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Great Barrier Reef Marine Park Authority

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Annual Report

# INSHORE CORAL REEF MONITORING

2020-21

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Front cover photo: A mixed coral community is regaining cover amongst macroalgae in the shallows of Geoffrey Bay, Magnetic Island , July 2021 © Australian Institute of Marine Science, Photographer: Angus Thompson

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



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# Commonly used abbreviations and acronyms

| | |
|---|---|
| AIMS | Australian Institute of Marine Science |
| 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 valuable contributions to data collection, sampling integrity and reporting of Stephen Neale and Damian Thomson over the first few years of the monitoring program. We thank Aaron Anderson, Tom Armstrong, Rebecca Forester, Joe Gioffre, Charlotte Johansson, Sam Noonan, Kate Osborne, Shawn Smith, and Irena Zagorskis for valuable field assistance over the years. We also thank Ed Butler, Terry Done, Johanna Johnson, Katherine Martin, Bronwyn Houlden, Carol Honchin, Daniela Ceccarelli and anonymous reviewers for their detailed reviews that improved the series of reports culminating in this current document.

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 nine 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 2021 (Figure 1). The cover of macroalgae, which compete with coral for space, has begun to decline leading to a modest improvement in scores for that indicator. The legacy of coral bleaching in 2020 and ongoing impacts of crown-of-thorns starfish in the Johnstone Russell–Mulgrave sub-region have precluded improvement in other indicators (Figure 1).

No severe climate related pressures affected the inshore Great Barrier Reef (the Reef) over the 2020/2021 summer. However, the legacy of above-average seawater temperatures in early 2020, which caused severe bleaching on some reefs in the Burdekin and Fitzroy regions, likely contributed to further decline in the condition of coral reefs in those regions. Where a high proportion of corals were bleached white at the time of surveys in 2020, any subsequent loss of coral cover was attributed to the longer-term stress of this bleaching event. Corallivorous crown-of-thorns starfish were again present on reefs in the Johnstone Russell–Mulgrave sub-region, where multiple cohorts of starfish, at 'outbreak' densities were observed. 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, since 2016, the cumulative pressures imposed by cyclones, high seawater temperatures, flooding, and crown-of-thorns starfish have contributed to a period of decline. The current poor condition of inshore coral communities shows that, over the 16 years of this monitoring program, the frequency and severity of disturbances exceeded the capacity of communities to recover.

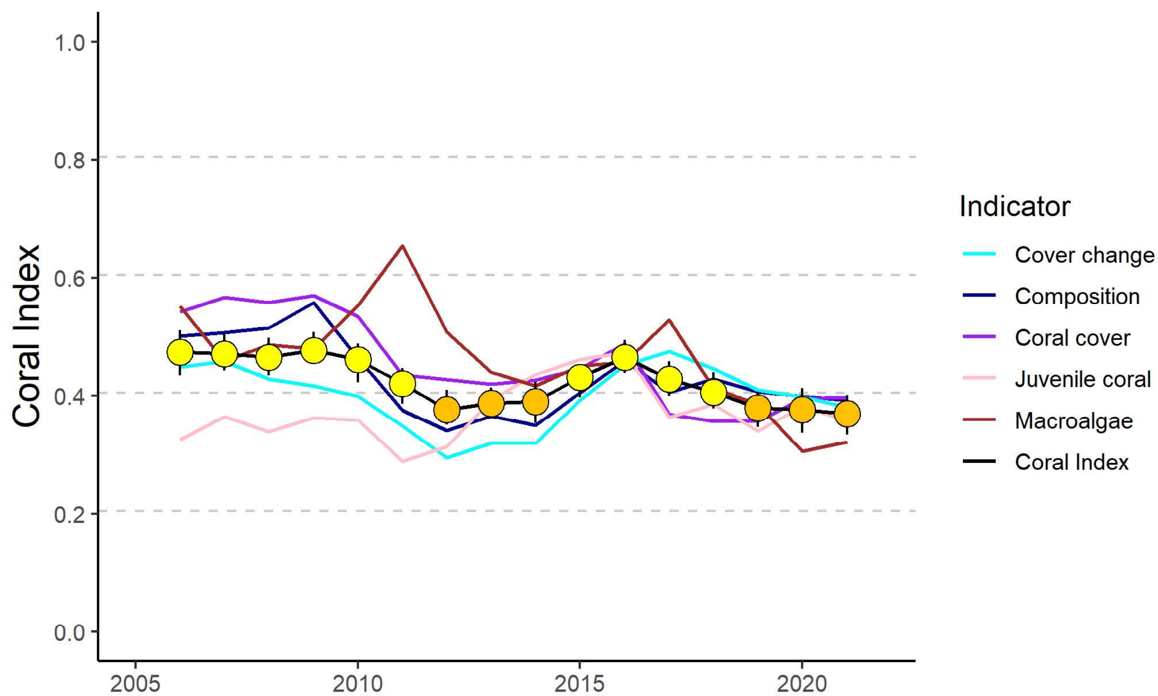


Figure 1 Trends in the Coral Index and contributing indicator scores for the inshore Reef. Coral Index scores are coloured according to Reef Water Quality Report Card categories: orange = 'poor', yellow='moderate'.

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 algal 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, coral community condition remained 'moderate'. Coral community condition has improved marginally since 2019, when flooding of the Daintree River and physical damage caused by cyclone Owen caused a slight decline. Low densities of juvenile corals and very high cover of macroalgae at shallow sites of Snapper North continue to influence scores.
- In the Johnstone Russell–Mulgrave sub-region coral community condition has fluctuated between 'moderate' and 'good' condition since 2016. This stability in condition 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 form Fitzroy Island and the Frankland Group by the Crown-of-thorns Starfish Control Program in the year prior to 2021 surveys and this will have reduced their impact on corals.
- In the Herbert Tully sub-region, coral community condition remained 'good' with coral cover recovering well since cyclone Yasi in 2011. High cover of macroalgae continues to limit the condition at Dunk Island and Bedarra Island.

## Burdekin region coral community condition

Inshore coral communities remain in 'moderate' condition. Coral communities continued to recover from a low point following the impact of cyclone Yasi in 2011. Thermal stress in early 2020 caused severe coral bleaching at most reefs. A slight decline in condition of coral communities in 2021 was attributed to the legacy of this thermal stress event.

Macroalgae scores improved but remained low at several reefs where high cover of large brown macroalgal species persists. Low densities of juvenile corals at most shallow sites continued to limit scores.

## Mackay–Whitsunday region coral community condition

Inshore coral community condition remained 'poor' although shows the first signs of recovery since the severe impact of cyclone Debbie in 2016. All indicators remain in poor condition, however indicators for macroalgae and juvenile corals improved slightly in 2021. At several severely impacted shallow sites macroalgae cover remains very high and recovery has not begun.

## Fitzroy region coral community condition

Inshore coral community condition remained ‘poor’ but has declined since 2020. Corals were severely bleached by high water temperatures in early 2020, the decline in condition through to 2021 was attributed to the longer-term impact of this event. Declines in the rate of change in coral cover and the density of juvenile corals where most influential in the decline in coral community condition.

The state of reefs varied markedly across the region. Coral cover was highest at the reef furthest from the coast, Barren Island (above 75 per cent at the 5 m depth). In contrast, coral cover remained ’poor’ or ‘very poor’ closer to the coast. Macroalgal cover remained high at almost all survey 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 community condition 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 make 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, 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. 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 pressures. To this end, the Australian Institute of Marine Science (AIMS) and the Great Barrier Reef Marine Park Authority (the Authority) have co-invested to provide inshore coral reef monitoring under the Great Barrier Reef Marine Monitoring Program (MMP) since 2005.

A key 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 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 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.

Macroalgae are monitored and reported on because they 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 recruitment of hard corals (Birrell *et al.* 2008a, Diaz-Pulido *et al.* 2010), although chemical cues from some species 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 (Fabricius 2005) and suppress coral recovery by altering microbial communities associated with corals (Morrow *et al*. 2012, Vega Thurber *et al.* 2012).

The composition of hard coral communities is monitored, as the selective pressure of water quality on coral communities is evident in 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).

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. 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 2021 Reef Water Quality Report Card. This report includes results from coral reefs monitored by AIMS as part of the MMP until July 2021 with inclusion of data from inshore reefs monitored by the AIMS Long-Term Monitoring Program (LTMP) from 2005 to 2021. 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 A 5).

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 A 1).

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 2). 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 recording temperature at 30 minute intervals until 2008 and 10 minute intervals thereafter (Table A 2).

Loggers were calibrated against a certified reference thermometer after each deployment and were 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 – 2021 | 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 – 2021 | | covariate in analysis of temporal change in Coral Index | MMP Water Quality (Moran *et al. 2022*) |
| Sea temperature | 2005 – 2021 | *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 – 2021 | remote sensing, ~4 km² 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 – 2021 | remote sensing | informing attribution of thermal stress, thermal stress maps | National Oceanographic and Atmospheric Administration |
| Cyclone tracks | 2005– 2021 | | informing attribution of storms as cause of observed coral loss, cyclone track maps | Bureau of Meteorology |
| Pollutant Exposure | 2005– 2021 | remote sensing and coupled niskin samples | covariate in analysis of temporal change in Coral Index | MMP Water Quality (Moran *et al. 2022*) |
| *Environment at coral sites* | | | | |
| Chlorophyll *a* and Total suspended solids | 2003 – 2020 | product of water colour classification derived from remote sensing and coupled niskin samples, resolution ~1 km² | Chl *a* exposure, mapping. Chl *a* and TSS concentrations covariates in analysis of variability in Coral Index score changes | MMP Water Quality (Waterhouse *et al. 2021*) |
| Non-algal particulate (NAP) | 2002 – 2018 | remote sensing adjacent to coral sites, resolution ~1 km² | Macroalgae and Composition metric thresholds, mapping | Bureau of Meteorology |
| Photosynthetically Active Radiation (PAR) | 2005 – 2020 | remote sensing adjacent to coral sites, resolution ~1 km² | covariate in analysis of spatial trends in index and indicator score, | Marites Magno-Canto (AIMS) |
| 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

Non-algal particulate (NAP) concentrations, derived from the MODIS satellite, were downloaded from the BoM[2]. For each monitoring location a square of nine 1 $km^2$ pixels were identified in closely adjacent waters from which daily medians were used to estimate monthly means. For use as a background to maps as an illustration of long-term mean gradients in conditions among monitoring sites monthly means for each pixel were aggregated to annual estimates and then long-term mean conditions (2003-2018). These data were not updated in this report as have not been available since 2019. Historically, the reef level NAP and Chl *a* estimates derived from the same source were also used to estimate thresholds for the macroalgae indicator.

Relative concentrations of TSS and Chl *a* at each reef in each year (2002-2020) were also estimated based on the methods developed by the water quality component of the MMP (Moran *et al*. 2022, Petus *et al*. 2016). In brief, MODIS aqua images were used to classify waters into one of six colour classes that range from those typical of primary (most turbid, colour classes 1–4), secondary (class 5), or tertiary (class 6) wet season water types, which reflect the influence of river discharge and resuspension events. The lowest (most turbid and nutrient rich) colour class for a given pixel was recorded as the exposure of that pixel in each week. Data for 2020/21 were not available compatibility with new estimates derived from the sentinel satellite was yet to be quantified.

It is important to note that waters can be classified into these colour classes when not exposed to flood plumes as non-plume processes, such as when wind driven resuspension produces waters with similar spectral signatures.

Water sampled from within colour classified water bodies provided mean concentrations of Chl *a* and TSS within each colour class. For each wet season (December – April, inclusive), multiplying the proportion of the wet season that each pixel was classified as a particular colour class by the concentration of Chl *a* or TSS in that colour class provided annual wet season estimates of the mean Chl *a* and TSS concentrations and also exposure to Chl *a* concentrations above wet-season guideline values (0.63 ug $L^{-1}$, GBRMPA 2010). Estimates were derived from the same nine pixels as described above for estimation of NAP concentration.

### 2.1.7 Pollutant exposure

As an extension of the colour class classifications of water bodies and measured concentrations of water quality parameters described above, annual pollutant exposure assessments were provided by the MMP water quality group (Moran *et al.* 2022). These assessments provide estimates of the proportion of coral reefs in each NRM region exposed to four categories of pollutant exposure risk. The four risk categories are based on a matrix of the cumulative magnitude of exposure to above Reef-wide guideline values for Chl a, particulate nitrogen, particulate phosphorus and total suspended solids and the frequency of exposure. Although the resulting risk categories remain untested in terms of actual risk to corals, they do provide a standardised summary of observed variability in water quality at the regional scale.

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[2] *Marine water quality indices produced by the Australian Bureau of Meteorology as a contribution to eReefs - a collaboration between the Great Barrier Reef Foundation, Australian Government, Bureau of Meteorology, Commonwealth Scientific and Industrial Research Organisation, Australian Institute of Marine Science and the Queensland Government. Data were acquired from NASA spacecraft. Note that this product has been discontinued.*

### 2.1.8 Light available for photosynthesis

The estimates of Chl a and TSS, or NAP, described above quantify the relative exposure to nutrients and suspended sediments. These and other optically active components of the water column interact to reduce the penetration of light at wavelengths necessary for photosynthesis (photosynthetically active radiation, PAR). As a proxy for relative light attenuation at the coral sites, daily estimates of PAR at 8 m depth below mean tide height were estimated based on an algorithm applied to MODIS aqua images (Magno-Canto *et al*. 2019) and extracted from the same pixels as used for the NAP and Chl *a* estimates. For each reef, annual water-year estimates (October through September) were derived as the mean of daily estimates capped at 16 µmol $m^{-2}$ $d^{-1}$.

## 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 2, 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 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 from 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 2) 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 2 Coral monitoring locations. 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. Shading indicates discontinued reefs. 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 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 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 2021. .

## 2.3 Coral community sampling methods

Two sampling methodologies were used to describe the benthic communities of inshore coral reefs (Table 3).

Table 3 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). 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 the proportion 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 Keppels 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).

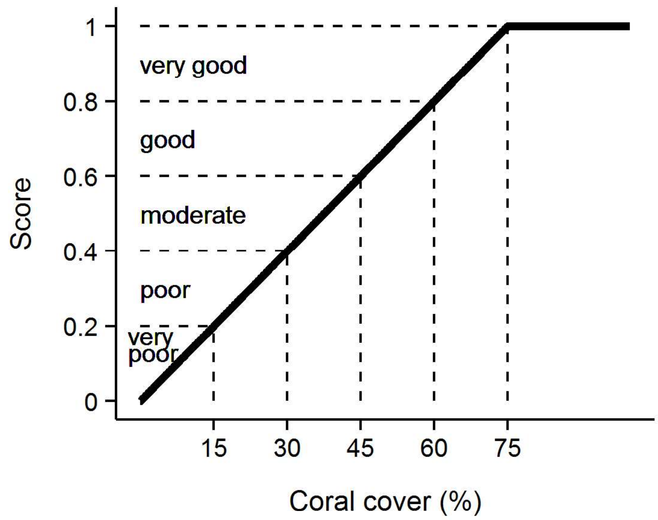


Figure 3 Scoring diagram for the coral cover indicator metric. Numeric scores and associated condition classifications based on observed coral cover are presented.

### 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:

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

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 A 3). 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).

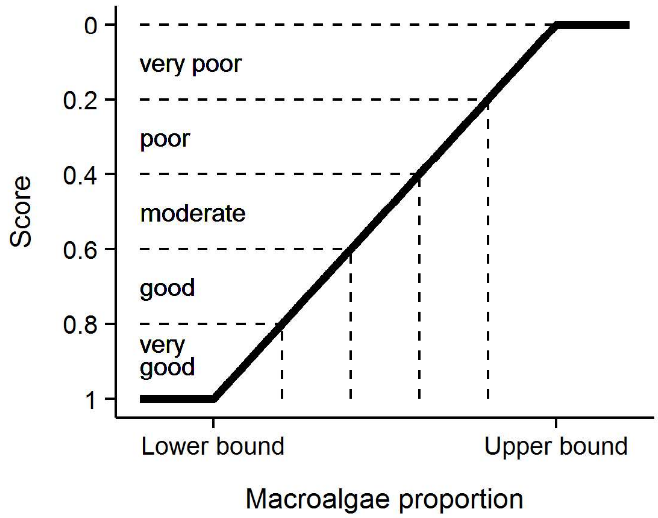


Figure 4 Scoring diagram for the Macroalgae indictor metric. Upper and lower threshold values are reef and depth specific. Numeric scores and associated condition classifications are presented. Note that for this metric the y-axis is inverted as high values reflect poor condition.

### 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)

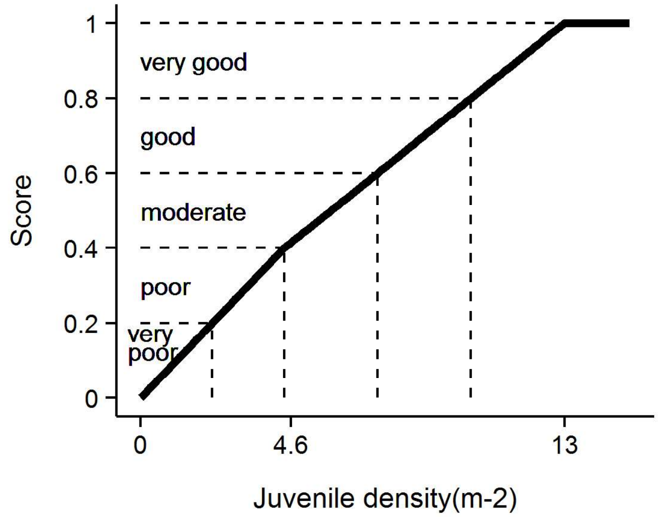


Figure 5 Scoring diagram for the Juvenile coral indicator metric. Numeric scores and associated condition classifications are presented.

### 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 form 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.

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

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

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

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

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

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

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

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

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

Note, the above 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.

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.

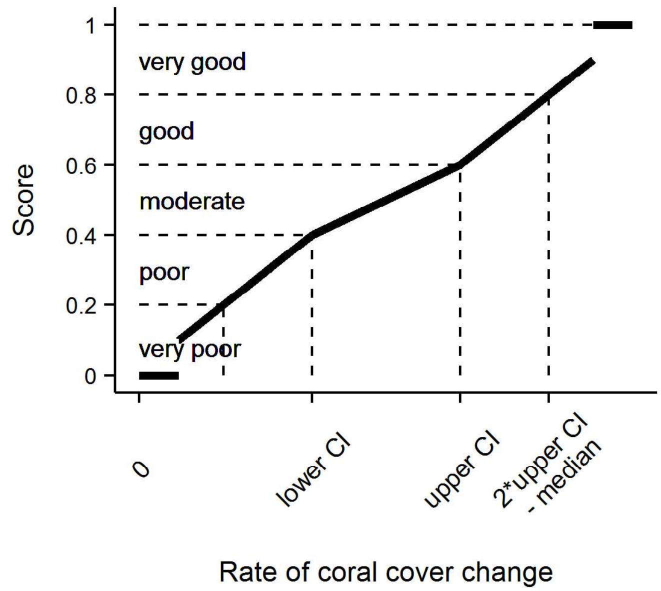


Figure 6 Scoring diagram for cover change indicator metric.

### 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 normal temperature (Hoegh-Guldberg 1999) or 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, because of 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. 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.

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).

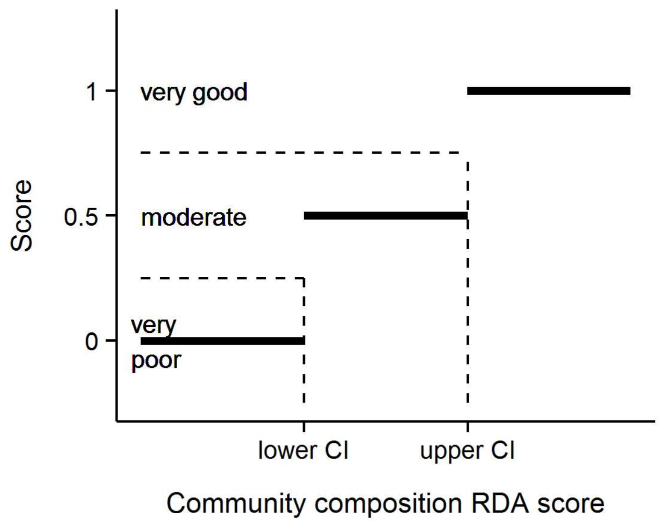


Figure 7 Scoring diagram for the composition indicator metric

### 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 4. 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 4 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 5).

Table 5 Format for presentation of community condition.

| Section | Scope | Scale | Covariates | Analyses/Presentation |
|---|---|---|---|---|
| 4.14.1 | Temporal trend in coral community condition | Reef | Major disturbances | Relative influence of major pressures over the time-series |
| 4.3, 4.4, 4.5, 4.64.3 | 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 (section 2.1.2),
- pollutant exposure (section 2.1.7).

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 there 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 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 A 1 to Figure A 6). 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 A 9 to Table A 11.

### 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 6) 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 6 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 2020-21

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

## 3.1 Cyclones

Tropical cyclones frequently cross the inshore Reef.

Over the 2020-21 reporting period the only cyclone likely to have produced damaging waves to the Reef was cyclone Kimi (Figure 8). This system entered the Reef off Cairns at category 1 intensity on the 18$^{th}$ of January 2021 and briefly intensified to category 2 as it tracked south over the day. By the 19$^{th}$ of January cyclone Kimi had degraded to a tropical low and tracked back north before dissipating off Cairns the following day.

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 A 3). 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 A 4)
- cyclone Debbie (2017) caused severe coral loss on reefs in the Mackay-Whitsunday region (Figure 8, Table A 6). There are yet to be clear signs of recovery of coral communities in the wake of this cyclone.

Numerous smaller cyclones have crossed the inshore Reef over the last decade (Figure 8) causing more moderate and localised damage (Table A 6, see also ((sub-)regional summaries section 4.3).

## 3.2 Sea temperature

Sea temperatures over the 2021 summer were slightly above long-term averages (Figure 9). However, temperatures remained below published thresholds of 60-100 degree heating days (Garde *et al.* 2014) or 4 degree heating weeks (NOAA 2018) that are likely to lead to significant coral bleaching. In contrast, high temperature stress occurred in 2020 and 2017 (Figure 9, Figure 10). 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).

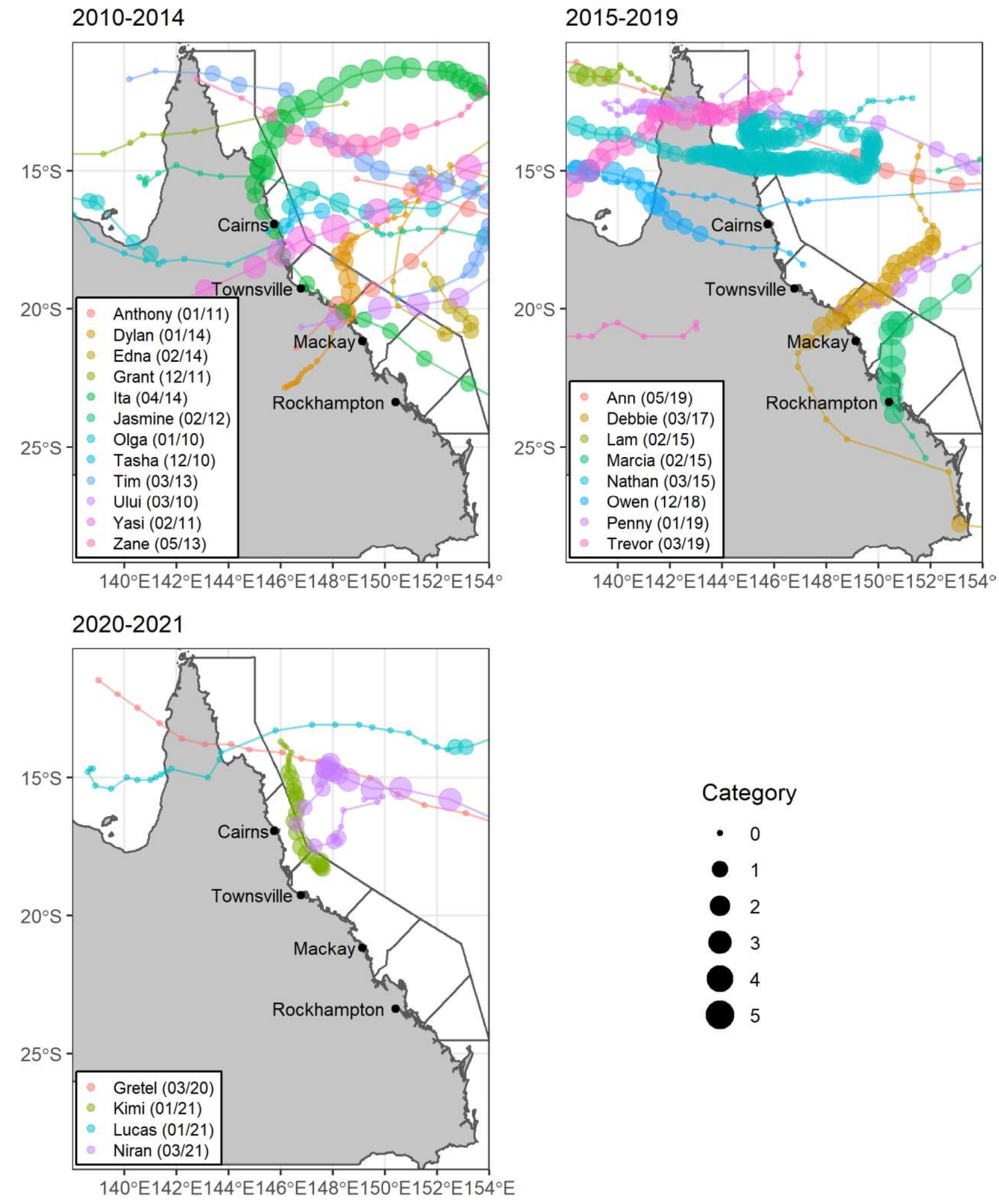


Figure 8 Cyclone tracks for systems crossing the inshore Reef over the last decade. Tracks sourced from the Bureau of Meteorology.

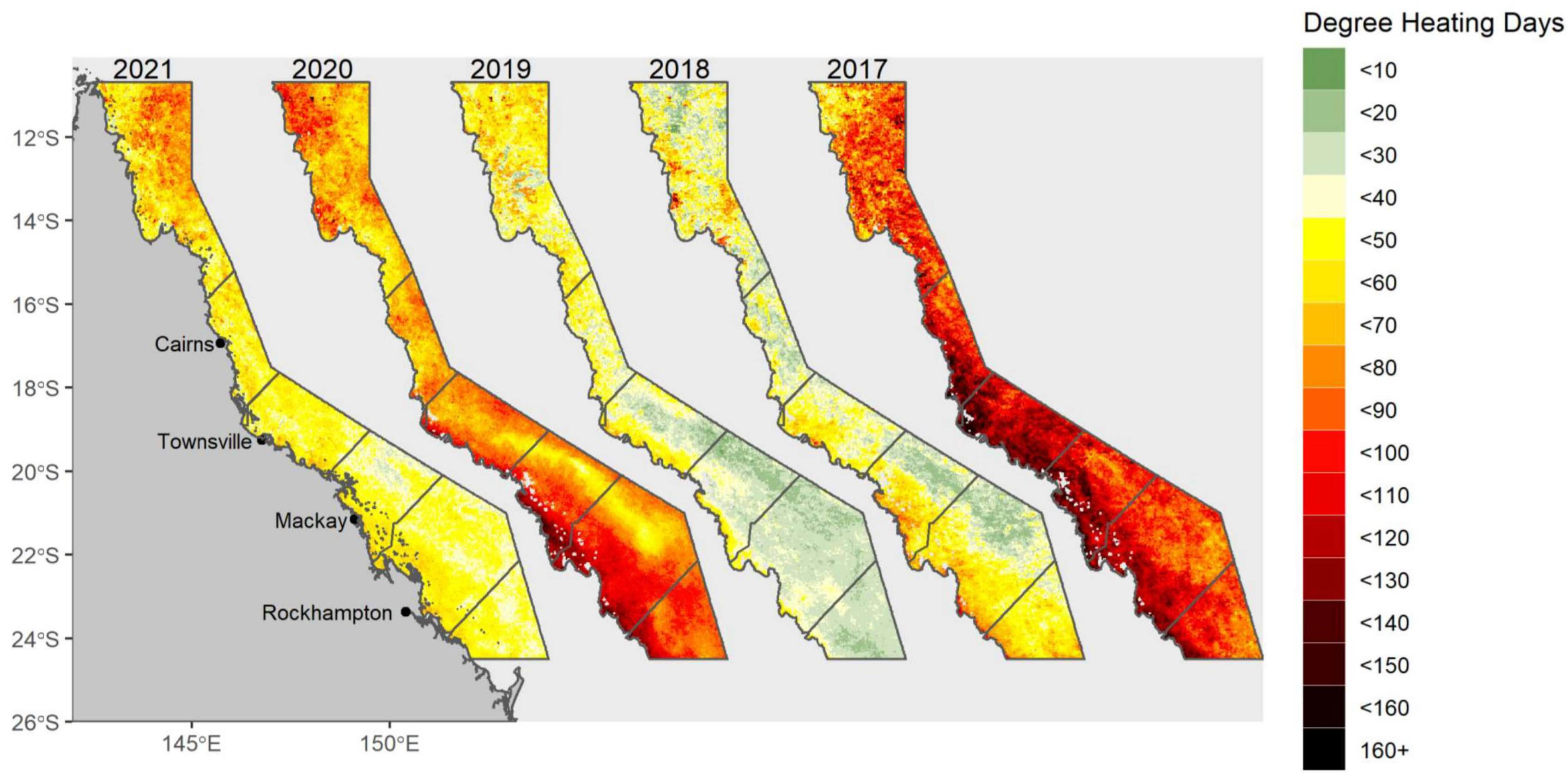


Figure 9 Annual degree heating day estimates for the Reef. Data are the annual degree heating day accumulations over the summer period (1 December to 31 March) for ~4 km$^2$ pixels based on temperatures exceeding 14 Day IMOS climatology. Data were sourced from the Australian Bureau of Meteorology ReefTemp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

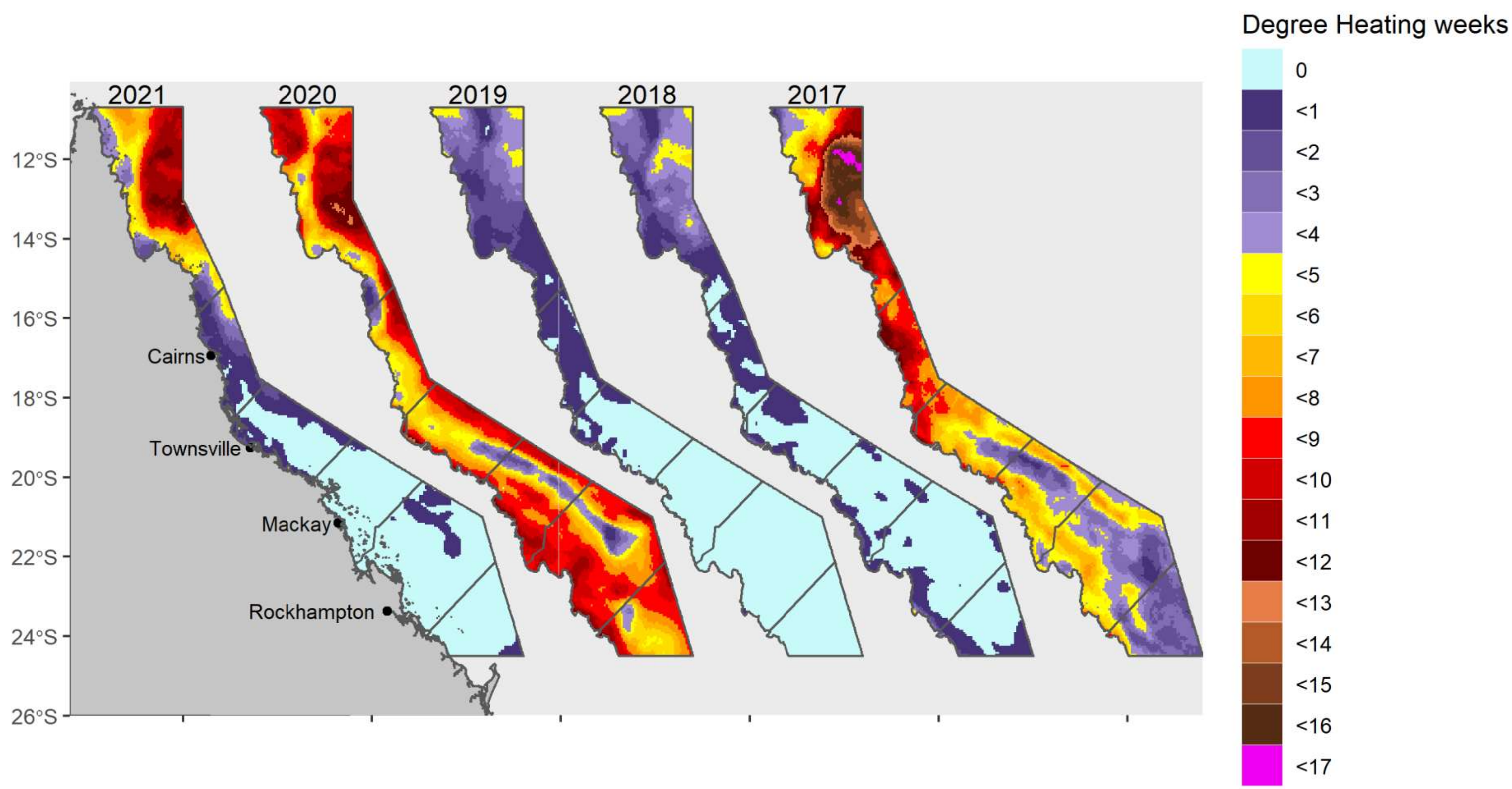


Figure 10 Annual degree heating week estimates for the Reef. Data are the annual maximum degree heating week estimates for each ~25 km$^2$ pixel. Date were sourced from NOAA coral reef watch.

## 3.3 Crown-of-thorns starfish

In 2021 the density of crown-of-thorns starfish remained a little above outbreak levels at Frankland Group East, Frankland Group West and High East, although numbers were substantially lower than observed in 2020. In 2021 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 on mid-shelf reefs south of Mackay (Pompey Sector 2020, Swains Sector 2021).

Since 2012 crown-of-thorns starfish have remained present on reefs in the Johnstone Russell-Mulgrave sub-region, with numbers increasing to outbreak levels (> 30 individuals per hectare) at five of the six reefs monitored in 2020 (Figure A 8). The crown-of-thorns starfish both observed by the MMP and removed by the Authority's control program ranged across several size cohorts indicating the ongoing recruitment and survival of crown-of-thorns starfish over recent years (Table 8). Similar recruitment of crown-of-thorns starfish was observed by the AIMS LTMP on mid-shelf reefs of the Innisfail sector, where juvenile and sub-adult crown-of-thorns starfish were noted during surveys in February 2020.

Table 7 Number of crown-of-thorns removed by the Crown-of-thorns Starfish Control Program. Figure 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** (1.1) | **6831** (3.36) |

Table 8 Size class distribution of crown-of-thorns starfish on inshore reefs in the Wet Tropics. Included are the percentages culled, as listed in Table 7, of cohorts 1-4, and percentage in size-classes observed by the MMP.

| 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 | 5-15 cm | 15-25 cm | >25 cm |
| 2012 | | | | | **54** | 40 | 2 |
| 2013 | 24 | **35** | 31 | 10 | 15 | **57** | 28 |
| 2014 | 12 | **42** | 36 | 10 | **57** | | 43 |
| 2015 | **41** | 39 | 16 | 4 | **50** | 17 | 33 |
| 2016 | **95** | 4 | 0 | 0 | **67** | 33 | |
| 2017 | **75** | 23 | 2 | 0 | **55** | 45 | |
| 2018 | 43 | **51** | 6 | 0 | 14 | 36 | **50** |
| 2019 | **84** | 14 | 2 | 0 | 33 | **67** | |
| 2020 | 24 | **62** | 13 | 1 | 27 | **49** | 24 |
| 2021 | 17 | **66** | 16 | 1 | 6 | 25 | **69** |

## 3.4 River discharge

Discharge in 2021 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 A 1). 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. Closer to the coast the authors' personal observations were that corals at Virago Shoal off the coast of Townsville were killed by floods of the Ross River, while corals along the eastern face of Cape Cleveland were killed by the plume of the Burdekin and / or Haughton rivers.

In previous years, the most extensive flood damage to monitored reefs occurred in 2011 in the Fitzroy region when there was very high mortality of corals at 2 m depths on reefs to the south of Great Keppel Island (Table A 6, Figure A 6). As at 2021 recovery from this event was occurring at Keppels South but limited, at best, at Pelican Island.

The influence of sediment and nutrient loads are not as overtly obvious as those associated with exposure 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.

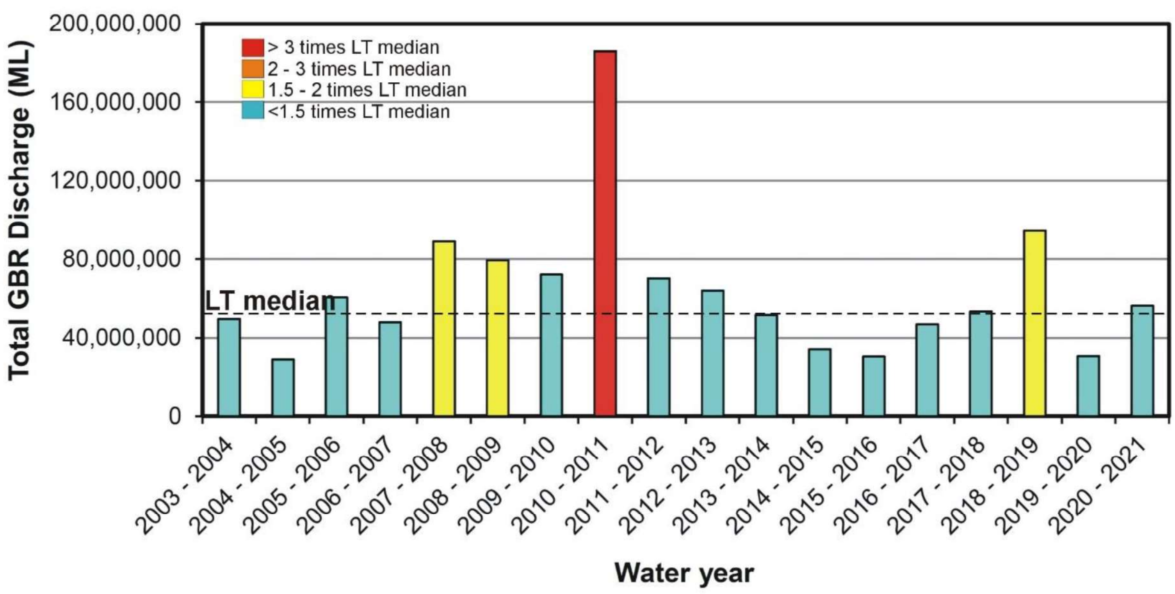


Figure 11 Annual total river discharge to the Reef. Annual estimates aggregate over the water year: 1 October to 30 September, for the 35 main Reef basins. Values are colour coded relative to proportion of long-term (LT) median (1986-2016) discharge. Figure source: Moran *et al*. 2022, data source: DNRM, 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 and trend (4.1)
- Reef-wide relative impact of disturbances (4.2)
- Coral community condition and trend in each(sub-)region (4.3 - 4.6)
- Response of coral communities to location 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 Reef scale, the Coral Index score remained largely unchanged from that observed in 2020 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 7, Table A 5).

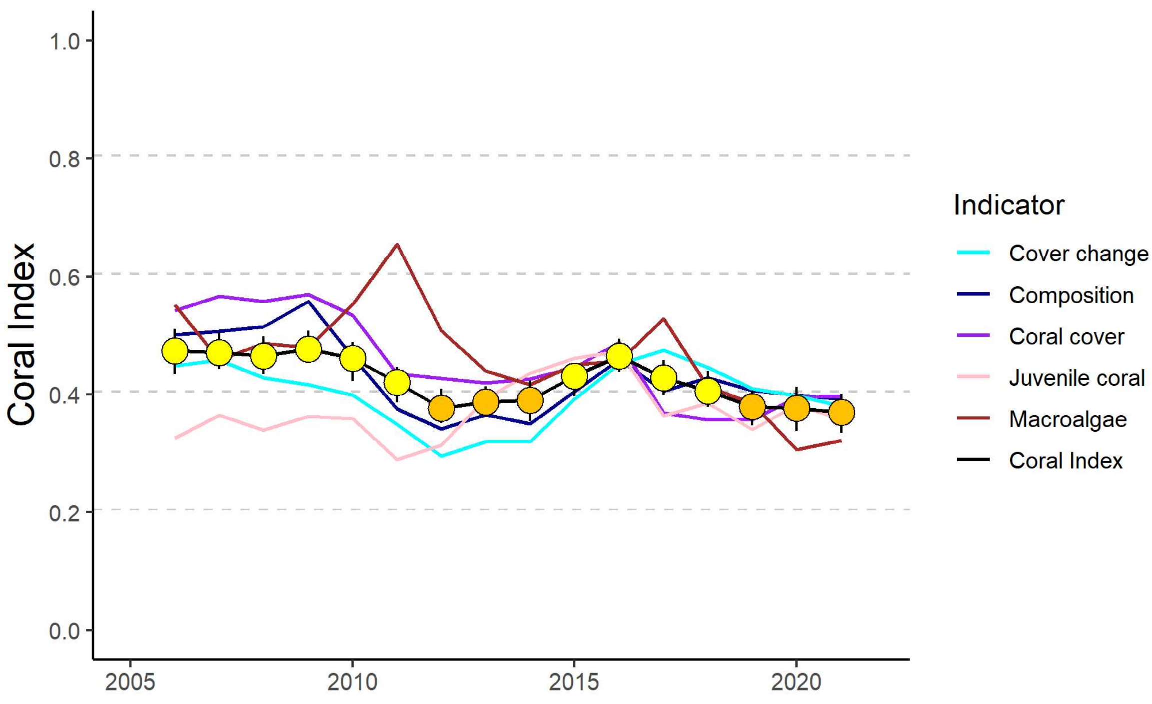


Figure 12 Reef level trend in Coral Index and indicator scores. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = ‘poor’, yellow=’moderate’. Error in Coral Index scores were derived from bootstrapped distribution of regional indicator scores weighted by the relative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

The recovery of coral communities between 2013 and 2016 demonstrated the inherent resilience of inshore coral communities. That the current condition is again low is unsurprising given the level of pressure imposed in recent years. A slight decline in the Coral Index in 2021 reflects small declines in scores for the juvenile coral and cover change indicators, both potentially linked to the stress imposed by marine heat wave conditions experienced in 2020. The ongoing presence of crown-of-thorns starfish in the Wet Tropics region continues to limit improvement in Coral Index scores. The macroalgae indicator scores, although showing a slight improvement from their lowest point in 2020, remain comparatively low with macroalgae continuing to put a downward pressure on coral community recovery.

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. 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 have caused almost half (46%) of all coral cover losses on inshore reefs (Figure 13, Table A 6). Unsurprisingly, the intense category 4 and 5 systems; cyclone Larry (Wet Tropics and Burdekin regions – 2006), cyclone Yasi (Wet Tropics and Burdekin regions – 2011), and cyclone Debbie (Whitsunday region – 2017) have caused the greatest losses. Changes in the composition indicator scores (Figure 12) following acute disturbances indicate that it is species sensitive to poor water quality (primarily *Acropora,* Table A 4) that are disproportionately impacted by these events.

When interpreting Figure 13 is important to note that the biennial sampling design 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 three 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 2020 (Figure 13). In these instances corals were still bleached at the time of surveys in 2017 and 2020 and the subsequent loss of cover attributed to the delayed response to thermal stress.

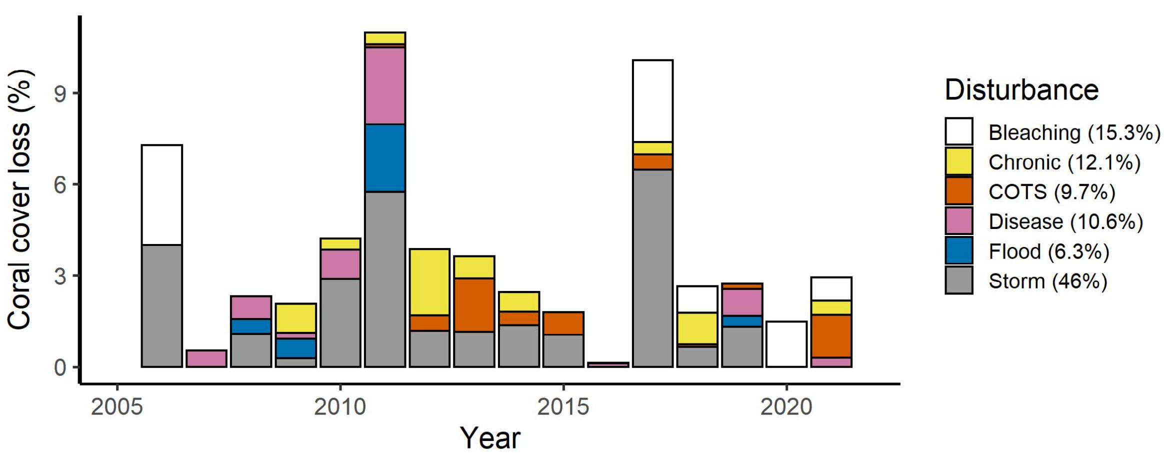


Figure 13 Hard coral cover loss by disturbance type across the inshore Reef. Length of bars represents the mean loss of cover across all reefs in each year. Colours represent the identified cause of cover loss.

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, and 2020 (Figure 13, Table A 6); the loss indicated in 2018 is due to some impacted reefs not being surveyed in 2017. This is not the case for bleaching impacts attributed in 2021 as all reefs that were sampled in 2021 were sampled in 2020. 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 was 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 in 2017, 2018 and 2021 were also influenced by stress imposed by high water temperatures.

While crown-of-thorns starfish have caused moderate losses (9.7%, Figure 13, Table A 6), their potential impact has been reduced by the removal of starfish by the Crown-of-thorns Starfish Control Program (Table 7). 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. 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 Coral Index scores in all regions.

The losses of coral cover attributed to disease and chronic pressures (22.7%, 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. High scores for the cover change indicator in recent years demonstrate the ongoing capacity for coral cover to rebound following disturbance events. The stable over-all condition, however, reflects a range of minor disturbances that have variously impacted reefs among the sub-regions, as detailed in the following sections. At the regional level, no indicator scores have fallen below moderate levels since 2014.

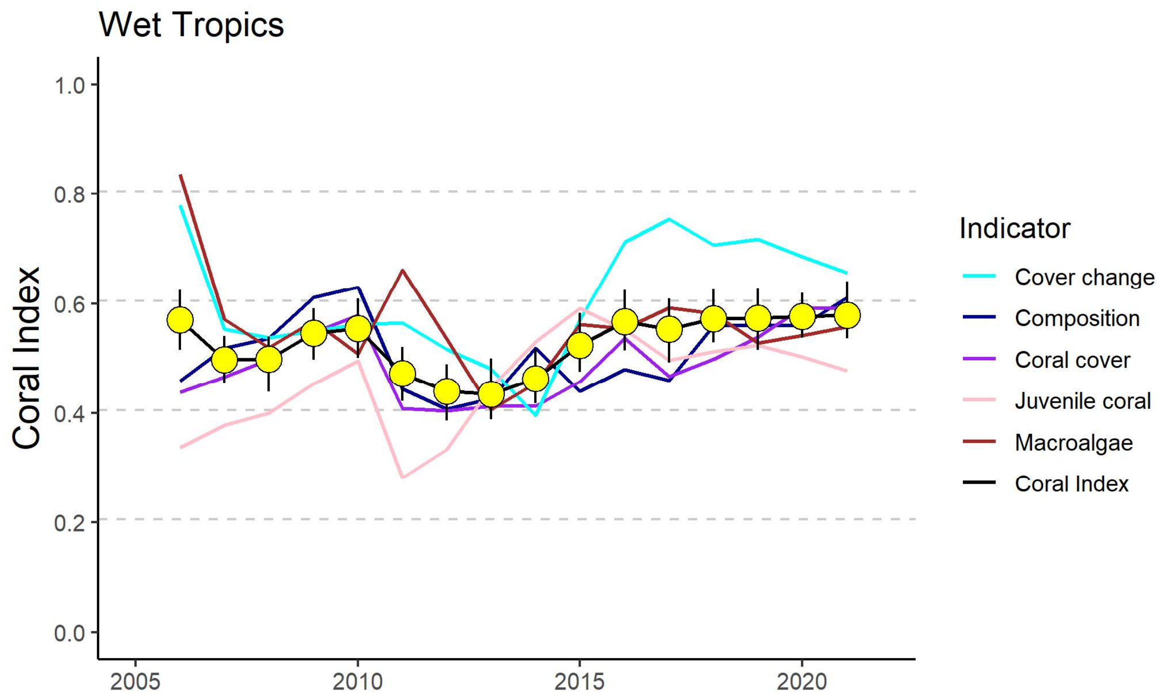


Figure 14 Trends in Coral Index and indicator scores for the Wet Tropics region. Coral Index scores are coloured by report card category: yellow=’moderate’ Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at individual reefs.

### 4.3.2 Wet Tropics region: Barron Daintree sub-region

The coral community condition remains ‘moderate’. 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 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 flood waters and cyclone Owen in 2019 (Figure 16, Figure 17, Table A 6).

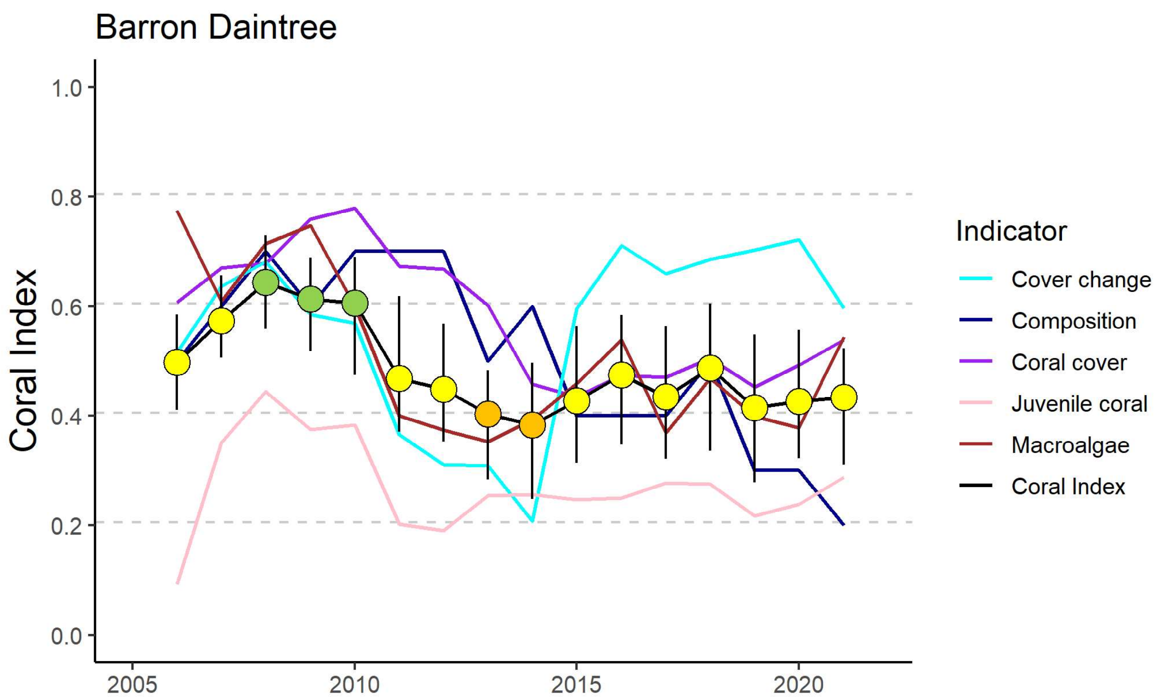


Figure 15 Trends in Coral Index and indicator scores for the Barron Daintree sub-region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = ‘poor’, yellow=’moderate’ and green=’good. Error in Coral Index scores were derived from bootstrapped distributions of indicator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

Table 9 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 | 0.99 | -0.50 | 1.00 |
| 2014 to 2021 | 2 | -0.07 | 0.72 | 0.11 | 0.80 | -0.05 | 0.76 | -0.07 | 0.70 | 0.64 | 0.98 | -1.00 | 1.00 |
| | 5 | 0.13 | 0.72 | 0.06 | 0.60 | 0.29 | 0.75 | 0.10 | 0.72 | 0.22 | 0.68 | 0 | 0 |

Most limiting to current Coral Index scores were scores for the composition and juvenile coral indicators. At 5 m depths the composition scores remain the same as they were in 2014 (Table 9) with hard coral communities having similar composition to those observed at the beginning of the program at Low Isles and Snapper South but having reduced representation of Acroporidae and Agariciidae at Snapper North (Figure A 1, Table A 7). At 2 m depths low scores for the composition indicator reflect disproportionate loss of Acroporidae corals in contrast to increased cover of Faviidae (Figure A 1), genus *Echinopora* at Snapper North (Table A 9Table A 9).

Except for a strong cohort of juvenile *Acropora* observed at 2 m depth at Snapper Island South in 2008, the density of juvenile corals at Snapper Island has been consistently low (Figure 17c, Figure A 1). The slight increase in juvenile coral scores in 2021 relative to 2014 at 5m depth (Table 9) does however capture marginally higher densities at Snapper North and Low Isles in recent years (Figure A 1).

The decline in the macroalgae score at 2 m depth since 2014 is due to the continued very high cover of macroalgae at Snapper North (Figure A 1). At 5 m depths macroalgae scores have improved since 2014 (Table 9) reflecting declines in cover of macroalgae at Snapper Island over recent years (Figure A 1).

Despite high freshwater discharge (Figure 16d) and associated high loads of nutrients and sediments delivered from adjacent catchments (Moran *et al.* 2022), the water quality index returned to ‘good’ condition in 2019 where it has remained in 2021 (Figure A 10). Not included in the water quality index are concentrations of dissolved organic carbon and oxidised nitrogen species (NOx), both of which show substantial increases since 2005 (Figure A 10). It remains unclear what has caused these increases (Moran *et al. 2022*) or what the ramifications for corals might be.

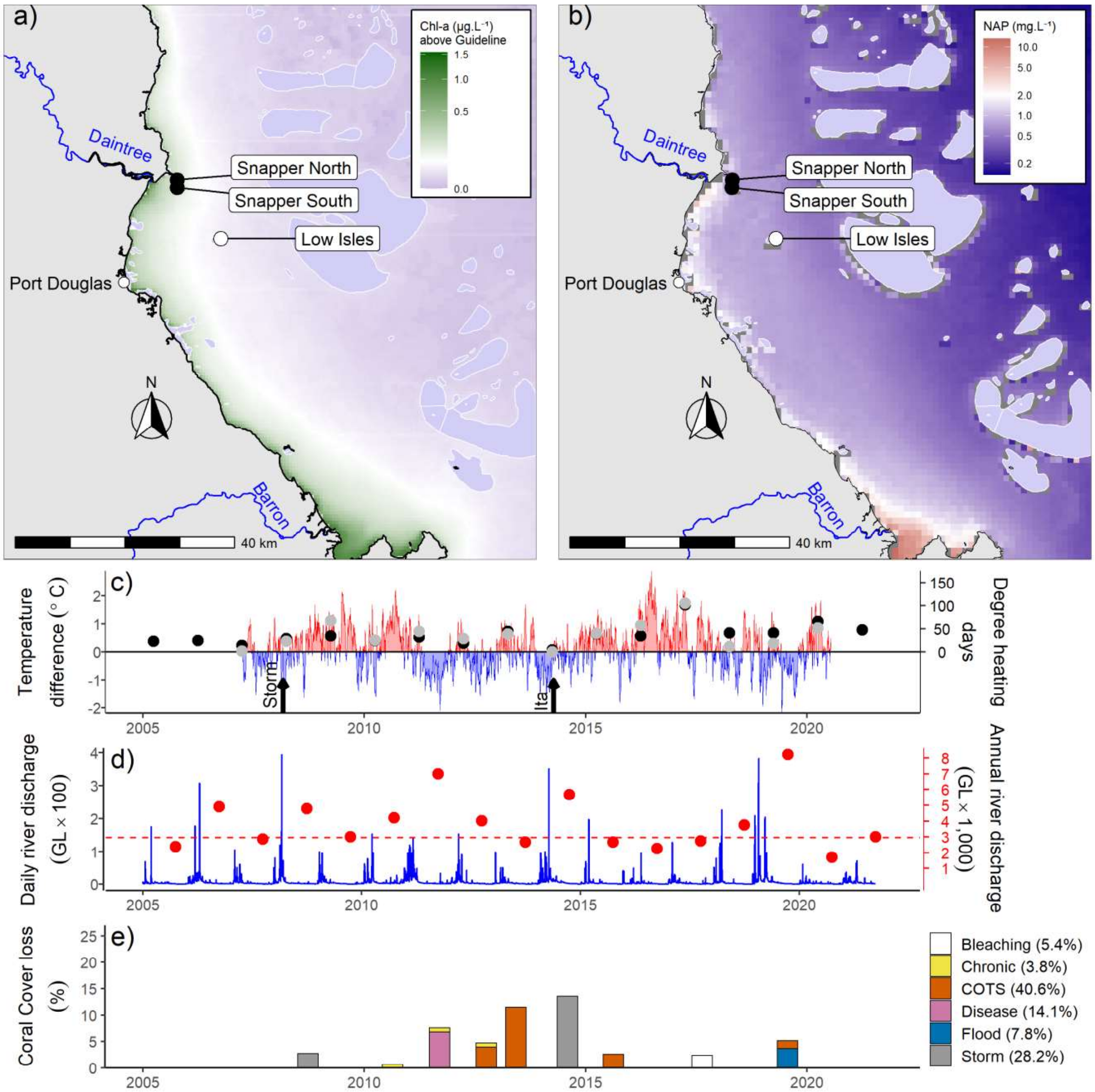


Figure 16 Barron Daintree sub-region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) mean chlorophyll *a* exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are mean levels 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.

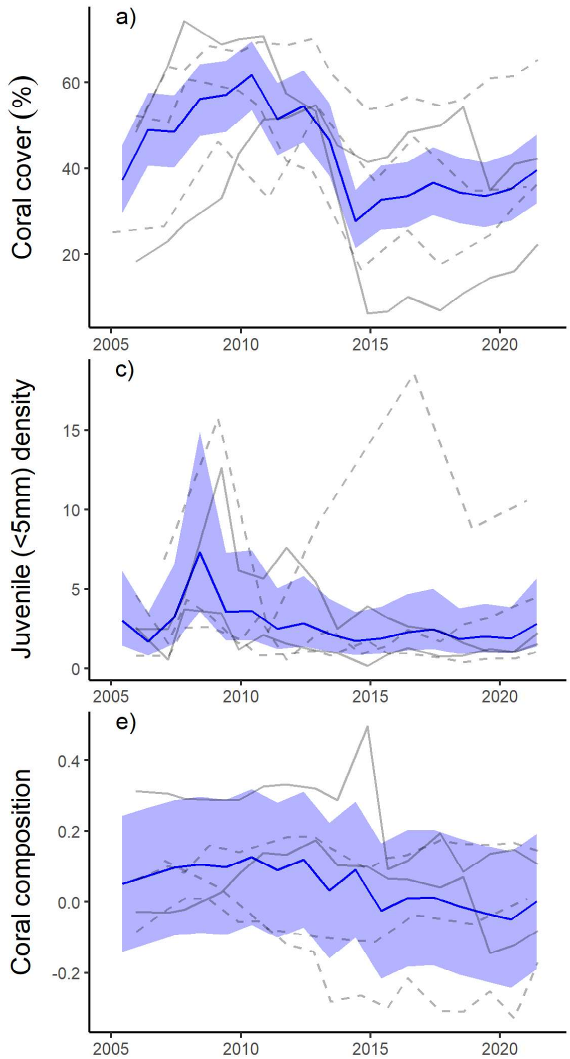


Figure 17 Barron Daintree sub-region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

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

The 2021 Coral Index score was categorised as 'good', with a marginal increase since 2020 (Figure 18).

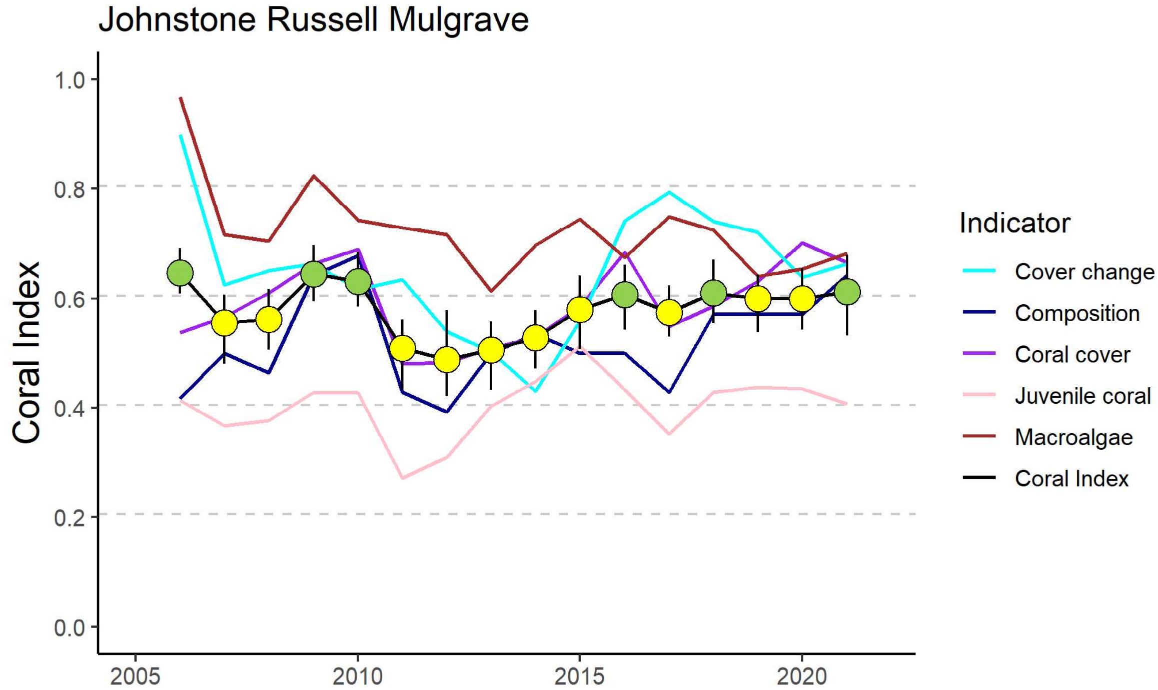


Figure 18 Trends in Coral Index and indicator scores for the Johnstone Russell-Mulgrave sub-region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = 'poor', yellow='moderate' and green='good. Error in Coral Index scores were derived from bootstrapped distributions 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 2021 (Figure 18). The slight increase in 2021 reflected improvement in the cover change, macroalgae and composition indicators in contrast to small declines in scores for coral cover and juvenile coral. Declines in hard coral cover at Fitzroy East, Franklands East 5m, High East and High West 5m (Figure A 2) were attributed to predation by crown-of-thorns starfish (Figure 19e). When not subjected to crown-of thorns starfish outbreaks hard coral cover continued to increase at or above expected rates (Figure 20d).

Crown-of-thorns starfish populations peaked in 2012 (Figure A 8), with a higher mean density than 2020, and were the primary cause of coral loss at Fitzroy Island and Green Island over the period 2012-2015 (Figure 19e, Figure A 2). Although the number of crown-of-thorns starfish observed during surveys in 2021 were lower than in 2020 (Figure A 8) this result is likely influenced by the large number removed by the Crown-of-thorns Starfish Control Program (Table 7). This removal will have substantially reduced the loss of coral cover at Fitzroy Island and in the Frankland Group.

Macroalgae scores remain good, with generally low cover of the persistent brown macroalgae typical of many inshore reef regions (Table A 11). At Franklands West and to a lesser degree High East 2 m red macroalgae form dense mats among corals (Table A 11) leading to low scores for macroalgae in those locations (Table A 7Table A 7), the cover of these algae continues to be highly variable among years (Figure 20b)

The juvenile coral indicator remains relatively low and declined into the 'poor' range in 2021 (Figure 18).

Table 10 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 2021 | 2 | 0.15 | 0.82 | 0.26 | 0.88 | 0.01 | 0.51 | 0.05 | 0.92 | 0.07 | 0.60 | 0.33 | 090 |
| | 5 | 0.09 | 0.76 | 0.12 | 0.84 | -0.07 | 0.57 | 0.14 | 0.95 | 0.07 | 0.58 | 0.19 | 0.78 |

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 A 2, Table A 6). The effects of cyclones were further compounded by the increased prevalence of disease in 2011 (Figure 19e, Table A 7). 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 A 6, Figure A 2). This outbreak of disease coincided with high discharge from local rivers (Figure 19d). The low point in the Coral Index reached in 2012 reflects decline in the cover change score in 2012 compounding reductions in other indicator scores in direct response to the cyclones in 2011 (

Table 10, Figure 20). The plateau in recovery of the coral communities in recent years has been influenced by thermal bleaching in the 2016-2017 years, when up to 23% of the cover of hard corals was lost at individual reefs (Figure 20a, Figure A 2), and ongoing predation of corals by crown-of-thorns starfish (Figure 19e, Figure A 8).

Discharge from rivers in the sub-region were at, or slightly above, median levels over the 2021 water year (Table A 5), however, peak flows remained relatively low (Figure 19d). No evidence of direct impacts due to exposure to low salinity plume waters was observed at High West. In 2020 and prior to 2018, annual discharge was at, or below, median levels since 2012 (Figure 19d) and under these conditions the coral communities demonstrated a clear recovery when not exposed to thermal stress or crown-of-thorns starfish.

The long-term water quality index in this sub-region has remained ‘good’ since monitoring began in 2008, with a slow decline stabilising in 2020 and 2021 (Figure A 11). This appears to be a balance between increasing nitrogen and turbidity, and declining levels of chlorophyll *a*, phosphate, and particulate carbon (Figure A 11). In 2021 most water quality parameters were near or below the guideline values (Figure A 11). Concentrations of dissolved and particulate organic carbon show substantial increase since 2005 (Figure A 11). It remains unclear what has caused this increase and what the ramifications for corals might be (Moran *et al. 2022*).

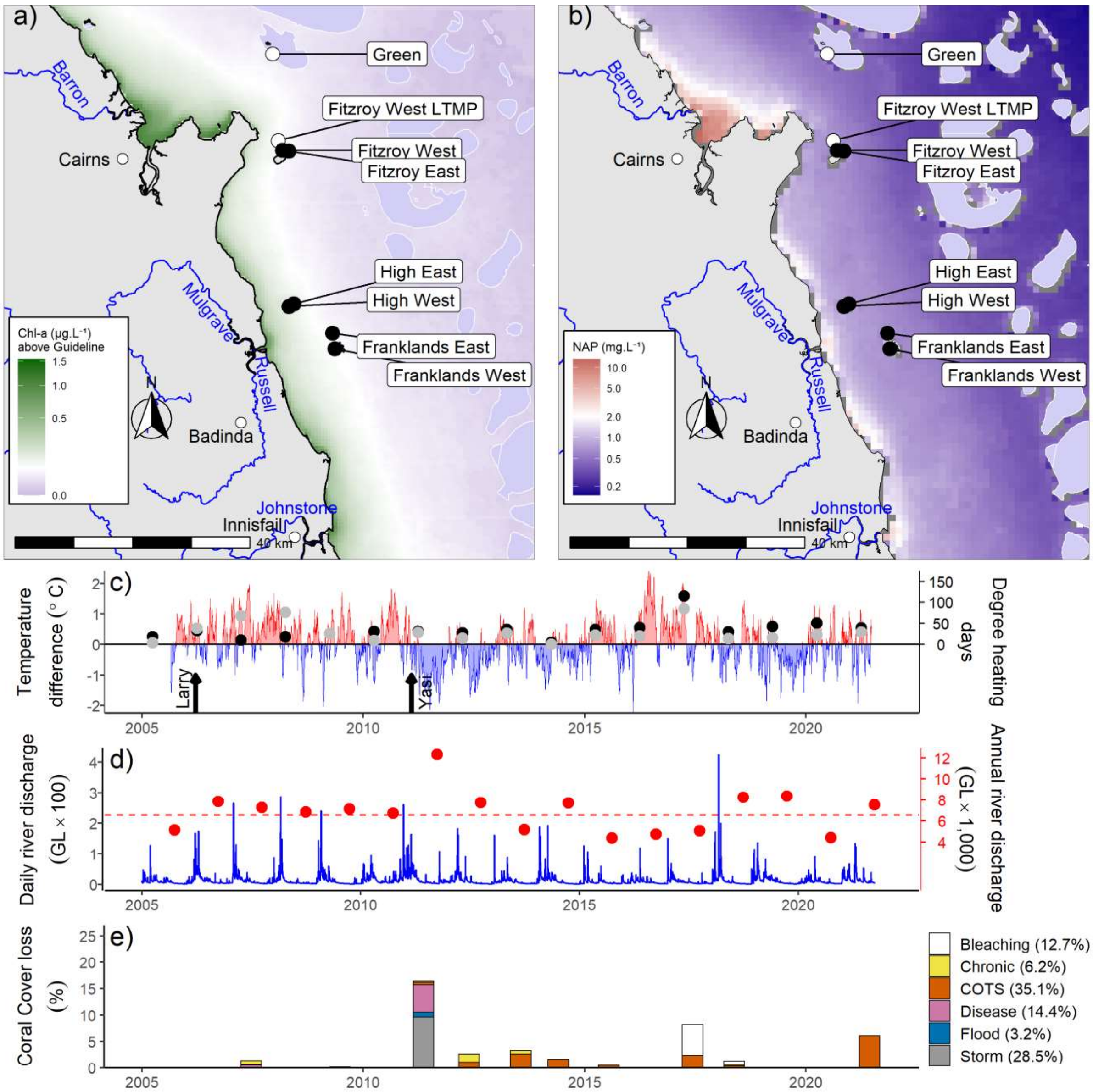


Figure 19 Johnstone Russell-Mulgrave sub-region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) mean chlorophyll *a* exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data 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.

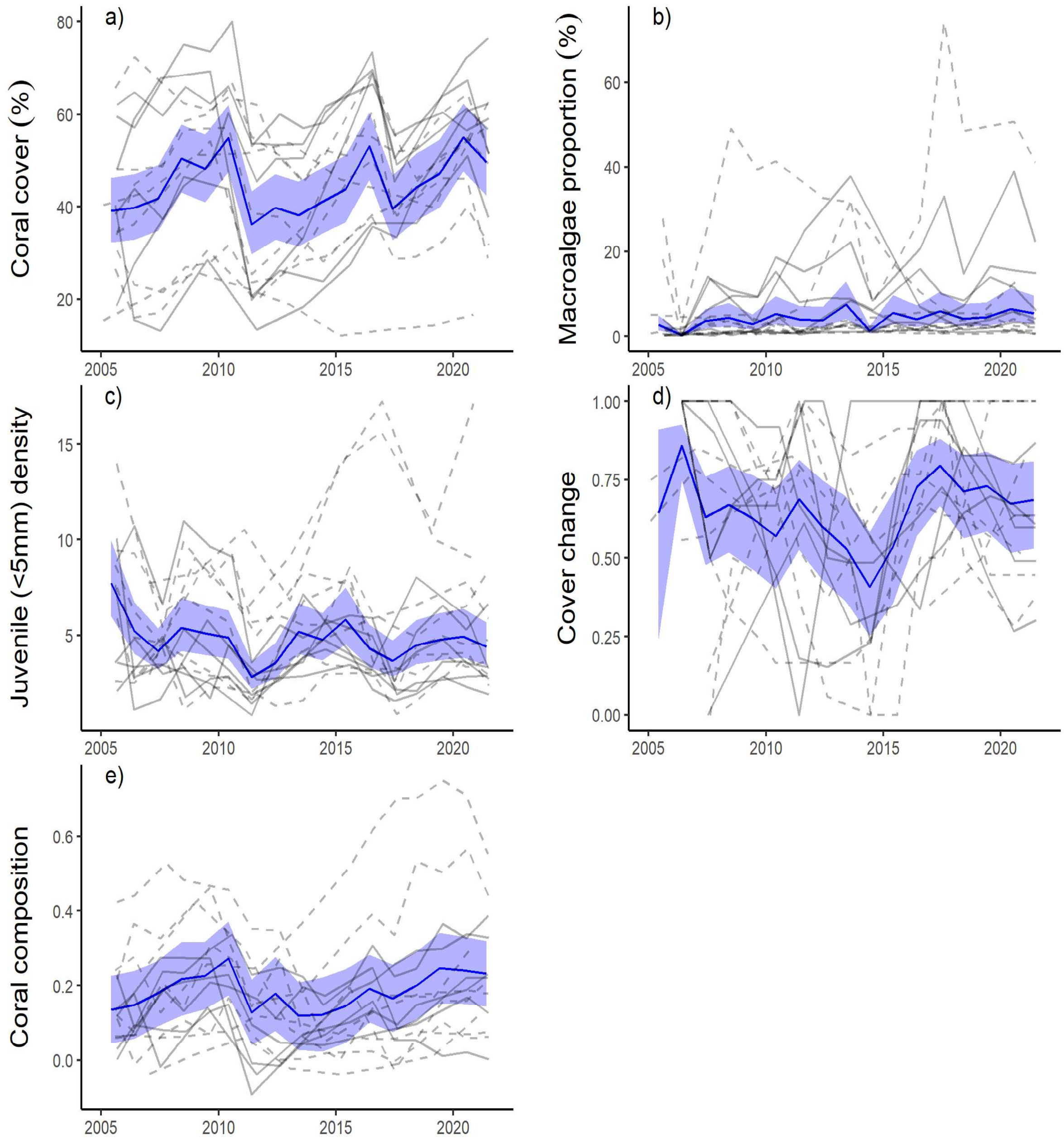


Figure 20 Johnstone Russell-Mulgrave sub-region indicator trends. a– e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

### 4.3.4 Wet Tropics region: Herbert Tully sub-region

The Coral Index in the Herbert Tully sub-region remains categorised as 'good', having declined marginally since 2020 as the rate of coral cover increase has slowed (Figure 21).

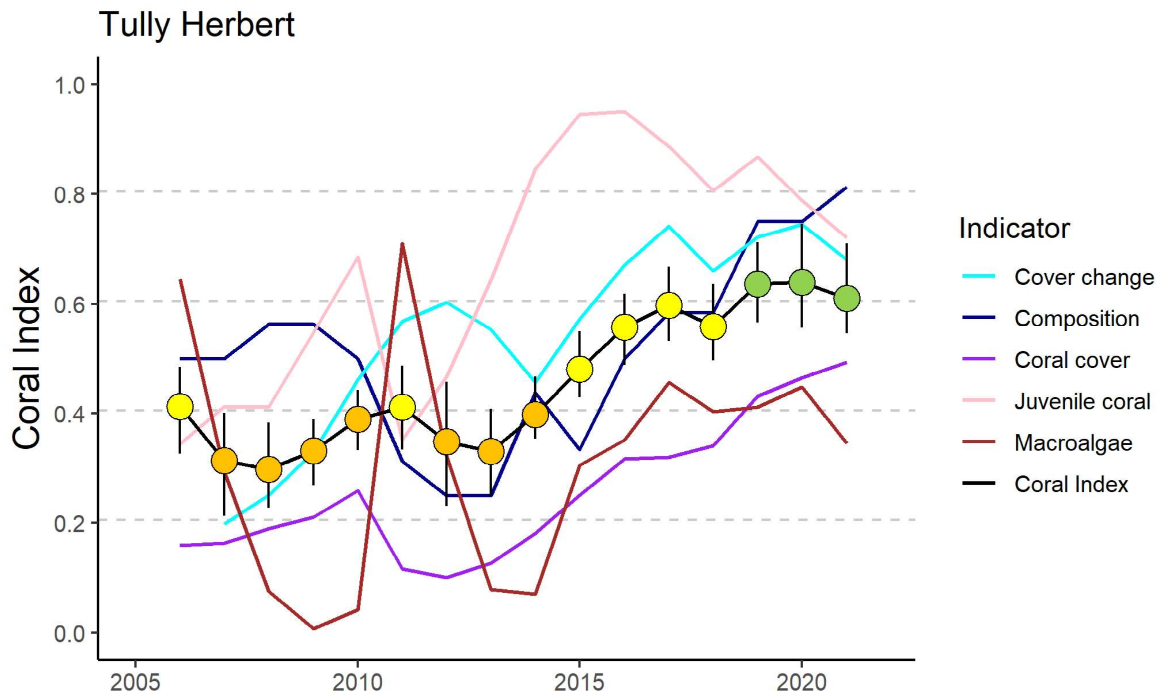


Figure 21 Trends in Coral Index and indicator scores for the Herbert Tully sub-region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = 'poor', yellow='moderate' and green='good. Error in Coral Index scores were derived from bootstrapped distributions of indicator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

The impressive rise of the Coral Index from 'poor' to 'good' (2013 – 2019) was supported by improvements in all indicators (Figure 21, Table 11). In 2021 coral communities remained in 'good' condition although the Coral Index had slightly declined due to declines in juvenile coral, macroalgae, and cover change indicators (Table 11).

Table 11 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 2013 | 2 | -0.04 | 0.64 | 0.02 | 0.64 | -0.67 | 0.92 | 0.39 | 0.83 | 0.05 | 0.58 | 0 | NA |
| | 5 | -0.12 | 0.88 | 0.01 | 0.54 | -0.59 | 0.90 | 0.20 | 0.75 | -0.08 | 0.60 | -0.13 | 0.70 |
| 2013 to 2019 | 2 | 0.38 | 1.00 | 0.39 | 1.00 | 0.40 | 0.79 | 0.04 | 0.58 | 0.39 | 0.99 | 0.67 | 0.99 |
| | 5 | 0.32 | 1.00 | 0.32 | 0.94 | 0.99 | 0.67 | 0.16 | 0.73 | 0.34 | 0.96 | 0.50 | 1.00 |
| 2019 to 2021 | 2 | 0.01 | 0.53 | 0.09 | 0.79 | -0.05 | 0.70 | -0.2 | 0.84 | 0.02 | 0.54 | 0 | NA |
| | 5 | -0.03 | 0.96 | 0.03 | 0.68 | -0.08 | 0.63 | -0.09 | 0.78 | -0.09 | 0.71 | 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, with rapid recovery occurring in both instances (Figure 23a). The combined impacts of these cyclones account for 72% of hard coral cover losses since 2005 (Figure 22e).

Following each cyclone, in addition to an immediate reduction, was a lagged decline in the Coral Index (Figure 21). This lagged response primarily reflects a temporary improvement and then rapid decline in the macroalgae indicator scores (Figure 23d, Table 11). 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 a lagged impact on Coral Index scores.

A strong contributor to the current score has been the rapid rate at which hard coral cover has recovered, as indicated by the high cover change, and increasing coral cover, indicator scores in recent years (Figure 21). Although the cover change score did decline slightly in 2021 (Figure 23d) predominantly at 5 m depths (Table 11). Comparing hard coral cover at the three reefs sampled since 2005 shows coral cover in 2021 is similar to levels observed in 2005, prior to the impact of cyclone Larry (Figure A 3).

Although remaining at 'good' levels, scores for the juvenile coral indicator have declined in recent years (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 A 3).

While the macroalgae indicator has improved since 2014, scores for this indicator remain at minimum levels of zero (Table A 7) at the 2 m depths of Bedarra, Dunk North and Dunk South. At these reefs, the macroalgae community is dominated by persistent brown algae of the genus *Sargassum* (Table A 11

| (sub-)region | Reef | Depth | Alcyoniidae | Anthothelidae | Briareidae | Clavulariinae | Heliporidae | Neptheidae |
|---|---|---|---|---|---|---|---|---|
| Fitzroy | Barren | 2 | 0.49 | 0 | 0.25 | 0 | 0 | 0 |
| | | 5 | 0.08 | 0 | 0 | 0 | 0 | 0 |
| | North Keppel | 2 | 0.01 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0.12 | 0 | 0 | 0 | 0 | 0 |
| | Middle | 2 | 0.05 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0.03 | 0 | 0 | 0 | 0 | 0 |
| | Keppels South | 2 | 0.1 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0.02 | 0 | 0 | 0 | 0 | 0 |
| | Pelican | 2 | 0.08 | 0.31 | 0 | 0 | 0 | 0 |
| | | 5 | 0.6 | 0.19 | 0.06 | 0 | 0 | 0.08 |

Table A 11).

At the reef-level, loss of corals was recorded at Barnards, principally among the *Acropora* at 2 m (Table A 9). Some *Acropora* were also lost at Dunk South 2 m. However, these losses were overshadowed by increasing cover at other sites (Figure 23a). While neither of these sites were strongly impacted by coral bleaching in 2020 (Thompson *et al* 2021), there was increased river discharge in the region during the water-year 2020-21 from the Tully, Murray, and Herbert, with levels above long-term medians (Table A 5). Levels of coral disease have been above the long-term mean in recent years (Figure A 7) implying an ongoing level of environmental stress.

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 (mean Chl *a* concentration over the wet season exceed the guideline) waters (Figure 22a, Table A 8). The combination of high turbidity and high nutrient availability (Figure A 12) is consistent with the prevalence of macroalgae observed in the shallow, but not deeper, depths at most reefs (Figure 23b, Figure A 3). The long-term water quality index for this sub-region remains poor (Figure A 12a). The short-term water quality index remains in moderate although has declined since 2020 reflecting higher inputs from the catchments in 2021 (Figure A 12, Figure 22d).

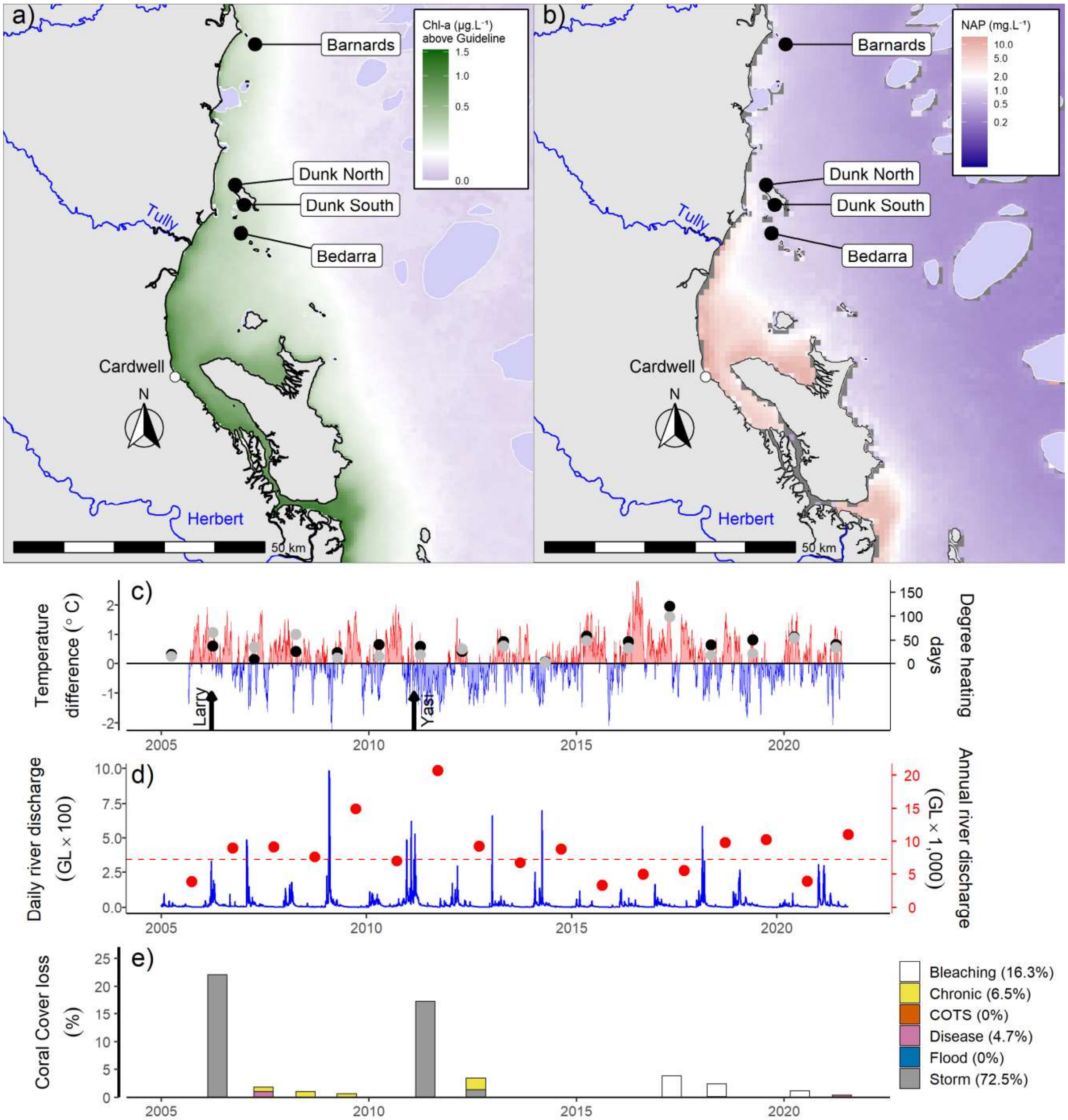


Figure 22 Herbert Tully sub-region environmental pressures. Maps show location of monitoring sites along with, a) mean chlorophyll a exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are mean levels over the period 2003–2018 (Chl) and 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.

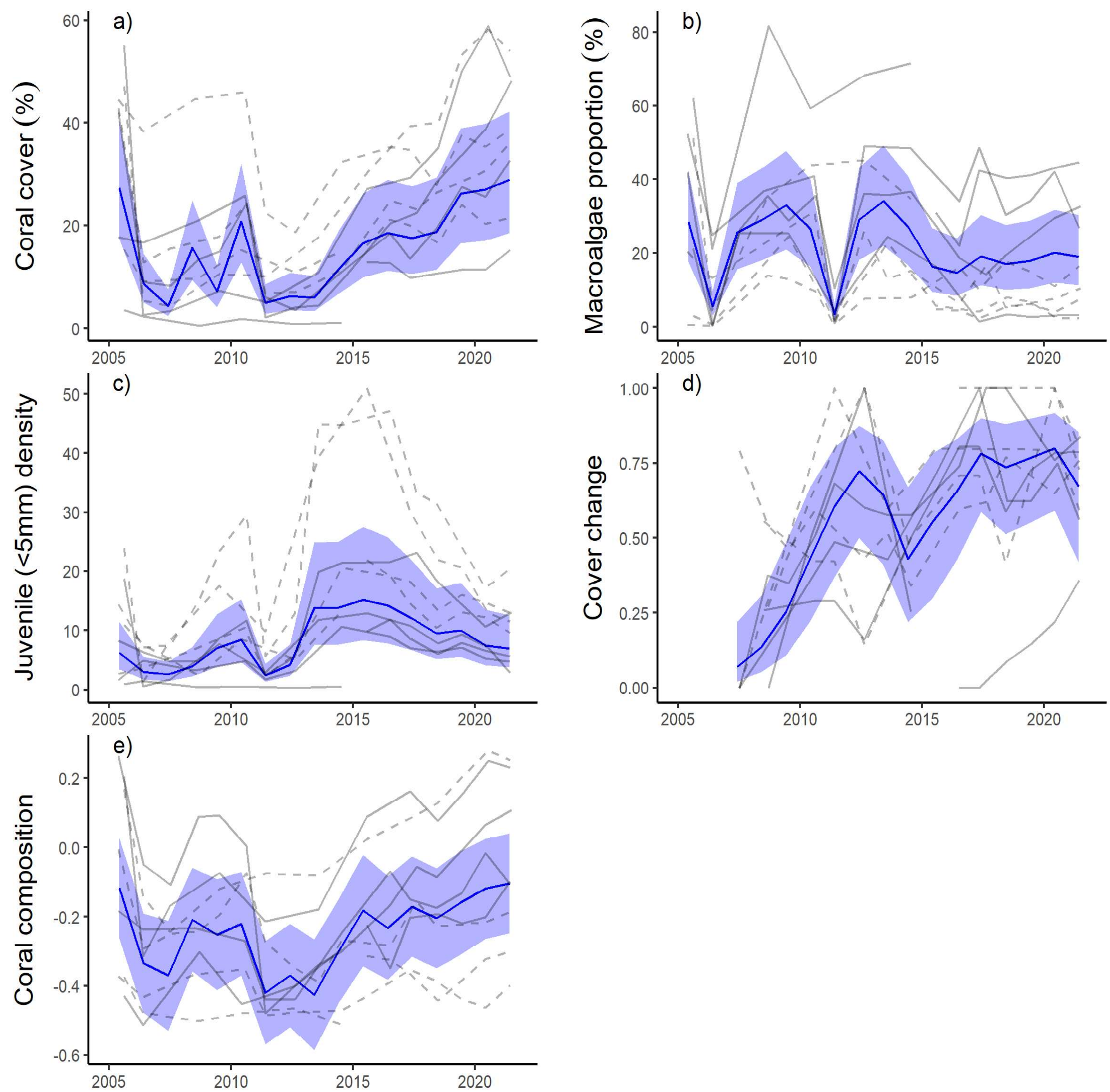


Figure 23 Herbert Tully sub-region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

## 4.4 Burdekin region

The Coral Index remained in moderate condition having declined slightly the peak reached in 2020 (Figure 24, Table 12).

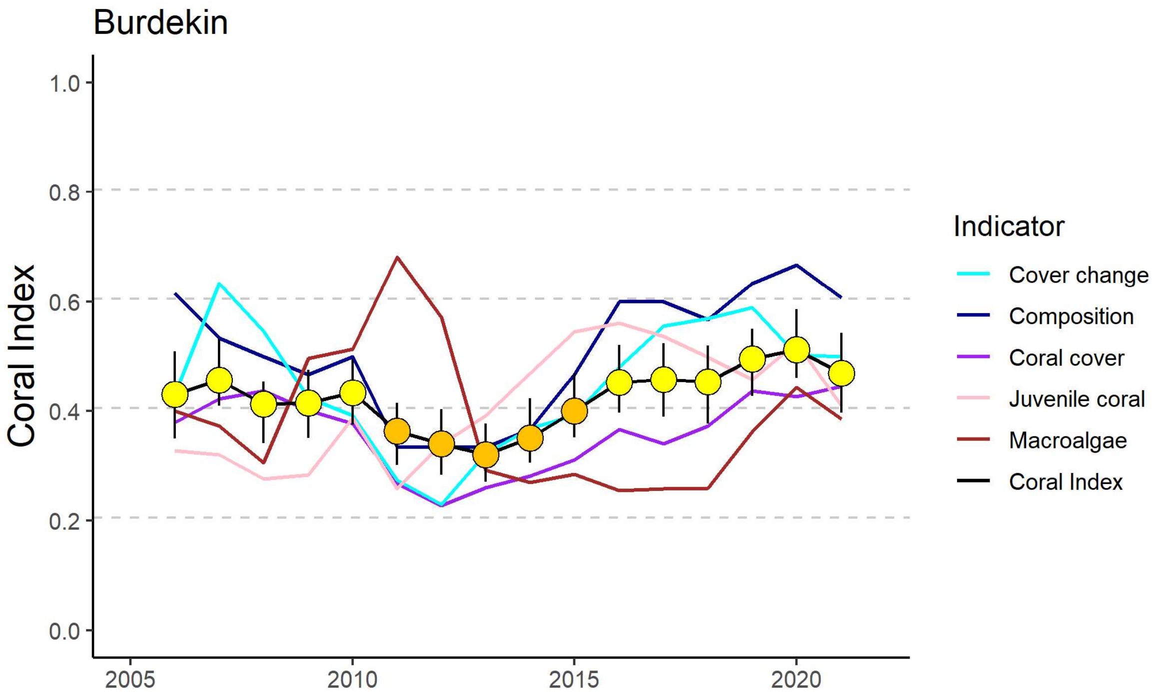


Figure 24 Trends in Coral Index and indicator scores for the Burdekin region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = 'poor', yellow='moderate'. Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at individual reefs.

Improvement in the Coral Index between 2013 to 2020 reflected improvements across all indicators (Table 12), as coral communities recovered from a period punctuated by high discharge from the regions catchments and exposure to physical damage from storms and cyclones (Figure 25c, d, e). The recent decline was most evident at 2 m depth where macroalgae, juvenile coral, and composition indicator scores declined (Table 12). In contrast, coral cover at 2 m depth increased (Figure 24, Table 12).

Table 12 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 | -0.07 | 0.70 | -0.09 | 0.64 | -0.17 | 0.71 | -0.04 | 0.61 | -0.03 | 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 | 0.89 | 0.18 | 0.77 | 0.26 | 0.87 | 0.33 | 0.89 | 0.31 | 0.76 |
| 2020-2021 | 2 | -0.11 | 0.75 | 0.05 | 0.82 | -0.26 | 0.73 | -0.08 | 0.75 | -0.01 | 0.54 | -0.25 | 0.73 |
| | 5 | -0.01 | 0.52 | 0.01 | 0.58 | 0.04 | 0.66 | -0.14 | 0.69 | 0.0 | 0.51 | 0.06 | 0.66 |

There were no major disturbances that impacted coral communities during the 2020-2021 monitoring season (Figure 26). However, there was a period of heavy rainfall and flooding brought to the region in early January (Figure 25) by the low-pressure system of cyclone Imogen, a short-lived category 1 cyclone in the Gulf of Carpentaria. The influence of this rainfall on rivers in the region was variable (Table A 5); most notable being the Burdekin River with a discharge of almost twice the long-term median.

The slight increase in coral cover across the region reflected modest increase, or similar coral cover to previous surveys at most reefs (Figure 26a, Figure A 4). Palms East is a notable exception; the *Acropora* dominated community continuing a strong recovery since cyclone Yasi (2011) with cover now exceeding 45% at both depths (Table A 9, Figure A 4Table A 9). Coral cover did, however, decline at three locations, each attributed to coral bleaching in 2020. Havannah North was not surveyed in 2020 and based on the level of bleaching observed at Havannah South, the loss of hard coral cover between 2019 and 2021 (Figure A 4, Table A 6) was attributed to the 2020 bleaching event. At both Havannah South 2 m and Magnetic 5 m a high proportion of coral was bleached at the time of survey in 2020 (Thompson *et al.* 2021) and the further loss of coral observed in 2021 (Figure A 4) was attributed to their subsequent mortality due to the stress imposed by the bleaching event (Table A 6).

The further loss of hard coral cover at Havannah South 2m in 2021 included a high proportion of branching *Acropora,* a group positively aligned to the water quality gradient underpinning the composition indicator (Table A 4). This loss, along with an increase in cover of M*ontipora,* a group negatively aligned with water quality (Table A 4), at Magnetic 2 m are responsible for the recent decline in the composition indicator.

The macroalgae indicator score declined slightly, returning the indicator into the ‘poor’ range (Table A 7Table A 7, Figure 24). Changes in macroalgae were variable among reefs (Figure 26a). Influential in the recent decline were increased cover of macroalgae at Havannah 2 m depth, where *Lobophora* had increased (Figure A 4, Table A 11Table A 11), and Lady Elliot 2 m depth, where the macroalgae community was dominated by the red algae *Hypnea* (Table A 11Table A 11). There was also an increase in cover of *Lobophora* at Havannah North and Havannah 5m (Figure A 4, Table A 11Table A 11) but in both cases this did not influence the scores relative to 2020 as the proportion of macroalgae already exceeded the threshold for a score of zero at these locations (Table A 3, Thompson *et al.* 2021). In contrast, quite large reductions in macroalgae (predominantly *Sargassum)* were observed at 2 m depths at Pandora and Magnetic, however this reduction did not change macroalgae scores as cover remained above the thresholds for a score of zero (Figure A 4, Table A 11Table A 11).

Since 2014 the density of juvenile corals has remained ‘moderate’ (Figure 24) although highly variable among reefs (Figure 26c). In 2021 the density of juvenile hard corals declined at most sites except for Palms West and Havannah North (Figure 26, Figure A 4).

While concentrations for most water-quality parameters declined in 2021, a sharp upturn in the concentration of NOx and turbidity levels (Figure A 13c, e) led to a decline in the short-term water quality index from ‘good’ to ‘moderate’ (Figure A 13a). Concentrations of NOx have remained above guideline values for the duration of the program (Figure A 13c). Concentrations of dissolved and particulate forms of organic carbon have markedly increased over the same period (Figure A 13), however these parameters do not contribute to water quality index scores (Gruber *et al.* 2020). At the regional scale, the poor scores for the macroalgae indicator, in particular, 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.

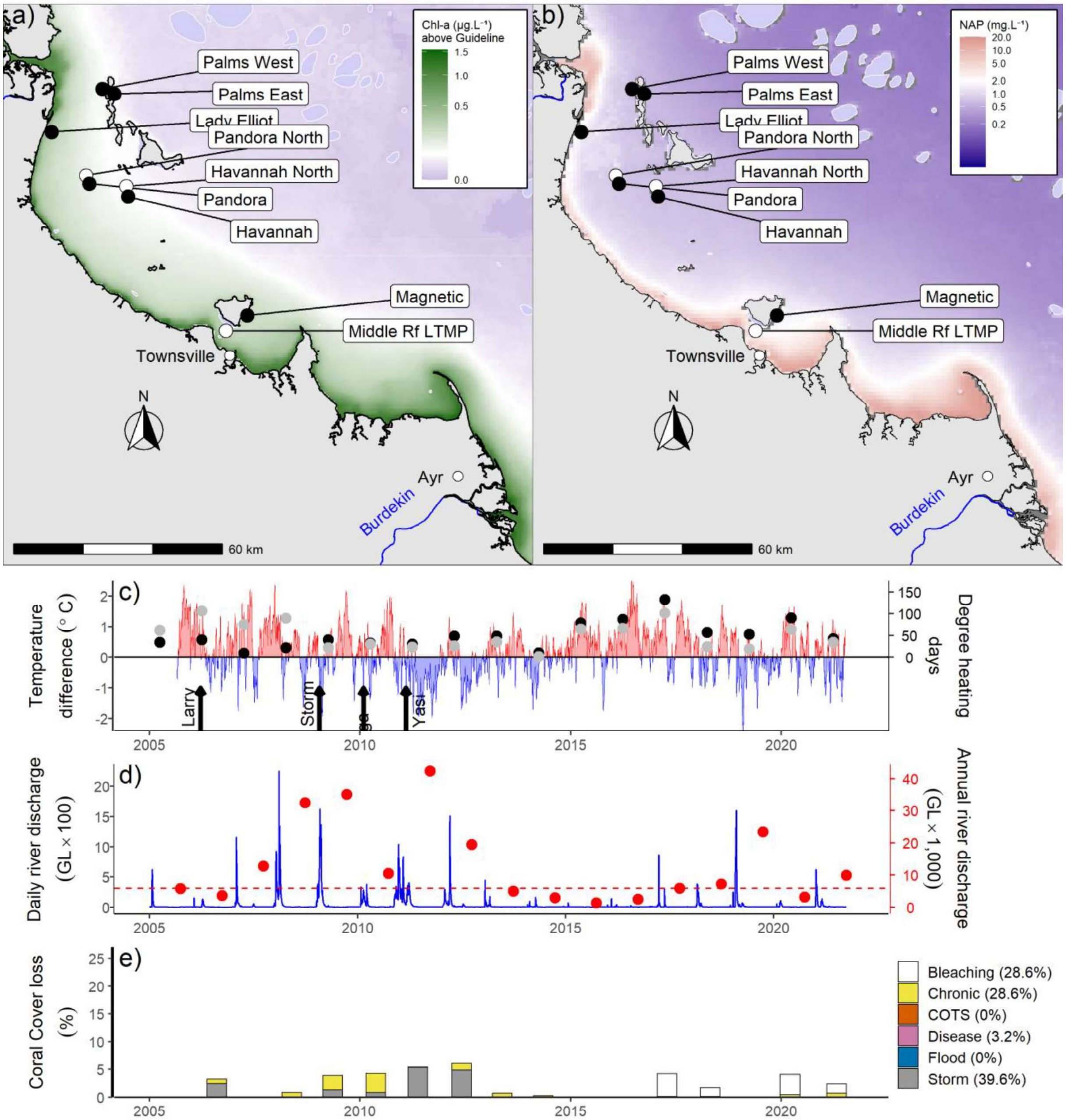


Figure 25 Burdekin region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) mean chlorophyll *a* exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are mean levels over the period 2003–2018 (Chl) 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.

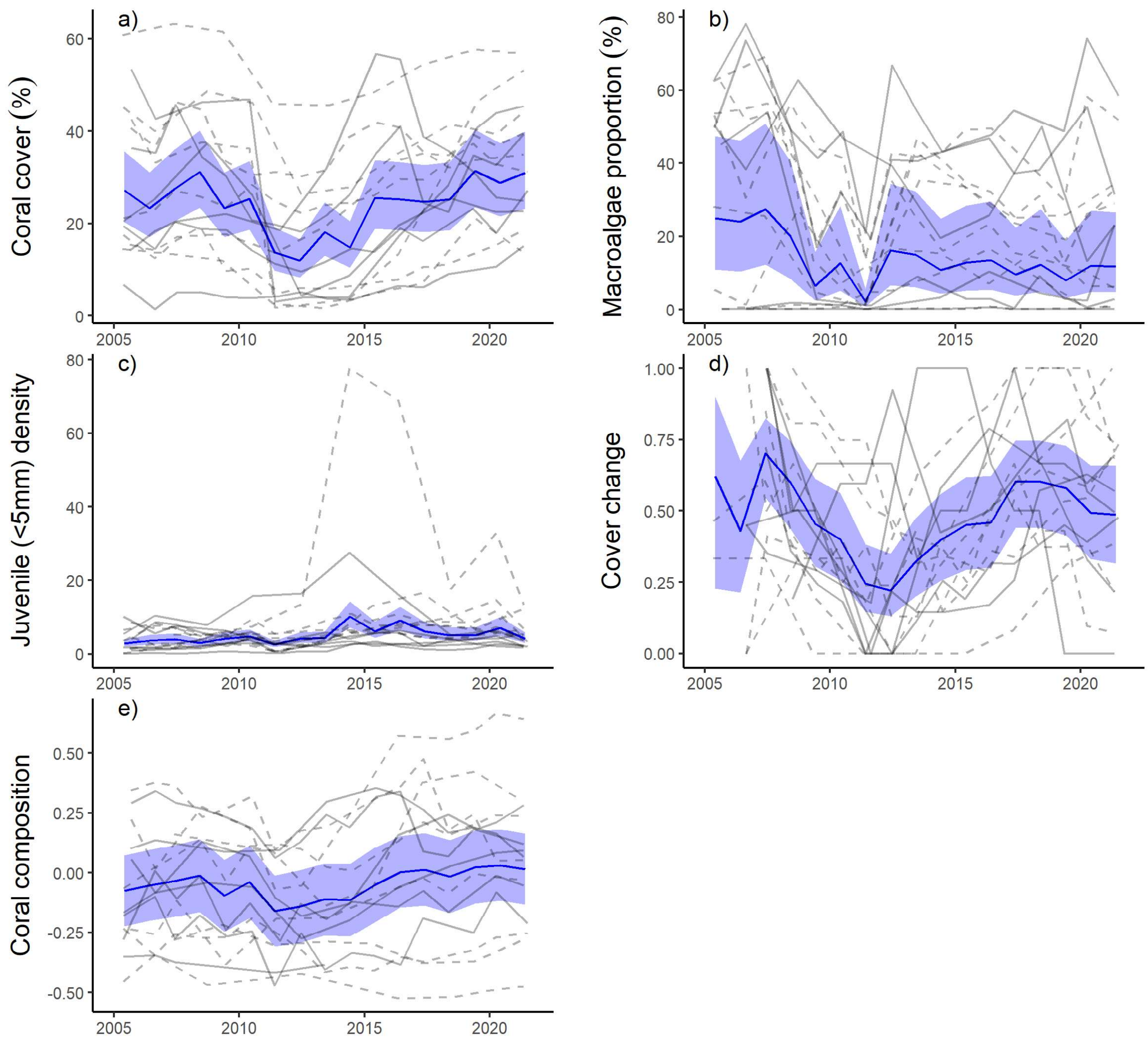


Figure 26 Burdekin region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

## 4.5 Mackay-Whitsunday region

The Coral Index score remained ‘poor’ in 2021, although shows a slight improvement since 2020 (Figure 27). This improvement breaks the continued decline in Coral Index scores since the severe impact of cyclone Debbie in 2017 (Figure 28e, Figure 27). Over the period 2016-2020 there were region-wide reductions in all indicators (Table 13). Most improved in 2021 were scores for the juvenile coral and macroalgae indicators, however all indicator scores remain in the ‘poor’ range (Figure 27, Table 13).

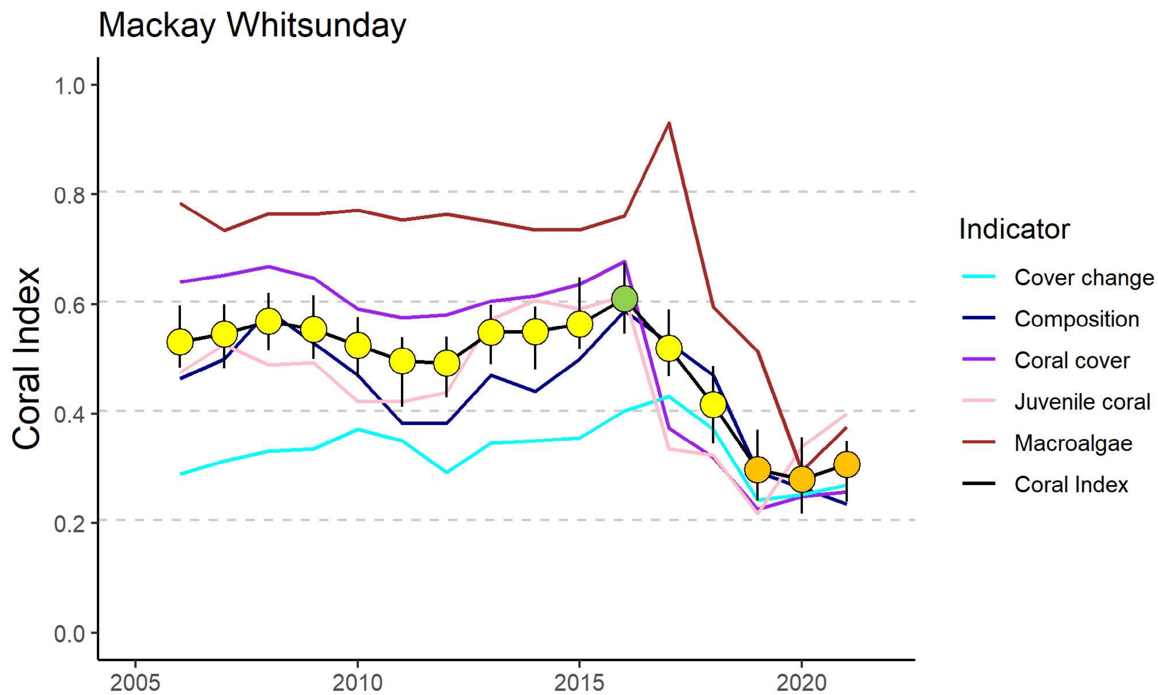


Figure 27 Trends in Coral Index and indicator scores for the Mackay-Whitsunday region. Coral Index scores are coloured by Reef Water Quality Report Card categories: orange = ‘poor’, yellow=’moderate’, green=’good’. Error in Coral Index scores were derived from bootstrapped distributions of indicator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

Table 13 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 2021 | 2 | -0.02 | 0.60 | 0.01 | 0.67 | -0.04 | 0.54 | -0.0 | 0.57 | -0.05 | 0.66 | -0.07 | 0.65 |
| | 5 | 0.06 | 0.72 | 0.0 | 0.54 | 0.16 | 0.71 | 0.11 | 0.70 | -0.04 | 0.57 | 0 | 0 |

Scores for the cover change indicator have been consistently low (Figure 27) and declined further in recent years (Table 13). These low scores demonstrate rates of hard coral cover recovery relative to other regions. Low scores for this indictor are of particular concern in this region where persistently high turbidity (Figure A 14) has selected for relatively slow growing taxa at many of the deeper sites (Figure A 5, Table A 9Table A 9). 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. The only locations at which hard coral cover was increasing at modelled rates in 2021 (cover change scores > 0.5) were Hook 5 m, where the cover of Poritidae (Figure A 5), genus *Porites* (Table A 9Table A 9) has increased, and Shute Harbour 2 m where stands of branching *Acropora* are recovering (Figure A 5, Table A 9).

Juvenile coral indicator scores declined steeply following cyclone Debbie (Figure 27), but are beginning to rebound (Figure 27, Figure 29c). Increases are seen at both depths (Figure 29c), with marked increases at Hayman, Border, and at 5 m depth at Daydream (Figure A 5). The rise in density of juvenile corals at Hayman was predominately among *Acropora*, a group previously well representing in the adult population (Figure A 5). 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).

Improvement in juvenile coral indicator scores have tracked the regional improvement in macroalgae scores (Figure 27). However, macroalgae scores remain well below those observed prior to cyclone Debbie (Figure 27). In 2021 macroalgae scores of zero were recorded at both 2 m and 5 m depths at Double Cone, Pine and Seaforth and at 2 m depth at Dent and Daydream (Table A 7Table A 7). At each of these location macroalgae cover had increased compared to levels observed in 2020, however these increases had very little or no influence on the indicator scores as, apart from Dent, scores were zero in 2020 (Thompson *et al.* 2021). Improvement in macroalgae scores was influenced by reduced cover of macroalgae at Hook and Shute and 5m depths of Dent and Daydream (Figure A 5), all reefs with relatively low cover of macroalgae.

High turbidity across the region (Figure A 14) 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. Over time, coral species tolerant of these conditions (Oculinidae, Pectiniidae, Agariciidae, Poritidae (genus *Goniopora*)). In contrast, Acroporidae and Poritidae (genus *Porites*) are most common at 2 m depths (Figure A 5). Reductions in the composition score following cyclones imply additional selective pressures on those species sensitive to poor water quality. The pressure imposed by the 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. have been selected for, however increased incidence of disease in 2021 (Figure A 7) suggest ongoing stress to corals With both the long-term and short-term water quality index scores showing water quality is generally below the GBRMPA guidelines in 2021 (

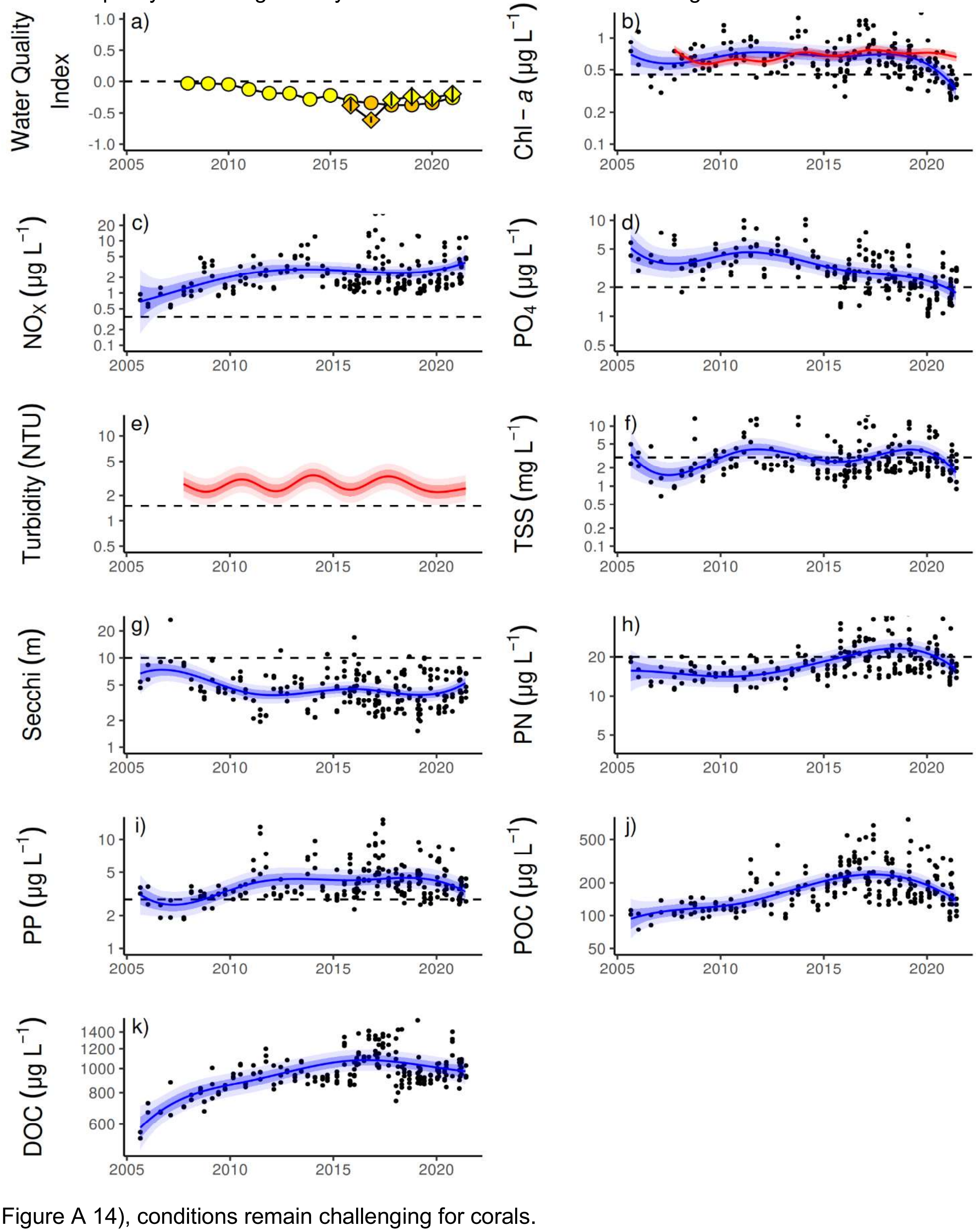


Figure A 14), conditions remain challenging for corals.

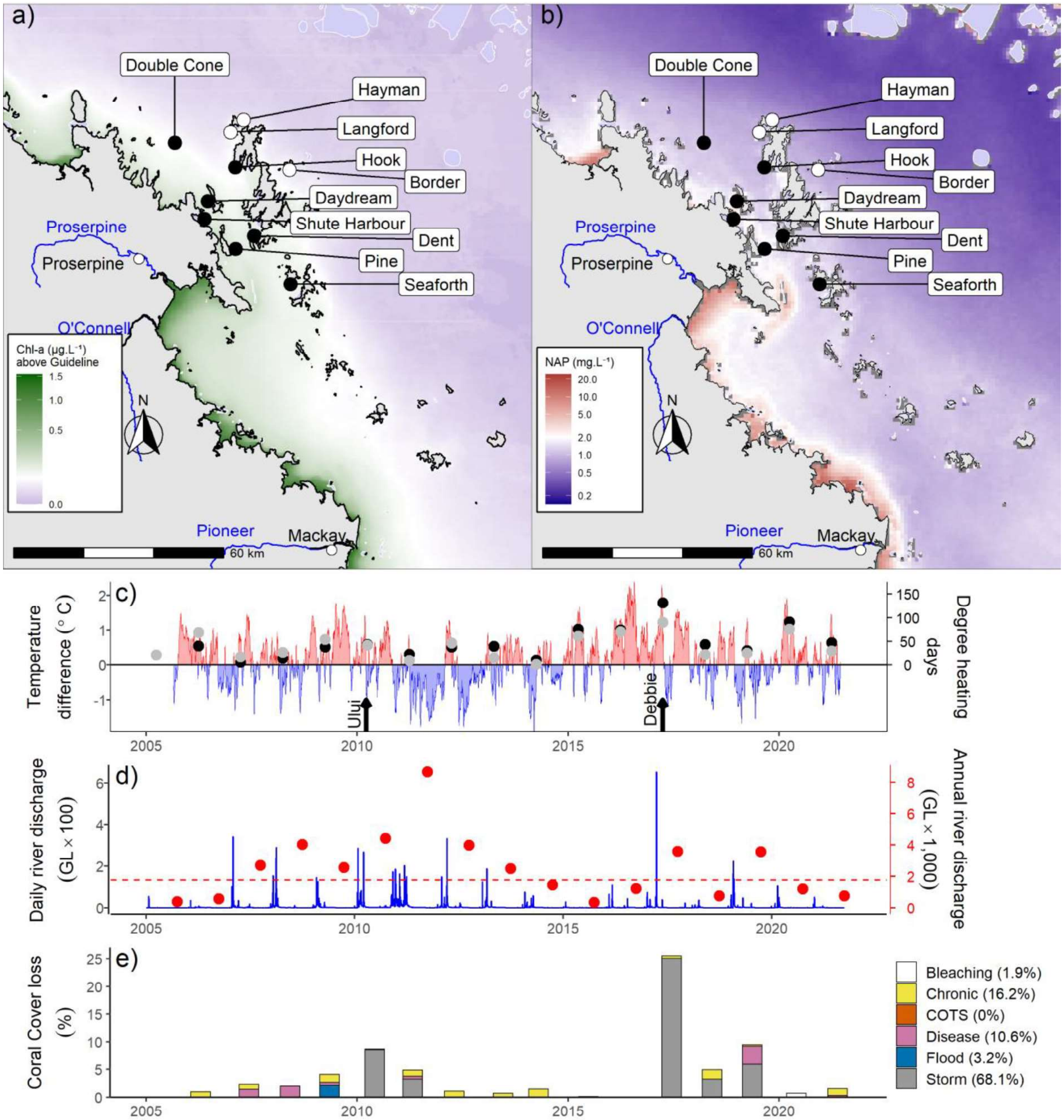


Figure 28 Mackay-Whitsunday region environmental pressures. Maps show location of monitoring sites, black symbols MMP, white symbols LTMP along with a) mean chlorophyll *a* exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are mean levels over the period 2003–2018 (Chl) 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.

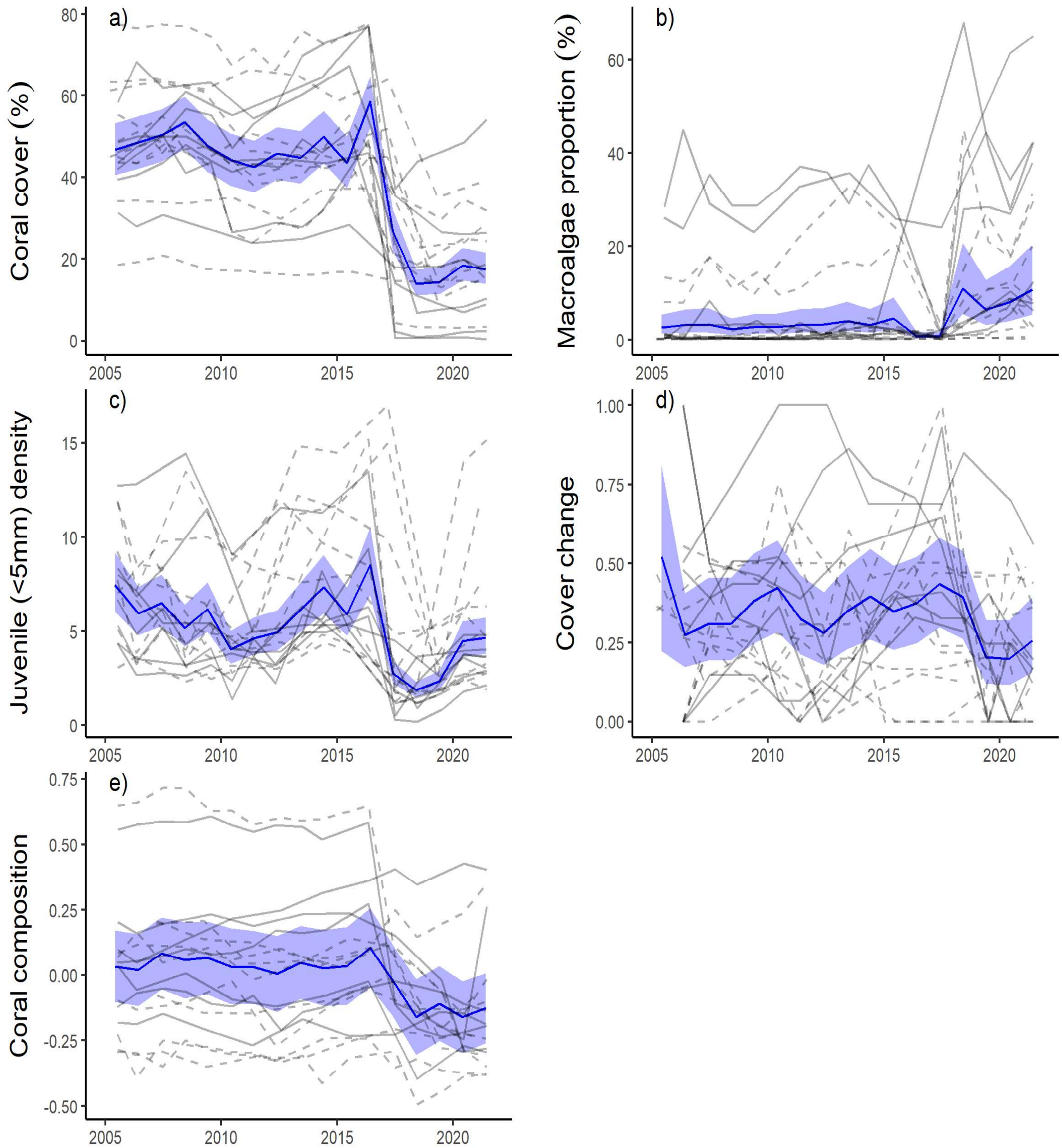


Figure 29 Mackay-Whitsunday region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

## 4.6 Fitzroy region

The Coral Index score in the Fitzroy region remains ‘poor’ having declined since 2020. This decline interrupts the improvement in scores between 2014 and 2020 (Figure 30, Table 14).

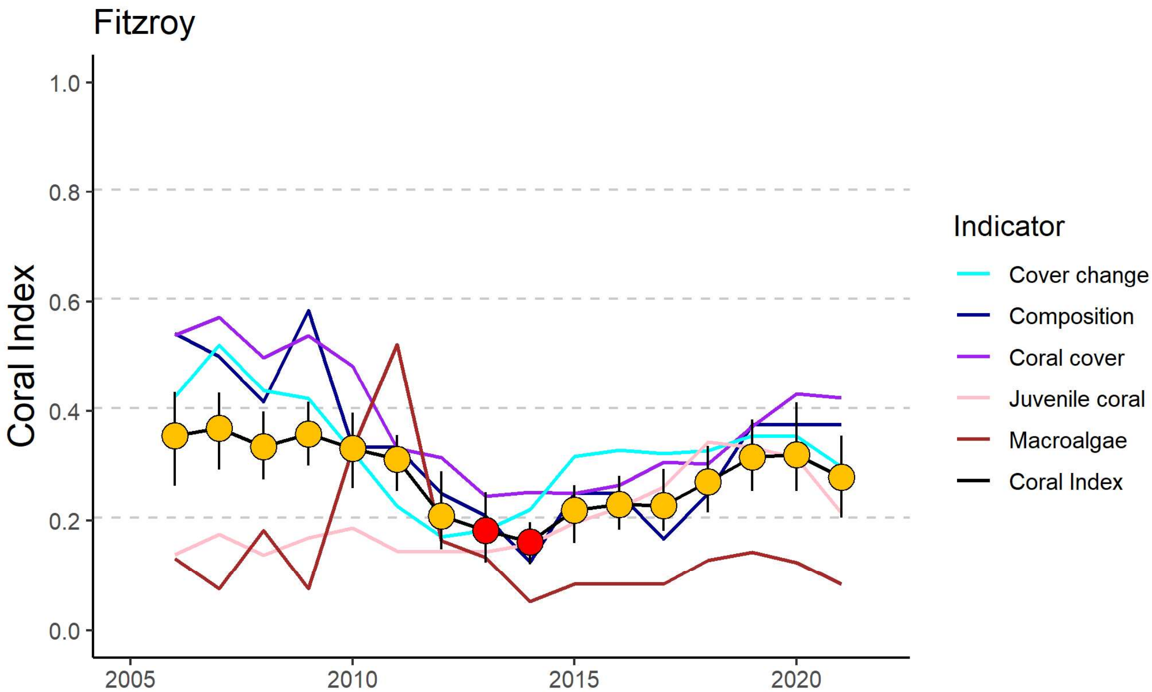


Figure 30 Trends in Coral Index and indicator scores for the Fitzroy region. Coral Index scores are coloured by Reef Water Quality Report Card categories: red=’very poor’, orange = ‘poor’. Error in Coral Index scores were derived from bootstrapped distributions of indicator scores at individual reefs.

Table 14 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 | 0.88 | -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 | 0.15 | 1.00 | 0.19 | 0.89 | 0.06 | 0.67 | 0.14 | 0.84 | 0.11 | 0.70 | 0.25 | 0.74 |
| | 5 | 0.17 | 0.87 | 0.17 | 0.82 | 0.08 | 0.70 | 0.17 | 0.76 | 0.16 | 0.77 | 0.25 | 0.70 |
| 2020 to 2021 | 2 | -0.02 | 0.83 | -0.0 | 0.54 | 0 | NA | -0.08 | 0.88 | -0.06 | 0.71 | 0 | NA |
| | 5 | -0.05 | 0.72 | -0.01 | 0.66 | -0.08 | 0.69 | -0.12 | 0.77 | -0.05 | 0.72 | 0 | 0.50 |

Across the region only the coral cover indicator was in the ‘moderate’ range (Figure 30). Most influential in the recent decline in the Coral Index have been reduced scores for the juvenile coral indicator, with the density of juvenile hard corals declining at most reefs and both 2 m and 5 m depths (Table 14, Figure 32a,c,d). Scores for the cover change indictor also declined and remain poor at the regional level (Figure 30, Table 14). For comparability in the Coral Index scores across the time-series, annual scores include caried forward scores for Peak Island, which was last surveyed in 2019. As such, the probability of changes in Coral Index and indicator scores (Table 14) will be buffered as there is no change possible for this reef. Previously the cover change indicator was reported as showing ongoing improvement between 2014 and 2020 (Thompson *et al.* 2021). While this was still the case at 5 m depths, in particular (Figure 30, Table 14), the magnitude of this improvement has been revised. Previous estimates of the cover change indicator excluded declines in cover at Middle Island and Pelican Island in 2018, the cause of which was classified as “unknown” as no acute event could be attributed. The estimates presented here (Figure 30) consider these declines as symptomatic of environmental stress resulting in a reduction in the cover change indicator scores compared to those previously reported for 2018-2020.

The coral reefs monitored are situated along a distinct environmental gradient within Keppel Bay. Peak and Pelican are 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; these four reefs share reasonably low levels of total suspended solids (Figure 31a, b, Table A 8). The gradient in water quality is clearly reflected in the benthic communities at these reefs . At Peak and Pelican benthic communities differ markedly between 2 m and 5 m depths (Figure A 6) 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 in contrast to fast-growing Acroporidae (*Acropora, Montipora* spp.) in the shallows; although these shallow communities were killed and replaced by macroalgae (*Sargassum* spp) following exposure to low salinity flood plumes in 2011 (Figure A 6). Closer to the Fitzroy River, Peak is characterised by low cover of corals, low density of juvenile corals and high cover of macroalgae (Figure A 6). 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 (section 9.3). In the less turbid waters surrounding the remaining reefs coral communities are dominated by Acroporidae (Figure A 6), principally, but not restricted to, the branching species *A. intermedia* and *A. muricata* (Table A 9).

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 14, Figure 32a). The disproportionate loss of *Acropora* (Figure A 6) resulted in a reduction in the composition indicator scores (Table 14, Figure 30). Compounding the impact of the acute disturbances were declines in the cover change scores between 2007 and 2014 (Table 14). These declines coincided with a period of relatively high discharge from the Fitzroy River and high levels of disease (Figure A 7), and are responsible for the “chronic” disturbances in Figure 31e.

In 2021 scores for the macroalgae indicator remain in the very poor range (Figure 30). It is only at 2 m depths at Barren Island that there is not a high proportion on macroalgae amongst the benthic algal community (Figure 32b, Table A 7Table A 7). The initial increase 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). Flooding in 2011 then killed corals and macroalgae at 2 m depths on reefs to the south of Great Keppel Island, causing a temporary improvement in macroalgae indicator scores. By 2012 macroalgae had recolonised these reefs and cover has remained high since. Of concern is that in recent years cover of macroalgae in the genus *Sargassum* has increased at Middle Island, and Humpy Island.

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 A 6). 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 (Figure 31d) did not result in substantial loss of coral cover, but are likely causes of observed low rates of increase in coral cover represented as chronic stress (Figure 31e). High water temperatures in 2020 resulted in extensive bleaching and observed mortality of corals at Barren Island. Despite this mortality, and the regional stress imposed by the 2020 bleaching event, coral cover across the region increased between 2019 and 2020 (Figure 32a). A lack of improvement in coral cover between 2020 and 2021 at the regional scale, masks differing trends among reefs (Figure 32a) with increases at Barren Island and 2 m depth at Keppels South contrasting a lack of recovery or declines in cover elsewhere (Figure A 6).

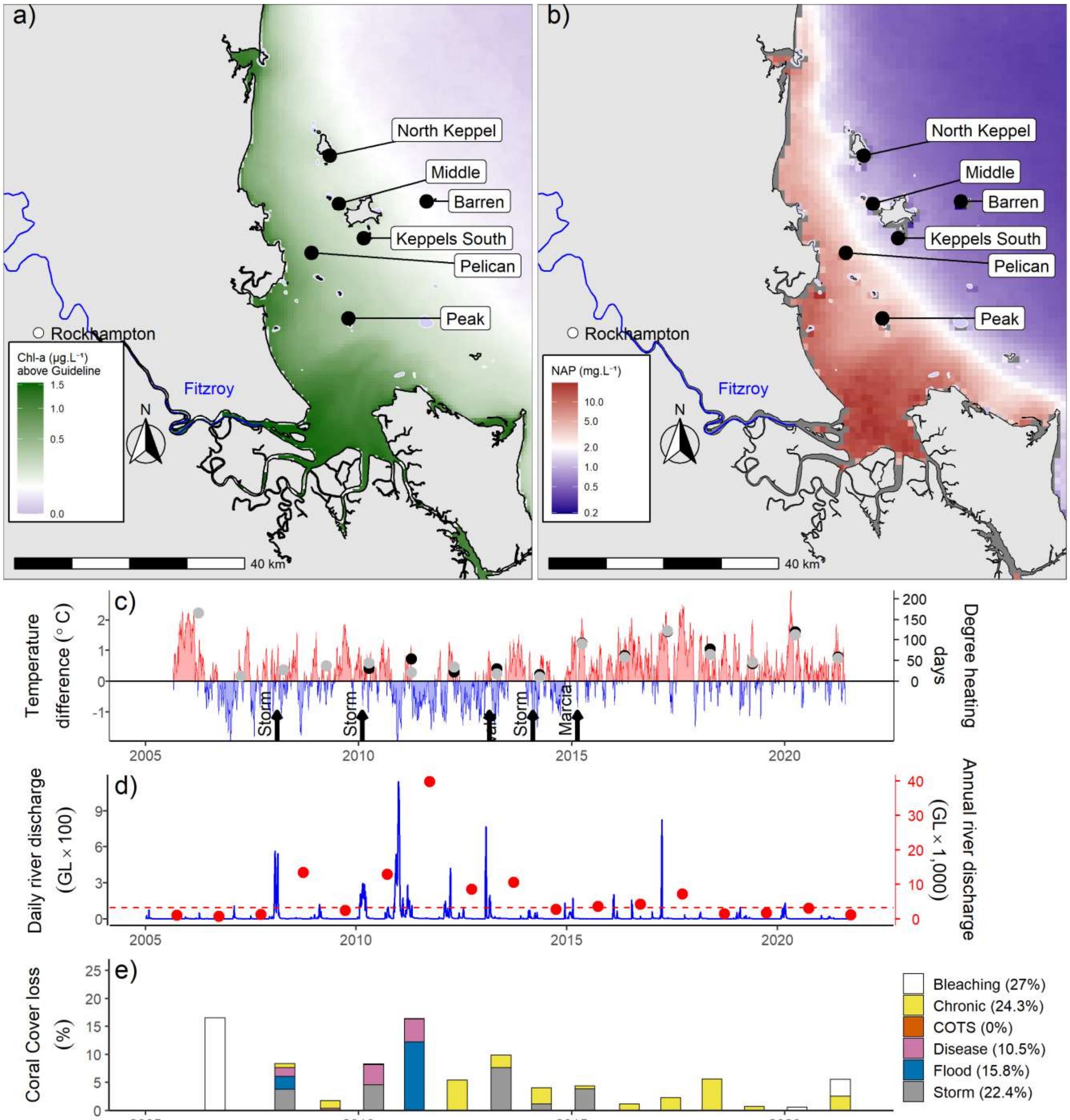


Figure 31 Fitzroy region environmental pressures. Maps show location of monitoring sites along with a) mean chlorophyll *a* exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are mean levels over the period 2003–2018 (Chl) and 2003–2018 (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.

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 (Figure A 15). 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).

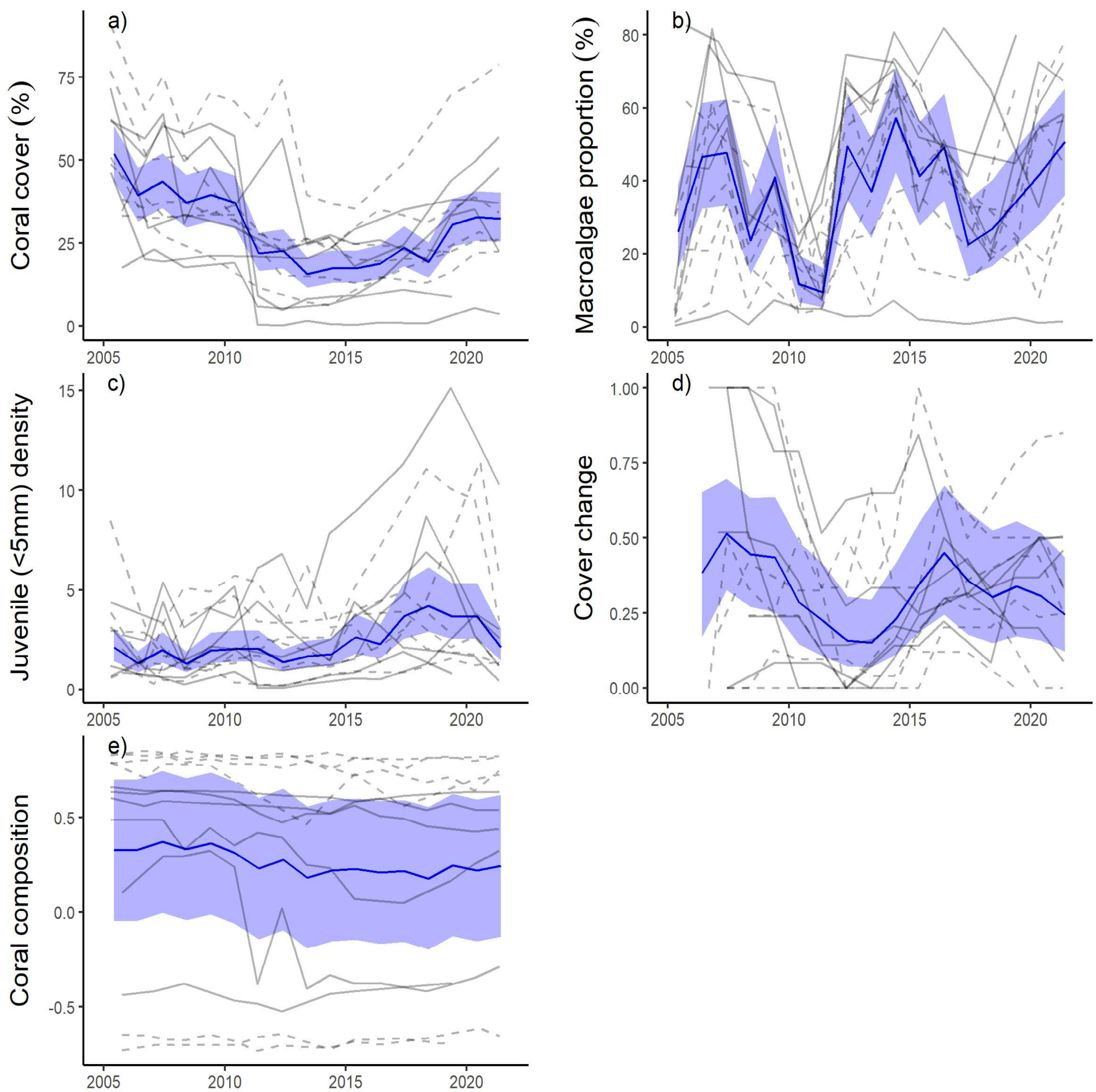


Figure 32 Fitzroy region indicator trends. a– e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

## 4.7 Response of coral communities to environmental conditions

### 4.7.1 Location along water quality gradients

The Reef-wide Coral Index scores in 2021 did not show consistent trends along water quality gradients. Of the three water quality variables, Chlorophyll *a* concentration (Chl *a*) in excess of wet season Guideline values ($0.63 \mu gL^{-1}$), Non algal particulate concentration (NAP) and the relative availability of light needed for photosynthesis (PAR), only Chl *a* related statistically to the variability in scores. The Reef-wide Coral Index scores at 2 m depths were negatively related to Chl *a*. (Table 15, Figure 33a).

Of the individual indicators:

- Scores for coral cover were negatively related to increasing concentration of Chl *a* at 2 m depths. This relationship strongest in the Wet Tropics regions (Table 15, Figure 33b).
- Scores for coral cover were also negatively related to increasing NAP concentration at both 2 m and 5 m depths (Table 15). This relationship was most evident at 2 m depth in Wet Tropics and at both 2 m at 5 m depth in the Fitzroy region (Table 15, Figure 34a b).
- Reef-wide scores for the macroalgae indicator were negatively related to Chl *a* concentration and positively related to PAR at 2 m depth (Table 15). These relationships were not statistically significant in any single region, although showed the same tendency as the reef-wide relationships everywhere except in the Mackay Whitsunday region (Figure 33c, d, Figure 34a).

Neither the juvenile coral, composition, or cover change indicator scores in 2021 varied predictably along water quality gradients.

Table 15 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 2021 and water quality proxies; mean wet season Chl *a* (2016-2020) and PAR (2015-2019) 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 Index | 2 | -4.4 | -0.3 | -4.5 | 1.1 | -6.6 | 2.1 | -7.6 | 16.9 | -8.8 | 0.2 |
| Coral cover score | 2 | -6.5 | -1.1 | -7.9 | -0.5 | -7.7 | 3.1 | -11.6 | 19.6 | -11.3 | 0.6 |
| Macroalgae score | 2 | -8.7 | -1.8 | -9.4 | 0.7 | -14.2 | 0.6 | -19.8 | 19.4 | -13.7 | 0.9 |
| | 5 | -6.0 | 0.2 | -6.2 | 3.0 | -11.0 | 3.8 | -15.9 | 1.5 | -7.2 | 7.2 |
| Photosynthetically active radiation | | | | | | | | | | | |
| Macroalgae score | 2 | 0.1 | 0.6 | -0.2 | 1.0 | -0.1 | 0.7 | -0.9 | 1.0 | -0.3 | 1.3 |
| Non Algal Particulates concentration | | | | | | | | | | | |
| Coral cover score | 2 | -3.0 | -0.6 | -10.3 | -1.5 | -2.3 | 1.7 | -2.7 | 3.7 | -5.8 | -0.1 |
| | 5 | -2.5 | -0.6 | -3.3 | 3.0 | -2.0 | 1.5 | -2.0 | 1.5 | -6.5 | -1.0 |

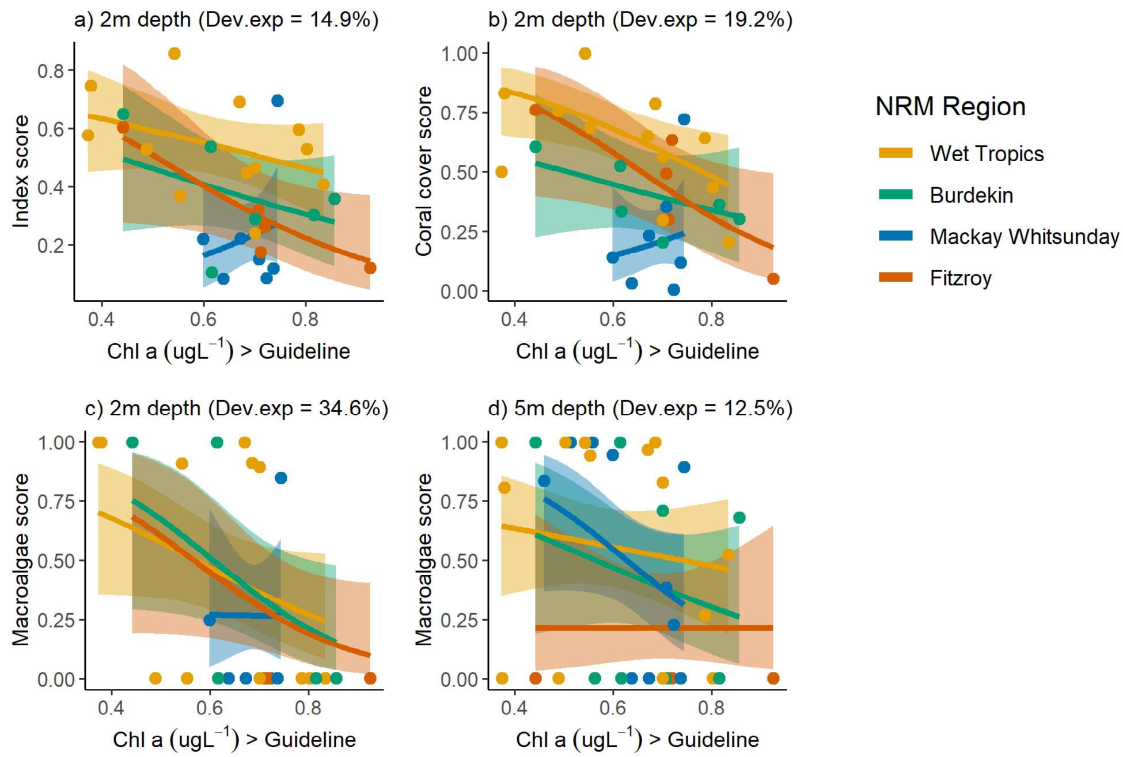


Figure 33 Coral Index and indicator score relationships to Chl *a* concentration. Combinations of Coral Index or indicator and depth are included where Reef wide relationships were indicated by models. Plots present predicted relationship within each region. Confidence intervals in predicted slopes are provided in Table 15. Chl *a* concentration expressed as mean wet-season exceedance of Guideline values (0.63µgL$^{-1}$).

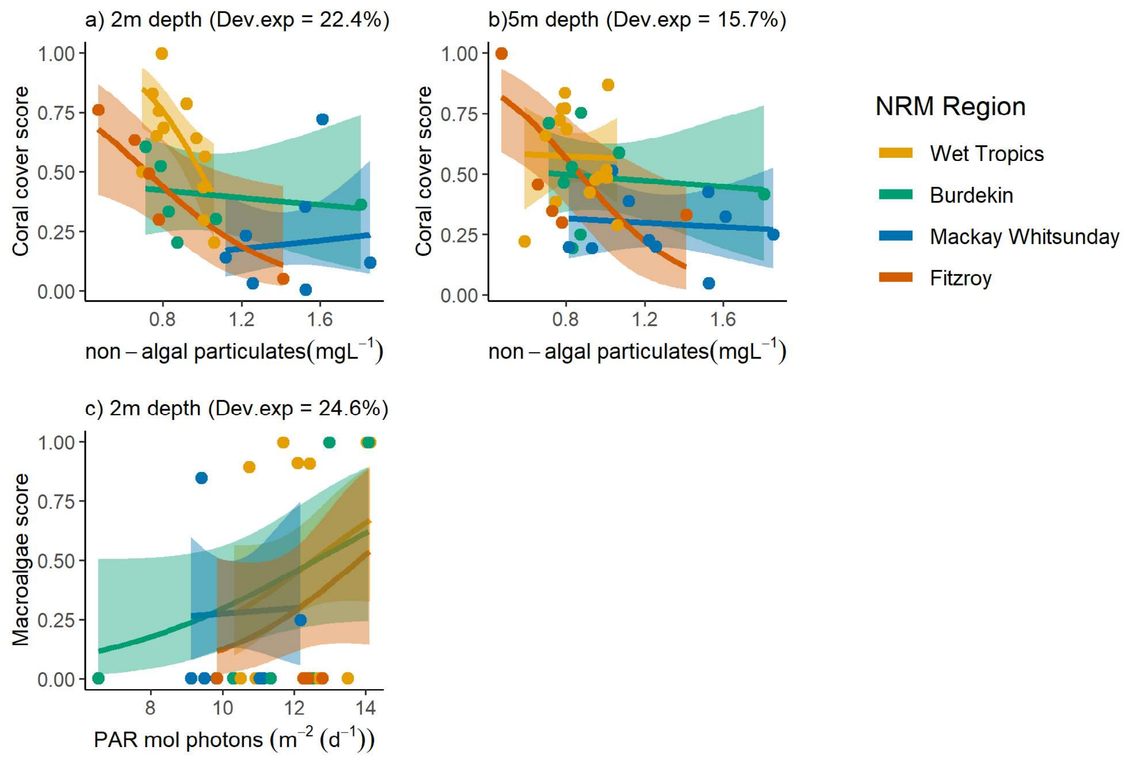


Figure 34 Coral indicator score relationships to PAR and Non algal particulates. Combinations of Coral Index or indicator and depth are included where Reef wide relationships were indicated by models. Plots present predicted relationship within each region. Confidence intervals in predicted slopes are provided in (Table 15).

Both the macroalgae and composition indicator scores are based on thresholds that vary along water quality gradients to ensure scores are sensitive to change at each reef. As such, the spatial analysis of scores masks underlying differences in the values underpinning these scores. Reef-wide, the proportion of algal cover classified as macroalgae shows a negative relationship to PAR and positive relationship to NAP at both 2 m and 5 m depth, and also a positive relationship to Chl *a* but only at 2 m depths (Table 16). These relationships are most evident in the Burdekin region and 2 m depths in the Wet tropics (Table 16,Figure 35).

Reef-wide community composition values were derived from the product of genus-level coral cover estimates and eigenvalues for the distribution of genera along WQ gradients (Table A 4). That community composition is negatively related to Chl *a* and NAP concentration and positively related to PAR (Figure 36) is entirely to be expected given the derivation and intent of this indicator. Limiting the sensitivity for change in community composition to water quality in the Mackay-Whitsunday region is the relatively short gradient in water quality compared to that observed in other regions.

Table 16 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 | 7.5 | 0.3 | 7.6 | -0.1 | 13.1 | -12.1 | 15.8 | 0.8 | 12.4 |
| Community composition | 2 | -1.3 | -0.03 | -1.5 | 0.1 | -2.0 | 0.4 | -0.1 | 6.7 | -2.4 | 0.2 |
| | 5 | -2.2 | -0.9 | -2.1 | -0.4 | -3.8 | -1.0 | -1.4 | 2.1 | -4.4 | -1.3 |
| Photosynthetically active radiation | | | | | | | | | | | |
| Macroalgae proportion | 2 | -0.6 | -0.1 | -1.0 | -0.1 | -0.7 | -0.1 | -0.7 | 0.6 | -1.3 | 0.02 |
| | 5 | -0.4 | -0.04 | -0.5 | 0.2 | -0.6 | -0.04 | -0.6 | 0.1 | -0.4 | 0.5 |
| Community composition | 2 | 0.02 | 0.1 | -0.01 | 0.2 | -0.01 | 0.1 | -0.3 | 0.03 | 0.03 | 0.3 |
| | 5 | 0.05 | 0.2 | 0.04 | 0.2 | 0.02 | 0.2 | -0.1 | 0.1 | 0.2 | 0.5 |
| Non Algal Particulates concentration | | | | | | | | | | | |
| Macroalgae proportion | 2 | 0.2 | 2.8 | 0.8 | 10.1 | 0.1 | 4.2 | -2.8 | 3.4 | -0.7 | 4.8 |
| | 5 | 0.04 | 2.1 | -2.4 | 4.9 | 0.3 | 3.6 | -0.5 | 3.2 | -2.2 | 1.8 |
| Community composition | 2 | -0.6 | -0.2 | -1.9 | -0.1 | -0.8 | 0.1 | -0.2 | 1.1 | -1.4 | -0.2 |
| | 5 | -1.0 | -0.3 | -2.4 | -0.5 | -1.1 | -0.02 | -0.4 | 0.5 | -2.3 | -1.1 |

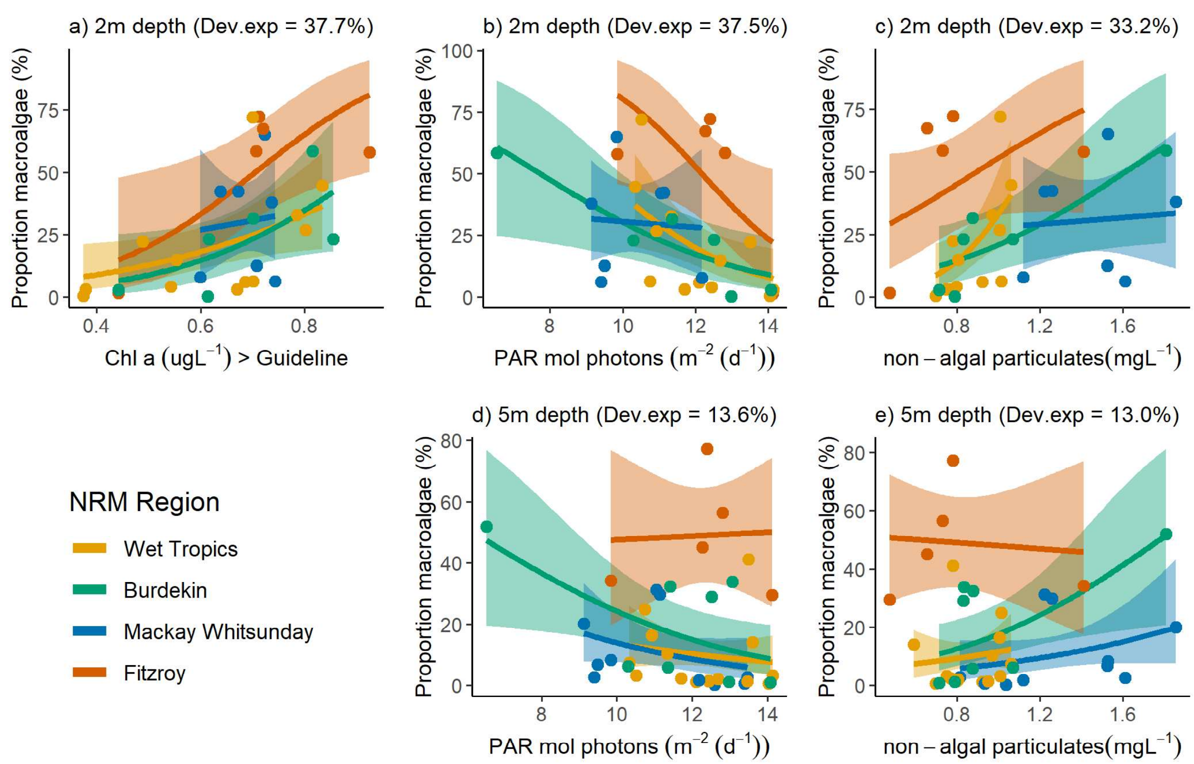


Figure 35 Relationship between the proportion of macroalgae indicator and water quality variables. Plots present predicted relationship within each region. Confidence intervals in predicted slopes are provided in (Table 15). Chl *a* concentration, expressed as mean wet-season exceedance of Guideline values (0.63µgL-1).

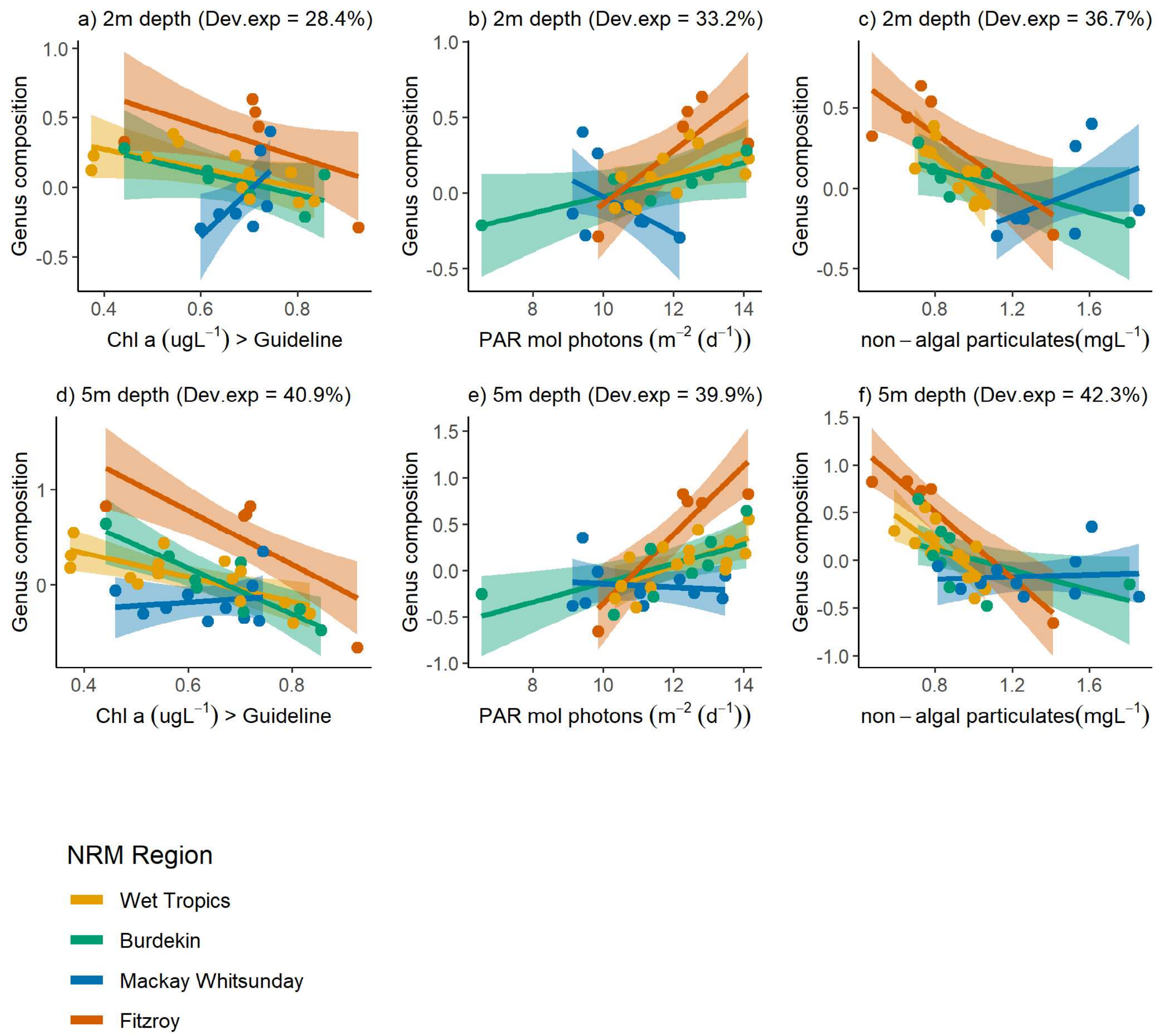


Figure 36 Relationship between coral community composition and water quality variables. Plots present predicted relationship within each region. Confidence intervals in predicted slopes are provided in Table 15. Colour coding for regional trends are consistent with those in above figures. Chl *a* concentration, expressed as mean wet-season exceedance of Guideline values (0.63µgL-1).

### 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 17, Figure 37). Importantly, these relationships consider only the contemporary influence of environmental conditions on the indicators during recovery periods. Any influence of water quality on the severity of response to disturbance events, or lagged responses of indicators will not be included. In the case of lagged influences, such as the initial decrease then post-disturbance increases in macroalgal cover that has been observed on several occasions following cyclones and floods, this will result in the underestimation of the response. Relationships between loads of particulate and dissolved nitrogen, total suspended solids and Coral Index change generally mirror those described for discharge (Table 17). This is not surprising as nutrient loads in rivers are correlated with river discharge.

The proportion of coral reefs exposed to water quality risk category 3, as estimated by frequency of exposure to different colour classes of water (Moran *et al.* 2022) was also negatively related to changes in the Coral Index the Wet Tropics, Burdekin and Fitzroy regions. In the Burdekin and Fitzroy regions this summary of observed water quality explained a similar proportion of the variability in Coral Index scores as end of catchment load and discharge estimates. The relationship in the Mackay Whitsunday was not monotonic with positive changes in Coral Index scores at both ends of the exposure range. Also investigated where exposure categories 2 and 4 neither explained more variability in index scores than category 3.

Table 17 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. A (#) marks relationships that are curved and do not indicate a unidirectional relationship. 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)) | Exposure Risk 3 |
|---|---|---|---|---|---|
| Wet Tropics | 16.7% | 16.2% | 14.1% | 16.2% | 5.8% |
| Burdekin | 13.7% | 10.4% | 13.4% | 9.4%* | 14.5 |
| Mackay-Whitsunday | | | 7.3%* | | 11.3%# |
| Fitzroy | 26% | 25.1% | 23.8% | 24.9% | 21.1% |

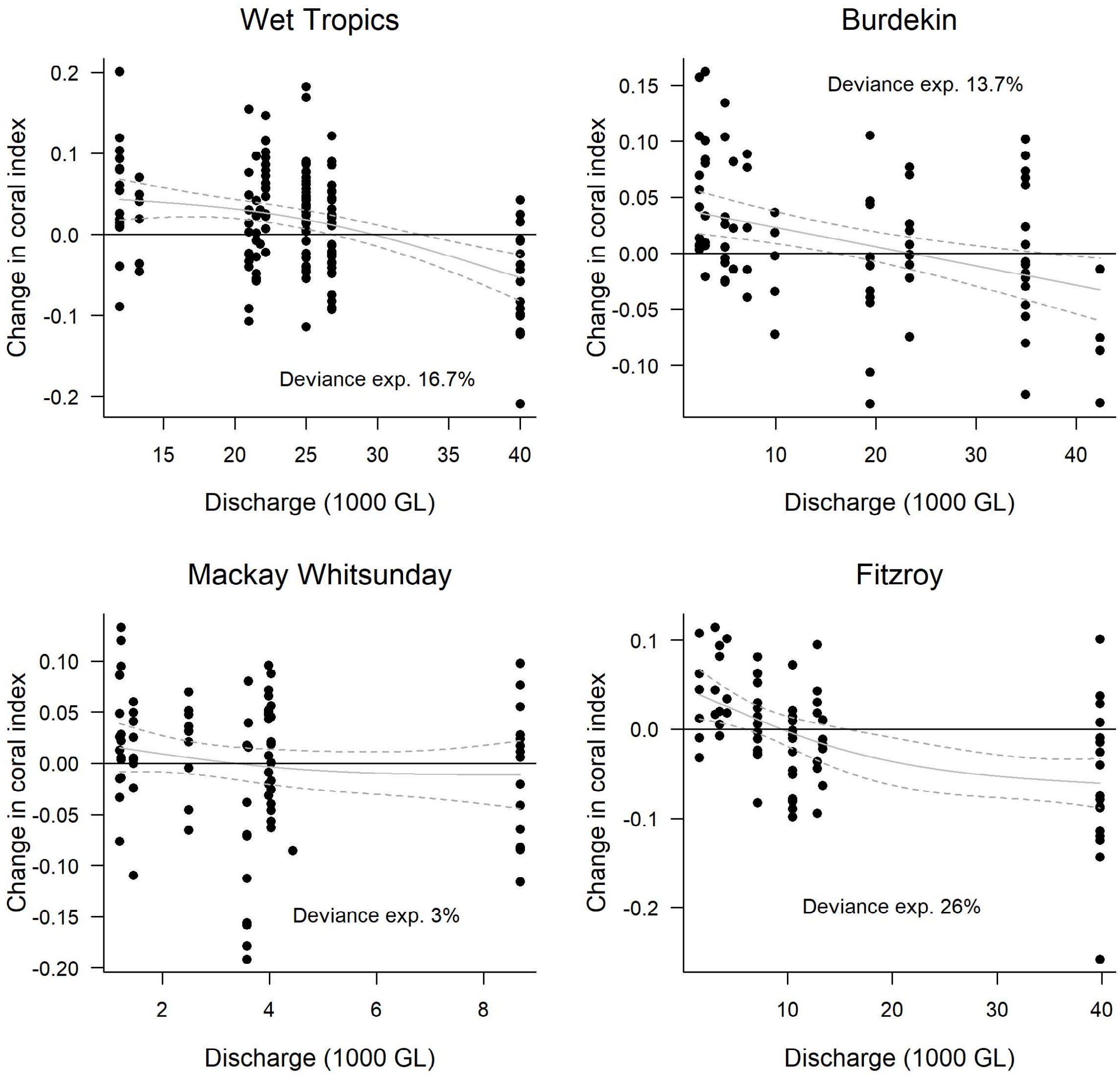


Figure 37 Relationship between the Coral Index and freshwater discharge from local catchments. Plotted points represent observed change in the Coral Index score at each reef and depth over a two-year period. Observations following years for which acute disturbances impacted communities in the period 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 and 2020 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 further loss of coral cover through to 2021 was attributed to the subsequent mortality of these stressed corals. Potentially confounding the influence of the 2020 coral bleaching event was that coral cover prior to the bleaching event was rapidly increasing. A case in point was Barren Island in the Fitzroy region. Here, surveys in 2020 revealed multiple recently dead and dying corals at 2 m depth, however, despite this clear mortality coral cover had increased in 2020, a trend that continued in 2021.

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 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 near 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 in the Wet Tropics and are there 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 form the adjacent catchments. Such exposure to low salinity waters add to the list of disturbances faced by coral communities near rivers and will 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 process 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 rivers. 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, 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 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*. 2022) 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.* 2022) 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 variability in ambient marine water quality. 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. Both the cycling of flood delivered material and the loads delivered during average river flows become a chronic pressure for inshore corals.

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.* 2022). 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 2021 (Moran *et al.* 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. 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.* 2013a, 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 2021 reflects both the strength of this recovery and the influence of more recent disturbance events.

In 2021:

- 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. Ongoing presence of crown-of-thorns starfish have limited the Coral Index score with coral cover tending to recover well when numbers of these starfish are low. Low densities of juvenile corals also supressed scores in the region.
- Herbert Tully sub-region score remains in 'good' condition following strong recovery from the 'poor' level observed in 2013. Thermal bleaching in 2017 and again in 2020 caused a slight pause to otherwise consistent improvement.
- Burdekin region score remains 'moderate'. A slight decline since 2020 was attributed to lagged impacts of coral bleaching in 2020. In the longer-term, Coral Index scores have substantially improved since 2013, although high levels of macroalgae continue to compete with corals at some reefs.
- The score for Mackay-Whitsunday region remains poor but has increased since 2020. This increase represents the first sign that coral communities are recovering since being severely impacted by cyclone Debbie in 2017.
- Slow recovery of reefs in the Fitzroy region stalled in 2021. A lagged impact of bleaching in 2020 compounded the pressures imposed by high cover of macroalgae to limit reef recovery.

Variability in the condition of coral communities along water quality gradients highlight the pressure that poor water quality imposes on coral communities. In 2021, Reef-wide Coral Index scores at 2 m depths declined with increasing concentration of Chl *a* in surrounding waters. Of the individual indicators this relationship was most evident in coral cover and macroalgae scores. Significant decline in coral cover scores with increasing Chl a concentration was observed the Wet Tropics regions. Reef-wide coral cover scores also showed a negative relationship to turbidity (non-algal particulate concentration) gradients. This relationship was observed at both 2m and 5 m depths and was most evident in the Wet Tropics and Fitzroy regions.

The relatively low variability in water quality conditions among MMP reefs in the Mackay-Whitsunday region reduces the scope for strong differentiation 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.

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. Relating the data underpinning the macroalgae indicator to reef-level water quality demonstrates there is a higher proportion of macroalgae in algal communities at reefs in high nutrient and turbid waters (high concentrations of Chl *a* and non-algal particulates and low levels PAR). Similarly, coral community composition changes along gradients of these same water quality variables. 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).

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 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. This combination, along with the distance of reefs from river mouths, will reduce the relative influence of run-off compared with hydrodynamic processes on the variability in conditions, and in particular turbidity, experienced by corals. Indeed, the 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 a selection for turbidity tolerance that is likely to offer a degree of resistance to additional pressures imposed by variable run-off; a point raised by Morgan *et al.* (2016). Influential in the results 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, our estimation of relative temperature stress - based on *in situ* loggers rather than satellites and expressed as degree heating days (available from the Bureau of Meteorology), implicate high summer temperatures as the likely stressor.

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 (McKenzie *et al.* 2022) 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 is 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 2021, coral cover was generally higher at reefs with low concentrations of non-algal particulates and, at 2 m depths, low Chl *a* concentration. 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.

### 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 2021 all reefs were surveyed. Observations excluded from informing the cover change indicator were from sites that had experienced an acute disturbance since the last survey: seven of fourteen reef and depth combinations in the Johnstone Russell Mulgrave sub-region (crown-of-thorns starfish), three of fourteen reef and depth combinations in the Burdekin region and two of ten reef and depth combinations in the Fitzroy region, both due to legacy of bleaching in 2020. Across all reef and depth combinations, the 2021 scores were based on single observations of change at 15 of the 70 locations with no scores based on caried forward estimates. This means that the cover change scores reported are based on recent estimates across all regions.

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 is that many of the coral communities are dominated by slow growing species, especially at five metre depths, suggesting 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.

In contrast, the moderate to high scores for the cover change indicator in Burdekin region and Wet Tropics sub-regions in 2021 continue to demonstrate the ongoing recovery potential exhibited by these of coral communities, especially those in less turbid waters.

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, discussed above, demonstrate the dynamic nature of coral community selection occurring on inshore reefs. Sensitive species gain a foothold during relatively benign conditions only to be removed during periods when environmental conditions are beyond their tolerance.

In 2021, the composition indicator scores remained relatively stable following a period of improvement over recent years. There were two exceptions. In Mackay-Whitsunday, composition scores declined through to 2020 following the impact of cyclone Debbie in 2017. In the Barron Daintree sub-region disproportional losses of *Acropora* relative to increases of other taxa have seen scores decline at 2 m depths. Apart from these exceptions, the results demonstrate that recovery of coral cover included increased representation of species sensitive to poor water quality. In general, the coral composition indicator has tended to track the trend in coral cover, indicating the disproportionate loss, and subsequent recovery, of genera sensitive to water quality. This does not necessarily imply poor water quality as a causative agent as the genus most susceptible to poor water quality, *Acropora,* is also susceptible to cyclones (Fabricius *et al*. 2008), thermal bleaching (Marshall & Baird 2000), and a preferred prey group for crown-of-thorns starfish (Pratchett 2007). Over the longer term, however, there is evidence that the representation of *Acropora* on reefs in the Burdekin region has declined since the mid-20th century, possibly due to increased run-off from the adjacent catchments (Roff *et al.* 2013). Branching *Acropora* were one group identified by Roff *et al.* (2013) as showing reductions in contemporary communities. While branching *Acropora* have recruited and contributed to increased coral cover across the region, losses of cover at two metre depth of Havannah Island since 2017 were primarily the result of large stands of *Acropora pulchra* being killed by disease. Bleaching in 2020 has further reduced the cover of remaining branching *Acropora* species at this reef.

That this indicator tends to reiterate changes in coral cover, due to its responsiveness to fluctuations in the cover of *Acropora,* means 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.

### 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) 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, for example after flood events, the persistence of macroalgae suggest that ambient water quality levels are important for the maintenance of high macroalgal cover. While reef-specific thresholds for macroalgae allow for increased abundance of macroalgae in response to naturally occurring gradients of water quality, their cover in 2021, 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 similar to macroalgae, 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 biological factors (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 and located reefs 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 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 a 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, adequate light (prevalence is limited by turbidity at five metre depths) and sufficient water movement to preclude the build-up of fine sediments on the substrate (Thompson *et al.* 2017).

In terms of light availability and water movement, the preferred habitat for brown macroalgae overlaps strongly with that of some corals, particularly the fast-growing Acroporidae, highlighting the direct competition for space between these groups. The correlation between high prevalence of macroalgae and Chl *a* concentration implies that a reduction in the availability of nutrients has the potential to shift the competitive relationship between macroalgae and coral and reducing 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) whereas 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 effects on larval settlement and survival (Tanner 1995, McCook *et al.* 2001, Birrell *et al.* 2005, 2008a, b, Johns *et al.* 2018). 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.

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.

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[3]) and differences in population genetics of corals (Mackenzie *et al.* 2004) in the Burdekin region both indicate limited connectivity between Halifax Bay and reefs further offshore. Perhaps the most compelling evidence for low larval supply to some inshore reefs has been observed at Snapper 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 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, Coral Index scores have remained relatively stable since 2016. In 2021, the cover change indicator remains categorised as good, the composition indicator increased to good, and all other indicators remained moderate. While there have been 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-regions. Reefs in this region escaped exposure to high levels of thermal stress in 2020 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[4] with 24,354 individuals removed from the monitoring reefs since 2013, 9789 of these from Fitzroy Island and the Frankland Group in the year preceding surveys in 2021. MMP surveys have continued to note the presence of crown-of-thorns starfish across a range of size classes with densities, in 2021 outbreak levels were observed in the Frankland Group and at High

[3] Connie 2.0, CSIRO Connectivity Interface, [CSIRO connie2]

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

East although numbers were well below those observed in 2020. That crown-of-thorns have not had greater impact in the Johnstone Russell-Mulgrave region appears due to most individuals both observed by the MMP and removed by the control program being in smaller size classes, which have lower feeding rates than larger individuals. This feeding will however have put downward pressure on the cover change and composition indicator scores. Also, the rapid growth of *Acropora* colonies that has been a feature of these reefs in recent years 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. However, persistently very poor scores for the macroalgae indicator at Dunk South and 2 m depths of Bedarra, and Snapper North limit the region's overall index scores and focus on the most direct influence of water quality to these locations.

### 5.3.2 Burdekin

The Coral Index score for the Burdekin region declined slightly in 2021 but remains 'moderate'. This decline represents a slight pause in the recovery of coral communities in the region since 2013. This slight decline is due in part to a lagged response to the thermal stress that led to coral bleaching at most reefs in 2020, but also the ongoing pressure imposed by high cover of macroalgae.

Influential in the recent decline has been loss of hard coral cover at the 2 m depth of Havannah in recent years. Between 2011 and 2015 hard coral cover rapidly increased, although this also occurred at other reefs the rate of increase at Havannah 2 m was remarkable. Contributing strongly to this increase were several species of branching (arborescent) *Acropora*. Since 2016 elevated temperatures in 2016, 2017 and 2020 led to bleaching and coincidental 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 1.6% in 2021. This disproportionate loss of branching *Acropora* reduced the composition score at this reef form 1 in 2020 to 0 in 2021. It appears several of the species that contributed to the very rapid recovery of 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).

Across the region there were small declines in the density of juvenile corals at most reefs. As the juvenile coral density estimates aggregate colonies that are likely to have settled over the preceding two or three summers, these declines will reflect any impact of the 2020 marine heat wave on the survival of juvenile corals, and particularly the newly settled cohort from the previous summer that are likely to be overlooked in their first year due to their very small size. A rise in the incidence of coral disease across the region in 2021 further implicate environmental stress as contributing to the downturn in juvenile coral scores.

There was also a slight reduction the macroalgae indicator due to increases in macroalgae at 2 m depths at Havannah and Lady Elliot. Across the region there is a clear distinction between the very low cover of macroalgae on the reefs at Palms East and Palms West compared to the reefs in more turbid and nutrient rich settings. Persistent stands of large brown-algae, *Sargassum* (Magnetic and Pandora 2 m) and *Lobophora* (Havannah 5 m, Havannah North) and red algae at Lady Elliot 2 m drive the relationship between high cover of macroalgae and poor water quality within the region. Johns *et al*. (2018) show that at Havannah North macroalgae were at sufficient density to severely limit juvenile settlement and this is likely to have trapped the benthic community in a macroalgae dominated phase for two decades. It is noteworthy that the surveys in 2019 demonstrated the first evidence of coral recovery at this reef in unison with declines in cover of macroalgae. However, this recovery was disrupted by the 2020 bleaching event with observations in 2021 recording a loss of coral cover and increase in macroalgae.

Historically, recovery from acute events in this region has been slow (Sweatman *et al*. 2007, Cheal *et al.* 2013). Monitoring of coral settlement during 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[5]) 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 are likely to directly influence larval supply. Exacerbating any supply-side limitation to coral recruitment is the persistently high cover of macroalgae at several reefs, which is likely to suppress recruitment success (Tanner 1995, McCook *et al.* 2001, Birrell *et al.* 2005, 2008a, b, Johns *et al.* 2018).

The recent improvement in Coral Index scores has coincided with a prolonged dry period, although punctuated by flooding 2019, in general this was a period of relatively low loads of nutrient and sediments being delivered from the adjacent catchments. Over the last sixteen years reef resilience has been inversely related to discharge, nutrient and sediment loads from the region's rivers. It was not until 2014, a year into a period of below median discharges from the region's rivers, that the average rates of hard coral cover increase began matching modelled expectations, and they continue to do so.

Stress to corals during periods of high catchment discharge were observed as increased disease prevalence in 2007-2009. Over that period discharge from the region's rivers were consistently above median levels, in contrast to the below median discharges of the preceding years. Moderate increases in coral disease were also noted in 2011 and 2019, again following high catchment discharges, although the severe impact of cyclone Yasi confounds the 2011 observation. In combination, these results are consistent with the well documented link between increased run-off and stress in coral communities, expressed as increased levels of coral disease (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). Increased levels of disease recorded in 2018 and again in 2021 are likely associated with the reduced fitness of corals due to thermal bleaching (Morris *et al*. 2019) in the previous year, although potentially exacerbated by above median discharges from the region's rivers.

### 5.3.3 Mackay-Whitsunday

The Coral Index in the Mackay-Whitsunday region declined dramatically through to 2019, due to the impacts of cyclone Debbie. In 2021 the coral communities showed the first 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. Conversely, coral cover scores were generally 'good', except for a short decline to 'moderate' levels due to damage imposed by cyclone Ului in 2010. 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 low capacity for rapid recovery of coral cover, especially at the five metre depths.

[5] Connie 2.0, CSIRO Connectivity Interface, [CSIRO connie2]

With the severe loss of coral cover at many sites successful recovery will rely heavily on the recruitment and survival of juvenile corals. Although the density of juvenile corals increased in 2021, they remain low at most reefs and suggest a bottleneck for the recovery of these communities.

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). Improved scores for the macroalgae indicator in 2021 largely reflect the return toward pre-disturbance levels of macroalgae at reefs where only a minor post-disturbance bloom occurred. 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 depths at Daydream Island and Double Cone Island and will almost certainly be putting downward pressure on the recovery of coral communities at these locations. Of concern is that at both reefs the algal community is changing toward higher proportion of brown algal species including *Sargassum* and *Lobophora* with *Lobophora* also notable among the algal community at Dent Island in 2021. 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 short and long-term water quality index are gradually improving with the long-term index returning to moderate in 2021, 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 on low densities of juvenile corals a slow recovery of coral communities at the worst impacted reefs remains likely.

### 5.3.4 Fitzroy

The Coral Index declined in 2021 with coral community condition remaining poor. In 2020 slow recovery of coral communities stalled as corals bleached in response to a marine heat wave. While the recent decline likely reflects the legacy of this thermal stress, the ongoing poor condition of communities suggests environmental conditions have been limiting their recovery.

In 2021 coral cover declined. In the absence of any severe weather, and with consideration of the level of coral bleaching observed in 2020 (Thompson *et al.* 2021), this decline likely reflects corals that failed to recover and subsequently died. The flow-on effect of this poor performance of corals since 2020 was a decline in the cover change score. Across the region, there was also a decline in the density of juvenile corals, that, as discussed previously (section 5.3.2) is likely to capture any impact of the 2020 heat wave on survival of newly settled corals.

The decline in densities of juvenile corals may also be related to the pressures imposed by macroalgae (see sections 5.2.5, 5.2.6). Although the score for the macroalgae indicator only declined at two reefs, this is because scores in 2020 were already at zero at most reefs. In 2021 only a single location, Barren 2 m had a score above zero for this indicator, and mean cover of macroalgae among the reefs surveyed had increased from 24% in 2020 to 30% in 2021. Across the region high cover of macroalgae was precipitated by loss of coral cover, as the algae rapidly occupied vacated space (Diaz-Pulido 2009, Ceccarelli *et al.* 2020) of concern is the persistence of these algae that continue to form dense mats that are almost certainly limiting the recovery of coral cover. Perhaps most concerning is Middle Island where, when first visited in 2005 there was almost no macroalgae. Cover of macroalgae at Middle Island is now of 50% and includes a high proportion of the persistent *Lobophora* and *Sargassum*.

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 the low salinity plumes that kill the corals (van Woesik 1991, data herein, Jones & Berkelmans 2014). In addition, the negative relationship between the rate of change in Coral Index scores and discharge from the Fitzroy River demonstrates the wider impact of major flood events on coral community condition within Keppel Bay. Of note were elevated levels of disease following major flood events supporting hypotheses that either reduced salinity (Haapkylä *et al.* 2011) or increased nutrient enrichment (Vega Thurber *et al.* 2013) were sufficiently stressful to facilitate coral disease. Reduction in light levels over extended periods of time due to increased concentrations of suspended sediments delivered by the floods, as well as dense plankton blooms following the floods, is another plausible explanation for reduced fitness of corals (Cooper *et al*. 2007) and is supported by the clear relationship between river derived loads and change in Coral Index scores in this region.

Variation among reefs in the recovery of coral communities further illustrates the role of water quality in supressing coral community resilience. 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 the complete lack of recovery of coral cover at Peak Island or Pelican Island between 2011 and 2019 although cover had shown a modest increase at Pelican in 2020.

A bottleneck for recovery of coral communities is the low density of juvenile corals. Although the juvenile coral indicator scores have improved since 2014, region densities of juvenile corals remain 'poor'. Recruitment of corals is likely limited by a combination of larval supply and the negative influence of high macroalgae cover. The prevalence of macroalgae is highly likely to be supressing recruitment processes (Johns *et al*. 2018). 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). From these results we cannot distinguish between the relative roles of reduced local brood-stock and high cover of macroalgae.

Despite the influences of macro-algae and low densities of juvenile corals and the repeated exposure to high thermal stress in recent years, coral cover had improved at almost all reefs since low points in 2014. This increasing recovery demonstrates the ongoing resilience of these communities when not exposed to the low salinity and high sediment and nutrient loads delivered by Fitzroy River floods.

## 5.4 Management response

Coral reefs in general are subjected to cumulative impacts of acute disturbances and environmental pressures. 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 2021 noted a decline in densities of crown-of-thorns starfish, however moderate densities across a range of size classes were still observed demonstrating an ongoing pressure 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 (Ceccarelli *et al.* 2018). Grazing by fish and urchins is also an important natural control for macroalgae and for any pressures influencing the ecosystem services offered by grazing organisms should be considered for suitable management responses.

In most NRM regions coral communities retain the ability to recover following impacts from acute disturbances. The rate of this recovery is however suppressed relative 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 that demonstrates 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 Conclusion

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 form 2021 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 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. Firstly, coral response-thresholds to the cumulative pressures associated with water quality will be spatially variable because of the selection and acclimatisation of corals in response to location-specific conditions. Secondly, extrinsic variability, due to weather, along with low concentrations for many constituents of water quality, limits the ability to quantify pressures resulting from run-off at scales relevant to the communities monitored. Finally, effects of interactions between water quality stressors and with other acute disturbances have only been quantified for a limited combination of pressures and few coral species (e.g., Uthicke *et al.* 2016). In combination, these knowledge gaps limit the ability to quantify thresholds for water quality that are appropriate to the diversity of coral communities found on inshore reefs. However, focusing on the response of the coral communities (as measured by differences in Coral Index 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 recently 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 have created 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 as observations to date suggest that low juvenile densities combined with low rates of cover increase will result in slow recovery of these communities. Of some 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 A 1 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 A 2 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 A 3 Thresholds for proportion of macroalgae in the algae communities.

| Reef | 2 m Depth | | 5 m Depth | | | Reef | 2 m Depth | | 5 m Depth | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Upper | Lower | Upper | Lower | | | Upper | Lower | Upper | Lower |
| Barnards | 23 | 4.8 | 20.8 | 1.7 | | Keppels South | 23 | 3.9 | 24 | 1.7 |
| Barren | 13 | 3.7 | 12.6 | 1.6 | | King | 23 | 6.2 | 24.8 | 1.8 |
| Bedarra | 23 | 5.3 | 15.6 | 1.9 | | Lady Elliot | 23 | 6.1 | 15.3 | 1.9 |
| Border | | | 8.2 | 1.4 | | Langford | | | 7.9 | 1.4 |
| Daydream | 13.5 | 3.5 | 10.4 | 1.5 | | Low Isles | | | 8.9 | 1.4 |
| Dent | 11.6 | 3.5 | 10.2 | 1.5 | | Magnetic | 23 | 6.4 | 19 | 2 |
| Double Cone | 8.9 | 3.4 | 7.6 | 1.4 | | Middle Rf | 21.9 | 5.5 | | |
| Dunk North | 23 | 4.6 | 13.5 | 1.7 | | Middle | 23 | 5.2 | 23 | 1.8 |
| Dunk South | 23 | 5.3 | 15.6 | 1.9 | | North Keppel | 23 | 5.1 | 22.6 | 1.8 |
| Fitzroy East | 11.7 | 3.5 | 10 | 1.5 | | Palms East | 12.2 | 3.6 | 10.5 | 1.5 |
| Fitzroy West | 12.5 | 3.3 | 13.3 | 1.5 | | Palms West | 12.8 | 3.4 | 17.5 | 1.5 |
| Franklands East | 12.2 | 3.4 | 10.5 | 1.5 | | Pandora North | | | 13.1 | 1.6 |
| Franklands West | 11.4 | 3.4 | 15.8 | 1.5 | | Pandora | 23 | 4.7 | 16.2 | 1.6 |
| Green | | | 11.9 | 1.6 | | Peak | 23 | 6.3 | 19.1 | 2 |
| Havannah North | | | 21.7 | 1.5 | | Pelican | 23 | 6.4 | 18.8 | 2 |
| Havannah | 18.2 | 3.4 | 25 | 1.6 | | Pine | 18.3 | 4.4 | 11.2 | 1.6 |
| Hayman | | | 9.4 | 1.4 | | Seaforth | 11.8 | 3.4 | 10.2 | 1.4 |
| High East | 11.2 | 3.4 | 13 | 1.4 | | Shute Harbour | 17.6 | 4.2 | 11.7 | 1.6 |
| High West | 22.4 | 4.4 | 12.1 | 1.6 | | Snapper North | 18.7 | 4.4 | 11.3 | 1.6 |
| Hook | 9.3 | 3.4 | 8.1 | 1.4 | | Snapper South | 23 | 4.4 | 13.1 | 1.6 |
| Keppels South | 23 | 3.9 | 24 | 1.7 | | | | | | |

Table A 4 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 A 5 Annual freshwater discharge for the major Reef Catchments. Values represented as proportional to the median (1986-2016). Flows corrected for ungauged area of catchments as per Moran *et al.* (2022). 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 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Wet Tropics | Daintree River | 1512054 | 1.8 | 1 | 1.3 | 0.9 | 1.6 | 2.2 | 1.3 | 1 | 2.4 | 1.1 | 0.9 | 1.1 | 1.2 | 3.3 | 0.6 | 1.0 |
| | Mossman River | 858320 | 1.5 | 1 | 1.1 | 0.9 | 1.3 | 1.7 | 1.3 | 1 | 1.6 | 0.7 | 0.9 | 1.0 | 1.2 | 1.9 | 0.6 | 1.1 |
| | Barron River | 574567 | 1.6 | 0.9 | 3.4 | 1.6 | 1 | 4 | 1.6 | 0.6 | 1.3 | 0.7 | 0.3 | 0.5 | 1.6 | 2.7 | 0.6 | 1.1 |
| | Russell - Mulgrave River | 2600465 | 1.2 | 1.1 | 1.1 | 1 | 1.1 | 1.8 | 1.3 | 0.8 | 1.2 | 0.7 | 0.7 | 0.7 | 1.3 | 1.4 | 0.7 | 1.2 |
| | Johnstone River | 3953262 | 1.2 | 1.1 | 1 | 1.1 | 1 | 2 | 1.1 | 0.8 | 1.1 | 0.6 | 0.7 | 0.8 | 1.2 | 1.2 | 0.7 | 1.1 |
| | Tully River | 3241383 | 1.2 | 1.3 | 1.1 | 1.2 | 1 | 2.1 | 1 | 0.9 | 1.2 | 0.8 | 0.8 | 0.9 | 1.2 | 1.2 | 0.7 | 1.3 |
| | Murray River | 380472 | 1.4 | 1.1 | 1 | 1.5 | 0.8 | 3.5 | 1.7 | 0.8 | 1.2 | 0.3 | 0.8 | 0.8 | 1.4 | 1.4 | 0.5 | 1.6 |
| | Herbert River | 3556376 | 1.2 | 1.2 | 1 | 2.9 | 1 | 3.6 | 1.3 | 0.9 | 1.2 | 0.3 | 0.5 | 0.6 | 1.5 | 1.6 | 0.4 | 1.8 |
| Burdekin | Black River | 208308 | 1 | 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 |
| | Ross River | 261907 | 0.8 | 1.7 | 2.3 | 3.2 | 1.4 | 3.0 | 2.2 | 0.6 | 0.7 | 0.2 | 0.4 | 0.4 | 0.1 | 6.3 | 0.4 | 0.2 |
| | Haughton River | 419051 | 1.1 | 2.2 | 3.3 | 4.4 | 2.1 | 4.7 | 3.2 | 0.9 | 1 | 0.3 | 0.5 | 0.7 | 1.4 | 5.6 | 0.6 | 1.1 |
| | Burdekin River | 4406780 | 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 | 0.5 | 1.9 |
| | Don River | 508117 | 0.9 | 2.1 | 4.8 | 3.8 | 1.6 | 7.1 | 2.2 | 1.4 | 0.8 | 0.4 | 0.3 | 2.2 | 0.6 | 2.7 | 0.8 | 0.9 |
| Mackay-Whitsunday | Proserpine River | 284542 | 0.8 | 1.6 | 2.3 | 1.7 | 2.9 | 5.2 | 2.4 | 1 | 0.8 | 0.2 | 0.4 | 1.9 | 0.6 | 3.0 | 0.7 | 0.5 |
| | O'Connell River | 478097 | 0.5 | 1.7 | 2.2 | 1.5 | 2.5 | 4.8 | 2 | 1.1 | 0.8 | 0.2 | 0.6 | 1.9 | 0.5 | 2.6 | 0.6 | 0.5 |
| | Pioneer River | 692342 | 0.1 | 1.4 | 2.2 | 1.4 | 2.3 | 5.2 | 2.3 | 1.7 | 0.9 | 0.2 | 0.9 | 2 | 0.4 | 1.7 | 0.6 | 0.3 |
| | Plane Creek | 309931 | 0.1 | 1.4 | 2.7 | 1.2 | 2.7 | 4.1 | 2.5 | 1.7 | 0.7 | 0.2 | 0.8 | 2.4 | 0.3 | 1.1 | 1.0 | 0.4 |
| Fitzroy | Water Park Creek | 97115 | 0.2 | 0.5 | 2.5 | 1 | 2.8 | 4.8 | 1.5 | 5.2 | 2.9 | 2.2 | 1.8 | 2.6 | 1.4 | 0.7 | 1.5 | 1.8 |
| | Fitzroy River | 2852307 | 0.2 | 0.4 | 4.4 | 0.7 | 4.1 | 13.3 | 2.8 | 3 | 0.6 | 0.9 | 1.2 | 2.2 | 0.3 | 0.5 | 0.9 | 0.1 |

Table A 6 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% 2m, 12% 5m) |
| | | 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 5m), COTS 2021 (45% 5m) |
| | | 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% 2m) |
| | | 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% 2m, 29% 5m) |
| | | 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% 5m) |
| | | Green | | | 12 % | COTS: 1994 (21%), 1997 (55%), 2011-2013 (44%), 2014-2015 (47%) |

Table A 6 continued

| (sub-)region | Reef | Bleaching | | | | Other recorded disturbances |
|---|---|---|---|---|---|---|
| | | 1998 | 2002 | 2017 | 2020 | |
| Herbert Tully | Barnards | 0.93 | 0.80 | 17% (2 m) | | cyclone Larry 2006 (95% at 2 m 87% at 5 m), cyclone Yasi 2011 (53% at 2 m, 24% at 5 m), Bleaching 2018 (10% at 5 m), Disease 2021 (18% 2m, 9% 5m) |
| | 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)<br>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)<br>6% (5 m) | 20% (2m)<br>12% (5m) | 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)<br>10% (5 m) | 16% (2m)<br>10% (5m) | Bleaching 2018 ongoing from 2017 (26% at 5 m) |

Table A 6 continued

| Region | Reef | Bleaching | | | | Other recorded disturbances |
|---|---|---|---|---|---|---|
| | | 1998 | 2002 | 2017 | 2020 | |
| Burdekin | Palms East | 0.93 | 0.80 | | | cyclone Larry 2006 (23% at 2 m, 39% at 5 m), cyclone Yasi 2011 (83% at 2 m and at 5 m) |
| | Palms West | 0.92 (83%) | 0.80 | 30% (2 m) 15% (5 m) | | Unknown 1995-1997 although possibly cyclone Justin (32%), cyclone Larry 2006 (15% at 2 m), Storm 2010 (68% at 2 m) |
| | Lady Elliott Reef | 0.93 | 0.85 | | 26% (2 m) 8% (5 m) | cyclone Yasi 2011 (86% at 2 m, 45% at 5 m) |
| | Pandora Reef | 0.93 (21%) | 0.85 (2%) | 33% (2 m) | 18% (2 m) | cyclone Tessie 2000 (9%), cyclone Larry 2006 (80% at 2 m, 34% at 5 m), Storm 2009 (37% at 2 m, 56% at 5 m), cyclone Yasi 2011 (30% at 2 m, 57% at 5 m) |
| | Pandora North | 11% | | 5 %* | n/a | cyclone Yasi 2011 (25%) |
| | Havannah | 0.93 | 0.95 | 37% (2 m) 11% (5 m) | 33% (2 m) 8% (5 m) | Combination of cyclone Tessie and COTS 1999-2001 (66%) cyclone Yasi 2011 (35% at 2 m, 34% at 5 m), Disease 2016 (9% at 2 m), Bleaching ongoing impact of 2017 recorded in 2018 (26% at 2 m, 16% at 5 m), Disease 2019 (23% at 2 m), Post 2020 bleaching (2021, 26% 2m) |
| | 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% 5m) |

Table A 6 continued

| Region | Reef | Bleaching | | | | Other recorded disturbances |
|---|---|---|---|---|---|---|
| | | 1998 | 2002 | 2017 | 2020 | |
| Mackay-Whitsunday | Hook | 0.57 | 1 | | 27% (2 m) 20% (5 m | Coral Bleaching Jan 2006, probable although not observed as we did not visit region at time of event. Same for other reefs in region, cyclone Ului 2010 (31% at 2 m,17% at 5 m), cyclone Debbie 2017 (recorded in 2018) (83% at 2 m, 45% at 5 m) |
| | Dent | 0.57 (32%) | 0.95 | ** | | Disease 2007(17% at 2 and at 5 m), cyclone Ului 2010 most likely although reef not surveyed in that year (21% at 2 m, 27% at 5 m), cyclone Debbie 2017 (48% at 2 m, 38% at 5 m), Cyclone Debbie 2017 (48% at 2 m, 38% at 5 m), Disease 2019 (44% at 2 m, 25% at 5 m), Disease 2021 (16% at 5 m) |
| | Seaforth | 0.57 | 0.95 | ** | 8% (2 m) | Flood 2009 (16% at 2 m,, 22% at 5 m), cyclone Debbie 2017 (45% at 2 m, 26% at 5 m) |
| | Double Cone | 0.57 | 1 | ** | 15% (2 m) 3% (5 m) | Flood 2009( 13% at 2 m), cyclone Ului 2010 (26% at 2 m, 12% at 5 m), cyclone Debbie 2017 (97% at 2 m, 74% at 5 m) |
| | Daydream | 0.31 (44%) | 1 | ** | 42% (2 m) 38% (5 m) | Disease 2008 (26% at 2 m, 20% at 5 m), cyclone Ului 2010 (47% at 2 m, 46% at 5 m), cyclone Debbie 2017 (98% at 2 m, 90% at 5 m) |
| | Shute Harbour | 0.57 | 1 | ** | 10% (2 m) | cyclone Ului 2010 (8% at 2 m), cyclone Debbie 2017 (48% at 2 m, 55% at 5 m) |
| | Pine | 0.31 | 1 | ** | 35% (2 m) | Flood 2009(14% at 2 and at 5 m), cyclone Ului 2010 (13% at 2 m, 10% at 5 m), Disease 2011(15% at 5 m), cyclone Debbie 2017 (74% at 2 m, 56% at 5 m), Disease 2019 (40% at 2 m, 29% at 5m) |
| | 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 A 6 continued

| Region | Reef | Bleaching | | | | Other recorded disturbances |
|---|---|---|---|---|---|---|
| | | 1998 | 2002 | 2006 | 2020 | |
| Fitzroy | Barren | 1 | 1 | 25% (2 m)<br>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)<br>41% (5 m) | 18% (2 m)<br>7% (5 m) | Storm Feb 2010 possible although not observed as site not surveyed that year. 2011 ongoing disease (26% at 2 m and 54% at 5 m) |
| | Middle Is | 1 (56%) | 1 (Nil) | 61% (2 m)<br>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% 2m) |
| | Keppels South | 1 (6%) | 1 (26%) | 27% (2 m)<br>28% (5 m) | 1% (2 m)<br>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% 5m) |
| | 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% 2m) |
| | 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 wide spread 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 A 7 Reef level Coral Index and indicator scores 2021. 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 | 0.48 | 0.79 | 1 | 0.14 | 0.5 | 0.58 |
| | Snapper North | 2 | 0.3 | 0.04 | 0 | 0.85 | 0 | 0.24 |
| | | 5 | 0.49 | 0.36 | 0.83 | 0.62 | 0 | 0.46 |
| | Snapper South | 2 | 0.56 | 0.17 | 0.89 | 0.69 | 0 | 0.47 |
| | | 5 | 0.87 | 0.08 | 0 | 0.68 | 0.5 | 0.43 |
| Moderate | | | **0.54** | **0.29** | **0.54** | **0.60** | **0.20** | **0.43** |
| Johnstone Russell-Mulgrave | Green | 5 | 0.22 | 1 | 0 | 0.52 | 0.5 | 0.45 |
| | Fitzroy East | 2 | 0.5 | 0.28 | 1 | 0.3 | 0.5 | 0.58 |
| | | 5 | 0.66 | 0.48 | 1 | 0.5 | 0 | 0.63 |
| | Fitzroy West | 2 | 1 | 0.51 | 0.91 | 0.87 | 1 | 0.86 |
| | | 5 | 0.84 | 0.65 | 1 | 1 | 0.5 | 0.8 |
| | Fitzroy West LTMP | 5 | 0.77 | 0.68 | 1 | 0.82 | 1 | 0.85 |
| | Franklands East | 2 | 0.83 | 0.29 | 1 | 0.61 | 1 | 0.75 |
| | | 5 | 0.39 | 0.41 | 0.81 | 0.45 | 1 | 0.61 |
| | Franklands West | 2 | 0.76 | 0.25 | 0 | 0.63 | 1 | 0.53 |
| | | 5 | 0.77 | 0.25 | 0 | 0.38 | 1 | 0.48 |
| | High East | 2 | 0.69 | 0.16 | 0 | 0.49 | 0.5 | 0.37 |
| | | 5 | 0.69 | 0.2 | 0.94 | 1 | 0.5 | 0.67 |
| | High West | 2 | 0.79 | 0.23 | 0.91 | 0.3 | 0 | 0.45 |
| | | 5 | 0.42 | 0.32 | 1 | 0.63 | 0.5 | 0.57 |
| Good | | | **0.66** | **0.41** | **0.68** | **0.66** | **0.64** | **0.61** |
| Herbert Tully | Barnards | 2 | 0.65 | 0.24 | 1 | 0.56 | 1 | 0.69 |
| | | 5 | 0.72 | 0.89 | 0.97 | 0.59 | 1 | 0.83 |
| | Dunk North | 2 | 0.64 | 1 | 0 | 0.84 | 0.5 | 0.6 |
| | | 5 | 0.49 | 1 | 0.27 | 0.77 | 0.5 | 0.6 |
| | Dunk South | 2 | 0.44 | 0.41 | 0 | 0.79 | 1 | 0.53 |
| | | 5 | 0.52 | 0.75 | 0 | 0.81 | 0.5 | 0.51 |
| | Bedarra | 2 | 0.2 | 0.47 | 0 | 0.36 | 1 | 0.41 |
| | | 5 | 0.29 | 0.99 | 0.53 | 0.73 | 1 | 0.71 |
| Good | | | **0.49** | **0.72** | **0.34** | **0.68** | **0.81** | **0.61** |
| Burdekin | Palms East | 2 | 0.61 | 0.15 | 1.00 | 0.50 | 1.00 | 0.65 |
| | | 5 | 0.71 | 0.43 | 1.00 | 0.74 | 1.00 | 0.78 |
| | Palms West | 2 | 0.53 | 0.46 | 1.00 | 0.70 | 0.00 | 0.54 |
| | | 5 | 0.47 | 0.54 | 1.00 | 0.07 | 0.00 | 0.42 |
| | Havannah North | 5 | 0.19 | 0.89 | 0.00 | 1.00 | 1.00 | 0.62 |
| | Havannah | 2 | 0.33 | 0.19 | 0.00 | 0.00 | 0.00 | 0.10 |
| | | 5 | 0.53 | 0.33 | 0.00 | 0.71 | 1.00 | 0.51 |
| | Pandora | 2 | 0.20 | 0.17 | 0.00 | 0.57 | 0.50 | 0.29 |
| | | 5 | 0.25 | 0.55 | 0.71 | 0.22 | 1.00 | 0.55 |
| | Pandora North | 5 | 0.75 | 0.33 | 0.00 | 0.38 | 0.50 | 0.39 |
| | Lady Elliot | 2 | 0.30 | 0.27 | 0.00 | 0.22 | 1.00 | 0.36 |
| | | 5 | 0.59 | 0.96 | 0.68 | 0.69 | 0.50 | 0.68 |
| | Magnetic | 2 | 0.36 | 0.17 | 0.00 | 0.48 | 0.50 | 0.30 |
| | | 5 | 0.42 | 0.29 | 0.00 | 0.73 | 0.50 | 0.39 |
| Moderate | | | **0.44** | **0.41** | **0.38** | **0.50** | **0.61** | **0.47** |

Table A 7continued

| Region | Reef | Depth | Coral cover | Juvenile | Macroalgae | Cover change | Composition | Coral Index |
|---|---|---|---|---|---|---|---|---|
| Mackay-Whitsunday | Hayman | 5 | 0.20 | 0.89 | 0.83 | 0.49 | 0.00 | 0.48 |
| | Langford | 5 | 0.19 | 0.42 | 1.00 | 0.00 | 0.00 | 0.32 |
| | Border | 5 | 0.52 | 0.60 | 1.00 | 0.00 | 0.00 | 0.42 |
| | Hook | 2 | 0.14 | 0.32 | 0.25 | 0.40 | 0.00 | 0.22 |
| | | 5 | 0.39 | 0.24 | 0.95 | 1.00 | 0.50 | 0.61 |
| | Double Cone | 2 | 0.03 | 0.18 | 0.00 | 0.20 | 0.00 | 0.08 |
| | | 5 | 0.20 | 0.16 | 0.00 | 0.00 | 0.00 | 0.07 |
| | Daydream | 2 | 0.00 | 0.41 | 0.00 | 0.00 | 0.00 | 0.08 |
| | | 5 | 0.05 | 1.00 | 0.23 | 0.13 | 0.00 | 0.28 |
| | Dent | 2 | 0.35 | 0.23 | 0.00 | 0.17 | 0.00 | 0.15 |
| | | 5 | 0.43 | 0.24 | 0.38 | 0.18 | 0.00 | 0.25 |
| | Shute Harbour | 2 | 0.72 | 0.34 | 0.85 | 0.56 | 1.00 | 0.70 |
| | | 5 | 0.32 | 0.52 | 0.89 | 0.44 | 1.00 | 0.63 |
| | Pine | 2 | 0.12 | 0.30 | 0.00 | 0.17 | 0.00 | 0.12 |
| | | 5 | 0.25 | 0.28 | 0.00 | 0.36 | 0.00 | 0.18 |
| | Seaforth | 2 | 0.23 | 0.26 | 0.00 | 0.12 | 0.50 | 0.22 |
| | | 5 | 0.23 | 0.41 | 0.00 | 0.37 | 1.00 | 0.40 |
| Poor | | | 0.26 | 0.40 | 0.38 | 0.27 | 0.24 | 0.31 |
| Fitzroy | Barren | 2 | 0.76 | 0.80 | 1.00 | 0.46 | 0.00 | 0.60 |
| | | 5 | 1.00 | 0.12 | 0.00 | 0.85 | 0.50 | 0.49 |
| | North Keppel | 2 | 0.49 | 0.02 | 0.00 | 0.09 | 1.00 | 0.32 |
| | | 5 | 0.35 | 0.04 | 0.00 | 0.25 | 0.50 | 0.23 |
| | Middle | 2 | 0.30 | 0.07 | 0.00 | 0.50 | 0.00 | 0.17 |
| | | 5 | 0.30 | 0.23 | 0.00 | 0.25 | 0.00 | 0.16 |
| | Keppels South | 2 | 0.64 | 0.16 | 0.00 | 0.51 | 0.00 | 0.26 |
| | | 5 | 0.46 | 0.10 | 0.00 | 0.00 | 0.50 | 0.21 |
| | Pelican | 2 | 0.12 | 0.07 | 0.00 | 0.00 | 1.00 | 0.12 |
| | | 5 | 0.30 | 0.24 | 0.00 | 0.00 | 0.50 | 0.33 |
| Poor | | | 0.42 | 0.21 | 0.08 | 0.30 | 0.38 | 0.28 |

Table A 8 Environmental covariates for coral locations. For chlorophyll a (Chl *a*), non-algal particulate (NAP) and Photosynthetically available radiation (PAR) a square of nine 1km square pixels was selected adjacent to each reef location. For Chl *a,* mean concentrations over the 2016-2020 wet seasons were estimated based proportion of time waters were classified into one of six colour classes (Petus et al. 2016) and the mean concentration of Chl *a* from MMP water samples taken within each colour class (Waterhouse et al. 2021). PAR values are mean values 2015-2019 form the same pixels estimated at 8m depth (Magno-Canto et al. 2019). Mean values for NAP over the period 2005-2018 were downloaded from the Bureau of Meteorology, Marine Water Quality Dashboard.

| (sub-)region | Reef | Wet season Chl *a* ($\mu gL^{-1}$) | NAP ($mgL^{-1}$) | PAR (mol photons $m^{-2}$, $d^{-1}$) |
|---|---|---|---|---|
| Barron Daintree | Low Isles | 0.50 | 0.95 | 13.47 |
| | Snapper North | 0.70 | 1.01 | 10.51 |
| | Snapper South | 0.70 | 1.01 | 10.75 |
| Johnstone Russell-Mulgrave | Fitzroy East | 0.37 | 0.69 | 14.03 |
| | Green | 0.37 | 0.59 | 13.60 |
| | Franklands East | 0.38 | 0.75 | 14.14 |
| | Franklands West | 0.49 | 0.78 | 13.50 |
| | Fitzroy West | 0.54 | 0.79 | 12.44 |
| | High East | 0.55 | 0.80 | 12.69 |
| | High West | 0.69 | 0.92 | 12.10 |
| Herbert Tully | Barnards | 0.67 | 0.77 | 11.70 |
| | Dunk North | 0.79 | 0.97 | 11.34 |
| | Dunk South | 0.80 | 1.00 | 10.93 |
| | Bedarra | 0.83 | 1.06 | 10.34 |
| Burdekin | Palms East | 0.44 | 0.71 | 14.08 |
| | Havannah North | 0.56 | 0.83 | 13.07 |
| | Palms West | 0.61 | 0.79 | 12.98 |
| | Havannah | 0.62 | 0.83 | 12.51 |
| | Pandora | 0.70 | 0.87 | 11.35 |
| | Pandora North | 0.71 | 0.88 | 11.42 |
| | Magnetic | 0.82 | 1.81 | 6.53 |
| | Lady Elliot | 0.86 | 1.07 | 10.29 |
| Mackay-Whitsunday | Hayman | 0.46 | 0.81 | 13.47 |
| | Langford | 0.51 | 0.93 | 13.39 |
| | Border | 0.56 | 1.04 | 12.59 |
| | Hook | 0.60 | 1.12 | 12.18 |
| | Double Cone | 0.64 | 1.26 | 11.13 |
| | Seaforth | 0.67 | 1.22 | 11.04 |
| | Dent | 0.71 | 1.52 | 9.50 |
| | Daydream | 0.72 | 1.53 | 9.84 |
| | Pine | 0.74 | 1.86 | 9.13 |
| | Shute Harbour | 0.74 | 1.61 | 9.40 |
| Fitzroy | Barren | 0.44 | 0.47 | 14.11 |
| | Keppels South | 0.71 | 0.73 | 12.80 |
| | North Keppel | 0.71 | 0.78 | 12.40 |
| | Middle | 0.72 | 0.66 | 12.27 |
| | Peak | 0.90 | 2.23 | 6.01 |
| | Pelican | 0.93 | 1.41 | 9.85 |

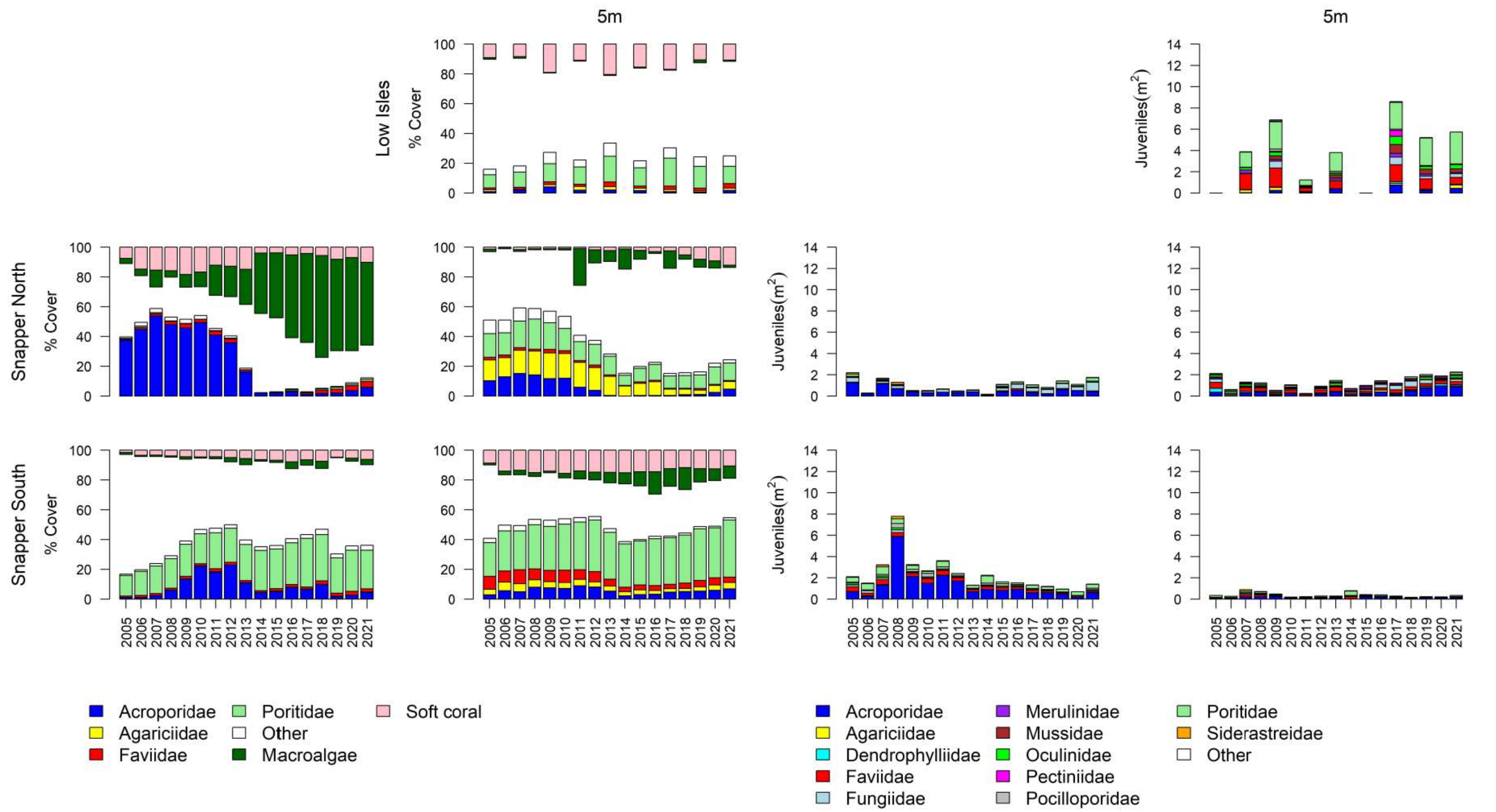


Figure A 1 Barron Daintree sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends relevant groupings for cover and juvenile density estimates are located beneath the relevant plots.

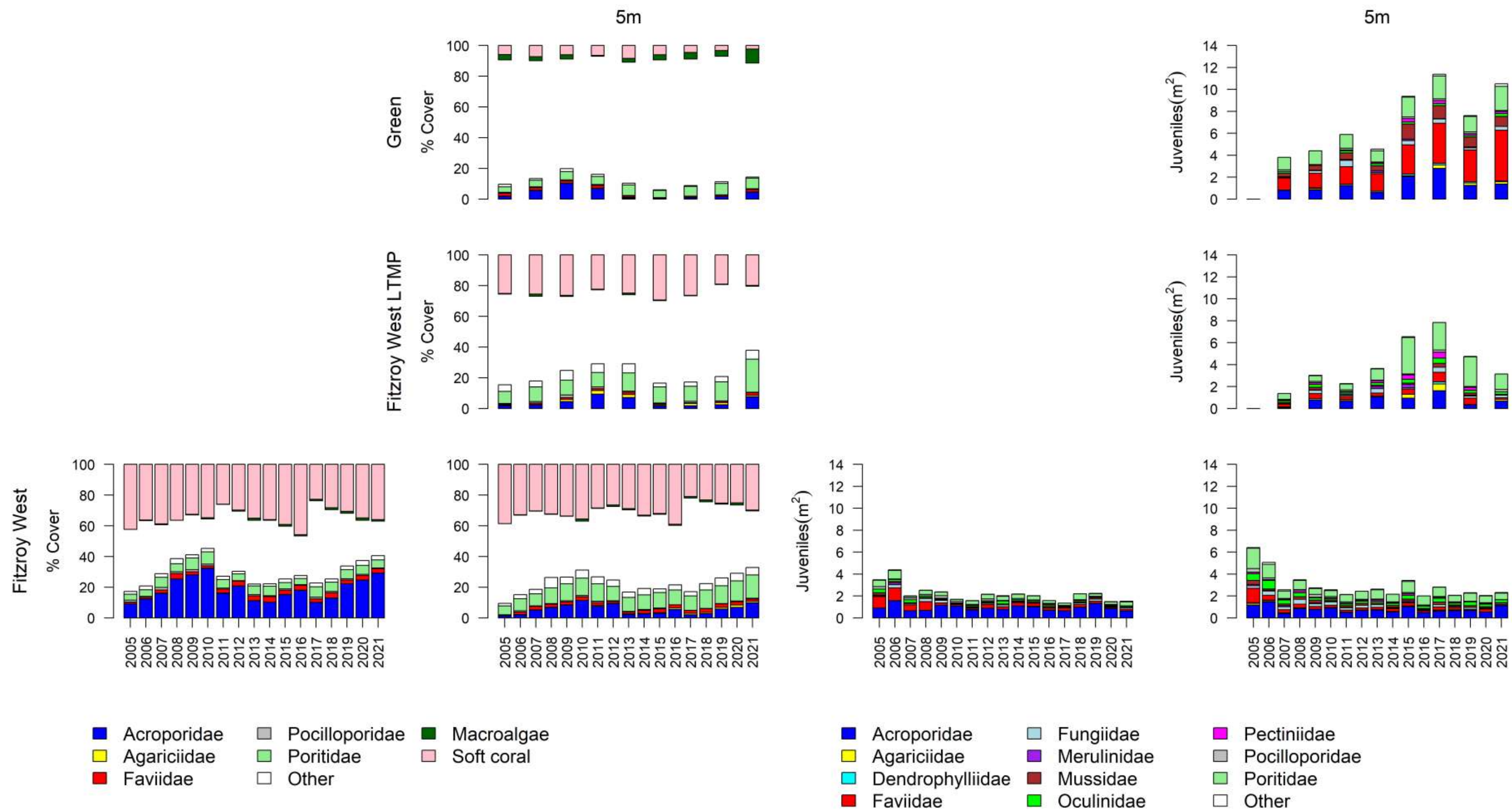


Figure A 2 Johnstone Russell-Mulgrave sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends relevant groupings for cover and juvenile density estimates are located beneath the relevant plots.

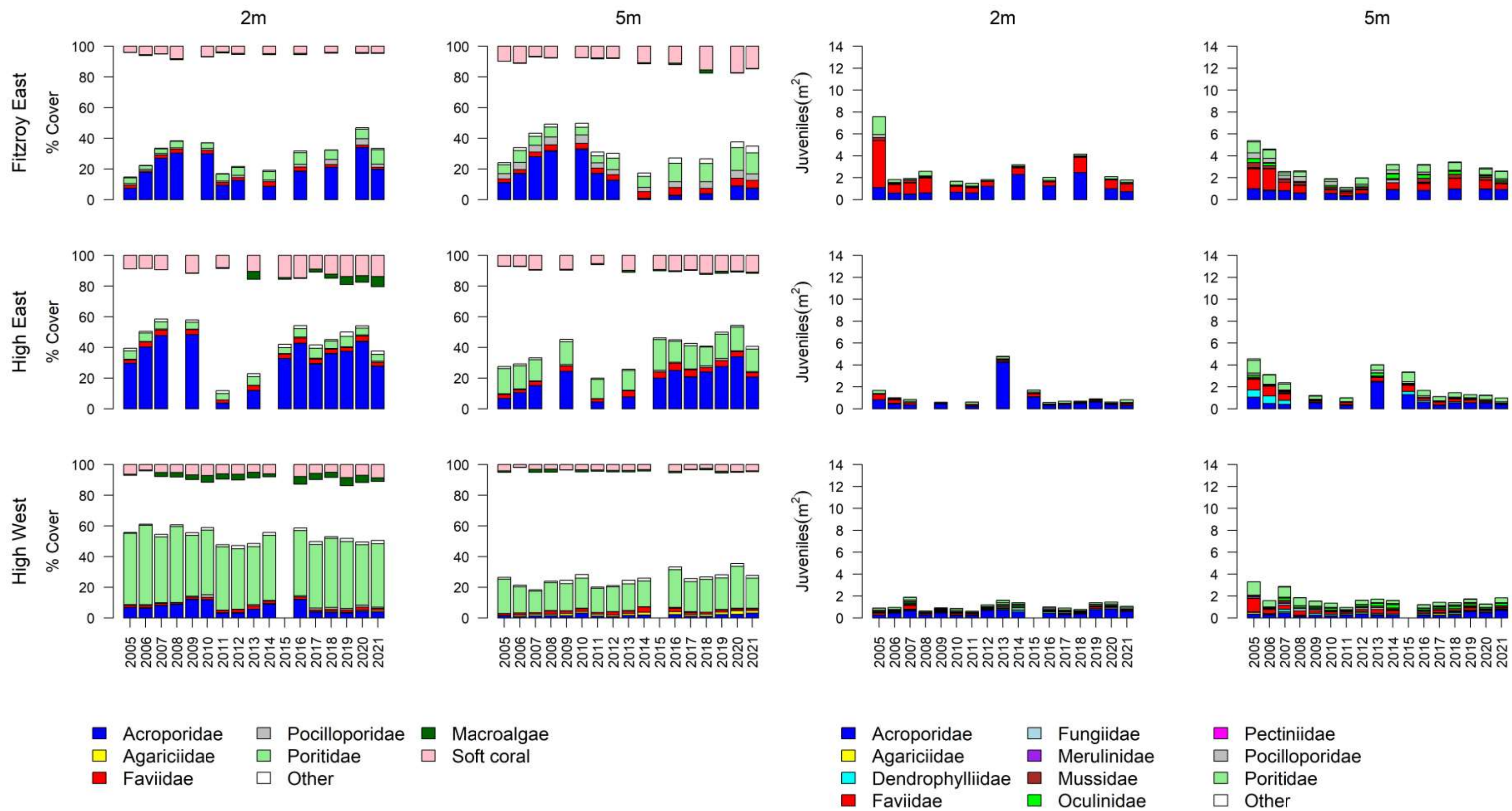


Figure A 2 continued

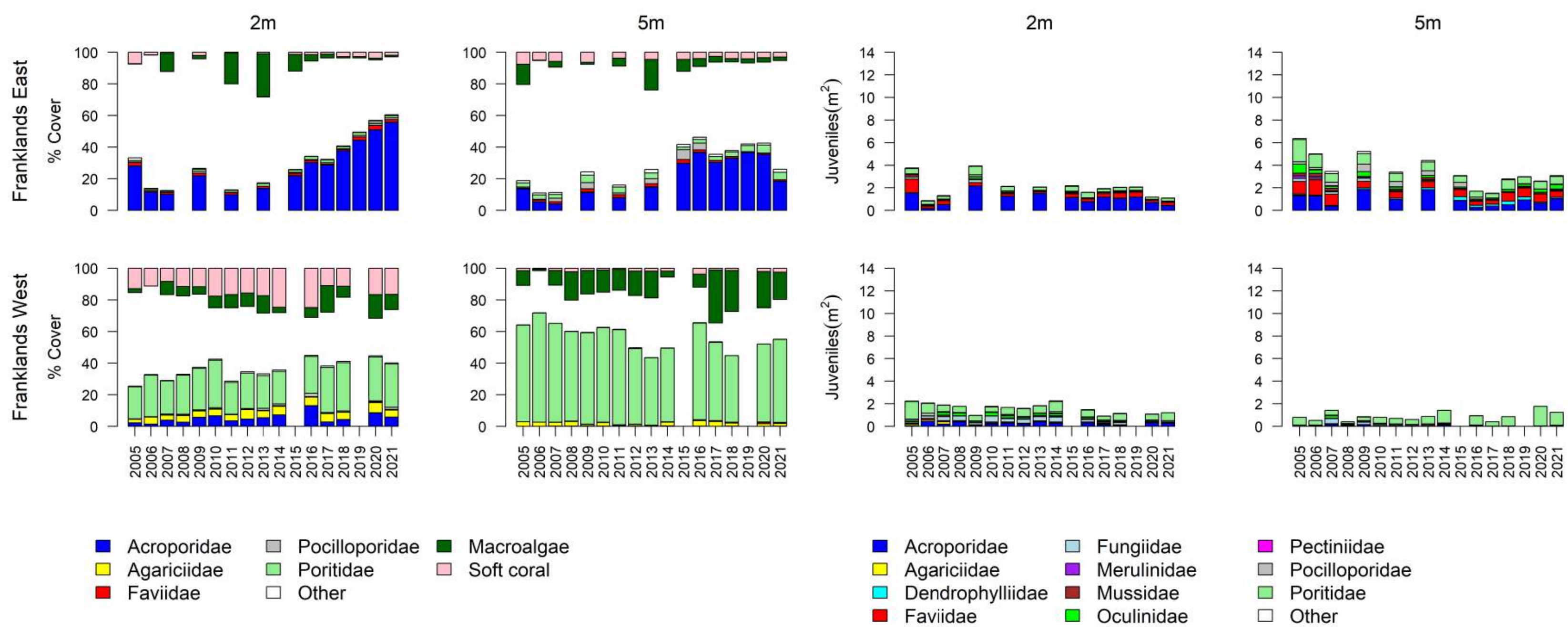


Figure A 2 continued

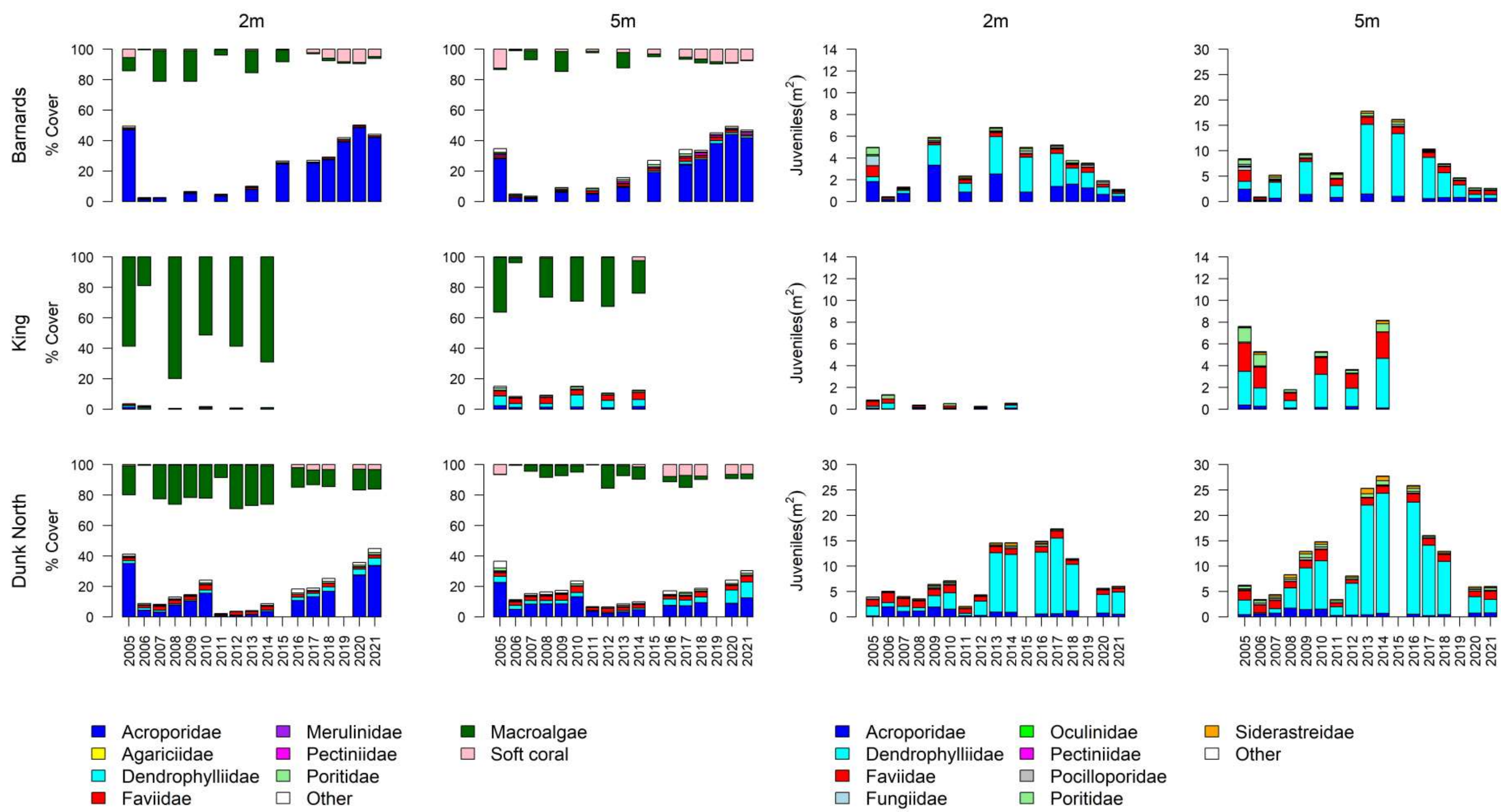


Figure A 3 Herbert Tully sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends with relevant groupings for cover and juvenile density estimates are located beneath the respective plots.

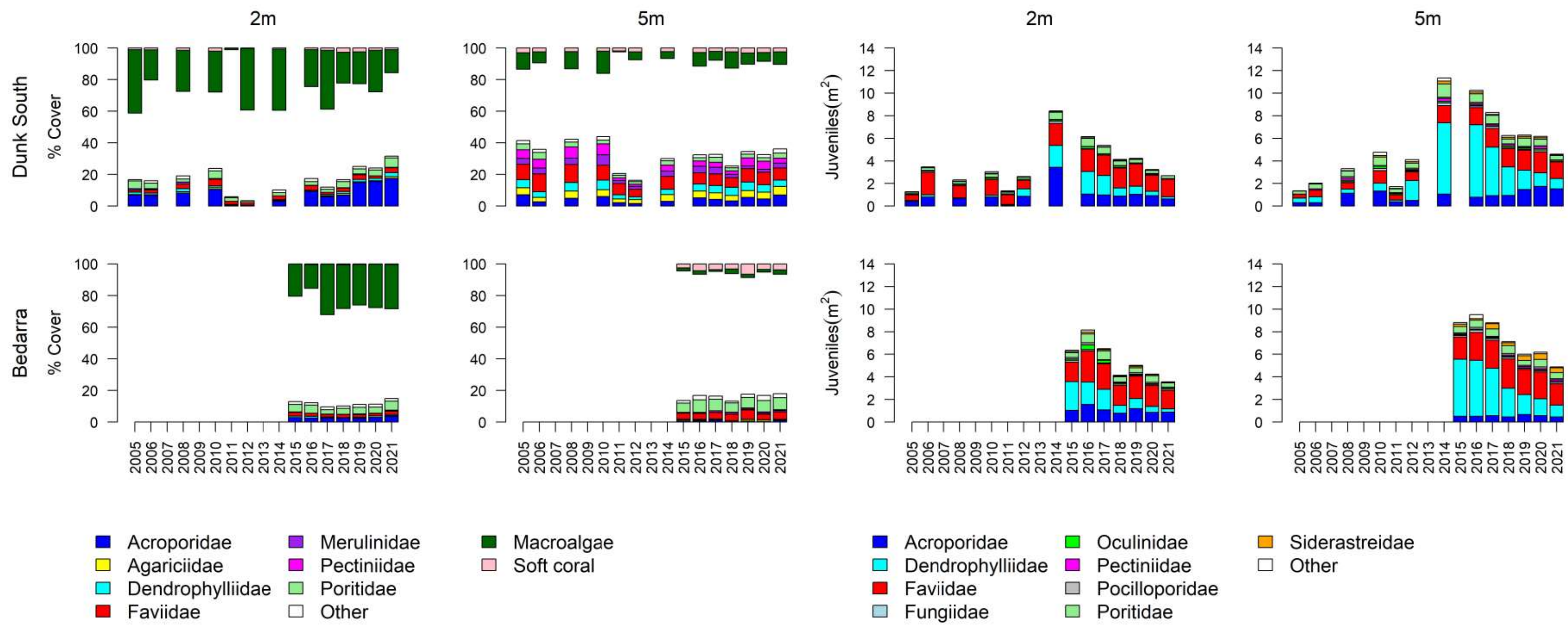


Figure A 3 continued

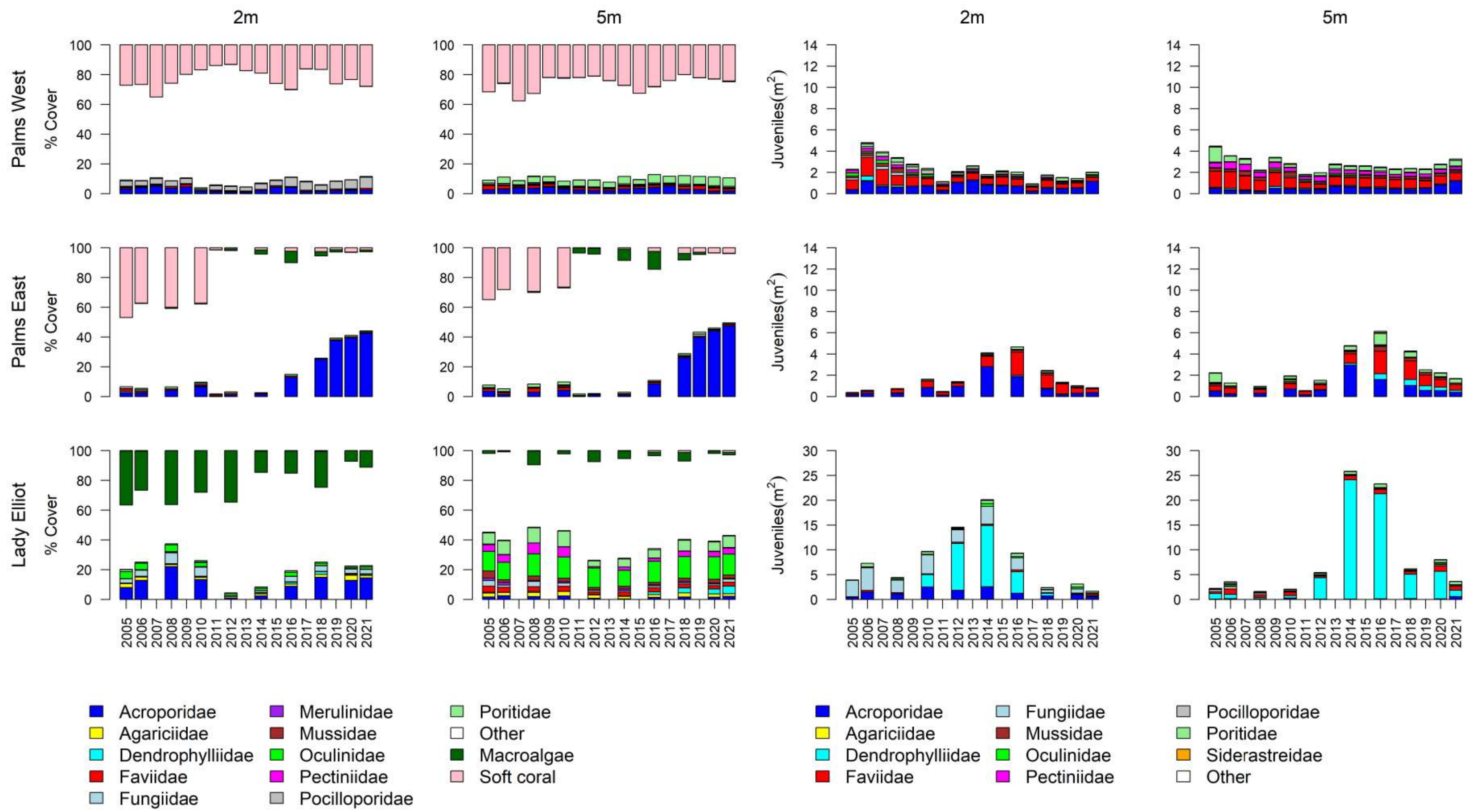


Figure A 4 Burdekin region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends with relevant groupings for cover and juvenile density estimates are located beneath the respective plots.

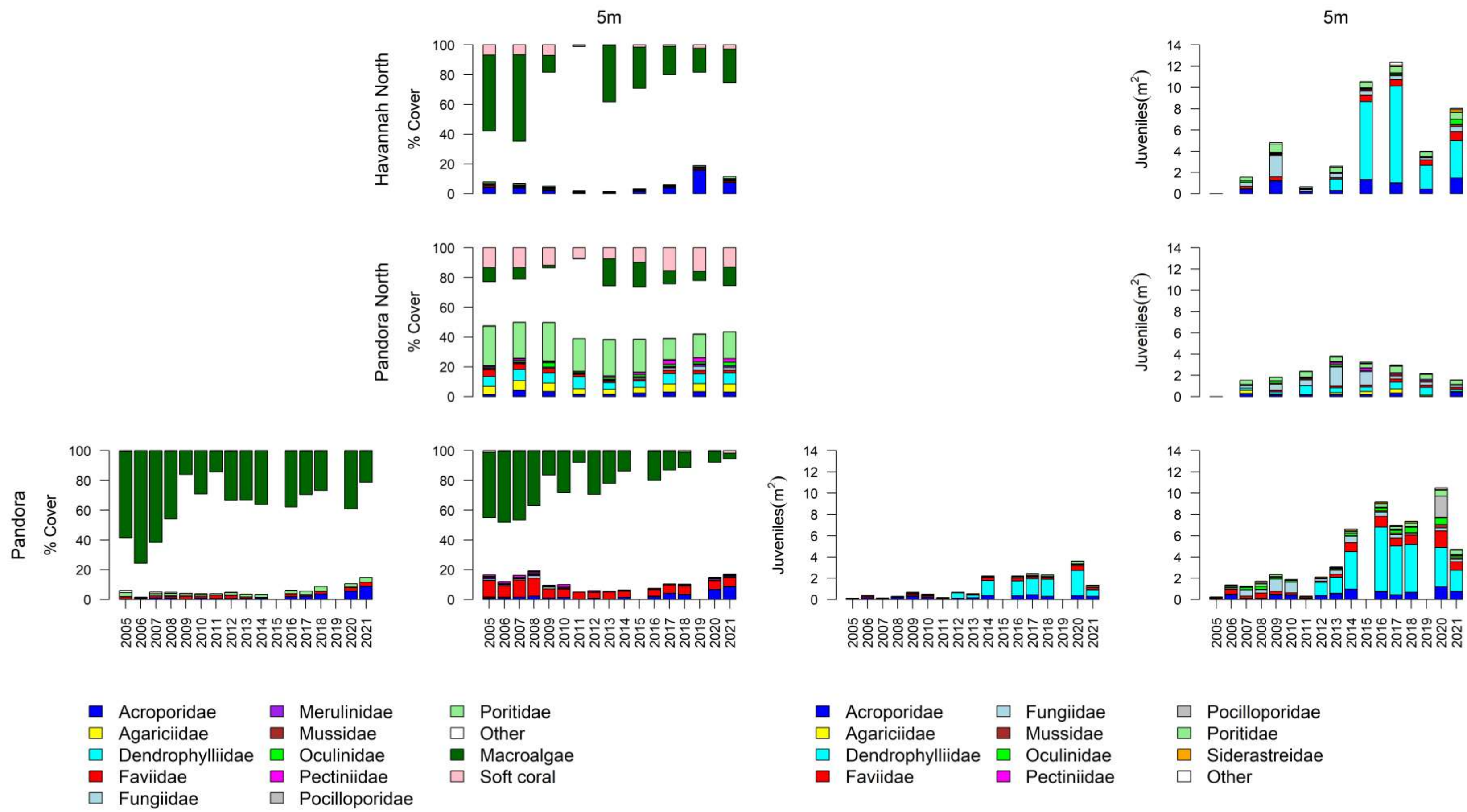


Figure A 4 continued

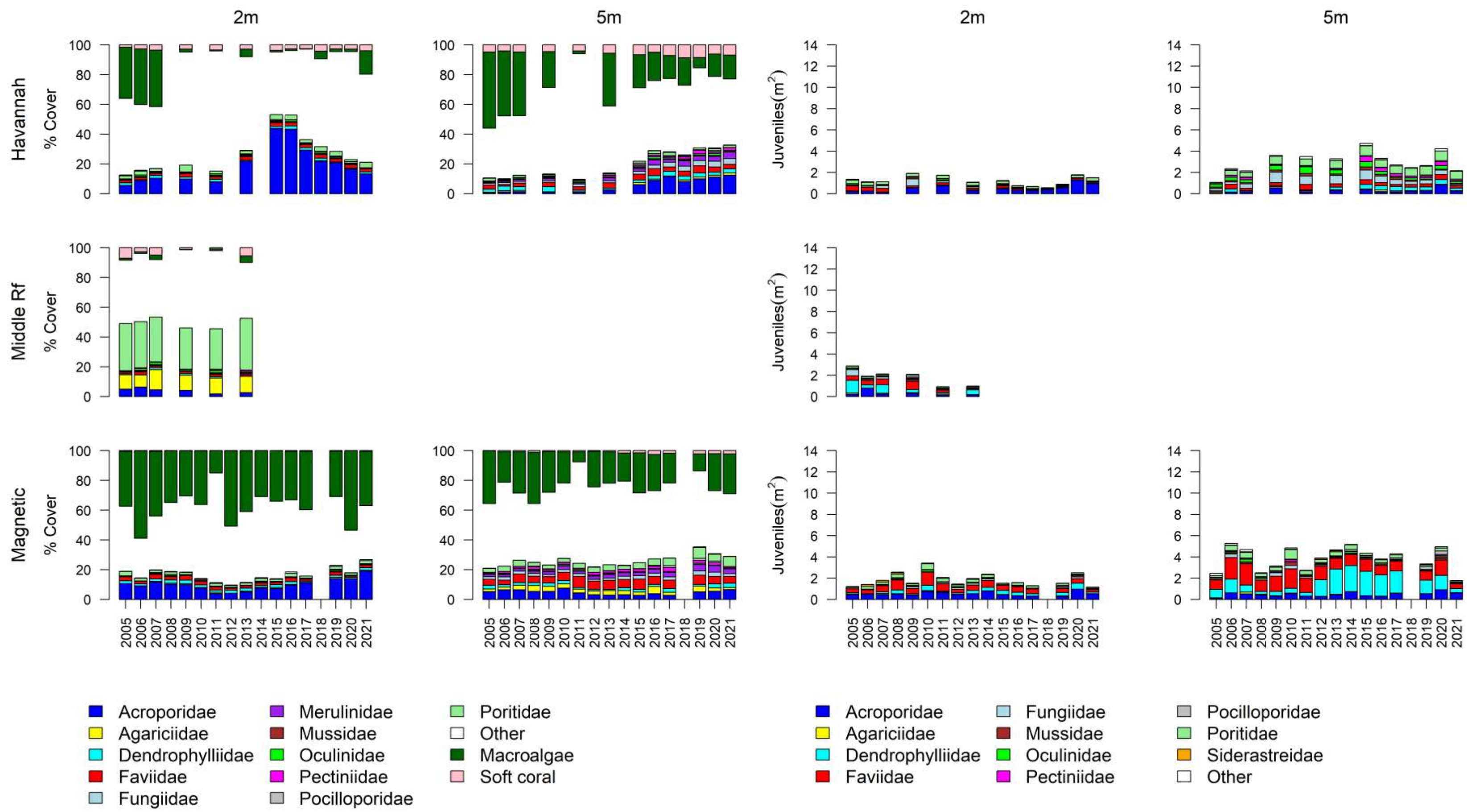


Figure A 4 continued

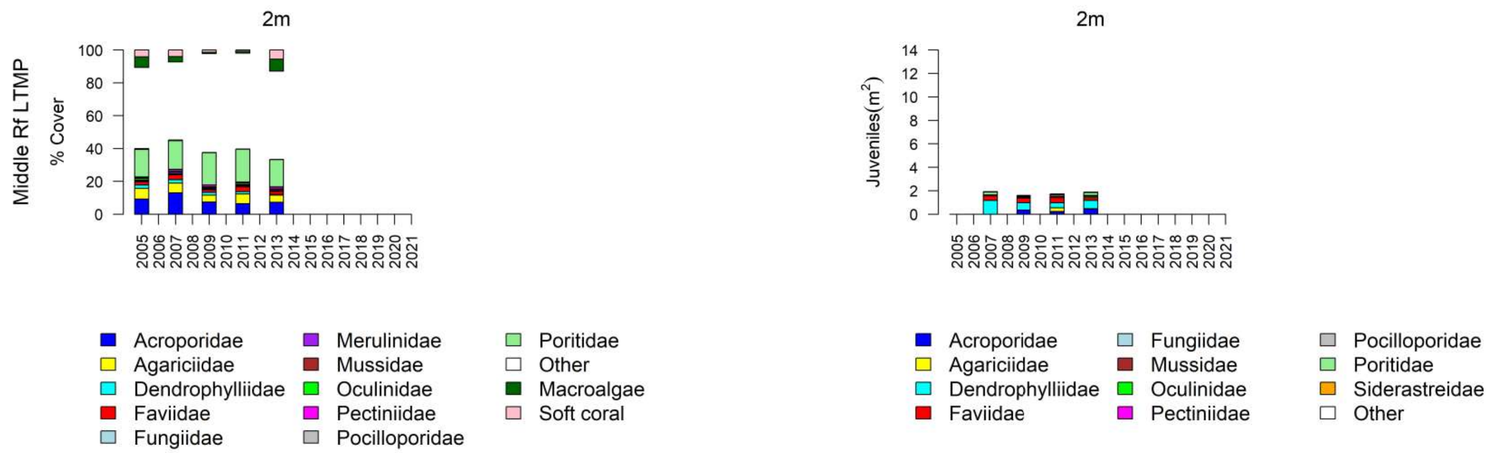


Figure A 4 continued

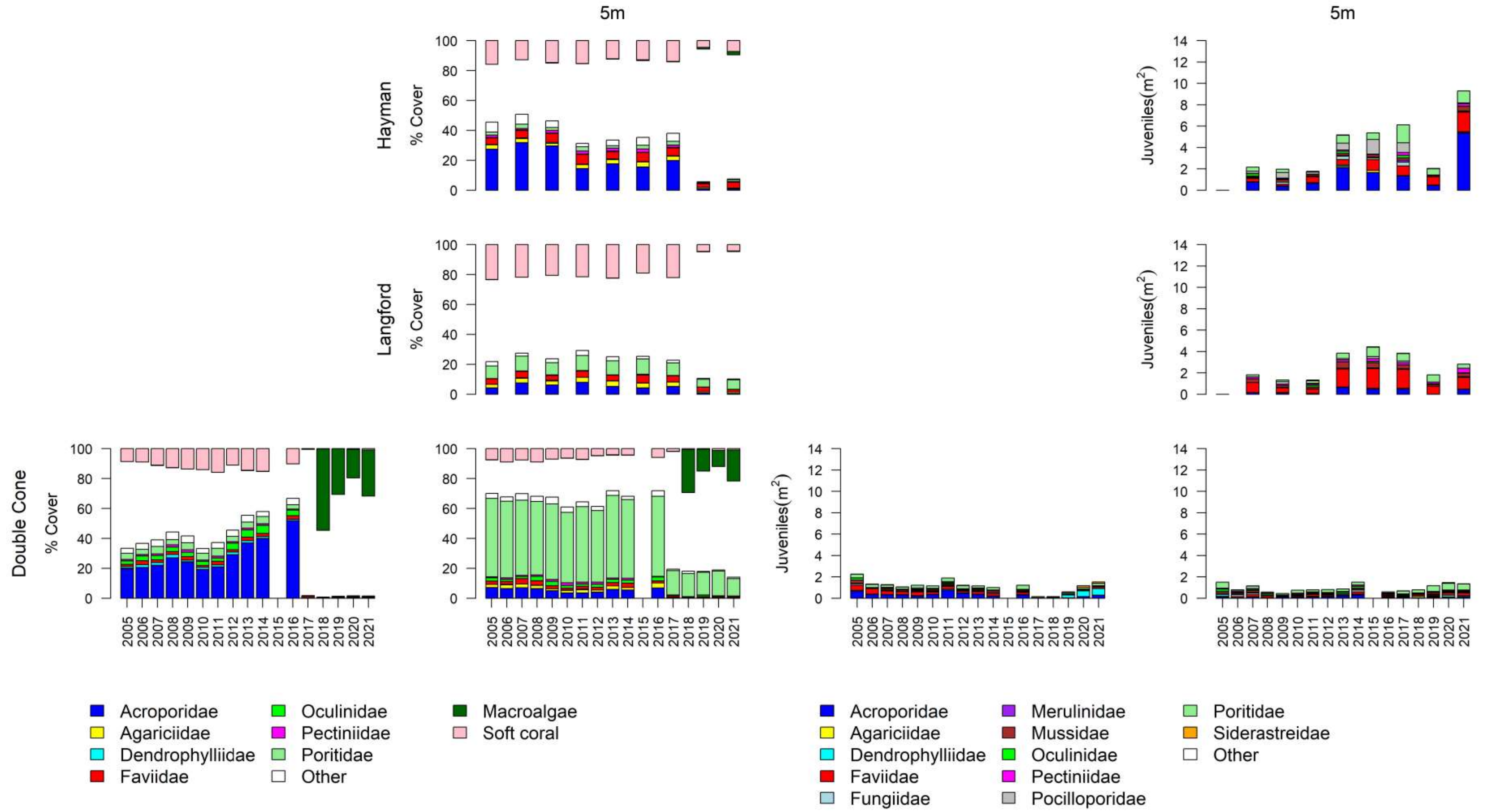


Figure A 5 Mackay-Whitsunday region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends with relevant groupings for cover and juvenile density estimates are located beneath the respective plots.

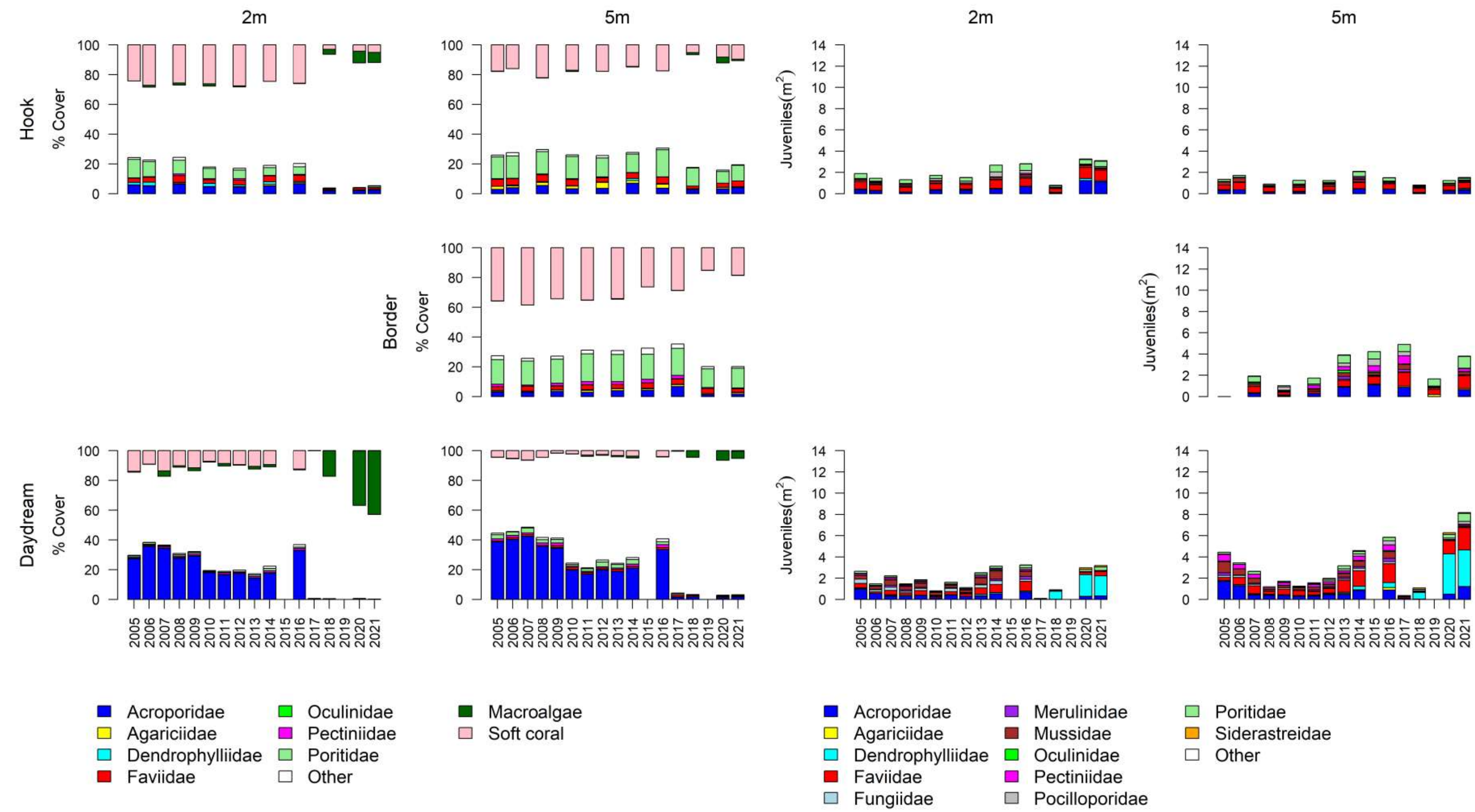


Figure A 5 continued

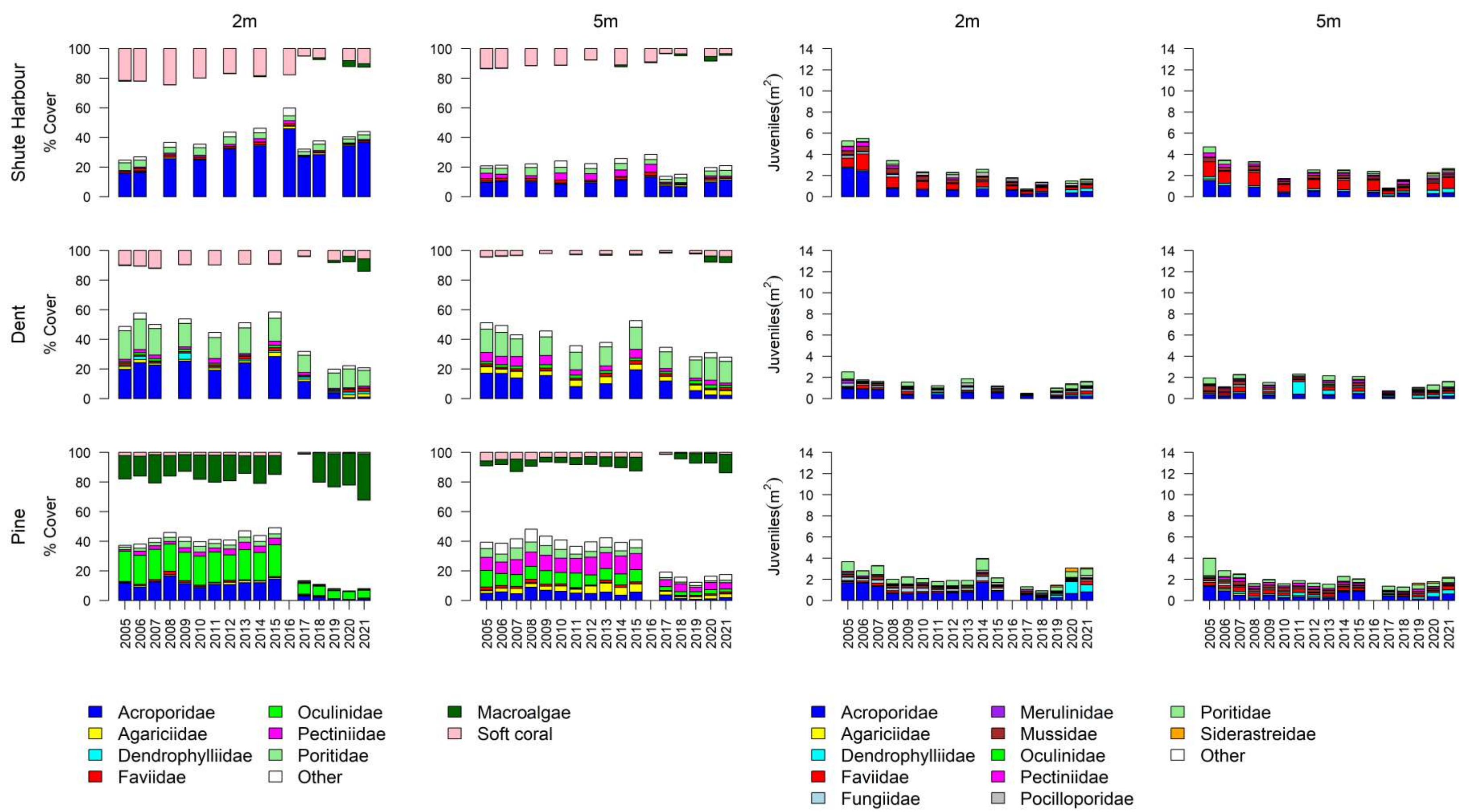


Figure A 5 continued

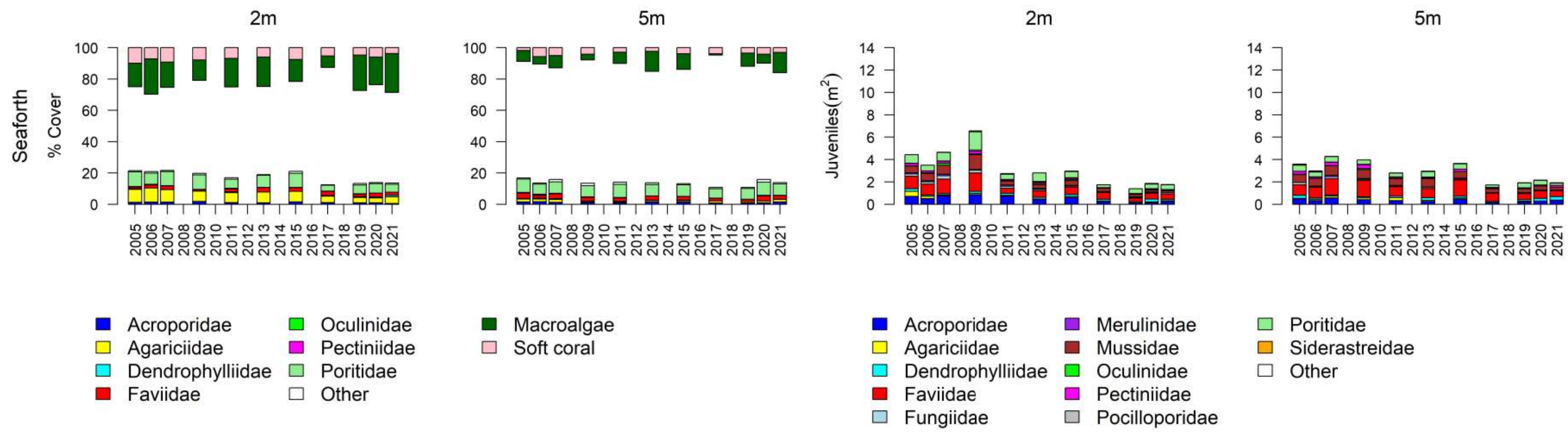


Figure A 5 continued

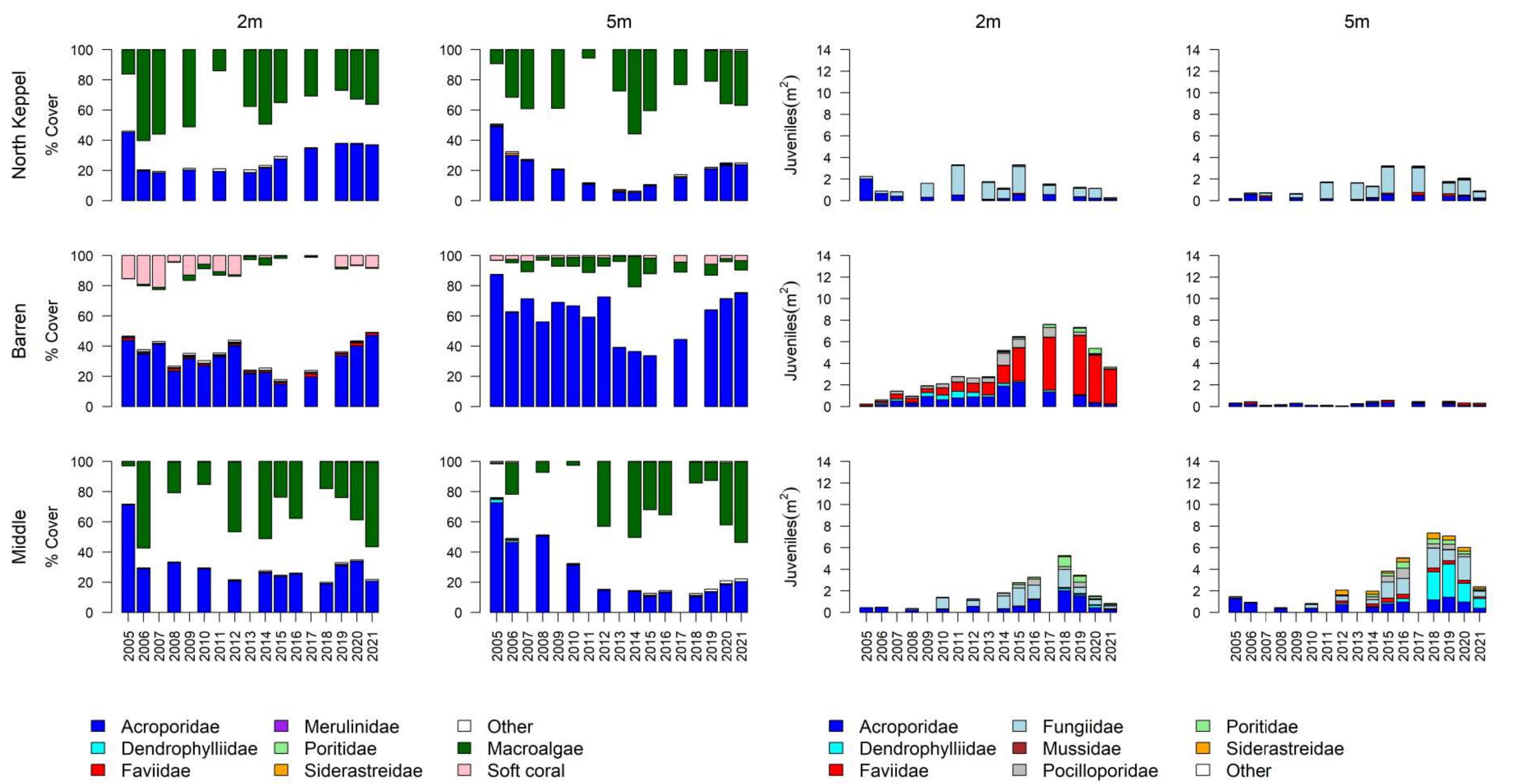


Figure A 6 Fitzroy region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends with relevant groupings for cover and juvenile density estimates are located beneath the respective plots.

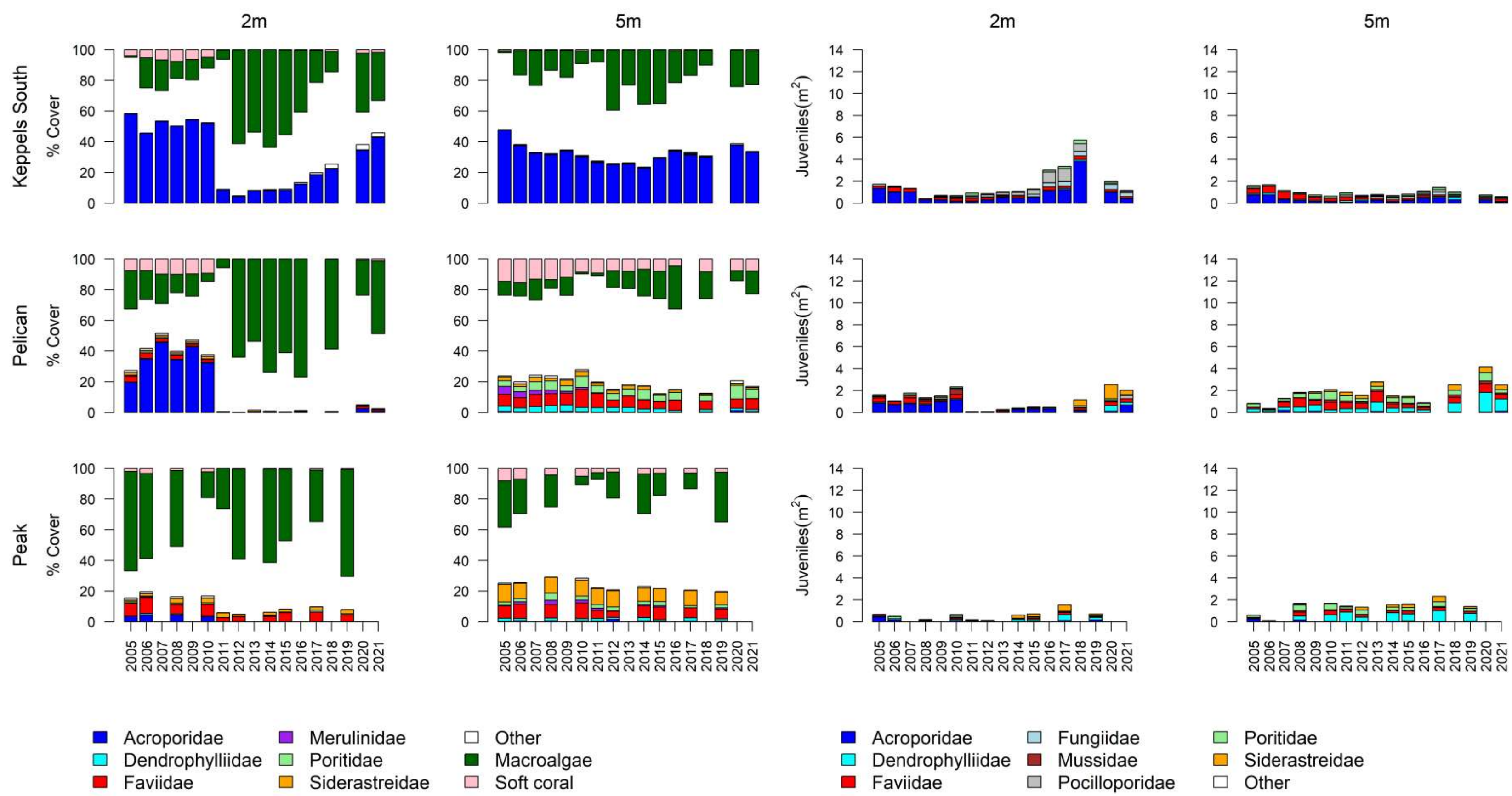


Figure A 6 continued

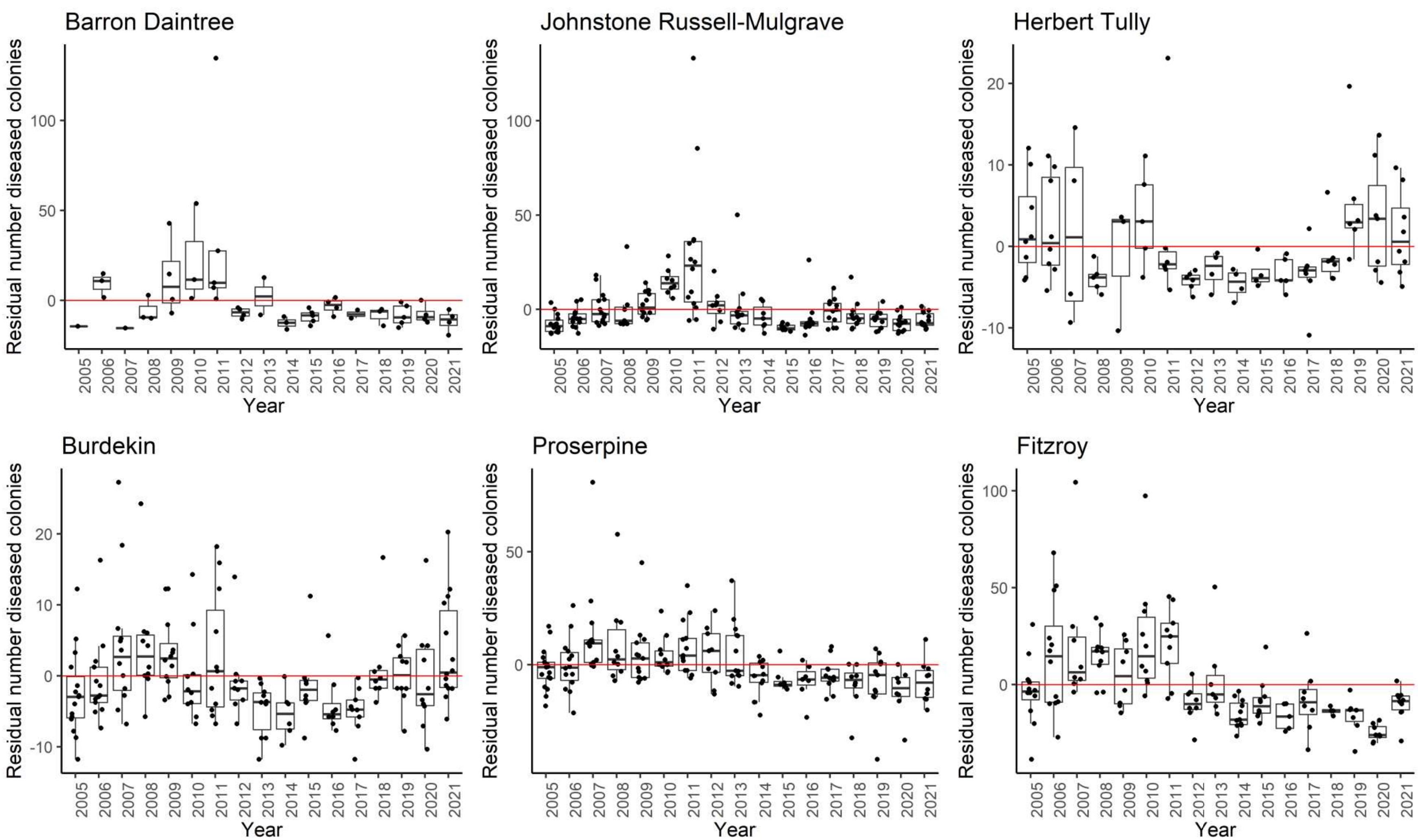


Figure A 7 Coral disease by year in each region. Boxplots include the number of coral colonies suffering ongoing mortality attributed to either disease, sedimentation or 'unknown causes' for each reef, depth and year. Data are standardised to the reef and depth mean across years.

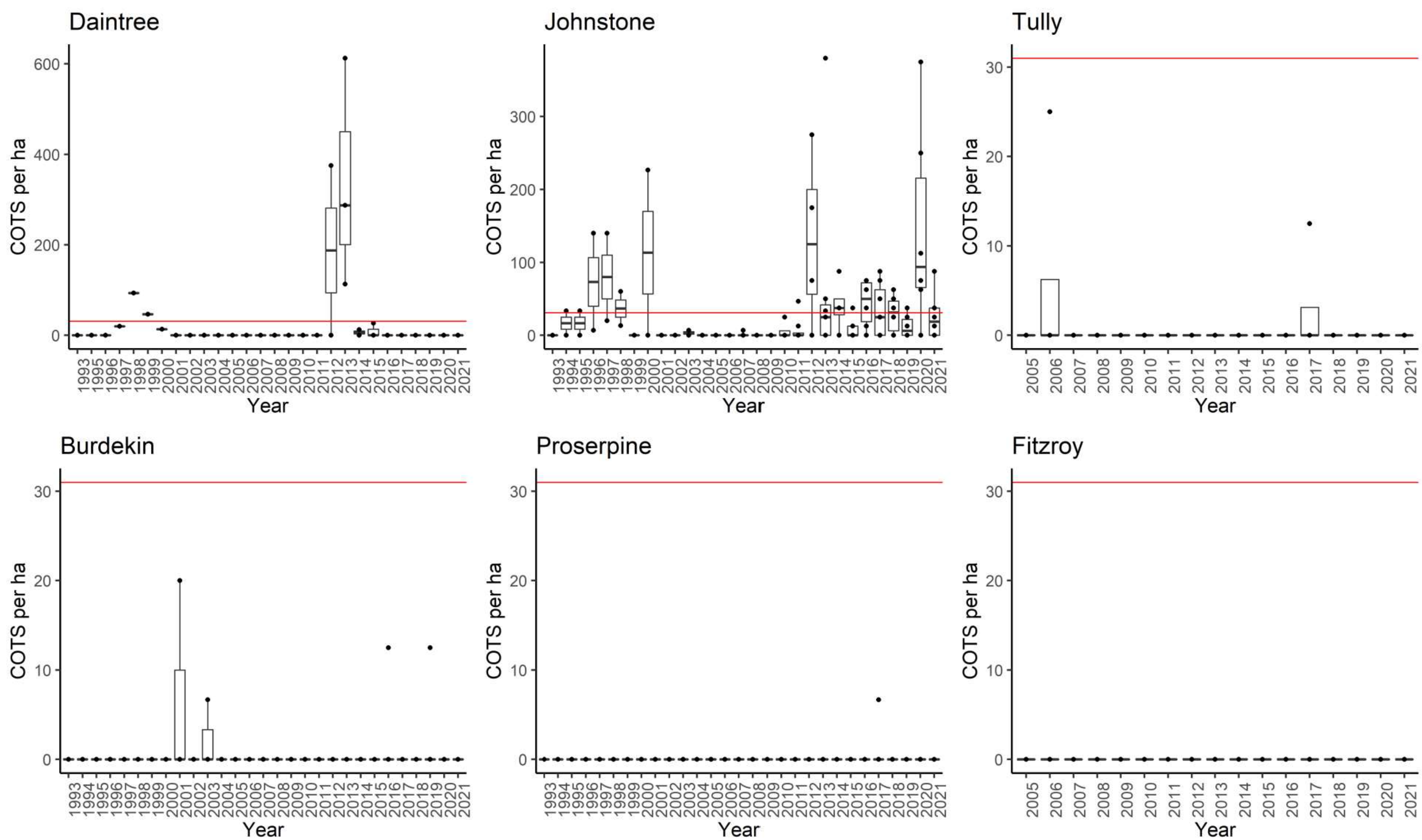


Figure A 8 Crown -of-thorn-starfish mean density (individuals/ha) by year in each region. Red line indicates outbreak densities of 31 individuals per hectare.

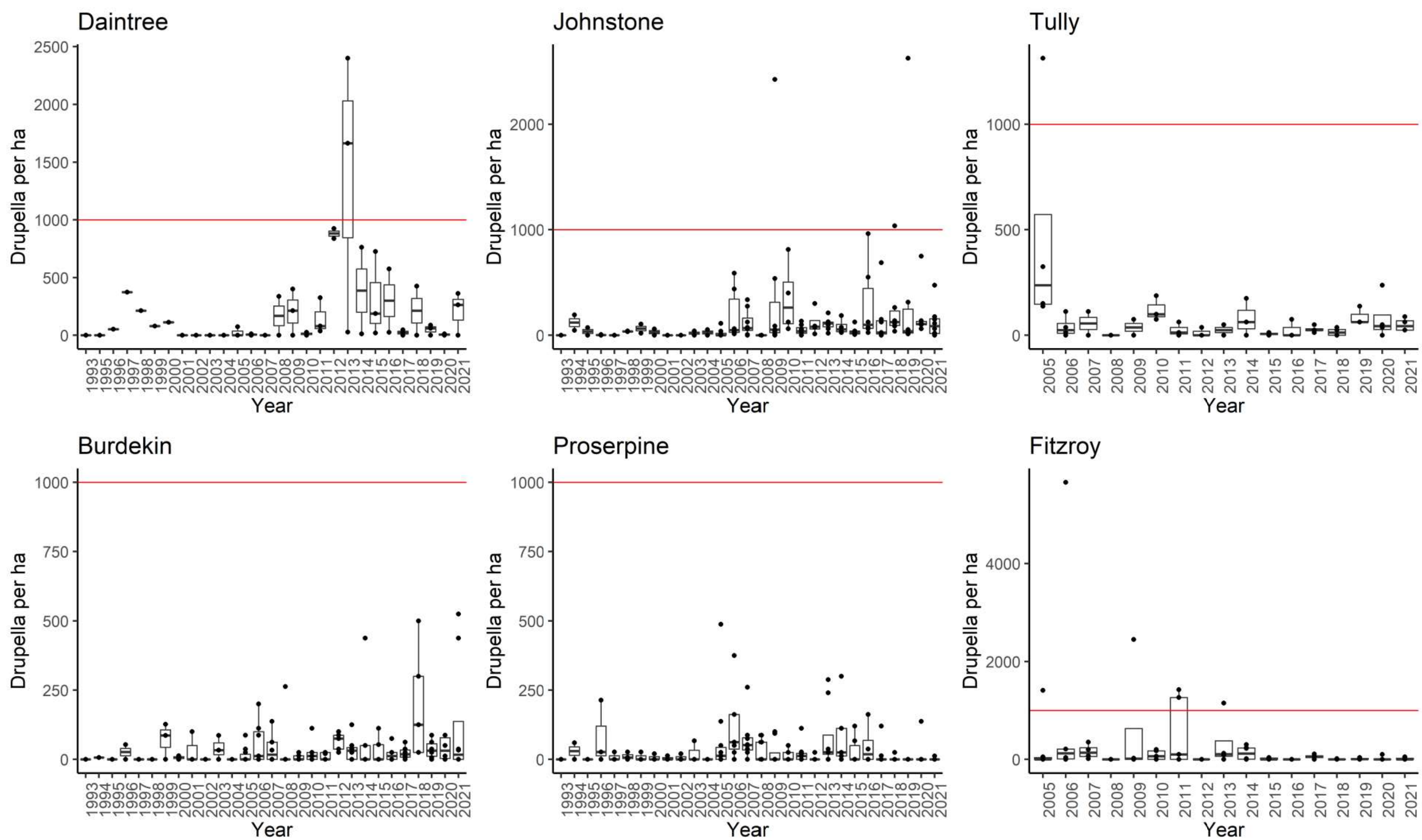


Figure A 9 Mean density of *Drupella* by year in each (sub-)region. Red line indicates densities of *Drupella* which have detrimental impact on coral communities.

Table A 9 Percent cover of hard coral genera 2021. Genera for which cover did not exceeded 1% on at least one reef or were unidentified to genus level are grouped as “other”.

| (sub-)region | Reef | Depth | *Acropora* | *Alveopora* | *Caulastrea* | *Cyphastrea* | *Diploastrea* | *Echinophyllia* | *Echinopora* | *Favia* | *Favites* | *Fungia* | *Galaxea* | *Goniastrea* | *Goniopora* | *Isopora* | *Lobophyllia* | *Merulina* | *Montipora* | *Mycedium* | *Oxypora* | *Pachyseris* | *Pavona* | *Pectinia* | *Platygyra* | *Plesiastrea* | *Pocillopora* | *Podobacia* | *Porites* | *Symphyllia* | *Turbinaria* | *Other* |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Barron Daintree | Low Isles | 5 | 1.39 | 0.03 | 0 | 0.07 | 0.47 | 0 | 1.44 | 0.23 | 0.07 | 0.37 | 3.24 | 0.17 | 0.5 | 0 | 1.1 | 0.3 | 0.23 | 0.1 | 0.13 | 1.3 | 0.23 | 0.13 | 0.27 | 0 | 0.03 | 0 | 11.15 | 0.13 | 0 | 1.83 |
| | Snapper North | 2 | 4.71 | 0 | 0.13 | 0 | 0 | 0 | 3.88 | 0 | 0 | 0.33 | 0.21 | 0 | 0.04 | 0.08 | 0.17 | 0.04 | 0.88 | 0 | 0 | 0 | 0.17 | 0 | 0 | 0 | 0.08 | 0 | 1.33 | 0 | 0 | 0.17 |
| | | 5 | 2.31 | 0 | 0.06 | 0 | 0 | 0 | 0 | 0.25 | 0.19 | 0.25 | 0.75 | 0 | 7.25 | 0.19 | 0.06 | 0.25 | 2.13 | 0.13 | 0.06 | 3.81 | 0.81 | 0.13 | 0.13 | 0 | 0 | 0 | 4.44 | 0 | 0 | 1.19 |
| | Snapper South | 2 | 1.96 | 0 | 0 | 0.13 | 0 | 0 | 0.08 | 0.21 | 0.21 | 0.25 | 2.29 | 1.21 | 0.08 | 0 | 0.04 | 0 | 2.58 | 0 | 0 | 0 | 0.21 | 0.04 | 0.04 | 0 | 0.04 | 0 | 25.88 | 0.17 | 0.04 | 0.63 |
| | | 5 | 5.75 | 0 | 3.25 | 0 | 0 | 0 | 0.19 | 0.13 | 0.06 | 0.06 | 0.06 | 0 | 3.44 | 0 | 0.13 | 0 | 1.19 | 0 | 0 | 3.38 | 0.88 | 0.06 | 0 | 0 | 0 | 0 | 34.94 | 0 | 0.25 | 0.88 |
| Johnstone Russell-Mulgrave | Green | 5 | 3.75 | 0 | 0 | 0.2 | 0.07 | 0 | 0.03 | 0.43 | 0.1 | 0.03 | 0.17 | 0.13 | 0.13 | 0 | 0.07 | 0 | 0.53 | 0 | 0.03 | 0.17 | 0.07 | 0.03 | 0.07 | 0 | 0.3 | 0 | 6.8 | 0.03 | 0.17 | 1.00 |
| | Fitzroy East | 2 | 14.25 | 0 | 0 | 0.06 | 0 | 0 | 0 | 0.56 | 0.13 | 0 | 0.13 | 0.19 | 0 | 0 | 0.06 | 0 | 5.19 | 0 | 0 | 0 | 0.13 | 0 | 0.19 | 0 | 2 | 0 | 9.38 | 0.38 | 0 | 0.69 |
| | | 5 | 6 | 0 | 0 | 0.06 | 1.13 | 0.13 | 1.5 | 0.75 | 0.19 | 0.06 | 1.25 | 0.31 | 0.31 | 0.06 | 0.88 | 0.06 | 0.88 | 0 | 0.13 | 0.25 | 0.06 | 0.19 | 0.38 | 0 | 4.31 | 0 | 13.38 | 0.25 | 0.06 | 2.31 |
| | Fitzroy West | 2 | 23.06 | 0 | 0 | 0 | 0.81 | 0 | 1.69 | 0.06 | 0.06 | 0.19 | 0.88 | 0.13 | 0.31 | 0 | 0.88 | 0.06 | 5.94 | 0 | 0.13 | 0 | 0 | 0 | 0 | 0 | 0.44 | 0 | 4.88 | 0.38 | 0 | 0.69 |
| | | 5 | 6.63 | 0 | 0.25 | 0 | 0.25 | 0.06 | 0.5 | 0.25 | 0.06 | 0.13 | 0.75 | 0 | 1.31 | 0 | 2.06 | 0.31 | 3.06 | 0.13 | 0.06 | 0.94 | 0 | 0.06 | 0.19 | 0 | 0.5 | 0 | 13.88 | 0 | 0 | 1.44 |
| | Fitzroy West LTMP | 5 | 0.42 | 0 | 0.31 | 0 | 0.09 | 1.01 | 0.25 | 0.17 | 0.08 | 0.23 | 1.46 | 0.08 | 0.34 | 0 | 0.43 | 0.66 | 2.1 | 0 | 0.34 | 3.18 | 0 | 0.09 | 0.15 | 0 | 0.17 | 0.39 | 15.14 | 0 | 0 | 1.06 |
| | Franklands East | 2 | 33.5 | 0 | 0 | 0.06 | 0 | 0 | 1.25 | 0.13 | 0.13 | 0 | 0.19 | 0 | 0.19 | 0.31 | 0 | 0.13 | 21.75 | 0 | 0 | 0 | 0.13 | 0 | 0.06 | 0 | 1.06 | 0 | 1.19 | 0 | 0 | 0.31 |
| | | 5 | 15.38 | 0 | 0 | 0.13 | 0 | 0 | 0.31 | 0.13 | 0.06 | 0.06 | 0.38 | 0.13 | 0 | 0 | 0.31 | 0.44 | 2.94 | 0 | 0 | 0 | 0 | 0 | 0.19 | 0 | 0.06 | 0 | 4.81 | 0 | 0 | 0.63 |
| | Franklands West | 2 | 5.5 | 0 | 0 | 0 | 0 | 0 | 0.31 | 0 | 0 | 0.31 | 0.13 | 0 | 0.25 | 0 | 0 | 0 | 0.31 | 0 | 0.06 | 4.44 | 0 | 0 | 0.13 | 0 | 0.5 | 0 | 27.13 | 0 | 0.06 | 1.06 |
| | | 5 | 0.13 | 0 | 0 | 0 | 0 | 0 | 0.31 | 0 | 0 | 0.06 | 0.06 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.75 | 0 | 0 | 0 | 0 | 0 | 0 | 52.5 | 0 | 0 | 0.31 |
| | High East | 2 | 19 | 0 | 0 | 0 | 0 | 0 | 1.06 | 0.06 | 0 | 0 | 0.06 | 0.5 | 0 | 0 | 0.38 | 0 | 8.56 | 0 | 0.06 | 0 | 0.25 | 0 | 0.44 | 0 | 0.81 | 0 | 4.38 | 1.38 | 0.19 | 0.56 |
| | | 5 | 14.06 | 0 | 0 | 0.06 | 0 | 0 | 2.25 | 0.06 | 0 | 0 | 0.44 | 0.25 | 0.56 | 0 | 0.13 | 0.38 | 6.56 | 0 | 0 | 0 | 0 | 0 | 0.13 | 0 | 0.75 | 0.56 | 14 | 0.06 | 0.19 | 0.19 |
| | High West | 2 | 2.88 | 0 | 0 | 0 | 0 | 0 | 0.19 | 0.25 | 0.44 | 0.19 | 0.31 | 0 | 1.94 | 0 | 0.06 | 0.56 | 1.06 | 0 | 0 | 0.44 | 0.5 | 0.13 | 0.25 | 0 | 0.81 | 0 | 39.45 | 0.25 | 0 | 0.75 |
| | | 5 | 2.19 | 0 | 0 | 0 | 0 | 0 | 0 | 0.5 | 0.25 | 0.25 | 0.75 | 0.06 | 5 | 0 | 0.13 | 0.06 | 0.5 | 0 | 0.06 | 0.5 | 1.13 | 0.06 | 0.25 | 0 | 0.44 | 0 | 14.5 | 0 | 0 | 1.13 |

| (sub-)region | Reef | Depth | *Acropora* | *Alveopora* | *Caulastrea* | *Cyphastrea* | *Diploastrea* | *Echinophyllia* | *Echinopora* | *Favia* | *Favites* | *Fungia* | *Galaxea* | *Goniastrea* | *Goniopora* | *Isopora* | *Lobophyllia* | *Merulina* | *Montipora* | *Mycedium* | *Oxypora* | *Pachyseris* | *Pavona* | *Pectinia* | *Platygyra* | *Plesiastrea* | *Pocillopora* | *Podobacia* | *Porites* | *Symphyllia* | *Turbinaria* | *Other* |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Herbert Tully | Barnards | 2 | 27.31 | 0 | 0 | 0.13 | 0 | 0 | 0.19 | 0.31 | 0 | 0.06 | 0.13 | 0 | 0 | 0 | 0 | 0 | 14.63 | 0 | 0 | 0 | 0 | 0 | 0 | 0.06 | 0.63 | 0 | 0 | 0 | 0.44 | 0.19 |
| | | 5 | 20.31 | 0 | 0 | 0.19 | 0 | 0.19 | 0.25 | 0.13 | 0.13 | 0 | 0 | 0.06 | 0.25 | 0 | 0 | 0 | 21.25 | 0.31 | 0.75 | 0.25 | 0 | 0.06 | 0.13 | 0 | 0.81 | 0 | 0.5 | 0 | 1.19 | 0.19 |
| | Dunk North | 2 | 28.13 | 0 | 0 | 0.5 | 0 | 0 | 0.63 | 0.13 | 0.13 | 0 | 0.31 | 0.25 | 0.19 | 0 | 0.13 | 0 | 5.44 | 0.06 | 0.06 | 0.19 | 0 | 0 | 0.06 | 0.06 | 1.81 | 0 | 1.19 | 0 | 4.94 | 0.69 |
| | | 5 | 6.75 | 0.19 | 0 | 0.5 | 0 | 0 | 0 | 0.63 | 1.19 | 0 | 0.06 | 0.13 | 0.25 | 0 | 0 | 0 | 5.38 | 0.25 | 0.31 | 0 | 0 | 0.13 | 0.75 | 0.06 | 0.5 | 0 | 0.81 | 0.31 | 10.44 | 1.75 |
| | Dunk South | 2 | 9.31 | 0 | 0 | 1.19 | 0 | 0 | 0.06 | 0.5 | 0.5 | 0.06 | 0.88 | 0.13 | 0.25 | 0 | 0.19 | 0 | 8.25 | 0 | 0 | 0 | 1.06 | 0 | 0 | 0.06 | 0 | 0 | 5.75 | 0 | 2.5 | 0.94 |
| | | 5 | 2.13 | 0 | 0.06 | 0.44 | 0.06 | 0 | 0.25 | 2.5 | 0.63 | 0.13 | 0.25 | 2.25 | 0.44 | 0 | 0.88 | 2.88 | 4.25 | 1.19 | 1.31 | 4.56 | 0.81 | 0.75 | 1 | 0.06 | 0.5 | 0.69 | 2.75 | 0.06 | 4 | 1.38 |
| | Bedarra | 2 | 3 | 0.06 | 0 | 0.69 | 0 | 0 | 0 | 0.38 | 0.31 | 0.31 | 0.38 | 0.13 | 0 | 0 | 0.5 | 0.13 | 0.63 | 0 | 0.06 | 0 | 0.38 | 0.06 | 0.25 | 0.13 | 0.25 | 0 | 5.63 | 0 | 0.75 | 0.88 |
| | | 5 | 0.5 | 0.06 | 0.06 | 0.06 | 0 | 0 | 0 | 3.19 | 0.44 | 0 | 0.06 | 0 | 3.44 | 0 | 1.63 | 0.75 | 0.5 | 0.44 | 0.13 | 0.31 | 0 | 0 | 0.19 | 0.19 | 0 | 0.25 | 4.06 | 0 | 0.56 | 1.06 |
| Burdekin | Palms East | 2 | 41.56 | 0 | 0 | 0.13 | 0 | 0 | 0 | 0.13 | 0.13 | 0 | 0.06 | 0.19 | 0 | 0 | 0 | 0 | 0.88 | 0 | 0 | 0 | 0 | 0 | 0.06 | 0 | 0.19 | 0 | 0.69 | 0 | 0 | 0.06 |
| | | 5 | 46 | 0 | 0 | 0 | 0 | 0 | 0.06 | 0.19 | 0.25 | 0 | 0 | 0.13 | 0 | 0 | 0 | 0 | 1.5 | 0 | 0 | 0 | 0 | 0 | 0.13 | 0 | 0.69 | 0 | 0.38 | 0.06 | 0 | 0.19 |
| | Palms West | 2 | 2 | 0 | 0 | 0 | 0.25 | 0 | 0.44 | 0.19 | 0.13 | 0 | 0 | 0 | 0.31 | 0.13 | 0 | 0 | 0.25 | 0 | 0 | 0 | 0 | 0 | 0.06 | 0.06 | 7.38 | 0 | 0.19 | 0 | 0 | 0.13 |
| | | 5 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0.25 | 0.25 | 0.38 | 0 | 0.25 | 1.69 | 0 | 0.38 | 0 | 0.75 | 0 | 0 | 0.06 | 0 | 0 | 0.19 | 0 | 0.81 | 0 | 4 | 0 | 0 | 0.63 |
| | Havannah North | 5 | 4.76 | 0 | 0 | 0.13 | 0 | 0 | 0.2 | 0.19 | 0.1 | 0.48 | 0.23 | 0 | 0.09 | 0 | 0.07 | 0.23 | 2.52 | 0.06 | 0.1 | 0.13 | 0 | 0 | 0.17 | 0 | 0 | 0 | 1.21 | 0 | 0.16 | 0.59 |
| | Havannah | 2 | 6.94 | 0.31 | 0 | 0 | 0 | 0 | 0.81 | 0.38 | 0.06 | 0.06 | 0.38 | 0 | 0.13 | 2.56 | 0 | 0 | 3.75 | 0 | 0 | 0 | 0 | 0.19 | 0.81 | 0 | 0 | 0 | 3.25 | 0 | 1.31 | 0.13 |
| | | 5 | 6.63 | 0.06 | 0.19 | 0.38 | 0.63 | 0.06 | 0.94 | 0.25 | 0.19 | 3 | 0.75 | 0.13 | 0.56 | 0 | 0.44 | 4 | 5.69 | 0 | 1.13 | 0.94 | 0.5 | 0.56 | 0.25 | 0 | 0.06 | 0.06 | 1 | 0 | 2.88 | 1.44 |
| | Pandora | 2 | 6.13 | 0 | 0 | 0.38 | 0 | 0 | 0 | 0.44 | 0.31 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2.5 | 0 | 0 | 0 | 0.06 | 0 | 0.06 | 1.25 | 0 | 0 | 3.19 | 0 | 0.25 | 0.25 |
| | | 5 | 5.38 | 0 | 0 | 0.5 | 3.38 | 0 | 0 | 0.38 | 0.5 | 0.5 | 0.25 | 0.06 | 0 | 0 | 0.13 | 0.5 | 3 | 0.06 | 0.31 | 0.25 | 0 | 0 | 0.25 | 0.06 | 0.06 | 0 | 0.31 | 0 | 0.19 | 1.00 |
| | Pandora North | 5 | 1.9 | 0 | 0.23 | 0 | 0 | 0.2 | 0.97 | 0.1 | 0.06 | 1.23 | 2.46 | 0.1 | 9.97 | 0.17 | 0.21 | 0.73 | 0.96 | 0.8 | 0.63 | 4.47 | 0.48 | 0.77 | 0.13 | 0 | 0 | 0 | 7.87 | 0 | 7.43 | 1.64 |
| | Lady Elliot | 2 | 8.94 | 0 | 0 | 0.13 | 0 | 0 | 0.06 | 0 | 0.13 | 2.88 | 1.44 | 0 | 0.19 | 0 | 0.25 | 0 | 5.56 | 0 | 0 | 0 | 1.56 | 0 | 0 | 0 | 0 | 0 | 0.63 | 0 | 0.81 | 0.13 |
| | | 5 | 1 | 0.38 | 0 | 0.13 | 0 | 0.13 | 0 | 0.81 | 0.75 | 0.13 | 14.06 | 0.13 | 3.63 | 0 | 2.25 | 0.44 | 1.06 | 1.63 | 1.13 | 1.75 | 0 | 1.19 | 0.06 | 0.19 | 0 | 1.63 | 3.69 | 0 | 5.13 | 1.81 |
| | Magnetic | 2 | 3.94 | 0 | 0 | 0.88 | 0 | 0 | 0.13 | 0.31 | 0.13 | 0 | 0.5 | 0 | 0.56 | 0 | 0 | 0.06 | 15.06 | 0 | 0 | 0.31 | 0.25 | 0 | 0.13 | 0.25 | 0 | 0.06 | 1.5 | 0 | 1.75 | 0.88 |
| | | 5 | 3.5 | 0 | 0 | 0.44 | 0 | 0.13 | 0.13 | 1.69 | 0.63 | 0.25 | 0.5 | 0 | 4.69 | 0 | 0.25 | 2.94 | 2.75 | 0 | 0.31 | 1.31 | 0 | 0.31 | 1.5 | 0 | 0 | 1.38 | 2.13 | 0 | 3 | 1.19 |

| (sub-)region | Reef | Depth | *Acropora* | *Alveopora* | *Caulastrea* | *Cyphastrea* | *Diploastrea* | *Echinophyllia* | *Echinopora* | *Favia* | *Favites* | *Fungia* | *Galaxea* | *Goniastrea* | *Goniopora* | *Isopora* | *Lobophyllia* | *Merulina* | *Montipora* | *Mycedium* | *Oxypora* | *Pachyseris* | *Pavona* | *Pectinia* | *Platygyra* | *Plesiastrea* | *Pocillopora* | *Podobacia* | *Porites* | *Symphyllia* | *Turbinaria* | *Other* |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Mackay-Whitsunday | Hayman | 5 | 0.13 | 0 | 0 | 0.17 | 2.13 | 0.1 | 0.23 | 0.13 | 0.33 | 0 | 0.1 | 0.36 | 0 | 0 | 0.07 | 0.2 | 0.69 | 0.1 | 0.1 | 0.4 | 0 | 0 | 0.23 | 0 | 0.13 | 0 | 1.13 | 0 | 0 | 0.79 |
| | Langford | 5 | 0.1 | 0.13 | 0 | 0.03 | 0.37 | 0.03 | 0.03 | 0.63 | 0.13 | 0 | 0 | 0.17 | 4.5 | 0 | 0.4 | 0 | 0.33 | 0 | 0 | 0.23 | 0.63 | 0.1 | 0.07 | 0 | 0.03 | 0 | 1.63 | 0 | 0.07 | 0.50 |
| | Border | 5 | 0.6 | 0.07 | 0 | 0 | 0.33 | 0.17 | 0.07 | 0.9 | 0.13 | 0 | 0.03 | 0.2 | 9.6 | 0 | 0.77 | 0.07 | 0.8 | 0 | 0.03 | 0.77 | 0.2 | 0.3 | 0.17 | 0 | 0.03 | 0.03 | 3.56 | 0 | 0.27 | 1.03 |
| | Hook | 2 | 0.13 | 0 | 0 | 0.06 | 0 | 0.06 | 0 | 0.5 | 0.19 | 0 | 0 | 0.13 | 0.19 | 0.06 | 0.13 | 0 | 2.25 | 0 | 0.06 | 0.31 | 0.06 | 0 | 0.19 | 0 | 0 | 0 | 0.75 | 0 | 0 | 0.25 |
| | | 5 | 0.31 | 0 | 0 | 0.31 | 0.56 | 0 | 0.19 | 1.19 | 0.81 | 0 | 0 | 0.06 | 1.25 | 0 | 0.19 | 0 | 3.44 | 0 | 0.06 | 0.38 | 0.06 | 0 | 0.19 | 0 | 0 | 0 | 9.19 | 0.06 | 0.19 | 1.00 |
| | Double Cone | 2 | 0 | 0 | 0 | 0.06 | 0 | 0 | 0.25 | 0.13 | 0 | 0 | 0 | 0.06 | 0 | 0 | 0 | 0.13 | 0.19 | 0 | 0 | 0.25 | 0 | 0 | 0 | 0 | 0.13 | 0 | 0.19 | 0 | 0 | 0.06 |
| | | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.56 | 0.38 | 10.94 | 0 | 0.44 | 0.25 | 0.19 | 0 | 0 | 0.13 | 0 | 0 | 0.19 | 0 | 0 | 0.25 | 0.63 | 0 | 0 | 0.06 |
| | Daydream | 2 | 0.13 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.06 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.06 |
| | | 5 | 0.44 | 0 | 0 | 0.19 | 0 | 0 | 0 | 0 | 0.19 | 0 | 0 | 0.19 | 0 | 0 | 0.06 | 0 | 1.06 | 0 | 0.44 | 0.13 | 0 | 0.06 | 0 | 0 | 0.06 | 0 | 0.19 | 0 | 0 | 0.13 |
| | Dent | 2 | 0.94 | 0 | 0 | 0.06 | 0 | 0 | 0.75 | 0.06 | 0 | 0.19 | 0.5 | 0.06 | 4.44 | 0 | 1.31 | 0.19 | 0.06 | 0 | 0.31 | 0.25 | 2 | 0.88 | 0.19 | 0 | 0 | 0 | 6.13 | 0 | 1.56 | 0.94 |
| | | 5 | 1.56 | 0.25 | 0 | 0 | 0 | 0.38 | 0.25 | 0.13 | 0.38 | 0.06 | 1.5 | 0.31 | 12.19 | 0 | 1.13 | 0.5 | 0.13 | 0.06 | 0.94 | 2.06 | 0.31 | 0.19 | 0.56 | 0 | 0.25 | 0.13 | 2.25 | 0 | 0.31 | 2.13 |
| | Shute Harbour | 2 | 32.82 | 0 | 0.38 | 0.06 | 0 | 0 | 0 | 0.19 | 0.19 | 0 | 0.25 | 0.06 | 3.19 | 0.13 | 0.88 | 0.25 | 3.56 | 0 | 0.19 | 0.06 | 0.13 | 0.44 | 0 | 0 | 0.56 | 0 | 0.06 | 0 | 0 | 0.56 |
| | | 5 | 9.06 | 0.06 | 0.06 | 0 | 0 | 0.06 | 0.31 | 0.06 | 0.06 | 0 | 0 | 0.06 | 2.5 | 0.19 | 1 | 0.25 | 2.06 | 0.13 | 0.56 | 0.25 | 0.63 | 0.13 | 0.06 | 0 | 0.75 | 0.13 | 1.38 | 0 | 0 | 1.13 |
| | Pine | 2 | 0.75 | 0 | 0 | 0 | 0 | 0 | 0 | 0.13 | 0.19 | 0 | 5.19 | 0.06 | 0.19 | 0 | 0.19 | 0 | 0.38 | 0 | 0 | 0.19 | 0 | 0.06 | 0 | 0 | 0 | 0 | 0.31 | 0 | 0 | 0.31 |
| | | 5 | 0.69 | 0 | 0 | 0.06 | 0 | 0.19 | 0.44 | 0.25 | 0.06 | 0.13 | 2 | 0 | 1 | 0 | 1 | 0.38 | 1.19 | 0.56 | 0.44 | 2.38 | 0 | 3.06 | 0.25 | 0 | 0 | 0.81 | 0.69 | 0 | 0.31 | 1.63 |
| | Seaforth | 2 | 0.44 | 0 | 0 | 0.13 | 0 | 0 | 0.38 | 0.13 | 0.31 | 0 | 0 | 0.44 | 0.25 | 0 | 0.56 | 0.06 | 0 | 0 | 0 | 0.63 | 3.63 | 0 | 0.06 | 0 | 0.13 | 0 | 4.88 | 0 | 0.94 | 0.56 |
| | | 5 | 0.75 | 0 | 0.5 | 0.06 | 0.94 | 0 | 0 | 0.25 | 0.06 | 0.25 | 0 | 0.13 | 6.44 | 0 | 0.19 | 0 | 0.25 | 0.06 | 0 | 0 | 1.13 | 0.13 | 0.13 | 0 | 0 | 0 | 0.81 | 0 | 0.25 | 1.50 |

| (sub-)region | Reef | Depth | *Acropora* | *Alveopora* | *Caulastrea* | *Cyphastrea* | *Diploastrea* | *Echinophyllia* | *Echinopora* | *Favia* | *Favites* | *Fungia* | *Galaxea* | *Goniastrea* | *Goniopora* | *Isopora* | *Lobophyllia* | *Merulina* | *Montipora* | *Mycedium* | *Oxypora* | *Pachyseris* | *Pavona* | *Pectinia* | *Platygyra* | *Plesiastrea* | *Pocillopora* | *Podobacia* | *Porites* | *Symphyllia* | *Turbinaria* | *Other* |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Fitzroy | Barren | 2 | 39.06 | 0 | 0 | 0.06 | 0 | 0 | 0 | 0 | 0.19 | 0 | 0 | 0.13 | 0 | 1.19 | 0 | 0 | 6.81 | 0 | 0 | 0 | 0 | 0 | 0.63 | 0 | 0.25 | 0 | 0.06 | 0.06 | 0 | 0.69 |
| | | 5 | 70.38 | 0 | 0 | 0.06 | 0 | 0 | 0 | 0 | 0.06 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4.56 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.31 | 0 | 0.13 | 0 | 0 | 0 |
| | North Keppel | 2 | 36.47 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.13 | 0 | 0 | 0 | 0 | 0 | 0 | 0.31 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.06 |
| | | 5 | 20.25 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.31 | 0 | 0 | 0 | 0 | 0.63 | 0 | 3.31 | 0.19 | 0 | 0 | 0 | 0 | 0.19 | 0 | 0 | 0 | 0 | 0 | 0 | 0.06 |
| | Middle | 2 | 18 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.06 | 0.5 | 0 | 0 | 0 | 0 | 0 | 0 | 2.63 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.63 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 15.5 | 0 | 0 | 0.13 | 0 | 0 | 0 | 0 | 0 | 0.5 | 0 | 0 | 0 | 0 | 0 | 0 | 4.75 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.38 | 0 | 0 | 0 | 0 | 0 |
| | Keppels South | 2 | 33.31 | 0 | 0 | 0.19 | 0 | 0 | 0 | 0 | 0 | 0.19 | 0 | 0 | 0.19 | 0 | 0 | 0 | 9.38 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2.38 | 0 | 0 | 0 | 0.06 | 0.06 |
| | | 5 | 30.31 | 0 | 0 | 0 | 0 | 0 | 0 | 0.06 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2.75 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.06 | 0 | 0.19 | 0 | 0 | 0.19 |
| | Pelican | 2 | 0.31 | 0.25 | 0 | 0.06 | 0 | 0 | 0 | 0.06 | 0.06 | 0 | 0 | 0.06 | 0 | 0 | 0 | 0 | 0.44 | 0 | 0 | 0 | 0 | 0 | 0 | 0.56 | 0.19 | 0 | 0 | 0 | 0 | 0.44 |
| | | 5 | 0 | 5.13 | 0 | 0.63 | 0 | 0.06 | 0 | 0 | 2.81 | 0 | 0 | 2 | 0.94 | 0 | 0.63 | 0 | 0.19 | 0 | 0 | 0 | 0 | 0 | 0.94 | 0.56 | 0.06 | 0 | 0.13 | 0 | 1.38 | 1.50 |

Table A 10 Percent cover of soft coral families 2021. Families for which cover did not exceeded 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 | Gorgoniidae like | Other |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Barron Daintree | Low Isles | 5 | 0.26 | 0 | 7.91 | 0.17 | 0 | 0 | 0 | 0.03 | 0.07 |
| | Snapper North | 2 | 0.06 | 0.04 | 4.08 | 2.63 | 0 | 0 | 0.02 | 0 | 0 |
| | | 5 | 0 | 0.75 | 0.06 | 0.06 | 0 | 0 | 1.59 | 0 | 0 |
| | Snapper South | 2 | 0.25 | 0 | 0.29 | 0.04 | 3.58 | 0 | 0 | 0 | 0 |
| | | 5 | 0.01 | 0 | 5.25 | 0 | 5.13 | 0 | 0 | 0 | 0.09 |
| Johnstone Russell-Mulgrave | Green | 5 | 0.23 | 0 | 0.17 | 0 | 0.03 | 0.01 | 0 | 0 | 0.03 |
| | Fitzroy East | 2 | 0.35 | 0 | 0.31 | 0.25 | 0 | 0.03 | 0 | 0 | 0 |
| | | 5 | 0.85 | 0 | 6.13 | 0.13 | 0 | 0.05 | 0 | 0 | 0 |
| | Fitzroy West | 2 | 3.93 | 0 | 0.5 | 0 | 0 | 0 | 0.01 | 0 | 0 |
| | | 5 | 3.3 | 0 | 0.13 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Fitzroy West LTMP | 5 | 1.77 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Franklands East | 2 | 0.08 | 0 | 0 | 0.38 | 0.25 | 0.01 | 0.02 | 0 | 0 |
| | | 5 | 0.2 | 0 | 0.19 | 0.5 | 0 | 0 | 0 | 0 | 0 |
| | Franklands West | 2 | 0.76 | 0 | 0 | 4.72 | 0 | 0.03 | 0 | 0 | 0 |
| | | 5 | 0.12 | 0 | 0 | 0.69 | 0.06 | 0 | 0 | 0 | 0 |
| | High East | 2 | 0.9 | 0 | 5.63 | 0.03 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0.06 | 0 | 10.38 | 0 | 0 | 0 | 0 | 0 | 0 |
| | High West | 2 | 0.51 | 0 | 0.06 | 0 | 4 | 0 | 0 | 0 | 0 |
| | | 5 | 0.26 | 0 | 7.91 | 0.17 | 1.56 | 0 | 0 | 0 | 0.06 |

| (sub-)region | Reef | Depth | Alcyoniidae | Anthothelidae | Briareidae | Clavulariinae | Heliporidae | Neptheidae | Xeniidae | Gorgoniidae like | Other |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Herbert Tully | Barnards | 2 | 0.08 | 0 | 2.19 | 0.03 | 0 | 0 | 0.28 | 0 | 0 |
| | | 5 | 0.09 | 0 | 3.5 | 0 | 0 | 0 | 0.4 | 0 | 0 |
| | Dunk North | 2 | 0.23 | 0 | 0.44 | 0.06 | 0 | 0 | 0.1 | 0 | 0 |
| | | 5 | 0.18 | 0 | 0.31 | 0 | 0 | 0 | 0.49 | 0.69 | 0.09 |
| | Dunk South | 2 | 0.02 | 0 | 0.63 | 0.13 | 0 | 0 | 0.01 | 0 | 0 |
| | | 5 | 0.04 | 0 | 2.13 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Bedarra | 2 | 0.03 | 0 | 0 | 0 | 0 | 0 | 0.01 | 0 | 0 |
| | | 5 | 0.06 | 0.31 | 2.56 | 0 | 0 | 0.02 | 0.01 | 0.13 | 0 |
| Burdekin | Palms East | 2 | 0.16 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0.4 | 0 | 0 | 0 | 0 | 0 | 0.01 | 0 | 0 |
| | Palms West | 2 | 2.1 | 0 | 0.44 | 0.31 | 0 | 0.89 | 0 | 0 | 0 |
| | | 5 | 1.7 | 0 | 5.75 | 0.22 | 0 | 0.26 | 0 | 0.31 | 0.09 |
| | Havannah North | 5 | 0.04 | 0 | 1.3 | 0.58 | 0 | 0 | 0 | 0 | 0 |
| | Havannah | 2 | 0.06 | 0 | 3.06 | 0 | 0 | 0.03 | 0 | 0 | 0 |
| | | 5 | 0.02 | 0 | 6.69 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Pandora | 2 | 0.04 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0.16 | 0 | 0 | 0.03 | 0 | 0 | 0 | 0 | 0.09 |
| | Pandora North | 5 | 0.07 | 0 | 7.74 | 2.28 | 0 | 0 | 0 | 0 | 0.02 |
| | Lady Elliot | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.03 |
| | | 5 | 0.02 | 0.63 | 0.19 | 0 | 0 | 0 | 0.01 | 0 | 0.03 |
| | Magnetic | 2 | 0.06 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0.21 | 0 | 0.06 | 0 | 0 | 0 | 0 | 0.25 | 0 |

| (sub-)region | Reef | Depth | Alcyoniidae | Anthothelidae | Briareidae | Clavulariinae | Heliporidae | Neptheidae | Xeniidae | Gorgoniidae like | Other |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Mackay Whitsunday | Hayman | 5 | 0.75 | 0 | 0.46 | 0 | 0 | 0.01 | 0 | 0 | 0 |
| | Langford | 5 | 0.43 | 0 | 0.33 | 0 | 0 | 0 | 0 | 0 | 0.03 |
| | Border | 5 | 1.98 | 0 | 0.17 | 0 | 0 | 0.03 | 0.01 | 0.03 | 0.2 |
| | Hook | 2 | 0.57 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0.92 | 0 | 1.31 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Double Cone | 2 | 0.09 | 0 | 0.06 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0.08 | 0 | 0.06 | 0 | 0 | 0.01 | 0 | 0 | 0 |
| | Daydream | 2 | 0 | 0 | 0 | 0 | 0 | 0.01 | 0 | 0 | 0 |
| | | 5 | 0.03 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Dent | 2 | 0.29 | 0 | 2.94 | 0 | 0 | 0.01 | 0 | 0 | 0 |
| | | 5 | 0.35 | 0.06 | 0.81 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Shute Harbour | 2 | 1.01 | 0 | 0.06 | 0 | 0 | 0.05 | 0.1 | 0 | 0 |
| | | 5 | 0.34 | 0.06 | 0 | 0 | 0 | 0.03 | 0 | 0 | 0 |
| | Pine | 2 | 0.08 | 0 | 0.25 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0.07 | 0 | 0.19 | 0 | 0 | 0 | 0.04 | 0 | 0 |
| | Seaforth | 2 | 0.29 | 0 | 1.19 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0.07 | 1.75 | 0.06 | 0 | 0 | 0 | 0.08 | 0 | 0.03 |

| (sub-)region | Reef | Depth | Alcyoniidae | Anthothelidae | Briareidae | Clavulariinae | Heliporidae | Neptheidae | Xeniidae | Gorgoniidae like | Other |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Fitzroy | Barren | 2 | 0.49 | 0 | 0.25 | 0 | 0 | 0 | 0.47 | 0 | 0 |
| | | 5 | 0.08 | 0 | 0 | 0 | 0 | 0 | 0.38 | 0 | 0 |
| | North Keppel | 2 | 0.01 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0.12 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Middle | 2 | 0.05 | 0 | 0 | 0 | 0 | 0 | 0.04 | 0 | 0 |
| | | 5 | 0.03 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Keppels South | 2 | 0.1 | 0 | 0 | 0 | 0 | 0 | 0.14 | 0 | 0 |
| | | 5 | 0.02 | 0 | 0 | 0 | 0 | 0 | 0.06 | 0 | 0 |
| | Pelican | 2 | 0.08 | 0.31 | 0 | 0 | 0 | 0 | 0.03 | 0 | 0.06 |
| | | 5 | 0.6 | 0.19 | 0.06 | 0 | 0 | 0.08 | 0.04 | 1.06 | 0.13 |

Table A 11 Percent cover of Macroalgae groups 2021. Genera for which cover exceeded 0.5% on at least one reef are included, rare or unidentified genera are grouped to 'Unidentified'.

| (sub)region | Reef | Depth | Rhodophyta (red algae) | | | | | Chlorophyta (green algae) | | | Phaeophyta (brown algae) | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | *Asparagopsis* | *Hypnea* | *Peyssonnelia* | *Calcareous* | Undefined | *Caulerpa* | *Halimeda* | Undefined | *Dictyota* | *Lobophora* | *Padina* | *Sargassum* | *Stypopodium* | Undefined |
| Barron Daintree | Low Isles | 5 | 0 | 0 | 0 | 0 | 0.17 | 0 | 0.33 | 0.03 | 0 | 0 | 0 | 0 | 0 | 0.07 |
| | Snapper North | 2 | 3.5 | 1 | 0.25 | 0.63 | 39.04 | 0 | 2.46 | 0.13 | 8.04 | 0 | 0.33 | 0 | 0 | 0.25 |
| | | 5 | 0.06 | 0 | 0 | 0.06 | 0.13 | 0 | 0.19 | 0 | 1.13 | 0 | 0.06 | 0 | 0 | 0 |
| | Snapper South | 2 | 0 | 0.58 | 0.54 | 0.42 | 1.63 | 0 | 0 | 0.25 | 0.08 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0 | 0.31 | 0.38 | 4.44 | 2.5 | 0 | 0 | 0.19 | 0.13 | 0.25 | 0 | 0 | 0 | 0 |
| Johnstone Russell-Mulgrave | Green | 5 | 0 | 0 | 0 | 0.05 | 0.7 | 0 | 4.14 | 0.25 | 0 | 0.05 | 7.4 | 0 | 0 | 5.33 |
| | Fitzroy East | 2 | 0 | 0.06 | 0 | 0.06 | 0.06 | 0 | 0 | 0.13 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0 | 0 | 0.13 | 0 | 0.19 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Fitzroy West | 2 | 0 | 0.31 | 0.19 | 0.06 | 0.38 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0 | 0 | 0.13 | 0 | 0.31 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Fitzroy West LTMP | 5 | 0 | 0 | 0.23 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Franklands East | 2 | 0 | 0.56 | 0 | 0 | 0.38 | 0 | 0 | 0 | 0 | 0.06 | 0 | 0 | 0 | 0 |
| | | 5 | 0 | 0 | 0.13 | 0.63 | 1.13 | 0 | 0 | 0.06 | 0 | 0.13 | 0 | 0 | 0 | 0.13 |
| | Franklands West | 2 | 0 | 1.06 | 0.06 | 0.19 | 7.88 | 0 | 0 | 0 | 0 | 0.31 | 0 | 0 | 0 | 0 |
| | | 5 | 0 | 0.25 | 0.19 | 1.38 | 13.75 | 0.19 | 0.75 | 0.13 | 0 | 0.56 | 0 | 0 | 0 | 0 |
| | High East | 2 | 0 | 3.13 | 0.06 | 0.06 | 3.31 | 0 | 0 | 0.13 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0 | 0.38 | 0.19 | 0 | 0.31 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | High West | 2 | 0 | 0.13 | 0.19 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0 | 0.13 | 0 | 0.06 | 0.38 | 0.06 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

| (sub)region | Reef | Depth | Rhodophyta (red algae) | | | | | Chlorophyta (green algae) | | | Phaeophyta (brown algae) | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | *Asparagopsis* | *Hypnea* | *Peyssonnelia* | *Calcareous* | Undefined | *Caulerpa* | *Halimeda* | Undefined | *Dictyota* | *Lobophora* | *Padina* | *Sargassum* | *Stypopodium* | Undefined |
| Herbert Tully | Barnards | 2 | 0 | 0.44 | 0.19 | 0 | 0.19 | 0.06 | 0 | 0.06 | 0.06 | 0.06 | 0 | 0 | 0 | 0.13 |
| | | 5 | 0 | 0 | 0.19 | 0 | 0.31 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Dunk North | 2 | 0 | 0 | 0 | 0 | 1.94 | 0 | 0 | 0.06 | 0.38 | 0.06 | 0.06 | 10.25 | 0 | 0 |
| | | 5 | 0 | 0 | 0 | 0 | 0.81 | 0 | 0 | 0 | 0.5 | 0.06 | 0 | 1.69 | 0 | 0.06 |
| | Dunk South | 2 | 0 | 0 | 0.06 | 0.19 | 1.56 | 0 | 0 | 0.19 | 0.25 | 3 | 0.06 | 8.31 | 0.56 | 0.38 |
| | | 5 | 0 | 0 | 0.38 | 0 | 1.38 | 0 | 0 | 0 | 0 | 6.06 | 0 | 0.06 | 0 | 0 |
| | Bedarra | 2 | 0 | 1.5 | 0 | 0.19 | 2.69 | 0 | 0.06 | 0.19 | 5.69 | 0.63 | 0.19 | 16.81 | 0 | 0.13 |
| | | 5 | 0 | 0 | 0.06 | 0 | 0.88 | 0 | 0 | 0 | 1 | 0.5 | 0 | 0.25 | 0.06 | 0 |
| Burdekin | Palms East | 2 | 0 | 0 | 0 | 0 | 0.06 | 0.88 | 0 | 0.31 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0 | 0 | 0.06 | 0 | 0.19 | 0.06 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | Palms West | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.13 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0 | 0 | 0.19 | 0 | 0.25 | 0.06 | 0 | 0.06 | 0 | 0.06 | 0 | 0 | 0 | 0 |
| | Havannah North | 5 | 0.1 | 0 | 1.05 | 0 | 0.46 | 2.04 | 0.05 | 0.1 | 0 | 28.15 | 0.14 | 11.31 | 0 | 0.44 |
| | Havannah | 2 | 0 | 0 | 1.13 | 0 | 0.38 | 0.19 | 0 | 0.13 | 1 | 8.88 | 0 | 0.81 | 3 | 0.31 |
| | | 5 | 0 | 0 | 0.13 | 0 | 0.13 | 0.13 | 0 | 0.06 | 0.06 | 13.56 | 0 | 1.63 | 0.13 | 0.19 |
| | Pandora | 2 | 0 | 0 | 0.06 | 0 | 0.44 | 0 | 0 | 0 | 1 | 1.81 | 0 | 17.38 | 0 | 0 |
| | | 5 | 0 | 0 | 0.06 | 0 | 0.25 | 0.06 | 0 | 0 | 2.19 | 1.25 | 0 | 0.19 | 0 | 0.1 |
| | Pandora North | 5 | 0.2 | 0 | 0.5 | 0.1 | 2.99 | 0 | 0 | 0 | 0 | 9.1 | 0 | 24.59 | 0 | 0.13 |
| | Lady Elliot | 2 | 0 | 5.94 | 1.13 | 0.06 | 1.75 | 0 | 0 | 0 | 1 | 0.13 | 0.13 | 0.81 | 0 | 0 |
| | | 5 | 0 | 0.19 | 0.88 | 0 | 0.63 | 0 | 0 | 0 | 0.06 | 0 | 0 | 0 | 0 | 1.75 |
| | Magnetic | 2 | 0 | 0.06 | 0.38 | 0 | 0.81 | 0 | 0 | 0 | 9.06 | 12.44 | 0.56 | 11.38 | 0 | 0.25 |
| | | 5 | 0 | 0 | 0.25 | 0.13 | 6.63 | 0 | 0 | 0 | 7.06 | 3.56 | 0.06 | 8.75 | 0 | 0.51 |

| (sub)region | Reef | Depth | Rhodophyta (red algae) | | | | | Chlorophyta (green algae) | | | Phaeophyta (brown algae) | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | *Asparagopsis* | *Hypnea* | *Peyssonnelia* | *Calcareous* | Undefined | *Caulerpa* | *Halimeda* | Undefined | *Dictyota* | *Lobophora* | *Padina* | *Sargassum* | *Stypopodium* | Undefined |
| Mackay Whitsunday | Hayman | 5 | 0 | 0 | 0.49 | 0 | 1.66 | 0 | 0 | 0 | 0 | 0.1 | 0 | 0 | 0 | 4.16 |
| | Langford | 5 | 0 | 0 | 0 | 0 | 0.17 | 0 | 0 | 0 | 0 | 0.03 | 0 | 0 | 0 | 0.17 |
| | Border | 5 | 0 | 0 | 0 | 0 | 0.15 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.05 |
| | Hook | 2 | 0 | 0 | 0 | 1.31 | 0.44 | 4.63 | 0 | 0.06 | 0.06 | 0.25 | 0 | 0 | 0 | 0 |
| | | 5 | 0 | 0 | 0.06 | 0.06 | 0.31 | 0.31 | 0 | 0 | 0.06 | 0.13 | 0 | 0 | 0 | 0 |
| | Double Cone | 2 | 0 | 0 | 0.13 | 6.25 | 3.38 | 0.25 | 0 | 0.13 | 5.81 | 5.44 | 0.44 | 7.06 | 0 | 2 |
| | | 5 | 0 | 0.06 | 0.19 | 1.63 | 5.06 | 0 | 0 | 0 | 4.13 | 0.94 | 0.25 | 6 | 0 | 2.38 |
| | Daydream | 2 | 0 | 0 | 0 | 1.19 | 18.88 | 0 | 0 | 0.06 | 0.5 | 2.56 | 0.81 | 9.25 | 0 | 9.5 |
| | | 5 | 0 | 0 | 0 | 1 | 0.56 | 0 | 0 | 0 | 0.13 | 2.75 | 0.19 | 0.06 | 0 | 0.19 |
| | Dent | 2 | 0 | 0 | 1 | 0.13 | 0.44 | 0 | 0 | 0 | 0.06 | 6.5 | 0 | 0.06 | 0 | 0.19 |
| | | 5 | 0 | 0 | 0.38 | 0.06 | 0.88 | 0 | 0 | 0 | 0 | 2.69 | 0 | 0 | 0 | 0 |
| | Shute Harbour | 2 | 0 | 0 | 0.06 | 0 | 0.19 | 0 | 0 | 0 | 0.06 | 1.25 | 0.06 | 0.44 | 0 | 0.19 |
| | | 5 | 0 | 0 | 0 | 0.06 | 0.25 | 0 | 0 | 0.06 | 0 | 0.5 | 0.06 | 0 | 0 | 0.19 |
| | Pine | 2 | 0 | 0 | 1.13 | 0.25 | 6 | 0.06 | 0.06 | 0.63 | 1.06 | 8.31 | 0.25 | 12.31 | 0 | 1.31 |
| | | 5 | 0 | 0 | 1.75 | 0 | 1 | 0 | 0.44 | 0.31 | 0 | 8.81 | 0 | 0.38 | 0 | 0 |
| | Seaforth | 2 | 0 | 0.63 | 0 | 4.31 | 6.25 | 0 | 0.13 | 0 | 0.31 | 2.81 | 0.75 | 5.69 | 0 | 3.88 |
| | | 5 | 0 | 0 | 0 | 4.81 | 1.19 | 0 | 0 | 0.13 | 0.38 | 3.81 | 1.13 | 0.94 | 0 | 0.56 |

| (sub)region | Reef | Depth | Rhodophyta (red algae) | | | | | Chlorophyta (green algae) | | | Phaeophyta (brown algae) | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | *Asparagopsis* | *Hypnea* | *Peyssonnelia* | *Calcareous* | Undefined | *Caulerpa* | *Halimeda* | Undefined | *Dictyota* | *Lobophora* | *Padina* | *Sargassum* | *Stypopodium* | Undefined |
| Fitzroy | Barren | 2 | 0 | 0.06 | 0.06 | 0 | 0.44 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | | 5 | 0 | 0 | 0.06 | 0 | 3.81 | 0 | 0 | 0 | 0 | 2.31 | 0 | 0 | 0 | 0 |
| | North Keppel | 2 | 0 | 0 | 0.38 | 0 | 0.56 | 0 | 0 | 0 | 0 | 35.27 | 0 | 0 | 0 | 0 |
| | | 5 | 0 | 0 | 1.56 | 0 | 0.19 | 0 | 0 | 0 | 2.81 | 31.31 | 0 | 0 | 0 | 0 |
| | Middle | 2 | 0 | 0 | 0.75 | 0 | 0.63 | 0 | 0 | 0 | 0 | 23.94 | 0 | 30.5 | 0 | 0.06 |
| | | 5 | 0 | 0 | 0.63 | 0 | 0.88 | 0 | 0 | 0 | 1.25 | 17.5 | 0 | 33 | 0 | 0.13 |
| | Keppels South | 2 | 0 | 0 | 1.19 | 0 | 0.63 | 0.13 | 0 | 0 | 0.5 | 13.25 | 0.81 | 14.44 | 0.06 | 0.19 |
| | | 5 | 0 | 0 | 1.13 | 0 | 0.5 | 0 | 0 | 0 | 0.63 | 19.5 | 0.06 | 0.06 | 0 | 0.06 |
| | Pelican | 2 | 0.19 | 0.13 | 0.38 | 4.75 | 3.94 | 0 | 0 | 0 | 2.56 | 30.81 | 0.19 | 3.63 | 0.25 | 0.63 |
| | | 5 | 0 | 0 | 0.69 | 0.13 | 2.69 | 0 | 0 | 0 | 1.38 | 9.5 | 0 | 0.25 | 0 | 0.19 |

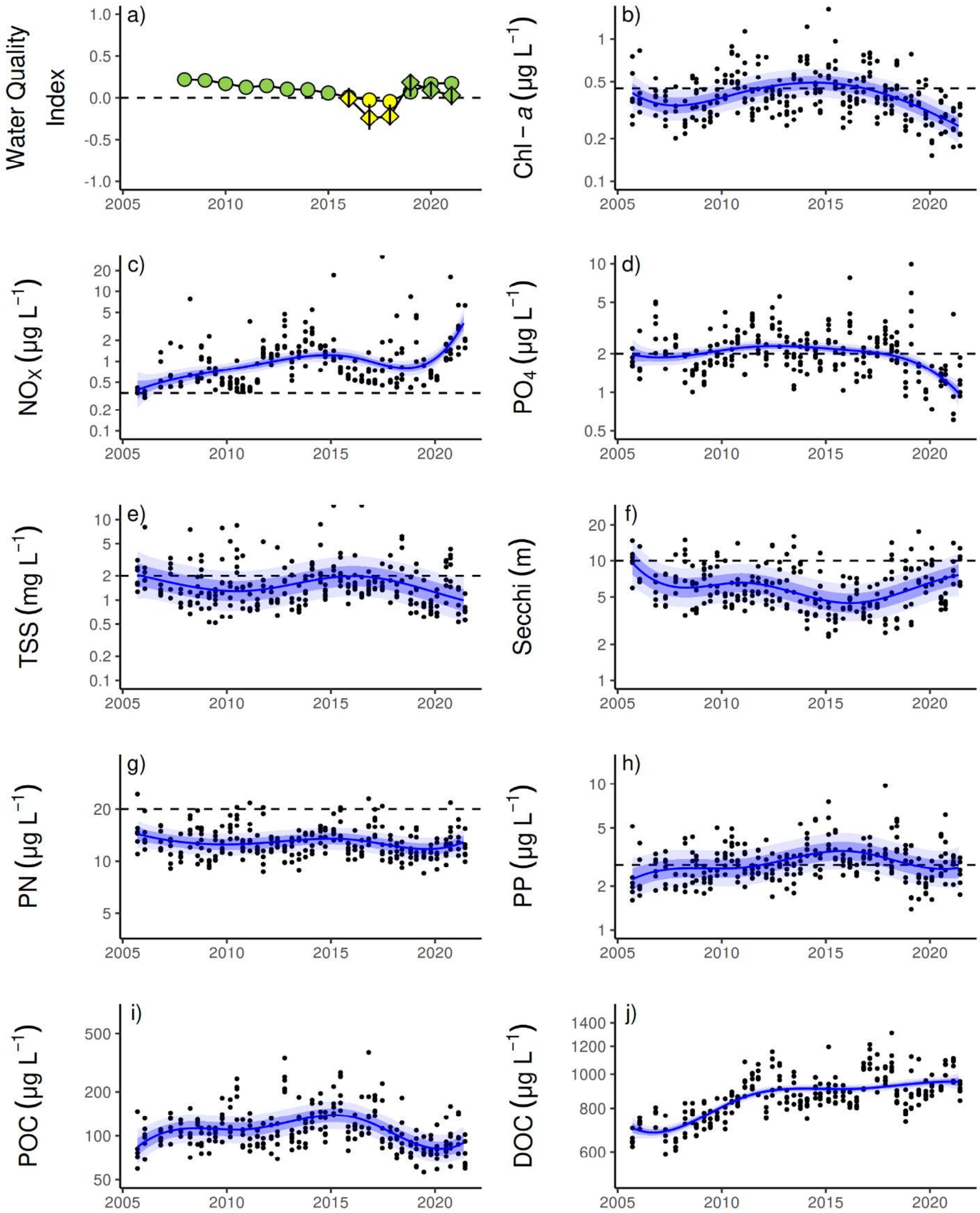


Figure A 10 Temporal trends in water quality: Barron Daintree sub-region. a) water quality index, b) chlorophyll *a*, c) nitrate/nitrite, d) Phosphate, e) total suspended solids, f) secchi depth, g) particulate nitrogen, h) particulate phosphorus, i) particulate organic carbon and j) dissolved organic carbon. Water quality index colour coding: dark green- 'very good'; light green – 'good'; yellow – 'moderate; orange – 'poor'; red – 'very poor'. The 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*. (2022).

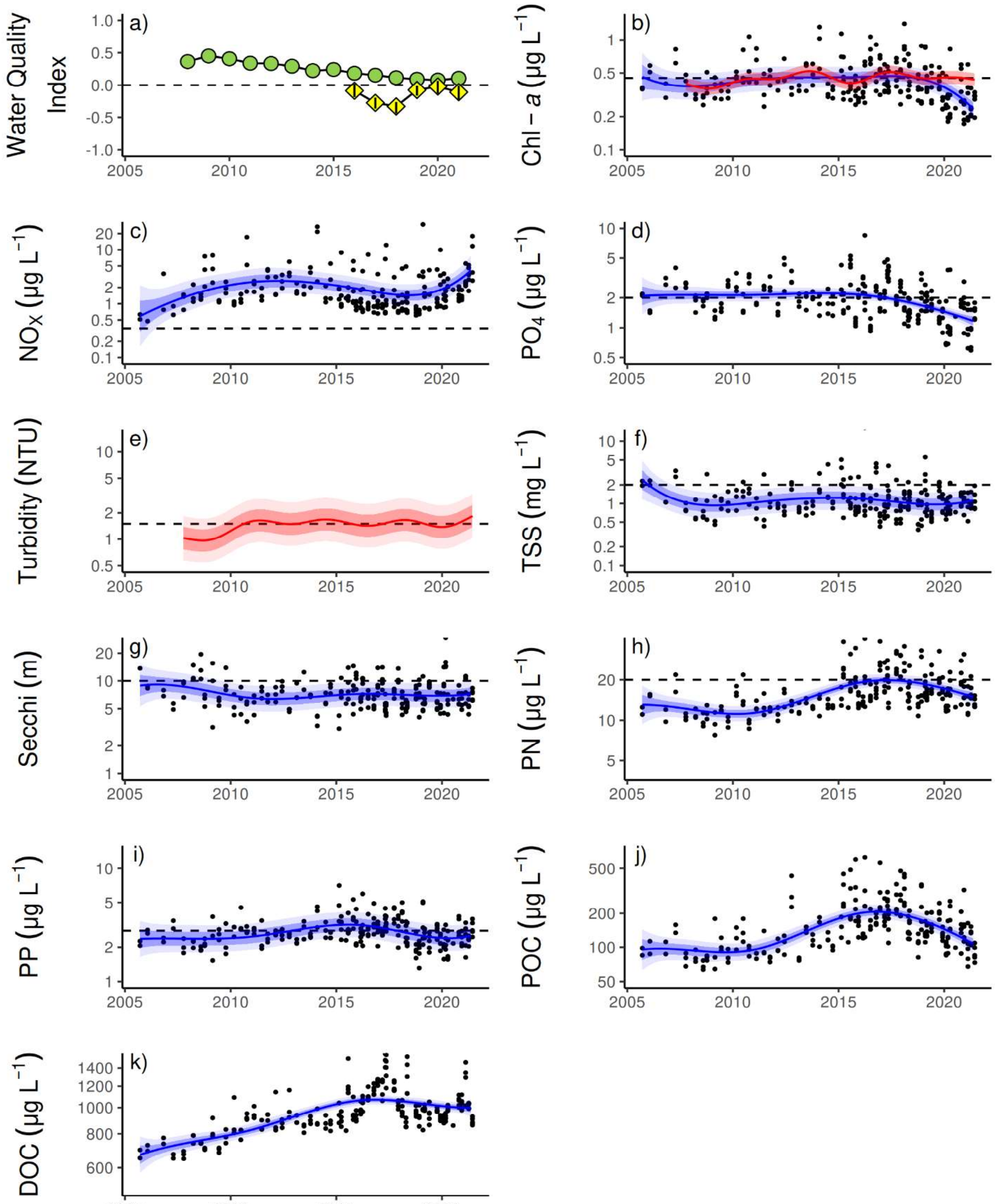


Figure A 11 Temporal trends in water quality: Johnstone Russell-Mulgrave sub-region. a) water quality index, b) chlorophyll *a*, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus j), particulate organic carbon 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 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.* (2022).

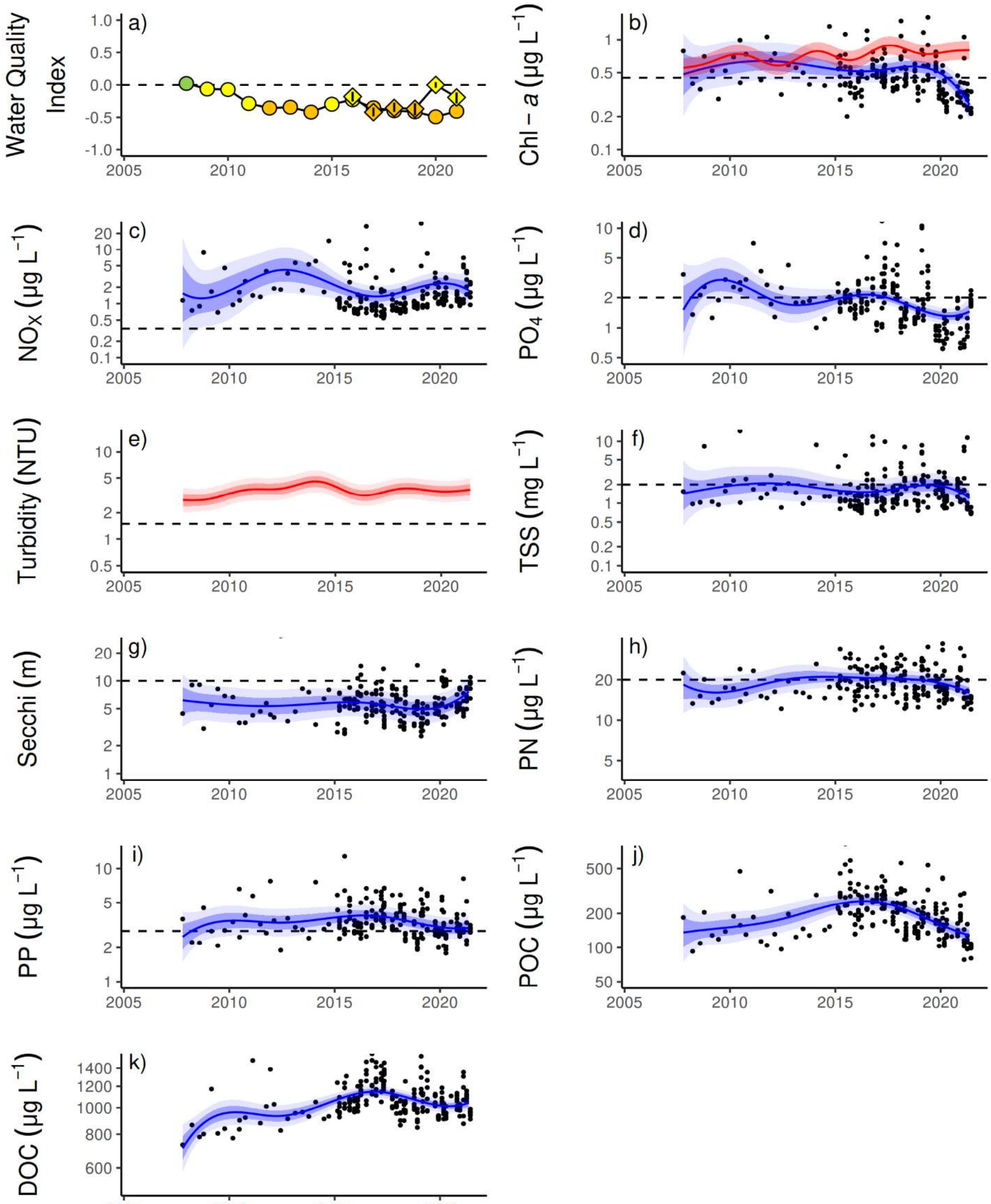


Figure A 12 Temporal trends in water quality: Herbert Tully sub-region. a) water quality index, b) chlorophyll *a*, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved organic carbon. Water quality index colour coding: dark green- 'very good'; light green – 'good'; yellow – 'moderate; orange – 'poor'; red – 'very poor'. The 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.* (2022).

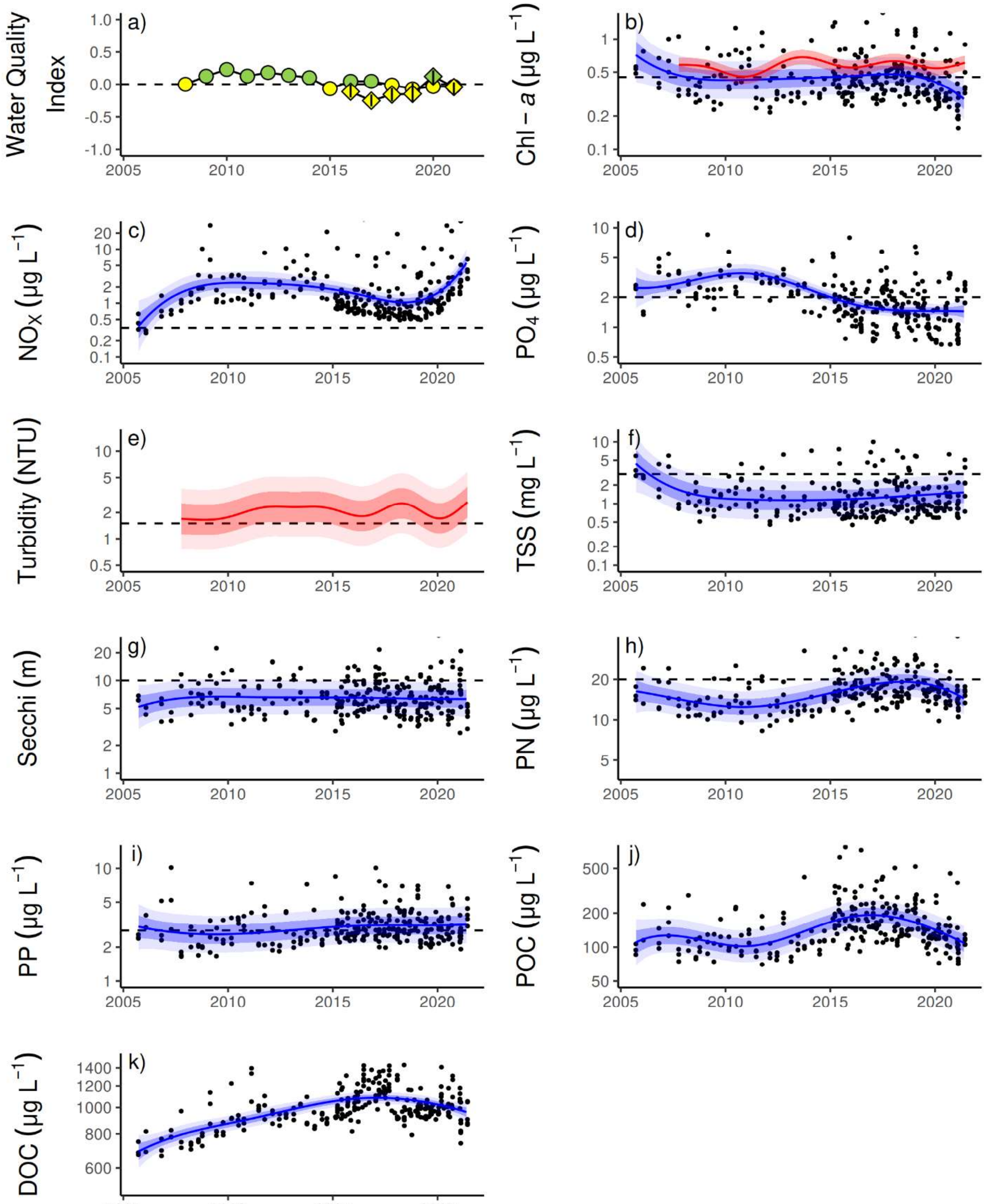


Figure A 13 Temporal trends in water quality: Burdekin region. a) water quality index, b) chlorophyll *a,* c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved organic carbon. Water quality index colour coding: dark green- 'very good'; light green – 'good'; yellow – 'moderate; orange – 'poor'; red – 'very poor'. The 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.* (2022).

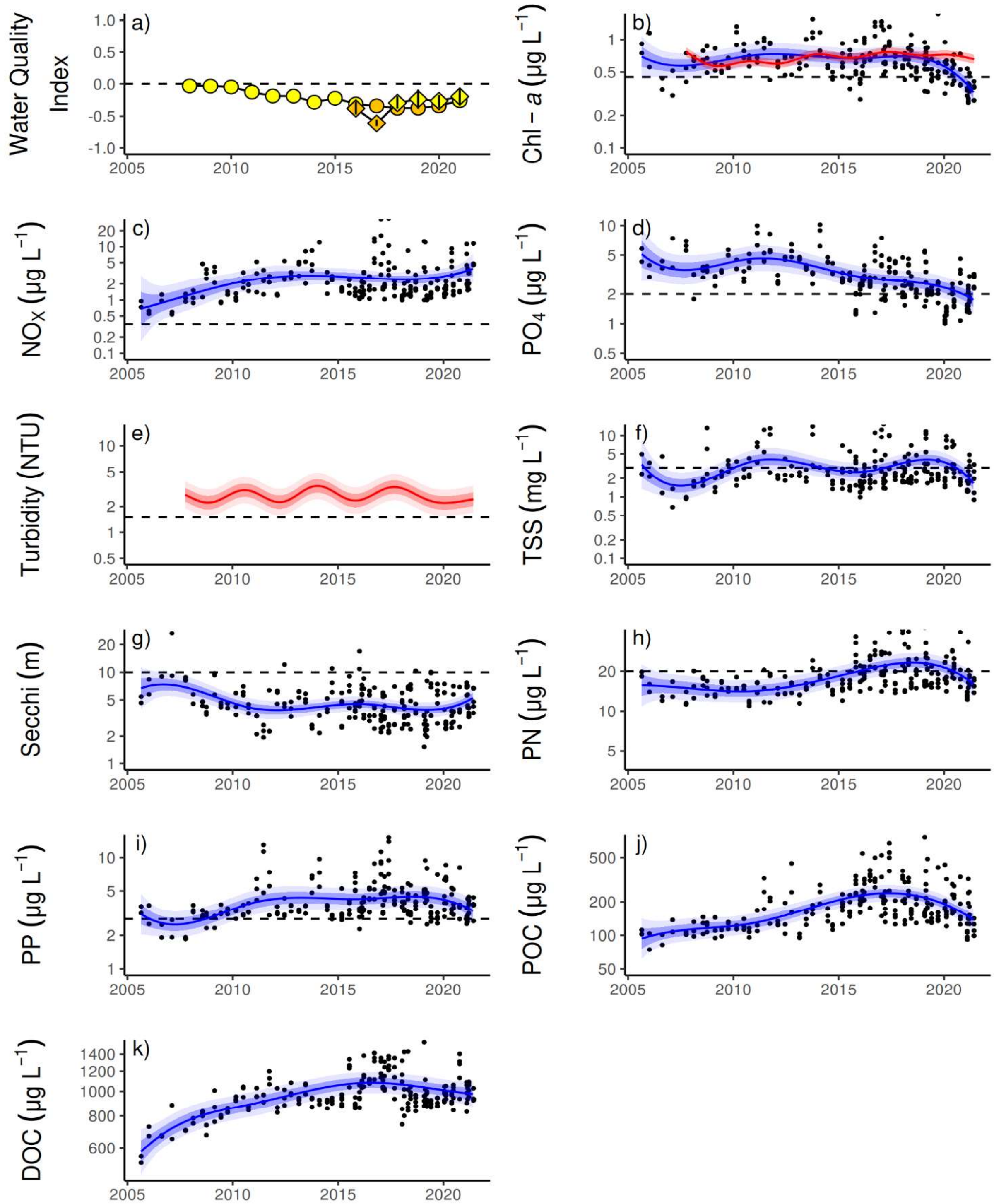


Figure A 14 Temporal trends in water quality: Mackay-Whitsunday Issac region.a) water quality index, b) chlorophyll *a*, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved organic carbon. Water quality index colour coding: dark green- 'very good'; light green – 'good'; yellow – 'moderate; orange – 'poor'; red – 'very poor'. The 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.* (2022).

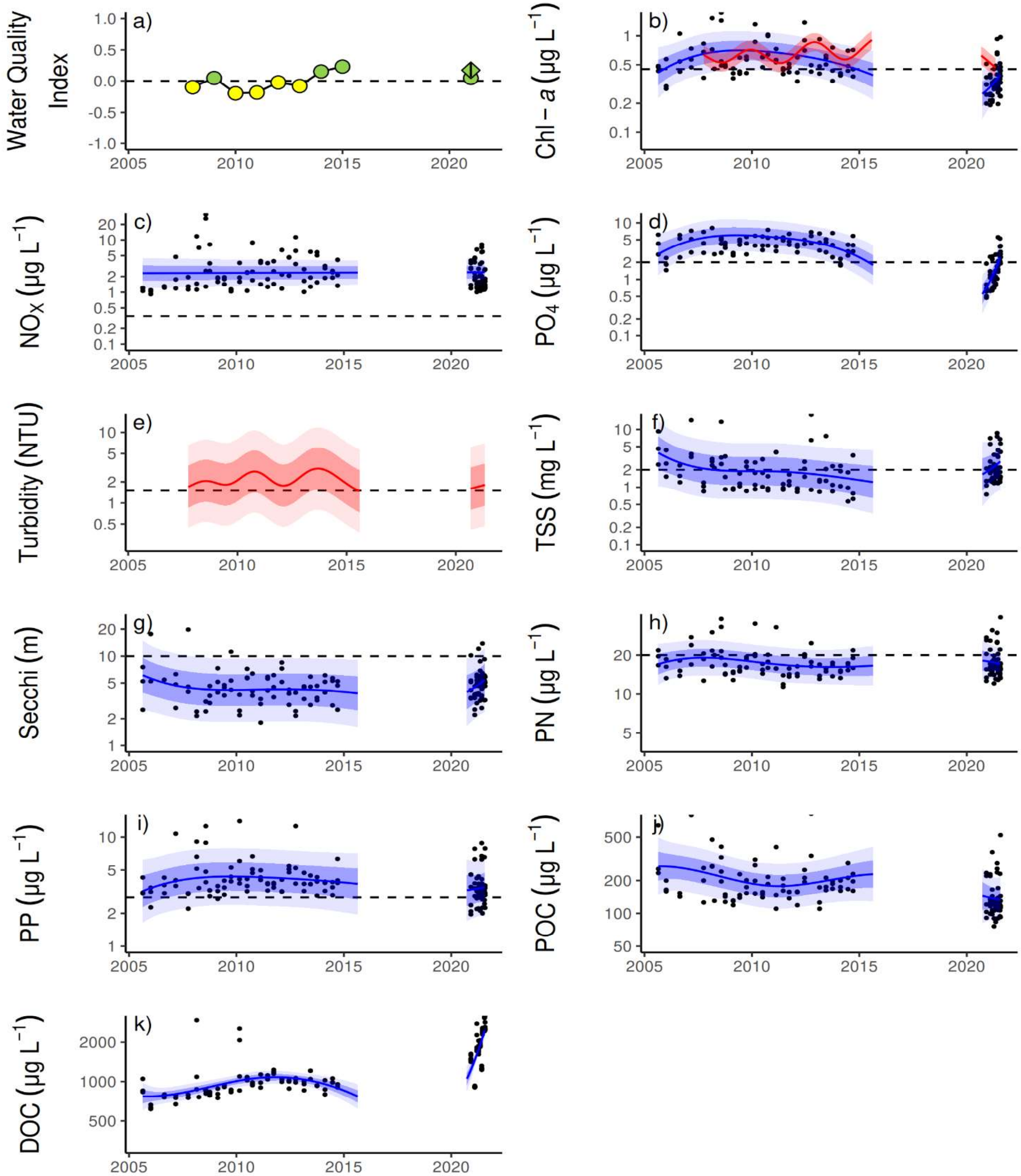


Figure A 15 Temporal trends in water quality: Fitzroy region.. a) water quality index, b) chlorophyll *a*, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved organic carbon. Water quality index colour coding: dark green- 'very good'; light green – 'good'; yellow – 'moderate; orange – 'poor'; red – 'very poor'. The 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*. (2021).

# 9 Appendix 3: Changes to spatial sampling design and influence on Coral Index scores

## 9.1 Abstract

Over the duration of the Great Barrier Reef Marine Monitoring Program (MMP) the coral reef sampling design has been altered in response to a combination of:

- safety concerns,
- the appropriateness of sites as being representative of inshore coral reef communities and,
- budgetary limitations.

The result is that six reefs included in the initial program design in 2005 are no longer included and one additional reef has been added. The reefs changed include:

- Three coastal fringing reefs in Baron Daintree sub-region. The reefs were only included in 2005 and 2006 and were removed due to concerns for diver safety.
- King Reef in the Herbert Tully sub-region that was excluded in 2015 in favour of Bedarra Island on the basis that King Reef was a rocky reef rather than coral reef habitat.
- Middle Reef in the Townsville region where sites were also monitored the Australian Institute of Marine Science's Long-term Monitoring Program (LTMP), however that Program has since removed the reef from their sampling design.
- Peak Island in the Fitzroy region removed in 2020, also on the basis that this did not represent a coral reef habitat.

These changes have little influence on the Coral Index scores in the respective regions and sub-regions.

## 9.2 Introduction

The overall intent of the sampling design for the coral reef component of the MMP was to spread sampling across naturally occurring gradients in water quality within each of the focus regions. In so doing, allowing changes in coral community condition to be assessed in response to both spatial and temporal variability in environmental conditions.

The initial selection of reefs sort to monitor coral communities from as close to the major rivers in each region out to those most distant, but still within the "inshore" zone where exposure to sediments and nutrients delivered by runoff was most likely. This constrained sampling to include reefs predominantly in the open coastal or mid-shelf water bodies. In some regions sites were included in areas where, although coral communities existed, there was no evidence of a carbonate substrate suggesting these communities were likely transient. As the program developed and the Coral Index was formulated as a way of summarising the condition of coral communities on inshore reefs the inclusion of these transient coral communities became problematic either from a workplace health and safety perspective or as outliers in the expectations underlying the Coral Index scoring, namely that 'coral reef' communities were being scored.

Changes to the spatial sampling design of the program have the potential to bias Coral Index scores relative to scores reported under prior designs. The purpose of this case study is to document the changes that have occurred over the duration of the MMP and summarise the impact of these changes on reported Coral Index scores.

## 9.3 Changes to reefs monitored by the MMP.

In 2005 and 2006 all reefs were scheduled for annual sampling. In 2007 budgetary limitations lead to the adoption of a two-tiered design with 14 'core' reefs, at which water quality was also monitored, continuing to be monitored annually and the remaining reefs dropping to a biennial cycle of monitoring. This change also gave pause to consider the sites that would be included into the future.

It was at this time that concerns for the safety of divers monitoring coastal fringing reefs of the Daintree were raised. Ever increasing saltwater crocodile populations were considered an unacceptable risk at these reefs. This risk, along with the cost saving of ceasing monitoring, saw the three coastal fringing reefs of the Daintree area removed from the MMP sampling design (Table 18).

A primary reporting product of the MMP is the Coral Index that was developed as a summary of coral reef community condition (Thompson *et al*, 2020). An underlying assumption in the application of this tool is that it is applied to coral reef communities. As the timeseries of the MMP progressed it became apparent that a small number of locations were not consistent with our view of a coral reef. These sites were originally selected with the intent of sampling across the full extent of coral reefs along water quality gradients in each region. However, this desire led to the selection of monitoring locations in conditions beyond those in which coral reefs had developed. This distinction made on the fact that there was little or no historical development of a calcium carbonate substrate, King Reef in the Herbert Tully sub-region and Peak Island in the Fitzroy region should be considered as rocky rather than coral reefs (Table 18).

Finally, Middle Reef in the Burdekin region was monitored by the LTMP. The site was included in the MMP primarily to infill the biennial sampling design of the LTMP. In 2013 this duplication across programs was removed and Middle Reef no longer scheduled as an MMP site. Unfortunately, weather and crocodile risks have precluded any subsequent sampling by the LTMP and Middle Reef has now been removed their sampling design (Table 18).

Table 18 Changes to the reefs monitored by the MMP.

| NRM region | Reef | First Survey | Last Survey | Reason |
|---|---|---|---|---|
| Wet Tropics | Cape Tribulation South | 2005 | 2006 | Initially included as a continuation of a historical data set collected by Sea Research. Coastal fringing reefs that are known crocodile habitat. Discontinued due to increasing crocodile population posing an unacceptable risk to divers. |
| | Cape Tribulation Mid | | | |
| | Cape Tribulation North | | | |
| | King Reef | 2005 | 2014 | Unsuitable habitat – Rock substrate, potentially negative bias on assessment of condition of coral reefs. |
| | Bedarra Island | 2015 | ongoing | Replaced King Reef as more suitable habitat and better alignment with water quality sampling design. |
| Burdekin | Middle Reef | 2005 | 2013 | Discontinued by MMP in 2015 (last survey 2013) due to duplicated effort as reef was also included in LTMP sampling design. However, LTMP have not surveyed the reef since 2013 due to a combination of logistical constraints (high turbidity) and safety concerns related to crocodiles. From 2015 to 2020 data for this reef were carried forward from 2013 LTMP surveys. LTMP have now formally removed this reef from their sampling design. |
| Fitzroy | Peak Island | 2005 | 2019 | Unsuitable habitat – Rock substrate, logistical constraints imposed by high turbidity. Removed from MMP sampling design in 2021. |

## 9.4 Consequence on Coral Index scores.

### 9.4.1 Wet Tropics – Barron Daintree sub-region

The discontinuation of sampling along the Cape Tribulation coastal reefs had no impact on Reef Water Quality Report Card scores as these reefs were removed from the sampling design prior to the implementation of Reef Water Quality Report Card summaries.

### 9.4.2 Wet Tropics – Herbert Tully sub-region

The reported Coral Index score for the Herbert Tully sub-region in 2015 of 0.48 included data from the first survey of Bedarra Island. By comparison, the 2015 Coral Index score, based instead on inclusion of carried forward data from King Reef was 0.45 (Figure A 16).

For the three indicators that were estimated from the first survey of Bedarra, scores were higher than 2014 estimates from King Reef for coral cover and juvenile coral indicators at 2 m depth and Macroalgae at 5 m depth and similar elsewhere (Table A 12)

As 2015 was the first survey for Bedarra Island scores for the cover change and composition indicators could not be estimated. As such the improvement in cover change scores when King reef data were excluded (Table A 12) reflects that estimates from King Reef in 2014 were below the mean of other reefs in 2015. Conversely higher scores for the composition indicator with the inclusion of King Reef means that composition scores for King Reef in 2014 were higher than the mean at other reefs in 2015, most notably at 2 m depth (Table A 12). The score for the composition indicator at 2 m at King Reef must be considered with some caution due to the very low coral cover in both in 2014 and over the baseline period 2005-2012.

Overall, the replacement of King Reef with Bedarra Island in the sampling design, is likely to have slightly improved the Coral Index scores for this sub-region. This bias should be viewed positively as the inclusion of King Reef was demonstrably underestimating the condition of coral reefs in the sub-region.

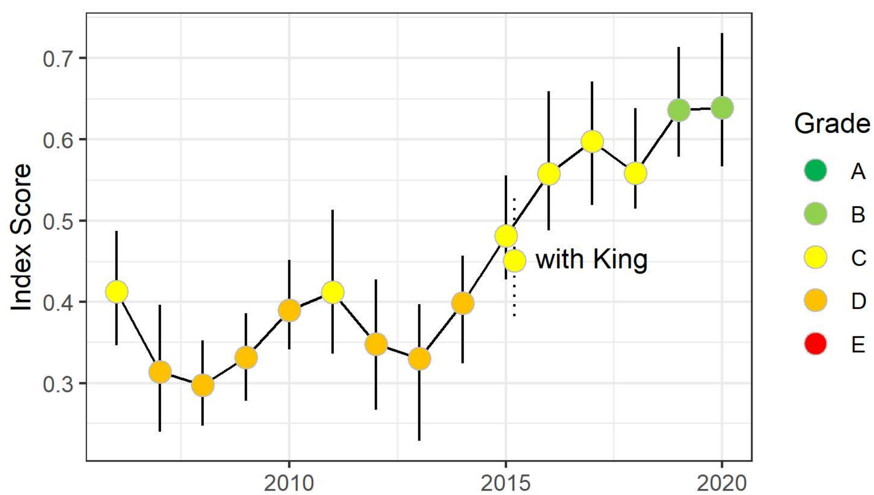


Figure A 16 Influence of sampling design change on Coral Index scores of the Herbert Tully sub-region. The trend is as reported and includes data from King Reef until 2014 and Bedarra Island thereafter. A second point for 2015 indicated the score for that year had data from King Reef been rolled forward from 2014 rather than data from Bedarra 2015.

Table A 12 Influence of sampling design change to indicator scores for Tully Herbert sub-region. Values are reported scores for 2015 based inclusion of Bedarra Island data, values in parentheses are the difference in scores estimated from carried forward data from King Reef in 2014.

| Depth | Coral cover | Juvenile coral | Macroalgae | Cover change | Composition |
|---|---|---|---|---|---|
| 2 m | 0.20 (-0.04) | 0.89 (-018) | 0.09 (0) | 0.60 (-0.09) | 0.33 (0.17) |
| 5 m | 0.30 (0) | 1 (0) | 0.52 (0.18) | 0.55 (-0.03) | 0.33 (0.05) |

### 9.4.3 Burdekin region

The inclusion of data from Middle Reef has had a variable impact on Coral Index scores. Over the period 2006-2010 Coral Index scores for the Burdekin region where higher when data from Middle Reef were included (Figure A 17). Scores in 2011 and 2012 were similar irrespective of whether Middle Reef was included, demonstrating the Coral Index score for Middle Reef was close to the regional mean. At the time of last survey in 2013, the Coral Index score for Middle Reef was again above the mean of other reefs in the region (Figure A 17). Since 2013, as the regional Coral Index score recovered the Coral Index scores that include or exclude Middle Reef converged by 2015 and then diverged as the mean Coral Index score for other reefs in the region improved to exceed the carried forward value for Middle Reef (Figure A 17). It is unknown if coral communities at Middle Reef also improved over this period, however, unquantified observations by the Author following localised flooding in the Townsville region and then thermal stress in early 2020 suggest a marked decline in the condition of coral communities at this reef.

Given the length of time since the monitoring sites at this reef have been surveyed it is not possible to infer the impact on recent Coral Index scores of removing Middle Reef from the sampling design. However, excluding carried forward data from 2020 scores does result in a small improvement in the regional Coral Index score (Figure A 17) due mostly to improved scores for the macroalgae indicator (Table A 13).

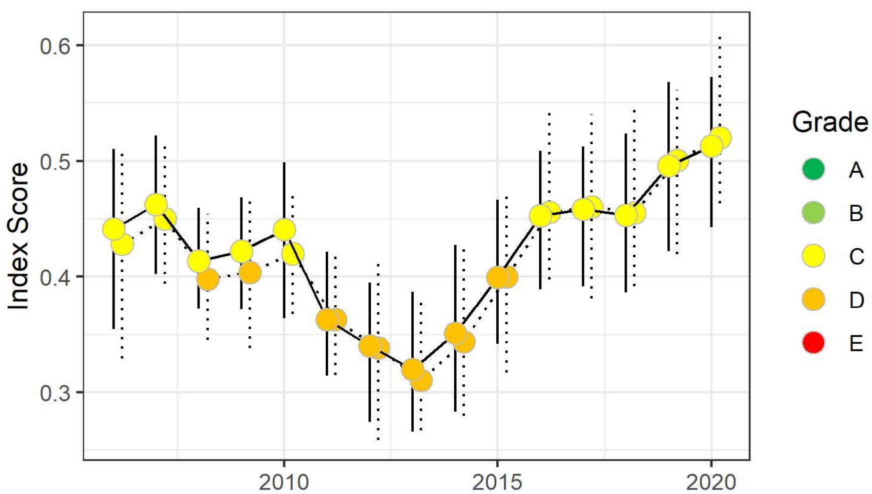


Figure A 17 Influence of sampling design change on Coral Index scores of the Burdekin region. The solid line represents reported Coral Index scores that include data from Middle Reef. Scores along the dotted line exclude data from Middle Reef across the time-series and estimates are offset along the x-axis for clarity.

Table A 13 Influence of sampling design change to indicator scores for Burdekin region. Values are reported scores for 2020 based on inclusion of carried forward 2013 LTMP data for Middle Reef, values in parentheses are the difference in scores with Middle Reef data excluded.

| Depth | Coral cover | Juvenile coral | Macroalgae | Cover change | Composition |
|---|---|---|---|---|---|
| 2 | 0.37 (-0.03) | 0.34 (-0.03) | 0.51 (0.09) | 0.42 (0) | 0.71 (0.04) |

### 9.4.4 Fitzroy region

Peak Island was last surveyed in 2019, not scheduled for survey in 2020 and then excluded from the sampling design of the MMP in 2021. Reported scores for 2020 and 2021 were based on caried forward data from 2019. It is intended that future reporting of the time-series of Coral Index scores exclude data from Peak Island to reflect the ongoing sampling design of the project.

Inclusion of data from Peak Island tends to reduce Coral Index scores except when regional scores are below ~ 0.25 (Figure A 18, solid line). In 2019 exclusion of data from Peak Island increased the regional Coral Index score from 0.34 to 0.36. Of the individual indicators it was only the composition score that did not improve with the exclusion of Peak Island (Table A 14). This result should be taken with some caution as the baseline composition of coral communities at Peak Island was already the result of selective processes and limited to a relatively few tough taxa, especially at 5 m depths.

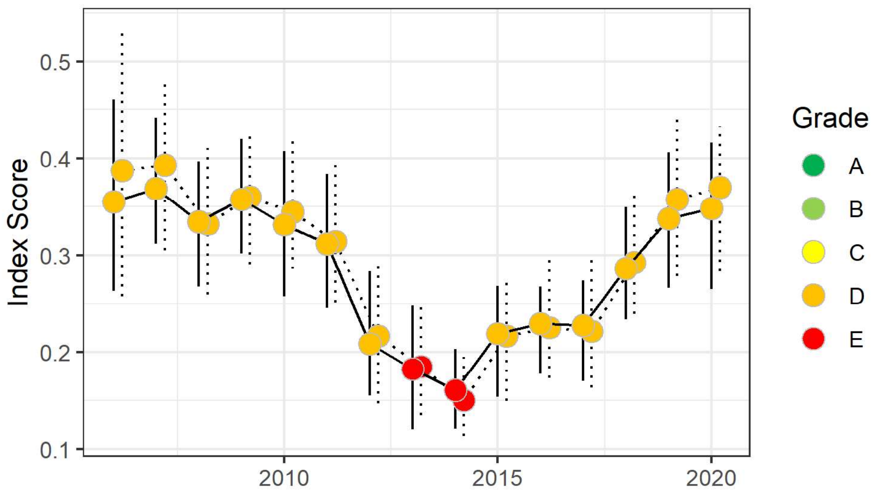


Figure A 18 Influence of sampling design change on Coral Index scores of the Fitzroy region. The solid line represents reported Coral Index scores that include data from Peak Island. Scores along the dotted line exclude data from Peak Island across the time-series and estimates are offset along the x-axis for clarity.

Table A 14 Influence of sampling design change to indicator scores for Fitzroy region. Values are reported scores for 2019 based on inclusion of data for Peak Island, values in parentheses are the difference in scores with Peak Island data excluded.

| Depth | Coral cover | Juvenile coral | Macroalgae | Cover change | Composition |
|---|---|---|---|---|---|
| 2 | 0.34 (0.04) | 0.38 (0.07) | 0.20 (0.04) | 0.47 (0.06) | 0.33 (-0.13) |
| 5 | 0.41 (0.02) | 0.28 (0.01) | 0.09 (0.01) | 0.47 (0.09) | 0.42 (-0.02) |

## 9.5 Conclusion

The substitution of King Reef for Bedarra Island as a monitoring location resulted in an improvement in Coral Index score but no change to the reported grade. An additional benefit of this location is that it more closely aligns with water quality monitoring sites. Future trends in Coral Index scores should continue to include scores for King Reef prior to 2015 and Bedarra Island thereafter.

Since 2013 Burdekin region Coral Index scores have caried forward information from Middle Reef with the expectation that the MMP would update this information once Middle Reef was resurveyed by the LTMP. However, Middle Reef has now been removed from the LTMP sampling design, and as such the 2013 data are unlikely to be updated nor do they represent the current state of this reef. As there remain eight reefs monitored by the LTMP and MMP in this region and the exclusion of Middle Reef has very little influence of Coral Index scores, we suggest removing Middle reef data from future calculation of Coral Index scores. A sensible approach would be to include data from Middle Reef until 2013, as these are valid estimates of coral reefs within the region but remove Middle Reef data from 2014 on. This will not change previously reported Coral Index Grades.

In general removal of Peak Island data form Coral Index scores in the Fitzroy region results in improved Coral Index scores, especially when the benthic communities at more developed coral reefs are in moderate to good condition. There is limited justification for applying the Coral Index scoring system to the benthic communities at Peak Island. We recommend this reef be removed from future reporting of the Coral Index in this region, noting this will more appropriately apply scores to coral reefs and will not change previously reported coral community grades.

# 10 Appendix 2: Publications and presentations 2020–2021

Castro-Sanguino C, Carlos Ortiz J, Thompson A, Wolff HH, Ferrari R, Robson B, Magno-Canto MM, Puotinen M, Fabricius KE, Uthicke S, 2021, Reef state and performance as indicators of cumulative impacts on coral reefs. Ecological Indicators 123: https://doi.org/10.1016/j.ecolind.2020.107335

Thompson, A., Martin, K., Logan, M. 2020, Development of the Coral Index, a summary of coral reef resilience as a guide for management, *Journal of Environmental Management* 271:111038.

State of coral communities in the Mackay Whitsunday Isaac Region 2020, Mackay Whitsunday Paddock to Reef Virtual Integrated Science Forum.

Warne DJ, Crossman KA, Jin W, Mengersen K, Osborne K, Simpson MJ, Thompson AA, Wu P, Ortiz JC 2021, Identification of two-phase recovery for interpretation of coral reef monitoring data. Journal of Applied Ecology. https://doi.org/10.1111/1365-2664.14039

State of inshore coral communities. Presentation to delegation from Pioneer Cane growers 2nd Nov 2020

State of inshore coral communities. Presentation to delegation from Sugar Research Australia 17th March 2021

State of inshore coral communities. Presentation to delegation from Women in Cane