practices influence the 'state' of marine water quality. The MMP river plume monitoring and exposure mapping (see Moran *et al.* 2023) 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. However, variability in nutrient loads delivered to the Reef has not been closely linked to observed variability in ambient marine water quality in most regions. This is not unexpected given the complexity of nutrient cycling that occurs in marine waters, dilution of plumes, and the necessarily sparse sampling regime of the long-term water quality monitoring program.

It is evident from the MMP marine water quality time-series that there were gradual declines in water quality over the period through to 2012 that saw high rainfall deliver relatively high loads of sediment and nutrients to the Reef. Water quality then continued to decline but has now stabilised or improved in recent years (Moran *et al.* 2023). 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). Concentrations for both these water quality parameters remain high in 2022. 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.

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 (Kline *et al.* 2006, Kuntz *et al.* 2005, Vega Thurber *et al.* 2009). An emerging concept is that dissolved inorganic nitrogen (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 required for recovery (Morris *et al*. 2019). A recently suggested mechanisms 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 enhances the availability of N to algal symbionts (Rädecker *et al.* 2015, Pogoreutz *et al*. 2017). In general, the water column NOx concentrations observed at MMP sites are low in comparison to P concentration and so unlikely to directly cause imbalance in N:P ratio (*pers. comm.* Renee Gruber). The role of increased DOC however, remains unknown.

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, Thompson *et al.* 2020, Fabricius *et al.* 2013, Fabricius *et al.* 2014, Fabricius *et al.* 2016). 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 relatively low inputs from most catchments in 2020 and 2021, and no major cyclones contributing to resuspension total suspended solids concentrations in 2021, have tended to decline in inshore waters (Moran *et al.* 2022).

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 2022 reflects both the strength of this recovery but also the influence of more recent disturbance events.

In 2022:

- 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 a series of losses caused by coral disease, crown-of-thorns starfish, floodwaters, and storms. Low scores for the Juvenile coral indicator suggest recruitment processes are limiting recovery, especially where there is a high cover of macroalgae.
- 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 as tended to recover well when numbers of these starfish are low, their feeding over recent years will have reduced the amount of cover currently observed. Low densities of juvenile corals also supressed scores in the region.
- Herbert Tully sub-region score has declined marginally to ‘moderate’ condition after holding ‘good’ condition from 2019-21. Most influential on this decline has been a recent decline in Coral change and Juvenile coral indicator scores, however, both remain in classified as ‘good’ in 2022.
- 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 limit the recovery of coral communities.
- The score for 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 early recovery of coral communities since being severely impacted by cyclone Debbie in 2017.
- Slow recovery of reefs in the Fitzroy region continues in 2022. The Coral Index score remains ‘poor’ even though there were significant increases to Coral cover and Cover change scores from 2020, with both in the ‘moderate’ range. Persistently high cover of macroalgae and low densities of juvenile corals at most reefs continue to limit coral community recovery.

Variability in the condition of coral communities along water quality gradients highlight the pressure that poor water quality imposes on coral communities. Reef-wide Coral cover scores decline with increasing Chl *a* concentration in surrounding waters. This relationship is most evident at 2 m depths, where a statistically significant relationship was observed in the Wet Tropics region and a similar tendency clear in the Fitzroy region. At 5 m depth the reef-wide relationship was less distinct, and statistically evident in the Fitzroy region only, although again, the Wet Tropics region showed a similar tendency. It should be acknowledged that within-region statistical estimates will have low power due to the small number of reefs sampled and be highly sensitive to the state of individual reefs. In the Fitzroy region the observed relationship is being driven by the very low coral cover at Pelican Island compared to other reefs in the region, while at 5 m depths it is the extremely high coral cover at Barren Island compared to the other reefs that drives the relationship. 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 MMP 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 (<1km) hydrodynamic processes that influence the conditions experienced by the corals, will not be resolved by satellite derived estimates of water quality. Further limiting the fidelity of satellite derived estimates of water quality currently is the relatively limited dataset that has been used to estimate water quality concentrations with each water type. It is planned for 2023 to include all available MMP niskin samples that can be matched to the observed water types to improve the estimation of mean concentrations of water quality parameters within water types.

Reef-wide scores for the Macroalgae indicator at 2 m depth were negatively related to Chl *a* concentration. While this tendency was common across regions, no individual regions returned slope estimates with error fully excluding zero. In part, the sensitivity of this relationship is limited by reefs returning scores of zero across much of the Chl *a* gradient.

Limited relationship between the Macroalgae scores and lack of relationship between Composition and Cover change 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).

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. Similarly, coral community composition changes along gradients of these same water quality variables. The Mackay-Whitsunday region remains an exception for these relationships.

Further, the single dimensional summaries of community composition reported were derived from the product of eigenvalues for each coral genus along water-quality gradients and the relative cover of those genera. Importantly, fast-growing *Acropora* score positively on this scale compared to the slower growing species of most other genera. The result is that while the Cover change score is standardised for community composition the actual rate of recovery of communities will be higher at reefs with a high proportion of *Acropora*. In short, the negative relationships between coral community composition and water quality variables are indicative of reduced recovery rates of coral cover as water quality declines.

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.

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 were declines in the Coral Index that occurred in

2006 when discharge was low. While the 2006 declines remain unexplained, they are best explained by 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, Kline *et al*. 2006, Kuntz *et al*. 2005, 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 and 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 environmental extremes but also maintain a competitive ability under ambient conditions. Although low scores for the Coral cover indicator in the Mackay-Whitsunday compared to the Wet Tropics and Burdekin regions are clearly influenced by the recent impact of cyclone Debbie, low cover, as a response to water quality pressures, can also be inferred from our analyses. In 2022, Coral cover was generally higher at reefs with low concentrations of TSS and Chl *a*. Poor water quality does not, however, 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 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. Conversely, high coral cover on reefs toward the better end of the water quality gradient have typically demonstrated strong recovery from observed disturbances.

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 without acute pressures. 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, and more recently a return to annual sampling of all reefs, to ensure visitation of reefs following disturbances, and so 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 predominantly comprised of slow growing species, as in these situations expected rates of increase are low relative to the precision of the sampling.

In 2022 all reefs were surveyed with only 2 reefs classified as having been impacted by an acute disturbance. Magnetic (5 m depth) was categorised as having been impacted by an acute event over the preceding summer. Coral cover at this reef declined with ~13% of corals still being bleached at the time of surveys and the focus of the 2021-22 marine heatwave on the inshore areas of Burdekin region was sufficient to categorise this loss of cover as having been caused by coral bleaching. While this categorisation appropriately excluded the changes between 2021 and 2022 at this location from informing the Cover change score, it is likely that changes at some other reefs in the region will have been suppressed by the heat wave conditions. Despite this probable limitation, the Coral change indicator improved slightly within the 'moderate' range demonstrating the ongoing recovery potential exhibited by these of coral communities.

Similarly, increases in coral cover at most reefs in the Johnstone Russell-Mulgrave subregion precluded categorising an impact. However, outbreak densities of crown-of-thorns starfish at High Island and those culled from Fitzroy Island and the Frankland Group are likely to have reduced the coral cover observed in 2022. Of these, hard coral cover only showed a clear decline at Fitzroy West LTMP and the reefs classified as having been impacted by crown-of-thorns starfish for the 2022 survey. That the scores for the Cover change indicator remained in the 'good' range despite likely localised losses do crown-of-thorns a positive sign for the corals in this region.

Poor scores for the Cover change indicator were returned for both the Mackay-Whitsunday and Fitzroy regions. Of concern is that in the Mackay-Whitsunday region many of the coral communities are dominated by slow growing species, especially at five metre depths. Poor Cover change scores for communities dominated by slow growing corals indicates that very slow recovery is occurring. In contrast, most reefs in the Fitzroy region include a high proportion fast growing *Acropora* meaning that the similarly low scores indicate faster recovery of coral cover than in the Mackay Whitsunday region.

Over the period of the MMP, temporal trends in the Cover change scores, as for the Coral Index, can be generalised as having declined to low points between 2012 and 2014 and subsequently improved. Exceptions were the Herbert Tully sub-region, where both the Coral Index and the Cover change indicator scores improved between 2008 and 2011, and the Mackay-Whitsunday region, where the Cover change score was consistently low prior to declining further since 2017. 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.

The conclusion is that environmental conditions associated with the increased loads of sediments and nutrients delivered by these floods were sufficiently stressful to limit the recovery of coral cover, and/or induce disease in susceptible species. This is consistent with previous observations linking nutrients and organic matter availability to higher incidence and severity of coral disease (Bruno *et al.* 2003, Haapkylä *et al*. 2011, Weber *et al.* 2012, Vega Thurber *et al.* 2013).

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.

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 2022, the Composition indicator score improved slightly but remained 'poor' however, there was substantial variability among the subregions. The 'moderate' (Johnstone Russell-Mulgrave) and 'good' (Tully Herbert and Burdekin) scores contrast the 'poor' scores in the Barron Daintree, 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), 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 are largely due to loss of *Acropora* cover following cyclone Debbie. Although early signs of *Acropora* recovery were observed in 2022 with low but increasing numbers of juveniles observed on some reefs, these have yet to translate in a recovery of the proportional cover of *Acropora* within the coral communities at most reefs.

In contrast, in the Barron Daintree sub-region while coral cover has, on average, recovered, the current coral communities include proportionately fewer *Acropora* with taxa such as *Porites* increasing in relative abundance. However, here there are also early indications that *Acropora* cover is beginning to recover. Similarly, 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 struggled to recover at several locations.

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).

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 thresholds. Work is being undertaken by AIMS and the University of Queensland to develop a more broad-based indicator of community change that may offer the ability to identify a greater range of changes in coral community composition once completed.

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, Doropoulos *et al*. *2021*) providing positive feedbacks to maintain communities in a macroalgae-dominated state (Johns *et al.* 2018). Significant relationships between Chl *a* concentration, a proxy for nutrient availability, and the proportion of macroalgae at 2 m depths, link nutrient availability to reduced coral community resilience in inshore areas of the Reef.

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 allow for increased abundance of macroalgae in response to naturally occurring gradients of water quality, their cover in 2022, where long-term Chl *a* concentration exceeds guideline levels, was often at levels likely to have detrimental influences on coral recruitment and growth.

It is important to note that the relationship between high Chl *a* concentration and macroalgae cover is correlative only and does not necessarily indicate a direct cause-effect relationship between nutrient concentration and pressures imposed by macroalgae. Chl *a* is a measure of phytoplankton biomass- these microalgae are likely to respond to environmental variables in a similar way to macroalgae, yet with more productivity and growth in suitable conditions. 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) 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 midshelf 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). 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 & 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).

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.

The correlation between high prevalence of macroalgae and Chl *a* concentration implies that a reduction in the availability of nutrients has the potential to reduce the competitive interactions between macroalgae and coral and reduce the potential for long-term phase shifts.

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*. 2021). 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 2022, declines in juvenile scores were evident in Tully Herbert, Burdekin and Fitzroy regions, in each case reductions in the density of *Turbinaria* juveniles was influential. An emerging pattern is that the coral genus *Turbinaria* recruits strongly to reefs following severe disturbance by cyclones. Highest 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, again in the Herbert Tully sub-region, and at Daydream Island following cyclone Debbie in 2017. In each case this genus was not well represented in the adult coral community 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). These *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. The abundance of *Turbinaria* on some reefs has the potential to mask trends in other species that may be responding to environmental gradients. A possible solution would be the development of an indicator metric that includes consideration of community composition in addition to abundance of juvenile corals, or focused on a group, such as *Acropora*, that are more indicative potential for rapid recovery of coral communities (Fabricius *et al.* 2012).

In general, juvenile coral densities have increased at most reefs over several years following the major disturbances that led to low points in the Coral Index scores between 2012 and 2014 in each region. While these increases demonstrate an ongoing capacity for recovery of communities via the recruitment of new colonies, there are some notable exceptions that suggest a limiting influence of water quality. 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 have cover distributions skewed toward poor water quality environments (Table A8).

Monitoring of coral settlement during 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 local loss of corals. These results suggest connectivity to broodstock may also play an important role in 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 Island 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 a single pulse of recruitment observed in 2008 (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. In 2022, the Cover change indicator remains categorised as good, the Coral cover indicator increased to good, and all other indicators remained moderate. 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.

The Barron Daintree sub-region saw reductions in scores due to coral bleaching in 2017 and then the combined influence of a flood of the Daintree River and cyclone Owen prior to 2019 surveys. Bleaching in 2017 also impacted scores in Johnstone Russel-Mulgrave and Herbert Tully sub-

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

regions. Reefs in this region escaped exposure to high levels of thermal stress in 2020 and 2022, with negligible impact observed.

This is the only region in which crown-of-thorns starfish have been common on inshore reefs. In recent years, the Crown-of-thorns Starfish Control Program has helped to mitigate the impact of crown-of-thorns starfish[3] with 24,354 individuals removed from the monitoring reefs since 2013, 10,646 of these from Fitzroy Island and the Frankland Group in the two years preceding the 2022 surveys. MMP surveys have continued to note the presence of crown-of-thorns starfish across a range of size classes though numbers were well below those observed in 2020, and in 2022 outbreak levels were only observed at High Island. It appears that impact of crown-of-thorns on the coral communities in the region has been limited by the size distribution of the population with many individuals removed prior to them reaching full adult size. Smaller starfish simply eat less coral and have smaller feeding scars than larger individuals. The rapid growth of *Acropora* colonies has also been a feature of these reefs in recent years and has helped to offset cover lost to predation.

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. The only reef to be categorised as being in 'poor' condition in 2022 was Snapper North (2 m). At this location there was a marked increase in the cover of macroalgae following a series of impacts including disease, crown-of-thorns starfish and cyclone Ita that reduced hard coral cover to very low levels in 2014. The persistence of high macroalgae cover has likely contributed to ongoing low densities of juvenile corals, however even here, coral cover has begun to improve in 2022.

5.3.2 Burdekin

The Coral Index score for the Burdekin region declined from a peak reached in 2020 and remains 'moderate' in 2022. The decline from 2020 is due primarily to declines in Juvenile coral and Macroalgae scores. In contrast, the mean cover of corals across the region in 2022 reached its highest level since the inception of the MMP in 2005. While attaining the highest level of coral cover observed of 18 years of monitoring is clearly 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 can be compared in their recovery since the low point reached following the impact of both cyclone Yasi and high discharge from the catchment in 2011. Between 2011 and 2015 there was a period of recovery and hard coral cover increased rapidly. Although this occurred at other reefs throughout the region, the rate of increase at Havannah 2 m was remarkable, from 15% in 2011, to 53% by 2015. Contributing strongly to this increase were several species of branching *Acropora*. Since 2016, elevated temperatures in 2016, 2017 and 2020 led to bleaching and coincidentally high levels of disease that more than halved the cover of Acroporidae, with the cover of branching *Acropora* declining from 23.6% in 2015 to just 0.81% in 2022. This disproportionate loss of branching *Acropora* reduced the Composition score at this reef from 1 in 2020 to 0 in 2021. It appears several of the 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).

Since 2015, it was the rapid increase in *Acropora* at Palms East that disproportionately contributed to increasing Coral cover scores. Here, the density of juvenile corals had been low prior to the moderate density of *Acropora* juveniles observed in 2014 that appear to have precipitated the rapid increase in cover of this fast-growing taxa. Most other reefs have had persistently low cover of these fast-growing corals and increases in coral cover have been less evident.

Suppressing the recovery of corals in recent years have been marine heat wave conditions that caused coral bleaching in 2017, 2020 and again in 2022. These events led to a loss of coral cover

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

variability across reefs within the region, but may also have influenced the recovery of communities. Across the region there were declines in the density of juvenile corals, particularly the *Acropora* spp., at all reefs since 2020. Juvenile density has always been variable among reefs and depths, but the consistent decline in the Burdekin region since the 2020 and 2022 bleaching event raises the potential for thermal stress to be impacting early life-history phases of corals, which culminates in reduced recruitment and survivorship of juvenile corals. Studies by Ward *et al* (2002) and Johnston *et al* (2020) describe the reduction in coral fecundity due to thermal stress leading to the duration of reduced reproduction increasing over subsequent spawning seasons.

Historically, recovery from acute events in this region has been slow (Sweatman *et al*. 2007, Cheal *et al*. 2013). 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. Low settlement would logically contribute to the low density of juveniles on most reefs. Preliminary hydrodynamic modelling (Luick *et al*. 2007, Connie 2.0[4]) and differences in population genetics of corals (Mackenzie *et al*. 2004) both indicate limited connectivity between Halifax Bay and reefs further offshore, meaning local fluctuations in coral cover also are likely to directly influence larval supply.

The high prevalence of macroalgae on many reefs are also likely to be suppressing the recovery potential of coral communities. Except for Palms East, Palms West and Lady Elliot (5 m) macroalgae are common among the reef's algae, as reflected in the poor score for Macroalgae. Although there is substantial variation in the mechanism and strength of interactions between macroalgae and the early life history stages of corals, it can be generally assumed that macroalgae will negatively influence the density of juvenile corals (Viera 2020, Doropoulos *et al*. 2021). The causes for the recent fluctuations in macroalgae composition were not readily ascertained. Levels of nutrients and turbidity are declining, except for the persistently over-abundant NOx. Water quality over the short term appears to be improving possibly reflecting reduced inputs from the catchments in recent years (Moran *et al*. 2022).

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 2022 the Coral Index has increased only marginally from 2021. However, coral communities are showing some signs of recovery on the back of increasing densities of juvenile corals and slight declines in macroalgae 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.

It is the consistently low scores for the Cover change indicator that pose the most concern for the recovery of coral communities that were severely impacted by cyclone Debbie. 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-Acroporid species, the consistently low scores observed over the duration of the MMP indicate particularly limited capacity for rapid recovery of coral cover, especially at the

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

five metre depths. However, there has been a gradual increase in water clarity since 2020 (Moran *et al*. 2022) allowing more light to reach the benthos. *Acropora* are fast growing corals that are intolerant of low-light environments (Luo *et al*. 2022), therefore water clarity will have a direct impact on the growth of these colonies and in turn, the Cover change indicator.

With the severe loss of coral cover at many sites, successful recovery will rely heavily on the recruitment and survival of juvenile corals. There has been a gradual movement of sediment deposited by cyclone Debbie away from the reef, allowing space for corals to settle, but they must compete for this new space with macroalgae. Although the density of juvenile corals continues to increase in 2022, this remains low at most reefs and suggests a bottleneck for the recovery of these communities. Included amongst the increasing numbers of juveniles are *Acropora* (a fast growing and functionally important genus), the survival and growth of these colonies will be central to the recovery trajectory of the coral communities in coming years.

Initial increase in macroalgae cover following disturbances is not uncommon as algae quickly establishes on the available space following the loss of coral (McManus & Polsenberg 2004, Ceccarelli *et al.* 2020). While the Macroalgae indicator score began improving in 2021, 2022 saw a return to a lower level resembling the score in 2020. Of concern is that prior to cyclone Debbie, persistently high cover of macroalgae was only present at 2 m depths at Pine Island and Seaforth Island. Since cyclone Debbie, similar levels of macroalgae cover have developed at 2 m depth at Daydream Island and Double Cone Island and will almost certainly be putting downward pressure on the recovery of coral communities at these locations. At both reefs, the algal community is changing toward a higher proportion of brown algal species including *Sargassum* and *Lobophora* - with *Lobophora* also notable among the algal community at Dent Island in 2022. This observation 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).

Prior to cyclone Debbie, consistent improvement in the Coral Index from 2012 to 2016 was observed, reflecting both the tolerance of coral communities to their environmental settings and the ability of these reefs to recover from, at least minor, disturbance events. Prior to 2017, the only other major disturbance event to impact this region, since LTMP monitoring commenced in 1992, was cyclone Ului in 2010, which contributed to the decline in the Coral Index through to 2012. Improvement in scores post-2012 was largely due to rapid recovery of communities at two metre depths, where cover of the family Acroporidae rapidly increased. Whilst impacts of cyclone Ului were widespread, they were substantially less severe than those imposed by cyclone Debbie.

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 (Moran *et al*. 2022). 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 2022 the index was almost back to the same level as was observed in 2010, prior to cyclone Ului, and consistent with levels in which prior, albeit slow, recovery of coral communities has been observed.

Although improvement in coral communities has occurred, the ongoing low scores for the Cover change indicator, persistently high cover of macroalgae, and low densities of juvenile corals, indicate that a slow recovery of coral communities at the worst impacted reefs remains likely.

5.3.4 Fitzroy

The Coral Index increased in 2022 towards levels that were observed in 2020 but remains ‘poor’. Both the Coral cover and Cover change indicators were in the ‘moderate’ ranges as coral communities resumed a recovery trajectory, interrupted by a decline in 2021 owing to persistent impacts of thermal stress undergone in 2020, whereby corals failed to recover and subsequently died.

Severely limiting Coral Index scores is the ongoing very high cover of macroalgae at most reefs. In 2022 only a single location, Barren 2 m, had a score above zero for the Macroalgae indicator. Most concerning is Middle Island, but also Keppels South, where, when first visited in 2005, there was almost no macroalgae. Cover of macroalgae at Middle Island is now 50% and includes a high proportion of the persistent brown algae *Lobophora* and *Sargassum*.

Across the region high cover of macroalgae was precipitated by loss of coral cover following a severe bleaching event in early 2006 (Diaz-Pulido 2009, Ceccarelli *et al.* 2020). The persistence of these algae are almost certainly limiting the recovery of coral cover. Across the region the density of juvenile corals has been consistently low where macroalgae cover is high. The highest scores for Juvenile corals occurred in 2019-2020 buoyed by high densities of *Leptastrea* at Barren (2 m) and *Turbinaria* at Middle Island (5 m), neither of which propagated a meaningful increase in coral cover of these genera. In 2022, the density of juvenile corals continued to decline and slipped into the 'very poor' category for the first time since 2015.

The current condition of reefs in the region is still influenced by the cumulative impacts of thermal stress in 2006, a series of cyclones and storms, and flooding of the Fitzroy River (Jones & Berkelmans 2014) that drove Coral Index scores to a 'very poor' level in 2014. The recovery from these pressures has been suppressed by high water temperatures in 2016 and 2017 (Kennedy 2018) and again in 2020.

Flooding of the Fitzroy River impacts coral communities in two primary ways. Corals in shallow waters, particularly those to the south of Great Keppel Island, have been repeatedly exposed to low salinity plumes that kill corals (van Woesik 1991, Jones & Berkelmans 2014, Figure 31). 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 reduced fitness of corals (Cooper *et al*. 2007, Jones *et al.* 2021). Annual change in Coral Index scores in this region show negative relationship to both discharge and contaminant loads from the Fitzroy River and concentrations of Chl a and TSS estimated from satellite imagery. Of note were elevated levels of disease following major flood events supporting hypotheses that either reduced salinity (Haapkylä *et al.* 2011), increased nutrient enrichment (Vega Thurber *et al.* 2013), or were sufficiently stressful to facilitate coral disease. Reduced light quality must also be considered as high turbidity reduces the quality of light reaching the benthos (Jones *et al*. 2021). Importantly, as they are integral to reef recovery, growth of adult *Acropora* declines with reduced light quality while their juveniles are more prone to overgrowth by algae (Noonan *et al*. 2022).

Variation among reefs in the recovery of coral communities further illustrates the role of water quality in supressing coral community resilience. The most offshore site, Barren Island (5 m) is the only location in the region that reached a rating of "good" in 2022. Following thermal bleaching in 2006, recovery of coral cover was inversely related to the persistence of macroalgae. At the three *Acropora* dominated communities on reefs surrounded by waters with Chl *a* concentration consistently above the wet season guideline level (Keppels South, Middle and North Keppel) macroalgae cover (predominantly *Lobophora*) rapidly increased and persisted at high levels; at the same time the rate of change in coral cover remained low or coral cover continued to decline. In contrast, at Barren, where Chl *a* concentration is lower, the *Lobophora* bloom was less pronounced, and recovery of the coral community clearly progressed. Most telling, however, has been negligible recovery of coral cover at Pelican Island, the reef situated in the most turbid waters. Here there was no appreciable recovery of coral cover between 2011 and 2019. Encouragingly, a small recovery was observed in 2020 and although cover dipped in 2021, this was regained in 2022. However, the state of coral communities at Pelican 2 m remain in a macroalgae dominated state, in stark contrast the community dominated by *Acropora* corals prior to 2011.

Adding to limitations to coral recruitment imposed by high cover of macroalgae, it the potential limited for limited larval supply. Following loss of corals in 2011 there was a substantial decline in the settlement of coral larvae, especially at Pelican Island where the cover of potential brood-stock was effectively eradicated (Davidson *et al*. 2019).

5.4 Management response

Coral reefs in general are subjected to cumulative impacts of acute disturbances and environmental pressures (Bozec *et al*. 2022). Simplistically, 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 2022 noted a decline in densities of crown-of-thorns starfish, however moderate densities across a range of size classes were still observed at High Island 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* suggest that nutrient loads entering the reef are a primary driver contributing to persistent macroalgae cover on these reefs. It must be noted however, that the environment occupied by many macroalgae is still suitable for corals and it may be that density-dependant 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). 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 NRM regions coral communities retain the ability to recover following impacts from acute disturbances. The rate of this recovery is however influenced by to the loads of nutrients and or sediment 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 such as climate change (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 has clearly impacted the condition of inshore reefs (Lam *et al.* 2018, Ceccarelli *et al.* 2020, Thompson *et al.* 2020). Results from 2022 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 (Abelson 2020, GBRMPA 2019).

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. First, 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. Second, 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 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 baseline.
- 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-dependant feedbacks bolster 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.

- 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 was observed in 2022, a necessary precursor to the recovery of severely damaged communities. 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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8 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

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

Reef	2 m Depth		5 m Depth	
	Upper	Lower	Upper	Lower
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 Moran *et al.* (2023). 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
Wet Tropics	Daintree River	1,918,174	1.7	1.0	1.2	0.8	1.6	2.1	1.3	0.9	2.3	1.0	0.8	1.0	1.1	3.2	0.6	1.0	1.3
	Mossman River	604,711	1.6	1.0	1.1	0.9	1.4	1.7	1.3	1.0	1.6	0.8	0.9	1.0	1.2	2.0	0.5	1.0	1.3
	Barron River	622,447	1.6	0.9	3.4	1.6	1.0	4.0	1.6	0.6	1.3	0.7	0.3	0.5	1.6	2.7	0.6	1.1	1.1
	Russell – Mulgrave River	3,772,711	1.3	1.2	1.2	1.1	1.2	1.9	1.4	0.9	1.3	0.8	0.8	0.8	1.4	1.4	0.8	1.3	1.1
	Johnstone River	4,257,163	1.3	1.2	1.1	1.2	1.1	2.1	1.2	0.9	1.3	0.8	0.8	1.0	1.4	1.3	0.8	1.3	1.1
	Tully River	3,393,025	1.2	1.3	1.0	1.2	1.0	1.9	1.2	0.9	1.2	0.6	0.8	0.9	1.1	1.2	0.6	1.2	0.9
	Murray River	884,246	2.1	1.8	1.7	2.2	1.4	4.1	2.3	1.5	1.9	1.0	1.5	1.4	2.0	2.0	1.2	2.2	1.4
	Herbert River	3,556,376	1.3	1.3	1.1	3.2	1.0	3.9	1.4	1.0	1.3	0.3	0.6	0.7	1.7	1.7	0.5	1.9	0.9
Burdekin	Black River	293,525	1.0	2.2	2.5	4.6	2.2	5.5	3.2	0.7	1.9	0.1	0.5	0.3	2.0	5.0	0.5	1.5	0.9
	Haughton River	558,735	1.1	2.2	3.3	4.4	2.1	4.7	3.2	0.9	1.0	0.3	0.5	0.7	1.4	5.6	0.6	1.1	1.3
	Burdekin River	4,406,780	0.5	2.2	6.2	6.7	1.8	7.9	3.6	0.8	0.4	0.2	0.4	1.0	1.3	4.0	0.5	1.9	1.2
	Don River	167,914	3.0	5.6	11.2	9.2	4.5	15.9	5.8	4.1	2.8	2.1	1.9	5.8	2.6	6.8	3.0	3.0	2.3
	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	0.8	0.6	0.5
	Pioneer River	616,216	0.1	1.6	2.4	1.6	2.6	5.9	2.5	1.9	1.0	0.2	1.0	2.3	0.4	1.9	0.6	0.4	0.5
	Plane Creek	848,985	0.4	1.8	3.3	1.6	3.4	4.9	3.0	2.2	1.0	0.5	1.1	3.0	0.5	1.5	1.3	0.7	0.6
Fitzroy	Waterpark Creek	349,614	0.3	0.6	2.6	1.1	3.0	5.0	1.6	5.3	3.0	2.3	2.0	2.7	1.5	0.8	1.6	1.9	2.3
	Fitzroy River	2,614,356	0.3	0.4	5.2	0.8	4.9	16.0	3.4	3.6	0.7	1.1	1.5	2.6	0.4	0.6	1.1	0.2	1.7

Table A6 Disturbance records for each survey reef. Tabulated losses of coral cover are calculated using the methods described in section (2.5.5) 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%)
		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)
		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)
		Green			12 %	COTS: 1994 (21%), 1997 (55%), 2011-2013 (44%), 2014-2015 (47%)

Table A6 continued

(sub-)region	Reef	Bleaching 1998	Bleaching 2002	Bleaching 2017	Bleaching 2020	Other recorded disturbances
Herbert Tully	Barnards	0.93	0.80	17% (2 m)		cyclone Larry 2006 (95% at 2 m 87% at 5 m), cyclone Yasi 2011 (53% at 2 m, 24% at 5 m), Bleaching 2018 (10% at 5 m), Disease 2021 (18% 2 m, 9% 5 m)
	King Reef	0.93	0.85	n/a		cyclone Larry 2006 (56% at 2 m,50% at 5 m), cyclone Yasi 2011 (71% at 2 m, 37% at 5 m)
	Dunk North	0.93	0.80	18% (2 m) 16% (5 m)		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 1998	Bleaching 2002	Bleaching 2017	Bleaching 2020	Other recorded disturbances
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 1998	Bleaching 2002	Bleaching 2017	Bleaching 2020	Other recorded disturbances
Mackay-Whitsunday	Hook	0.57	1		27% (2 m) 20% (5 m	Coral Bleaching Jan 2006, probable although not observed as we did not visit region at time of event. Same for other reefs in region, cyclone Ului 2010 (31% at 2 m,17% at 5 m), cyclone Debbie 2017 (recorded in 2018) (83% at 2 m, 45% at 5 m)
	Dent	0.57 (32%)	0.95	**		Disease 2007(17% at 2 and at 5 m), cyclone Ului 2010 most likely although reef not surveyed in that year (21% at 2 m, 27% at 5 m), cyclone Debbie 2017 (48% at 2 m, 38% at 5 m), Cyclone Debbie 2017 (48% at 2 m, 38% at 5 m), Disease 2019 (44% at 2 m, 25% at 5 m), Disease 2021 (16% at 5 m)
	Seaforth	0.57	0.95	**	8% (2 m)	Flood 2009 (16% at 2 m,, 22% at 5 m), cyclone Debbie 2017 (45% at 2 m, 26% at 5 m)
	Double Cone	0.57	1	**	15% (2 m) 3% (5 m)	Flood 2009(13% at 2 m), cyclone Ului 2010 (26% at 2 m, 12% at 5 m), cyclone Debbie 2017 (97% at 2 m, 74% at 5 m)
	Daydream	0.31 (44%)	1	**	42% (2 m) 38% (5 m)	Disease 2008 (26% at 2 m, 20% at 5 m), cyclone Ului 2010 (47% at 2 m, 46% at 5 m), cyclone Debbie 2017 (98% at 2 m, 90% at 5 m)
	Shute Harbour	0.57	1	**	10% (2 m)	cyclone Ului 2010 (8% at 2 m), cyclone Debbie 2017 (48% at 2 m, 55% at 5 m)
	Pine	0.31	1	**	35% (2 m)	Flood 2009(14% at 2 and at 5 m), cyclone Ului 2010 (13% at 2 m, 10% at 5 m), Disease 2011(15% at 5 m), cyclone Debbie 2017 (74% at 2 m, 56% at 5 m), Disease 2019 (40% at 2 m, 29% at 5 m)
	Hayman					cyclone Ului 2010 (36%), cyclone Debbie 2017 (recorded 2019) (86%)
	Langford					cyclone Debbie 2017 (recorded 2019) (56%)
	Border		(11%)			cyclone Debbie 2017 (recorded 2019) (45%)

Table A6 continued

Region	Reef	Bleaching				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
	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)
	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 2022. 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	0.74	1.00	0.86	0.57	0.50	0.73
	Snapper North	2	0.40	0.07	0.00	0.74	0.00	0.24
		5	0.56	0.32	1.00	0.76	0.00	0.53
	Snapper South	2	0.65	0.44	0.88	0.74	0.00	0.54
		5	0.90	0.05	0.00	0.67	1.00	0.52
Moderate			**0.65**	**0.38**	**0.55**	**0.70**	**0.30**	**0.51**
Johnstone Russell-Mulgrave	Fitzroy East	2	0.49	0.22	1.00	0.30	0.50	0.50
		5	0.72	0.49	0.93	0.76	0.00	0.58
	Fitzroy West	2	1.00	0.53	0.24	1.00	1.00	0.75
		5	0.85	0.57	0.76	1.00	0.50	0.74
	Fitzroy West LTMP	5	0.67	0.77	1.00	0.82	0.00	0.65
	Franklands East	2	0.87	0.33	0.87	0.46	1.00	0.71
		5	0.44	0.38	0.64	0.46	1.00	0.58
	Franklands West	2	0.89	0.29	0.00	0.71	0.50	0.48
		5	0.90	0.28	0.00	0.71	1.00	0.58
	High East	2	0.72	0.20	0.00	0.39	1.00	0.46
		5	0.68	0.27	0.59	0.68	0.50	0.54
	High West	2	0.82	0.30	0.70	0.35	0.50	0.53
		5	0.43	0.26	1.00	0.64	0.00	0.47
Moderate			**0.73**	**0.38**	**0.60**	**0.64**	**0.58**	**0.58**
Herbert Tully	Barnards	2	0.78	0.40	0.99	0.75	1.00	0.78
		5	0.73	0.72	0.93	0.45	1.00	0.77
	Dunk North	2	0.67	0.75	0.00	0.66	0.50	0.52
		5	0.52	1.00	0.46	0.61	0.50	0.62
	Dunk South	2	0.48	0.32	0.11	0.71	1.00	0.52
		5	0.57	0.59	0.00	0.74	0.50	0.48
	Bedarra	2	0.22	0.59	0.00	0.40	1.00	0.44
		5	0.36	1.00	0.74	0.77	0.50	0.67
Moderate			**0.54**	**0.67**	**0.40**	**0.64**	**0.75**	**0.60**
Burdekin	Palms East	2	0.58	0.13	1.00	0.38	1.00	0.62
		5	0.70	0.25	0.86	0.53	1.00	0.67
	Palms West	2	0.50	0.40	1.00	0.95	0.00	0.57
		5	0.43	0.45	1.00	0.25	0.00	0.43
	Havannah North	5	0.17	0.88	0.00	0.5	1.00	0.51
	Havannah	2	0.43	0.21	0.00	0.37	0.00	0.20
		5	0.56	0.26	0.00	0.66	1.00	0.50
	Pandora	2	0.23	0.16	0.00	0.40	0.50	0.26
		5	0.33	0.43	0.31	0.63	1.00	0.54
	Pandora North	5	0.74	0.53	0.00	0.26	0.00	0.31
	Lady Elliot	2	0.35	0.14	0.00	0.22	1.00	0.34
		5	0.59	0.54	0.41	0.54	0.50	0.52
	Magnetic	2	0.38	0.14	0.00	0.39	0.50	0.28
		5	0.36	0.26	0.00	0.73	1.00	0.47
Moderate			**0.45**	**0.34**	**0.33**	**0.49**	**0.61**	**0.44**

Table A7 continued

Region	Reef	Depth	Coral cover	Juvenile	Macroalgae	Cover change	Composition	Coral Index
Mackay-Whitsunday	Hayman	5	0.20	1.00	0.98	0.68	0.00	0.57
	Border	5	0.47	0.54	1.00	0.00	0.50	0.50
	Hook	2	0.16	0.48	0.83	0.46	0.00	0.39
		5	0.37	0.28	0.73	0.56	0.50	0.49
	Double Cone	2	0.03	0.23	0.00	0.20	0.00	0.09
		5	0.25	0.19	0.00	0.28	0.00	0.14
	Daydream	2	0.02	0.57	0.00	0.50	0.00	0.22
		5	0.07	1.00	0.00	0.47	0.00	0.31
	Dent	2	0.36	0.19	0.00	0.12	0.00	0.13
		5	0.42	0.22	0.00	0.13	0.00	0.16
	Shute Harbour	2	0.73	0.38	0.52	0.42	1.00	0.61
		5	0.35	0.51	0.86	0.41	1.00	0.62
	Pine	2	0.12	0.35	0.00	0.17	1.00	0.33
		5	0.25	0.30	0.00	0.40	0.00	0.19
	Seaforth	2	0.20	0.25	0.00	0.08	0.50	0.21
		5	0.24	0.42	0.00	0.28	0.50	0.29
Poor			0.26	0.43	0.31	0.32	0.31	0.33
Fitzroy	Barren	2	0.78	0.48	1.00	0.42	0.00	0.54
		5	1.00	0.17	0.00	0.89	1.00	0.61
	North Keppel	2	0.61	0.04	0.00	0.22	1.00	0.37
		5	0.37	0.06	0.00	0.28	0.50	0.24
	Middle	2	0.36	0.16	0.00	0.68	0.00	0.24
		5	0.28	0.32	0.00	0.25	0.00	0.17
	Keppels South	2	0.68	0.18	0.00	0.47	0.00	0.27
		5	0.48	0.19	0.00	0.39	0.00	0.21
	Pelican	2	0.06	0.16	0.00	0.53	0.00	0.15
		5	0.37	0.29	0.00	0.54	1.00	0.44
Poor			0.50	0.20	0.10	0.47	0.35	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 concentrations over the 2019-2022 wet seasons were estimated based on the product of the proportion of time waters were classified into one of four water-types based on water colour extracted from Sentinel satellite imagery (Table 2 ,Moran *et al*. 2022) and the mean concentration of Chl *a* and TSS from MMP water samples taken within each colour class (Waterhouse *et al*. 2021).

(sub-)region	**Reef**	Wet season Chl *a* (µgL^{-1})	Wet season TSS (mgL^{-1})
Barron Daintree	Low Isles (LTMP)	0.58	4.62
	Snapper North	0.75	5.96
	Snapper South	0.77	6.22
Johnstone Russell-Mulgrave	Fitzroy East	0.55	4.05
	Franklands East	0.51	3.82
	Franklands West	0.56	4.22
	Fitzroy West (LTMP)	0.62	4.57
	High East	0.61	4.57
	High West	0.74	5.87
Herbert Tully	Barnards	0.67	5.1
	Dunk North	0.73	5.58
	Dunk South	0.76	5.9
	Bedarra	0.82	6.73
Burdekin	Palms East	0.52	3.96
	Havannah North (LTMP)	0.59	4.5
	Palms West	0.61	4.83
	Havannah	0.61	4.61
	Pandora	0.69	5.21
	Pandora North	0.68	5.18
	Magnetic	0.88	7.27
	Lady Elliot	0.92	7.84
Mackay-Whitsunday	Hayman (LTMP)	0.5	3.74
	Border (LTMP)	0.54	4.18
	Hook (LTMP)	0.57	4.35
	Double Cone	0.6	4.56
	Seaforth	0.66	4.96
	Dent	0.7	5.21
	Daydream	0.66	4.99
	Pine	0.75	5.73
	Shute Harbour	0.7	5.23
Fitzroy	Barren	0.54	4.08
	Keppels South	0.72	5.61
	North Keppel	0.71	5.49
	Middle	0.7	5.29
	Pelican	0.95	8.46

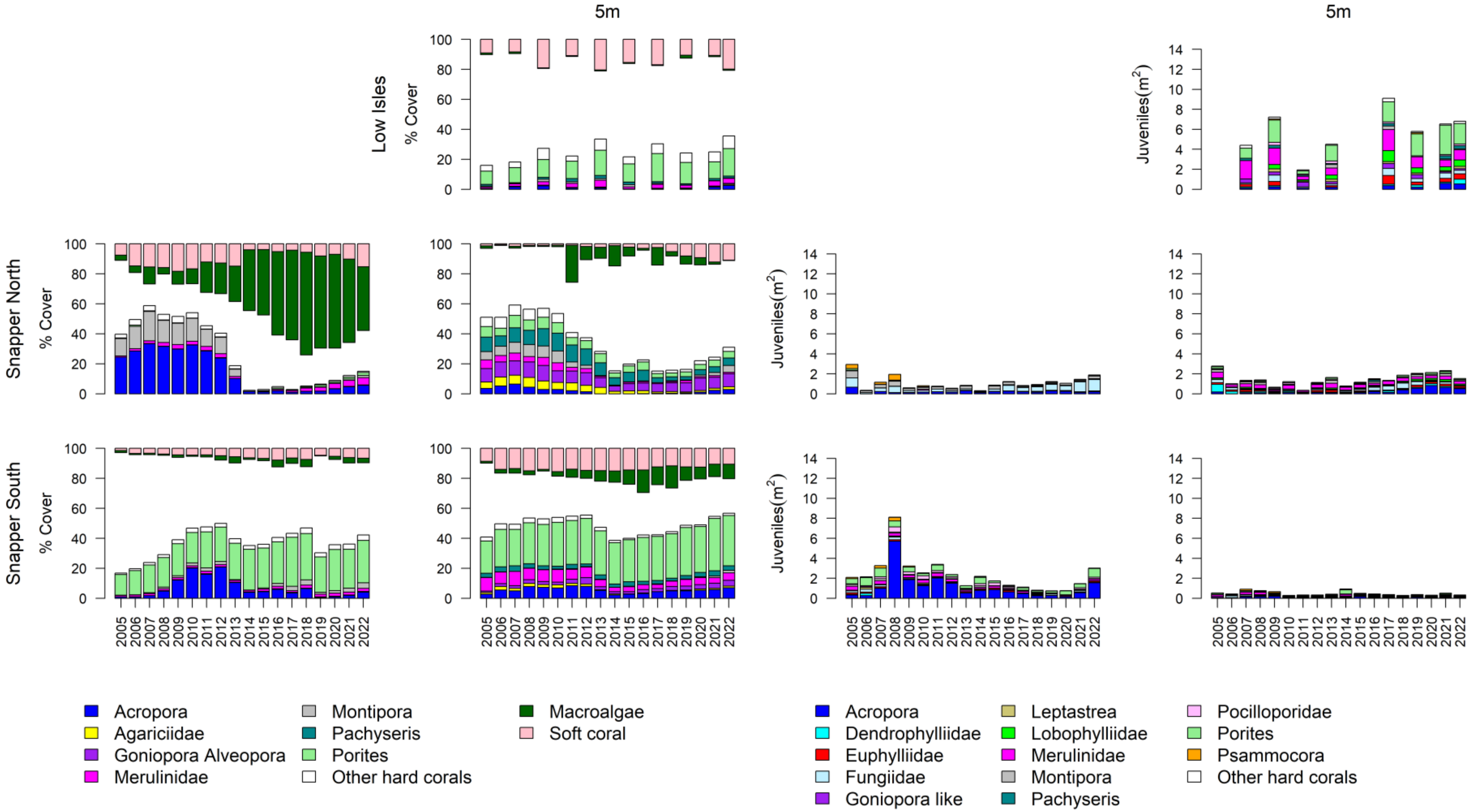

Figure A1 Barron Daintree sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends relevant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

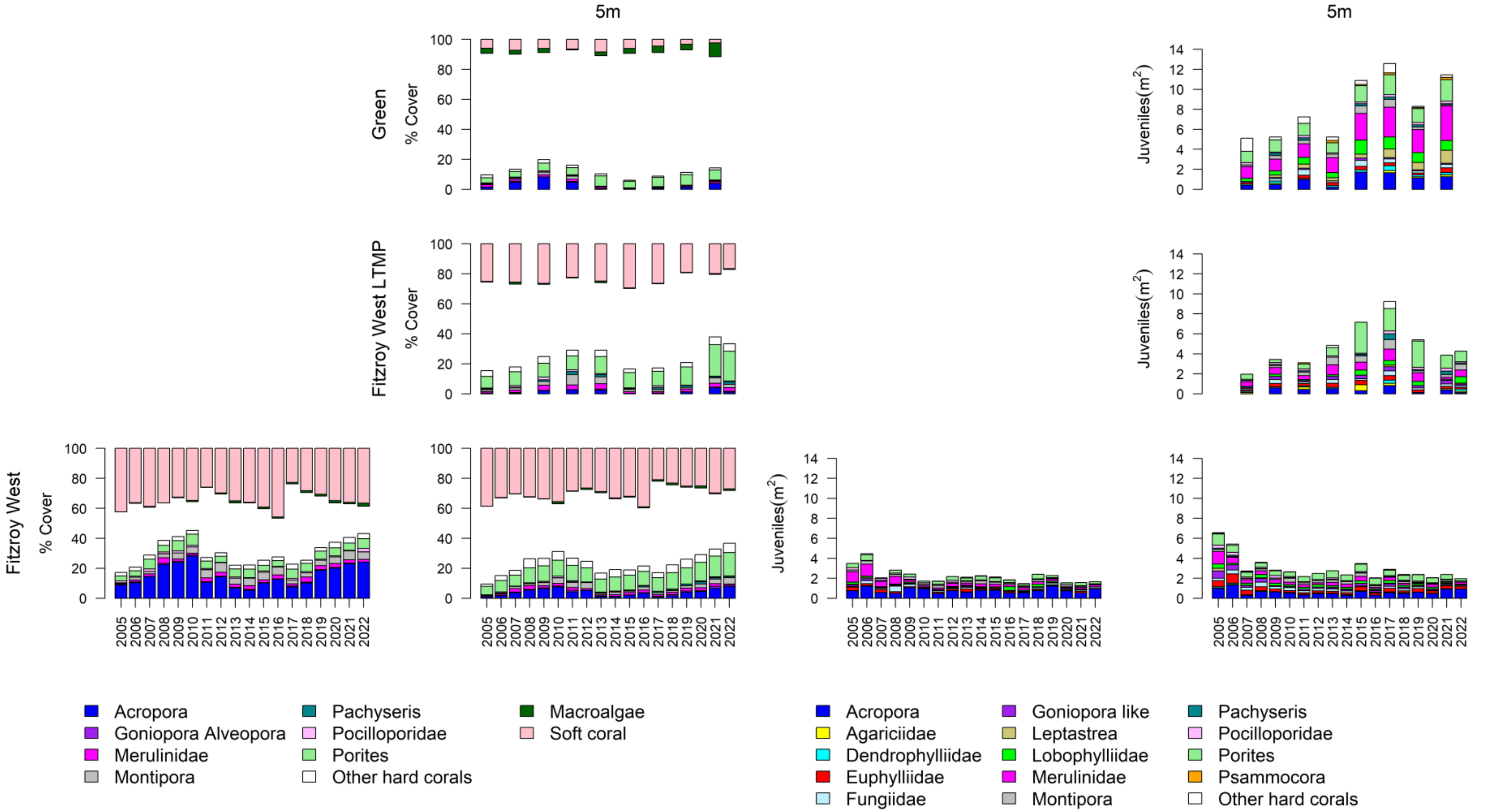

Figure A2 Johnstone Russell-Mulgrave sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate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

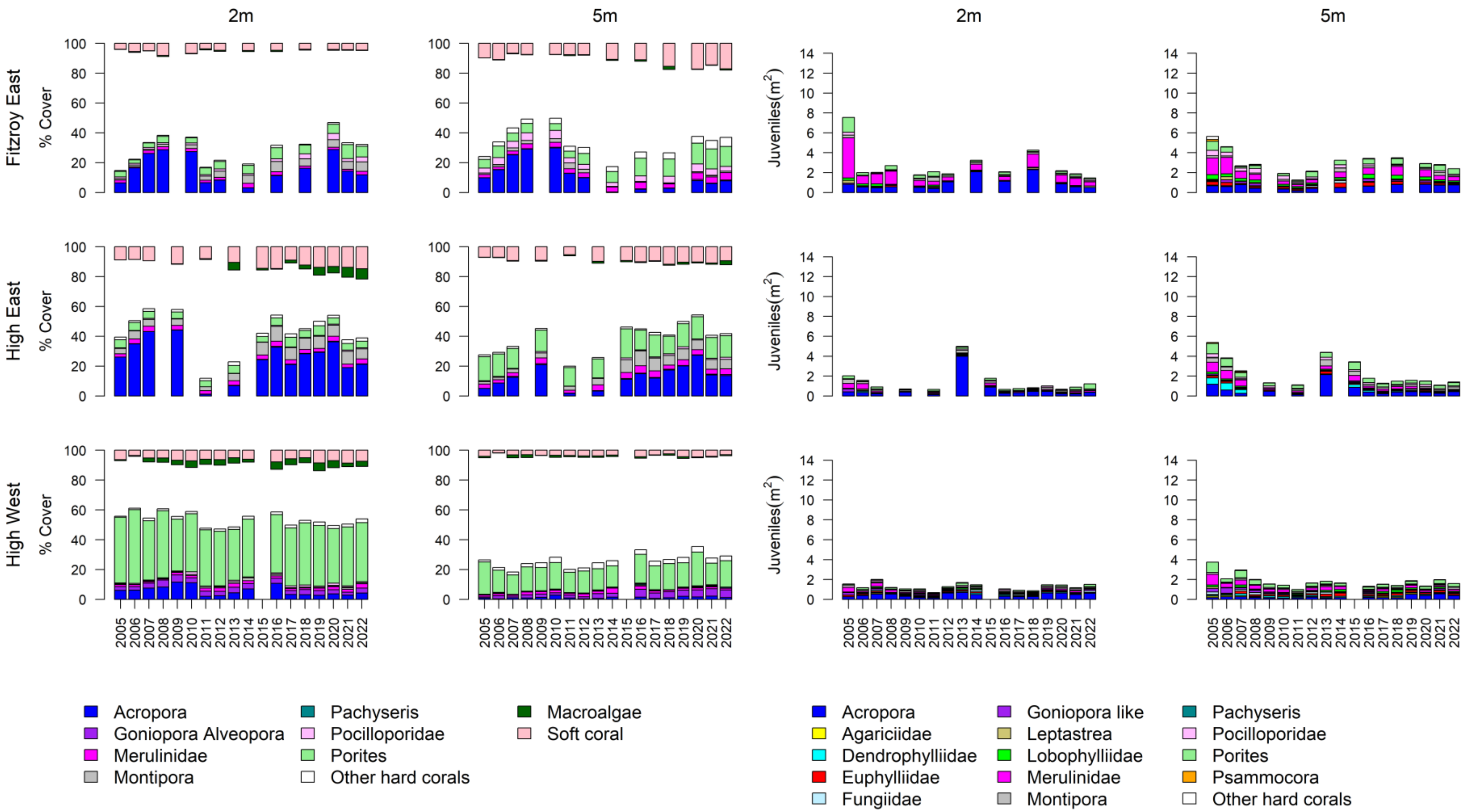

Figure A2 continued

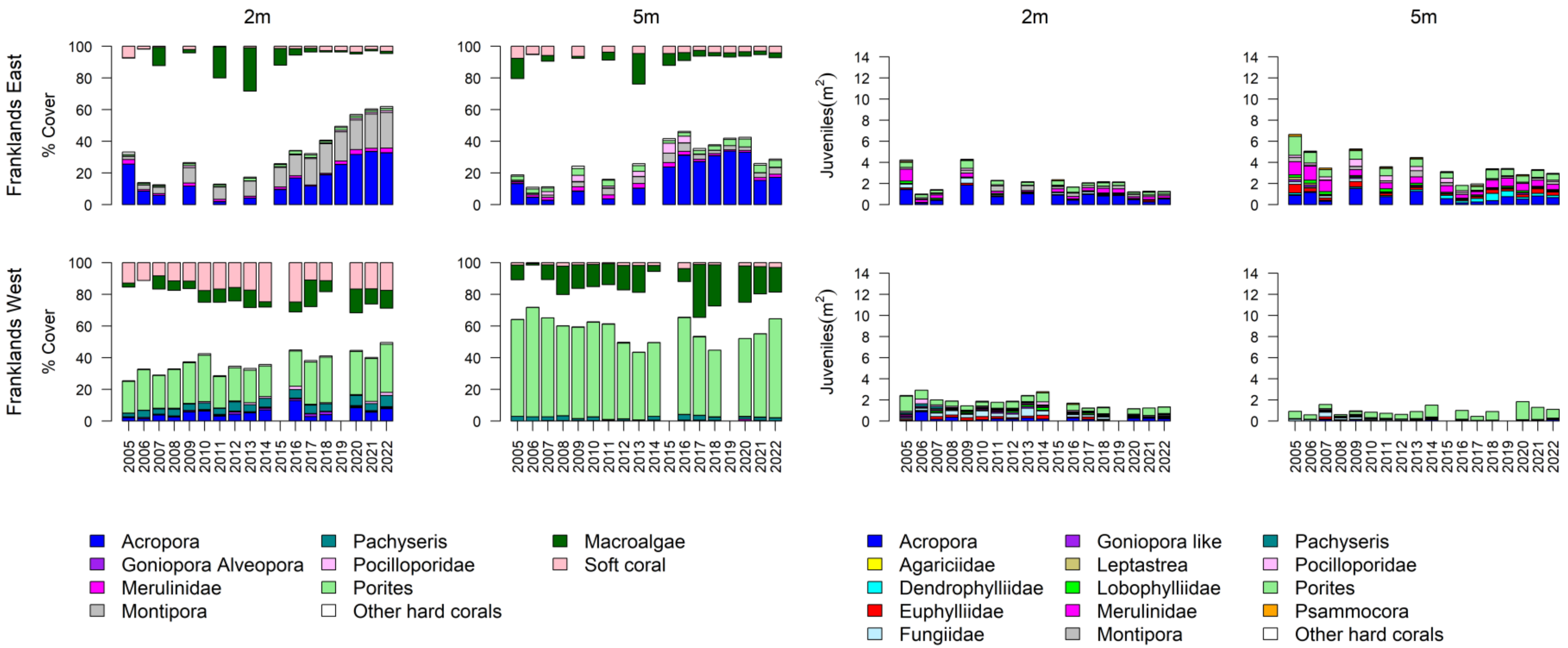

Figure A2 continued

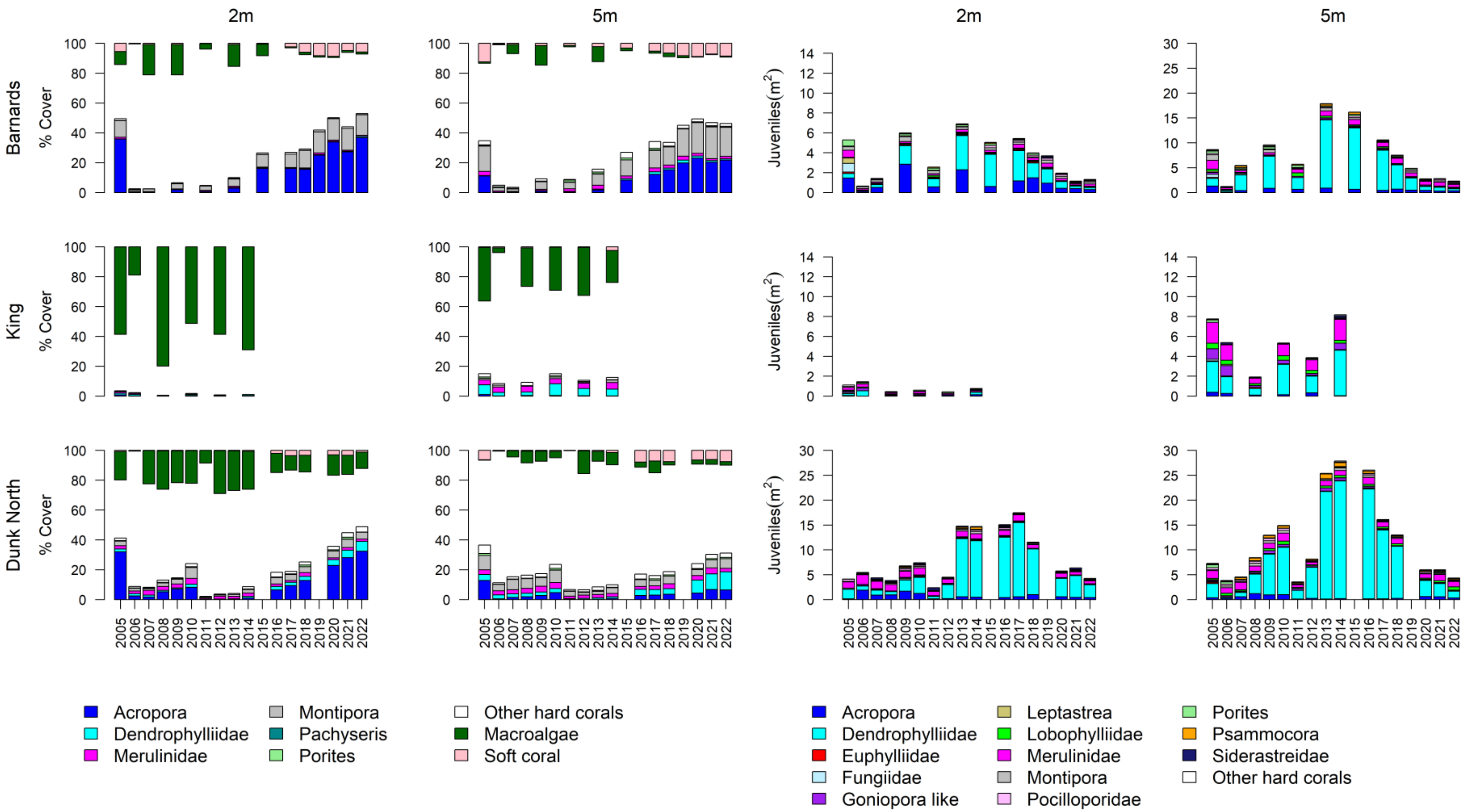

Figure A3 Herbert Tully sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends with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

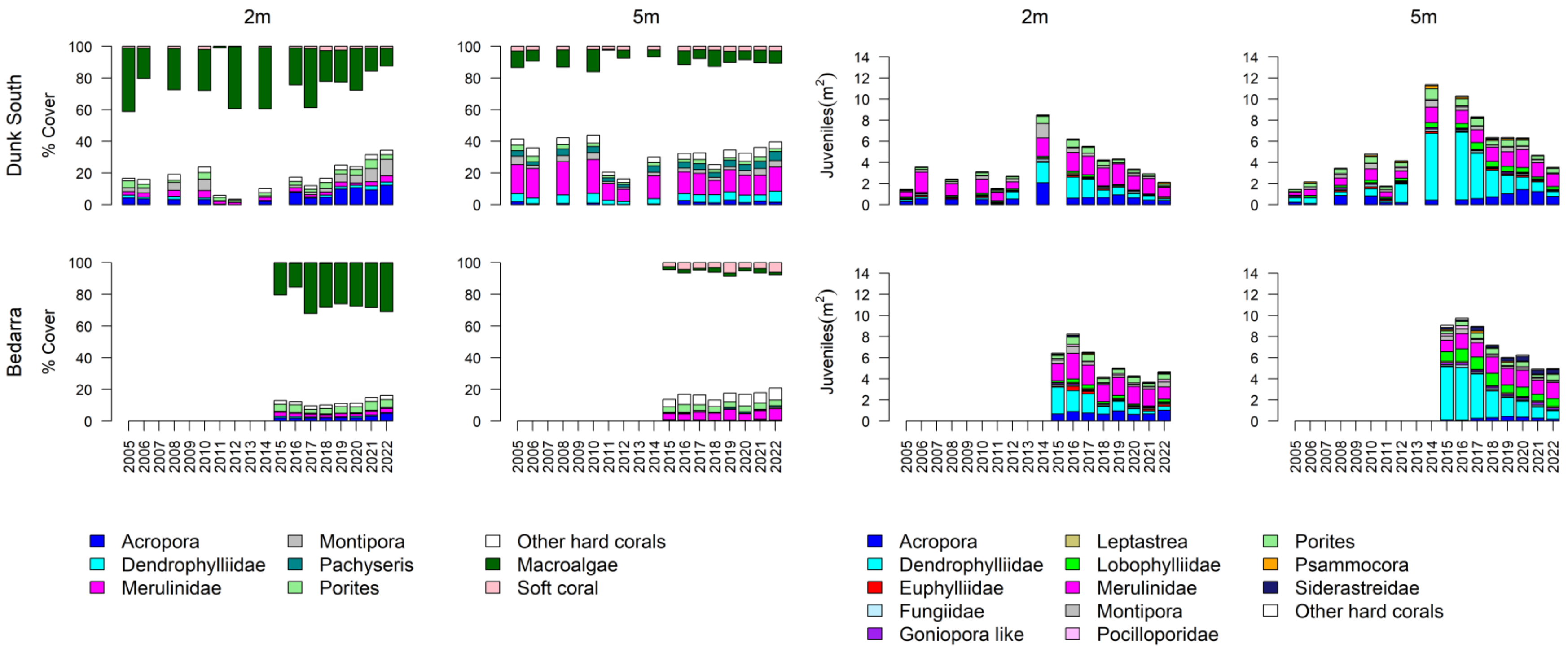

Figure A3 continued

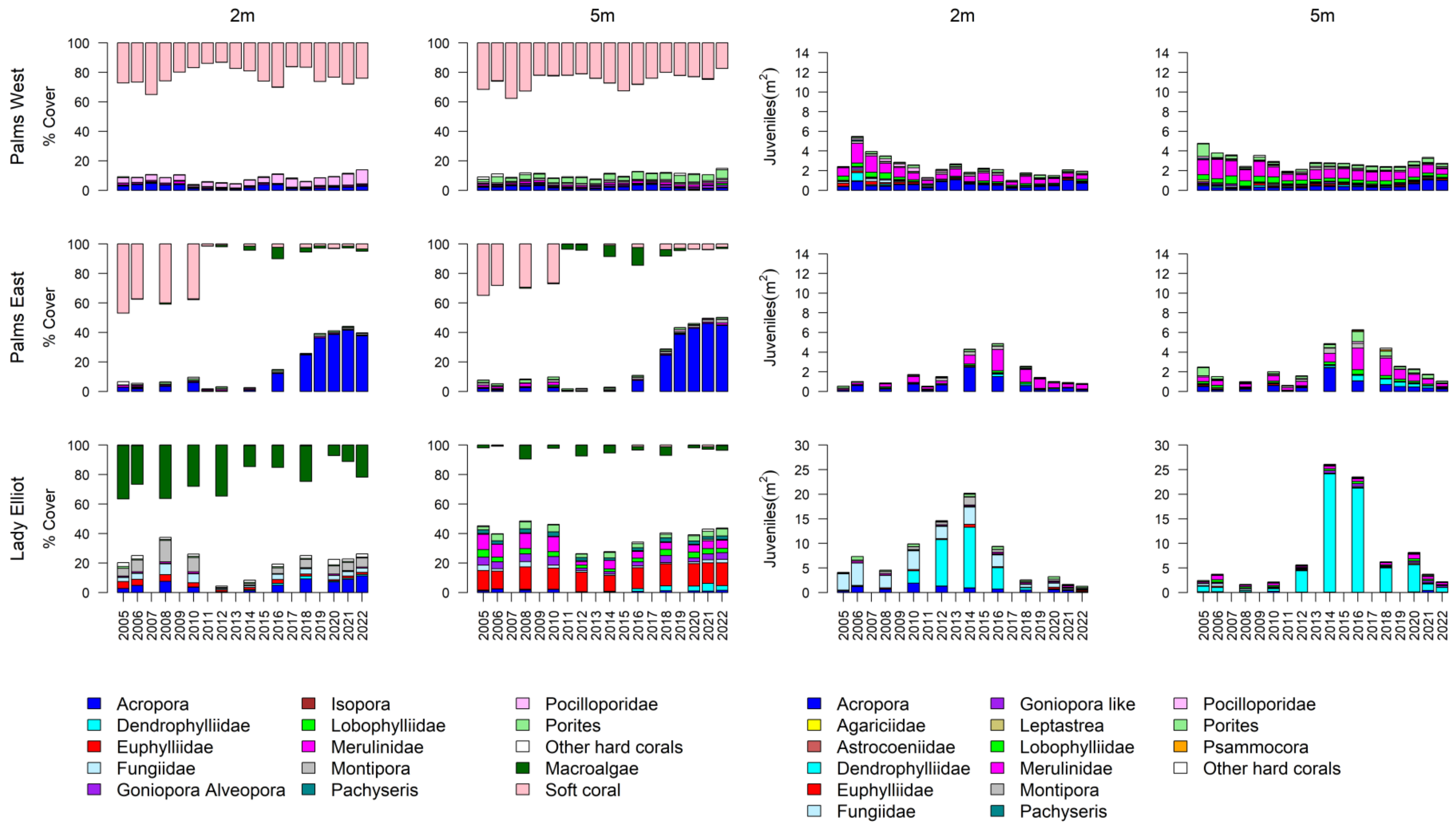

Figure A4 Burdekin region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends with relevant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

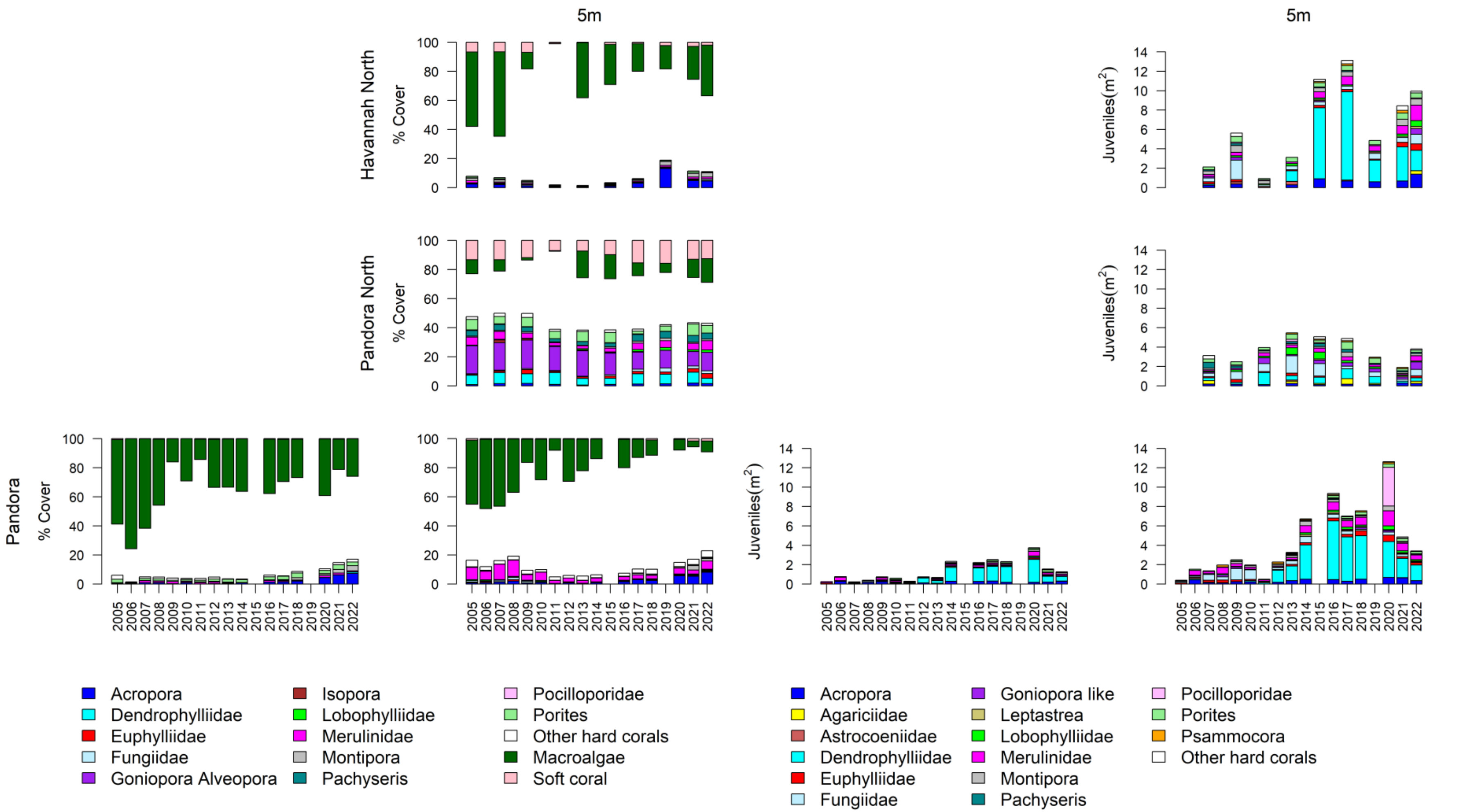

Figure A4 continued

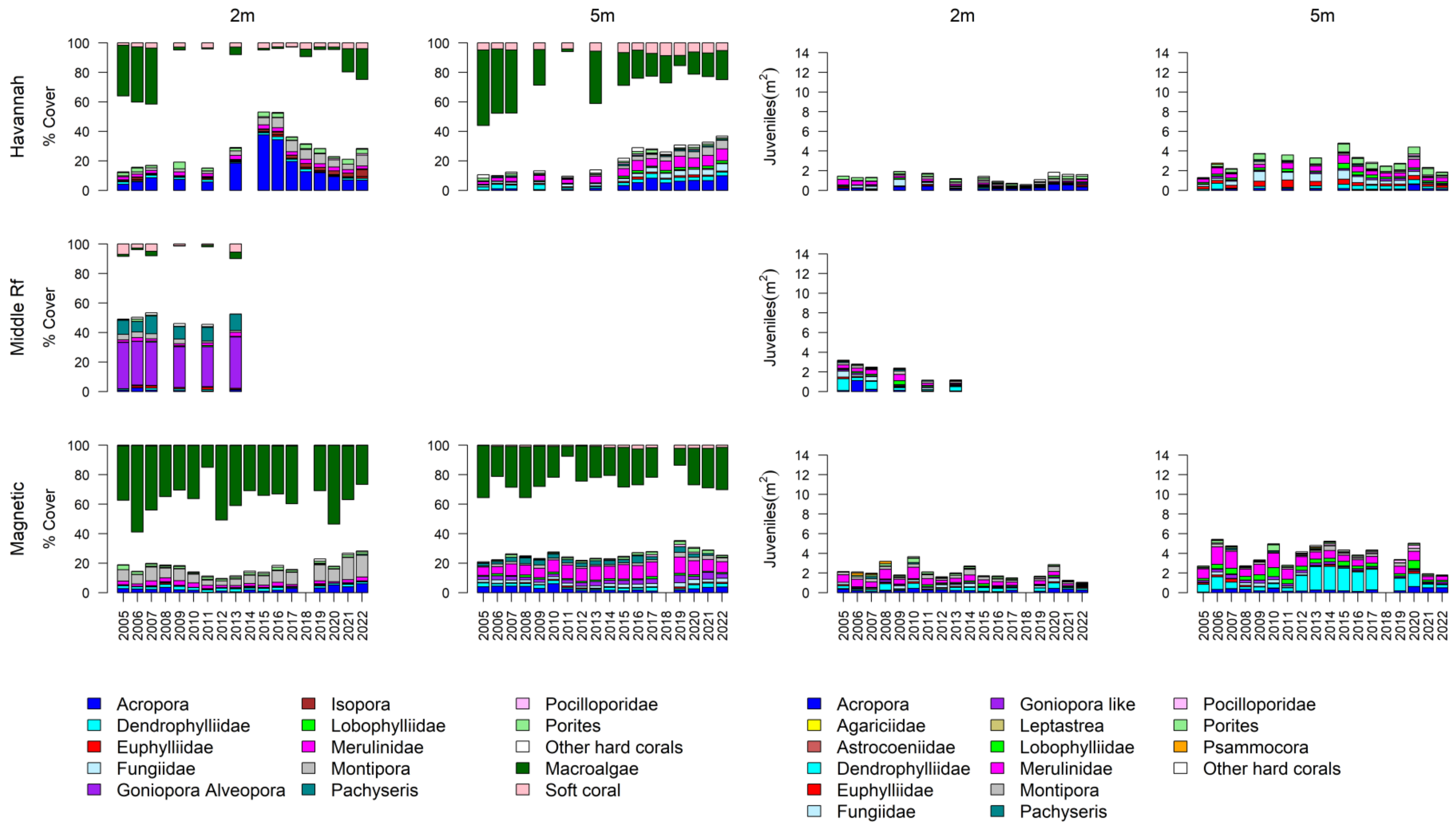

Figure A4 continued

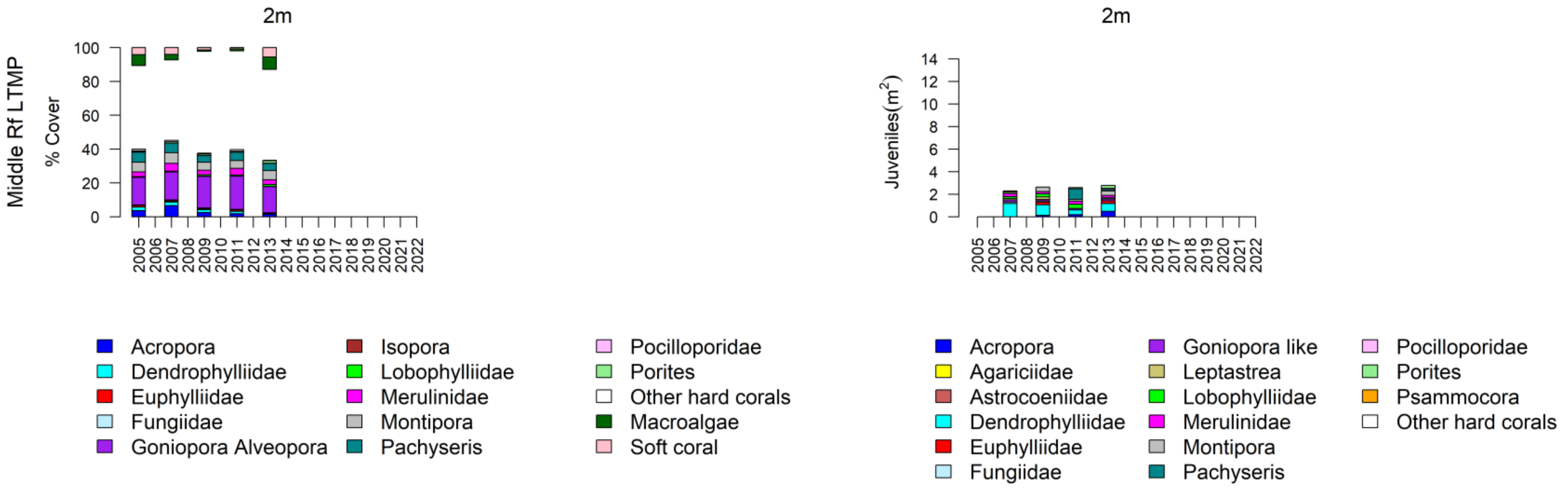

Figure A4 continued

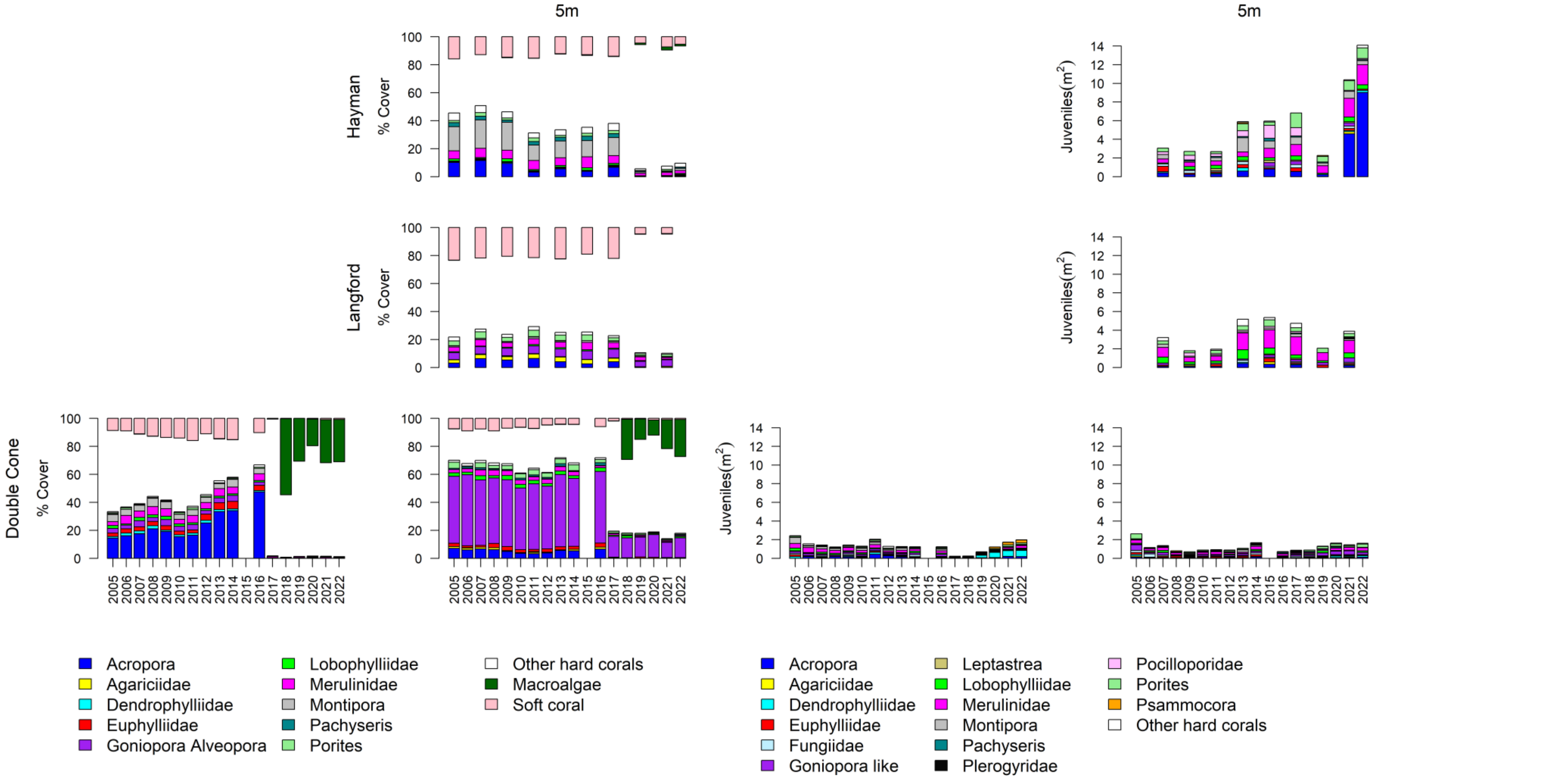

Figure A5 Mackay-Whitsunday region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends with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

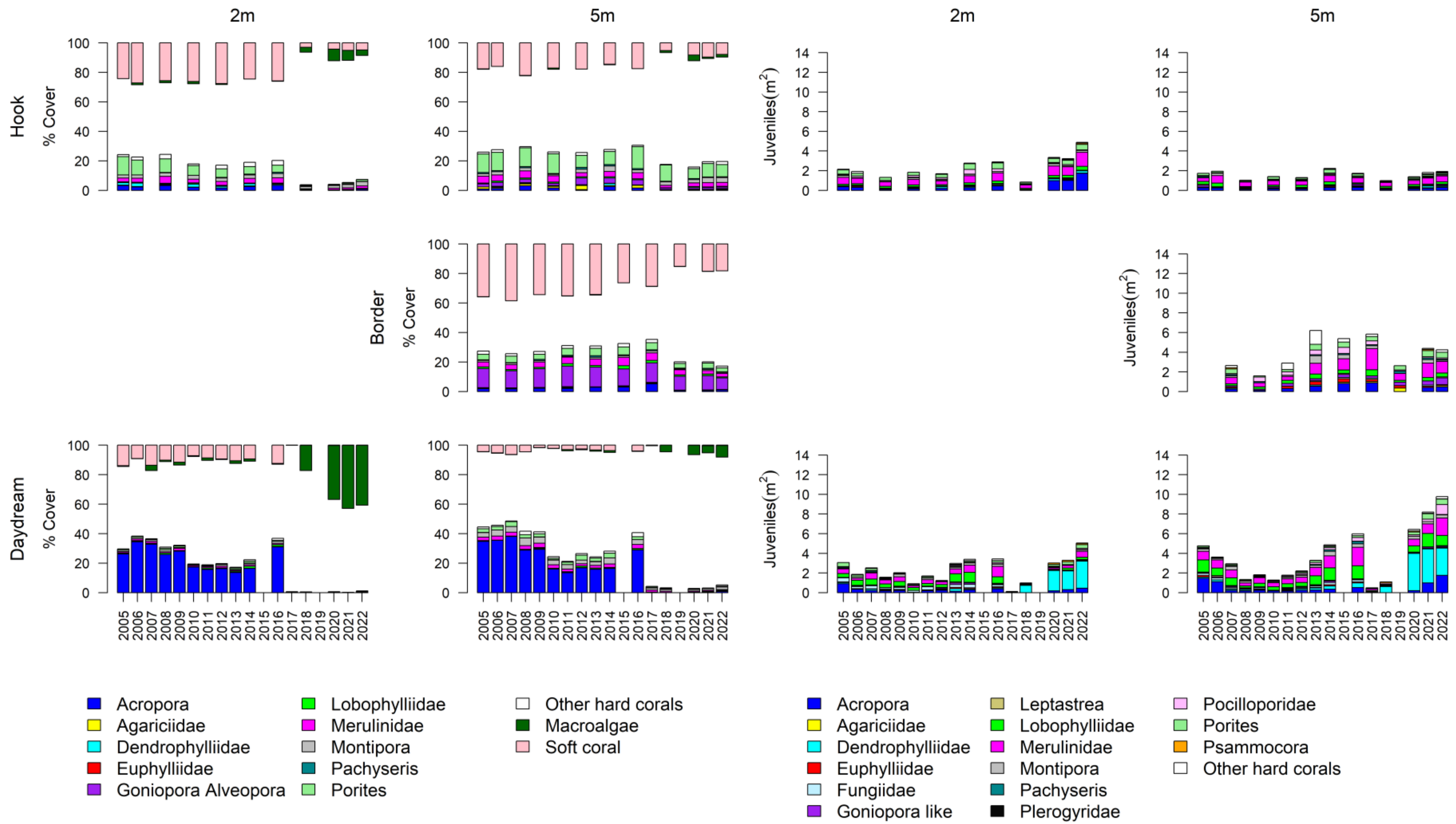

Figure A5 continued

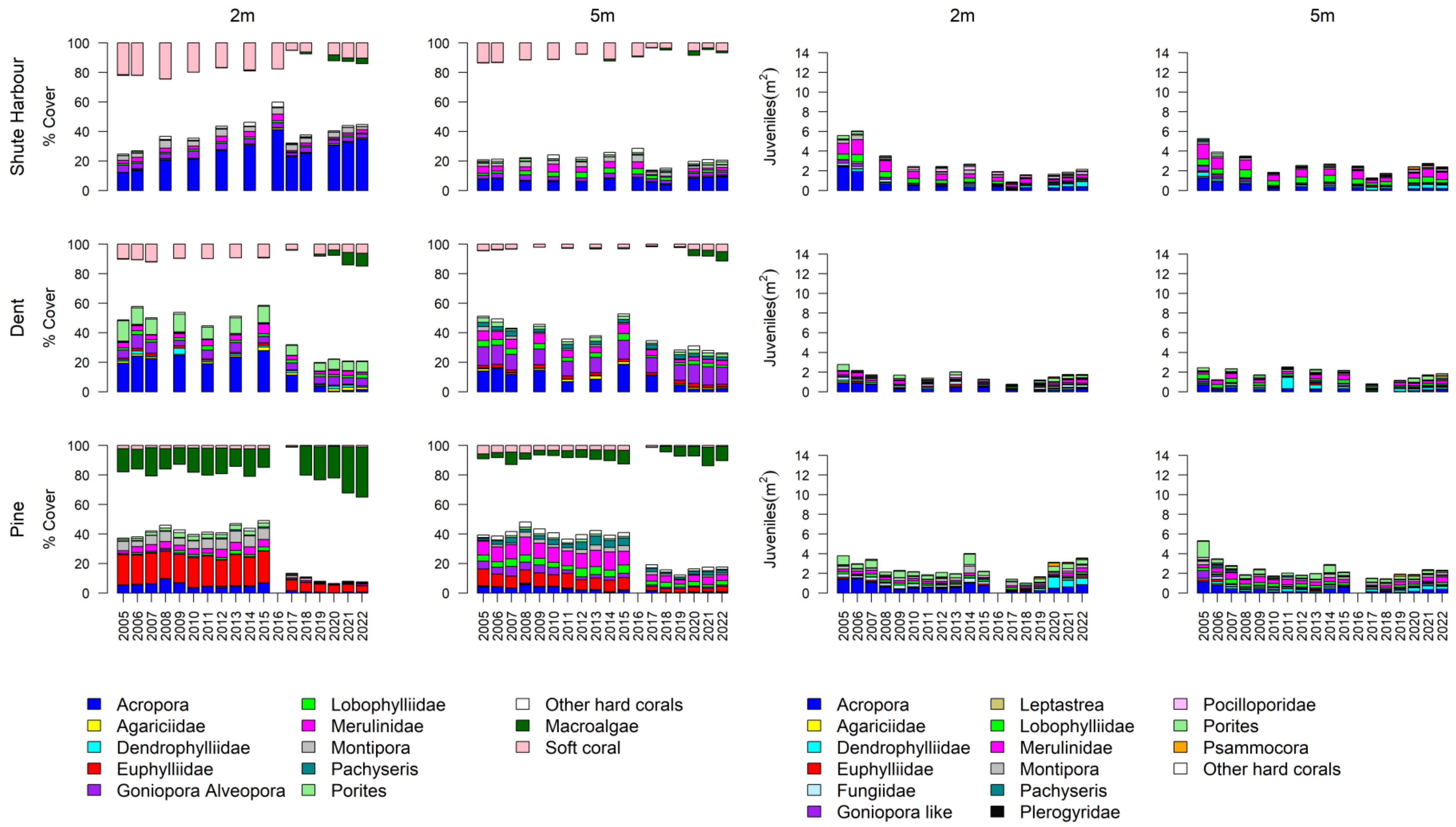

Figure A5 continued

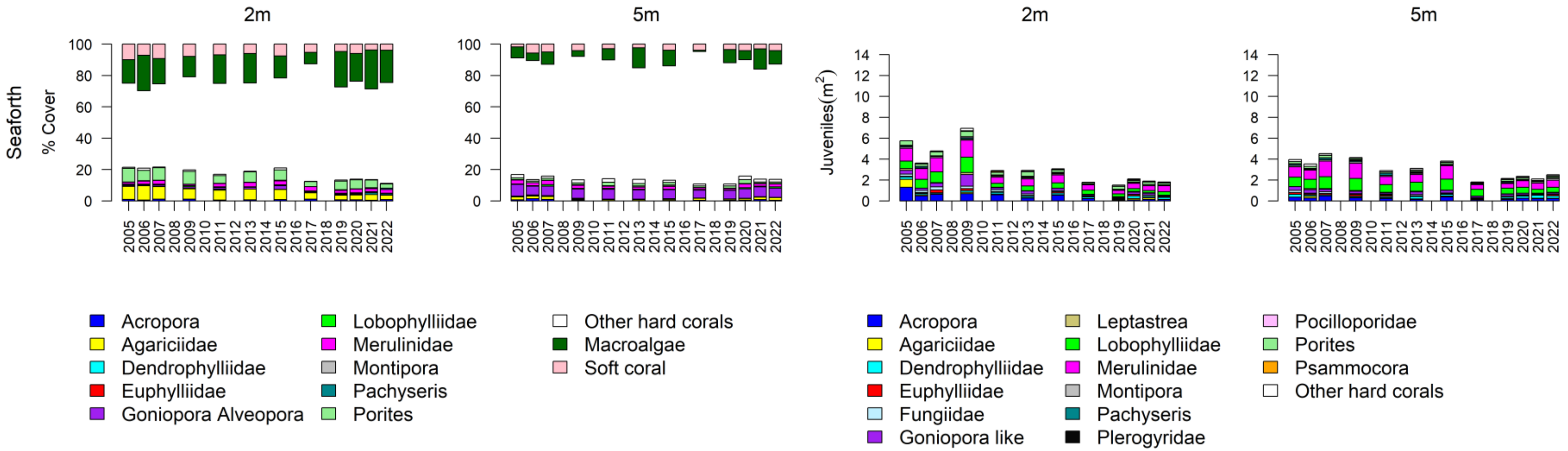

Figure A5 continued

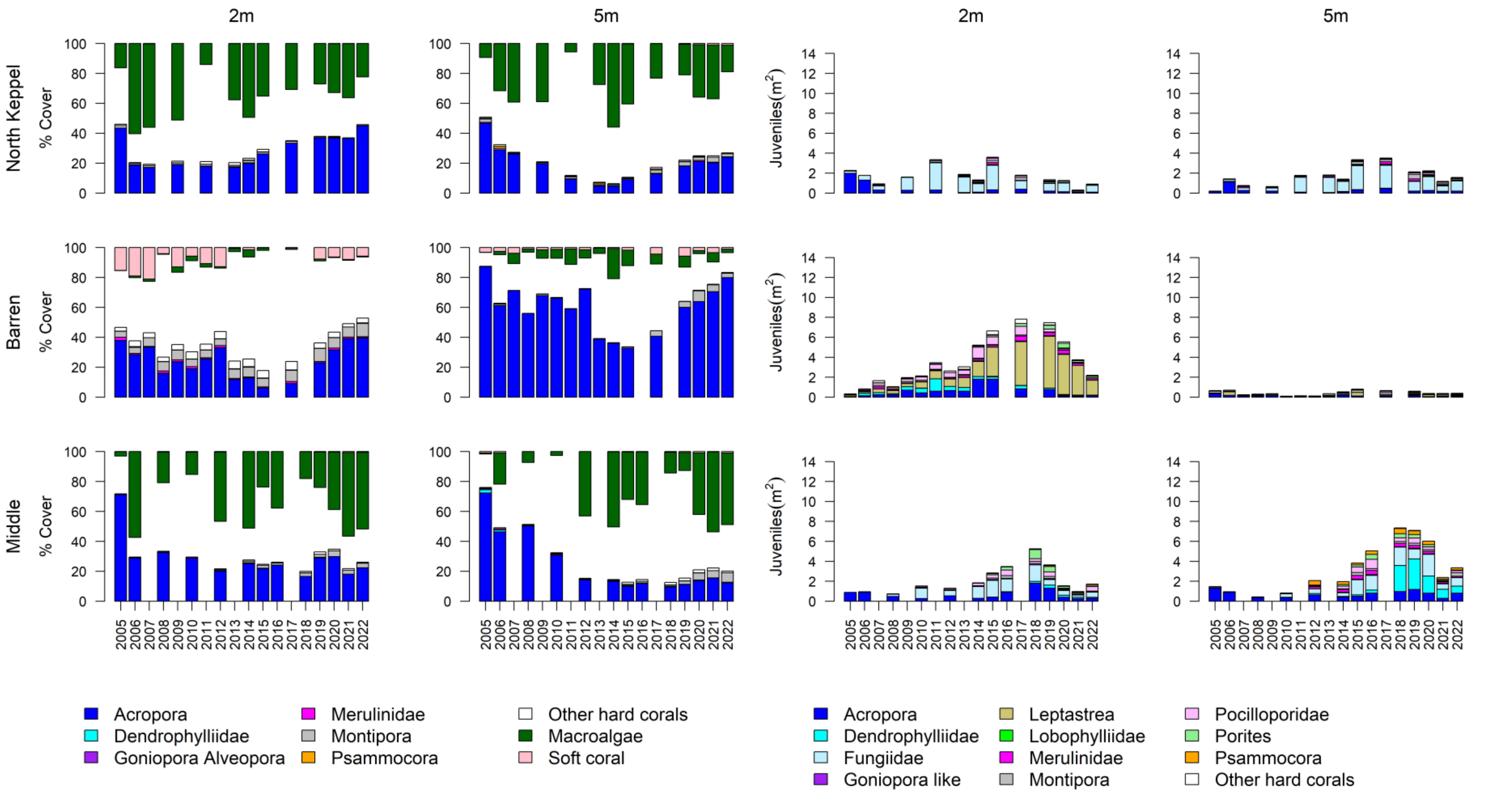

Figure A6 Fitzroy region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends with relevant groupings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

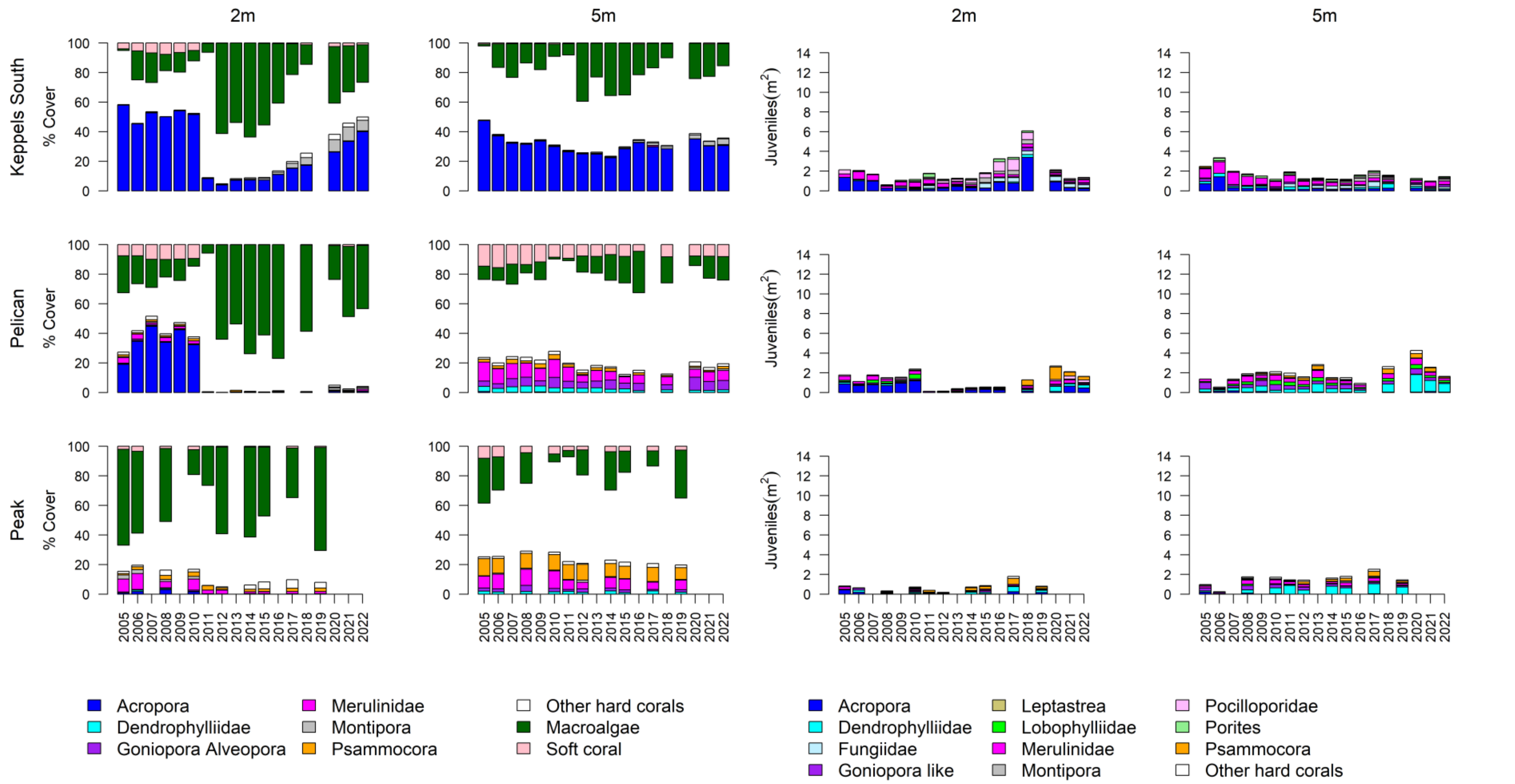

Figure A6 continued

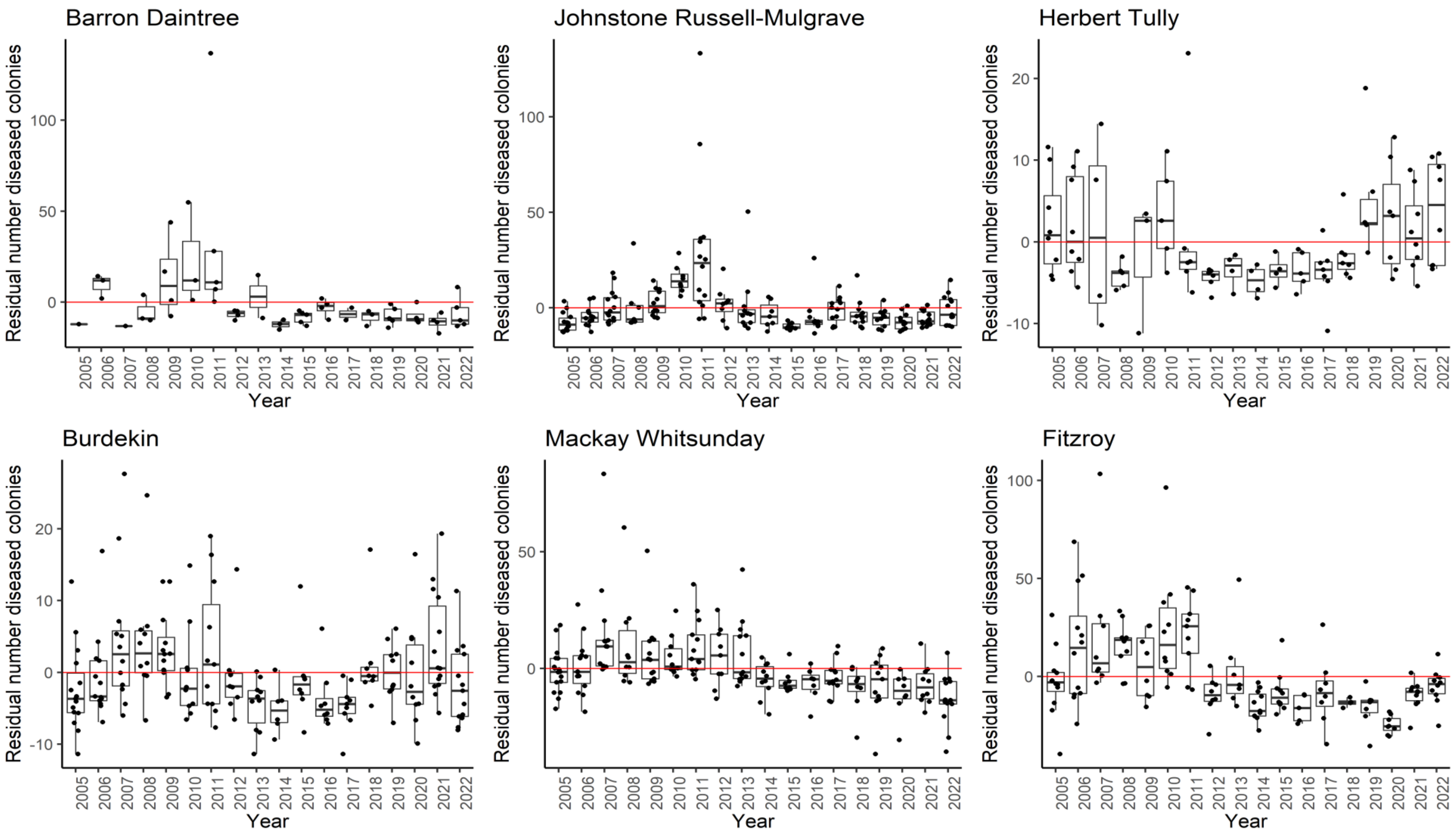

Figure A7 Coral disease by year in each region. Boxplots include the number of coral colonies suffering ongoing mortality attributed to either disease, sedimentation or 'unknown causes' for each reef, depth and year. Data are standardised to the reef and depth mean across years (see section 2.3.3).

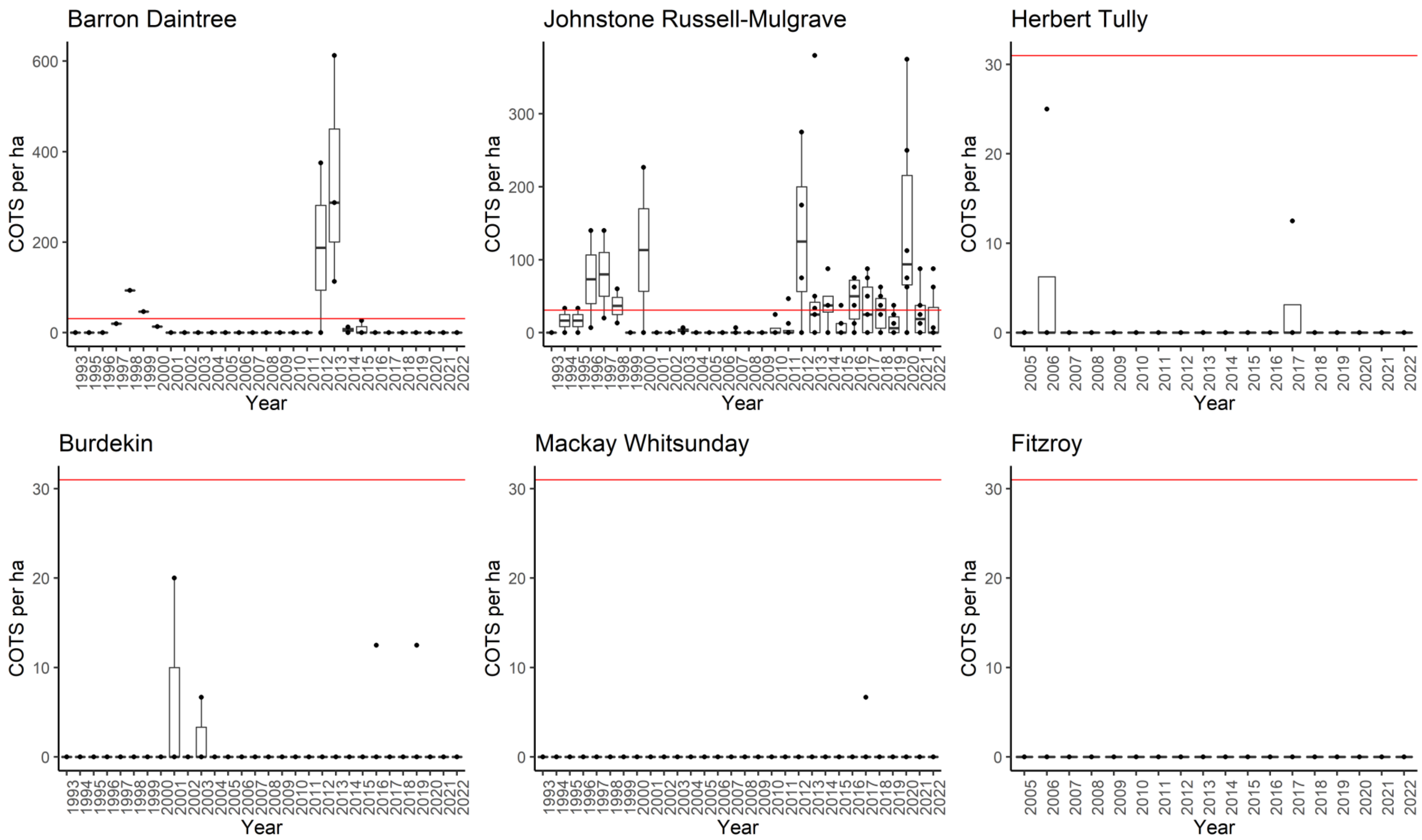

Figure A8 Crown -of-thorn-starfish mean density (individuals/ha) by year in each region. Red line indicates outbreak densities of 31 individuals per hectare (see section 2.3.3 for derivation).

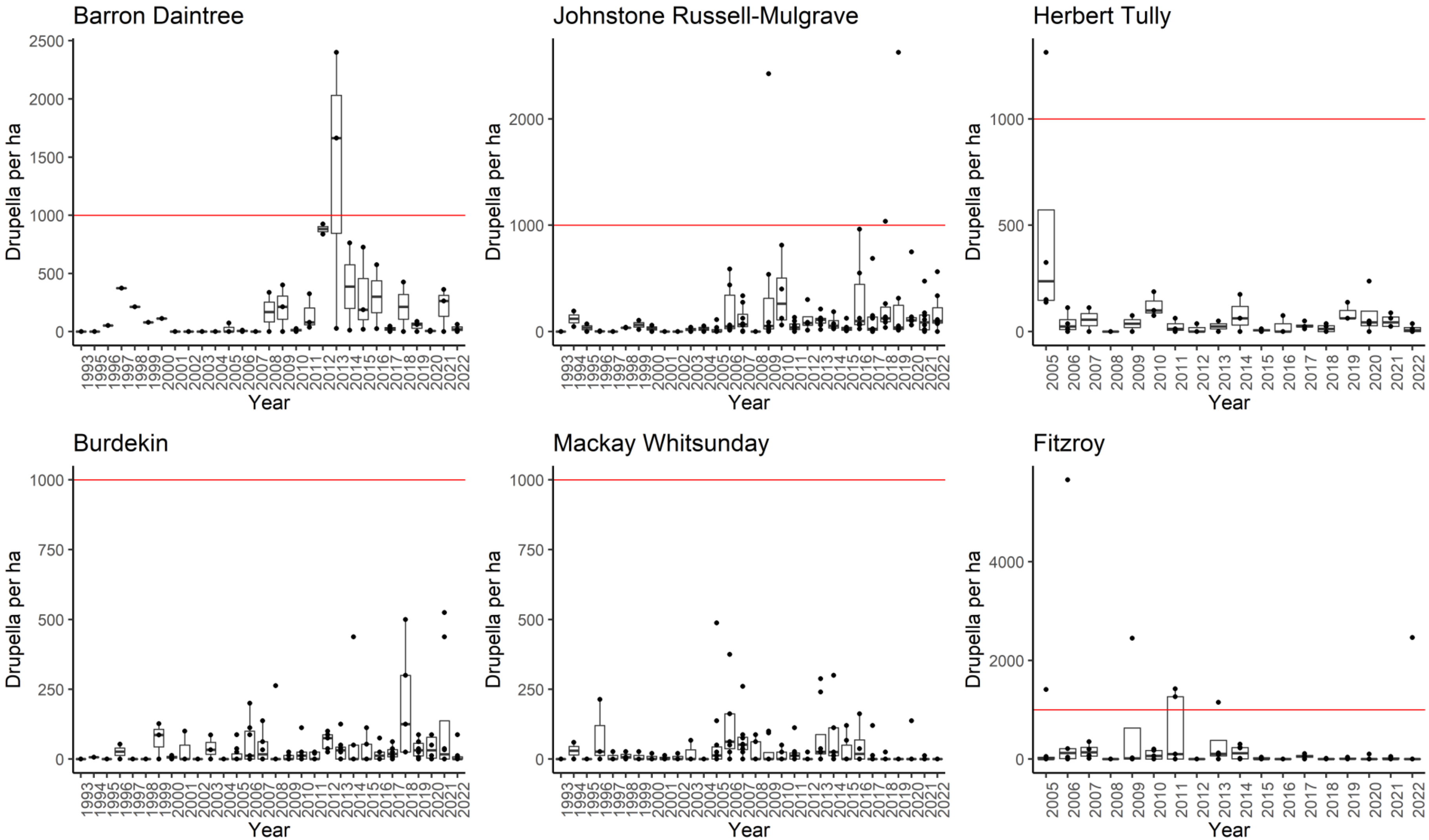

Figure A9 Mean density of *Drupella* by year in each (sub-)region. Red line indicates densities of *Drupella* which have detrimental impact on coral communities (see section 2.3.3 for derivation).

Table A9 Percent cover of hard coral genera 2022. 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*	*Astreopora*	*Caulastrea*	*Cyphastrea*	*Diploastrea*	*Dipsastraea*	*Duncanop-sammia*	*Echinopora*	*Favites*	*Galaxea*	*Goniastrea*	*Goniopora*	*Isopora*	*Lobophyllia*	*Merulina*	*Montipora*	*Mycedium*	*Oxypora*	*Pachyseris*	*Paragoniastrea*	*Pavona*	*Pectinia*	*Platygyra*	*Plesiastrea*	*Pocillopora*	*Podobacia*	*Porites*	*Psammocora*	*Seriatopora*	*Turbinaria*	Other
Barron Daintree	Low Isles	5	2.61	0.00	0.10	0.03	0.03	0.60	0.29	0.00	1.20	0.10	3.51	0.13	0.66	0.00	1.46	0.53	0.33	0.27	0.17	1.16	0.00	0.30	0.23	0.37	0.00	0.10	0.30	18.44	0.17	0.60	0.03	1.83
	Snapper North	2	5.68	0.00	0.00	0.17	0.00	0.00	0.04	0.00	4.67	0.00	0.08	0.00	0.04	0.13	0.08	0.00	1.47	0.00	0.00	0.00	0.00	0.13	0.00	0.00	0.00	0.00	0.00	2.04	0.00	0.00	0.00	0.42
		5	2.88	0.00	0.00	0.00	0.00	0.00	0.13	0.00	0.06	0.13	0.75	0.06	8.69	0.56	0.00	0.06	4.44	0.31	0.00	5.00	0.06	1.31	0.00	0.19	0.00	0.00	0.25	4.38	0.00	0.00	0.00	1.81
	Snapper South	2	4.13	0.00	0.00	0.00	0.21	0.00	0.13	0.00	0.04	0.25	2.33	1.25	0.25	0.00	0.25	0.00	3.71	0.00	0.00	0.00	0.00	0.17	0.00	0.00	0.00	0.21	0.00	28.31	0.33	0.00	0.04	0.67
		5	6.97	0.00	0.00	4.73	0.00	0.00	0.00	0.00	0.00	0.06	0.25	0.00	3.88	0.00	0.06	0.06	1.70	0.06	0.00	3.06	0.00	1.13	0.00	0.06	0.00	0.00	0.00	33.43	0.44	0.00	0.25	0.50
Johnstone Russell-Mulgrave	Fitzroy East	2	11.75	0.00	0.06	0.00	0.00	0.00	0.75	0.00	0.13	0.44	0.19	0.19	0.06	0.00	0.31	0.00	6.31	0.00	0.00	0.00	0.00	0.38	0.00	0.38	0.00	3.31	0.00	7.19	0.25	0.00	0.00	0.50
		5	7.75	0.00	0.19	0.06	0.00	2.00	1.06	0.00	1.00	0.38	1.44	0.81	0.63	0.00	1.88	0.00	0.94	0.00	0.13	0.00	0.13	0.06	0.00	0.56	0.00	3.19	0.00	13.38	0.19	0.00	0.00	1.19
	Fitzroy West	2	24.13	0.00	0.00	0.00	0.00	1.19	0.00	0.00	1.19	0.06	0.56	0.00	0.06	0.00	1.19	0.00	5.00	0.00	0.00	0.06	0.00	0.19	0.00	0.06	0.00	2.25	0.00	6.44	0.06	0.00	0.00	0.75
		5	7.81	0.00	0.00	0.00	0.00	1.19	0.13	0.00	0.31	0.00	0.75	0.00	0.81	0.00	2.06	0.13	4.25	0.13	0.00	0.81	0.00	0.06	0.13	0.06	0.00	0.25	0.25	15.56	0.56	0.00	0.00	1.44
	Fitzroy West LTMP	5	0.42	0.00	0.31	0.00	0.09	1.01	0.25	0.17	0.08	0.23	1.46	0.08	0.34	0.00	0.43	0.66	2.10	0.00	0.34	3.18	0.00	0.09	0.15	0.00	0.17	0.39	15.14	0.00	0.00	0.00	0.34	1.06
	Franklands East	2	32.69	0.00	0.00	0.06	0.00	0.00	0.44	0.00	1.94	0.19	0.19	0.00	0.06	0.31	0.19	0.06	22.50	0.00	0.00	0.00	0.00	0.13	0.00	0.13	0.06	1.19	0.00	1.38	0.00	0.00	0.06	0.38
		5	17.19	0.00	0.00	0.00	0.19	0.00	0.31	0.00	0.25	0.13	0.50	0.06	0.06	0.00	0.13	0.25	4.06	0.00	0.00	0.00	0.00	0.00	0.25	0.31	0.00	0.25	0.00	4.31	0.00	0.00	0.00	0.56
	Franklands West	2	7.75	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.44	0.00	0.31	0.00	0.63	0.00	0.31	0.00	0.25	0.00	0.00	7.13	0.00	0.00	0.00	0.00	0.00	0.31	0.00	30.44	0.13	1.56	0.00	0.31
		5	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.81	0.00	0.00	0.00	0.00	0.00	0.00	0.00	62.56	0.00	0.06	0.00	0.00
	High East	2	21.19	0.00	0.00	0.00	0.00	0.00	0.06	0.00	1.44	0.19	0.00	0.38	0.25	0.00	0.94	0.00	6.81	0.00	0.00	0.00	0.00	0.38	0.00	0.63	0.00	0.63	0.00	4.44	0.00	0.00	0.13	1.44
		5	13.69	0.00	0.00	0.00	0.06	0.00	0.06	0.00	2.75	0.56	0.19	0.06	0.69	0.00	0.31	0.00	6.44	0.00	0.00	0.00	0.06	0.13	0.00	0.13	0.00	0.81	0.19	14.50	0.38	0.00	0.00	0.75
	High West	2	4.32	0.00	0.06	0.06	0.00	0.00	0.44	0.00	0.56	0.19	0.69	0.00	3.25	0.00	0.06	0.50	0.69	0.00	0.00	0.00	0.25	0.38	0.00	0.31	0.00	0.63	0.00	39.71	0.06	0.00	0.00	1.81
		5	1.19	0.00	0.19	0.00	0.00	0.00	0.31	0.00	0.00	0.31	0.81	0.00	5.19	0.00	0.00	0.06	0.00	0.06	0.00	0.63	0.00	1.00	0.06	0.19	0.00	0.25	0.00	17.60	0.19	0.00	0.06	1.01

(sub-)region	Reef	Depth	*Acropora*	*Alveopora*	*Astreopora*	*Caulastrea*	*Cyphastrea*	*Diploastrea*	*Dipsastraea*	*Duncanop-sammia*	*Echinopora*	*Favites*	*Galaxea*	*Goniastrea*	*Goniopora*	*Isopora*	*Lobophyllia*	*Merulina*	*Montipora*	*Mycedium*	*Oxypora*	*Pachyseris*	*Paragoniastrea*	*Pavona*	*Pectinia*	*Platygyra*	*Plesiastrea*	*Pocillopora*	*Podobacia*	*Porites*	*Psammocora*	*Seriatopora*	*Turbinaria*	Other
Herbert Tully	Barnards	2	36.81	0.00	0.00	0.00	0.06	0.00	0.13	0.31	0.25	0.00	0.13	0.00	0.00	0.00	0.19	0.00	13.88	0.00	0.00	0.00	0.06	0.00	0.00	0.06	0.00	0.25	0.00	0.19	0.00	0.00	0.44	0.19
		5	21.77	0.06	0.06	0.00	0.06	0.00	0.06	0.06	0.50	0.13	0.06	0.00	0.13	0.00	0.19	0.06	19.42	0.06	0.13	0.19	0.06	0.00	0.19	0.31	0.00	0.75	0.06	0.50	0.19	0.00	0.69	0.69
	Dunk North	2	32.38	0.00	0.00	0.00	0.38	0.00	0.13	1.19	0.00	0.19	0.56	0.00	0.06	0.00	0.06	0.00	4.50	0.00	0.00	0.00	0.00	0.38	0.00	0.31	0.06	1.69	0.13	0.19	0.31	0.00	5.50	0.81
		5	6.50	0.06	0.63	0.00	0.25	0.00	0.44	0.56	0.00	1.25	0.19	0.00	0.06	0.00	0.06	0.06	6.44	0.00	0.19	0.00	0.13	0.00	0.00	0.00	0.00	1.06	0.06	0.81	0.06	0.00	11.56	0.81
	Dunk South	2	12.25	0.00	0.00	0.00	1.94	0.00	0.63	0.19	0.06	0.63	1.75	0.00	0.19	0.00	0.06	0.00	10.44	0.00	0.00	0.00	0.06	0.44	0.00	0.44	0.13	0.06	0.00	2.81	0.00	0.00	1.75	0.50
		5	1.50	0.00	0.00	0.06	0.31	0.00	3.63	0.06	0.44	1.75	0.31	1.13	0.25	0.00	0.94	2.00	3.94	3.06	0.69	5.81	0.31	0.56	0.88	1.31	0.00	0.00	0.19	1.81	0.13	0.00	7.00	1.56
	Bedarra	2	4.94	0.00	0.19	0.00	0.94	0.00	0.50	0.00	0.00	0.19	0.56	0.06	0.19	0.00	0.44	0.19	0.69	0.00	0.00	0.00	0.06	0.50	0.00	0.31	0.00	0.50	0.00	4.94	0.06	0.00	0.38	0.50
		5	0.25	0.06	0.00	0.19	0.19	0.06	3.94	0.06	0.00	0.94	0.00	0.25	3.56	0.00	1.25	0.88	0.81	0.75	0.00	1.00	0.00	0.38	0.00	0.19	0.00	0.13	0.75	3.56	0.00	0.00	0.19	1.44
Burdekin	Palms East	2	37.63	0.00	0.00	0.00	0.06	0.00	0.19	0.00	0.00	0.06	0.00	0.00	0.00	0.06	0.00	0.00	1.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.63	0.00	0.00	0.00	0.06
		5	44.75	0.00	0.06	0.00	0.06	0.00	0.19	0.00	0.00	0.25	0.06	0.06	0.00	0.00	0.19	0.25	2.38	0.00	0.00	0.00	0.06	0.00	0.00	0.38	0.00	0.25	0.00	0.94	0.13	0.00	0.00	0.19
	Palms West	2	2.69	0.00	0.00	0.00	0.00	0.00	0.25	0.00	0.38	0.13	0.06	0.00	0.19	0.13	0.00	0.00	0.31	0.00	0.00	0.13	0.00	0.00	0.00	0.00	0.00	9.56	0.00	0.00	0.00	0.00	0.00	0.13
		5	1.56	0.00	0.13	0.00	0.00	0.56	0.13	0.00	0.13	0.19	0.00	0.13	1.44	0.00	0.31	0.00	1.31	0.00	0.25	0.13	0.00	0.13	0.00	0.38	0.00	0.94	0.00	5.81	0.00	0.00	0.13	1.19
	Havannah North	5	4.49	0.07	0.00	0.00	0.10	0.00	0.23	0.00	0.07	0.03	0.27	0.03	0.00	0.00	0.03	0.40	2.95	0.00	0.03	0.03	0.00	0.00	0.03	0.13	0.00	0.07	0.03	0.40	0.13	0.00	0.20	1.30
	Havannah	2	6.81	0.00	0.00	0.00	0.00	0.00	0.06	0.81	0.75	0.06	1.00	0.06	0.19	4.69	0.50	0.25	7.44	0.00	0.00	0.00	0.00	0.50	0.13	0.31	0.00	1.06	0.00	2.94	0.06	0.00	0.50	0.31
		5	9.88	0.00	0.13	0.00	0.38	0.00	0.69	0.00	0.94	0.19	0.69	0.06	0.19	0.19	0.50	3.81	5.75	0.38	1.13	0.81	0.06	0.38	1.19	0.00	0.00	0.06	0.63	0.75	0.69	0.00	2.44	4.94
	Pandora	2	7.25	0.00	0.00	0.00	0.25	0.00	0.13	0.06	0.00	0.31	0.06	0.00	0.19	0.00	0.00	0.00	3.50	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.25	0.06	0.00	2.44	0.00	0.00	0.88	0.69
		5	8.13	0.00	0.00	0.00	0.56	3.25	2.06	0.00	0.25	0.38	0.63	0.19	0.31	0.00	0.13	0.44	1.69	0.19	0.13	0.38	0.00	0.00	0.00	0.88	0.00	0.44	0.00	0.19	0.13	0.00	0.38	2.19
	Pandora North	5	1.44	0.00	0.07	0.17	0.27	0.00	0.47	0.00	1.86	0.17	3.10	0.17	12.66	0.03	0.50	1.17	1.04	1.35	1.11	4.10	0.00	0.23	0.71	0.10	0.00	0.03	0.37	5.27	0.17	0.00	3.76	2.72
	Lady Elliot	2	11.25	0.00	0.00	0.00	0.06	0.00	0.25	0.00	0.00	0.06	1.38	0.00	0.00	0.00	0.00	0.00	6.13	0.00	0.00	0.00	0.00	2.13	0.00	0.13	0.00	0.00	0.00	0.63	0.00	0.00	1.06	3.13
		5	1.50	0.38	0.00	0.00	0.31	0.00	0.69	0.25	0.00	1.00	15.38	0.00	4.25	0.00	1.06	0.00	0.63	1.94	1.69	2.31	0.00	0.00	1.13	0.00	0.13	0.00	2.19	4.88	0.06	0.00	2.94	1.06
	Magnetic	2	6.00	0.00	0.00	0.00	1.19	0.00	0.19	0.00	0.06	0.94	0.38	0.00	0.00	0.00	0.00	0.06	14.81	0.00	0.00	0.63	0.00	0.38	0.00	0.13	0.00	0.00	0.00	1.56	0.31	0.00	1.31	0.38
		5	3.81	0.00	0.00	0.00	0.31	0.00	2.13	0.13	0.00	0.81	0.69	0.00	3.06	0.00	0.31	2.38	2.63	0.06	0.25	0.38	0.00	0.13	0.19	1.13	0.00	0.00	2.06	1.19	0.00	0.00	2.19	1.50

(sub-)region	Reef	Depth	*Acropora*	*Alveopora*	*Astreopora*	*Caulastrea*	*Cyphastrea*	*Diploastrea*	*Dipsastraea*	*Duncanop-sammia*	*Echinopora*	*Favites*	*Galaxea*	*Goniastrea*	*Goniopora*	*Isopora*	*Lobophyllia*	*Merulina*	*Montipora*	*Mycedium*	*Oxypora*	*Pachyseris*	*Paragoniastrea*	*Pavona*	*Pectinia*	*Platygyra*	*Plesiastrea*	*Pocillopora*	*Podobacia*	*Porites*	*Psammocora*	*Seriatopora*	*Turbinaria*	Other
Mackay-Whitsunday	Hayman	5	0.40	0.00	0.07	0.00	0.03	2.23	0.30	0.00	0.67	0.27	0.57	0.07	0.00	0.00	0.50	0.17	1.53	0.10	0.07	0.80	0.07	0.30	0.03	0.30	0.00	0.20	0.00	0.13	0.13	0.00	0.17	0.57
	Border	5	1.00	0.10	0.07	0.00	0.03	0.80	0.76	0.03	0.17	0.13	0.03	0.37	7.82	0.00	0.60	0.10	0.67	0.00	0.00	0.37	0.07	0.23	0.10	0.23	0.13	0.00	0.00	2.59	0.00	0.13	0.03	0.66
	Hook	2	0.75	0.00	0.00	0.00	0.44	0.00	0.50	0.00	0.00	0.44	0.00	0.19	0.00	0.00	0.25	0.00	2.81	0.00	0.00	0.25	0.13	0.19	0.00	0.00	0.00	0.00	0.00	1.25	0.00	0.00	0.06	0.13
		5	0.38	0.00	1.32	0.00	0.06	0.50	1.00	0.00	0.00	0.94	0.06	0.00	1.13	0.00	0.25	0.00	3.38	0.00	0.13	0.50	0.25	0.50	0.06	0.13	0.00	0.00	0.00	8.20	0.06	0.00	0.19	0.56
	Double Cone	2	0.13	0.00	0.00	0.00	0.06	0.00	0.06	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.00	0.00	0.19	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.44	0.00	0.00	0.00	0.00
		5	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.00	0.06	0.00	0.50	0.25	14.06	0.00	0.75	0.00	0.19	0.00	0.25	0.63	0.00	0.00	0.06	0.06	0.56	0.00	0.00	0.19	0.00	0.00	0.00	0.19
	Daydream	2	0.50	0.00	0.00	0.00	0.00	0.00	0.06	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.00	0.00	0.13	0.13
		5	0.75	0.00	0.00	0.00	0.06	0.00	0.13	0.25	0.00	0.19	0.00	0.00	0.00	0.00	0.13	0.00	1.88	0.00	0.00	0.13	0.00	0.00	0.19	0.00	0.00	0.06	0.06	0.38	0.00	0.75	0.13	0.19
	Dent	2	1.44	0.00	0.00	0.06	0.00	0.00	0.06	0.00	0.44	0.00	0.06	0.00	5.25	0.00	2.63	0.25	0.13	0.00	0.06	0.06	0.00	1.50	0.31	0.00	0.00	0.06	0.00	7.00	0.06	0.00	0.94	0.44
		5	2.25	0.13	0.19	0.00	0.00	0.00	0.31	0.00	0.69	0.31	1.63	0.25	11.31	0.06	0.63	0.94	0.13	0.00	0.81	2.13	0.00	0.69	0.50	0.00	0.00	0.19	0.13	2.06	0.00	0.00	0.25	0.75
	Shute Harbour	2	34.68	0.00	0.19	0.75	0.06	0.00	0.06	0.00	0.06	0.06	0.13	0.13	2.81	0.13	0.44	0.44	1.88	0.00	0.06	0.06	0.00	0.75	0.56	0.00	0.00	0.25	0.00	0.06	0.00	0.00	0.00	1.06
		5	9.19	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.13	0.00	0.69	0.19	2.13	0.13	0.69	0.19	1.88	0.50	0.31	0.25	0.13	0.50	0.31	0.25	0.00	0.06	0.06	1.81	0.00	0.00	0.00	1.13
	Pine	2	0.69	0.00	0.00	0.13	0.00	0.00	0.00	0.00	0.13	0.00	4.31	0.00	0.00	0.00	0.19	0.00	0.31	0.00	0.00	0.13	0.13	0.00	0.50	0.00	0.00	0.19	0.00	0.44	0.00	0.00	0.00	0.44
		5	0.56	0.06	0.00	0.00	0.00	0.00	0.06	0.00	0.38	0.25	3.44	0.06	0.88	0.00	1.63	0.19	1.00	1.00	0.38	1.94	0.00	0.00	2.13	0.06	0.00	0.06	0.25	0.63	0.00	0.00	0.19	2.56
	Seaforth	2	0.75	0.00	0.19	0.00	0.06	0.00	0.81	0.63	0.31	0.63	0.00	0.06	0.38	0.00	0.13	0.06	0.00	0.00	0.00	0.81	0.00	2.88	0.00	0.00	0.00	0.13	0.00	2.63	0.00	0.00	0.00	0.75
		5	0.44	0.13	0.44	0.25	0.06	0.94	1.00	0.00	0.00	0.25	0.00	0.13	5.69	0.00	0.38	0.00	0.13	0.00	0.00	0.13	0.00	1.19	0.19	0.06	0.06	0.00	0.19	0.94	0.00	0.00	0.06	1.06

(sub-)region	Reef	Depth	*Acropora*	*Alveopora*	*Astreopora*	*Caulastrea*	*Cyphastrea*	*Diploastrea*	*Dipsastraea*	*Duncanop-sammia*	*Echinopora*	*Favites*	*Galaxea*	*Goniastrea*	*Goniopora*	*Isopora*	*Lobophyllia*	*Merulina*	*Montipora*	*Mycedium*	*Oxypora*	*Pachyseris*	*Paragoniastrea*	*Pavona*	*Pectinia*	*Platygyra*	*Plesiastrea*	*Pocillopora*	*Podobacia*	*Porites*	*Psammocora*	*Seriatopora*	*Turbinaria*	Other
Fitzroy	Barren	2	39.50	0.25	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.06	0.00	1.06	0.13	0.00	8.81	0.00	0.00	0.00	0.06	0.06	0.00	0.31	0.00	0.88	0.00	0.50	0.38	0.00	0.00	0.75
		5	79.91	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	2.94	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.00	0.00	0.00	0.19
	North Keppel	2	45.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.63	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.06	0.00	0.00	0.00
		5	24.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	2.01	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.00	0.19	0.00	0.06	0.13	0.00	0.00	0.25
	Middle	2	22.25	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.06	0.00	0.00	0.00	0.00	3.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.00	0.19	0.13	0.00	0.00	0.06
		5	12.25	0.00	0.00	0.00	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	6.38	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.88	0.00	0.25	0.00	0.00	0.25	0.00
	Keppels South	2	40.04	0.00	0.00	0.00	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	7.14	0.00	0.00	0.00	0.13	0.00	0.00	0.19	0.00	1.94	0.06	0.25	0.00	0.00	0.00	0.06
		5	30.56	0.00	0.00	0.00	0.25	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.88	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.00	0.25	0.06	0.00	0.25	0.31
	Pelican	2	0.69	0.00	0.00	0.00	0.50	0.00	0.00	0.00	0.00	0.50	0.00	0.00	0.19	0.00	0.00	0.00	0.88	0.00	0.00	0.00	0.13	0.00	0.00	0.00	0.13	0.25	0.00	0.06	0.31	0.00	0.13	0.31
		5	0.13	5.34	0.56	0.00	0.25	0.00	0.06	0.13	0.00	4.05	0.00	0.00	1.07	0.00	0.38	0.00	1.14	0.00	0.00	0.00	1.44	0.00	0.00	0.83	0.19	0.00	0.00	0.06	1.69	0.00	1.50	0.63

Table A10 Percent cover of soft coral families 2022. 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	Alcyoniidae	Anthothelidae	Briareidae	Clavulariinae	Heliporidae	Neptheidae	Xeniidae	Other
Barron Daintree	Low Isles	5	3.25	0.00	16.18	0.13	0.07	0.00	0.03	0.16
	Snapper North	2	0.79	0.00	4.42	10.07	0.00	0.00	0.00	0.00
		5	0.12	0.00	1.19	0.12	0.00	0.00	9.44	0.00
	Snapper South	2	1.96	0.00	0.46	0.21	3.96	0.00	0.00	0.00
		5	0.12	0.00	4.83	0.00	5.51	0.00	0.00	0.06
Johnstone Russell-Mulgrave	Fitzroy East	2	1.75	0.00	0.56	1.81	0.00	0.38	0.00	0.00
		5	9.25	0.00	7.19	0.25	0.00	0.38	0.00	0.00
	Fitzroy West	2	36.38	0.00	0.12	0.00	0.00	0.00	0.00	0.00
		5	26.94	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Fitzroy West LTMP	5	15.81	0.00	0.63	0.03	0.03	0.03	0.00	0.00
	Franklands East	2	1.38	0.00	0.06	1.19	0.50	0.00	0.00	0.00
		5	2.31	0.00	0.62	0.94	0.00	0.00	0.25	0.00
	Franklands West	2	7.12	0.00	0.00	9.88	0.00	0.50	0.00	0.00
		5	1.69	0.00	0.00	1.19	0.19	0.00	0.00	0.00
	High East	2	9.19	0.00	5.56	0.00	0.00	0.00	0.00	0.00
		5	0.56	0.00	8.75	0.00	0.00	0.00	0.00	0.00
	High West	2	5.00	0.00	0.00	0.00	2.44	0.00	0.00	0.00
		5	1.25	0.00	0.44	0.00	1.12	0.00	0.12	0.00

(sub-)region	Reef	Depth	Alcyoniidae	Anthothelidae	Briareidae	Clavulariinae	Heliporidae	Neptheidae	Xeniidae	Other
Herbert Tully	Barnards	2	1.94	0.00	1.19	0.00	0.00	0.00	2.69	0.00
		5	2.31	0.00	2.69	0.25	0.00	0.00	3.31	0.00
	Dunk North	2	0.44	0.00	0.12	0.25	0.00	0.00	0.44	0.00
		5	1.25	0.00	0.12	0.00	0.00	0.00	5.19	0.94
	Dunk South	2	0.56	0.00	0.31	0.56	0.00	0.00	0.00	0.00
		5	0.69	0.00	2.19	0.00	0.00	0.00	0.00	0.00
	Bedarra	2	0.06	0.00	0.19	0.00	0.00	0.00	0.00	0.00
		5	0.50	1.75	3.31	0.00	0.00	0.31	0.00	0.25
Burdekin	Palms East	2	3.38	0.00	0.00	0.00	0.00	0.00	0.00	0.00
		5	2.06	0.00	0.00	0.06	0.00	0.00	0.00	0.00
	Palms West	2	15.56	0.00	0.19	1.25	0.00	6.88	0.00	0.06
		5	11.12	0.00	2.69	0.81	0.00	2.62	0.00	0.06
	Havannah North	5	0.50	0.00	0.60	0.79	0.00	0.00	0.07	0.03
	Havannah	2	0.56	0.00	3.38	0.00	0.00	0.00	0.00	0.00
		5	0.19	0.00	5.06	0.00	0.00	0.00	0.00	0.00
	Pandora	2	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00
		5	1.50	0.00	0.00	0.00	0.00	0.00	0.00	0.12
	Pandora North	5	2.43	0.00	6.85	3.30	0.00	0.00	0.00	0.00
	Lady Elliot	2	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
		5	0.25	0.00	0.00	0.00	0.00	0.00	0.00	0.12
	Magnetic	2	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00
		5	1.19	0.00	0.19	0.00	0.00	0.00	0.00	0.06

(sub-)region	Reef	Depth	Alcyoniidae	Anthothelidae	Briareidae	Clavulariinae	Heliporidae	Neptheidae	Xeniidae	Other
Mackay Whitsunday	Hayman	5	5.13	0.00	0.23	0.00	0.00	0.00	0.00	0.00
	Border	5	17.36	0.00	0.20	0.00	0.00	0.13	0.20	0.27
	Hook	2	4.31	0.00	0.50	0.00	0.00	0.06	0.00	0.00
		5	5.89	0.06	1.94	0.00	0.00	0.00	0.00	0.00
	Double Cone	2	0.56	0.00	0.19	0.00	0.00	0.00	0.00	0.00
		5	0.44	0.00	0.31	0.00	0.00	0.06	0.00	0.00
	Daydream	2	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00
		5	0.25	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Dent	2	2.19	0.00	3.94	0.00	0.00	0.00	0.00	0.00
		5	4.50	0.00	0.56	0.00	0.00	0.00	0.00	0.00
	Shute Harbour	2	7.70	0.25	0.13	0.00	0.00	1.38	0.94	0.00
		5	4.44	0.19	0.00	0.00	0.00	0.44	0.31	0.19
	Pine	2	1.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00
		5	0.44	0.00	0.25	0.00	0.00	0.00	0.00	0.00
	Seaforth	2	1.94	0.12	1.81	0.00	0.00	0.00	0.00	0.06
		5	0.31	3.75	0.00	0.00	0.00	0.00	0.06	0.00

(sub-)region	Reef	Depth	Alcyoniidae	Anthothelidae	Briareidae	Clavulariinae	Heliporidae	Neptheidae	Xeniidae	Other
Fitzroy	Barren	2	3.44	0.00	0.06	0.00	0.00	0.00	2.25	0.00
		5	0.25	0.00	0.00	0.00	0.00	0.00	0.81	0.00
	North Keppel	2	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
		5	1.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Middle	2	0.56	0.00	0.00	0.00	0.00	0.00	0.00	0.00
		5	0.88	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Keppels South	2	0.50	0.00	0.00	0.00	0.00	0.00	0.75	0.00
		5	0.31	0.00	0.00	0.00	0.00	0.00	0.25	0.00
	Pelican	2	0.31	0.00	0.00	0.00	0.00	0.06	0.19	0.00
		5	5.26	0.00	0.00	0.00	0.00	0.44	0.00	2.44

Table A11 Percent cover of macroalgae groups 2022. Genera for which cover exceeded 0.5% on at least one reef are included, rare or unidentified genera are grouped to 'Undefined'.

			Rhodophyta (red algae)					Chlorophyta (green algae)			Phaeophyta (brown algae)							Turf Algae
(sub)region	Reef	Depth	*Acanthophora*	*Hypnea*	*Peyssonnelia*	Crustose Coralline	Undefined	*Caulerpa*	*Halimeda*	Undefined	*Dictyopteris*	*Dictyota*	*Lobophora*	*Padina*	*Sargassum*	*Stypopodium*	Undefined	
Barron Daintree	Low Isles	5	0.00	0.00	0.07	2.72	0.43	0.00	0.20	0.13	0.00	0.00	0.03	0.00	0.00	0.00	0.03	33.14
	Snapper North	2	2.17	1.65	0.46	7.33	33.77	0.04	1.96	0.12	0.00	2.13	0.00	0.04	0.00	0.00	0.17	19.42
		5	0.00	0.00	0.00	3.38	0.12	0.00	0.06	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	34.88
	Snapper South	2	0.00	0.12	0.38	3.79	2.04	0.00	0.00	0.54	0.00	0.00	0.00	0.00	0.00	0.00	0.00	40.31
		5	0.00	0.38	0.44	5.48	8.14	0.00	0.00	0.00	0.00	0.00	0.82	0.00	0.00	0.00	0.00	16.39
Johnstone Russell-Mulgrave	Fitzroy East	2	0.00	0.00	0.00	0.81	0.19	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	55.25
		5	0.00	0.00	0.06	2.88	0.69	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	35.31
	Fitzroy West	2	0.00	0.94	0.12	1.44	1.00	0.00	0.00	1.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	16.50
		5	0.00	0.12	0.19	3.12	0.88	0.00	0.00	0.88	0.00	0.00	0.00	0.00	0.00	0.00	0.00	22.88
	Fitzroy West LTMP	5	0.00	0.00	0.10	1.56	0.33	0.00	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	34.87
	Franklands East	2	0.00	0.56	0.00	1.00	0.56	0.00	0.00	0.12	0.00	0.12	0.00	0.00	0.00	0.00	0.00	27.94
		5	0.00	0.06	0.38	3.56	2.19	0.00	0.00	0.00	0.00	0.19	0.06	0.00	0.00	0.00	0.19	57.88
	Franklands West	2	0.00	3.12	0.00	1.81	7.56	0.00	0.25	0.06	0.00	0.25	0.00	0.00	0.00	0.00	0.00	19.19
		5	0.00	0.12	0.12	2.12	13.81	0.00	1.12	0.00	0.00	0.06	0.19	0.00	0.00	0.00	0.06	14.25
	High East	2	0.00	2.94	0.00	2.88	3.94	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	33.31
		5	0.00	0.19	0.12	3.81	2.25	0.00	0.00	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	36.62
	High West	2	0.00	0.69	0.00	4.69	2.69	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	26.90

(sub)region	Reef	Depth	Rhodophyta (red algae)					Chlorophyta (green algae)			Phaeophyta (brown algae)							Turf Algae
			Acanthophora	*Hypnea*	*Peyssonnelia*	Crustose Coralline	Undefined	*Caulerpa*	*Halimeda*	Undefined	*Dictyopteris*	*Dictyota*	*Lobophora*	*Padina*	*Sargassum*	*Stypopodium*	Undefined	
		5	0.00	0.00	0.00	3.45	0.69	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	46.26
Herbert Tully	Barnards	2	0.00	0.38	0.00	1.12	0.81	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.12	24.88
		5	0.00	0.00	0.13	1.00	0.56	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	20.27
	Dunk North	2	0.00	0.00	0.31	0.88	1.94	0.00	0.00	0.00	0.00	0.00	0.44	0.00	3.25	0.00	4.88	25.69
		5	0.00	0.00	0.12	1.81	1.06	0.00	0.00	0.00	0.00	0.00	0.19	0.00	0.31	0.00	0.69	25.12
	Dunk South	2	0.00	0.00	0.25	2.12	1.56	0.00	0.00	0.06	0.00	0.00	4.12	0.00	1.44	0.81	2.81	39.38
		5	0.00	0.00	0.06	1.31	1.12	0.00	0.00	0.00	0.00	0.00	6.50	0.00	0.00	0.00	0.12	37.69
	Bedarra	2	0.00	0.25	0.06	1.31	1.06	0.00	0.00	0.06	0.00	0.00	1.12	0.12	27.50	0.00	0.50	30.81
		5	0.00	0.00	0.12	0.62	0.12	0.00	0.00	0.00	0.00	0.00	0.75	0.00	0.44	0.00	0.12	32.00
Burdekin	Palms East	2	0.00	0.00	0.00	0.50	0.12	1.25	0.00	0.25	0.00	0.00	0.00	0.00	0.00	0.00	0.00	44.56
		5	0.00	0.00	0.00	0.25	0.00	0.94	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	35.25
	Palms West	2	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	39.88
		5	0.00	0.00	0.06	0.75	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	48.75
	Havannah North	5	0.00	0.00	0.37	1.06	0.20	0.83	0.03	0.43	0.00	0.17	10.83	0.13	20.69	0.00	1.19	36.49
	Havannah	2	0.00	0.00	0.31	1.44	0.31	1.00	0.00	0.06	0.00	2.25	10.00	0.00	0.81	5.50	0.69	39.12
		5	0.00	0.00	0.00	0.44	0.19	0.00	0.00	0.00	0.00	4.31	8.19	0.00	5.56	1.38	0.12	33.25
	Pandora	2	0.00	0.00	0.00	0.94	0.38	0.00	0.00	0.00	0.00	2.19	4.06	0.12	18.69	0.06	0.25	34.94
		5	0.00	0.00	0.00	0.50	0.12	0.00	0.00	0.00	0.00	6.31	0.62	0.00	0.19	0.00	0.12	55.69
	Pandora North	5	0.00	0.00	0.17	1.51	0.77	0.00	0.00	0.00	0.00	1.42	4.30	0.10	9.17	0.00	0.10	20.21
	Lady Elliot	2	0.06	8.93	0.69	1.52	3.36	0.00	0.00	0.00	0.00	4.73	0.84	0.38	1.97	0.00	0.75	31.30
		5	0.00	0.12	0.44	0.25	1.44	0.00	0.00	0.00	0.00	0.88	0.12	0.00	0.00	0.00	0.12	28.44

(sub)region	Reef	Depth	Rhodophyta (red algae)					Chlorophyta (green algae)			Phaeophyta (brown algae)							Turf Algae
			Acanthophora	*Hypnea*	*Peyssonnelia*	Crustose Coralline	Undefined	*Caulerpa*	*Halimeda*	Undefined	*Dictyopteris*	*Dictyota*	*Lobophora*	*Padina*	*Sargassum*	*Stypopodium*	Undefined	
	Magnetic	2	0.00	0.38	0.38	3.12	1.12	0.00	0.00	0.06	0.00	1.19	10.31	0.00	13.06	0.00	0.00	30.75
	Magnetic	5	0.00	0.62	1.06	0.56	4.62	0.00	0.00	0.00	0.00	4.94	2.06	0.00	15.00	0.06	0.31	27.50
Mackay Whitsunday	Hayman	5	0.00	0.00	0.10	1.27	0.73	0.00	0.00	0.00	0.00	0.00	0.13	0.00	0.00	0.00	0.27	77.40
Mackay Whitsunday	Border	5	0.00	0.00	0.00	0.17	0.00	0.00	0.00	0.03	0.00	0.00	0.03	0.00	0.00	0.00	0.00	52.56
Mackay Whitsunday	Hook	2	0.00	0.00	0.00	0.31	0.56	2.38	0.00	0.00	0.00	0.12	0.56	0.00	0.00	0.00	0.00	78.69
Mackay Whitsunday	Hook	5	0.00	0.00	0.00	0.62	0.62	1.12	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.00	54.02
Mackay Whitsunday	Double Cone	2	0.00	0.00	0.19	0.56	3.12	0.44	0.00	0.31	0.19	7.00	5.12	0.19	13.31	0.00	0.44	36.00
Mackay Whitsunday	Double Cone	5	0.00	0.25	0.25	0.38	2.00	0.00	0.00	0.06	0.06	10.31	1.81	0.25	11.00	0.12	0.38	41.69
Mackay Whitsunday	Daydream	2	0.00	0.00	0.00	0.19	15.62	0.00	0.00	0.06	3.38	0.44	5.19	2.19	11.62	0.00	2.00	29.56
Mackay Whitsunday	Daydream	5	0.00	0.00	0.00	0.06	0.50	0.00	0.00	0.06	0.00	0.56	5.44	0.06	0.12	0.00	1.06	54.50
Mackay Whitsunday	Dent	2	0.00	0.00	0.50	2.75	1.44	0.00	0.00	0.00	0.00	0.06	6.38	0.12	0.19	0.00	0.12	55.31
Mackay Whitsunday	Dent	5	0.00	0.00	0.31	2.56	0.88	0.06	0.00	0.19	0.00	0.00	4.88	0.00	0.00	0.00	0.00	52.12
Mackay Whitsunday	Shute Harbour	2	0.00	0.00	0.00	0.31	0.50	0.00	0.00	0.00	0.00	0.56	1.94	0.06	0.62	0.00	0.00	30.78
Mackay Whitsunday	Shute Harbour	5	0.00	0.00	0.06	0.19	0.12	0.00	0.00	0.00	0.00	0.25	0.62	0.06	0.12	0.00	0.00	40.00
Mackay Whitsunday	Pine	2	0.00	0.25	0.56	4.88	12.06	0.19	0.19	0.31	0.00	1.62	10.75	0.12	7.12	0.00	0.81	44.38

(sub)region	Reef	Depth	Rhodophyta (red algae)					Chlorophyta (green algae)			Phaeophyta (brown algae)							Turf Algae
			Acanthophora	*Hypnea*	*Peyssonnelia*	Crustose Coralline	Undefined	*Caulerpa*	*Halimeda*	Undefined	*Dictyopteris*	*Dictyota*	*Lobophora*	*Padina*	*Sargassum*	*Stypopodium*	Undefined	
		5	0.00	0.00	1.50	3.75	1.69	0.00	0.38	0.19	0.00	0.00	5.57	0.00	0.19	0.00	0.12	48.28
	Seaforth	2	0.00	0.62	0.19	1.31	5.63	0.00	0.12	0.12	0.00	0.69	4.32	1.00	5.19	0.00	2.69	39.28
		5	0.00	0.00	0.12	0.25	2.75	0.06	0.12	0.00	0.00	0.06	4.31	0.19	0.38	0.00	0.69	39.88
Fitzroy	Barren	2	0.00	0.00	0.00	1.69	0.62	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	32.62
		5	0.00	0.00	0.06	5.33	0.44	0.00	0.00	0.00	0.00	0.00	1.82	0.00	0.00	0.00	0.00	7.69
	North Keppel	2	0.00	0.00	0.19	4.89	0.77	0.00	0.00	0.00	0.00	0.00	21.38	0.00	0.00	0.00	0.00	25.94
		5	0.00	0.00	1.06	3.70	0.75	0.00	0.00	0.00	0.00	0.25	15.81	0.00	0.00	0.00	0.00	38.88
	Middle	2	0.00	0.00	1.44	4.44	2.25	0.00	0.00	0.06	0.00	0.00	21.31	0.00	26.12	0.00	0.00	17.12
		5	0.00	0.00	1.88	3.25	3.12	0.00	0.00	0.00	0.00	0.62	13.62	0.00	28.62	0.00	0.00	17.75
	Keppels South	2	0.00	0.06	1.94	0.69	0.69	0.00	0.00	0.00	0.06	0.50	8.69	0.12	13.19	0.00	0.06	16.57
		5	0.00	0.12	2.06	2.25	0.94	0.00	0.00	0.00	0.00	0.12	11.50	0.00	0.00	0.00	0.06	29.63
	Pelican	2	0.00	0.00	0.25	1.00	4.81	0.00	0.00	0.19	0.38	4.25	16.31	0.19	14.31	0.19	1.88	41.69
		5	0.00	0.00	0.38	0.56	2.51	0.00	0.00	0.12	0.00	1.63	9.60	0.00	1.38	0.00	0.19	31.47

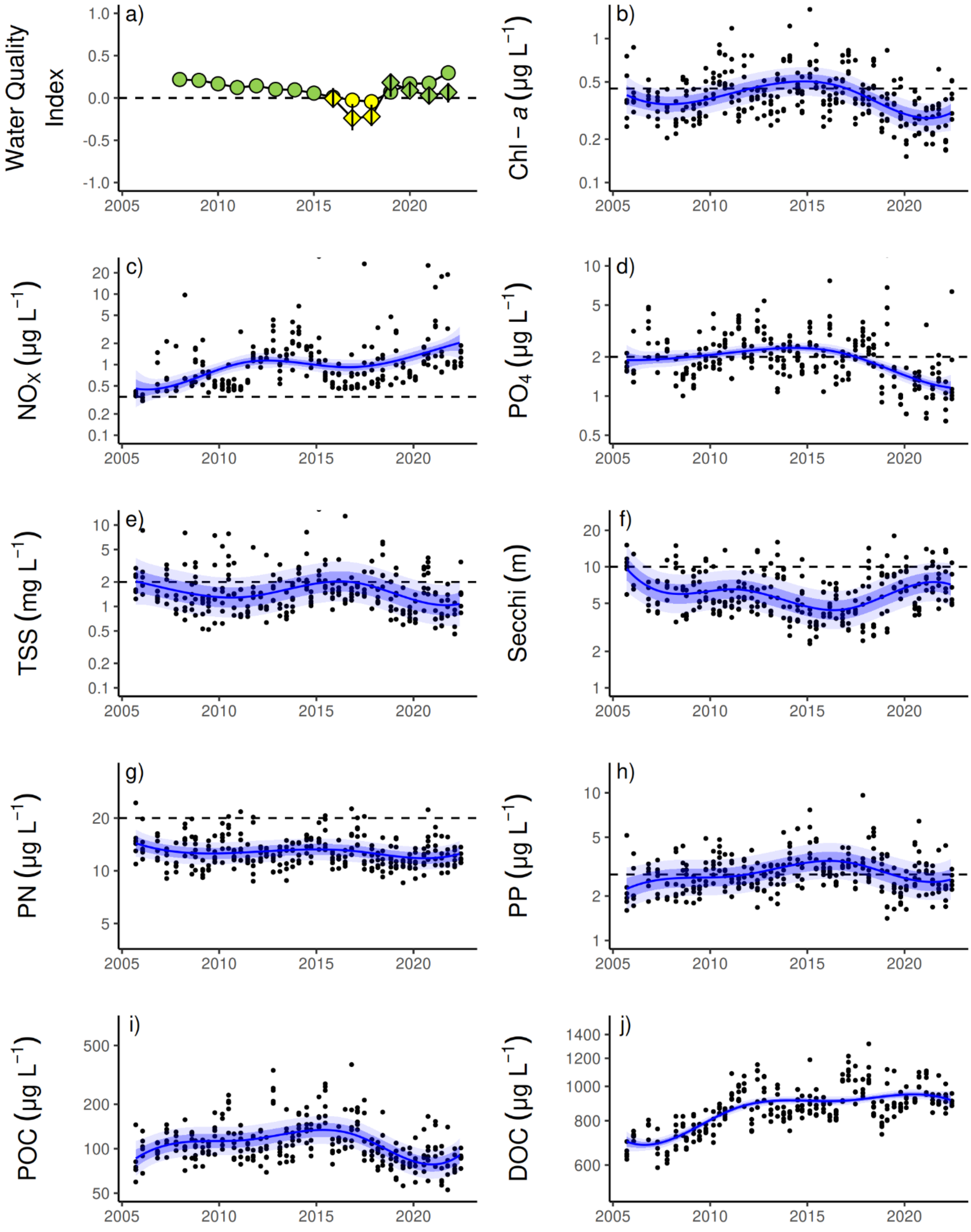

Figure A10 Temporal trends in water quality: Barron Daintree sub-region. a) water quality index, b) chlorophyll *a*, c) nitrate/nitrite, d) Phosphate, e) total suspended solids, f) secchi depth, g) particulate nitrogen, h) particulate phosphorus, i) particulate organic carbon and j) dissolved organic 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 Moran *et al*. (2023).

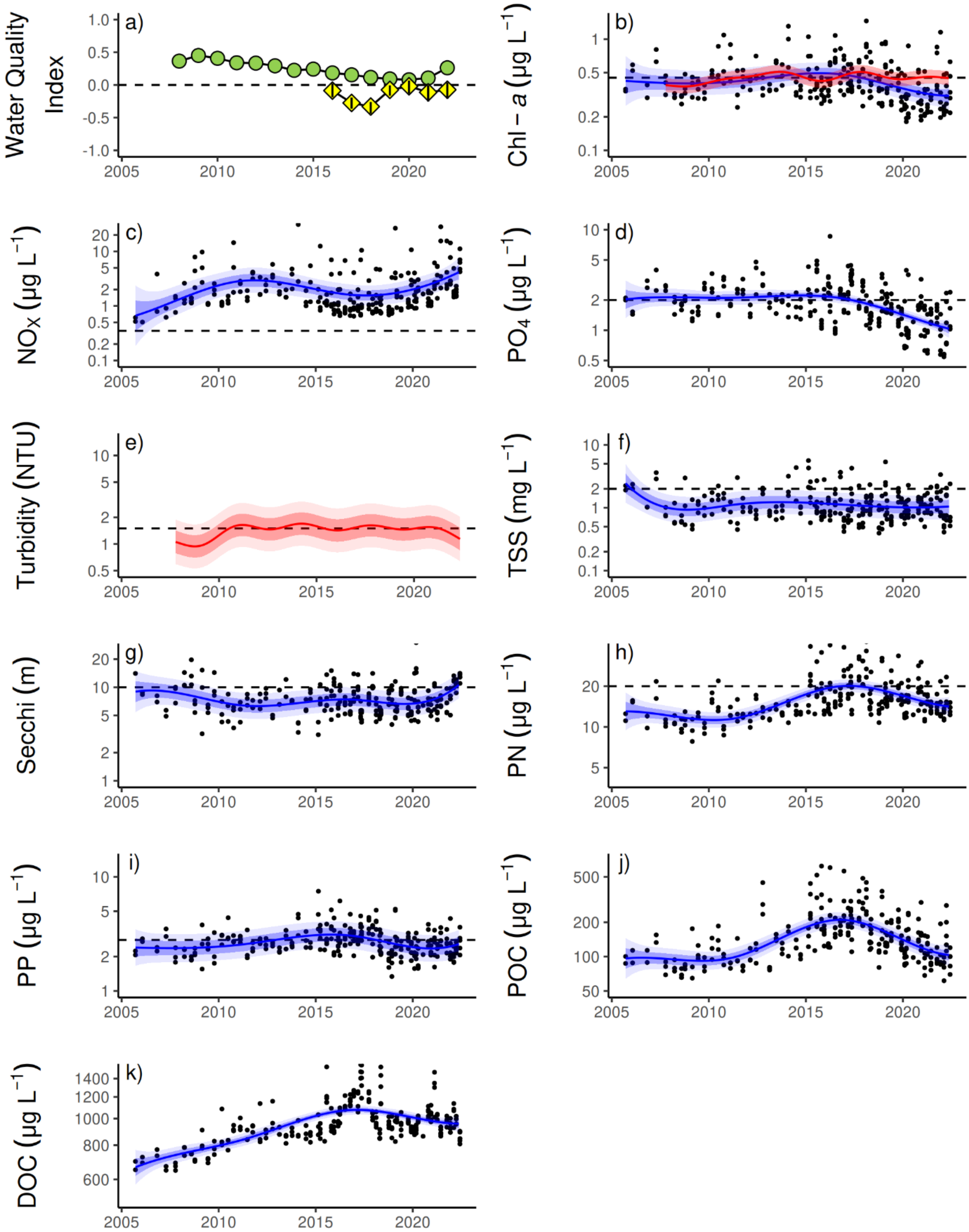

Figure A11 Temporal trends in water quality: Johnstone Russell-Mulgrave sub-region. a) water quality index, b) chlorophyll *a*, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus j), particulate organic carbon 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 Moran *et al.* (2023).

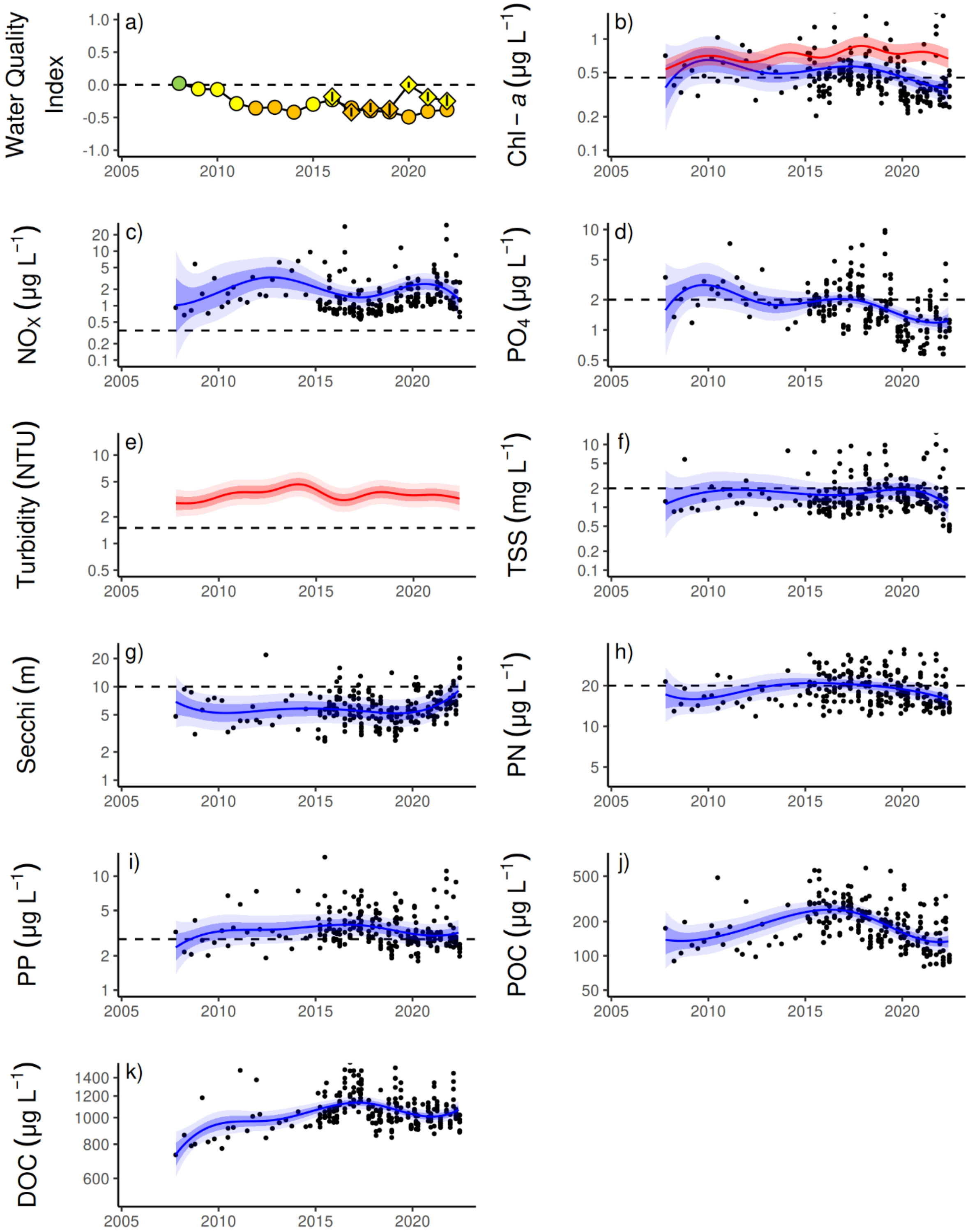

Figure A12 Temporal trends in water quality: Herbert Tully sub-region. a) water quality index, b) chlorophyll *a*, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved 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 Moran *et al.* (2023).

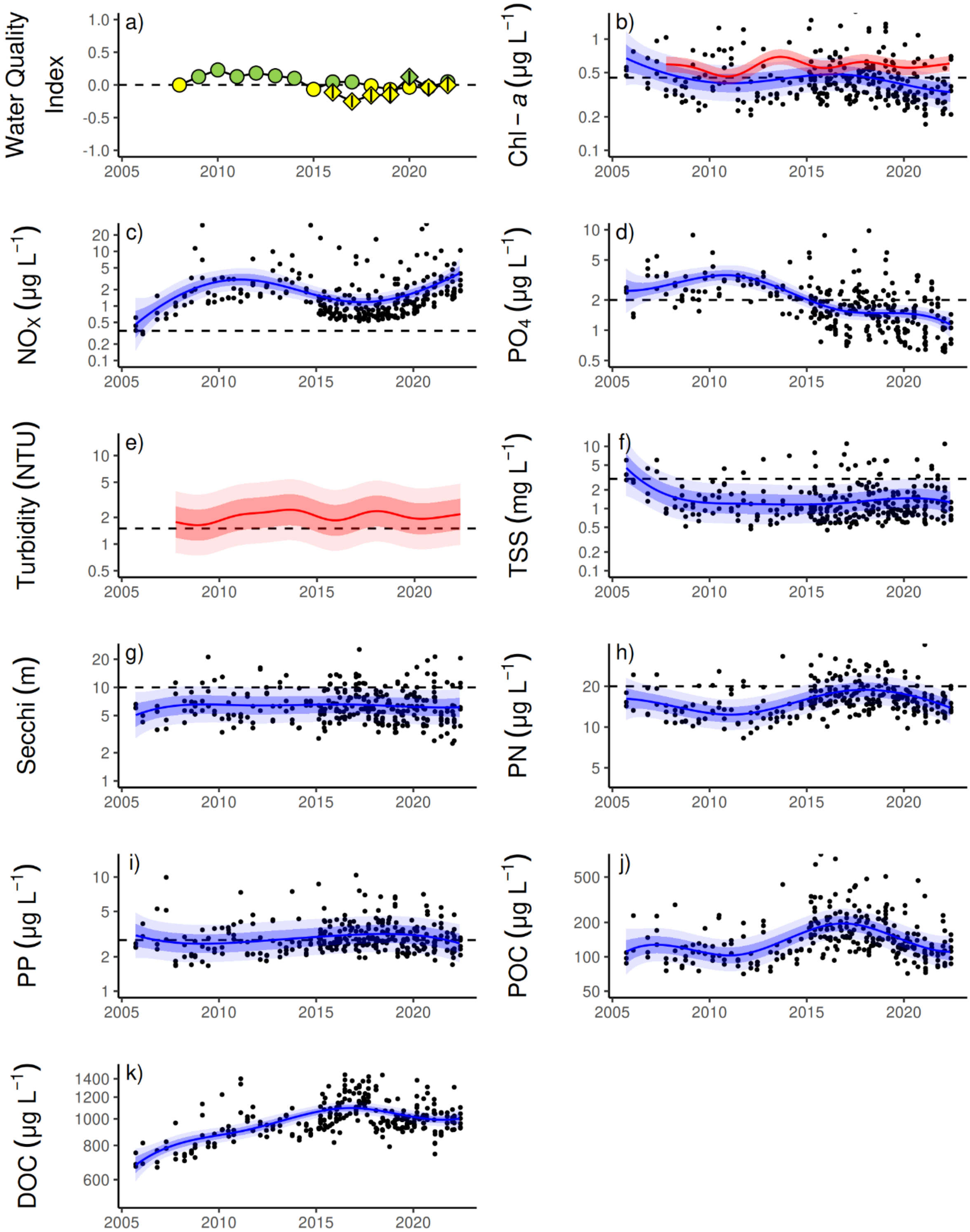

Figure A13 Temporal trends in water quality: Burdekin region. a) water quality index, b) chlorophyll *a*, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved organic 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 Moran *et al.* (2023).

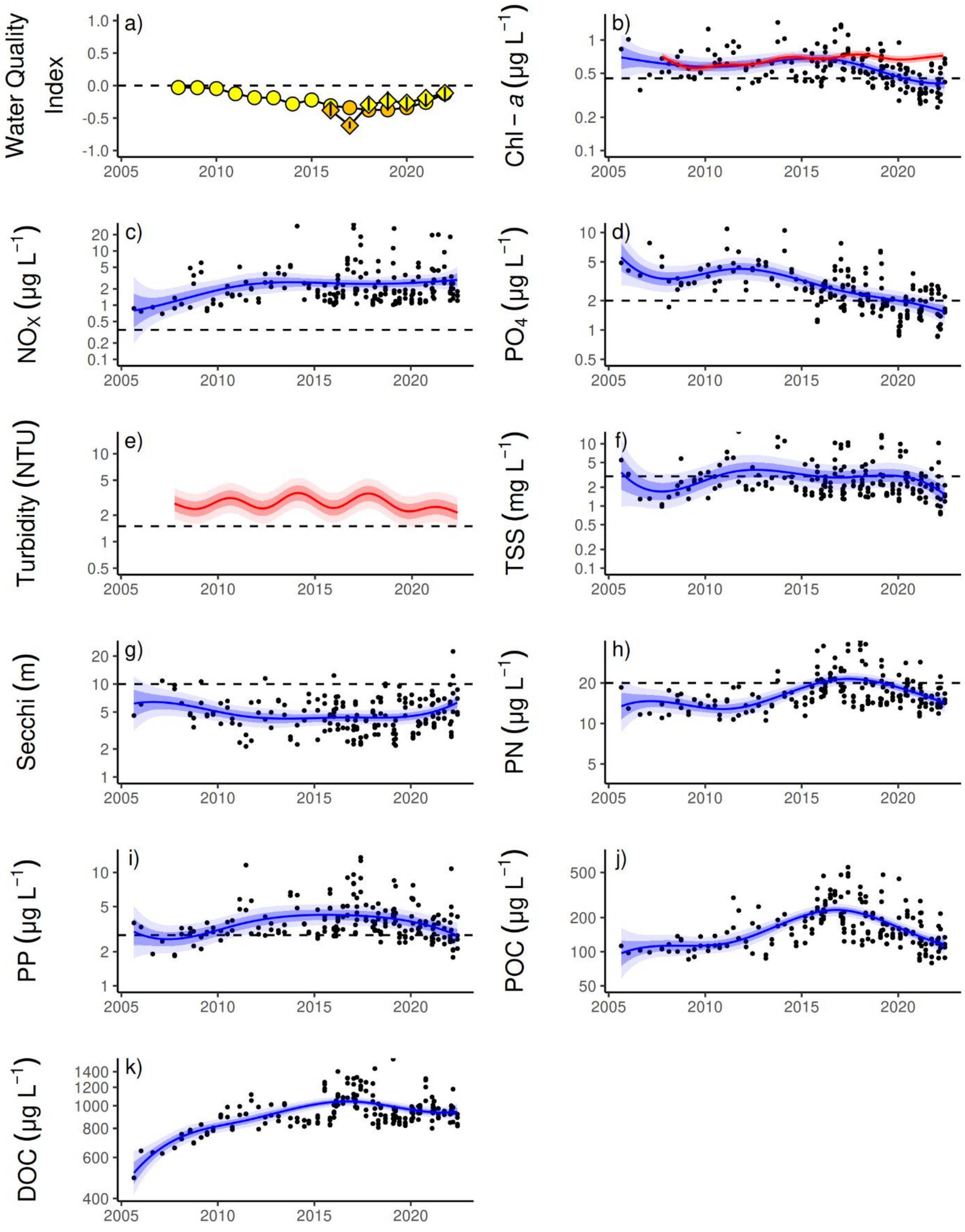

Figure A14 Temporal trends in water quality: Mackay-Whitsunday Issac region.a) water quality index, b) chlorophyll *a*, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved 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 Moran *et al*. (2023).

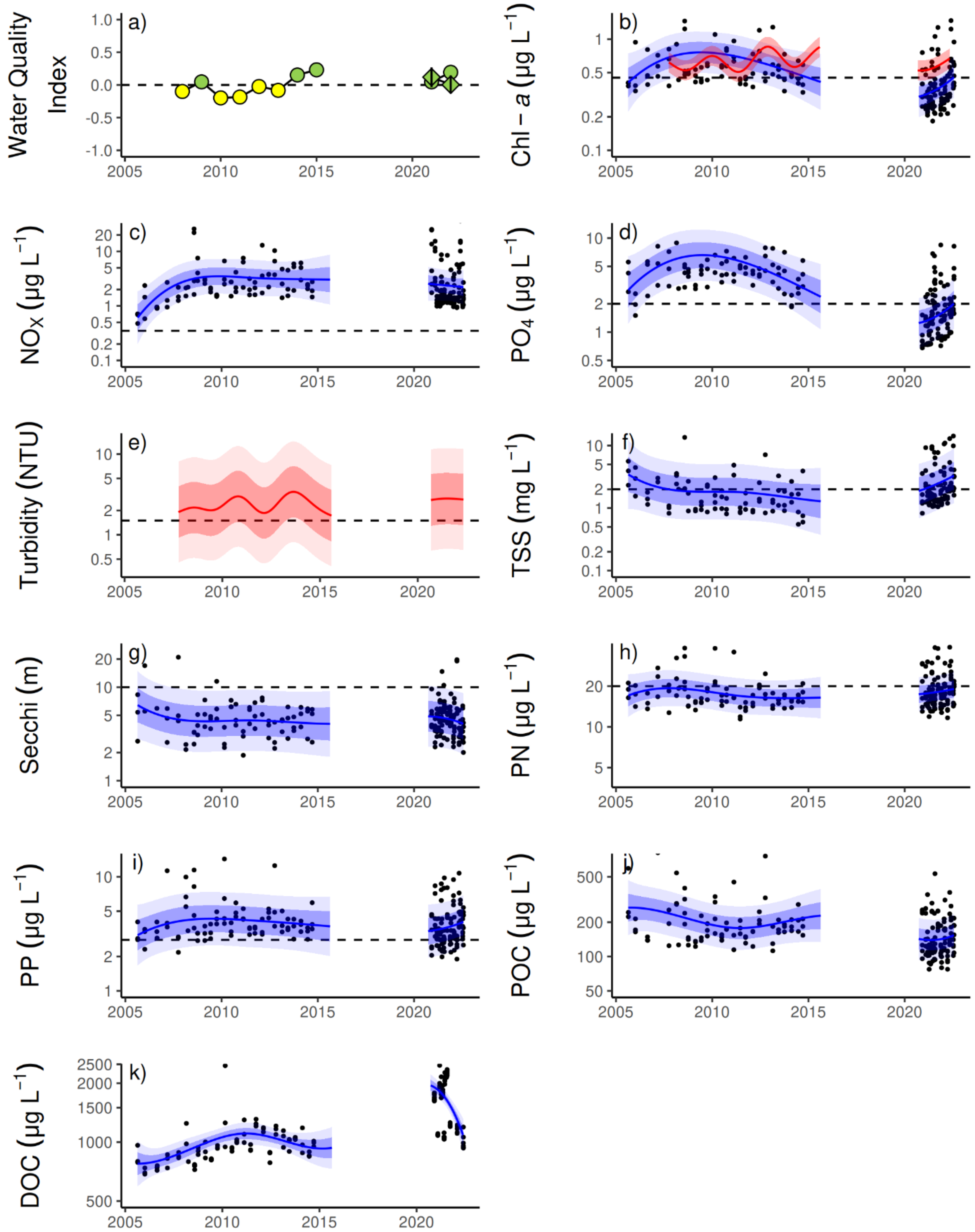

Figure A15 Temporal trends in water quality: Fitzroy region.. a) water quality index, b) chlorophyll *a*, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved organic 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 Moran *et al*. (2023).

9 Appendix 2: Publications and presentations 2021–2022

State of inshore coral communities in the Mackay Whitsunday Region. Presentation to Mackay Whitsunday Isaac P2R regional science forum 16th May 2022

State of inshore coral communities in the Tully Herbert and Burdekin regions. Presentation to Herbert River Science forum 24th May 2022

Bozec YM, Hock K, Mason RA, Baird ME, Castro-Sanguino C, Condie SA, Puotinen M, Thompson A, Mumby PJ. Cumulative impacts across Australia's Great Barrier Reef: A mechanistic evaluation. Ecological Monographs. 2022 Feb;92(1):e01494.