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Annual Report for

INSHORE CORAL REEF MONITORING

2018-19

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Front cover photo: A resilient reef, vibrant and diverse healthy corals thrive despite the presence of crown-of-thorns starfish at Franklands East July 2019. © Australian Institute of Marine Science, Photographer: Paul Costello

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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Glossary

AIMS	Australian Institute of Marine Science
Authority	Great Barrier Reef Marine Park Authority
BOM	Bureau of Meteorology
Chl *a*	Chlorophyll *a*
LTMP	Long-Term Monitoring Program
MMP	Marine Monitoring Program
NAP	Non-algal particulate
Reef 2050 WQIP	Reef 2050 Water Quality Improvement Plan
The Reef	Great Barrier Reef

Acknowledgments

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Executive Summary

This report details the condition of 31 inshore coral reefs monitored under the Great Barrier Reef Marine Park Authority's 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.

Environmental conditions over the 2018–19 summer were relatively benign for inshore corals. There were no severe disturbances associated with tropical cyclones or high seawater temperatures impacting the reefs monitored. A tropical low that developed into cyclone Owen is likely to have caused minor damage at a single reef in the Wet Tropics. Flooding in the Burdekin region and Daintree sub-region caused some mortality of corals.

Inshore corals declined to an overall 'poor' condition in 2019 (Figure 1). The declining scores since 2016 primarily reflect the severe impact of cyclone Debbie in the Mackay–Whitsunday region, while elsewhere scores have remained stable or improved over this period. Over the longer-term, coral cover and macroalgae indicator scores in 2019 were at the lowest levels recorded since the beginning of the Marine Monitoring Program in 2005 (Figure 1). Coral communities are naturally dynamic going through periods of recovery following acute disturbances such as cyclones. Improvement of coral condition scores from a low point reached in 2011 demonstrates the ongoing ability of coral communities to recover. However, extended periods of low scores for the cover change indicator and the overall decline in coral cover suggest a potential mismatch between disturbances and the rate at which coral communities are recovering.

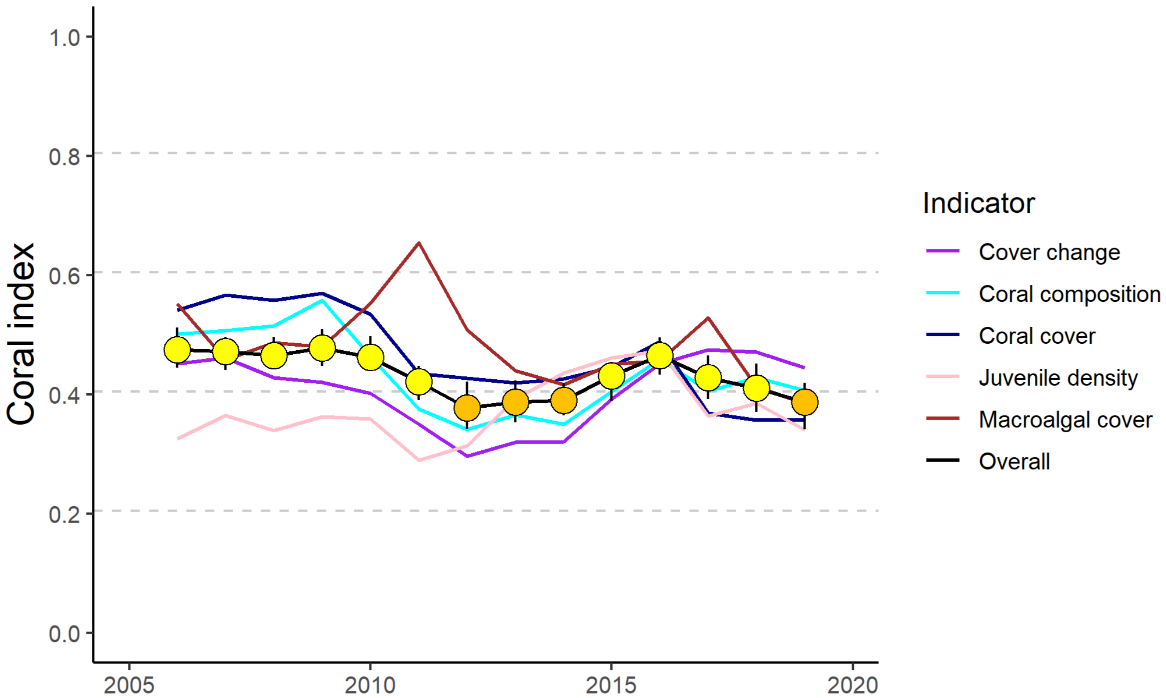

Figure 1 Trends in Coral index and contributing indicator scores for the inshore Reef. Coral index scores are coloured by report card categories: orange = 'poor', yellow='moderate'. Error in index score derived from bootstrapped distribution of regional indicator scores weighted by the relative area of inshore coral reefs in each region.

Coral 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 coral community resilience. 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 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 **coral cover change** as an indicator of the recovery potential of coral communities due to growth
- **community composition** as an indicator of selective pressures imposed by the environmental conditions at a reef.

The coral index score is published in the Great Barrier Reef Report Card and contributes to the marine condition score. Index scores based largely on Marine Monitoring Program data, but also locally relevant data sources, are also published in regional report cards. Regional level coral community condition and trend within regions are summarised below.

Wet Tropics region condition

Inshore coral remains in ‘moderate’ condition. The stable condition observed since 2016 masks differing trends within sub-regions.

- In the Barron Daintree sub-region, coral condition remained ‘moderate’ with some decline. Flooding of the Daintree River, physical damage caused by cyclone Owen and thermal bleaching in 2017 are likely to have contributed to the observed decline. Very high cover of macroalgae at shallow sites of Snapper North continue to influence scores.
- In the Johnstone Russell-Mulgrave sub-region coral condition has fluctuated about the threshold between ‘moderate’ and ‘good’ condition since 2016. Ongoing low scores for the macroalgae and juvenile coral indicators at Franklands West continue to limit condition within the sub-region. Crown-of-thorns starfish were observed at High Island and in the Frankland Group and while coral cover increased, predation of corals by these starfish will have contributed to reduced scores for the change in cover indicator.
- In the Herbert Tully sub-region, coral condition has improved to ‘good’ as coral communities continue to recover from the severe impact of cyclone Yasi in 2011. Recovery has been slowest at Bedarra and Dunk South were macroalgae cover is high.

Burdekin region condition

Inshore coral communities remain in ‘moderate’ condition. Coral communities continue to recover from a low point following the impact of cyclone Yasi in 2011. Thermal stress leading to coral bleaching in 2017 temporarily stalled this recovery.

At the shallower sites, macroalgae indicator scores remain low due to high cover of macroalgae at reefs inshore of the Palm Island group. Low juvenile densities at these shallow sites also continue to limit scores.

Mackay–Whitsunday region condition

Inshore coral condition has declined to ‘poor’. The coral index has declined sharply since 2016 following cyclone Debbie. The seemingly prolonged decline reflects the biennial sampling designs of both the Marine Monitoring Program and Long-Term Monitoring Program – the 2019 results are the first to include post-cyclone observations from all monitored reefs.

Reduction in the coral cover and composition indicators are to be expected following a severe cyclone. The subsequent decline in juvenile density and macroalgae indicators and the further decline in the coral cover change indicator, that was already low, suggest rapid recovery is unlikely.

Fitzroy region condition

Inshore coral condition remains ‘poor’ but has continued to improve from the ‘very poor’ condition observed in 2013 and 2014. Although all indicators have improved over recent years it is only the coral cover change score that has reached ‘moderate’ levels at the regional scale.

The state of reefs varies markedly across the region. Coral cover is highest at the most offshore reef, Barren Island (above 60 per cent at the 5 metre sites). In contrast, coral cover remains ’poor’ or ‘very poor’ closer to the coast. Macroalgae cover remains high at almost all reefs.

Role of water quality on inshore reef resilience

The primary premise of the *Reef 2050 Water Quality Improvement Plan* — that load reduction will have downstream environmental benefits for the Reef — is supported by:

- spatial analyses that demonstrated improved coral index scores along a declining gradient in suspended particulates, and low cover of macroalgae where chlorophyll *a* concentration met guideline values,
- negative relationships between the recovery of coral communities and pressures from catchment run-off. By focusing on periods free from acute disturbance events we demonstrate that incremental changes in the coral index, during periods when reefs were recovering, were inversely related to discharge from local catchments in three of the four regions monitored (the exception being the Mackay-Whitsunday region).

1 Introduction

The proximity of inshore reefs to the coast make them highly accessible; a factor elevating their social, economic and cultural importance beyond of their relatively 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 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 Authority have co-invested to provide inshore coral reef monitoring under the 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 that contributes to the Reef report card. The coral index is designed to capture key aspects of coral community condition and resilience that can be used to both track trends in community condition but also 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 in a cycle of recovery punctuated by acute disturbance events. Common disturbances to inshore reefs include cyclones (often coinciding with flooding), high water temperatures, and outbreaks of crown-of-thorns starfish, all of which can result in widespread mortality of corals (e.g. Sweatman *et al.* 2007, Osborne *et al.* 2011). While, elevated 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), promoting outbreaks of crown-of-thorns starfish (Wooldridge & Brodie 2015) and increasing susceptibility to thermal stress (Wooldridge & Done 2004, Wiedenmann *et al.* 2013). It is the potential for pollutants in run-off to suppress the recovery of coral communities (Schaffelke *et al.* 2013) 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, agrochemicals, 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 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 is included as a key indicator of the endpoint of recruitment processes.

Macroalgae are monitored and reported as they have higher abundance 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) by increased competition for space or 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 of 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).

Hard coral community composition is monitored as the selective pressure of water quality on coral communities is clearly 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, and from the photosynthesis of their symbiotic algae (zooxanthellae). The ability to compensate by feeding, where there is a reduction in energy derived from photosynthesis, e.g. as a result of light attenuation in turbid waters (Bessell-Brown *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 is that reduced energy, or completion for space is likely to reduce the rate corals grow or result in increased rates of partial colony mortality of corals, or increased susceptibility to disease. The rate that coral cover recovers may decline under increase water quality pressure. A derivative of coral cover is an indicator based on expected rate of coral cover increase.

1.2 Purpose of this report

The purpose of this report is to provide the data, analyses, and interpretation underpinning coral index scores included in the 2019 Reef report card. This report covers inshore coral reef monitoring conducted by AIMS as part of the Authority's MMP until August 2019, with inclusion of data from reefs monitored by the AIMS Long-Term Monitoring Program (LTMP) from 2005 to 2019. The coral indicator and index scores reported here are 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 communities 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.

1.3 Structure of the report

In keeping with these objectives listed above, the report is structured to present firstly data relating to key pressures likely to have influenced the condition of coral communities (section 3), followed by the coral community responses to those pressures. Coral community condition summaries (section 4) are reported at the scale of the inshore Reef followed by more detailed exploration of regional patterns and finally responses of communities to water quality. These results are discussed in section 5. Finer scale trends in condition for individual reefs are available on-line at http://apps.aims.gov.au/reef-monitoring/.

2 Methods

This section provides and 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 is summarised in Table 1.

2.1.1 River discharge

Daily records of river discharge 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 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 (Gruber *et al*. 2020, Table A 6).

For each (sub-)region time-series of daily discharge were estimated as the sum of gauged values from gauging stations nearest 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 index scores were considered due to the underlying sampling design of the program. To match this sampling frequency, the mean of the total annual discharge from all rivers discharging into a given region for each two-year period between 2006 and 2019 was calculated.

2.1.2 River Nutrient Loads

Loads of total nitrogen (N) and phosphorous (P) delivered by rivers were sourced from the Great Barrier Reef Catchment Loads Monitoring Program. In short, annual loads are estimated from concentrations measured across the year's hydrological cycle. Time series of loads that span the coral monitoring data are limited to those detailed in (Table A 2).

The load time series from the Great Barrier Reef Catchment Loads Monitoring Program were supplied by the State of Queensland, Department of Environment and Science. Loads for 2016-17 and 2017-18 are available from the © State of Queensland (Department of Environment and Science) 2019.

2.1.3 Sea temperature

To assess variability in temperature stress 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 (Table A 3).

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 lead to coral bleaching in 2016 and 2017. For the Fitzroy Region 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

Two 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 a particular day (i), and Tci is (sub-)region's climatological temperature for that day of the year. Only positive anomalies are summed.

The second, degree heating days (DHD) were 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 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.

The primary difference between these degree heating day estimates are the underlying climatology used and the focus on deviation from the maximum rather than seasonal temperature profile.

2.1.5 Cyclone tracks

Cyclone tracks and intensity are downloaded from the Australian Bureau of Meteorology at http://www.bom.gov.au/cyclone/history/index.shtml. These tracks are primarily used as a double check of 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) concentration, a proxy for total suspended sediments, derived from the MODIS aqua satellite mounted sensor were downloaded from the Australian Bureau of Meteorology[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 temporal covariate these monthly means were aggregated to annual estimates, and for spatial analysis of current indicator scores aggregated to reef level means over the last five years (2015 – 2019).

Relative exposure to Chl *a* at each reef in each year, was estimated based on the methods developed by the water quality component of the MMP (Gruber *et al*. 2020, Petus *et al*. 2016). In

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[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).

[2] [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 are acquired from NASA spacecraft. http://www.bom.gov.au/marinewaterquality/. Although the confidence in individual estimates of Chl a in turbid inshore waters is low the time averaged conditions do describe gradient that correspond to differences in benthic communities.*

brief, MODIS aqua images were used to classify waters into one of six colour classes that range from those typical of primary (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 is recorded as the exposure of that pixel in each week.

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 wind driven resuspension, produce waters with similar spectral signatures.

Matching *in situ* sampling with the classified colour of the water at the date and location of the sample provided estimates of the mean concentration of water quality parameters for each colour class. The Chl *a* estimates in this report are expressed as the mean exposure to Chl *a* concentrations above wet-season guideline values (0.63 ugL^{-1}, GBRMPA 2010) over the wet-season (December – April, inclusive) preceding annual coral surveys. Estimates were derived from the same nine pixels as described above for estimation of NAP concentration.

As a background to regional maps of sampling locations, mean Chl *a* and NAP concentrations over the period (2003–2018) were derived for all inshore waters using the same methods as described above and scaled to visually demonstrate concentrations relative to Guideline values.

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

	Data range	Method	Usage	Data source
Climate				
Riverine discharge	1980 – 2019	water gauging stations closest to river mouth, adjusted for ungauged area of catchment	regional discharge plots and table, covariate in analysis of temporal change in Coral index	DNRME, adjustment as tabulated by (Gruber *et al. 2020.*)
Riverine Total N and Total P loads	2006 – 2018		covariate in analysis of temporal change in Coral index	Data sourced from the State of Queensland, Great Barrier Reef Catchment Loads Monitoring Program
Sea temperature	2005 – 2019	*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 – 2019	remote sensing, ~4 km^2 pixels adjacent to coral sites	regional plots, thermal bleaching disturbance categorisation	Bureau of Meteorology
Cyclone tracks	2005-2019		informing attribution of storms as cause of observed coral loss	Bureau of Meteorology
Environment at coral sites				
Chlorophyll *a* exposure	2003 – 2019	product of water colour classification derived from remote sensing and coupled niskin samples, resolution ~1 km^2	mapping, covariate in analysis of temporal variability in index score changes	MMP Water Quality (Gruber *et al. 2020.*)
Non-algal particulate (NAP) Chlorophyll *a*	2002 – 2019	remote sensing adjacent to coral sites, resolution ~1 km^2	Macroalgae and Community Composition metric thresholds, mapping, covariate in analysis of spatial trends in index and indicator score, covariate in analysis of temporal variability in index score changes	Bureau of Meteorology
Photosynthetically Active Radiation (PAR)	2005 – 2019	remote sensing adjacent to coral sites, resolution ~1 km^2	covariate in analysis of spatial trends in index and indicator score, covariate in analysis of temporal variability in index score changes	Marites Magno-Canto (AIMS)
Sediment grain size	2006 – 2017	optical and sieve analysis of samples from coral sites	analysis covariate, Macroalgae metric thresholds	MMP Inshore Coral monitoring

2.1.7 Light available for photosynthesis

The estimates of Chl *a* and NAP described above quantify the relative exposure to nutrient 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 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.

2.1.8 Sediment characteristics

The proportion of sediments with grainsize < 63µm (clay and silt) from the reefs sites was used as a proxy for exposure to wave and tide mediated resuspension. These estimates were used as covariates in analysis of spatial distributions of index and indicator scores and in analyses that determined reef level thresholds for macroalgae (Thompson *et al.* 2016).

Grainsize distribution of sediments was estimated from samples collected from the 5 m depth MMP sites at the time of coral sampling until 2014. At each site five 60 ml syringe tubes were used to collect cores of surface sediment from available deposits along the site. The end of the syringe tube was cut away to produce a uniform cylinder. Sediment was collected by pushing the tube into the sediment being careful not to suck sediment and pore-water into the tube with the plunger. A rubber stopper was then inserted to trap the sediment plug. The surface centimetre of sediment was retained and grainsize distribution determined by a combination of sieving and laser analysis carried out by the School of Earth Sciences, James Cook University (2005–2009) and subsequently by Geoscience Australia.

For LTMP sites the clay and silt content of sediments was estimated by interpolating between MMP reefs with similar exposure to the south-east as the predominant direction of wave energy in the Reef. Estimated sediment composition was verified by visually checking images including sediment from photo transects against images from MMP reefs with similar exposure.

For the new site at Bedarra sediment samples collected in 2015 were passed through a 63 µm sieve to estimate the clay and silt grain-sized proportion of the sample.

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 inshore coral reef communities occurs at reefs adjacent to four of the six natural resource management regions with catchments draining into the Reef: Wet Tropics, Burdekin, Mackay-Whitsunday and Fitzroy. 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.

Sub-regions were included in the Wet Tropics Region to more closely align reefs with the combined catchments of the: Barron and Daintree rivers, the Johnstone and Russell-Mulgrave Rivers, and the Herbert and Tully rivers.

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 distance from the coast and increasing distance from rivers, particularly in a northerly direction (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 facilitate the teasing out of water quality associated impacts.
2. There was either an existing coral reef 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. Coral reef 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, e.g. sediments, fresh water, nutrients or toxins imported by flood events, 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 frequency of sampling by the MMP has varied between annual and biennial, LTMP samples on a biennial basis. Sampling undertaken at each reef is detailed in Table 2 and the location of those reefs presented in Figure 2. In 2019 the LTMP sampled three inshore reefs in both the Wet Tropics and Mackay-Whitsunday regions and two of three reefs in the Burdekin region. The number of reefs sampled by the MMP in 2019 were: Wet Tropics (9 of 13), Burdekin (4 of 6), Mackay-Whitsunday (4 of 7) and Fitzroy (4 of 6).

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. At each reef surveys of juvenile coral densities, benthic cover estimates derived from photo point intercept transects and scuba searches for incidence of coral mortality are undertaken. 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.

(Sub-) Region	Reef	Program	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
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	●	●		●		●	+	●		●	+		●		●

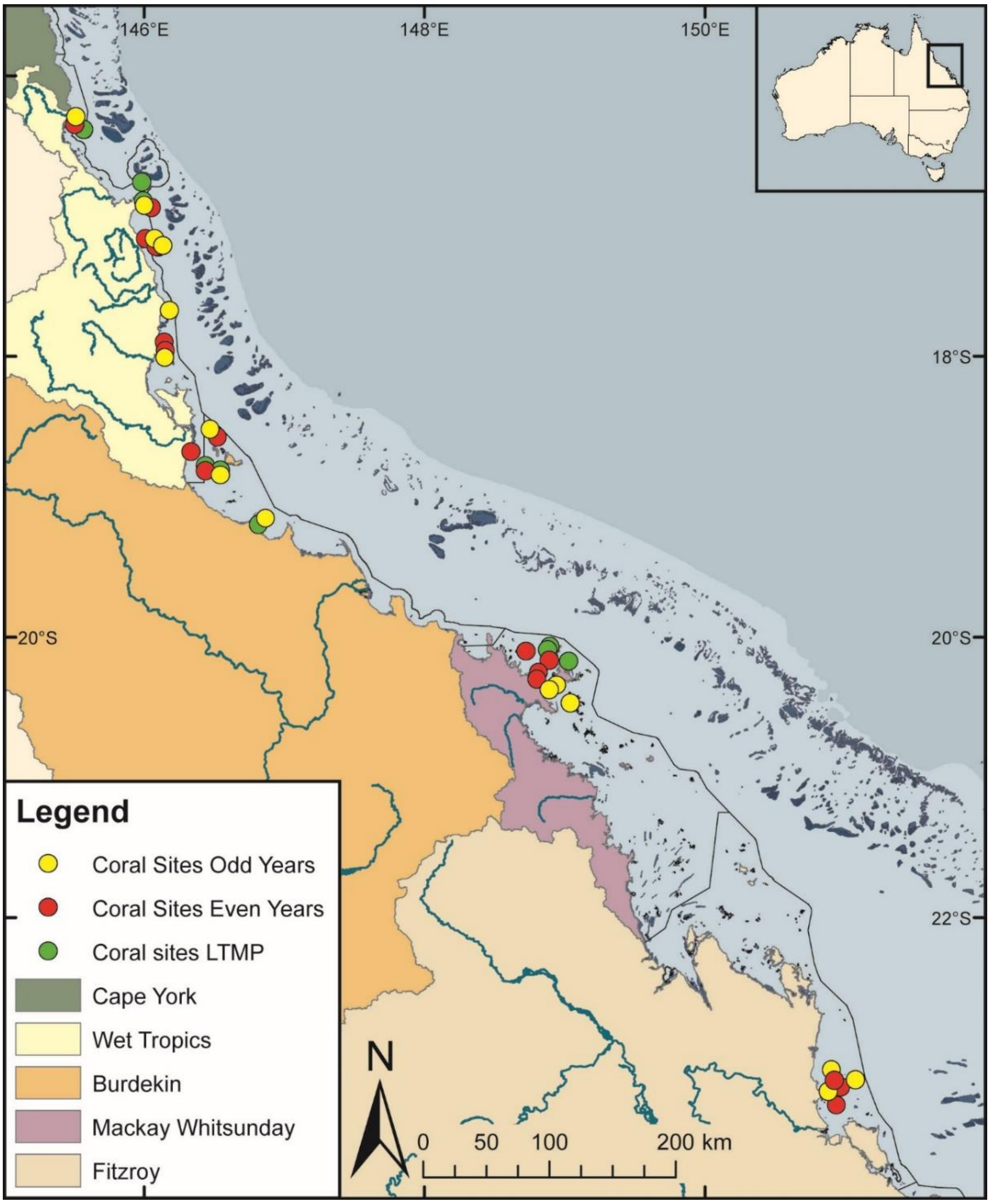

Figure 2 Sampling locations of the coral monitoring. Natural resource management region boundaries are represented by coloured catchment areas.

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 selected 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 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 surveys were 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 survey has changed gradually over time 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 to gain the best estimate of the impact of the acute event and book-end the start of the recovery period. Further funding reductions necessitated the adoption of a biennial sampling cycle for all reefs, although a contingency for the out-of-phase resampling of reefs impacted by acute disturbance was maintained.

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 covers of the substratum of major benthic habitat components.	Approximately 34cm belt along upslope side of transect sampled at 50 cm intervals from which 32 frames are sampled.	Approximately 34cm belt along upslope side of transect sampled at 1m intervals from which 40 frames are sampled.
Demography	Size structure and density of juvenile coral communities.	34cm belt along the upslope side of transect. Size classes: 0–2 cm, 2–5 cm, 5–10 cm.	34cm belt along the upslope side of the first 5 m of transect. Size class: 0–5 cm.

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 followed closely 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 cover of benthic community components 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 at least 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 surveys

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 settlement and growth through to 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. Importantly, this method aims to record only those small colonies assessed as juveniles, i.e. which result 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 recorded.

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 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 follows closely 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 were 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 were also recorded.

Finally, an 11-point scale was used to record the proportions of the coral community 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 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 report card scores

Coral community condition is summarised as an index score that aggregates scores for five indicators of reef ecosystem state. The resulting coral index score provides the coral component of the Reef report card. 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 communities represent different processes that contribute to coral community resilience that are potentially influenced 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 at which coral cover increases as an indicator of the recovery potential of coral communities due to growth, and
- 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 metric

High coral cover is a highly desirable state for coral reefs both in providing essential ecological goods and services related to habitat complexity but also 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 (1992-2014), MMP data (2005-2014) and surveys from Cape Flattery to the Keppel’s by Sea Research prior to 1998 (Ayling 1997) which identified a mean of >50% for combined coral cover on 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 is considered to capture 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 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).

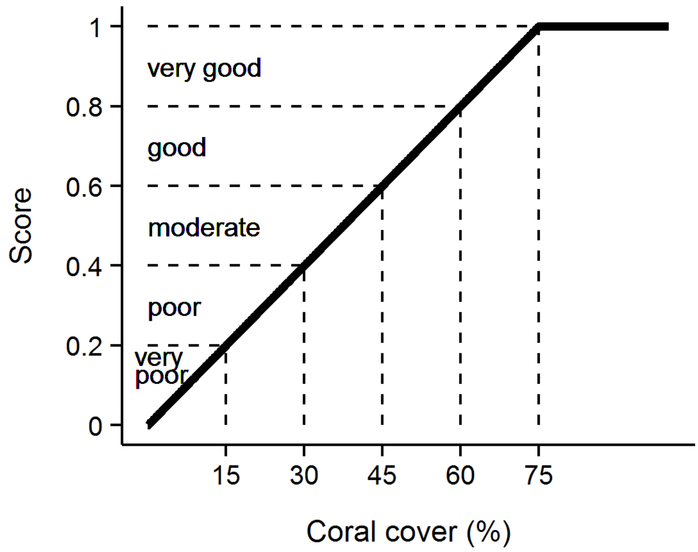

Figure 3 Scoring diagram for the coral cover metric. Numeric scores and associated condition classifications based on observed coral cover are presented.

2.4.2 Macroalgae 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 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 4). The use of separate thresholds ensures 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 reflect 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 density, 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 Macroalgae metric 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).

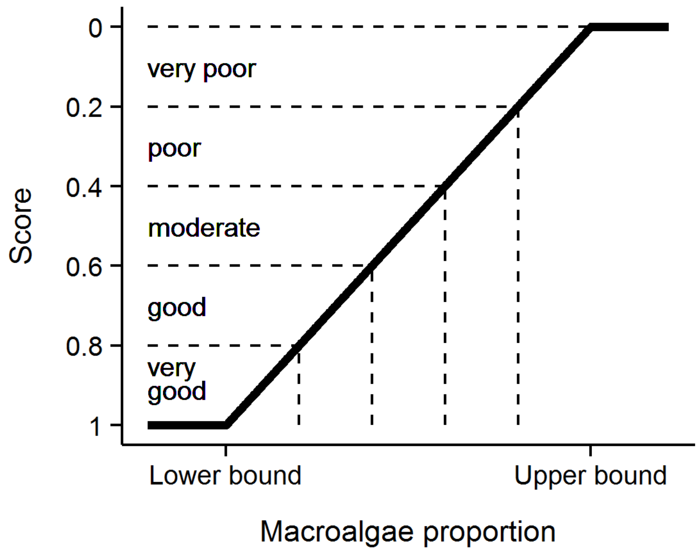

Figure 4 Scoring diagram for the Macroalgae metric. Upper and lower threshold values are reef and depth specific. Numeric scores and associated condition classifications are presented. Note that for this metric the y-axis is inverted as high values reflect poor condition.

2.4.3 Density of juvenile hard corals metric

For coral communities to recover rapidly from disturbance events requires 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 through to 0.4 at a density of 4.6 colonies per m^2 then linearly through to a score of 1 when the density was 13 colonies per m^2 or above (Figure 5).

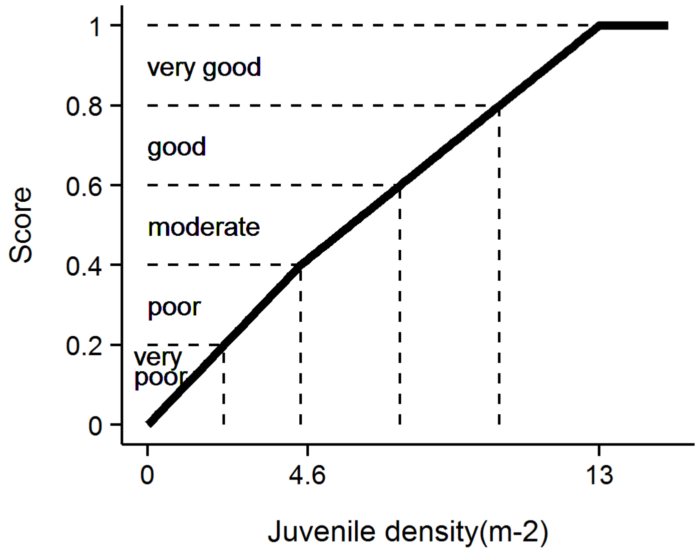

Figure 5 Scoring diagram for the Juvenile metric. Numeric scores and associated condition classifications are presented.

2.4.4 Cover change 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 that coral cover increases and indicate a lack of resilience. The change in coral cover indicator score is derived from the comparison of the observed change in coral cover between two visits and the change in cover predicted by Gompertz growth equations parameterised from time-series of coral cover available on inshore reefs up 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 inability to gain precise estimates of cover increase when cover is low — as it is for most taxa at most reefs.

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

As model predictions relate to annual changes in 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 warm up 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 Coral predicted cover were combined into posterior predictions of total coral cover from which the mean, median and 95% Highest Probability Density (HPD) intervals were calculated.

As changes in 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 metric the following process was applied (Figure 6):

- If coral cover declined between surveys, a score of 0 was applied.
- If cover change was between 0 and the lower HPD interval of predicted total 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 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 cover change was greater than the upper HPD interval of predicted change and less than double the upper interval, scores where 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.

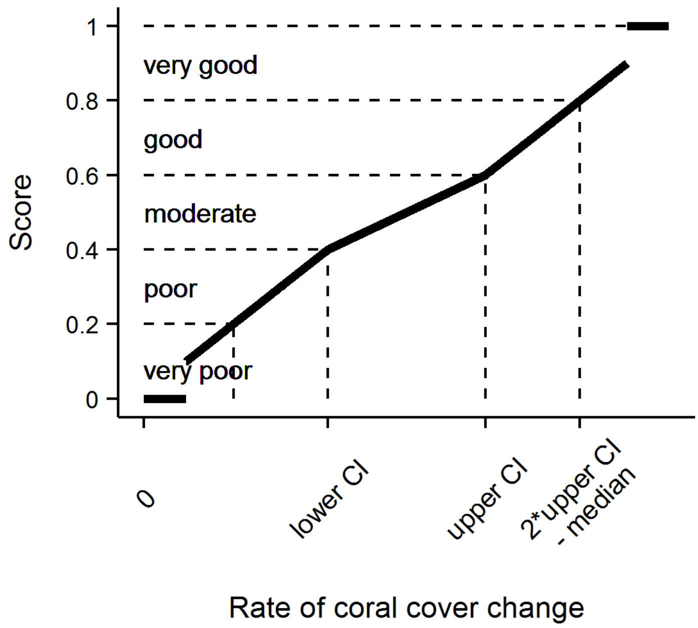

Figure 6 Scoring diagram for Cover Change metric.

2.4.5 Community composition metric

The coral communities monitored by the MMP vary considerably in the relative composition of coral species (Uthicke *et al*. 2010, Thompson *et al.* 2014b). 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, as a result 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 principle 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.* 2014b) 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 6).

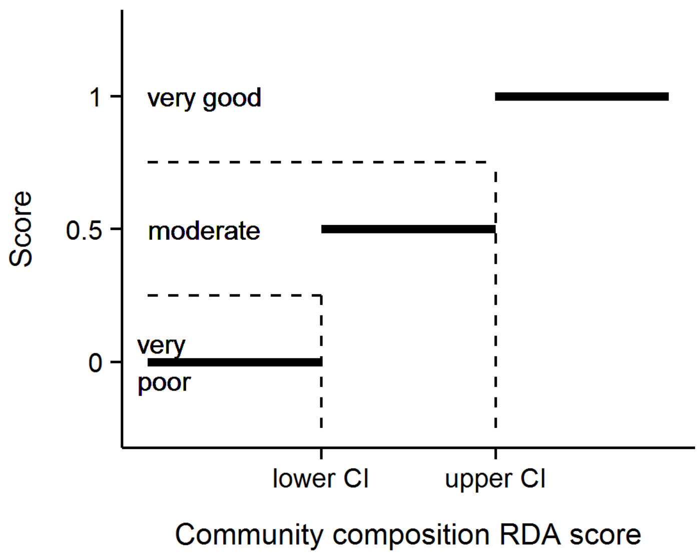

Figure 7 Scoring diagram for community composition metric

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 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 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 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 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 community 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
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
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 m2 of available substrate
	Continuous between 0.4 and 1	4.6 to 13 juveniles per m2 of available substrate
	Continuous between 0 and 0.4	0 to 4.6 juveniles per m2 of available substrate
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
4.1	Temporal trend in coral condition	Reef	Major disturbances	Relative influence of major pressures over the time-series
4.3	Trends in Coral index and individual indicators	(Sub-)regional		Generalised linear mixed models; pairwise comparisons
4.7.1	Coral index and indicator scores in 2019	Reef and Regional	Chl *a*, Non-algal particulate concentration	Generalised linear mixed models, predicted responses
4.7.2	Temporal variability in Coral index in relation to water quality	Regional	Regional riverine: discharge, Total N and Total P loads. Chl *a* exposure, NAP concentration, PAR	Generalised additive models, predicted responses
Appendix 1: Additional Information	Trends in benthic community composition.	Reef/Depth		Plots
	Summaries of 2019 observations	Reef/Depth		Observed values

2.5.1 Variation in index and indicator scores to gradients in water quality

The relationships between the most recent index or indicator scores, at each depth, to location of reefs along water quality gradients were explored via generalised linear mixed models. Each combination of index or indicator score, and depth, were fit separately to two water quality proxies; mean Chl *a* and mean NAP concentration. Identification of 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 algae 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. Macroalgae proportion was also fit using a beta distribution and a gaussian distribution was used for genus composition values.

Where relationships between 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 describe above. The results of these models are plotted and confidence intervals for slopes within each region estimated to identify the regions contribution most to the general Reef-wide trends. Generalised linear mixed models were fit via the mgcv package (Wood 2019) while the probit model for 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 between index and indicator scores and temporal variability in environmental conditions

The response of coral communities to variation in environmental conditions was assessed by comparing changes in index scores to:

- annual discharge and total N and P loads estimated from the adjacent catchments
- exposure to above Guideline concentrations of Chl *a* over the wet season
- NAP concentrations.

For these analyses Generalised Additive Models (GAMs) were applied separately to results from each Region. The response variable was the biennial change in the 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 covariate in each model were summed over the two water years ending in the survey year (y+2). To reduce confounding between the response of the index to acute disturbances, observations of change in the 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 condition 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.

For each of the five indicators that inform the coral index, temporal trends and their 95% confidence intervals 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. To account for the sampling design that samples reefs on a biennial cycle, missing data were infilled with observations from the preceding year as is done for the estimation of annual index scores.

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

A more detailed summary of raw data for benthic cover and juvenile density at each reef and depth combination is presented as bar plots in Appendix (Figure A 1 to Figure A 6). These additional plots breakdown 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 10 to Table A 12.

2.5.4 Analysis of change in index and indicator scores

Differences in the index, or individual indicator scores were estimated between focal years identified as local maxima or minima within the time-series of the index scores within each (sub-)region. Confidence in the magnitude of these differences are 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 within each (sub-)region is presented as a bar plot of annual hard coral cover loss. The height of the bar represents the mean 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 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). The proportion of coral cover lost per region for each disturbance type is subsequently calculated as:

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

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

For reference among (sub-)regions the y axis of each plot was scaled to the maximum mean hard coral cover loss observed across regions in a single year (22% loss of coral cover within the Tully region in 2011).

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 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 2018-19

The condition of coral communities is impacted by a range of environmental pressures. Interpreting the impact of pressures associated with water quality relies on first understanding the impacts of confounding 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 section 4.3.

3.1 Cyclones

Tropical cyclones frequently cross the inshore Reef.

Over the 2018-19 reporting period cyclone activity was focused on northern areas of the Reef (Figure 8). It is highly likely that cyclone Trevor will have impacted reefs proximal to its track. For the remainder of the Reef the only storm of any significance was the tropical low that contributed localised loss of cover at Snapper Island before forming into cyclone Owen (Figure 8). No impact of the remnant low associated with cyclone Penny was recorded as this system was relatively week when crossing the inshore Reef in the Mackay-Whitsunday region and shared a similar path to cyclone Debbie that had already severely damaged reefs in the region.

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 subregion have yet to regain the coral cover lost to the combined impacts of these cyclones (Figure A 3). In general, clear recovery from these cyclones is evident.
- cyclone Debbie (2017) caused severe coral loss on reefs in the Mackay-Whitsunday Region (Figure 8, Table A 7,). There is 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 7, see also ((sub-)regional summaries section 4.3).

3.2 Sea temperature

Temperatures over the 2017-18 and 2018-19 summers were not warm enough to cause severe coral bleaching and mortality at monitored sites. Maximum degree heating day estimates reported by the Bureau of Meteorology, adjacent to MMP monitoring sites, were in the Fitzroy Region (mean 78 degree heating days) in 2018 and Burdekin Region (mean 52 degree heating days) in 2019 (Figure 9).

In contrast, the summer of 2016-17 was the hottest recorded over the MMP time series for the Wet Tropics, Burdekin and Whitsunday Regions where mean degree heating day estimates reported by the Bureau of meteorology ranged between 101 in the Barron Daintree sub-region and 132 in the Burdekin Region (Figure 9). Coral cover losses attributed to bleaching in both 2017 and 2018 within this report were the result of this event. Results from 2019 (Section 4) suggest recovery from this event is occurring, with the minor losses of coral cover observed largely regained.

Loss of coral cover attributed to coral bleaching in response to high water temperatures occurred in the Fitzroy region in 2006. This event was not recorded in the Bureau of Meteorology time-series, though was clear in temperature logger records (Figure 30). Coral cover losses attributed to this event were most severe at North Keppel and Middle Islands with recovery slow and confounded by subsequent pressures (Figure A 6, section 4.).

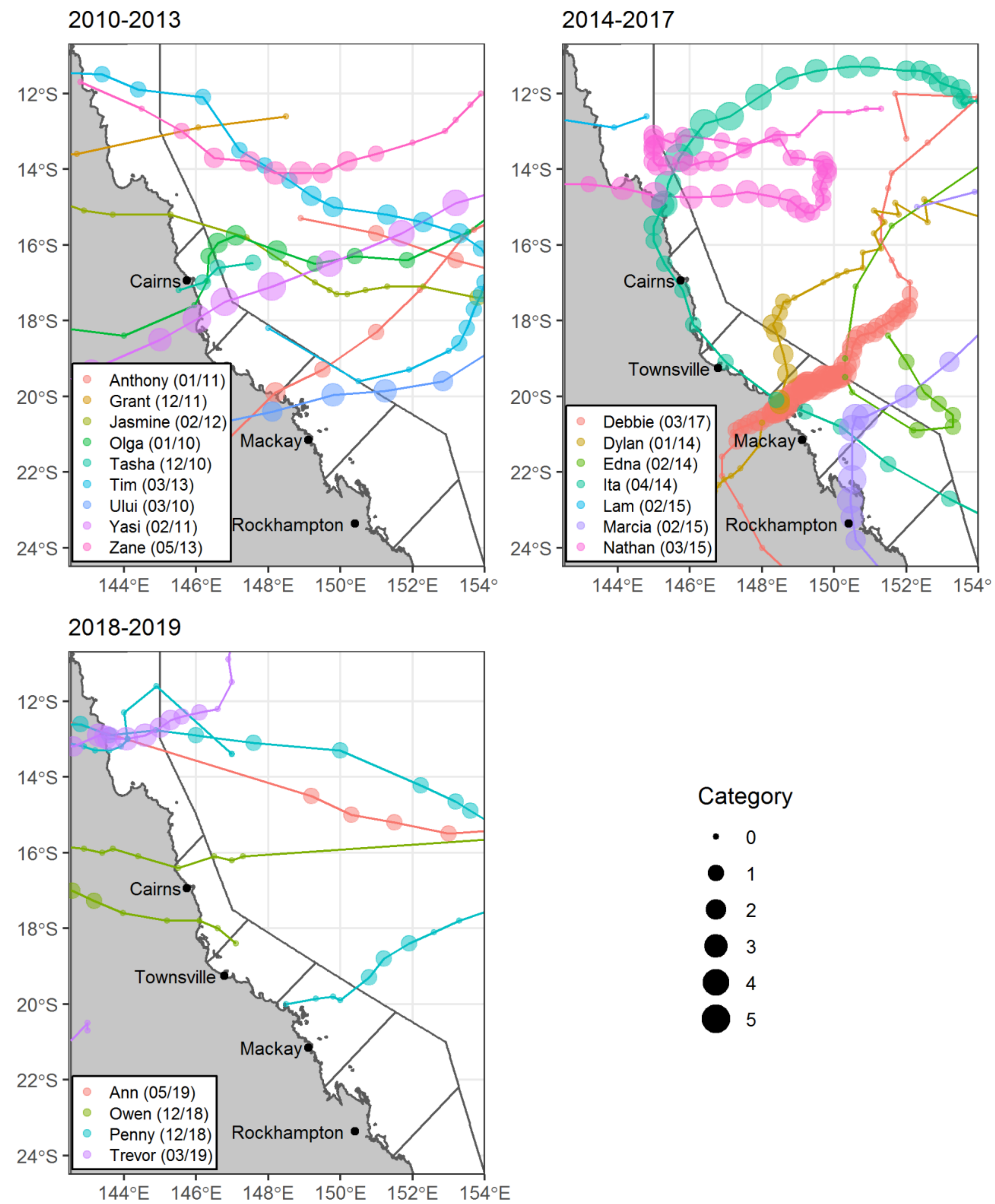

Figure 8 Cyclone tracks for systems crossing the inshore Reef over the last decade. Tracks sourced from the Bureau of Meteorology.

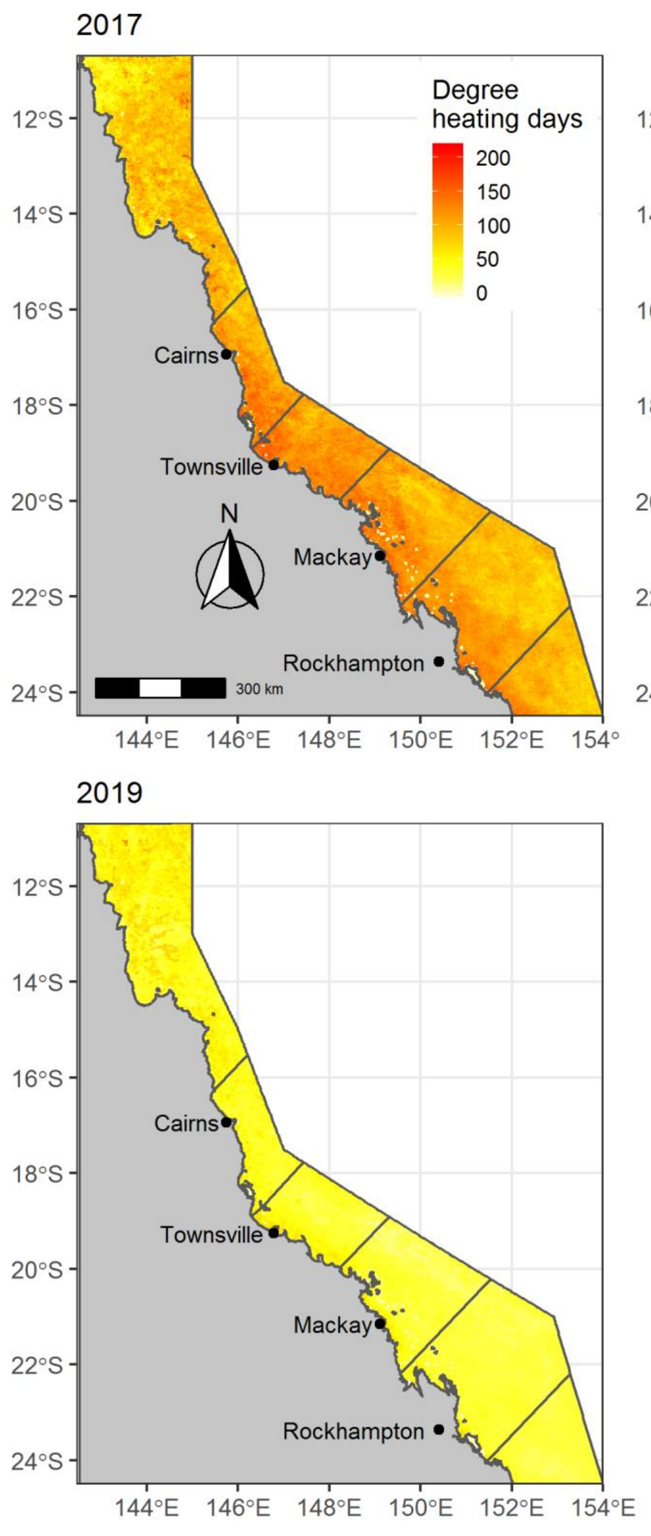

Figure 9 Annual degree heating day estimates for the Reef. Data are the annual Degree heating day accumulations over the summer period (1-December to 31 March) for ~4 km^2 pixels. Maps were sourced from Bureau of Meteorology and include estimates based on 14 Day IMOS climatology.

3.3 Crown-of-thorns starfish

Elevated populations of COTS on the inshore reefs have been limited to reefs in the Barron-Daintree and Johnstone Russell-Mulgrave sub-regions where they caused 37.5% and 17.5%, respectively, of coral cover losses since 2005 (Figure 15, Figure 18). These losses occurred over the period 2012 to 2015. The potential impact of COTS on these reefs was almost certainly mitigated by the removal of starfish under the Authority's crown-of-thorns control program (Table 7).

Since 2015 COTS have remained present on reefs in the Johnstone Russell-Mulgrave sub-region, although numbers have continued to decline (Table 7, Figure A 8). The impact of these on coral cover was limited by the small size of these starfish (Table 8). In the smaller size classes, COTS tend to feed in the understory of the coral community, and this, along with the relatively small size of their feeding scars, limits the loss of coral observed using photo transects. This feeding will however have put downward pressure on the cover change indicator scores at these reefs.

No control was undertaken on the Johnstone Russell-Mulgrave reefs in 2018-19. MMP surveys noted COTS at Fitzroy Island, High Island and the Frankland Group in 2019 (Figure A 8). The individuals observed at both High Island and in the Frankland Group ranged across all size classes, at Fitzroy Island individuals were in the lower two size classes. These results suggest the ongoing recruitment of COTS onto these reefs and potential for future damage.

In 2019 a single large COTS was observed during MMP surveys of Palms West in the Burdekin Region, with feeding scars also observed at Palms East. Sighting data supplied by the Authority also indicate low numbers of COTS at Hayman Island, Hook Island and Border Island in the Mackay-Whitsunday region.

In contrast population outbreaks of COTS were recorded on mid shelf reefs between Innisfail and Townsville, as well as off Princess Charlotte Bay in the North and Mackay to Rockhampton in the south during 2018 and 2019(AIMS LTMP).

Table 7 Number of crown-of-thorns removed by the Authority's crown-of-thorns control program. Figure in bold are the number of individuals removed. The catch rate per diver hour is given in bracket to provide an idea of relative population density.

Reef	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**
Fitzroy Island	**961** (8.6)	**2761** (11.9)	**793** (6.8)	**175** (1.6)	**385** (5.7)	**71** (0.6)	**3** (0.04)
Green Island	**2838** (17.6)	**2889** (10.2)	**758** (4.9)	**352** (3.2)	**579** (4.9)	**170** (2.1)	**2** (0.04)
Low Isles		**405** (23.7)	**296** (2.3)	**144** (2.2)	**1** (0.1)		
Frankland Group						**770** (5.6)	**73** (0.8)
Snapper Island		**135** (16.2)					

Table 8 Size class distribution of crown-of-thorns removed as listed in (Table 7). Figures represent percent of individuals that where within each size class across all Wet Tropics inshore reefs.

Year	pC1 0-15 cm	pC2 15-25 cm	pC3 25-40 cm	pC4 >40 cm
2012	25	27	34	14
2013	26	45	24	5
2014	39	36	19	6
2015	69	27	3	>1
2016	88	11	>1	0
2017	67	30	3	0
2018	55	44	1	0

3.4 River discharge

Discharge from the catchments adjacent to the Reef has the potential to impact coral communities either by exposing corals to lethally low salinity in flood plumes or via the increased loads of sediments and nutrients delivered to inshore waters. The likelihood of negative impacts associated with runoff is logically related to the volume of freshwater and contaminants delivered both of which vary with river discharge.

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 10).

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 two 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, as 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 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 where 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 year the most extensive 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 7, Figure A 6). As at 2019 recovery from this event was occurring at Keppels South but limited at best at Peak and Pelican Islands.

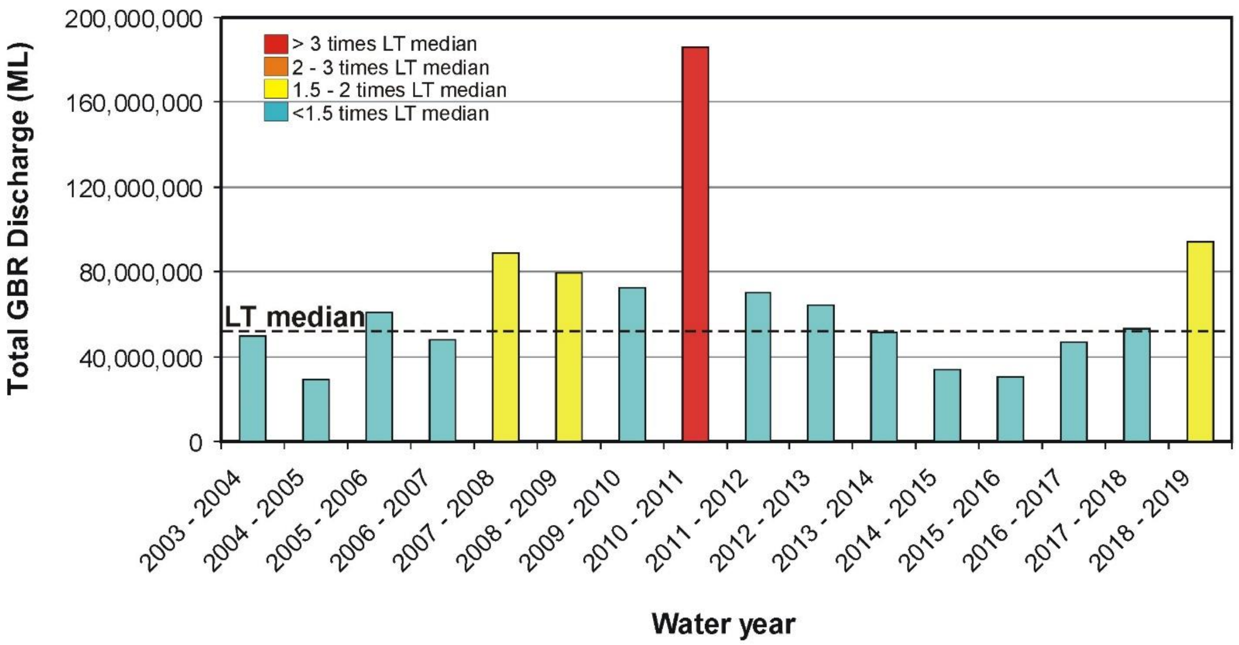

Figure 10 Annual total river discharge to the Reef Annual estimates aggregate over the water year: 1 October to 30 September, for the 35 main Reef basins. Values are colour coded relative to proportion of long-term (LT) median (1986-2016) discharge. Figure source: Gruber et al. 2020, data source: DNRM, http://watermonitoring.dnrm.qld.gov.au/host.htm

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 4.7.14.7.

4 Coral condition and trend

Results are presented in the following sequence:

- Reef level trend in coral condition and indicator scores (4.1)
- Reef level attribution of pressures causing coral cover losses (4.2)
- (sub-)Regional level trend in coral condition and indicator scores (4.3 - 4.6)
- relationship between the condition of coral communities in 2019 and water quality gradients (4.7.1)
- the influence of river discharge and reef-level water quality on change in community condition (4.7.2)

The above results highlight that pressures and current coral 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 on line at http://apps.aims.gov.au/reef-monitoring/.

4.1 Reef-wide coral condition and trend

At the Reef scale, the coral index score declined into the "poor' category having declined consistently from the 'moderate' condition recorded in 2016 (Figure 11). Precipitating this decline was the combination of high sea temperatures causing coral bleaching and the severe impact of cyclone Debbie (Figure 9, Figure 8). That the index has continued to decline is due in part to the biennial sampling design of both the MMP and LTMP that sees lagged recording of these impacts at some reefs, as discussed in the previous section.

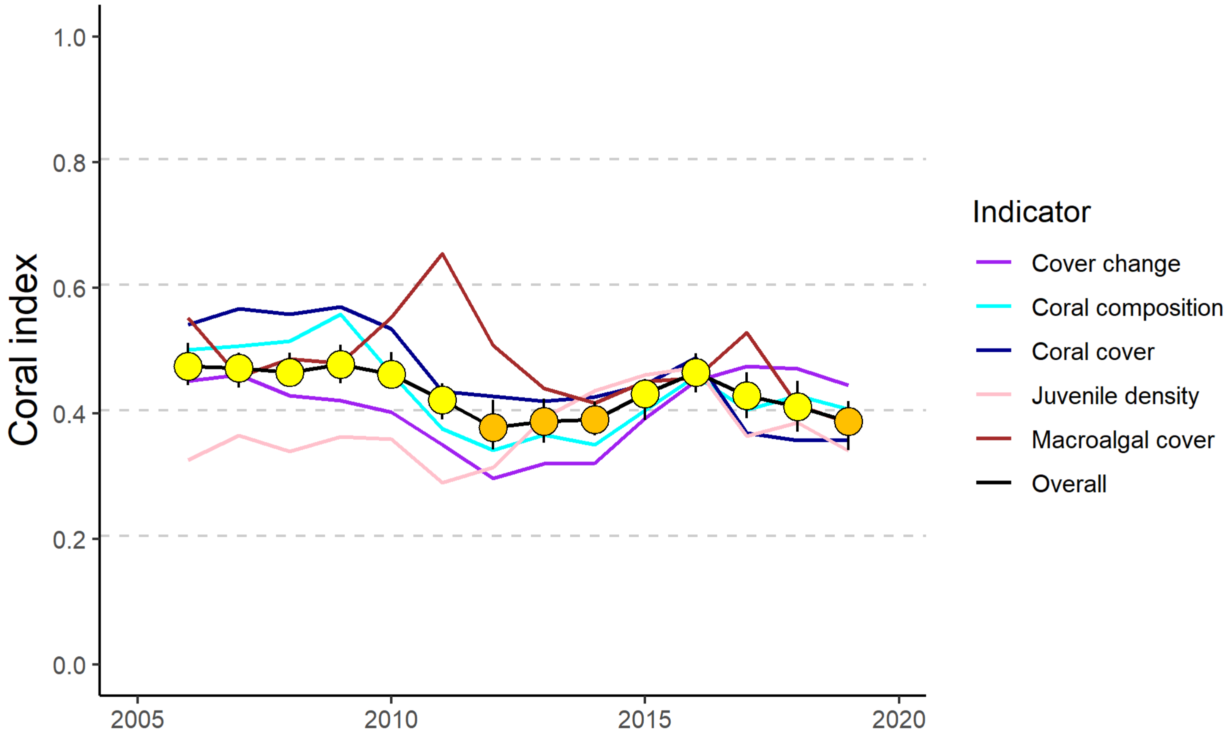

Figure 11 Reef level trend in coral index and indicator scores. Coral index scores are coloured by report card categories: orange = 'poor', yellow='moderate'. Error in index score derived from bootstrapped distribution of regional indicator scores weighted by the relative area of inshore coral reefs in each region.

A lagged response in coral index scores following cyclones, and floods, can also occur due to the stripping of macroalgae cover. Peaks in macroalgae indicator scores observed in 2011 (Figure 11) reflect reduced cover of macroalgae in surveys following cyclones Yasi and, in the Fitzroy region, flooding, followed by their rapid recolonisation. The smaller peak observed in 2017 was partially influenced by a similar process occurring following cyclone Debbie.

The recovery of coral communities between 2013 and 2016 demonstrate the recent resilience of inshore coral communities. That the current condition is again low is unsurprising given the level of pressure imposed in 2017. Of concern is that coral cover had not recovered to levels observed prior to the similar level of pressure imposed in 2011 (Figure 11). A positive indication for future recovery is that the cover change, juvenile density and community composition indicator scores all remain above those observed in 2012 from which point recovery progressed. Conversely, macroalgae indicator scores are lower suggesting a downward pressure on recovery.

Ultimately, the Reef level coral community condition averages over 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 inshore reefs have been exposed to a barrage of disturbance events, from which recovery has been, on average below expected rates.

4.2 Relative impact of disturbances

The most directly observable impact of acute disturbance events is the loss of coral cover they cause. Over the period of the MMP cyclones and storms have caused almost half (48%) of all coral cover losses on inshore reefs since 2005 (Figure 12). 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. Changes in the community composition indicator scores (Figure 11) following acute disturbances indicate that it is species sensitive to poor water quality (primarily *Acropora,* Table A 5) that are disproportionately impacted by these events.

Of note when interpreting Figure 12 is that the biennial sampling design of both the MMP and LTMP results in some lagged attribution of coral loss. For example, loss of coral cover attributed to cyclone Debbie at three reefs monitored by the LTMP in the Mackay-Whitsunday region impacted attributed to cyclone Debbie were not quantified until 2019. Similarly, while the MMP sampling design includes capacity for contingency sampling in the event of acute disturbances so as to limit this lagged assessment, cyclone and bleaching impacts recorded in 2018 (Figure 12) occurred during 2017.

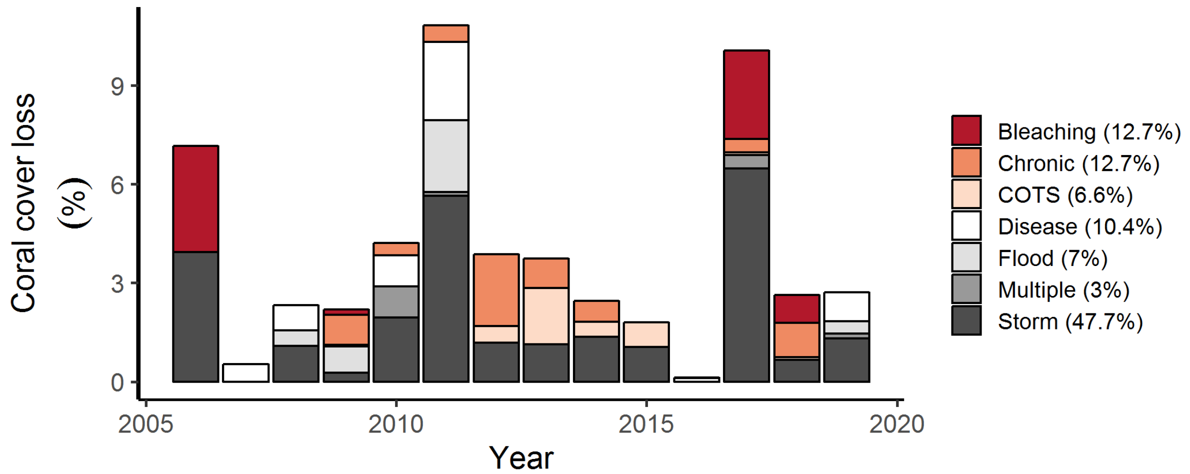

Figure 12 Hard coral cover loss by disturbance type across the inshore Reef. Length of bars represents the mean loss of cover across all reefs in each year. Colours represent the identified cause of cover loss.

Thermal bleaching events have been less frequent although locally severe and caused 13% of coral cover loss (Figure 12). While crown-of-thorns starfish have caused moderate losses (7%, (Figure 12), their effect has likely been limited by active removal to control populations. These figures contrast 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 due to direct exposure to low salinity flood waters has been limited to two metre depths on reefs most proximal to rivers during major flood events. This is unsurprising as more frequent exposure would be expected to precluded reef development. Indeed, the reefs most impacted, Peak Island and Pelican Island in the Fitzroy region, demonstrate minimal development of a carbonate substrate, questioning their appropriate inclusion as coral reef monitoring locations.

In combination, these acute disturbance events contribute strongly to the declines in the coral cover (Lam *et al.* 2018) and index scores in all regions.

The losses of coral cover attributed to disease, and chronic (23%, Figure 12) are more likely to reflect the impacts of water quality. However, by categorising response to primary disturbance-type ignores potential compounding of impacts associated with acute disturbances as elevated levels of nutrients 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), promote outbreaks of crown-of-thorns starfish (Wooldridge & Brodie 2015) and increase susceptibility to thermal stress (Wooldridge & Done 2004, Wiedenmann *et al.* 2013).

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 outbreak dynamics remains unresolved (Pratchett *et al.* 2017) and will not be dwelt on here.

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

4.3.1 Wet Tropics region

Inshore coral remains in 'moderate' condition. The stable condition observed since 2016 masks differing trends within sub-regions (Figure 13). That scores for the cover change indicator have been high in recent years demonstrates the ongoing capacity for coral cover to recover. The stable overall condition, however, reflects a range of minor disturbances that have impacted reefs variously among the sub-regions as detailed in the following sections.

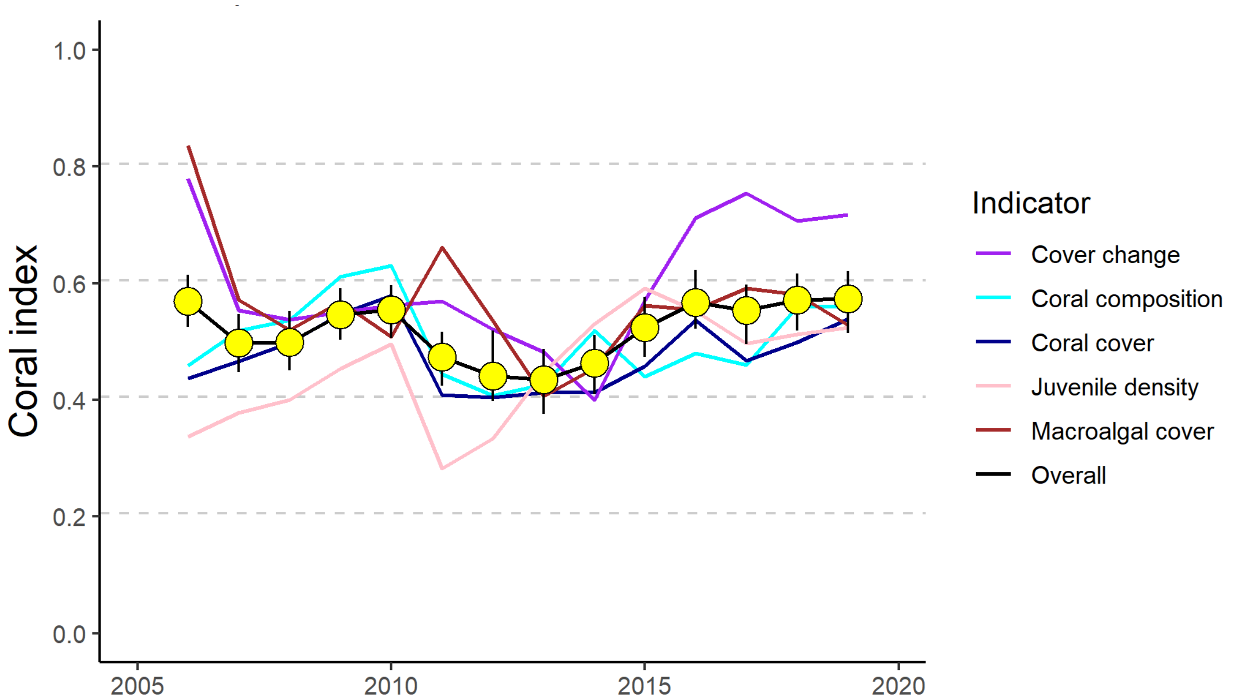

Figure 13 Trends in Coral index and contributing indicator scores for the Wet Tropics region. Coral index scores are coloured by report card category: yellow='moderate' Error in index score derived from bootstrapped distributions of indicator scores at individual reefs.

4.3.2 Wet Tropics region: Barron Daintree sub-region

In 2019 the coral condition index remains 'moderate' (Figure 14) although most indicators declined at both 2 m and 5 m depths (Table 9, Figure 16a).

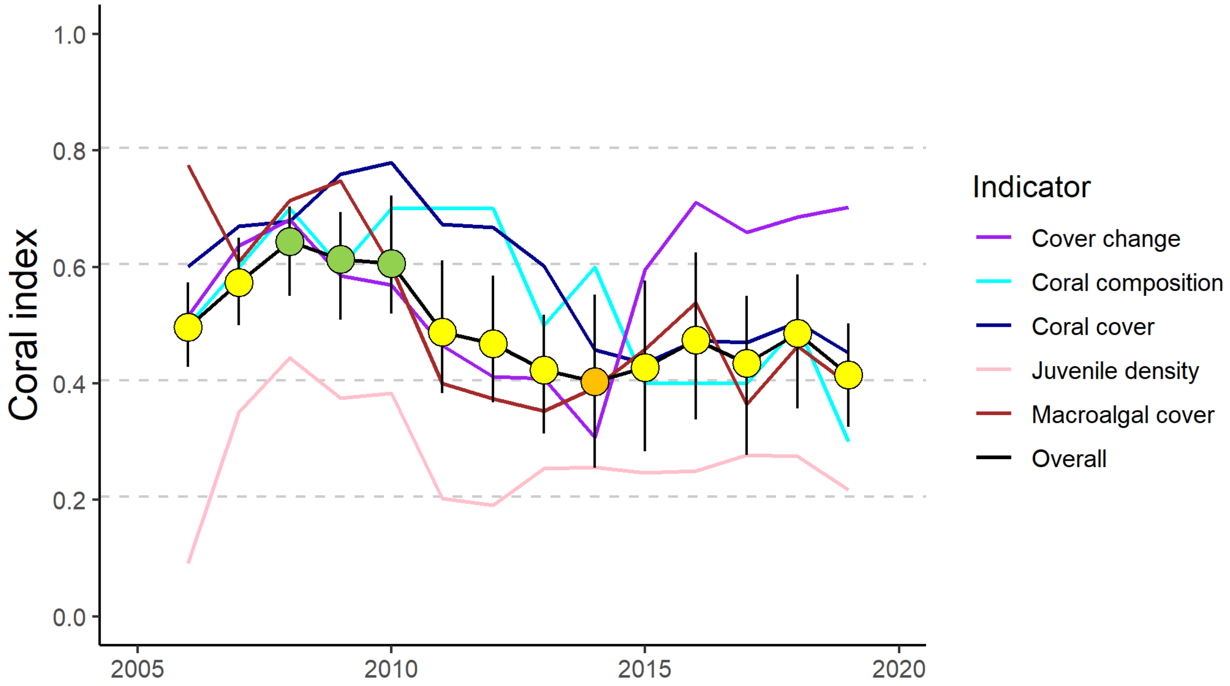

Figure 14 Trends in Coral index and contributing indicator scores for the Barron Daintree subregion. Coral index scores are coloured by report card categories: orange = 'poor', yellow='moderate' and green='good. Error in index score derived from bootstrapped distributions of indicator scores at individual reefs.

At 2 m depth improvement in cover change scores, and macroalgae scores at most reefs, contrasted decline in composition scores and variable responses for coral cover and Juvenile indicators that combined to reduce condition score relative to 2018. At 5 m depth the 2019 results contrast the consistent improvements in most indicators between 2014 and 2018 (Table 9, Figure A 1).

Table 9 Index and indicator score comparisons in the Barren Daintree sub-region. Data compare the changes in scores between local maxima and minima in the index time-series. For the 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	Condition 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	-0.26	0.81	-0.13	0.61	-0.42	0.81	-0.04	0.58	-0.21	0.70	-0.50	1.00
2014 to 2018	2	-0.03	0.87	0.12	0.93	-0.18	0.76	-0.09	0.73	0.52	0.99	-0.50	0.76
	5	0.16	0.87	0.00	0.51	0.24	0.74	0.09	0.70	0.28	0.79	0.17	0.73
2018 to 2019	2	-0.07	0.72	-0.10	0.69	0.18	0.77	-0.02	0.65	0.08	0.90	-0.50	0.76
	5	-0.07	0.76	-0.02	0.55	-0.23	0.85	-0.08	0.66	-0.02	0.55	0.00	0.00

The decline in the index reflects a combination of disturbances in recent years. The recent survey of Low Isles, was the first resurvey since late 2016 and documents a 24% reduction in coral cover due to the cumulative impact of thermal bleaching in 2017 and the ongoing presence of crown-of-thorns-starfish (Figure A 1). Coral cover also declined at the 2 m sites at Snapper Island where exposure to low salinity flood waters during record discharge from the Daintree River, and wave damage attributed to pre-cyclone Owen (Figure 8) resulted in a 38% loss of coral cover (Table A 7, Figure A 1). Elsewhere, recovery of the coral communities is continuing in the absence of acute pressures.

Sub-regional coral cover reached a low in 2015 (Figure 14, Figure 16), reflecting the cumulative impacts of crown-of-thorns starfish, cyclone Ita, and losses attributed to disease and chronic pressures that coincided with relatively high discharge from the Barron and Daintree rivers (Figure 15c, d, e). The influence of chronic pressures and disease is evident in the decline in cover change scores through to 2014 (Figure 16e). During that time, there was an increase in the cover of macroalgae at Snapper North in 2011 (peaks at 2 m and 5 m reefs, Figure 16c, Figure A 1) implicating pressures associated with nutrient availability, consistent with the long-term exposure to above guideline concentrations of Chl *a* at Snapper Island (Figure 15a). In 2019 the proportion of macroalgae in the algal communities remained elevated at Snapper North and is likely limiting the recovery process (Figure 16c, Figure A 1) despite Chl *a* concentrations returning to below guideline values.

Scores for the juvenile indicator remain poor at Snapper Island (Figure 16, Table A 8). It is reasonable to consider that the high macroalgae cover and regional declines in coral cover following consecutive thermal bleaching events in 2016 and 2017 across the northern Great Barrier Reef contribute to this result.

Despite high discharge (Figure 15d), and associated high loads of nutrients and sediments delivered from adjacent catchments (Gruber *et al.* 2020), the water quality index returned to ‘good’ condition in 2019 following two slightly lower years (Figure A 10). Not included in the index are concentrations of dissolved organic carbon and NOx, both of which show substantial increase since 2005 (Figure A 10). It remains unclear what has caused this increase and what the ramifications for corals might be (Gruber *et al. 2020*).

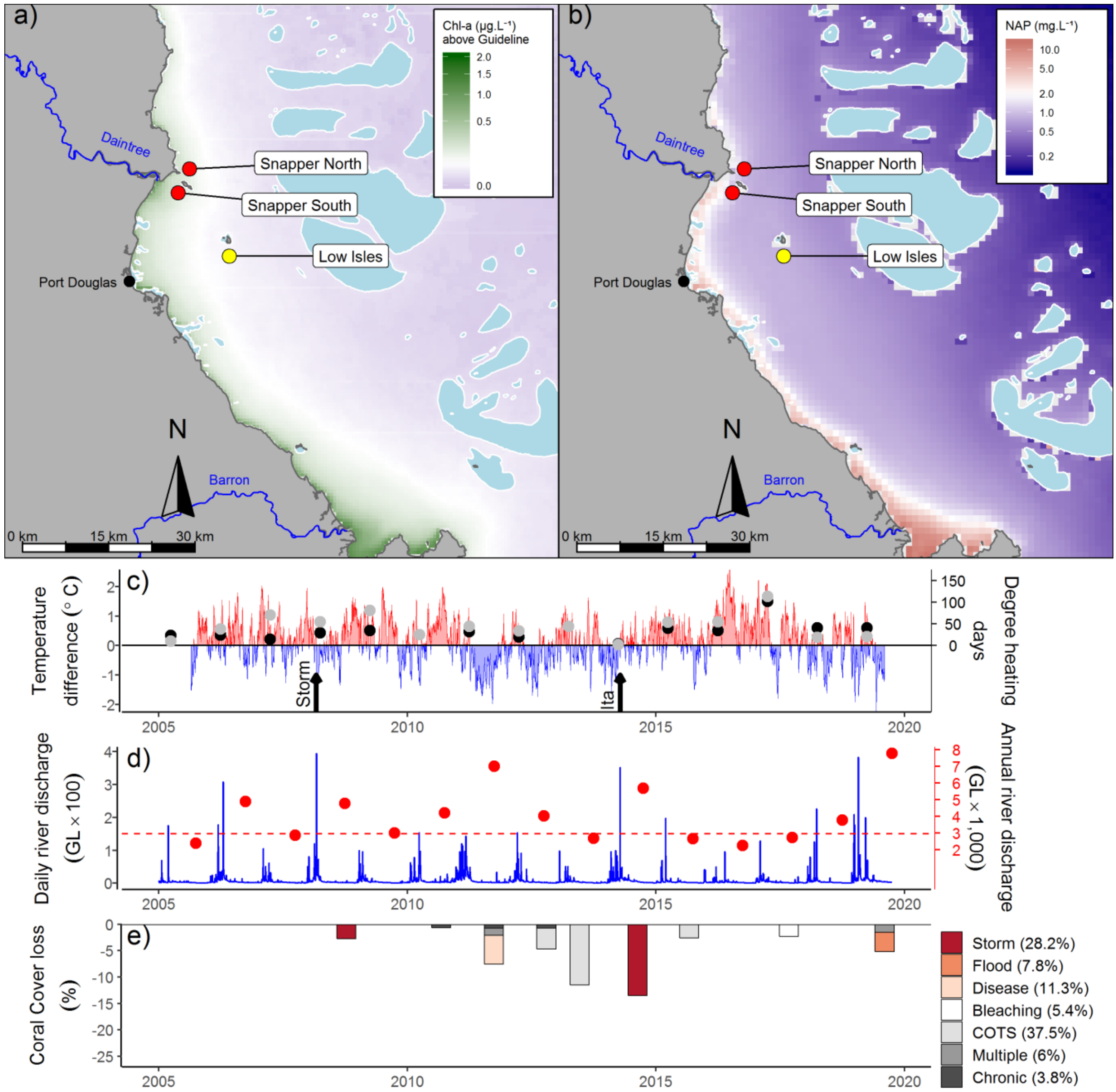

Figure 15 Barron Daintree sub-region environmental pressures. Maps show location of monitoring sites, red symbols MMP, yellow symbols LTMP along with a) mean chlorophyll a exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are mean levels 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.

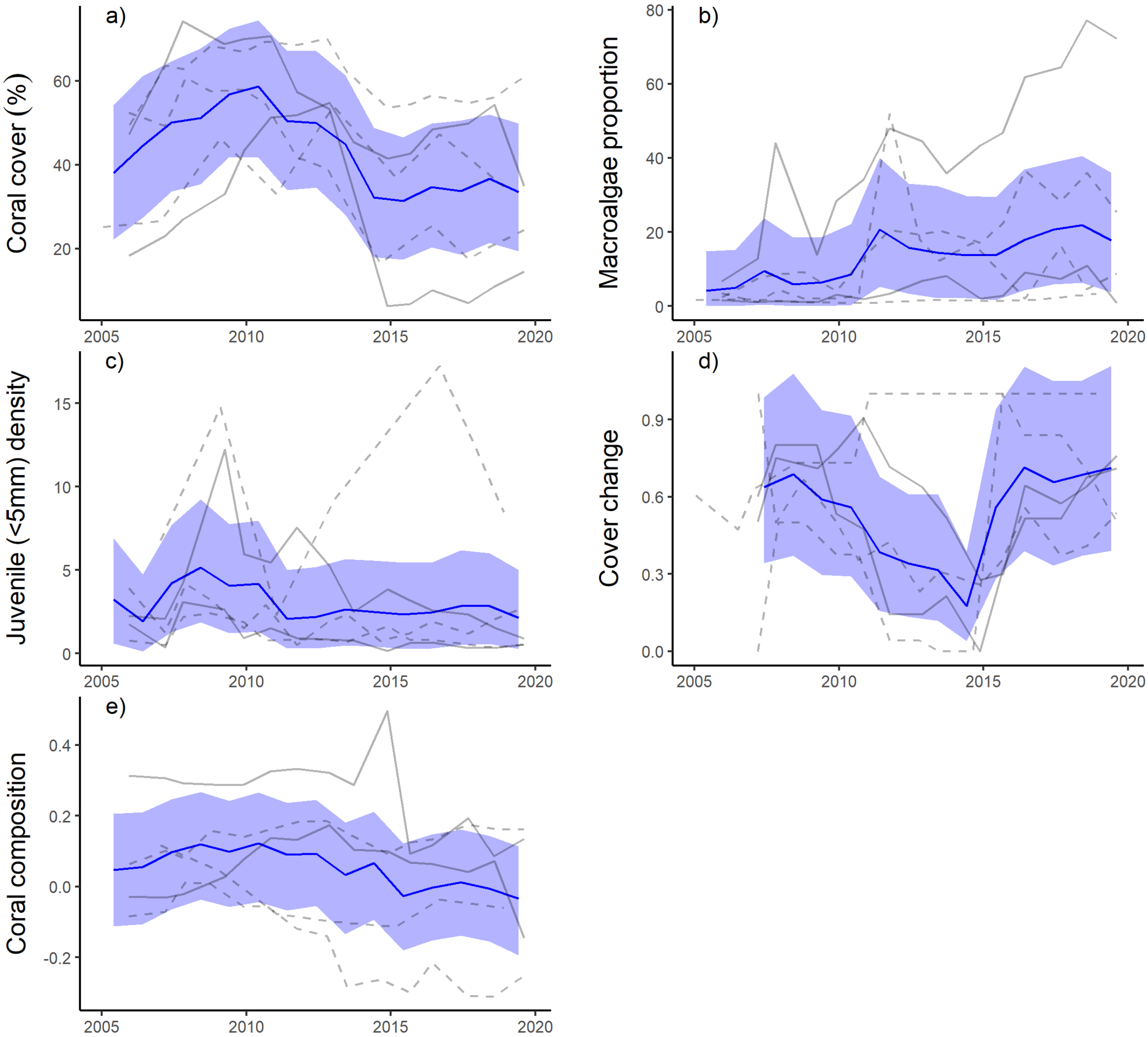

Figure 16 Barron Daintree sub-region index and indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

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

The 2019 coral index score was categorised as 'moderate', a very slight decline from the 'good' score observed in 2018 (Figure 19a).

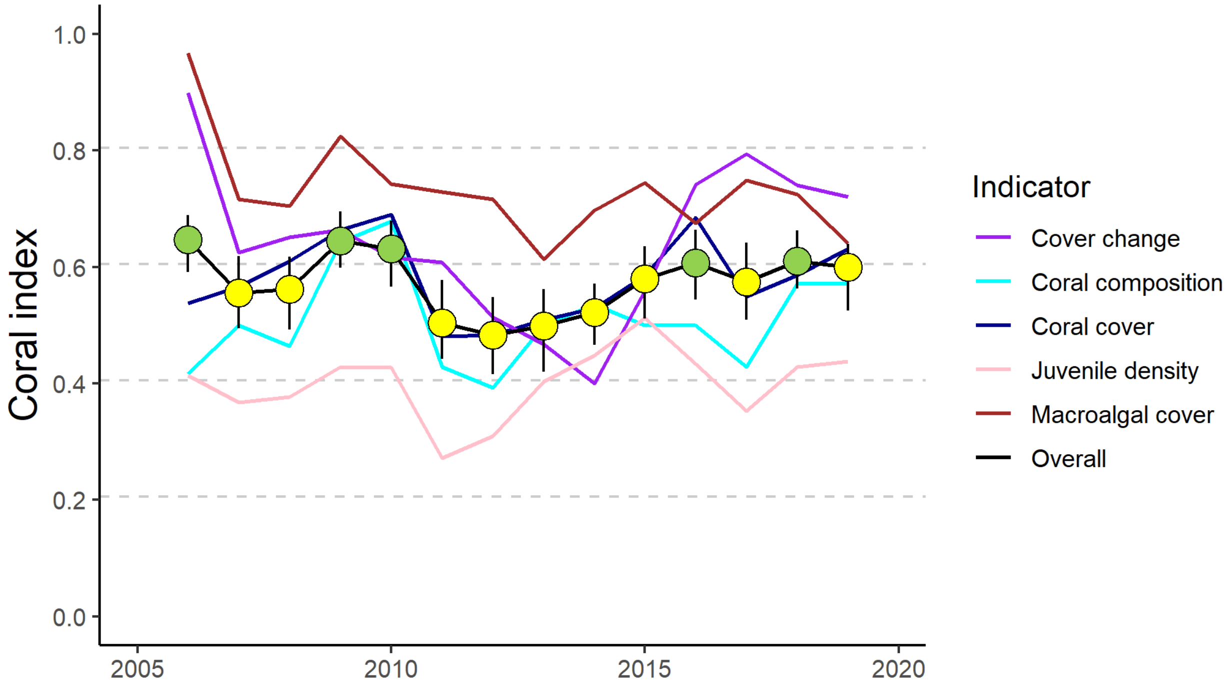

Figure 17 Trends in Coral index and contributing indicator scores for the Johnstone Russell-Mulgrave subregion. Coral index scores are coloured by report card categories: orange = 'poor', yellow='moderate' and green='good. Error in index score derived from bootstrapped distributions of indicator scores at individual reefs.

Contributing to this decline are decreased juvenile scores at two of the eight reefs in the region (Figure 19c) and a drop in macroalgae scores at High East and High West (Figure 19Figure 18a). While coral cover has generally increased across the region, the rate of increase was low leading to reduced scores for the cover change indicator at several reefs (Figure 19f). Overall the index has improved since the low levels of 2012 (Table 10) and has fluctuated about the threshold between 'moderate' and 'good' scores since 2016.

Table 10 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 index time-series. For the 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	Condition Index		Coral Cover		Macroalgae		Juvenile		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	-0.13	0.79	-0.14	0.87	-0.03	0.55	-0.12	0.82	-0.10	0.60	-0.25	0.71
2012 to 2019	2	0.15	0.83	0.24	0.83	-0.06	0.53	0.14	0.82	0.28	0.76	0.17	0.76
	5	0.09	0.76	0.08	0.66	-0.09	0.69	0.12	0.86	0.16	0.64	0.19	0.67

In general, the trend in the coral index in the sub-region reflects the impact, and subsequent recovery, of coral communities following the severe impacts associated with cyclones Tasha and Yasi in 2011 (Figure 18). 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 7). The effects of cyclones were further compounded by the increased prevalence of disease in 2011 (Figure 18e). Fitzroy Island, which had escaped serious damage from the cyclones lost a substantial proportion of the 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 7, Figure A 2). This outbreak of disease coincided with high discharge from local rivers (Figure 18d). The low point in the 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 19). 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 19a, Figure A 2).

Crown-of-thorns starfish populations peaked in 2012 (Figure A 8) and were the primary cause of coral loss at Fitzroy Island and Green Island over the period 2012-2015 (Figure 18e, Figure A 2). The impact of crown-o-thorns feeding over this period was almost certainly reduce by the Authorities crown-of-thorns control program. It is possible that the losses of coral cover attributed to thermal bleaching in 2017 included some loss due to crown-of-thorns starfish. In 2019 numbers of crown-of-thorns starfish were variable between reefs. A maximum density of 75 per ha was recorded at Frankland Island East (Figure A 8), however their impact of appears to be minimal at this stage with coral cover generally high at affected reefs.

Coral consumed by crown-of-thorns starfish will have contributed to the reduced coral change scores in the region (Figure 19d). Previously, active control programs have helped mitigate the impact from crown-of-thorns starfish, however no population control has been conducted at any of our monitoring sites over 2018 –19 (Table 7).

Discharge from rivers in the sub-region was above median levels over the 2019 water year. However, peak flows remained relatively low (Figure 18d) and evidence of direct impacts due to exposure to low salinity plume waters was limited to minimal damage suspected at 2 m depth at High West. Prior to 2018, discharge had been at, or below, median levels since 2012 (Figure 18d) and under these conditions the coral communities demonstrated a clear recovery.

The long-term water quality index in this sub-region has remained in ‘good’ condition and although there appears to be a slight declining trend in the long-term index the short-term index suggested improved conditions in 2018-2019 (Figure A 11). Not included in the index are concentrations of dissolved and particulate organic carbon, both of which show substantial increase since 2005 (Figure A 11). It remains unclear what has caused this increase and what the ramifications for corals might be (Gruber *et al. 2020*).

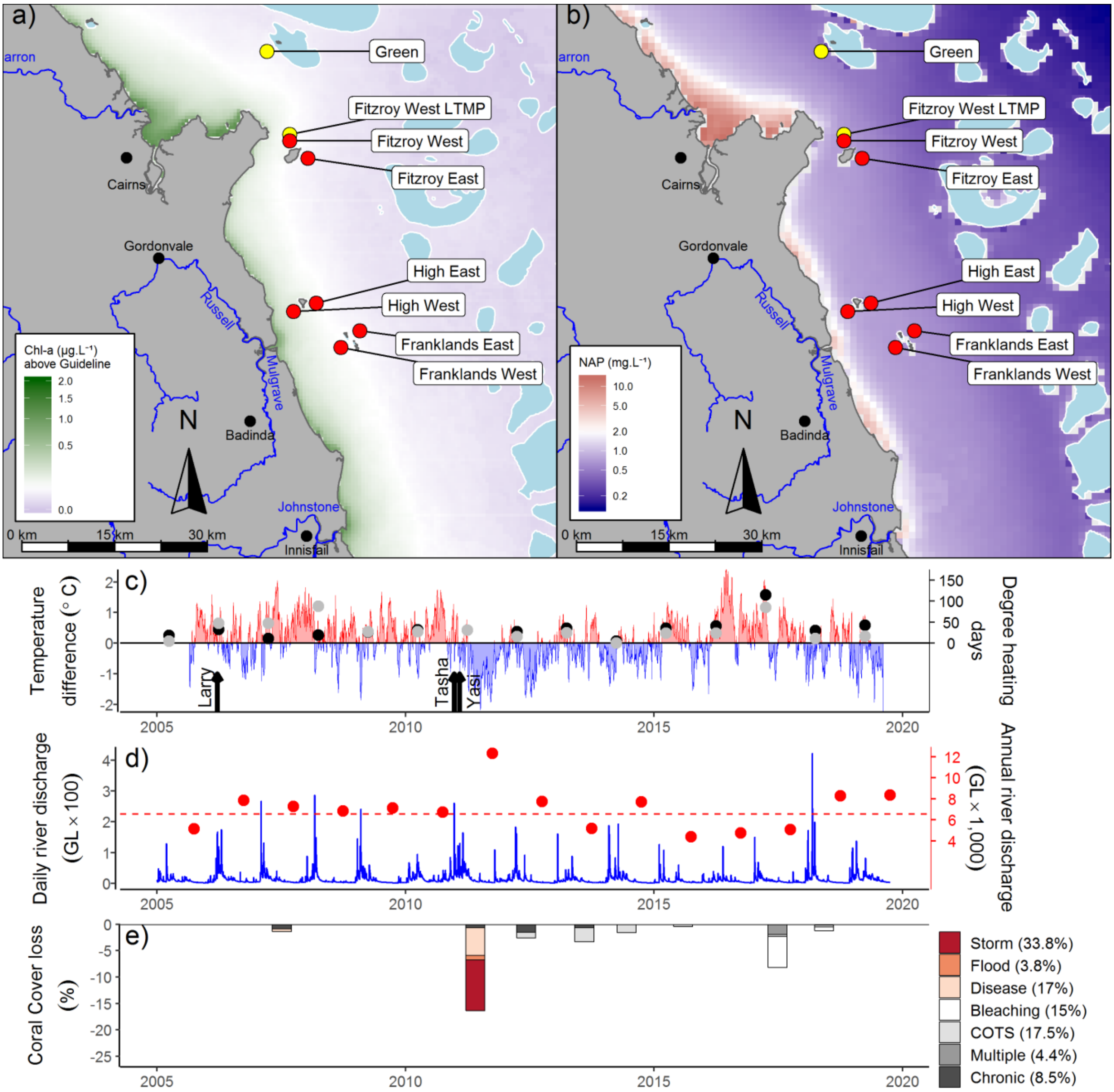

Figure 18 Johnstone Russell-Mulgrave sub-region environmental pressures. Maps show location of monitoring sites, red symbols MMP, yellow symbols LTMP along with a) mean chlorophyll a exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are 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.

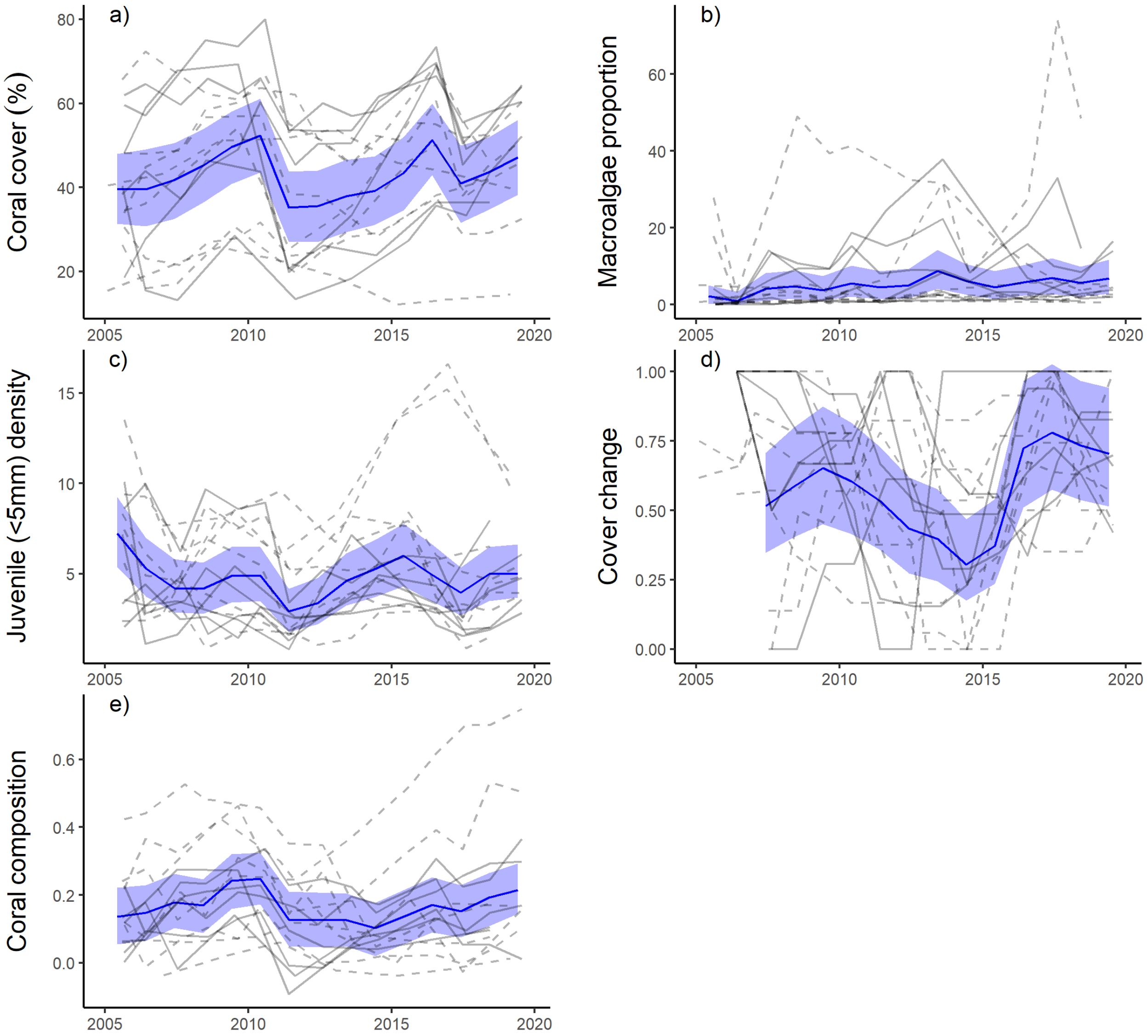

Figure 19 Johnstone Russell-Mulgrave sub-region index and indicator trends. a– e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.3.4 Wet Tropics region: Herbert Tully sub-region

The coral condition index in the Herbert Tully sub-region has improved to be categorised as ‘good’ (Figure 20).

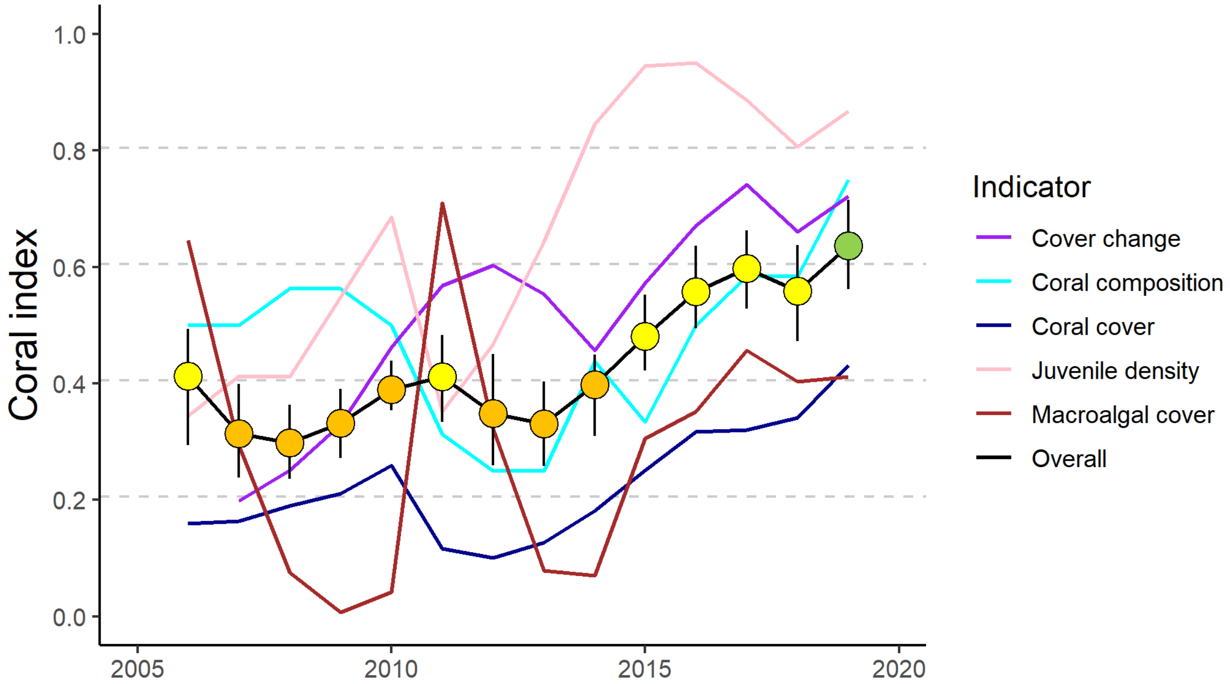

Figure 20 Trends in Coral index and contributing indicator scores for the Herbert Tully subregion. Coral index scores are coloured by report card categories: orange = ‘poor’, yellow='moderate’ and green='good. Error in index score derived from bootstrapped distributions of indicator scores at individual reefs.

The improved index scores reflect consistent improvements in most metrics since 2013 at both 2 m and 5 m depths. Exceptions were juvenile scores at 2 m and macroalgae scores at 5 m where trends were variable between reefs (Table 11).

Table 11 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 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	Condition Index		Coral Cover		Macroalgae		Juvenile		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	0.99	0.29	0.67	0.17	0.73	0.34	0.96	0.50	1.00

Since monitoring began in 2005 changes in the coral index identify 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 22a). The combined impacts of these cyclones account for 83% of hard coral cover losses since 2005 (Figure 21e). At the three reefs samples in both 2005 and 2018 or 2019 coral cover was 36% representing a recovery to 86% of the 42% cover observed in 2005.

Of note is that following each cyclone, in addition to an immediate reduction, was a lagged decline in the index scores (Figure 20). This lagged response reflects temporary improvement in the macroalgae indicator score in the first post-cyclone survey (Figure 22d). During cyclones, macroalgae are stripped from the substrate, temporarily reducing their abundance. Subsequent colonisation of space made available to algae due to reduced coral cover results in a lagged impact on index scores.

A strong contributor to the current score has been the rapid rate at which coral cover has recovered, influencing both the cover change and coral cover indicators (Figure 20). In contrast there has been a decline in the density of juvenile corals (Figure 22c), although scores remain high across the sub-region (Figure 20). The decline in juvenile densities reflects a reduction in *Turbinaria* recruitment following an exceptionally high recruitment pulse in 2013 (Figure 22c, Figure A 3) – it is not expected that this decline will have drastic implications for future recovery of these coral communities.

No major floods were observed across the region in 2019 (Figure 21d) with annual discharge from adjacent catchments only a little over median levels. No direct impacts of flood waters were observed at the time of coral surveys. It is possible, however, that the continued poor performance of coral communities at Bedarra, the closest site to rivers, reflects regular exposure to low salinity waters and/or poor water quality (Table A 9, Figure 21a) even in the absence of major floods.

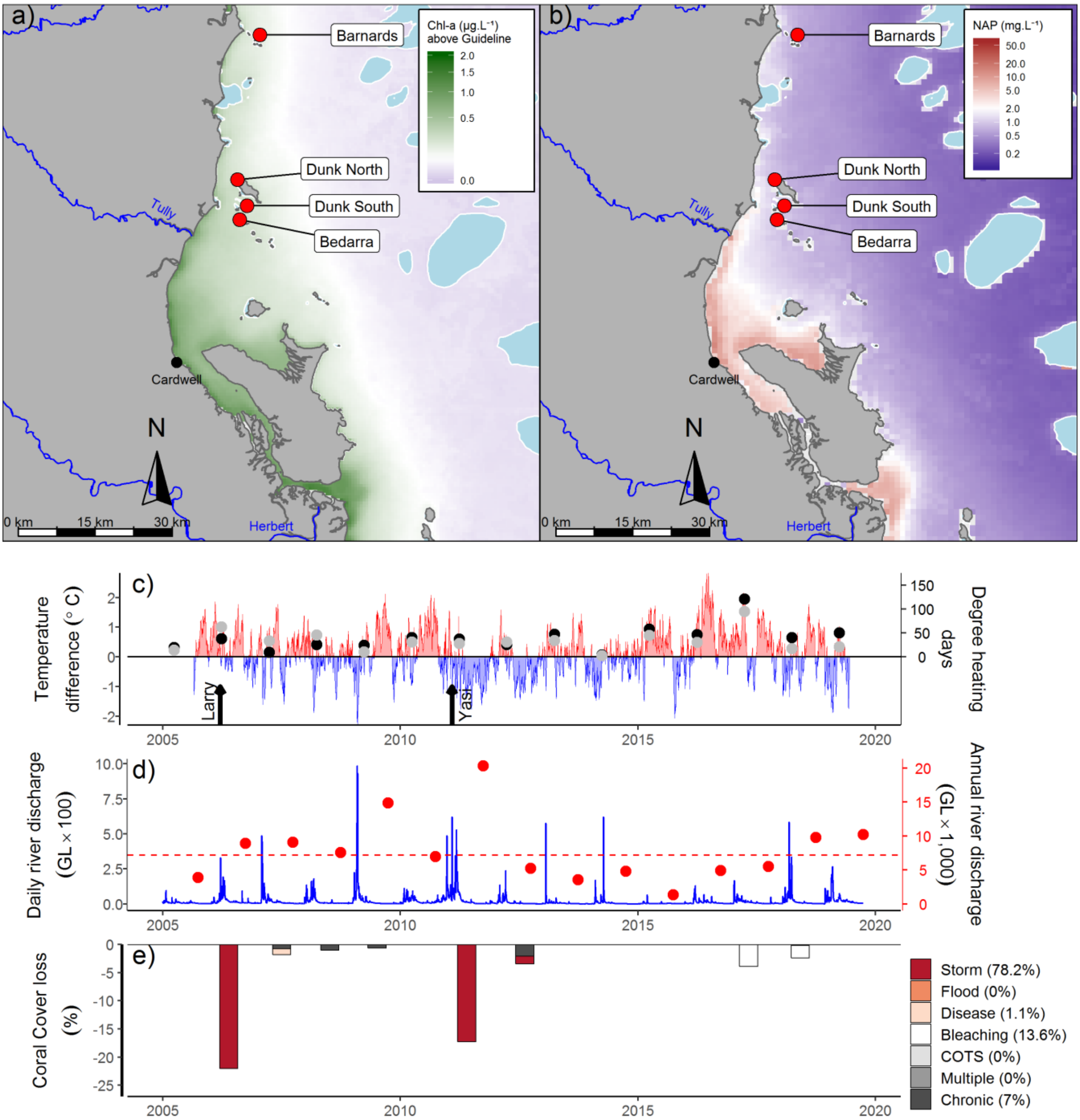

Figure 21 Herbert Tully sub-region environmental pressures. Maps show location of monitoring sites, red symbols MMP, yellow symbols LTMP along with, a) mean chlorophyll a exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are mean levels 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 (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.

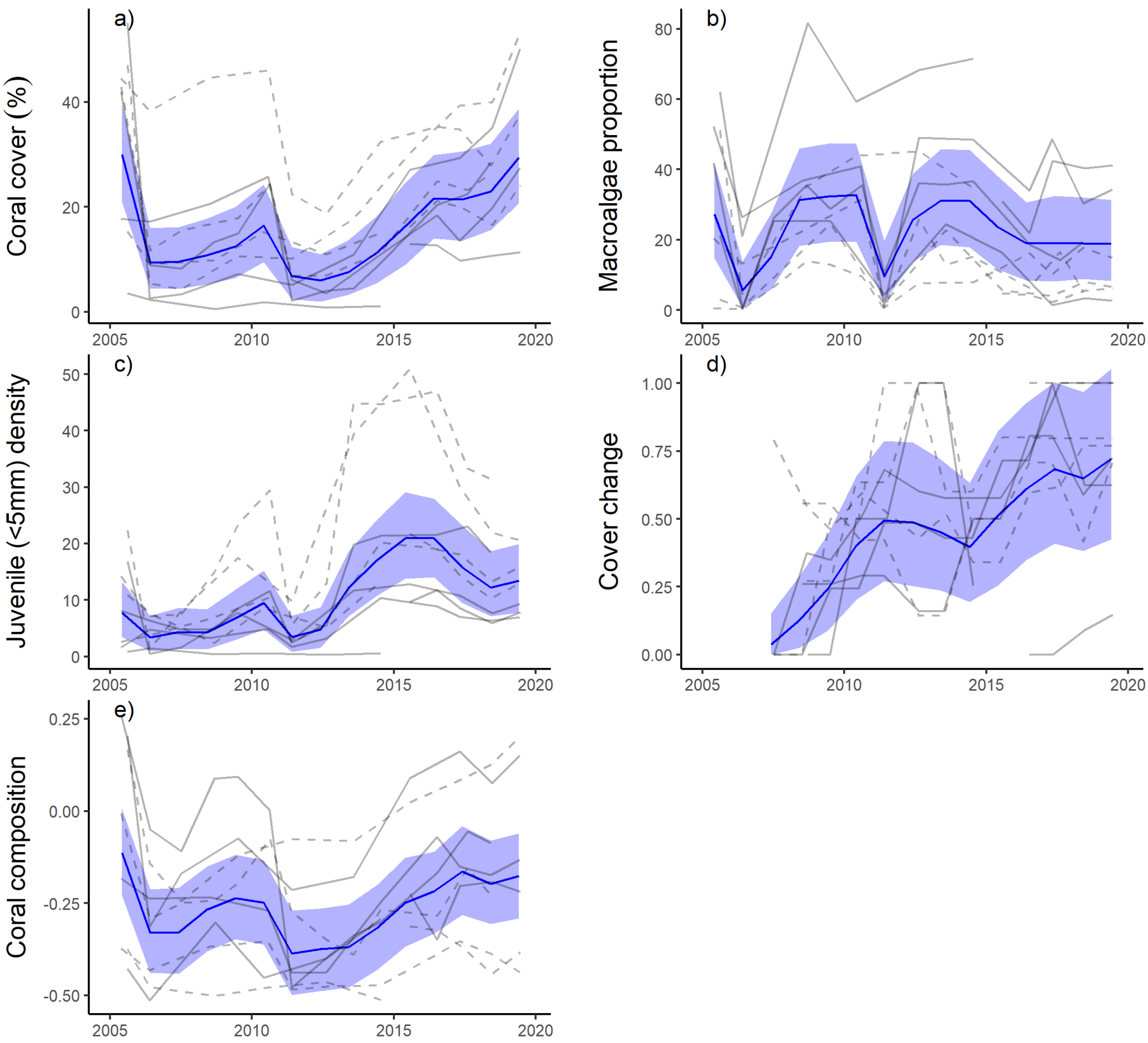

Figure 22 Herbert Tully sub-region indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

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* concentrations exceed the guideline) waters (Figure 21a, Table A 9). The combination of low turbidity and high nutrient availability is consistent with the prevalence of macroalgae observed in the shallow depths at most reefs (Figure 22b, Figure A 3) and reflected in the ongoing poor scores for both the long-term and short-term water quality index for this sub-region (Figure A 12

Figure A 12).

4.4 Burdekin region

The coral condition index has remained stable and categorised as ‘moderate’ since 2016, which represents an improvement since 2013 (Figure 23, Table 12).

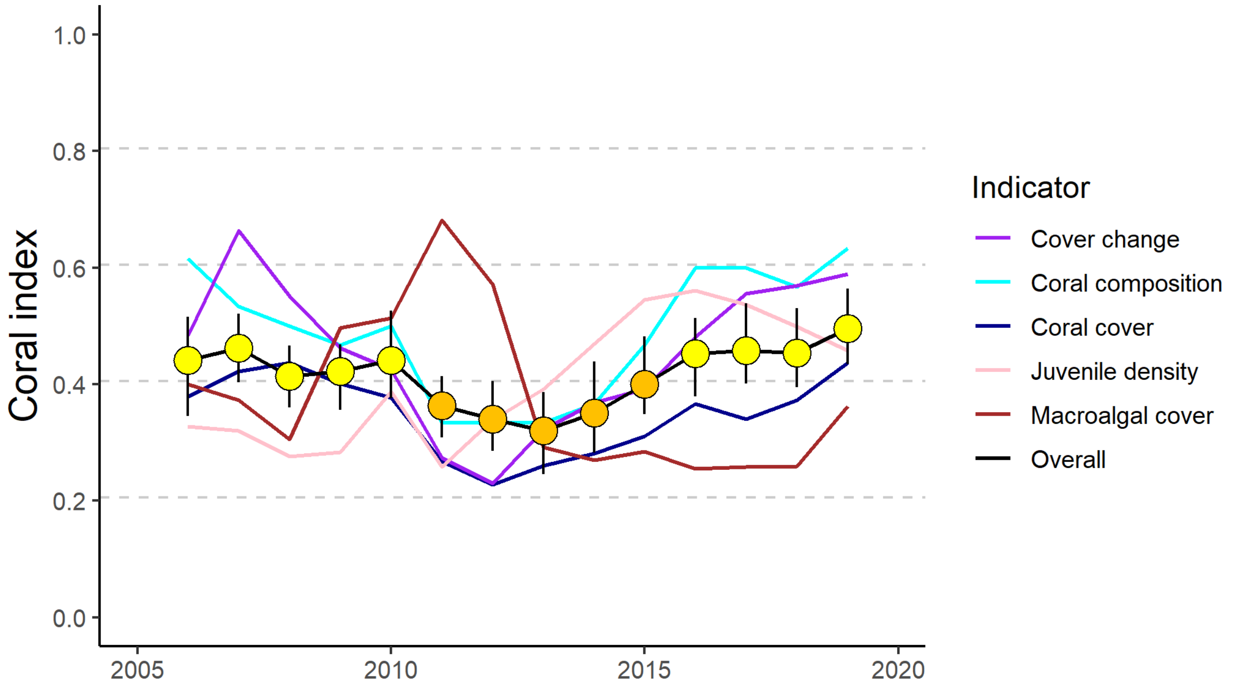

Figure 23 Trends in Coral index and contributing indicator scores for the Burdekin region. Coral index scores are coloured by report card categories: orange = ‘poor’, yellow=’moderate’. Error in index score derived from bootstrapped distributions of indicator scores at individual reefs.

Increased indicator scores were most evident at 5 m depths where all indicators have shown consistent improvement (Table 12). At 2 m depths the continued improvement in the index since 2013 is driven most consistently by the coral cover and composition metrics. The other indicators have remained stable at the regional scale (Table 12).

Overall, modest declines in macroalgae cover at reefs surveyed in 2019 resulted in macroalgae scores improving within the ‘poor’ range (Figure 23, Figure 25b). The proportional cover of macroalgae has been variable over time with a low point recorded in 2009 (the reason for this decline remains unexplained), and then again in 2011 because of macroalgae being stripped from some reefs during cyclone Yasi (Figure 25b). By 2012 macroalgae had re-established. The recent, albeit modest, decline in macroalgae observed in 2019 is perhaps unexpected given the major flooding of the local rivers over the 2018-19 wet season (Figure 24) although it is possible this is a short-term response. Surveys in 2020 will be informative as to whether these declines occurred at reefs not surveyed in 2019, which are closer to the coast.

Improvement in the coral index prior to 2017 coincided with a period of below median discharge from the region’s rivers (Figure 24d) and absence of acute disturbances (Figure 24c). Despite widespread thermal bleaching over the 2016–2017 summer, which accounted for 18% of the coral cover lost since 2005, the index score remained relatively stable. Since 2017 coral cover has continued to increase at rates that support high scores for the cover change indicator at most reefs (Figure 23). Notably, marked increases in coral cover continue at Palms East, demonstrating the strong recovery of this coral community following the severe impact of cyclone Yasi (Figure A 4).

In 2019 it was only at Havannah, 2 m depth, that coral cover declined – corals at this site were severely bleached in 2017 and disease has since killed large stands of branching *Acropora*, in particular, *A. pulchra*. This progression of disease and loss of coral cover is influential in the inconsistent trend in the cover change indicator at 2 m depths (Table 12).

In contrast, improvement in the composition indicator score (Table 12) reflects that cover increase regionally includes recovery of taxa sensitive to poor water quality, in particular, cover of *Acropora* spp. has increased notably at some reefs (Figure A 4, Table A 10).

Juvenile indicator scores have improved since 2013 at 5 m depths (Table 12) but have declined in recent over the last three years at the regional scale (Figure 23) as a strong recruitment cohort of *Turbinaria* (Family Dendrophylliidae) (Figure A 4) passes through the juvenile size class at some reefs.

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 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.

	Depth	Condition Index		Coral Cover		Macroalgae		Juvenile		Cover Change		Composition	
)		Score	P	Score	P	Score	P	Score	P	Score	P	Score	P
2010 to 2013	2	-0.08	0.70	-0.09	0.64	-0.17	0.71	-0.04	0.61	-0.05	0.54	-0.07	0.57
	5	-0.15	0.86	-0.14	0.82	-0.26	0.82	0.04	0.61	-0.15	0.80	-0.25	0.71
2013 to 2019	2	0.12	0.78	0.19	0.87	0.06	0.67	-0.09	0.62	0.11	0.58	0.33	0.72
	5	0.24	0.93	0.22	0.90	0.11	0.73	0.17	0.81	0.40	0.95	0.31	0.73

Declines in the index through to 2013 coincided with the combined influence of cyclone Yasi and a period of very high discharge from the region's rivers (Figure 24d, e). Since 2005, cyclones and storms have accounted for 50% of hard coral losses (Figure 24e). The lagged influence from cyclone Yasi noted in 2012 (Figure 24e), is due to LTMP surveys post-Yasi not occurring until that year. East–facing locations, such as Palms East and Lady Elliot (2 m), were particularly exposed to storm driven seas, and show the impacts of cyclone Larry (2006) and cyclone Yasi (2011) (Figure A 4, Table A 7). At both these reefs coral cover has returned to similar levels as observed in 2005. At Lady Elliot the community composition is similar to that present in 2005 while. At Palms East the 2005 community dominated by soft corals has been replaces by a hard-coral community dominated by Acroporidae (Figure A 4)

The last outbreak of crown-of-thorns starfish on the inshore reefs in this region occurred at Havannah in 2001. A single crown-of-thorns starfish was observed during surveys at Palms West in 2019 with little evidence of impacts at this stage. Populations of crown-of-thorns starfish remain in outbreak densities at some mid-shelf reefs in the region (AIMS LTMP).

The period 2010 to 2013 saw a reduction in scores for the cover change indicator at 5 m depths (Table 12). This reduced rate of increase in coral cover saw observed cover fall below that predicted by coral growth models and accounts for most of the cover loss since 2005 attributed to chronic pressures (Figure 24e). Although not categorised as a disease outbreak for disturbance estimation, elevated levels of disease were observed from 2007 to 2009 (Figure A 7) and will have contributed to the chronic disturbances recorded over the period 2008 to 2010 (Figure 24e).

Chronic pressures are assumed when there is no evidence for impacts associated with acute disturbance. Rather, they represent the cumulative impacts of environmental pressures that suppress the annual increments in cover that are the basis of the cover change scores. As *Acropora* and *Montipora* were the genera most infected by disease, the disproportional loss of these groups will have contributed to the decline in the composition indicator score.

Major flooding occurred in the region in February 2019 (Figure 24d). Despite this event there was no direct impacts to coral communities surveyed in 2019. However, incidence of disease did increase slightly in 2019 which may be attributed to the decreased water quality associated with the floods. Impacts of disease in 2019 were most evident at Havannah, where coral cover declined by 23% at 2m (Figure A 4). While this is likely to be a continuation of disease observed at this reef following the thermal bleaching in 2017 it is possible that reduced fitness from plume water exposure exacerbated the effects. Informal observations by the author in the weeks following the floods noted mortality of most corals along the south-eastern rocky shores of Cape Cleveland and at Virago Shoal that was almost certainly caused by exposure to low salinity.

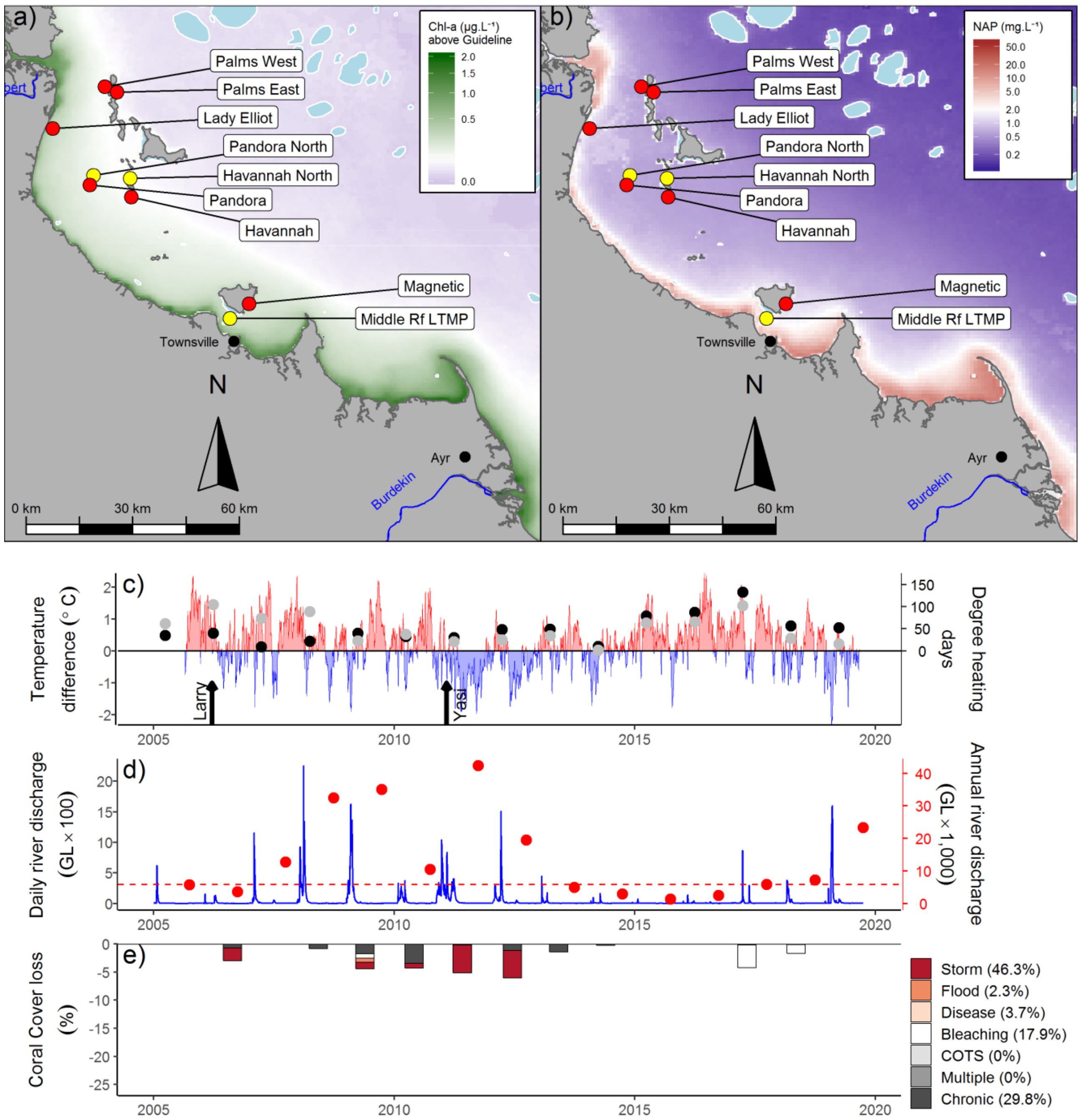

Figure 24 Burdekin Region environmental pressures. Maps show location of monitoring sites, red symbols MMP, yellow symbols LTMP along with a) mean chlorophyll a exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are mean levels over the 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.

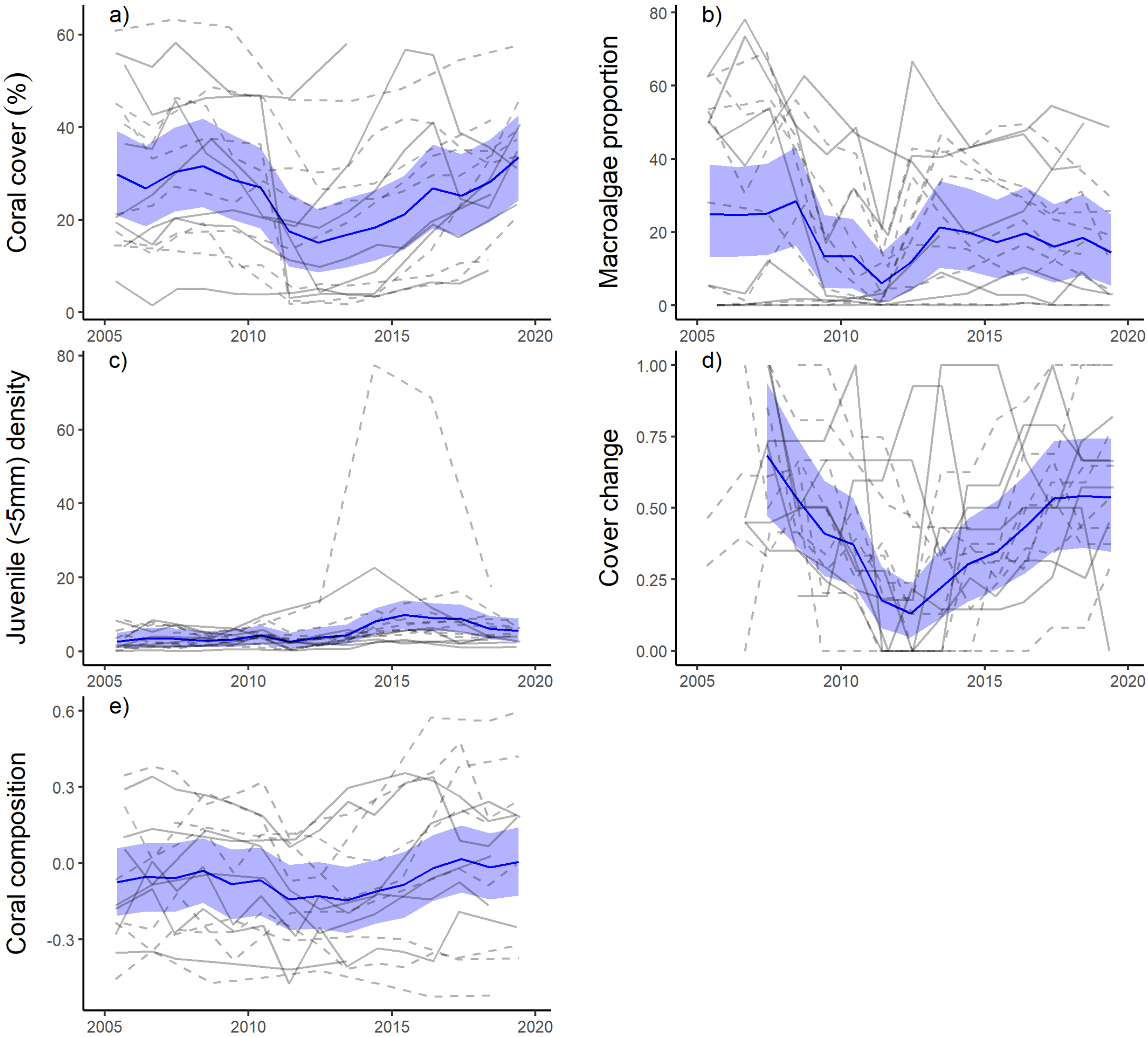

Figure 25 Burdekin Region index and indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

Improvement in the coral index between 2013 and 2016 (Figure 25a) coincided with low discharge, and corresponding relatively low loads of nutrients and sediments being delivered to the Reef (Figure 24d, Gruber *et al.* 2020). Despite variability in loads and demonstrated coral recovery there was no observable improvement in the region's water quality index over this time with measured water quality constituents appearing to remain stable or increase (Figure A 13). The lack of direct response between measured water quality and loads limits the ability to explicitly link improvement in the index scores to water quality drivers.

4.5 Mackay-Whitsunday region

The coral index has continued to decline and is now 'poor' (Figure 26, Table 13).

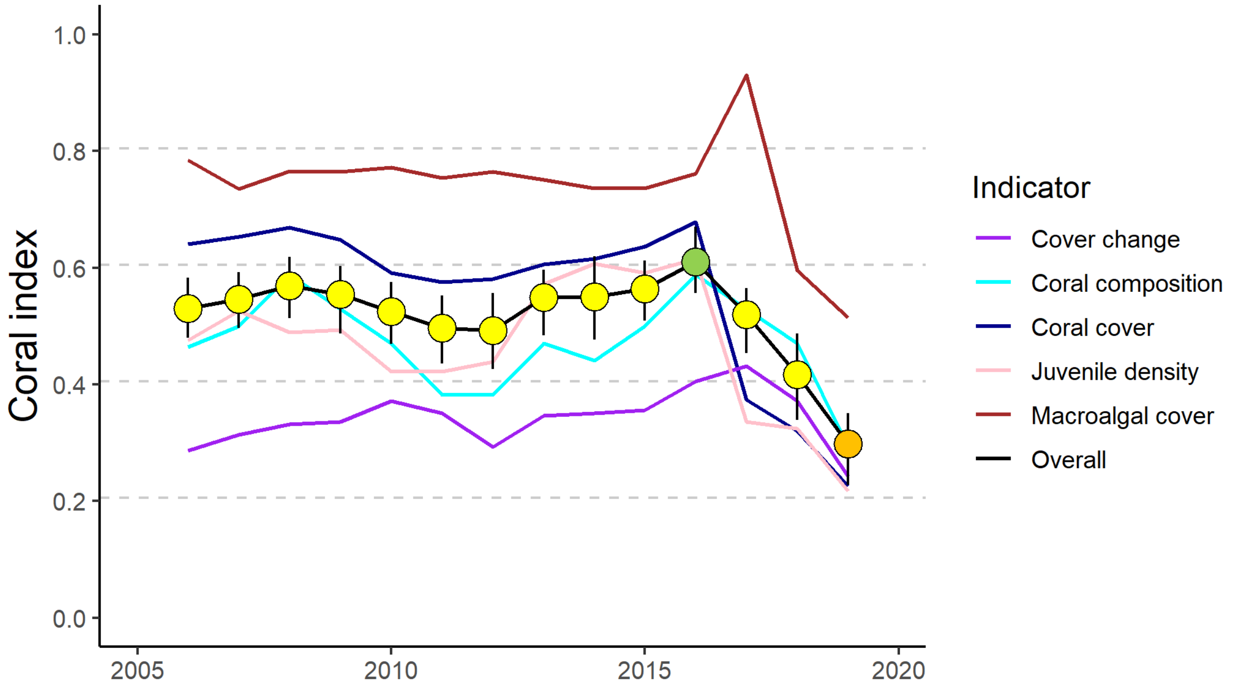

Figure 26 Trends in Coral index and contributing indicator scores for the Mackay-Whitsunday region. Coral index scores are coloured by report card categories: orange = 'poor', yellow='moderate', green='good'. Error in index score derived from bootstrapped distributions of indicator scores at individual reefs.

The continued decline reflects the final documentation of direct impacts of cyclone Debbie and the lack of recovery to-date (Figure 27e, Figure 26). Strongly influencing the decline in the index in 2019 was the addition of current survey data from the AIMS LTMP sites at Hayman, Langford, and Border that were last monitored in the month prior to cyclone Debbie. All three of these reefs were severely impacted, most notably Hayman where 86% of coral cover was lost (Figure A 5, Figure 28a). The impact of cyclone Debbie has resulted in regional declines in all indicator scores (Table 13). Only scores for the cover change indicator have not consistently declined, however scores for this indicator have been consistently low since surveys began in 2005 (Figure 26).

The bloom in macroalgae observed in 2018 following the lagged colonisation of available space (Figure 28b) continues to affect scores for this indicator. Since cyclone Debbie, macroalgal communities have been in a state of succession. At Double Cone macroalgae cover has declined as thick mats of red macroalgae observed in 2018 are being replaced by a lower cover of potentially persistent brown macroalgae *Lobophora* and *Sargassum* (Table A 12). Similarly, at Pine the initial increase in macroalgae cover was dominated by *Lobophora* and red macroalgae species that are now being replaced by *Sargassum* which was common prior to 2017 (Table A 12).

Prior to cyclone Debbie, the only acute disturbance events recorded since 2005 were flooding in 2009 and cyclone Ului in 2010 (Figure 27e). These contributed to a slight decline in the coral index through to 2012. Daydream was severely impacted by cyclone Ului, losing 47% of the coral cover at 5 m depth (Figure A 5, Table A 7). By 2016, coral cover at Daydream had recovered to its former level. Following cyclone Debbie, cyclones now contribute to 73% of coral loss since 2005 (Figure 27e).

Table 13 Index and indicator score comparisons in the Mackay-Whitsunday Region. Data compare the changes in scores between local maxima and minima in the index time-series. For the 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	Condition Index		Coral Cover		Macroalgae		Juvenile		Cover Change		Composition	
		Score	P	Score	P	Score	P	Score	P	Score	P	Score	P
2008 to 2012	2	-0.07	0.77	-0.07	0.91	0.00	0.00	-0.08	0.80	-0.05	0.60	-0.14	0.72
	5	-0.08	0.80	-0.10	0.87	0.00	0.63	-0.03	0.61	-0.03	0.53	-0.25	0.83
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 2019	2	-0.37	0.92	-0.55	0.98	-0.30	0.73	-0.39	0.98	-0.24	0.81	-0.36	0.74
	5	-0.27	0.93	-0.39	0.97	-0.21	0.72	-0.40	0.93	-0.11	0.67	-0.25	0.76

Chronic stress remains the second highest contributor to the loss of hard coral in the region (Figure 27e). The influence of chronic environmental pressures in the region is demonstrated by the marked differences in the composition of coral communities between 2 m and 5 m depths (Figure 28e, Figure A 5). High turbidity at most of the MMP reef sites (Table A 9), in combination with limited exposure to wave energy among the Whitsunday Islands, results in reduced availability of light and accumulation of fine sediments at 5 m depths and the selection for corals 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 metric 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 26), that in turn contribute to the frequently categorised chronic stresses (Figure 27e). This is particularly concern for reefs dominated by corals other than Acroporidae as their growth expectation is low within the model. With a decline in the long-term water quality index and short-term index value of poor in 2019 (

Figure A 14) there is little prospect for an improvement on this front.

High incidence of coral disease were observed in 2007 and 2008 as discharge from local catchments rose to above median levels (Figure 27d), and then in the two years following cyclone Ului, again coinciding with high river discharges (Figure 27d, e, Figure A 7). Disease was noted as a cause for decline in coral cover at Dent Island in 2019 again coinciding with above median discharges from the local catchment, however this did not translate into a region-wide increase in disease, primarily due to the current low coral cover on most reefs (Figure A 7, Figure 28a).

Direct impacts due to flooding were recorded only in 2009 (Figure 27e), attributed primarily to the high loads of sediments observed on corals during surveys. The source of these sediments is not clear as the local rivers did not experience extreme flooding over the preceding summer (Figure 27d), although local heavy rainfall did result in several land-slides along the adjacent ranges.

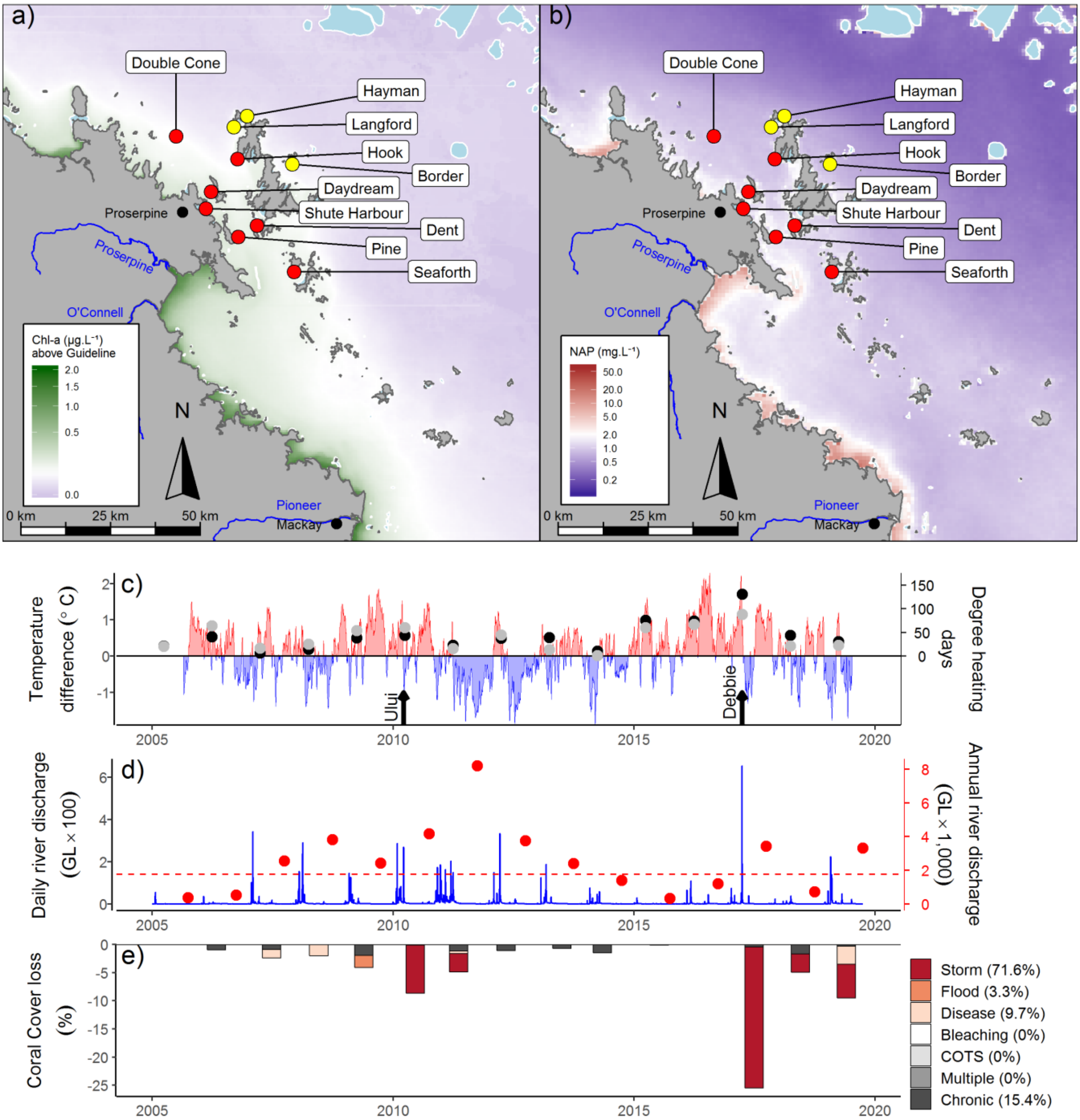

Figure 27 Mackay-Whitsunday Region environmental pressures. Maps show location of monitoring sites, red symbols MMP, yellow symbols LTMP along with a) mean chlorophyll a exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are mean levels 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.

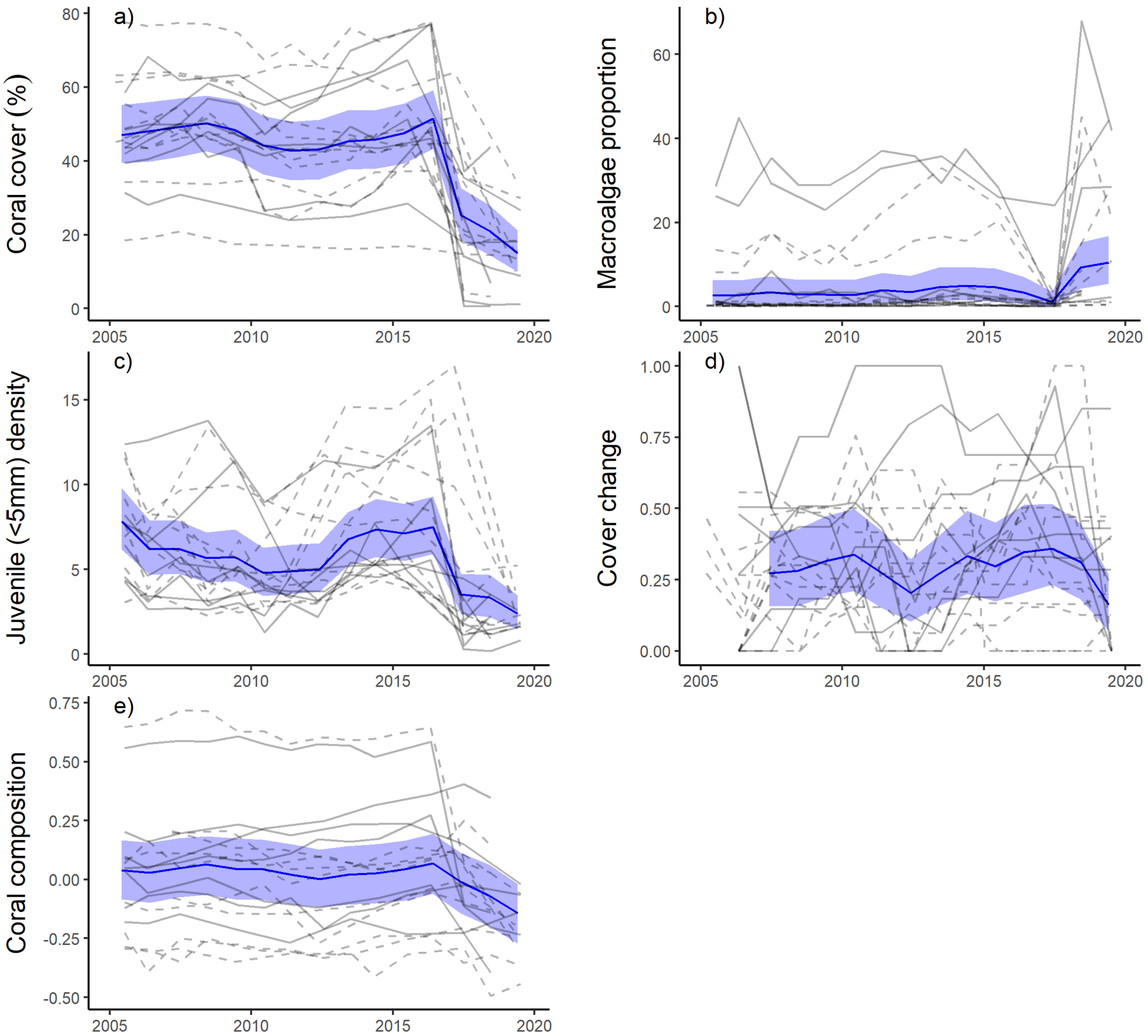

Figure 28 Mackay-Whitsunday Region index and indicator trends. a – e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.6 Fitzroy region

The coral index score in the Fitzroy Region remains 'poor' but continues to improve from the 'very poor' condition observed in 2014 (Figure 29, Table 14).

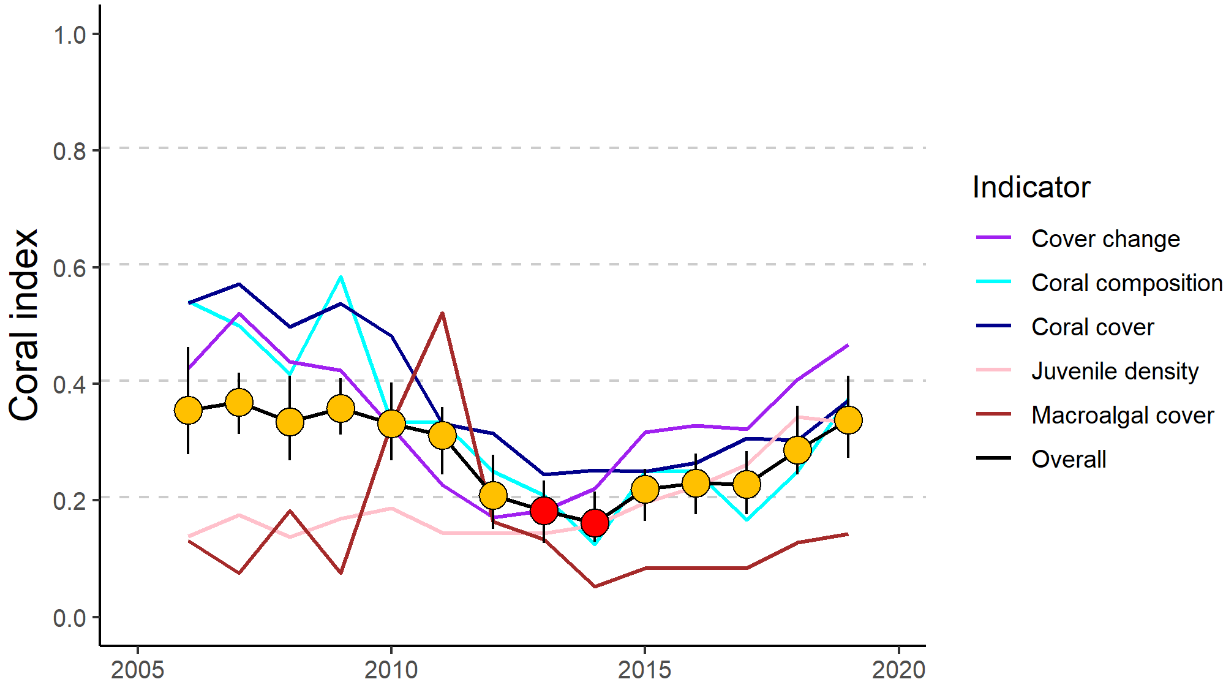

Figure 29 Trends in Coral index and contributing indicator scores for the Fitzroy region. Coral index scores are coloured by report card categories: red='very poor', orange = 'poor'. Error in index score derived from bootstrapped distributions of indicator scores at individual reefs.

Improvements in index scores reflect improvement in scores for all indicators at both 2 m and 5 m depths of most reefs (Table 14, Figure 31a,c,d). Despite these improvements the scores for individual indicators range from 'very poor' for macroalgae to 'moderate' for change in cover, the remaining indicators being 'poor' (Figure 29, Table A 8).

Table 14 Index and indicator score comparisons in the Fitzroy Region. Data compare the changes in scores between local maxima and minima in the index time-series. For the 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	Condition Index		Coral Cover		Macroalgae		Juvenile		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 2019	2	0.18	1.00	0.13	0.89	0.09	0.74	0.23	0.84	0.20	0.75	0.25	0.74
	5	0.17	0.92	0.11	0.74	0.09	0.75	0.13	0.75	0.29	0.83	0.25	0.82

The coral communities 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 30a, 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 30a, b, Table A 9). The gradient in water quality is clearly reflected in the benthic communities. 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 high turbidity. The differences in community composition are evident in the baseline conditions for the composition indicator (Figure 31e). Pelican has a highly stratified environment, supporting slow growing, low-light tolerant corals at depth, and 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 defined by low cover of corals, low density of juvenile corals and high cover of macroalgae (Figure A 6). A lack of substantial reef development at both Peak and Pelican suggests that the environmental conditions at these locations are marginal for most corals; questioning their suitability as inshore coral reef monitoring locations.

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 10).

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 30c-e). These disturbances resulted in a clear reduction in coral cover (Table 14, Figure 31a). The disproportionate loss of *Acropora* (Figure A 6) resulted in a reduction in the community composition indicator scores (Table 14, Figure 29). Compounding the impact of the acute disturbances was a low rate of recovery of coral cover demonstrating the effect of chronic impacts (Figure 30e), evidenced by high levels of disease (Figure A 7) and declines in the cover change scores between 2007 and 2014 (Table 14). During this period of slow recovery annual discharge from the Fitzroy River was mostly well above median levels (Figure 30d). No assessment of change in the macroalgae indicator scores between 2007 and 2014 at 5 m depth was possible (Table 14) as scores were zero at all reefs in both years: further implicating availability of nutrients in the observed slow recovery. 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 31c, Diaz-Pulido *et al.* 2009).

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 7). 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.

Recent improvement in the coral change indicator (Table 14) to “moderate’ levels reflects improvement at several reefs, however across the region, recovery remained slightly below expected rates. A maximum score of 1 for the coral change indicator returned for Pelican Island, 2 m depth (Table A 8), should be treated with caution as expected increase from the almost zero coral cover at that location is very low and within sampling error of the coral surveys (Figure 30d).

Elevated water temperatures (2016 and 2017, Figure 30c) and exceedance of median discharge levels from the local catchment (in 2017, Figure 30d) 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 30e).

Water quality monitoring (in-situ) was discontinued in the Keppels region in 2015. The final year of water quality sampling saw an improvement in the water quality index (Lønborg *et al.* 2015). Measured levels of Chl *a*, showed a slight downturn coinciding with a respite from flooding in the region since 2012 (Lønborg *et al.* 2015). Modelling of total suspended solids and dissolved inorganic nitrogen indicate substantially lower concentrations in the region from 2014 to 2016 compared to those associated with the high discharge years of 2010, 2011 and 2013 (Waterhouse *et al.* 2017).

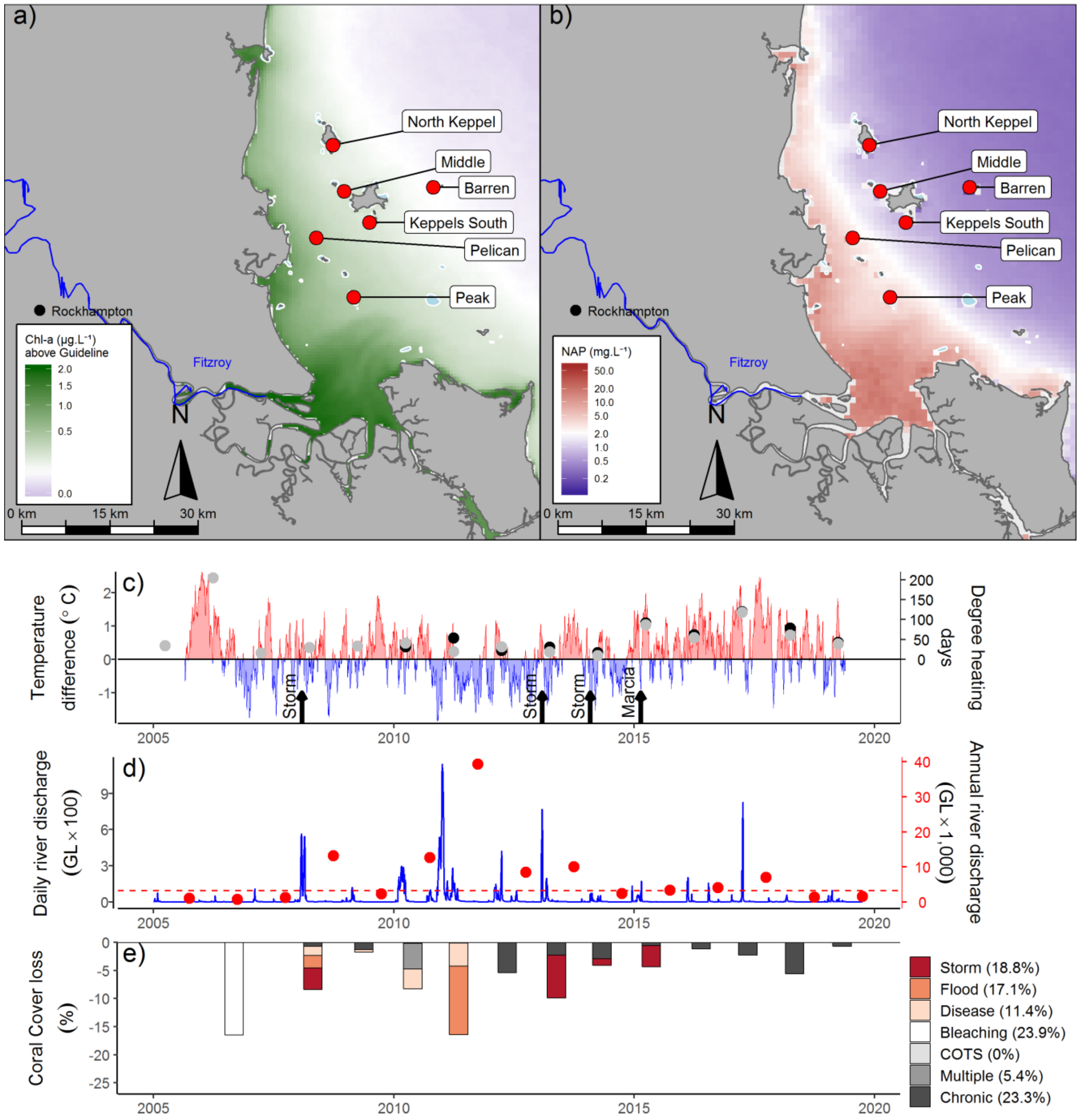

Figure 30 Fitzroy Region environmental pressures. Maps show location of monitoring sites, red symbols MMP, yellow symbols LTMP along with a) mean chlorophyll a exceedance of wet season Guideline (0.63ugL-1) and b) mean Non Algal Particulate concentrations. Water quality data are mean levels over the 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 Calliope and Fitzroy rivers and Waterpark Creek, red dashed line represents long-term median discharge (1986–2016). e) break-down of hard coral cover loss by disturbance type; length of bars represents the mean loss of cover across all reefs.

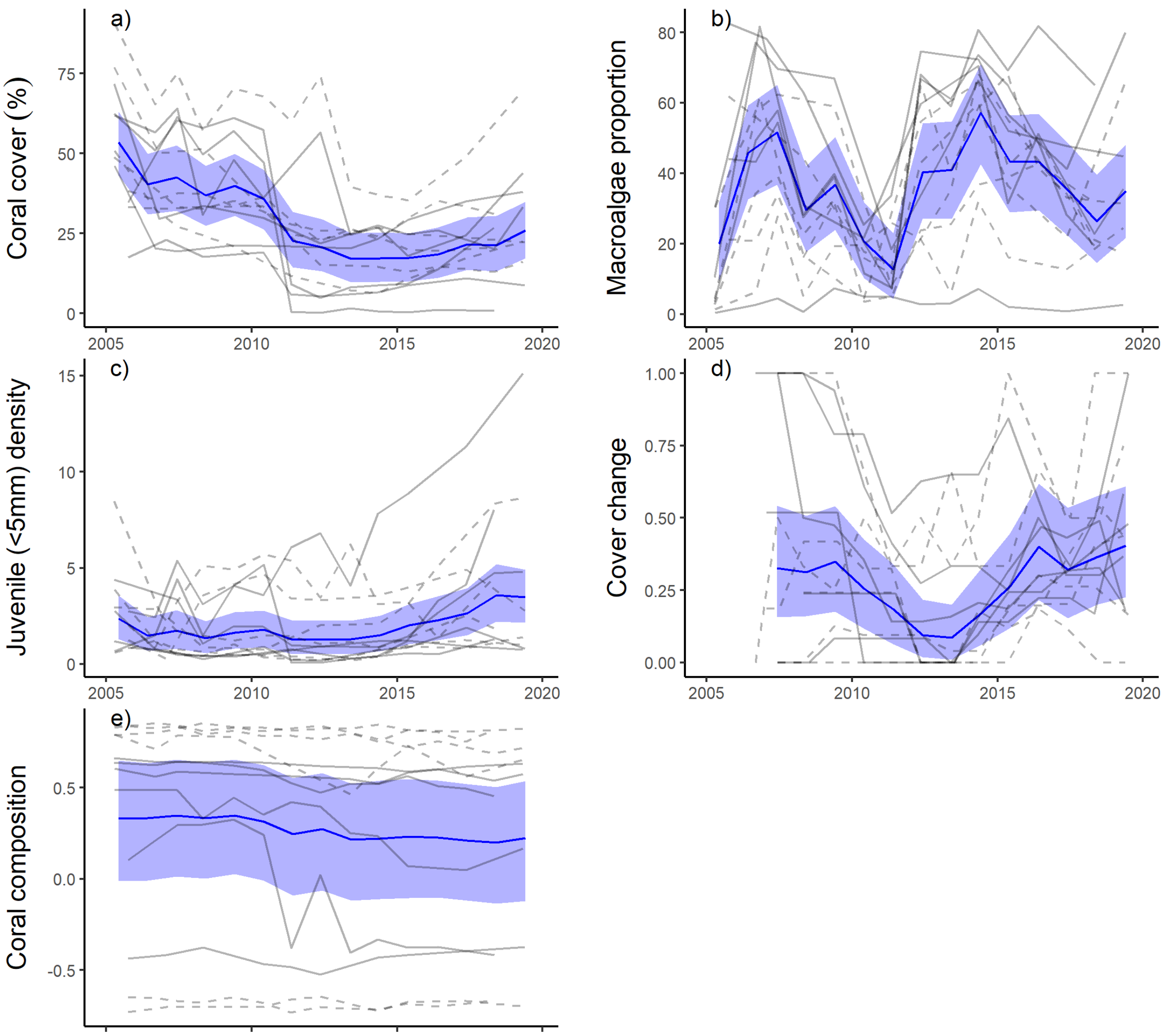

Figure 31 Fitzroy Region index and indicator trends. a– e) trends in individual indicators, (blue lines) bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5 m (dashed) and 2 m (solid) depths for individual reefs.

4.7 Response of coral communities to environmental conditions

4.7.1 Location along water quality gradients

Reef-wide index scores at 2 m depth show a negative relationship to both Chl *a* and NAP concentration in surrounding waters (Table 15, Figure 32a, Figure 33a). However, for both these water quality proxies it was only in the Wet Tropics that this relationship was statistically significant (Table 15). The low number of reefs in other regions will have reduced the precision of within region confidence intervals for observed relationships. It was clear, however that for most regions index scores in 2019 did not reflect relative exposure to pressures associated with water quality (Table 15, Figure 32a, Figure 33a). For the Mackay-Whitsunday region similarly low Chl *a* concentration among reefs limit the sensitivity of these analyses. Recent severe impacts associated with cyclone Debbie across will also confound the state of scores responses to water quality.

Table 15 Relationship between index and indicator scores and gradients in water quality in 2019. Tabulated values are upper and lower confidence intervals of the trend in scores for each combination of index or indicator score and depth for which Reef-wide relations ships between score and water quality proxies; Chl a and NAP where observed. Slopes for which confidence intervals did not include zero are shaded to highlight the direction of the relationship.

Response	Depth	Reef-wide		Wet Tropics		Burdekin		Mackay-Whitsunday		Fitzroy	
		l	u	l	u	l	u	l	u	l	u
Chlorophyll *a* concentration											
Index	2	-4.7	-0.5	-6.9	-0.2	-6.5	1.3	-8.1	19.1	-7.22	1.8
Coral cover score	2	-7.0	-1.4	-10.3	-1.7	-6.6	3.2	-11.8	22.8	-12.1	-0.6
Macroalgae score	2	-8.4	-1.0	-10.8	2.1	-13.8	0	-22.2	26.6	-12.3	3.1
	5	-7.6	-1.0	-8.2	3.2	-11.5	1.6	-25.8	4.1	-8.2	6.3
Juvenile score	5	2.7	8.8	3.8	12.9	0.8	12.3	-12.1	13.4	-5.1	8.1
Non-algal particulate concentration											
Index	2	-1.5	-0.2	--6.3	-0.5	-2.2	0.6	-1.9	2.9	-1.5	0.6
Coral cover score	2	-2.2	-0.5	-7.4	0.3	-2.1	1.6	-2.4	4.0	-2.7	0
Juvenile score	2	-2.1	-0.1	-4.5	5.0	-2.3	2.2	-3.2	4.3	-3.2	-0.1

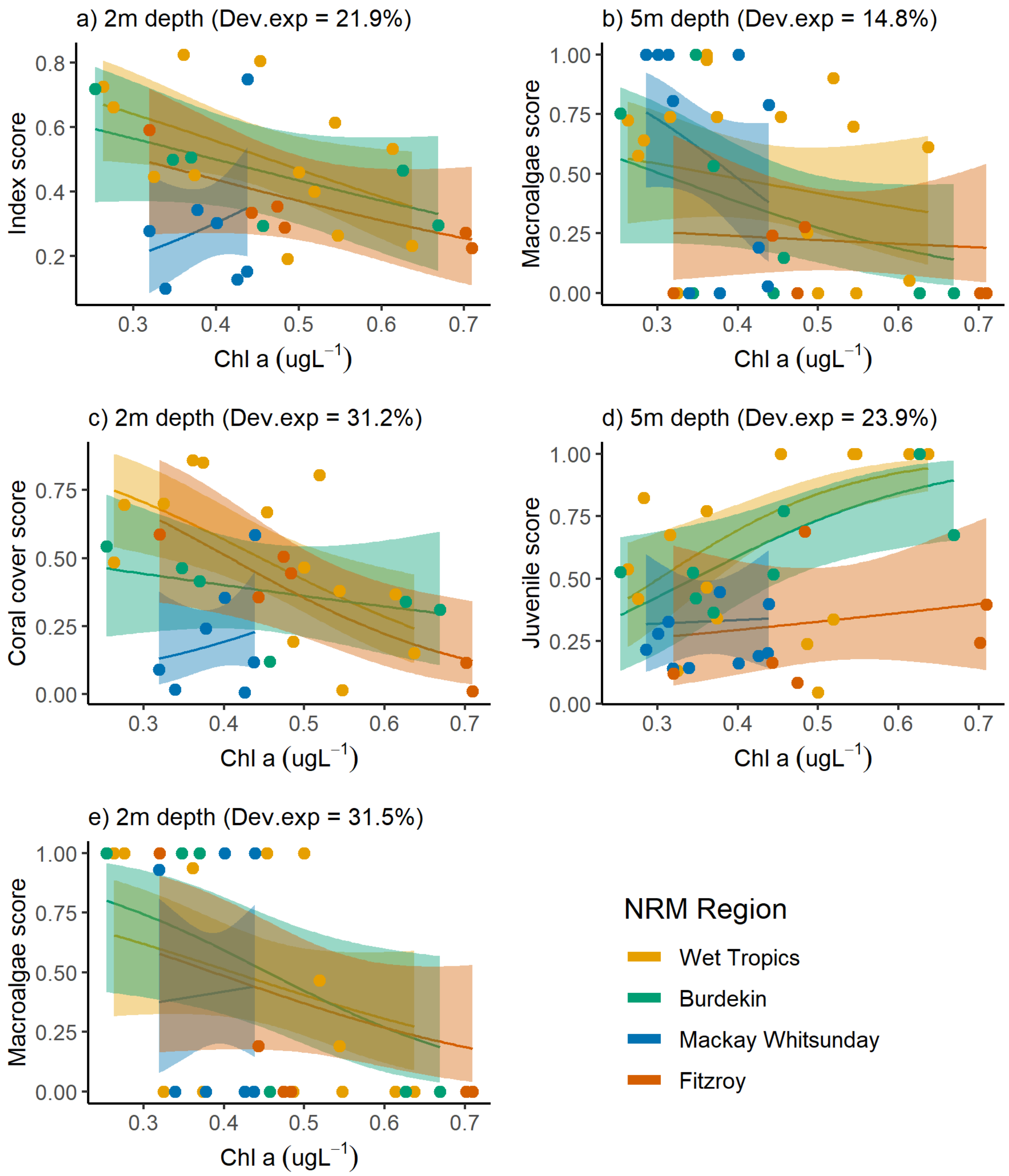

Figure 32 Coral index and indicator score relationships to Chl *a* concentration. Combinations of coral index or indicator and depth are included where Reef wide relationships were. Plots present predicted relationship within each region. Confidence intervals in predicated slopes are provided in (Table 15).

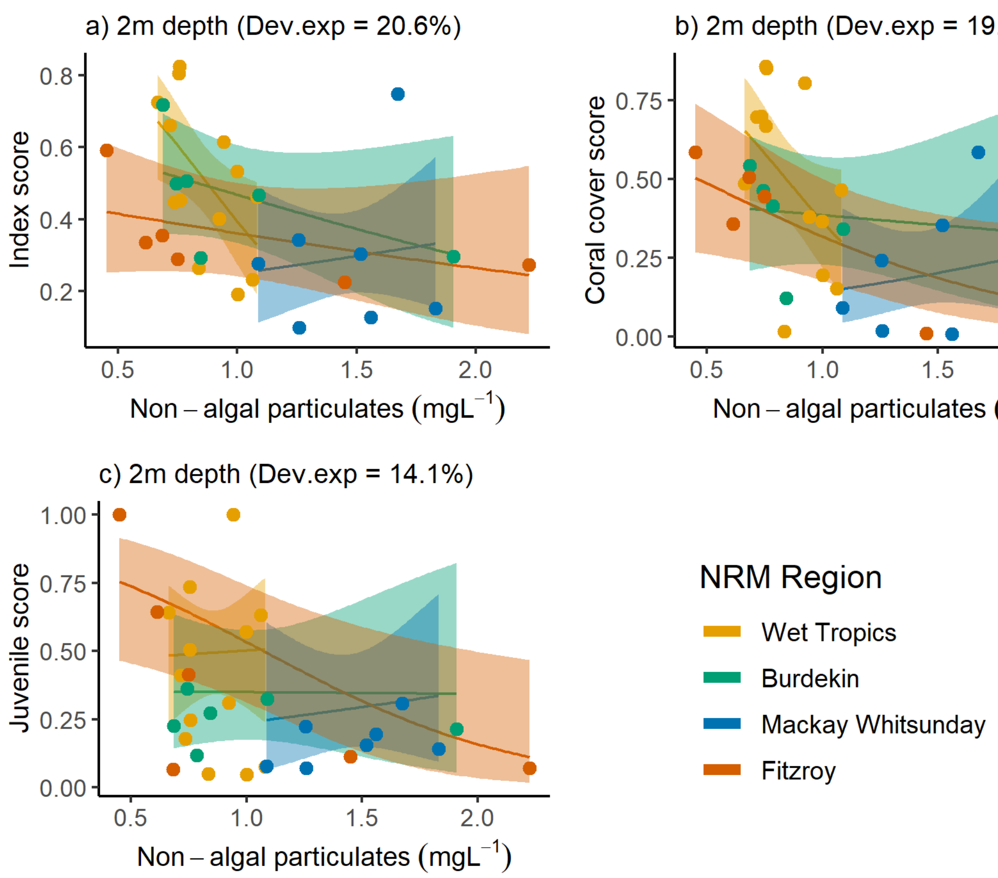

Figure 33 Coral index and indicator score relationships to NAP concentration. Combinations of coral index or indicator and depth are included where Reef wide relationships were. Plots present predicted relationship within each region. Confidence intervals in predicated slopes are provided in (Table 15).

Of the individual indicators:

- Scores for coral cover were negatively related to increasing concentration of both Chl *a* and NAP at 2 m depth, this relationship strongest for Chl *a*, in the Wet Tropics and Fitzroy regions (Table 15, Figure 32c, Figure 33b).
- Reef-wide, scores for macroalgae were negatively related to Chl *a* concentration at both 2 m and 5 m depths, however, although mean slopes were negative in all regions confidence intervals of predicted slopes did not exclude zero (Figure 32b, e, Table 15).

The juvenile indicator at 5 m depths was the only indicator for which Reef-wide scores were positively related to Chl *a* concentration. A relationship evident in both the Wet Tropics and Burdekin Regions (Table 15, Figure 32d). Very high densities of *Turbinaria* spp juveniles at reefs with relatively high Chl *a* concentration in the Herbert Tully sub-region and Burdekin Region are influential in these results (

- Figure A 12, Figure A 13).
- Reef-wide scores for the juvenile indicator at 2m depth were negatively related to NAP concentration a result driven by low densities of juvenile corals at the more turbid reefs in the Fitzroy region (Table 15, Figure 33c)
- Neither the cover change nor composition indicator scores were related to relative exposure to water quality pressures. This is not surprising as both indicators are scaled reflect change from baseline coral community compositions that include a higher proportion of slow growing species tolerant to poor water quality along water quality gradients.

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 algae cover classified as macroalgae shows a positive relationship to Chl *a* at both 2 m and 5 m depths, and NAP at 2 m depth (Table 16). Within regions, it is only in the Mackay-Whitsunday region that no relationship was observed between macroalgae representation in the algal community and either water quality proxy (Table 16). Relationships between macroalgal representation and Chl *a* were evident at 2 m depth in all other regions (Figure 34a). While mean trends were positive at 5 m depth confidence intervals about these trends demonstrate variability in all regions (Figure 34b, Table 16). Similarly, despite consistently positive trends between macroalgae representation and NAP lower confidence intervals of these trends were only positive in the Fitzroy region and at 2m depth in the Burdekin (Table 16, Figure 34c).

Table 16 Relationship between indicator values underpinning the macroalgae and composition indicator scores. Tabulated values are upper and lower confidence intervals of the trend in values for each combination of indicator value and depth. Slopes for which confidence intervals did not include zero are shaded to highlight the direction of the relationship.

Response	Depth	Reef-wide		Wet Tropics		Burdekin		Mackay-Whitsunday		Fitzroy	
		l	u	l	u	l	u	l	u	l	u
Chlorophyll *a* concentration											
Macroalgae proportion	2	3.9	9.2	0.9	10.0	2.9	13.4	-17	16.7	1.3	12.7
	5	1.4	5.7	-1.7	5.5	-0.1	8.1	-4.2	16.6	-0.9	7.3
Genus composition	2	-1.8	-0.5	-2.1	-0.2	-2.0	0.2	0.1	7.4	-3.5	-1.0
	5	-3.2	-1.6	-2.9	-1.0	-3.6	-1.1	-1.6	3.8	-5.9	-3.2
Non-algal particulate concentration											
Macroalgae proportion	5	0.4	1.7	-1.5	4.3	-0.1	2.6	-0.7	2.8	0.1	1.8
Genus composition	2	-0.6	-0.2	-1.7	-0.1	-0.7	0.1	0	1.3	-0.8	-0.3
	5	-0.9	-0.4	-2.2	-0.3	-1.1	0	-0.4	0.6	-1.4	-0.7

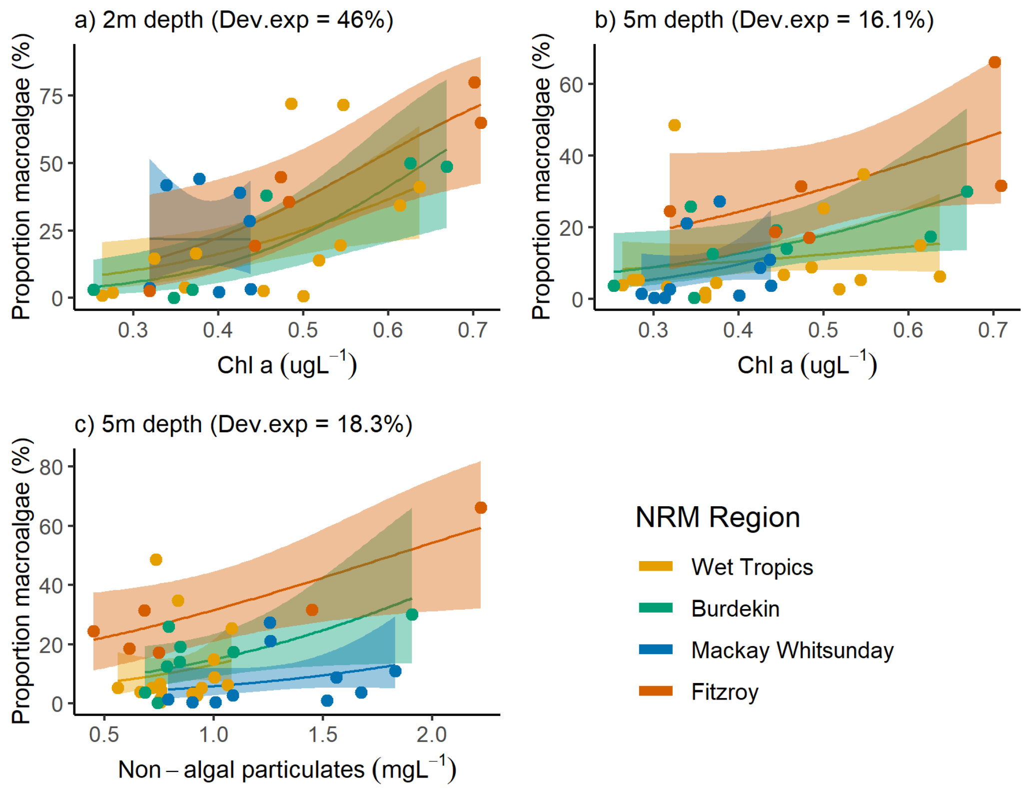

Figure 34 Relationship between proportions of algae cover classified as macroalgae and water quality proxies. Plots present predicted relationship within each region. Confidence intervals in predicated slopes are provided in (Table 15).

Reef-wide genus composition values derived from the product of genus level coral cover estimates and eigenvalues for the distribution of genera along WQ gradients (Table A 5), are negatively related to the water quality proxies at both 2 m and 5 m depths (Table 15). This estimation of community composition is based on a constrained ordination of the baseline condition of coral communities at the start of the MMP, positive eigenvalues are assigned to genera that occurred disproportionately at reefs with better water quality.

At 2 m depth community composition is negatively related to Chl *a* and NAP in both the Wet Tropics and Fitzroy regions and positively related to Chl *a* in the Mackay-Whitsunday region (Figure 35, Table 15). At 5 m depths these negative relationships are evident in all regions other than Mackay-Whitsunday (Figure 35, Table 15). Limiting the sensitivity of the analyses in the Mackay-Whitsunday region is the relatively short gradient in water quality compared to other regions.

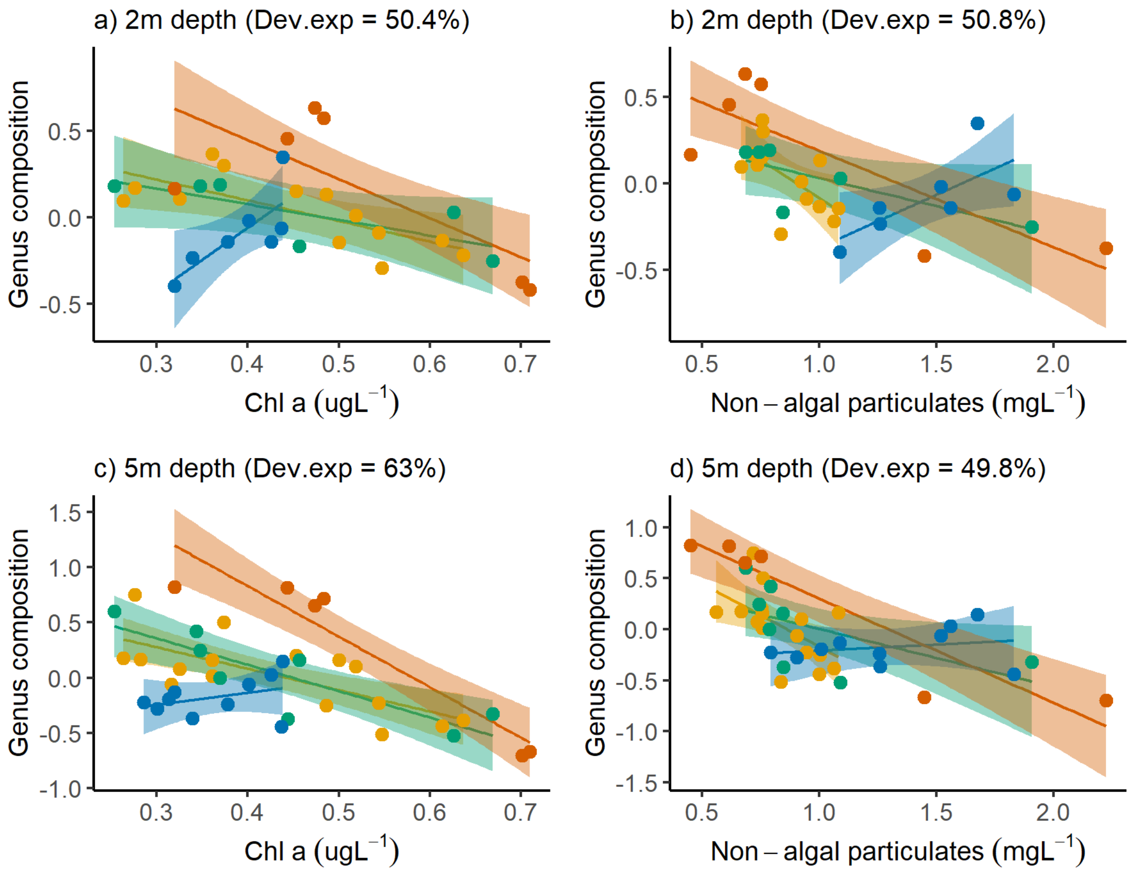

Figure 35 Relationship between coral community composition and water quality proxies. Plots present predicted relationship within each region. Confidence intervals in predicated slopes are provided in (Table 15). Colour coding for regional trends are consistent with those in above figures.

4.7.2 Influence of discharge, catchment loads and water quality on reef recovery

During periods free from acute disturbances (cyclones, thermal bleaching, crown-of-thorns starfish or direct exposure to low salinity floodwaters) the recovery of reefs, as measured by biennial change in index scores, were negatively related to discharge from the local catchments in three of the four regions monitored (Mackay-Whitsunday Region showed no relationship (Table 17, Figure 36). Importantly, these relationships consider only the contemporary influence of environmental conditions on the indicators during recovery periods. Any influence of water quality on the severity of response to disturbance events, or lagged responses of indicators will not be included. In the case of lagged influences, such as the initial decrease then post disturbance increase in macroalgae cover described above, this will result in the underestimation of the response.

For both the Wet Tropics and Fitzroy regions the trend was not completely monotonic, however changes where mostly positive at low discharge and negative at higher discharge (Figure 36). It is not surprising that relationships between Total P and Total N and coral index change were generally like those described for discharge as nutrient loads in rivers are correlated with river discharge.

Relationships between satellite derived water quality variables estimated from the waters adjacent to the coral monitoring sites; NAP, Chl *a* and PAR and changes in coral index scores were most evident in the Fitzroy Region and were consistently negative (Table 17). The strongest relationship between water quality on reef recovery in the Whitsunday Region was for river loads of dissolved inorganic nitrogen (DIN) that were not monotonic although the lowest recovery did correspond to high DIN load (Table 17, Figure 36). Similarly, the relationship between NAP and changes in coral index scores in the Mackay-Whitsunday Region was again not monotonic (Table 17) but demonstrated a clear decline in index score change at higher NAP exposure. Uncertainty in the covariate estimation, and necessity to derive estimates from waters approximately 1-3 km off the reefs that may not describe the exposure at the coral, limits the potential to uncover relationships between coral community responses and satellite derived environmental covariates.

Table 17 Relationship between changes in index scores and environmental conditions. Tabulated are the proportion of deviance explained by models fit to relationships between the time-series of index score changes during non-disturbance periods and summaries of environmental condition during those periods. Bolding indicates statistical significance, P-values<0.05 in bold, P-values between 0.05 and 0.1 not bold. Shading indicates the relationship was monotonic with higher increase in index scores at lower exposures to the environmental pressure. A (*) marks relationships that where not monotonic although either, the most negative index score changes were observed at high exposures, or most positive changes occurred at lower exposures. Blank cells indicate no relationship was observed with AICc not more than 2 units lower than null models.

Region	Freshwater Discharge	Total N river load	Total P river load	DIN Region load	Non-algal particulates (reef)	Chlorophyll (reef)	Light (reef)
Wet Tropics	**15.4%***	**16%**	**18.1%**				**3.3%**
Burdekin	**14.8%**	**15.5%***	**15.9%**			9.6%*	5.0%
Mackay-Whitsunday			5.3%	**17.2%***	**13.3%***		
Fitzroy	**30.1%***	**28%***	**27.7%***	**20.1%***	**6.1%**	**22.3%**	**8.8%**

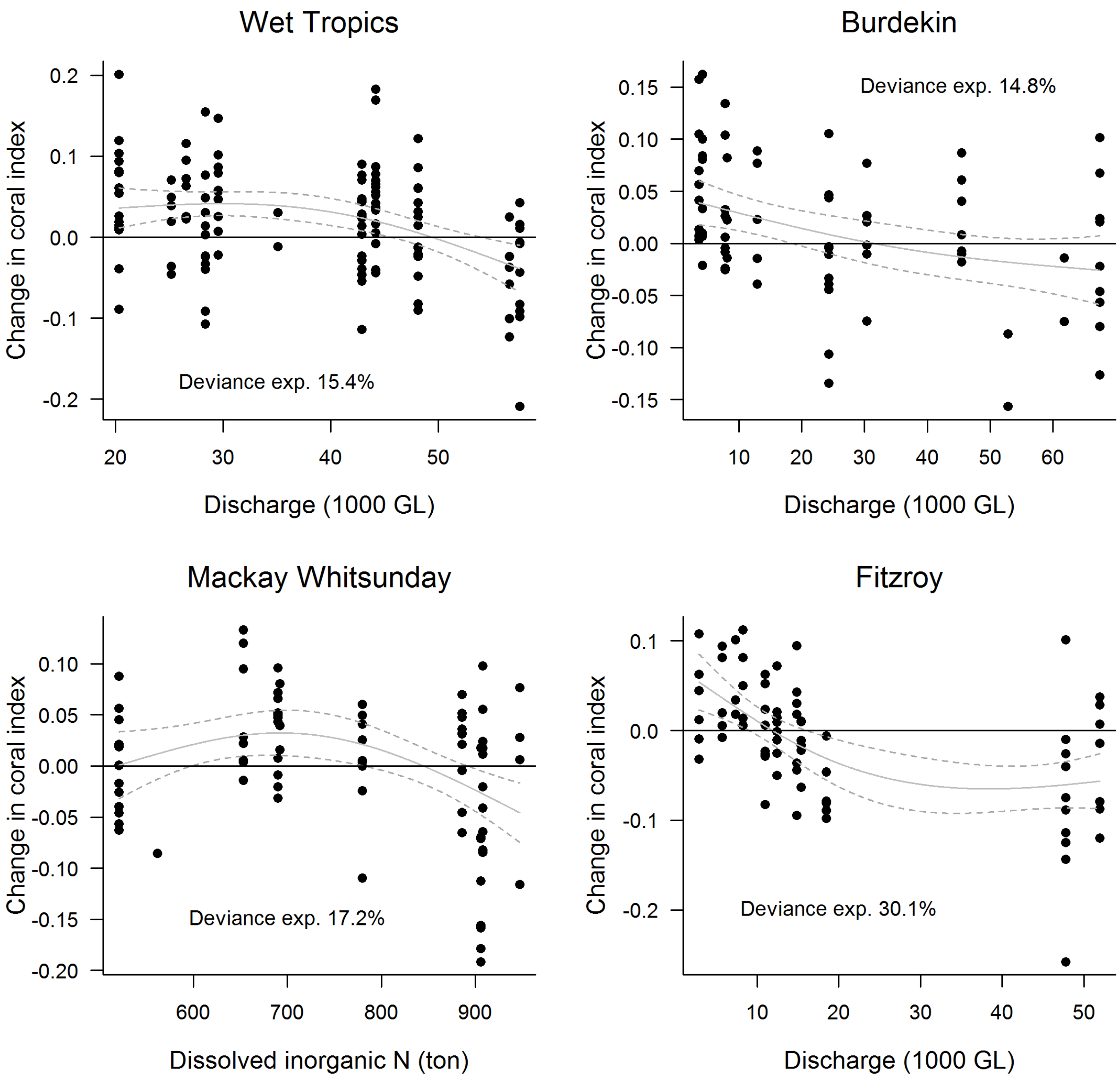

Figure 36 Relationship between the coral index and run-off from local catchments. Plotted points represent observed change in the index score at each reef and depth over a two-year period. Observations following years for which acute disturbances impacted communities in the period between samples were excluded. Discharge values represent the cumulative discharge from the region's major rivers over the two-year period corresponding to index changes. Trend lines represent the predicted change in index scores (solid line) and the 95% confidence intervals of the prediction (dashed lines)

5 Discussion

As naturally dynamic systems that alternate between impacts and periods of recovery (Connell 1978), it is critical for the persistence of coral communities that there is a long-term balance between disturbance and recovery processes. 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 condition, and management responses required to maintain a healthy Reef.

- 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 direct drivers such as global climate change. In this context, there is a distinction between pressures arising from climate change that are beyond the realm of management under Reef 2050 Water Quality Improvement Plan, such as acute disturbances associated with severe storms or thermal bleaching events, and those related more tangibly to water quality, and as such, expected to be 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 decisions for *response* such as policy or regulatory actions to alleviate that *impact*.

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 (48%) 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.

Loss of coral cover attributed to thermal stress leading to coral bleaching has caused 12.7% of coral cover loss; primarily during 2006 in the Fitzroy region and 2017 in the Wet Tropics region.

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 2011, and at Snapper Island in 2019. In combination these disturbance events contribute strongly to the declines in the coral cover (Lam *et al.* 2018) and index scores in all regions.

The maintenance of coral community condition requires that recovery process keep pace with the impact of disturbances. For the reef monitoring program, it is important that acute disturbances are identified and quantified primarily to allow the quantification of the resilience of communities to their cumulative impacts.

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 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 will interact. In short quantification of acute pressures will include the cumulative response of the identified pressure and any additional sensitivity of the coral community to that pressure as a result of local environmental conditions. Similarly, minor acute pressures that go unnoticed will potentially confound estimates of chronic pressure.

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

Such gradients are a natural part of the Reef ecosystem, albeit with lower levels of input of run-off-derived pollutants than presently occurs (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. It is the quantification of the compounding influence of run-off on the naturally occurring gradients, and any subsequent improvement under the Reef 2050 WQIP, that is the core focus of the water quality monitoring component of the MMP (see separate report by Gruber *et al*. 2020).

For corals, the pressure relating to land management practices is the 'state' of marine water quality, which in turn is influenced by the pressure of contaminant loads entering marine waters as run-off. The MMP river plume monitoring (see Gruber *et al.* 2020) clearly shows that inshore reefs are directly exposed to elevated loads of sediments and nutrients carried by flood plumes. 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 conditions. 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 loads and the loads delivered during more moderate river flows become a chronic pressure for inshore corals.

It is evident from the MMP marine water quality time-series, that there have been general increases in oxidised forms of dissolved nitrogen (NOx) and dissolved organic carbon (DOC). Lønborg *et al.* (2015) suggest that these observations indicate 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 results in reduced flow of carbon between the coral and symbionts that in turn increases their susceptibility to thermal stress and reduces energy required for recovery (Morris *et al*. 2019). Perversely, given energy supplied to the coral is in the form of DOC, elevated water column concentration of DOC has also been shown to stimulate nitrogen fixation by cyanobacteria within corals that further enhances the availability of N to algal symbionts (Pogoreutz *et al*. 2017). In general, the NOx concentrations observed 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). 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*. 2012a, Thompson *et al.* 2014a, 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).

5.2 Ecosystem State

5.2.1 Coral condition based on the index

In 2019 coral condition scores at two metre depths were inversely related to the concentration of Chlorophyll *a* (Chl *a*) and non-algal particulates (NAP) in surrounding waters. This relationship was not consistent among regions or the individual indicators, but most strongly influenced by negative relationships between coral cover and water quality in the Wet Tropics and Fitzroy regions. At five metre depth no overall relationship between index scores and Chl *a* level was observed, however macroalgae scores did decline with increasing Chl *a* concentration, although this relationship was, again, variable within regions. In contrast, juvenile scores increased with Chl *a* concentration, due largely to high recruitment of the *Turbinaria* at some high Chl *a* concentration reefs in the Herbert Tully sub-region and Burdekin region.

Stronger relationships between current scores and water quality gradients are limited by two primary factors. Firstly, the severe impact of disturbance events such as cyclone Debbie at some Mackay-Whitsunday, set low scores for indicators at reefs independent of their location along water-quality gradients. In addition, the relatively low variability in water quality conditions among some reefs likely reduces the scope for strong differentiation of condition. Compounding this lack of differentiation among sites is that satellite derived estimates of water quality are necessarily derived from open waters adjacent to the sampled reefs, assimilating estimates from waters ~ 1-3 km from the coral sites. Fine-scale hydrodynamic processes may divorce the conditions estimated in waters adjacent to the reefs from those experienced by the corals.

Secondly, the index has been designed to be responsive to change in environmental pressures and this required ensuring reef level scores for each indicator had the potential to either improve or decline. This desire for a responsive index required setting location-specific thresholds for scores of the cover change, composition and macroalgae indicators as water quality pressures unequivocally influence the underlying values of these indicators. Analysis of the values underpinning the macroalgae and composition indicators to reef-level water quality demonstrates both the higher proportion of macroalgae in algal communities with increased concentrations of Chl a and NAP, and changes in community composition that are driven by water quality.

Further, the single dimensional summaries of community composition reported were derived from the product of eigenvalues for each genus along water-quality gradients and the relative cover of those genera. Importantly, the 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 genus composition and water quality variables are indicative of reduced recovery rates of coral cover as water quality declines.

Temporal trends in coral index scores reflect the cumulative influence of multiple acute disturbances and the mediation of recovery because of environmental pressures. In all regions index scores reached a low point between 2012 and 2014 following a period of acute disturbances, but also high discharge and the associated nutrient and sediment loads from adjacent catchments. In all regions recovery was observed and the condition in 2019 reflects both the strength of this recovery but also the influence of more recent disturbance events.

In 2019 index scores varied among regions:

- The cumulative impacts of thermal stress in 2017, exposure to flooding of the Daintree River in 2019 and crown-of-thorns starfish resulted in a decline in scores in the Barron Daintree sub-region. The score in 2019 was only marginally higher than the 2014 low.

- Johnstone Russell-Mulgrave sub-region scores have varied about the threshold between ‘moderate’ and ‘good’ since 2015. Thermal bleaching in 2017 in combination with ongoing presence of crown-of-thorns starfish have limited increase in index scores.
- Herbert Tully sub-region score reached ‘good’ condition in 2019 following strong recovery from the ‘poor’ level observed in 2013. Thermal bleaching in 2017 caused a slight interruption to an otherwise consistent improvement.
- The index has regained a positive trajectory in the Burdekin Region following impacts attributed to thermal stress in 2017.
- Cyclone Debbie severely impacted reefs in the Mackay-Whitsunday region in 2017, resulting in a steep decline in index score from ‘good’ in 2016 to ‘poor’ in 2019.
- The index has continued to improve in the Fitzroy Region.

Acute disturbance events are primarily responsible for the loss of coral cover at most reefs (Lam *et al.* 2018). Our analysis of recovery periods, when reefs were not exposed to acute disturbances, demonstrates that the rate of change in index scores, which we consider as representative of community resilience, were negatively associated with catchment discharge in the Wet Tropics, Burdekin and Fitzroy regions. Similar relationships were evident for end-of-catchment loads of total phosphorus and total nitrogen in these regions. There was some evidence that the relationships between river inputs were realised as reduced light levels at the reefs in these regions and, in the Fitzroy region, negative relationships between Chl *a* and NAP concentrations.

The observed relationship between discharge and changes in the coral condition index implies that the cumulative impacts of river-delivered contaminants suppress the resilience of coral communities. We are mindful, however, that temporal responses of the index to water quality or discharge varied among reefs. This is expected as index scores at any point in space or time will reflect the cumulative responses of the communities to: past disturbance events and chronic pressures, selective pressures imposed by ambient conditions, and 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. What this means, is 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.* 2017) indices in the MMP, as tools to identify where, and when, environmental stress is occurring (Karr 2006, Crain *et al.* 2008). A potential way forward is to consider reef level responses within a decision tree framework that is explicitly aimed at identification of likely drivers of any observed lack of resilience (Flower *et al.* 2017).

In general, the spatial and temporal variability in 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). Failure to observe a clear relationship between discharge and change in the index scores in the Mackay-Whitsunday Region is likely due to the relatively low discharge but high tidal range 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 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 five metre 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 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.

Changes in index scores attributed to acute disturbances are also likely to be confounded by water quality pressures. In addition to reducing capacity for recovery, degraded water quality may also increase the susceptibility of corals to acute disturbance events. Evidence from recent research into the interactions between water quality and temperature suggests that coral's tolerance to heat stress is reduced by exposure to contaminants including nutrients, herbicides and suspended particulate matter (Wooldridge & Done 2004, Negri *et al*. 2011, Wiedenmann *et al*. 2013, Fabricius *et al.* 2013b, Wooldridge 2016, Bessell-Browne *et al*. 2017b, Morris *et al.* 2019), although during the widespread and severe 2016 thermal bleaching no such effect was detected (Hughes *et al.* 2017). With widespread thermal bleaching events impacting the Reef in 2016 and 2017 (Hughes *et al.* 2018) a likely precursor of increasingly frequent bleaching events (van Hooidonk *et al*. 2017) any interaction between water quality conditions and temperature on the fate of corals remains an ongoing concern. Similarly, the increased stress to corals in response to run-off, discussed above, may compound losses of coral cover attributed to cyclones, floods, or crown-of-thorns starfish.

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 2019, coral cover at 2 m depths was generally higher at reefs with low Chl a or NAP concentrations. High turbidity or nutrient levels do 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, Brown *et al.* 2010, Morgan *et al.* 2016) that reefs in highly turbid settings can support very high cover of species tolerant to those conditions. Despite claims for high diversity in turbid habitats based on aggregated diversity over a variety of microhabitats (Brown *et al.* 2010, Morgan *et al.* 2016), from sites that control for depth and exposure to wave energy, it is evident that as turbidity increases, high coral cover typically results from relatively few species tolerant of their local environment, particularly at deeper depths (De Vantier *et al.* 2006, Sweatman *et al.* 2007). These selective pressures will contribute to the observed negative relationship between coral cover and NAP as not only do they limit the number of species that can contribute to recovery of coral cover following disturbance events, they also limit the rate at which corals grow (see below). In combination this leads to prolonged recovery of coral cover and, on balance, lower cover in areas subject to higher water quality pressures.

5.2.3 Rate of change in coral cover

The cover change metric assesses the rate of change in coral cover (growth) during years free from acute disturbances. An adequate rate of coral cover increase is essential to ensure the long-term balance between cover lost to disturbances and that regained under ambient conditions. Within regions, the cover change indicator scores are often highly variable. Such variability is likely due to a combination of both sampling error and real responses as communities are differentially exposed to pressures in both space and time. The formulation of this metric includes the averaging of estimates over a four-year period intended to allow averaging over potential sampling error. Unfortunately, the move to a biennial sampling and the multiple disturbances recorded over the life of the program mean that the scores over a four-year period may represent estimates derived from a single observation of cover change. It was partly to account for this that the program adopted a contingent sampling design to ensure visitation of reefs following disturbances and so improve the data available from which to estimate scores for this indicator.

In 2019 cover change indicator scores were lowest in the Mackay-Whitsunday Region, where coral cover has shown no sign of recovery in the two years since cyclone Debbie. Of concern is that the cover change indicator score has been consistently low in the Mackay-Whitsunday Region and, in combination with communities dominated by slow growing species at five metre depths, suggests slow recovery is likely.

In contrast, the moderate to high scores for the cover change indicator in Burdekin and Wet Tropics in 2019 demonstrate the ongoing potential for recovery of coral communities especially those in less turbid waters. Of note is that ongoing presence of low densities of crown-of-thorn starfish at High Island and the Frankland Group are likely to have reduced the rate of increase in cover at these reefs meaning that the cover change score may be slightly underestimated. For the Fitzroy region although the indicator score remains poor, it has improved since 2014.

Over the period of the MMP, temporal trends in cover change indicator scores can be generalised as having declined to low points in the coral index between 2012 and 2014 and subsequently improved. Exceptions were, the Herbert Tully sub-region where both the index and 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 since 2017. The general decline in the cover change indicator scores coincided with a period during which high river discharge delivered 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 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 reduce growth rates, 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 scores improved suggesting a link between coral community recovery and catchment inputs. Interestingly, although coral recovery had improved, suggesting at least a partial release from chronic pressures, there is no clear signal for reduced concentrations of any of the monitored water quality variables in the surrounding waters. Rather, concentrations appear to have remained stable at elevated levels, or continued to increase.

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 foot-hold during relatively benign conditions only to be removed during periods when environmental conditions are beyond their tolerance.

In 2019, the composition indicator scores improved in all regions except in Mackay-Whitsunday, demonstrating that recovery of coral cover includes increased representation of species sensitive to poor water quality. In general, the coral community 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 clear reduction 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 the *Acropora pulchra*, a branching species, being killed by disease.

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 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.

In light of these realisations, consideration will be given to removing this indicator from the annually reported coral index and investing effort into the development of a less constrained method for the identification of changes in community composition, the cause and change could be investigated using post hoc analyses focused on likely drivers.

5.2.5 Macroalgae

Macroalgae generally benefit from increased nutrient availability due to run-off (e.g. Schaffelke *et al.* 2005). 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). Clear relationships between Chl *a* concentration, a proxy for nutrient availability, and the proportion of macroalgae 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, 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 2019, 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* may be a proxy for environmental variables or ecological processes other than the direct availability of nutrients that influence macroalgae abundance. 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) occurred on the LTMP survey reefs included in this report and are among the reefs toward the better end of the strong water quality gradient in inshore waters. The higher turbidity at most reefs surveyed under 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. 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 mechanism that limits 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 correspondence 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 reducing potential for long-term phase shifts. Alternatively, the persistence of macroalgae may be supported by density dependant feedbacks that help to maintain the macroalgal dominated state (Roff *et al*. 2015, Johns *et al.* 2018) in which case management of nutrient levels to mitigate initial blooms or management of blooms may provide scope for coral recovery.

5.2.6 Juvenile 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). That the juvenile 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.

Some of the highest densities of juvenile corals occur in the Herbert Tully and Burdekin (sub-)regions on reefs where the genus *Turbinaria* recruits in vast numbers. As this genus was not well represented in the adult community prior to successive cyclonic disturbances in 2006 and 2011, it is unclear whether this recruitment pattern is simply due to natural variability 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. This shift in species composition along environmental gradients has the potential to mask trends in sensitive species. A possible solution would be the development of a metric that includes consideration of community composition in addition to abundance of juveniles, or focused on a group, such as, *Acropora* that is important for recovery of coral communities (Fabricius *et al.* 2012).

In general, juvenile densities have increased at most reefs over several years following the major disturbances that led to low points in condition 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 density indicator were also very poor. Where this relationship is not evident higher juvenile 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 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.

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

5.3 Regional summaries

5.3.1 Wet Tropics

Index scores remain moderate in both the Barron Daintree and the Johnstone Russell-Mulgrave sub-regions and have improved to good in the Herbert Tully Region. There have been no severe disturbances in recent years although discharge from the region's rivers did cause some loss of coral in shallow water on reefs in close proximity to rivers over the 2018-19 wet season. Flooding may also have contributed to elevated levels of disease.

This is the only region in which crown-of-thorns starfish have been common on inshore reefs. In recent years the Great Barrier Reef Marine Park Authority's crown-of-thorns control program has helped to mitigate the impact of crown-of-thorns starfish[4] with 15,067 individuals removed from the monitoring reefs prior to 2019. There were no was no culling on inshore reefs in this region over the 2018-2019 financial year. MMP surveys in 2019 again noted low densities of crown-of-thorns starfish across a range of size classes demonstrating the ongoing risk posed by these coral predators.

Most reefs have demonstrated a clear potential for recovery with coral cover increasing during periods free from acute disturbance. However, persistently very poor scores for the macroalgae indicator at Bedarra, Dunk South and Snapper Island appropriately limit the regions over all overall index scores as at these reefs coral recovery is suppressed.

5.3.2 Burdekin

The coral index score for the Burdekin region improved in 2019 and remains 'moderate'. Only the juvenile indicator score did not improve in 2019, rather, has steadily declined since 2016. Much of the decline in juvenile score in recent years has been due to a return to more typical densities of *Turbinaria* juveniles following very strong recruitment in recent years at Lady Elliot and Havannah North and, to a lesser degree, Magnetic Island. Juvenile densities at Palms East have also declined noticeably since 2017.

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.

Exacerbating any supply-side limitation to coral recruitment is the persistently high cover of macroalgae at several reefs that is likely to further suppress recruitment success (Tanner 1995, McCook *et al.* 2001, Birrell *et al.* 2005, 2008a, b, Johns *et al.* 2018).

Across the region, the clear relationship between the representation of macroalgal species in the algal community and gradients in water quality at two metre depths provides strong evidence that nutrient levels are influencing the score for this indicator. 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 demonstrate the first evidence of coral recovery at this reef. However, the long-term persistence of high macroalgae is common within the region with scores for the macroalgae indicator remaining 'very poor' at all reefs other than Palms East, Palms West and

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[4] Australian Government crown-of-thorns starfish management programme data supplied by Great Barrier Reef Marine Park Authority, Eye on the Reef.

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

Havannah. Persistent cover of macroalgae implies high nutrient availability. Alternatively, the persistence of macroalgae may represent an alternate stable state whereby the biomass of algae that occupied space following severe disturbance to the coral communities has created density dependant feedbacks that help to maintain the macroalgal dominated state (Roff *et al*. 2015, Johns *et al.* 2018) in which case management of nutrient levels to mitigate initial blooms or management of blooms may provide scope for coral recovery.

The influence of water quality on reef resilience is also demonstrated by the inverse relationship between discharge from the Region's rivers and the rate of improvement in the coral index. 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.

In addition to generally low rates of cover increase, stress to corals during periods of high catchment discharge were observed as increased levels of disease 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 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 discharge for the first time since 2013.

5.3.3 Mackay-Whitsunday

The coral index in the Mackay-Whitsunday Region has sharply declined to a 'poor' rating as the full extent of cyclone Debbie's impact was realised. Three reefs in the region monitored by the LTMP have been surveyed for the first time since the passage of cyclone Debbie. All three reefs were heavily impacted with substantial losses in coral cover and reduced juvenile densities, driving the decline in scores for the associated indicators in 2019. Reduced scores in 2019 also capture the ongoing loss of cover at Dent Island, Pine Island, and two metre depth at Seaforth Island.

The only indicator to remain in moderate condition in 2019 was macroalgae. Prior to cyclone Debbie most reefs in the region had low cover of macroalgae; it was only Seaforth and Pine islands where macroalgae were present at levels sufficient to reduce indicator scores below the 'very good' range. At these reefs macroalgae was removed by cyclone Debbie resulting in an improvement in the macroalgae score in 2017. At both reefs macroalgae cover had returned to pre-cyclone levels in 2019. Driving the macroalgae indicator lower has been the colonisation of macroalgae at Daydream and Double Cone islands.

Initial increase in macroalgae cover following disturbances is not uncommon as algae quickly establishes on the available space following the loss of coral (McManus & Plosenberg 2004). The post cyclone algal community at Double Cone was dominated by a mix of small red algae species that formed a thick mat over the substrate, especially at two metre depth. Although cover of macroalgae had declined in 2019 the algal community had shifted to include a higher proportion of brown algal species including *Sargassum* and *Lobophora* that, once established, have proven persistent at other reefs monitored by the MMP. Although still in relatively low abundance in 2019, the appearance of these large brown macroalgae species is concerning. Once established, high macroalgal cover can lead to feedback loops that constrain coral recovery, potentially trapping benthic communities in a macroalgal dominated state (Mumby *et al.* 2013, Johns *et al*. 2018).

Environmental conditions will play a large role in determining the future of these coral communities. 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 that is clearly illustrated by the marked differences in coral community composition between two metre and five metre depths at most reefs. Although, these species are tolerant of the highly turbid conditions, they tend to be slow growing. Despite the cover change indicator score being calibrated to account for the slower growth of these species, a feature of coral community dynamics in this region has been consistently low scores for cover change. The combination of low rates of cover increase for slow growing species implies recovery will be slow, especially at the five metre depths.

Despite the clear pressures imposed by the environmental conditions, 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 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 index through to 2012. Improvement in the coral index 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 incurred during cyclone Debbie.

The decline in the short-term water quality index (Gruber *et al*. 2020) captures anecdotal observations from commercial users, suggesting high turbidity persisted for several months in the aftermath of cyclone Debbie. At the time of coral surveys in July 2017 turbidity was noticeably high and sedimentation to the substrate was ongoing. It is highly likely that these conditions precipitated the further loss of coral cover observed at reefs resurveyed in 2018. Both the long-term and short-term water quality indices demonstrate ongoing poor water-quality across the region.

Given past observations of low scores for the cover change indicator, the unsuitable nature of the substrate for coral settlement (Ricardo *et al.* 2017), and the regionally reduced brood-stock, a slow recovery of coral communities at the worst impacted reefs appears likely.

5.3.4 Fitzroy

The coral index continued to improve in 2019, although coral condition remains poor. 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 that exposed corals to lethally low levels of salinity (Jones & Berkelmans 2014) and introduced high loads of nutrients and suspended sediments into Keppel Bay. These pressures substantially reduced coral cover across the region through to 2014. The recovery from these pressures was limited by high water temperatures in 2016 and 2017. While not resulting in substantial loss of cover, high temperatures did result in coral bleaching (Kennedy 2018) and are likely to have reduced the rate of coral cover increase.

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 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) that may have supressed the recovery of coral cover.

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 spp.) rapidly increased and persisted at high densities; at the same time the rate of change in coral cover remained low or coral cover continued to decline. In contrast, at Barren, where Chl *a* concentration is lower, the *Lobophora* bloom was less pronounced and recovery of the coral community clearly progressed.

There is clear evidence that the abundance of macroalgae on the Reef is higher where mean Chl *a* concentration (as a proxy for nutrients) is above the annual guideline values for coastal and mid-shelf waters of 0.45µgL^{-1} (De'ath & Fabricius 2008, Thompson *et al.* 2017). In 2019 scores for the macroalgae indicator did improve at some reefs where proportional cover declined, however, regionally the scores for this indicator remain very poor, and macroalgae continue to limit the recovery of coral communities.

A bottleneck for recovery of coral communities is the low density of juvenile corals. Although the juvenile density indicator scores continue to improve relative to the low scores of 2014, scores in 2019 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. What is evident, is that benthic communities at several reefs appear locked in a high macroalgae-low coral state.

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 the coral losses and subsequent recovery. The breakdown of causes of coral loss and relationships between recovery and environmental conditions provide emerging from the MMP provide guidance for management.

In terms of limiting of coral loss the Authority's crown-of-thorns control program has helped to mitigate the impact of crown-of-thorns starfish in the Wet Tropics Region. The small size and isolation of many inshore reefs may make such controls particularly feasible. MMP surveys in 2019 again noted low densities of crown-of-thorns starfish across a range of size classes demonstrating an ongoing pressure warranting management consideration.

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 ongoing efforts to reduce nutrient loads entering the reef are warranted so as to reduce the potential for the initial establishment of these algae. It must be noted however, that the environment occupied by many macroalgae is still suitable for corals and it is potentially density dependant feed backs that maintain these high covers 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.

That time-series of coral recovery are negatively related to the loads of nutrients and or sediment entering the Reef in most regions, which supports the management of regional activities associated with catchment development (including agriculture through the Reef 2050 Water Quality Improvement Plan). 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 global stressors such as climate change (Bellwood *et al.* 2004, Marshall & Johnson 2007, Carpenter *et al.* 2008, Mora 2008, Hughes *et al*. 2010).

Benthic communities in inshore areas of the Reef show clear responses to gradients in water quality (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.* 2012a 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).

6 Conclusion

The cumulative impacts of tropical cyclones and storms, predation by crown-of-thorns starfish, thermal stress and exposure to low salinity flood plumes has clearly impacted the condition of inshore reefs (Lam *et al.* 2018). Compounding the impact of these acute events are the chronic pressures of water quality that operate both spatially along gradients in water quality and temporally in response to variability in the loads of sediments and nutrients delivered by rivers. These chronic pressures supress the recovery of coral communities following 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 or other changes to 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 of climate change and other human activities (Steffen *et al*. 2013, Halpern *et al.* 2015, Hughes *et al.* 2018), the focus on supporting recovery in a climate of increasing disturbance is ever-sharpening (GBRMPA 2019).

Disentangling the influence of run-off in the observed declines in coral condition, or the ability of communities to recover, remains difficult for several reasons. Firstly, coral response-thresholds to the cumulative pressures associated with water quality are likely to be spatially variable because of the selection and acclimatisation of corals in response to location-specific conditions. Secondly, extrinsic variability, along with low concentrations of many constituents of water quality, limits the ability to quantify additional pressures resulting from run-off at scales relevant to the communities monitored. Finally, effects of interactions between water quality stressors have only been quantified for a limited combination of pressures and few coral species (e.g. Uthicke *et al.* 2016). The response of coral reef communities to the cumulative impacts of the range of sediment, nutrient and pesticide contaminants carried by rivers, along with climate change, is still only semi-quantitatively understood (although it is unlikely that pesticides are reaching our monitored coral sites at concentrations that are known to cause any impact). 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 index scores) does identify both spatial and temporal responses of coral communities to variation in water quality.

Spatially, results from this project reiterate that macroalgae abundance is enhanced, to the detriment of corals, in areas exposed to the chronic pressure of high nutrient availability (Fabricius *et al.* 2005). Temporally, the recovery of coral communities, assessed as rate of increase in index scores, also shows a negative relationship to river discharge and the corresponding loads of sediments and nutrients carried therein. In combination these results highlight the detrimental influence that the chronic pressure of water quality has on recover of coral communities following inevitable exposure to acute pressures.

As the time series for the MMP increases some pertinent observations relating to the balance between disturbance regimes 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 high run-off. This indicates the pressures imposed on community recovery by increased loads of sediments and/or nutrients from the adjacent catchments. On balance, condition index scores have returned to those observed at the beginning of the project. However, in 2005 the condition of some reefs in these regions was poor and as such the 2005 condition may not be an appropriate aspirational baseline.
- At reefs where high levels of macroalgae become established the recovery of coral communities can be stalled. There is potential that the density-dependant feedbacks that promote the ongoing persistence of macroalgae have resulted in phase shifts at these reefs.

The effect of these phase shifts is that although high nutrient levels may have promoted the initial bloom of macroalgae, the persistence of macroalgae will reduce the sensitivity of measures of recovery to variations in water quality. As a result, the strength of relationship between changes in index scores and environmental variability may be underestimated.

- In the Mackay-Whitsunday Region high turbidity coupled with the sheltered nature of many reefs promote challenging conditions for most corals at deeper sites. Despite these conditions large colonies of tolerant species are found. The magnitude of impact from cyclone Debbie is unprecedented in the monitoring time-series from this region. It will be informative to observe how these communities recover. Data to date suggest that low juvenile densities and low rates of cover increase will result in slow recovery of these communities. Of concern is the colonisation by macroalgae at some reefs and the persistence of these algae should be a focus of monitoring in the medium term.

While the results presented here do not provide clear guidance in terms of load reductions required to improve coral condition in the inshore Reef, they do support the premise of the Reef 2050 WQIP 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 expected increase in disturbance frequency, reinforces the importance of reducing local pressures to support the long-term maintenance of these communities.

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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 Source of river loads time series used for analysis

Basin	Gauging station	Site name	Years available
Barron	110001D	Barron River at Myola	2008-2018
Johnstone	112101B	South Johnstone River at Upstream Central Mill	2007-2018
Tully	113006A	Tully River at Euramo	2007-2018
Burdekin	120001A	Burdekin River at Home Hill	2007-2018
Pioneer	125013A	Pioneer River at Dumbleton Pump Station	2007-2018
Fitzroy	1300000	Fitzroy River at Rockhampton	2007-2018

Table A 3 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 4 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 5 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 *	1	0
Turbinaria	-0.279	-0.307	Ctenactis *	0.016	1
Goniopora	-0.320	-0.304	Anacropora *		1
Goniastrea	-0.115	-0.278	Physogyra	0	1
Pachyseris	-0.077	-0.235	Cynarina *	-0	4
Favites	-0.096	-0.230	Sandalolitha*	3	5
Alveopora	-0.076	-0.221	Montastrea	0.019	5
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	-2	-0.11	Montipora	-0.131	0.045
Moseleya *	-0.058	-0.091	Leptastrea *	0.022	0.048
Oxypora	-8	-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	7	-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	3	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 *	2	-0.021			
Polyphyllia *	0	-0.020			
Heliofungia	0.015	-7			
Catalaphyllia *	-2	-6			
Stylocoeniella *	4	-6			
Pseudosiderastrea *	-1	-6			
Gardineroseris *	-4				
Submassive Porites	-0.047	-5			
Submassive Acropora	0.043	-4			
Halomitra *		-2			
Plerogyra	2	-1			
Lithophyllon*		-1			
Tubastrea*	5	-0			

Table A 6 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 Gruber et al. (2020). 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
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	3.1
	Mossman River	858320	1.5	1	1.1	0.9	1.3	1.7	1.3	1	1.6	0.7	1	0.9	1.2	2.2
	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
	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
	Johnstone River	3953262	1.2	1.1	1	1.1	1	2	1.1	0.8	1.1	0.6	0.7	0.8	1.3	1.2
	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
	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
	Herbert River	3556376	1.2	1.2	1	2.9	1	3.5	1.3	0.9	1.2	0.3	0.5	0.6	1.8	1.6
Burdekin	Black River	208308	1	2.2	2.5	4.6	2.2	5.5	3.2	0.8	1.8	0.1	0.5	0.3	1.9	4.6
	Ross River	261907	0.7	1.1	1.2	3.3	1.3	3.5	1.1	2.8	0.4	0.1	0.1	0.1	0.2	6.3
	Haughton River	419051	1.1	2.2	3.3	4.4	2.1	4.7	3.2	1	1	0.3	0.5	0.7	1.4	5.6
	Burdekin River	4406780	0.5	2.2	6.2	6.7	1.8	7.9	3.5	0.8	0.3	0.2	0.4	0.9	1.3	4
	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	2.1	0.6	2.7
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	2.9
	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
	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
	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	2.5	0.2	1.1
Fitzroy	Water Park Creek	97115	0.2	0.5	2.5	1	2.8	4.8	1.5	5.2	2.9	2	1.8	2.7	1.4	0.7
	Fitzroy River	2852307	0.2	0.4	4.4	0.7	4.1	13.3	2.8	3	0.6	0.9	1.3	2.2	0.3	0.5

Table A 7 Disturbance records for each reef. Tabulated losses of coral cover are calculated using the methods described in section 2.5.5 of this report and represent the proportion of hard coral lost compared to projected cover based on previous observations as opposed to reduction in observed cover that does not account for expected increase in cover because of growth between surveys. * represent cases where bleaching was the likely primary cause of loss although other factors may have contributed, ** bleaching likely however impact confounded by other severe disturbance. Bleaching events that occurred beyond the span of available time-series indicated by n/a.

Sub-Region		Reef	Bleaching 1998	Bleaching 2002	Bleaching 2017	Other recorded disturbances
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 2009 (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
Barron	Daintree	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)
Barron	Daintree	Low Islets				COTS 1997-1999 (69%), Multiple disturbances (Cyclone Rona, crown-of-thorns) 1999-2000 (61%), Multiple disturbances (Cyclone Yasi, bleaching and disease) 2009-2011 (23%), COTS 2013-2015 (38%),
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
Johnstone	Russell-Mulgrave	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)
Johnstone	Russell-Mulgrave	Fitzroy West LTMP	12%			COTS and continued bleaching 2000 (80%), COTS: 2013 (6%), 2014-15(46%)
Johnstone	Russell-Mulgrave	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
Johnstone	Russell-Mulgrave	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
Johnstone	Russell-Mulgrave	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)
Johnstone	Russell-Mulgrave	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)
Johnstone	Russell-Mulgrave	Green			12 %	COTS: 1994 (21%), 1997 (55%), 2011-2013 (44%), 2014-2015 (47%)

Table A 4 continued

Sub-Region	Reef	Bleaching			Other recorded disturbances
		1998	2002	2017	
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)
	King Reef	0.93	0.85	n/a	Cyclone Larry 2006 (56% at 2 m,50% at 5 m), Cyclone Yasi 2011 (71% at 2 m, 37% at 5 m)
	Dunk North	0.93	0.80	18% (2 m) 16% (5 m)	Cyclone Larry 2006 (81% at 2 m, 71% at 5 m), Disease 2007 (34% at 2 m), Cyclone Yasi 2011 (93% at 2 m, 75% at 5 m)
	Dunk South	0.93	0.85	45% (2 m) 6% (5 m)	Cyclone Larry 2006 (23% at 2 m, 19% at 5 m), Cyclone Yasi 2011 (79% at 2 m, 56% at 5 m), Bleaching 2018 (28% at 5 m)
	Bedarra	n/a	n/a	36% (2 m) 10% (5 m)	Bleaching 2018 (26% at 5 m)

Table A 4 continued

Region	Reef	Bleaching			Other recorded disturbances
		1998	2002	2017	
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		Cyclone Yasi 2011 (86% at 2 m, 45% at 5 m)
	Pandora Reef	0.93 (21%)	0.85 (2%)	33% (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 %*	Cyclone Yasi 2011 (25%)
	Havannah	0.93	0.95	37% (2 m) 11% (5 m)	Combination of Cyclone Tessie and Crown-of-thorns 1999-2001 (66%) Cyclone Yasi 2011 (35% at 2 m, 34% at 5 m), Disease 2016 (9% at 2 m), Bleaching 2018 (26% at 2 m, 16% at 5 m), Disease 2019 (23% at 2 m)
	Havannah North	49%	21%		Cyclone Tessie 2000 (54%), 2001 COTS (44%) Cyclone Yasi 2011 (69%)
	Middle Reef LTMP	(7%)	(12%)	n/a	Flood 2009 (20%)
	Magnetic	0.93 (24%)	0.95 (37%)	32% (2 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)

Table A 4 continued

Region	Reef	Bleaching 1998	Bleaching 2002	Bleaching 2017	Other recorded disturbances
Mackay-Whitsunday	Hook	0.57	1		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)
	Seaforth	0.57	0.95	**	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	**	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	**	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	**	Cyclone Ului 2010 (8% at 2 m), Cyclone Debbie 2017 (48% at 2 m, 55% at 5 m)
	Pine	0.31	1	**	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 4 continued

Region	Reef	Bleaching			Other recorded disturbances
		1998	2002	2006	
Fitzroy	Barren	1	1	25% (2 m) 30% (5 m)	Storm Feb 2008 (43% at 2 m, 24% at 5 m), Storm Feb 2010 plus disease (25% at 2 m,8% at 5 m), Storm Feb 2013 (51% at 2 m, 48% at 5 m)), Storm Feb 2014 (18% at 2 m and at 5 m), Cyclone Marcia 2015 (45% at 2 m, 20% at 5 m)
	North Keppel	1 (15%)	0.89 (36%)	61% (2 m) 41% (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) 38% (5 m)	Storm Feb 2010 plus disease (29% at 2 m, 42% at 5 m) Cyclone Marcia 2015 (30% at 2 m, 32% at 5 m)
	Keppels South	1 (6%)	1 (26%)	27% (2 m) 28% (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)
	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)
	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 8 Reef level Coral index and indicator scores 2019. Coral index and (sub-)regional indicator scores are colour coded by condition categories: red = very poor, orange = poor, yellow = moderate, light green = good and dark green = very good.

Region	Reef	Depth	Coral Cover	Juvenile	Macroalgae	Cover Change	Composition	Coral index
Barron Daintree	Low Isles	5	0.46	0.68	0.74	1.00	0.50	0.68
	Snapper North	2	0.19	0.05	0.00	0.71	0.00	0.19
		5	0.33	0.24	0.25	0.50	0.00	0.26
	Snapper South	2	0.47	0.08	1.00	0.76	0.00	0.46
		5	0.81	0.05	0.00	0.54	1.00	0.48
Report Card Score - Moderate			0.45	0.22	0.40	0.70	0.30	0.41
Johnstone Russell-Mulgrave	Green	5	0.19	0.82	0.64	0.62	0.50	0.56
	Fitzroy East	2	0.49	0.64	1.00	1.00	0.50	0.73
		5	0.56	0.54	0.72	1.00	0.00	0.56
	Fitzroy West	2	0.86	0.51	0.94	0.83	1.00	0.83
		5	0.68	0.47	0.98	1.00	0.50	0.72
	Fitzroy West LTMP	5	0.53	0.77	1.00	0.50	0.50	0.66
	Franklands East	2	0.70	0.41	1.00	0.70	0.50	0.66
		5	0.61	0.42	0.58	0.45	1.00	0.61
	Franklands West	2	0.70	0.18	0.00	0.85	0.50	0.45
		5	0.62	0.13	0.00	0.50	1.00	0.45
	High East	2	0.85	0.25	0.00	0.66	0.50	0.45
		5	0.80	0.34	0.74	1.00	1.00	0.78
	High West	2	0.81	0.31	0.47	0.42	0.00	0.40
		5	0.43	0.34	0.90	0.57	0.50	0.55
Report Card Score - Moderate			0.63	0.44	0.64	0.72	0.57	0.60
Herbert Tully	Barnards	2	0.67	0.74	1.00	0.62	1.00	0.81
		5	0.71	1.00	0.74	0.80	1.00	0.85
	Dunk North	2	0.38	1.00	0.19	1.00	0.50	0.61
		5	0.35	1.00	0.70	0.77	0.50	0.66
	Dunk South	2	0.37	0.57	0.00	0.73	1.00	0.53
		5	0.50	1.00	0.05	0.71	0.50	0.55
	Bedarra	2	0.15	0.63	0.00	0.15	NA	0.23
		5	0.32	1.00	0.61	1.00	NA	0.73
Report Card Score – Good			0.43	0.87	0.41	0.72	0.75	0.64
Burdekin	Palms East	2	0.54	0.23	1.00	0.82	1.00	0.72
		5	0.62	0.53	0.75	1.00	1.00	0.78
	Palms West	2	0.46	0.36	1.00	0.67	0.00	0.50
		5	0.45	0.42	1.00	0.43	0.50	0.56
	Havannah North	5	0.28	0.52	0.00	0.77	1.00	0.52
	Havannah	2	0.42	0.12	1.00	0.00	1.00	0.51
		5	0.52	0.36	0.53	1.00	1.00	0.68
	Pandora	2	0.12	0.27	0.00	0.57	0.50	0.29
		5	0.15	0.77	0.15	0.37	1.00	0.49
	Pandora North	5	0.77	0.52	0.00	0.31	0.00	0.32
	Lady Elliot	2	0.34	0.33	0.00	0.66	1.00	0.47
		5	0.55	1.00	0.00	0.65	0.00	0.44
	Magnetic	2	0.31	0.22	0.00	0.45	0.50	0.30
		5	0.50	0.67	0.00	0.53	0.50	0.44
	Middle Rf LTMP	2	0.52	0.54	0.00	NA	0.50	0.39
Report Card Score – Moderate			0.44	0.46	0.36	0.59	0.63	0.50

Table A 5 continued

Region	Reef	Depth	Coral cover	Juvenile	Macroalgae	Cover Change	Composition	Coral index
Mackay-Whitsunday	Hayman	5	0.13	0.22	1.00	0.20	0.00	0.31
	Langford	5	0.20	0.28	1.00	0.00	0.00	0.30
	Border	5	0.47	0.33	1.00	0.47	0.00	0.45
	Hook	2	0.09	0.08	0.93	0.28	0.00	0.28
		5	0.30	0.14	0.81	0.13	0.50	0.38
	Double Cone	2	0.02	0.07	0.00	0.40	0.00	0.10
		5	0.24	0.14	0.00	0.16	0.00	0.11
	Daydream	2	0.01	0.20	0.00	0.43	0.00	0.13
		5	0.04	0.19	0.19	0.39	0.00	0.16
	Dent	2	0.35	0.16	1.00	0.00	0.00	0.30
		5	0.40	0.16	1.00	0.00	0.00	0.31
	Shute Harbour	2	0.59	0.31	1.00	0.85	1.00	0.75
		5	0.25	0.40	0.79	0.46	1.00	0.58
	Pine	2	0.12	0.14	0.00	0.00	0.50	0.15
		5	0.18	0.20	0.03	0.00	0.00	0.08
	Seaforth	2	0.24	0.22	0.00	0.25	1.00	0.34
		5	0.19	0.45	0.00	0.10	1.00	0.35
Report Card Score – Poor			0.22	0.22	0.51	0.24	0.29	0.30
Fitzroy	Barren	2	0.59	1.00	1.00	0.37	0.00	0.59
		5	0.93	0.12	0.00	0.75	0.50	0.46
	North Keppel	2	0.51	0.07	0.00	0.20	1.00	0.35
		5	0.30	0.08	0.00	0.44	0.50	0.26
	Middle	2	0.45	0.41	0.00	0.58	0.00	0.29
		5	0.22	0.69	0.28	0.44	0.00	0.32
	Keppels South	2	0.36	0.64	0.19	0.48	0.00	0.33
		5	0.42	0.16	0.24	0.17	0.50	0.30
	Pelican	2	0.01	0.11	0.00	1.00	0.00	0.22
		5	0.28	0.40	0.00	1.00	0.50	0.43
	Peak	2	0.12	0.07	0.00	0.18	1.00	0.27
		5	0.30	0.24	0.00	0.00	0.50	0.21
Report Card Score – Poor			0.37	0.33	0.14	0.47	0.38	0.34

Table A 9 Environmental covariates for coral locations. For chlorophyll a (Chl *a*) and Non algal particulates (NAP) a square of nine 1km square pixels was selected adjacent to each reef location. From these pixels the mean concentrations for over the period 2015-2019 were downloaded from the Bureau of Meteorology, Marine Water Quality Dashboard. And average to month of the year and then reef. Clay and silt is the mean proportion of sediments from reef sites with grainsize < 63um. Within (sub-)regions, reefs are ordered by Chl a concentration.

(sub) Region	Reef	Chl a (µgL-1)	Nap (mgL-1)	Clay and silt (%)
Barron Daintree	Low Isles	0.32	0.90	7.5
	Snapper North	0.49	1.00	40.5
	Snapper South	0.50	1.08	11.2
Johnstone Russell-Mulgrave	Fitzroy East	0.26	0.66	1.7
	Franklands East	0.28	0.72	3.2
	Green	0.28	0.56	6.5
	Franklands West	0.33	0.73	31.3
	Fitzroy West	0.36	0.76	9.3
	High East	0.37	0.76	1.3
	High West	0.52	0.92	12.8
Herbert Tully	Barnards	0.45	0.75	6.1
	Dunk North	0.54	0.94	12.3
	Dunk South	0.61	1.00	12.1
	Bedarra	0.64	1.06	42.3
Burdekin	Palms East	0.25	0.69	0.5
	Havannah North	0.34	0.79	7.1
	Palms West	0.35	0.74	5.6
	Havannah	0.37	0.79	7.0
	Pandora North	0.44	0.85	46.0
	Pandora	0.46	0.84	4.1
	Lady Elliot	0.63	1.09	14.5
	Magnetic	0.67	1.91	10.0
	Middle Rf	0.89	3.39	51.5
Mackay-Whitsunday	Hayman	0.29	0.79	8.0
	Langford	0.30	0.90	46.0
	Border	0.31	1.01	12.5
	Hook	0.32	1.09	35.6
	Double Cone	0.34	1.26	36.1
	Seaforth	0.38	1.26	37.1
	Dent	0.40	1.52	53.8
	Daydream	0.43	1.56	72.4
	Pine	0.44	1.83	61.0
	Shute Harbour	0.44	1.67	53.9
Fitzroy	Barren	0.32	0.45	4.2
	Keppels South	0.44	0.61	9.8
	North Keppel	0.47	0.68	21.3
	Middle	0.48	0.75	4.8
	Peak	0.70	2.22	9.5
	Pelican	0.71	1.45	2.1

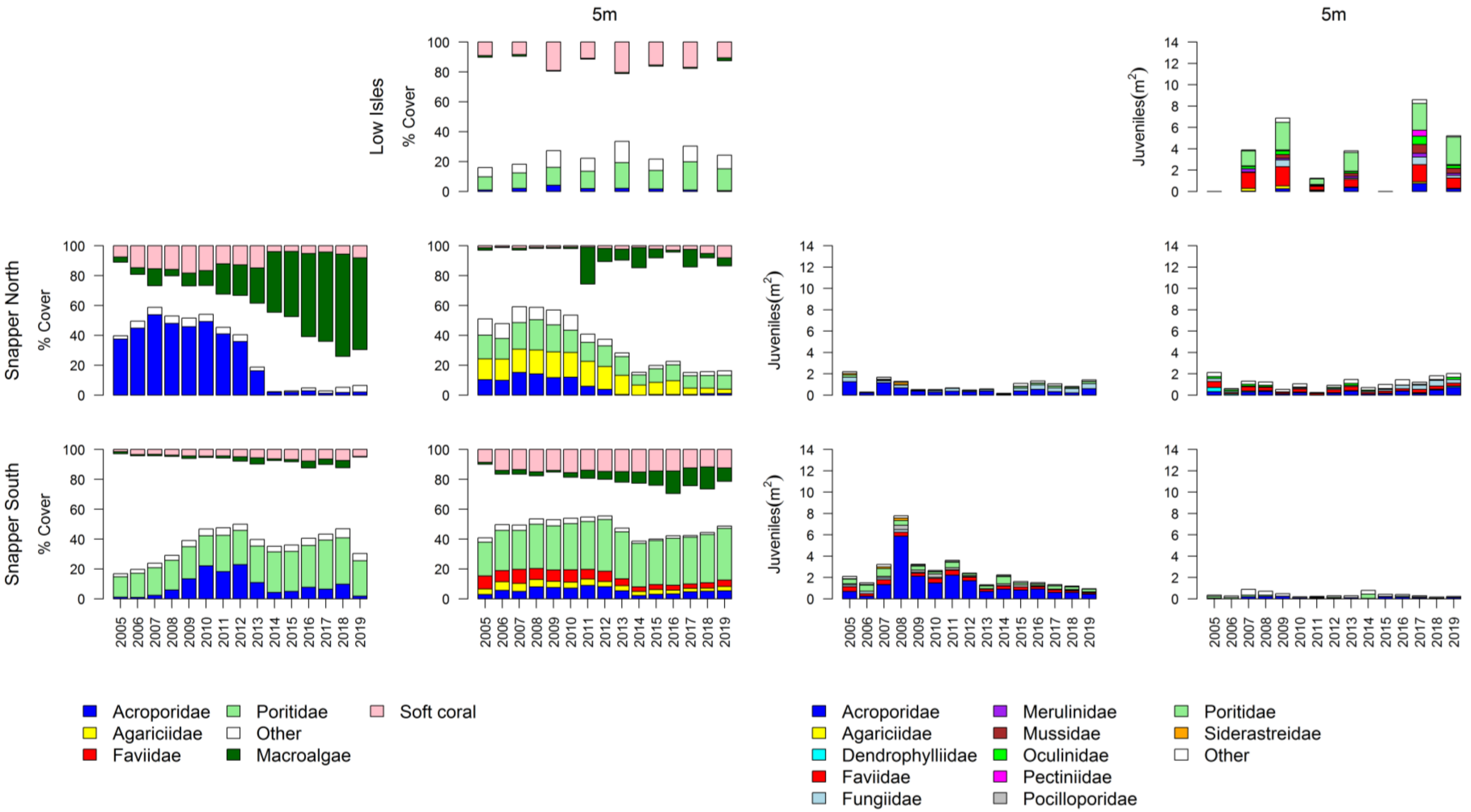


Figure A 1 Barron Daintree sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends relevant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

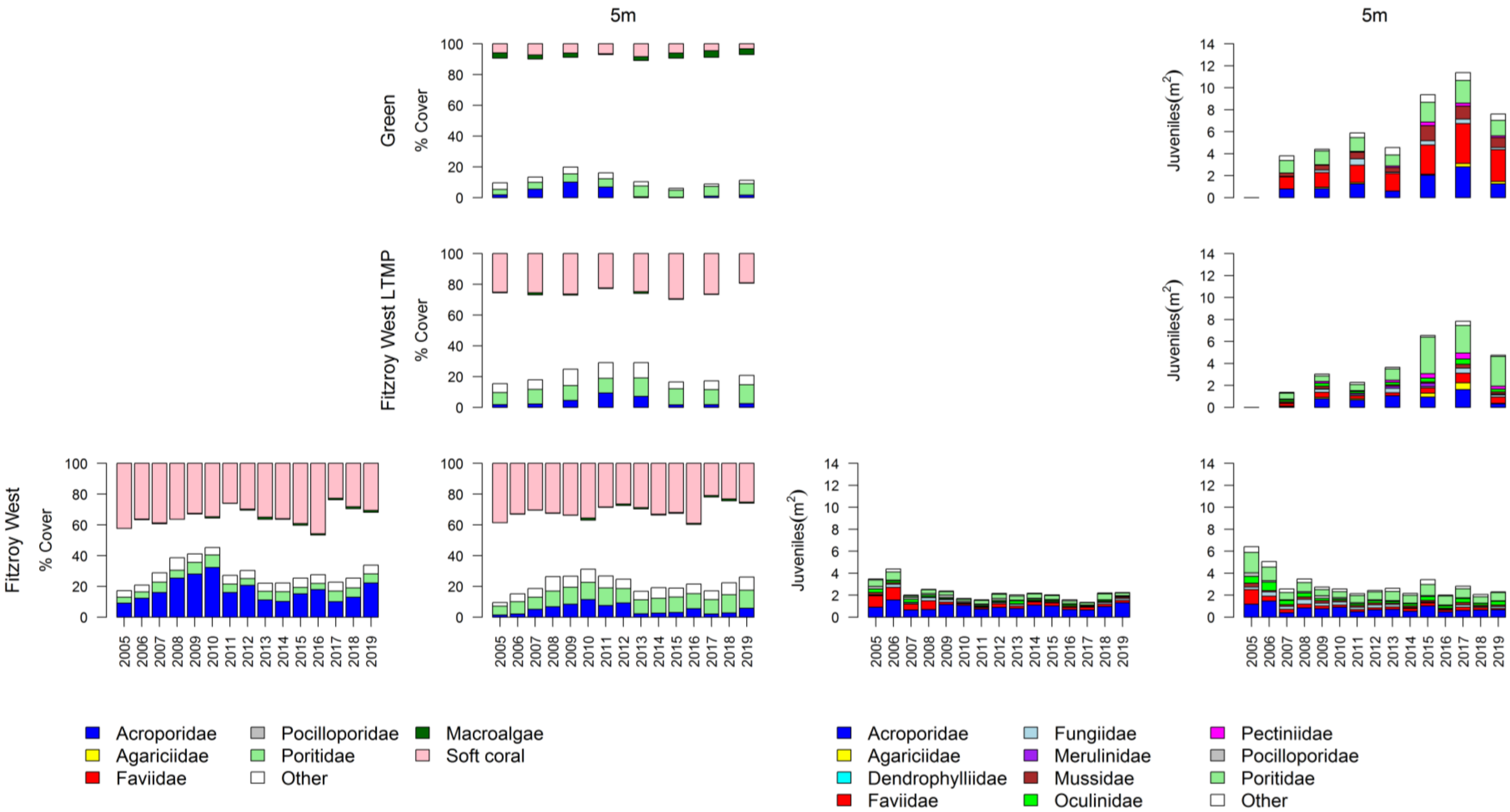


Figure A 2 Johnstone Russell-Mulgrave sub-region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate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

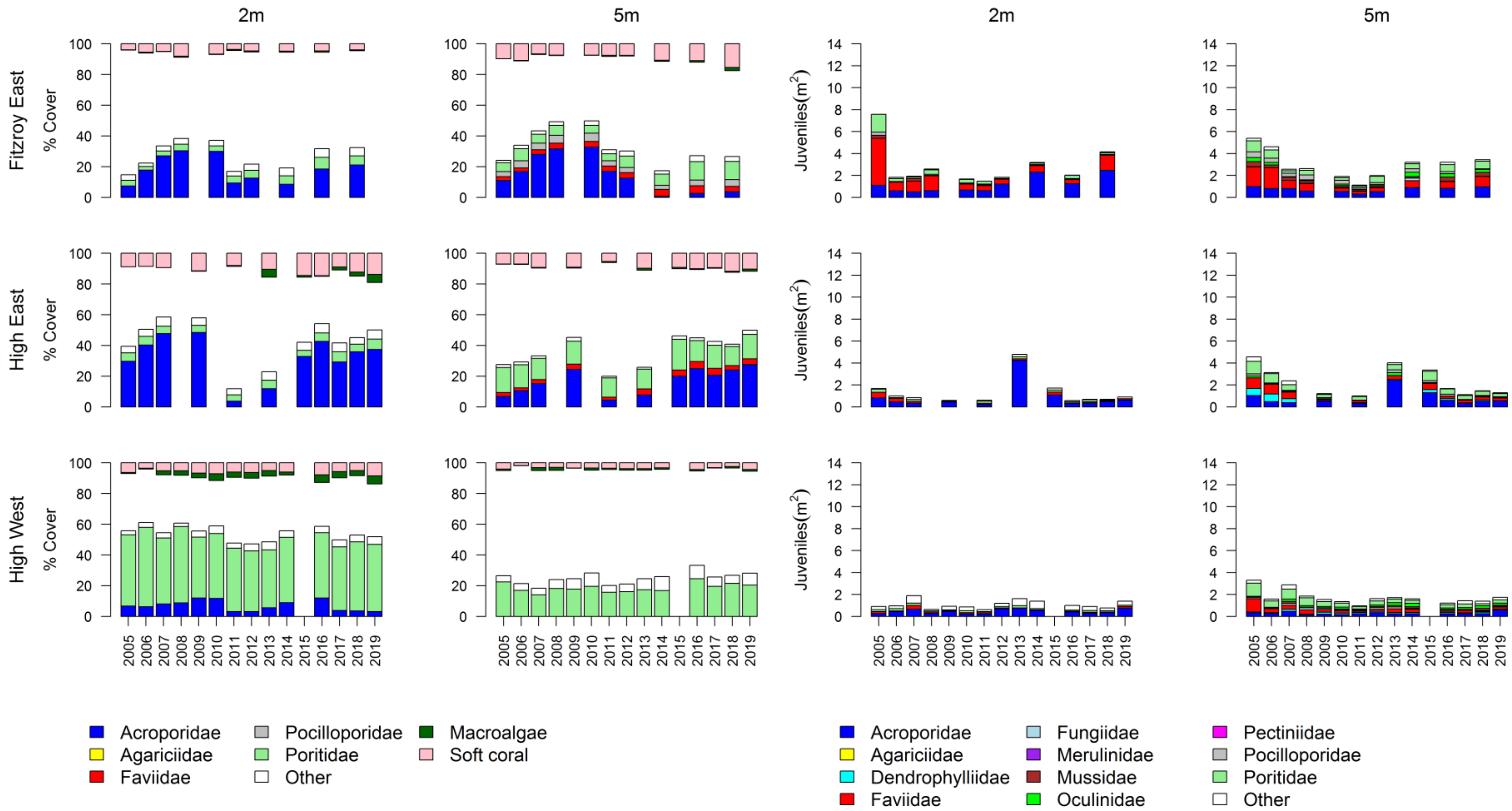


Figure A 2 continued

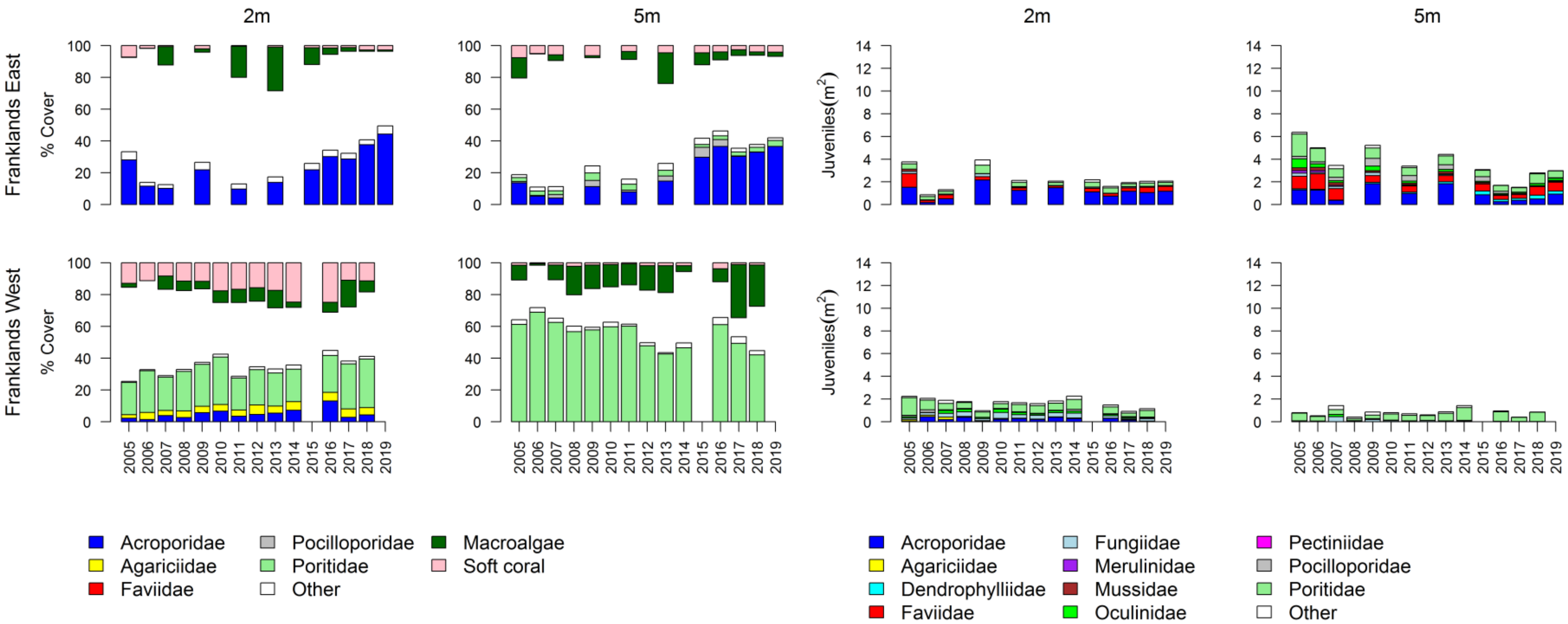


Figure A 2 continued

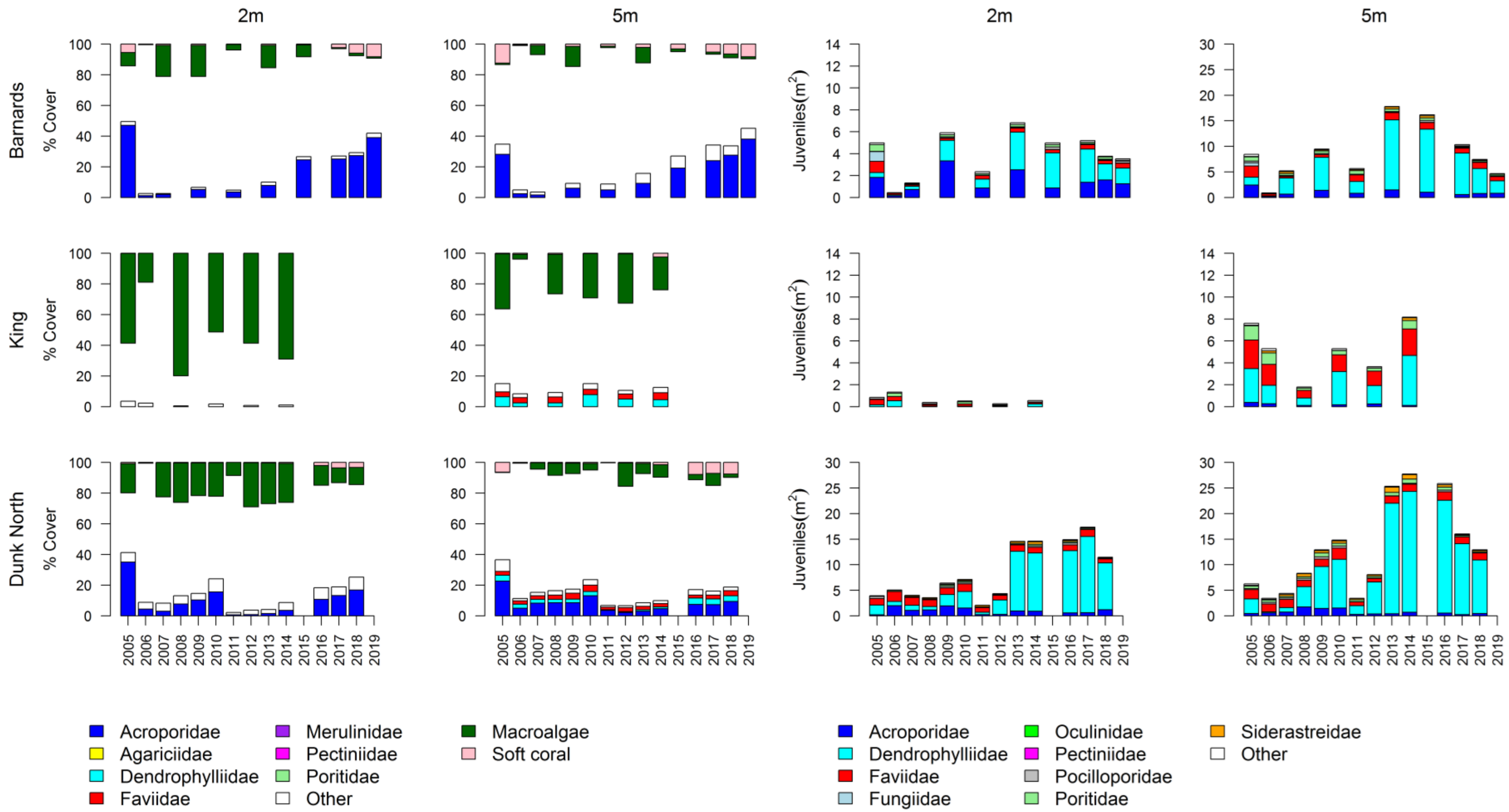


Figure A 3 Herbert Tully sub- region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends with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

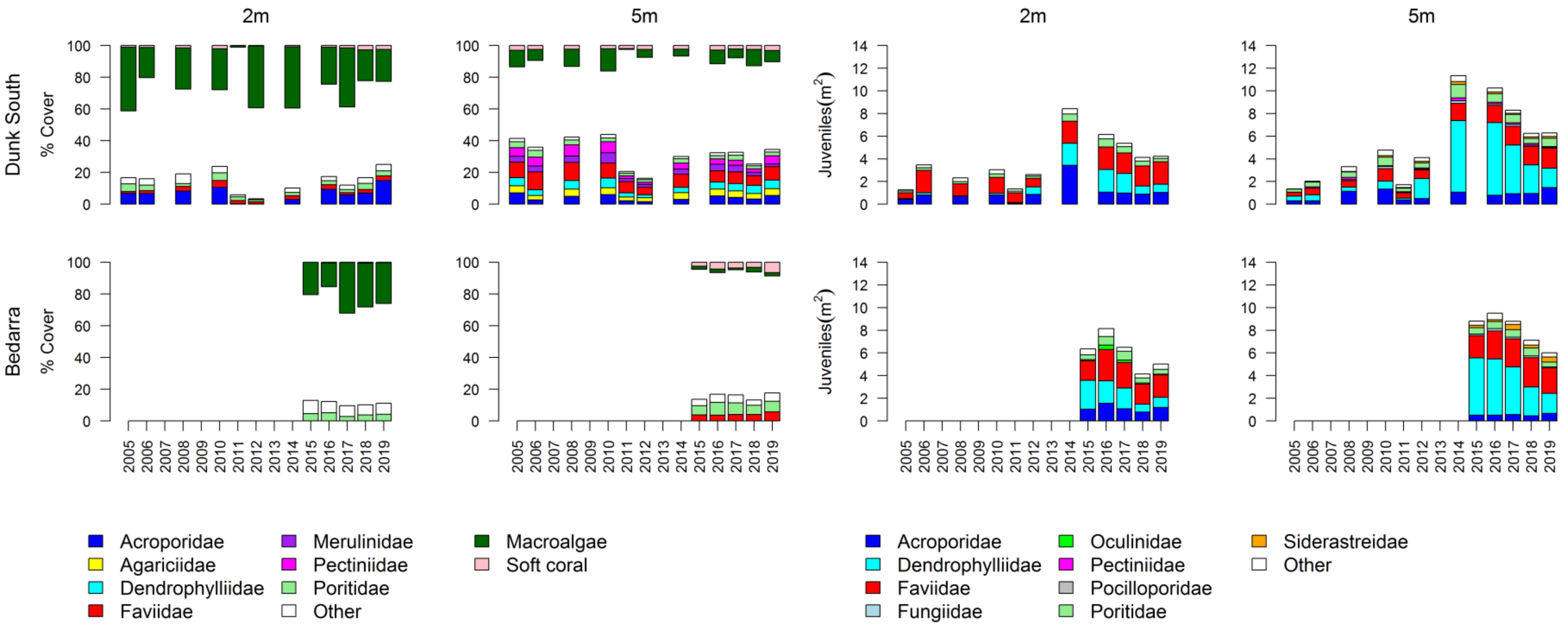


Figure A 3 continued

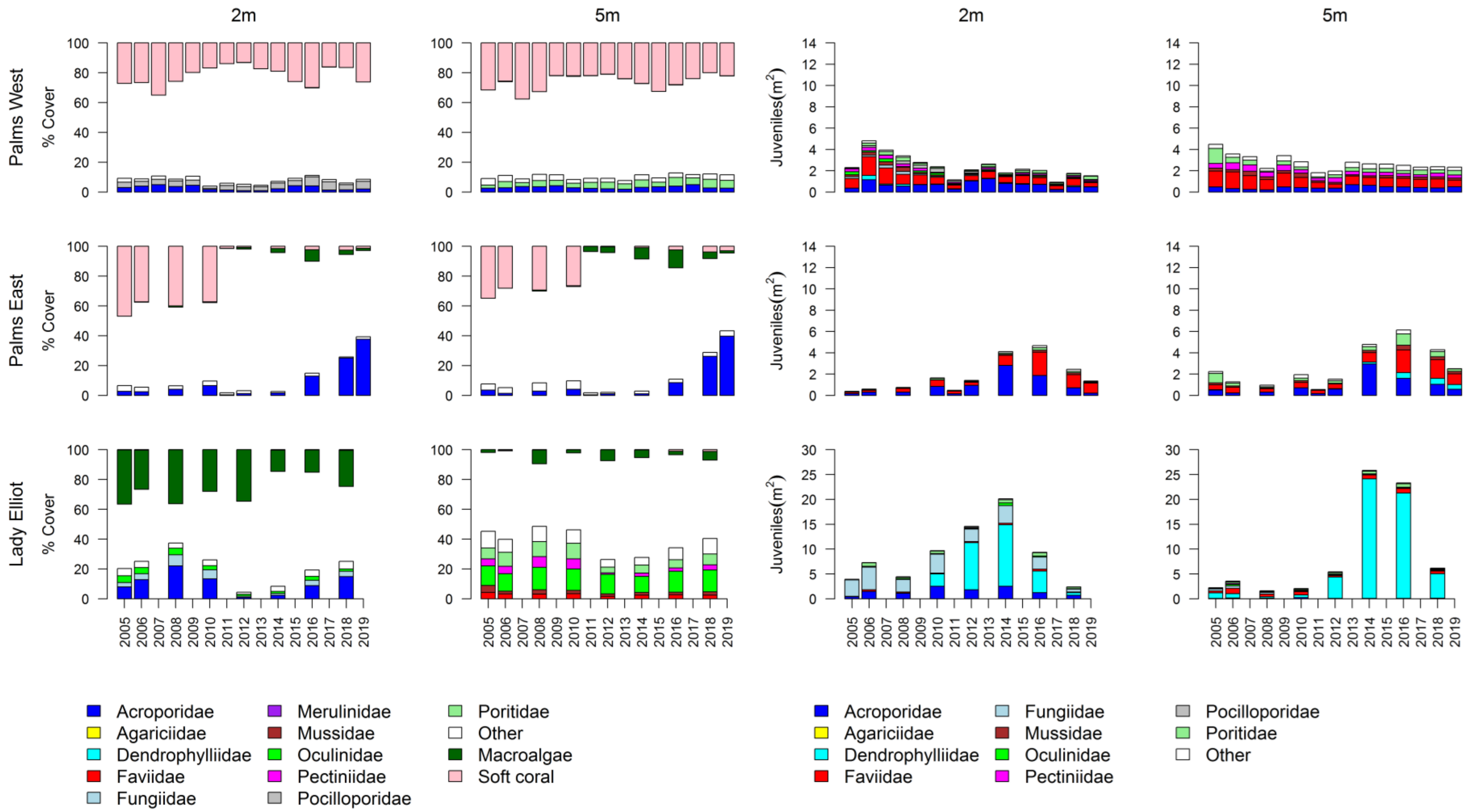


Figure A 4 Burdekin Region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends with relevant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

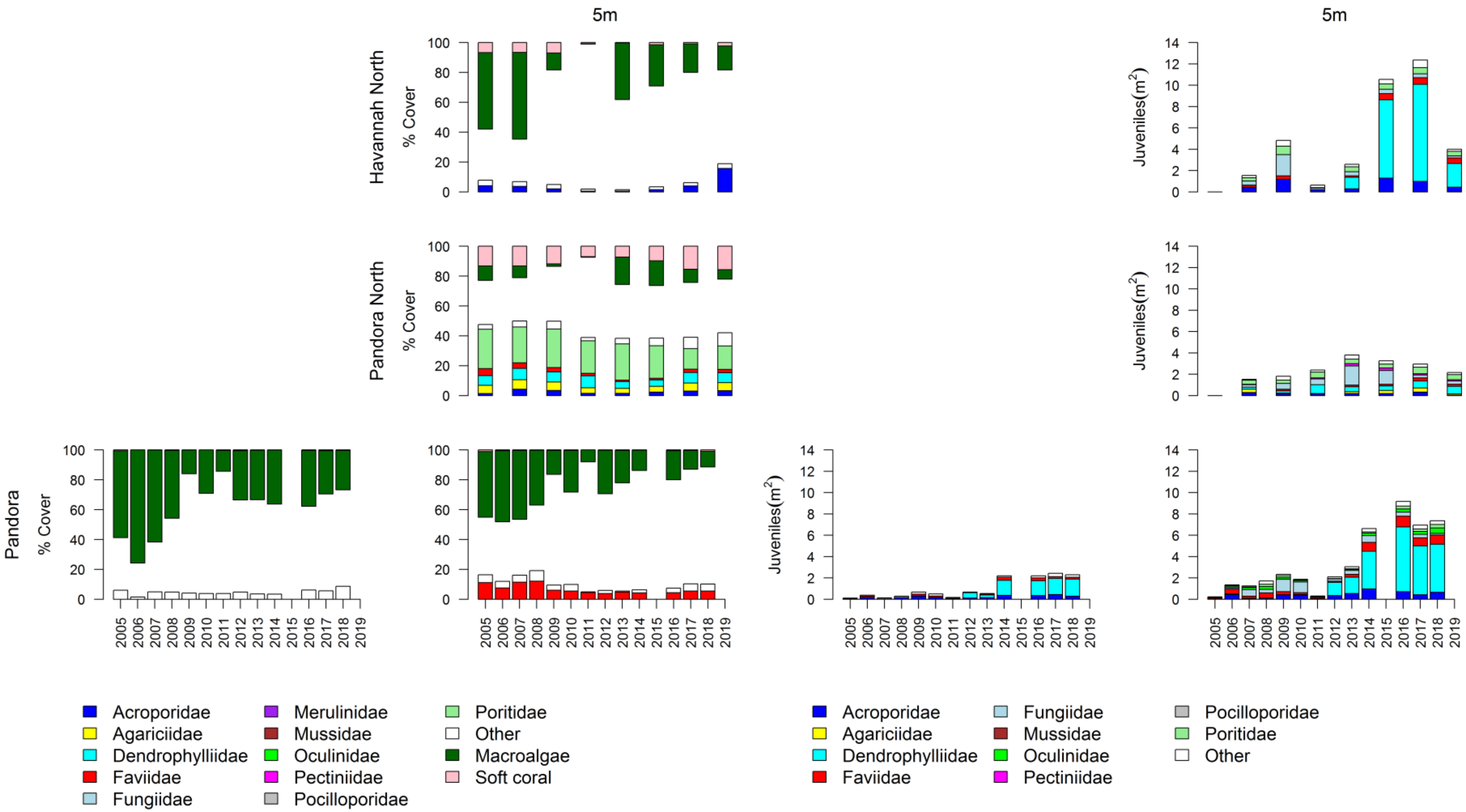


Figure A 4 continued

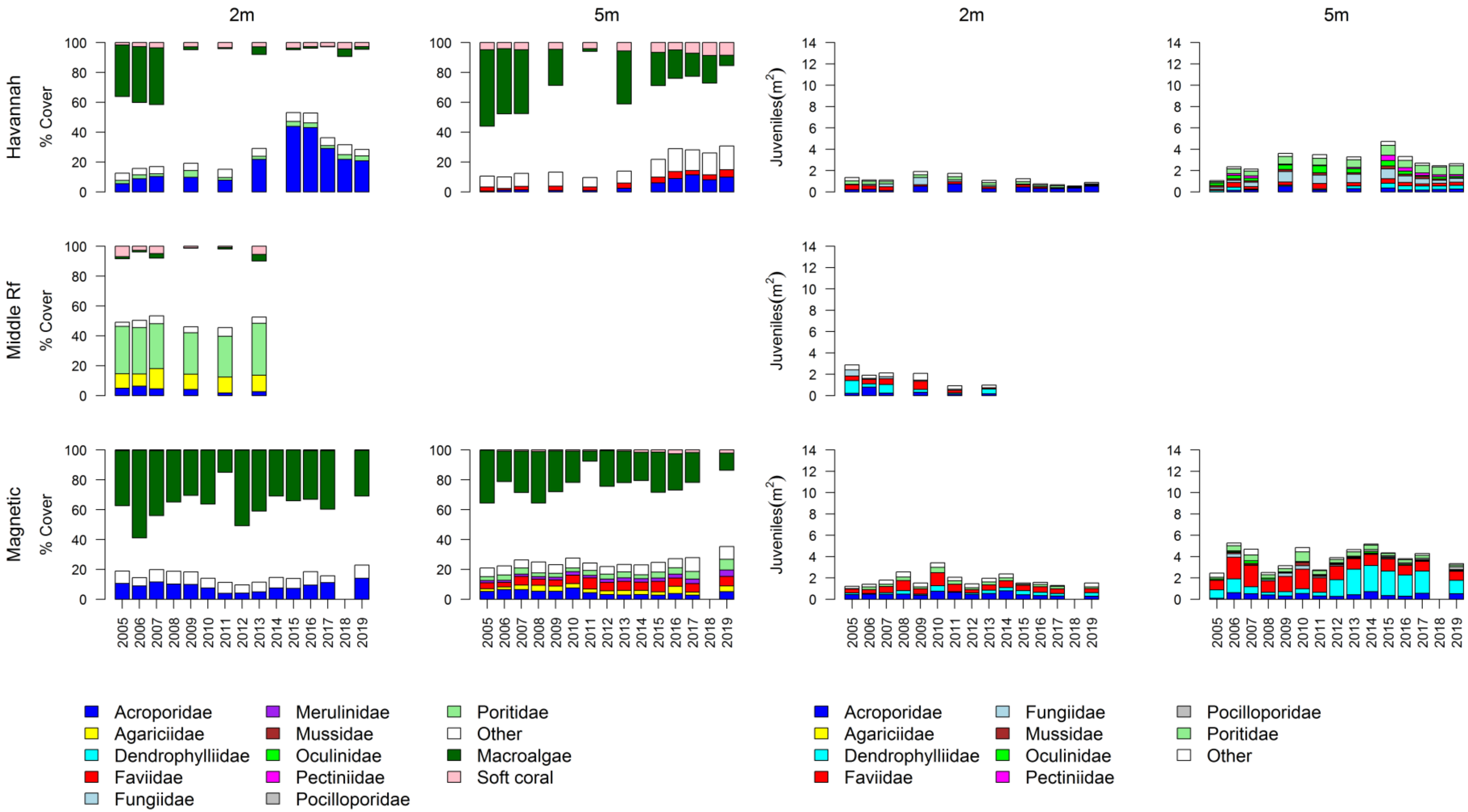
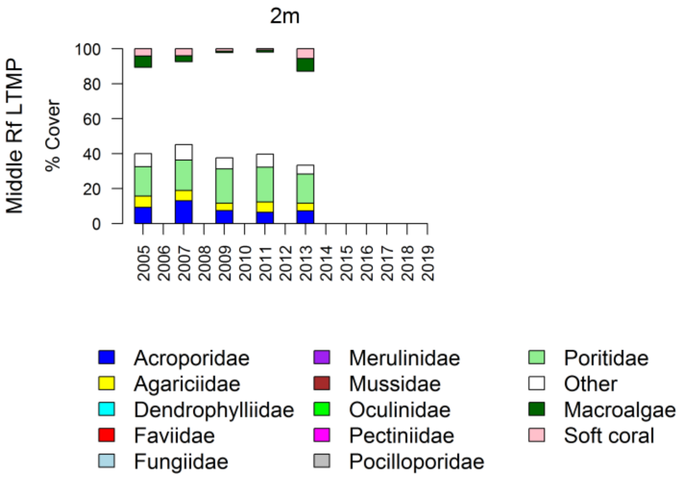


Figure A 4 continued



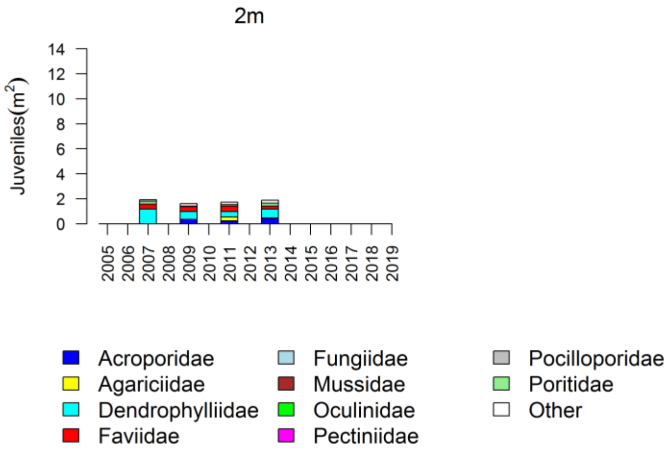


Figure A 4 continued

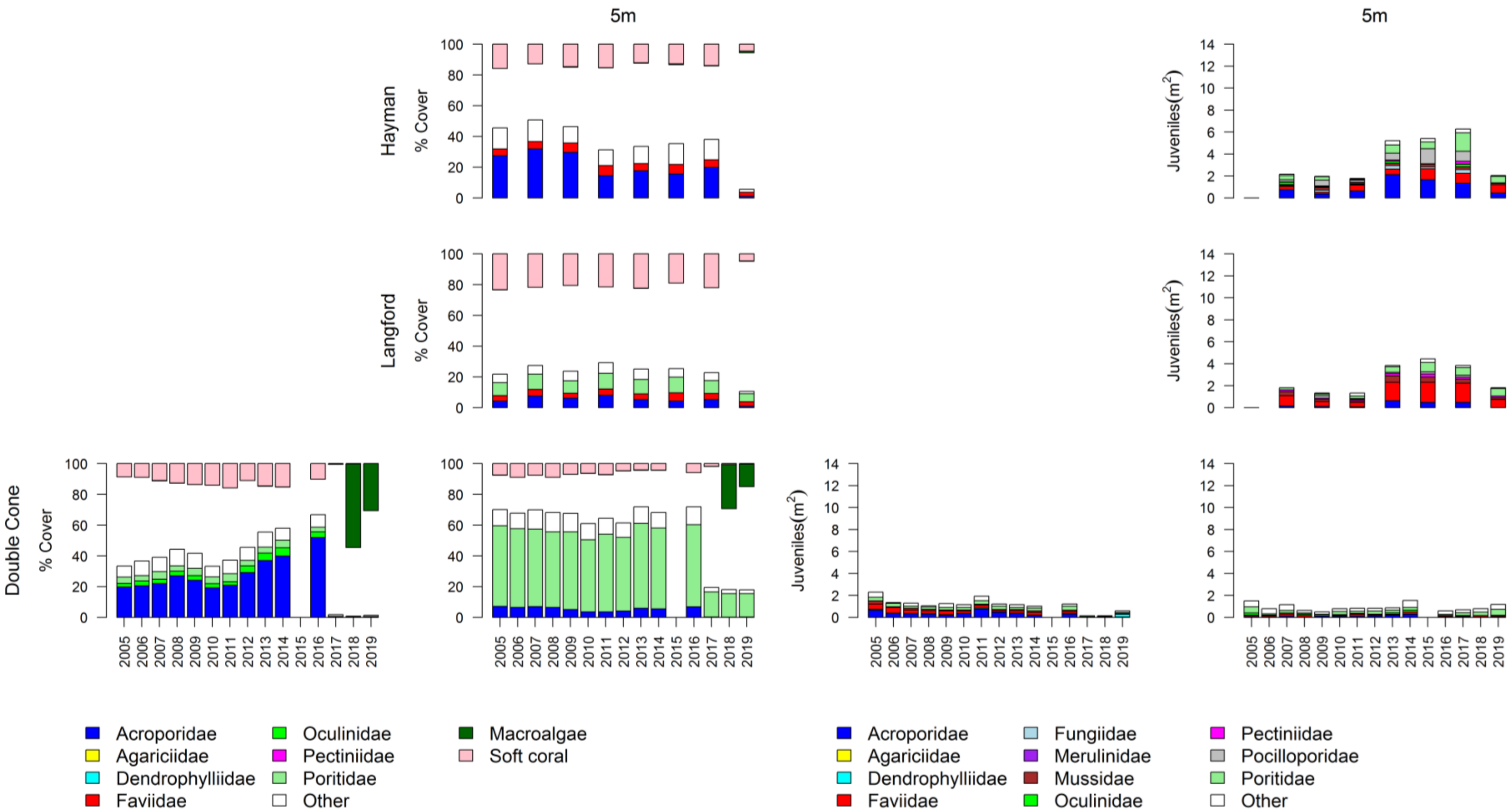


Figure A 5 Mackay-Whitsunday Region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends with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

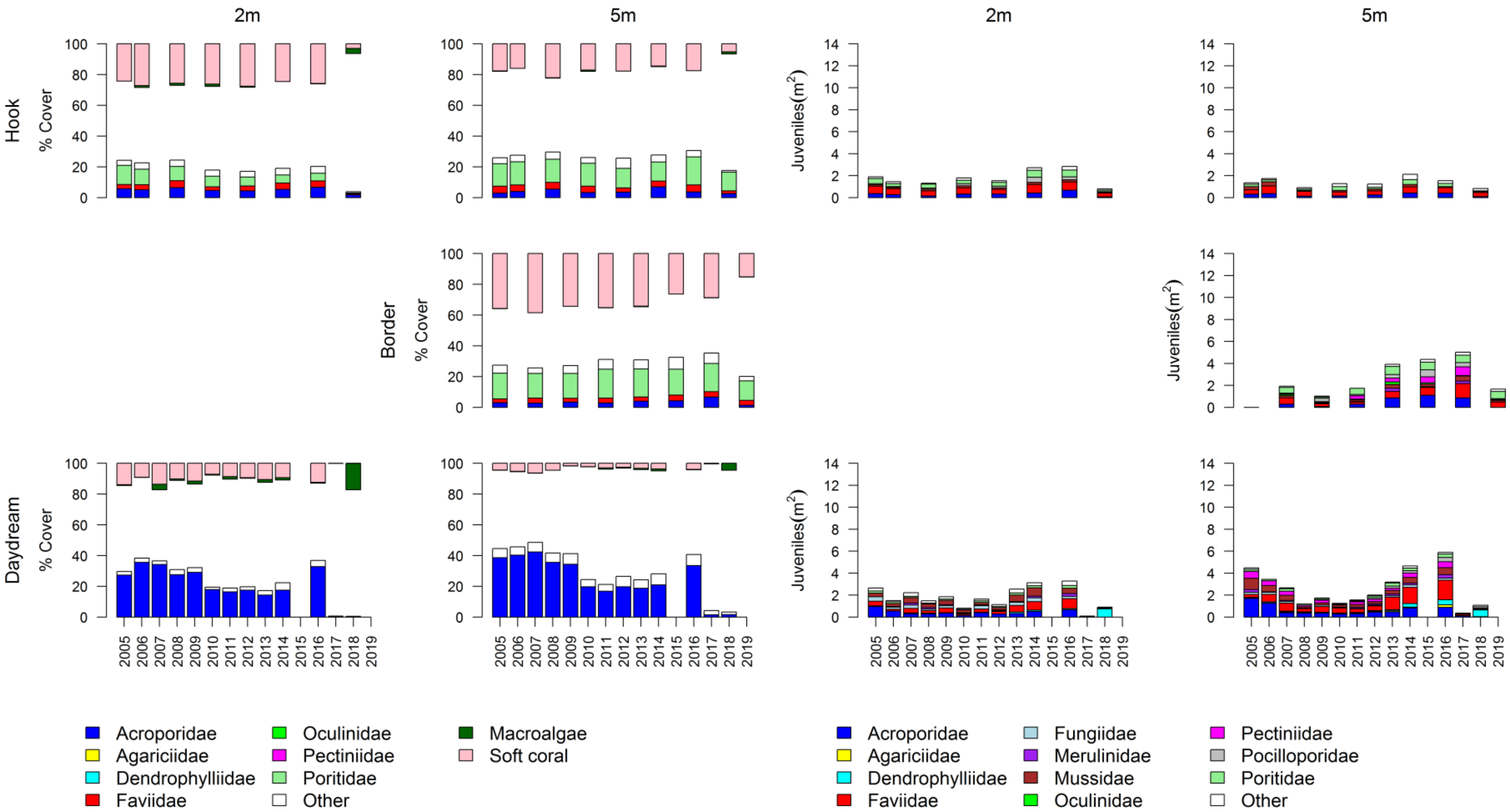


Figure A 5 continued

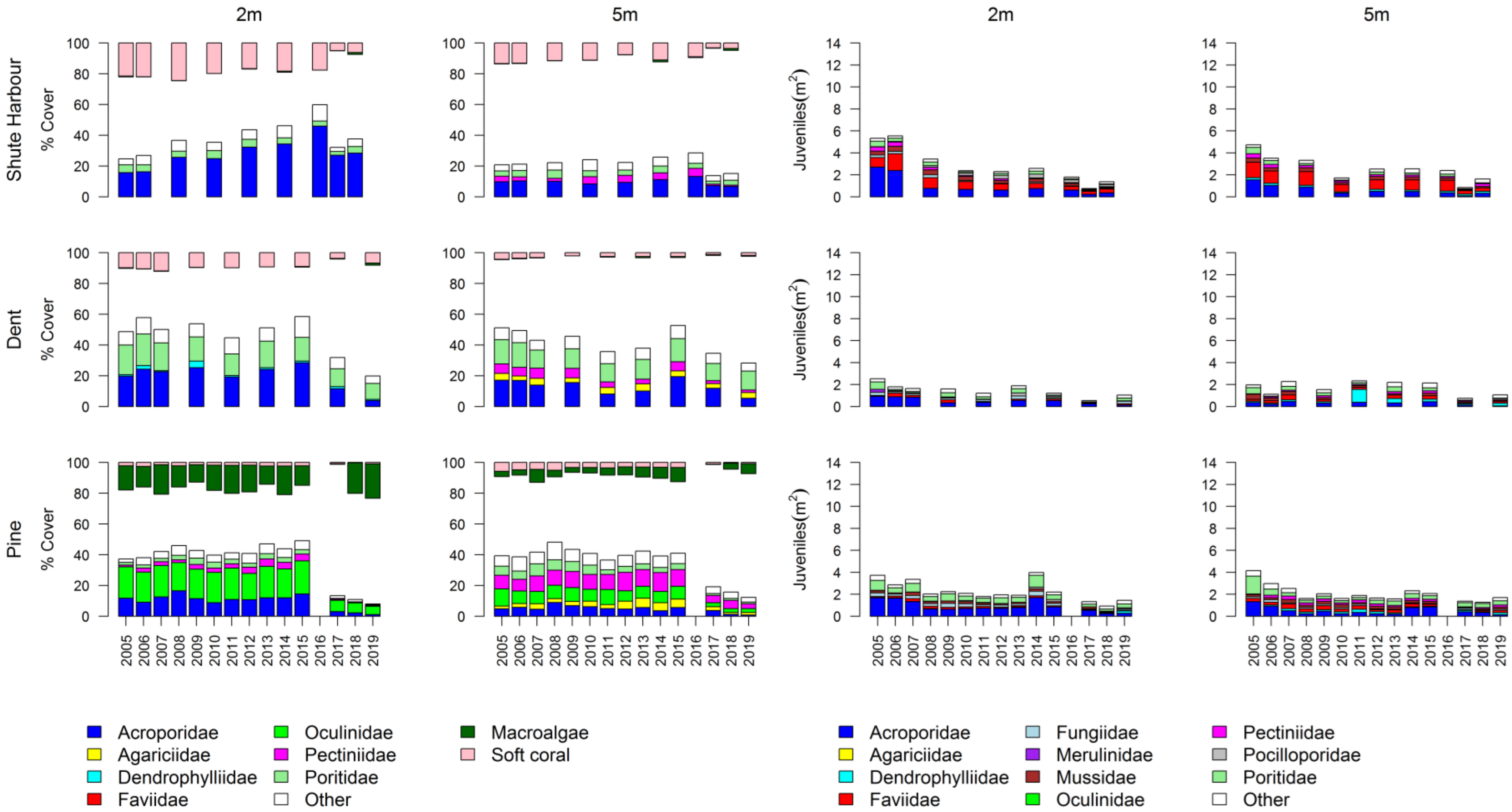


Figure A 5 continued

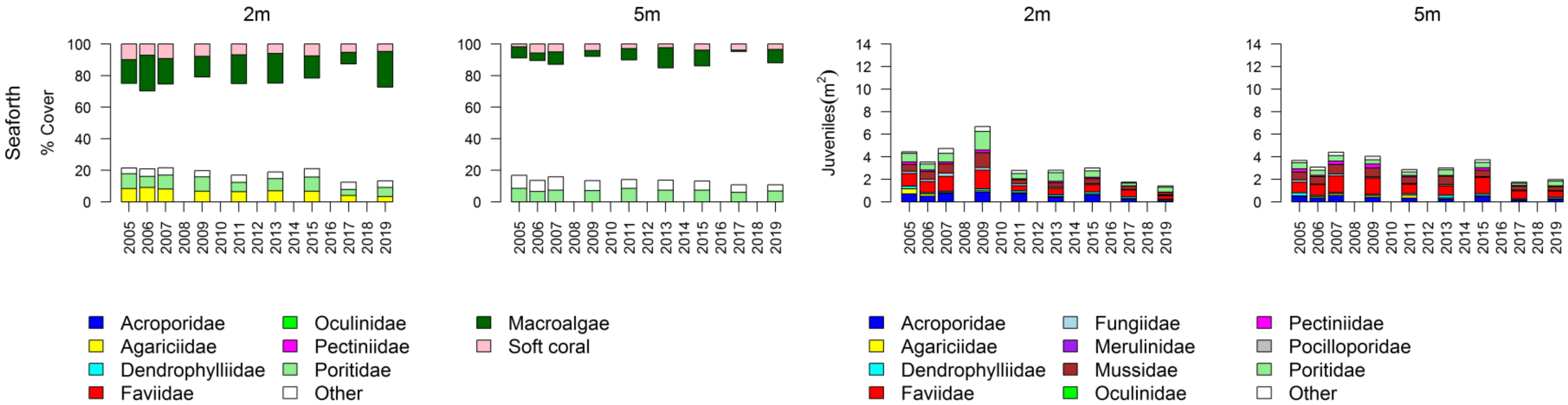


Figure A 5 continued

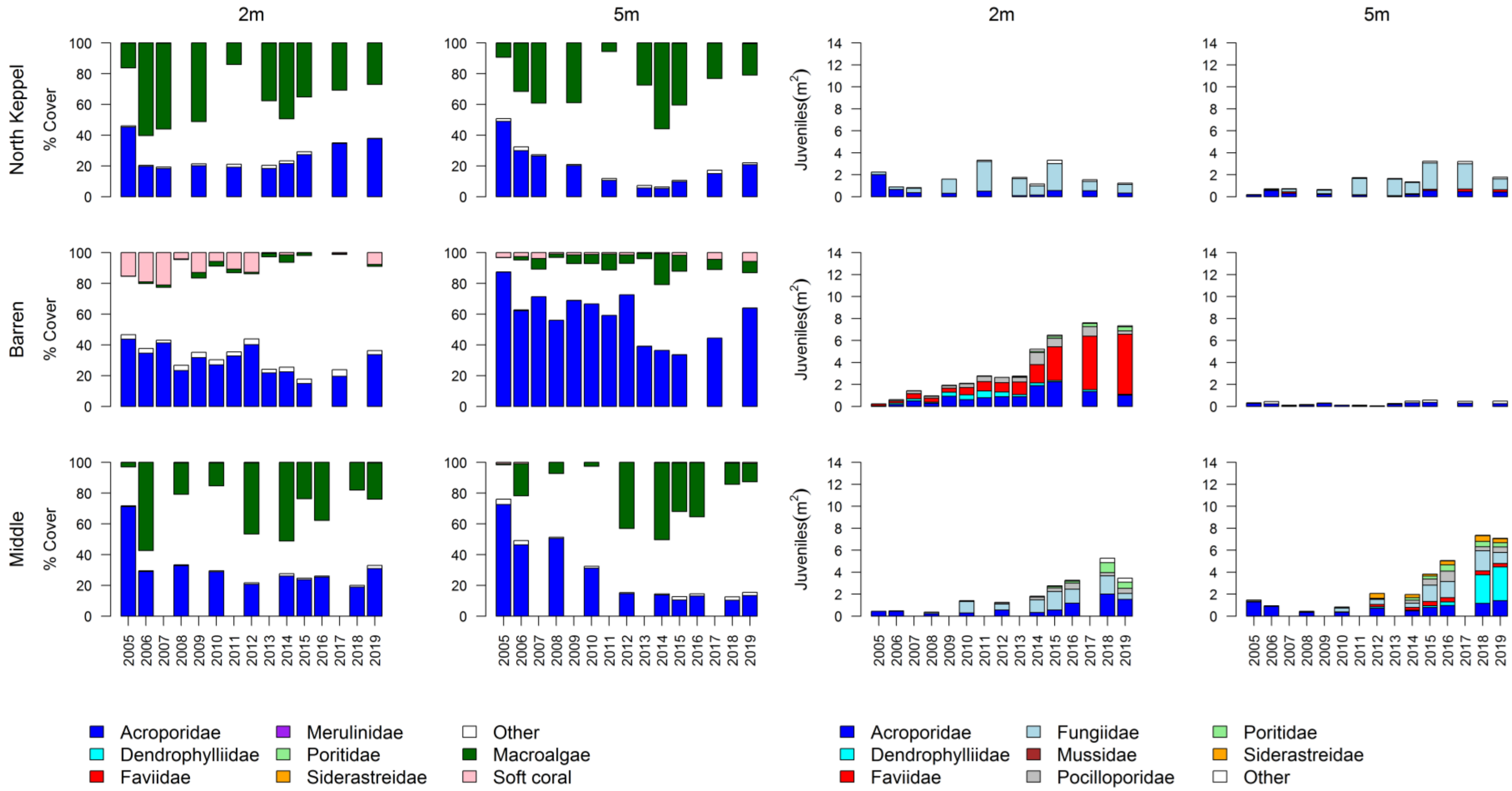


Figure A 6 Fitzroy Region benthic community composition. Cover estimates are separated into regionally abundant hard coral families and the total cover for soft corals and macroalgae (hanging). Juvenile density estimates are for regionally abundant hard coral families. Separate legends with relevant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

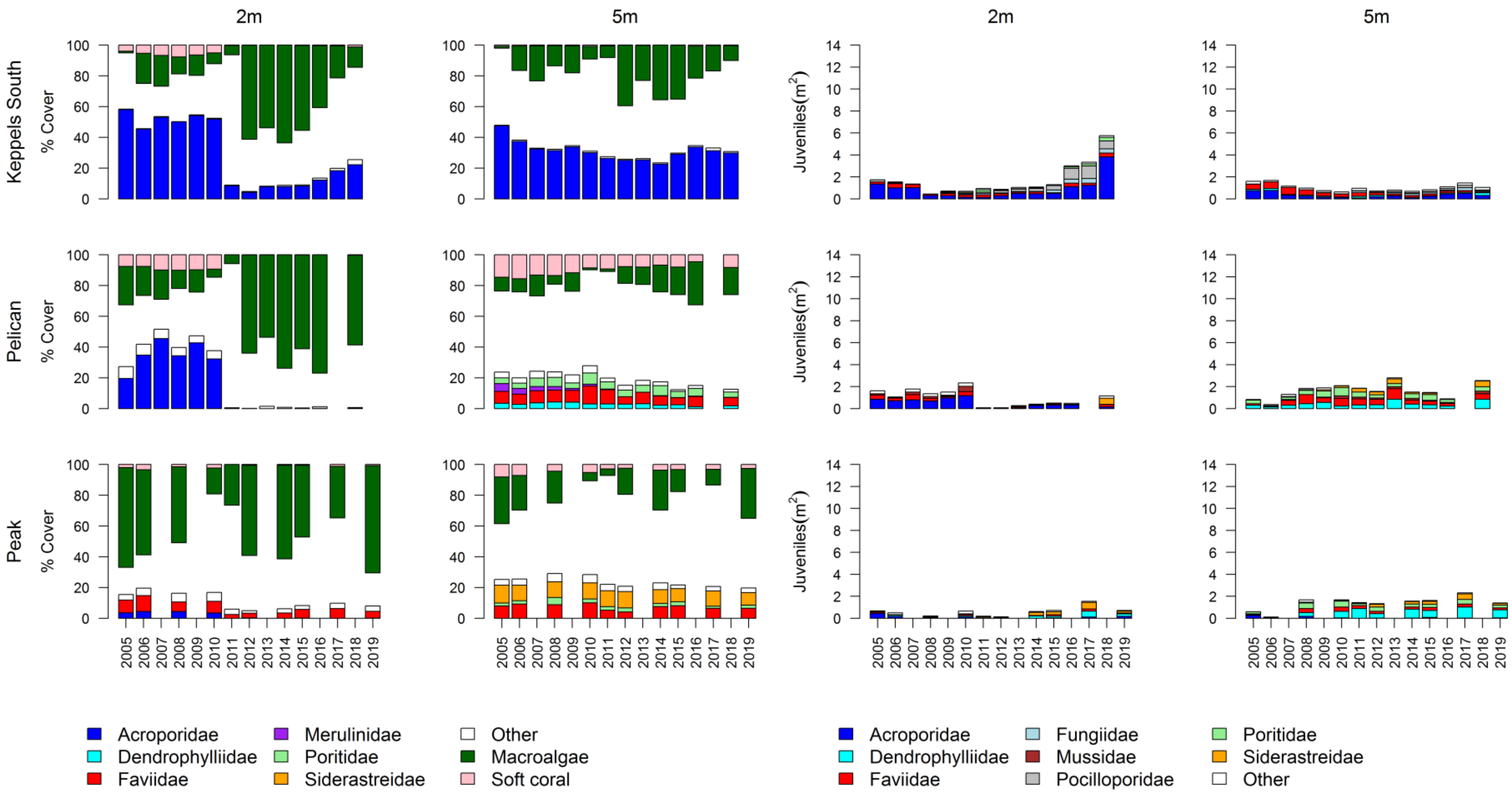


Figure A 6 continued

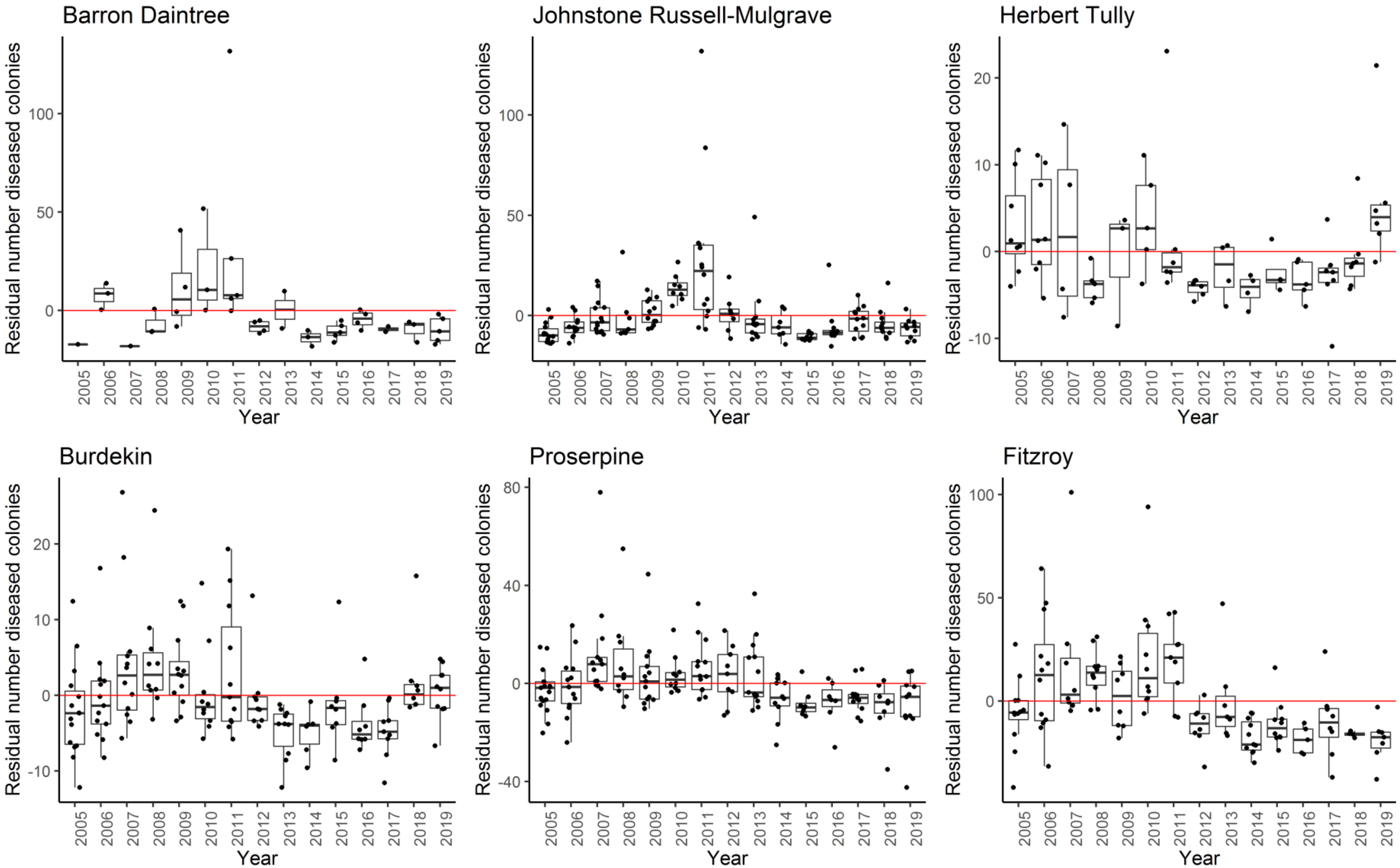


Figure A 7 Coral disease by year in each region. Boxplots include the number of coral colonies suffering ongoing mortality attributed to either disease, sedimentation or 'unknown causes' for each reef, depth and year. Data are standardised to the reef and depth mean across years.

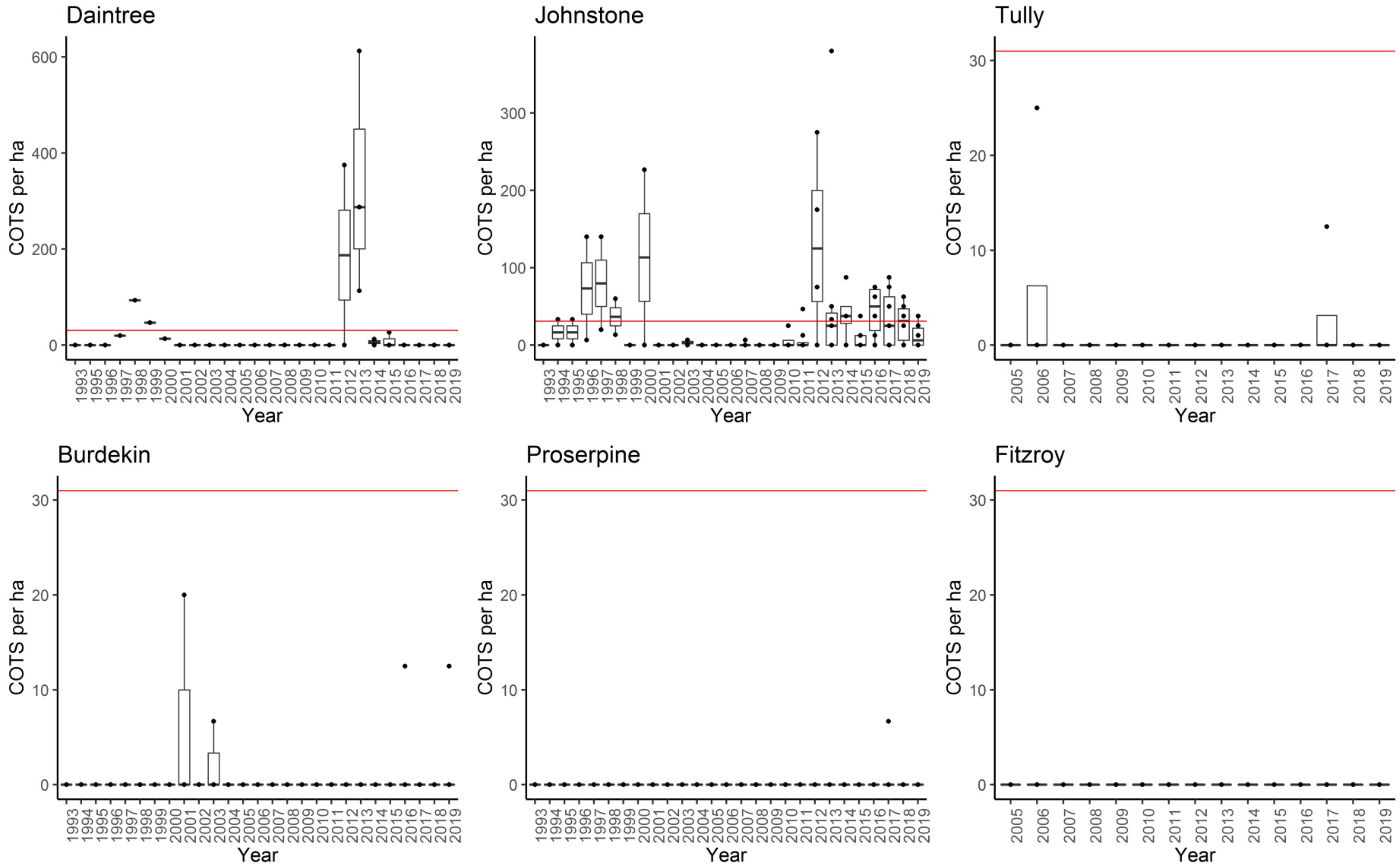


Figure A 8 Crown -of-thorn-starfish mean density (P/ha) by year in each region. Red line indicates outbreak densities of 31 P/ha.

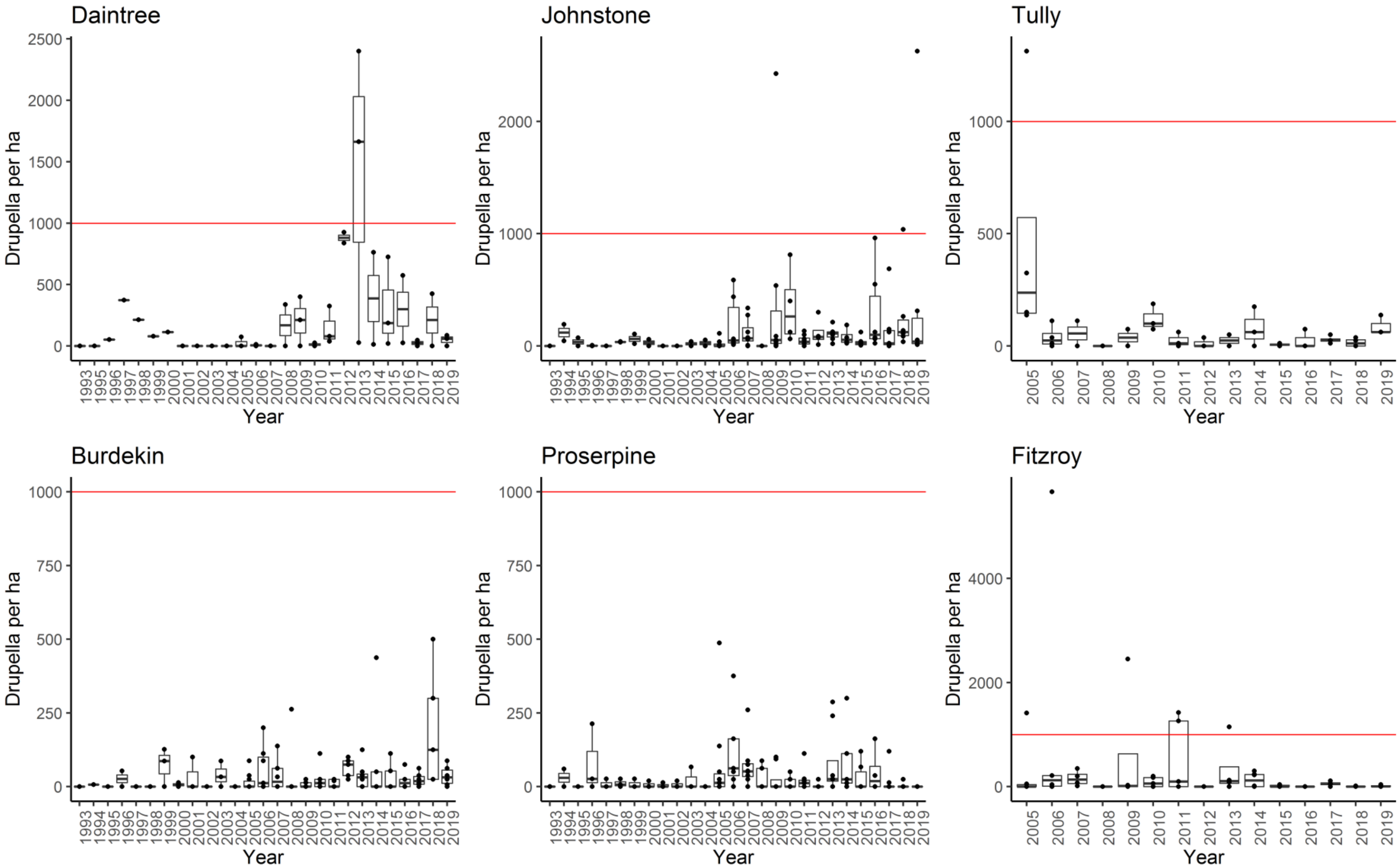


Figure A 9 Mean density of Drupella (P/ha) by year in each (sub)-region. Red line indicates densities of *Drupella* which have detrimental impact on coral communities.

Table A 10 Percent cover of hard coral genera 2019. 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*	*Echinopora*	*Favia*	*Favites*	*Fungia*	*Galaxea*	*Goniastrea*	*Goniopora*	*Isopora*	*Lobophyllia*	*Merulina*	*Montipora*	*Mycedium*	*Oxypora*	*Pachyseris*	*Pavona*	*Pectinia*	*Platygyra*	*Plesiastrea*	*Pocillopora*	*Podobacia*	*Porites*	*Psammocora*	*Symphyllia*	*Turbinaria*	*Other*
Barron Daintree	Low Isles	5	0.30		0.03	0.03	0.20	1.00	0.07		0.60	2.24	0.13	0.37		1.79	0.17	0.10	0.07	0.03	0.77	0.03		0.17		0.23	0.20	14.29	0.03	0.07	0.07	1.20
	Snapper North	2	1.71		0.13			2.42			0.13	0.25				0.13		0.25						0.08		0.04		1.25	0.08			
		5	0.56		0.06	0.25			0.19	0.19	0.44	0.56		6.00	0.06		0.19	0.38	0.31		2.19	0.44		0.13		0.13		3.19			0.13	0.94
	Snapper South	2	0.42			0.08			0.21	0.13	0.08	2.25	1.21	0.21				1.29				0.25		0.17				23.58	0.21	0.04	0.04	0.21
		5	4.94		3.94	0.06		0.25		0.06	0.13	0.13		2.50		0.06		0.38			2.25	0.63		0.06				32.13	0.38		0.25	0.56
Johnstone Russell-Mulgrave	Green	5	1.40				0.06	0.13	0.23	0.03	0.03	0.17	0.06	0.35		0.26		0.20			0.03			0.10		0.03	0.03	7.11		0.28	0.10	0.58
	Fitzroy East	2	16.19			0.06			0.13	0.38			0.44	0.06				4.88				0.06		0.13		3.31		5.94				0.81
		5	2.81				0.63	0.75	0.44	0.25	0.19	1.13	0.44	0.19		0.56	0.06	0.56			0.06	0.06		0.31		4.44		11.69	0.75	0.25		1.06
	Fitzroy West	2	18.63				0.13	1.81	0.06	0.13	0.38	0.63		0.50		0.50	0.06	3.56	0.06				0.06	0.06		0.75		5.44	0.06	0.38		0.63
		5	3.94		0.13		0.88	1.00	0.13		0.56	0.94		0.88		1.88		1.75	0.44	0.31	1.00			0.19		0.06		10.82	0.50	0.06	0.06	0.50
	Fitzroy West LTMP	5	1.19		0.10	0.03	0.07	0.48	0.03	0.03	0.26	0.79	0.03	0.16		1.12	0.26	1.19	0.20	0.03	1.33	0.07	0.32	0.17		0.20	0.07	12.18			0.07	0.36
	Franklands East	2	25.38			0.19		1.13	0.19	0.25		0.13	0.13	0.06	0.50			18.44						0.13		1.00		1.69				0.31
		5	33.75		0.06	0.13			0.13	0.06		0.31				0.06	0.25	2.63					0.06	0.19				3.88		0.06	0.19	0.19
	Franklands West	2	4.00							0.06	0.25	0.31		1.81		0.06		0.25			4.69					0.56		28.69		0.06		0.38
		5	0.13					0.31				0.06									1.94							42.06				0.25
	High East	2	29.25					1.13	0.13	0.13		0.38	0.44	0.25		0.50		8.13		0.06		0.13		0.25		0.56		6.56	0.13	1.56	0.25	0.31
		5	20.06			0.31		2.00	0.06	0.44		0.44	0.06	0.31		0.19	0.13	7.31	0.06			0.13		0.38		1.00	0.31	15.50	0.06		0.06	1.06
	High West	2	2.69		0.06			0.44	0.19	0.06	0.19	0.44	0.56	3.25		0.56	0.13	0.44			0.13	0.50		0.06		0.88	0.06	40.50		0.19		0.56
		5	2.00			0.13	0.06	0.06	0.44	0.06	0.31	0.81	0.13	4.19		0.06		0.13	0.06	0.19	0.38	1.19		0.13		0.31		16.25				1.31

(sub)-region	Reef	Depth	*Acropora*	*Alveopora*	*Caulastrea*	*Cyphastrea*	*Diploastrea*	*Echinopora*	*Favia*	*Favites*	*Fungia*	*Galaxea*	*Goniastrea*	*Goniopora*	*Isopora*	*Lobophyllia*	*Merulina*	*Montipora*	*Mycedium*	*Oxypora*	*Pachyseris*	*Pavona*	*Pectinia*	*Platygyra*	*Plesiastrea*	*Pocillopora*	*Podobacia*	*Porites*	*Psammocora*	*Symphyllia*	*Turbinaria*	*Other*
Herbert Tully	Barnards	2	25.06			0.31			0.13	0.19		0.13						14.00						0.31		0.69		0.13		0.25	0.44	0.25
		5	19.81			0.25		0.50	0.25	0.38	0.13	0.13		0.13		0.19	0.19	18.25	0.31	0.56				0.56		0.44		0.50	0.19		1.88	0.44
	Dunk North	2	12.81			0.56		0.31	0.44	0.50		0.25	0.31	0.06				4.06						0.13	0.13	1.38		1.06	0.19		2.81	0.25
		5	3.63	0.13		0.25			0.25	1.25	0.06	0.13	0.19	0.13		0.13		5.13	0.13	0.06				1.06	0.06	0.25		0.75	0.38		3.63	1.25
	Dunk South	2	9.94			1.50			0.38	0.19	0.06	1.13		0.31		0.38		4.94				0.50		0.25	0.38			2.81			1.50	0.81
		5	2.81		0.06	0.44		0.56	3.69	0.94	0.06		1.38	0.69		0.50	1.75	1.94	2.56	1.44	4.19	0.19	1.13	1.00	0.13	0.31	0.63	1.56	0.06		5.31	1.13
	Bedarra	2	2.19			0.44			0.38	0.44	0.13	0.06	0.19	0.06		0.44		0.25	0.19			0.19				0.44	0.13	4.06	0.25	0.13	0.50	0.63
		5	0.19			0.31			4.06	0.13			0.38	3.13		1.25	0.56	0.25	0.69	0.06	0.94	0.06		0.50		0.13	0.44	3.44			0.38	0.75
Burdekin	Palms East	2	36.31			0.13			0.13	0.19			0.31					1.31										0.94				
		5	38.66							0.25		0.25	0.13	0.19				1.00						0.19		1.31		1.19				0.13
	Palms West	2	1.50			0.06		0.25	0.06	0.06				0.13				0.44			0.13		0.06	0.25		5.13		0.13		0.06		0.25
		5	1.38				0.94	0.19	0.31	0.13	0.13		0.06	0.31		0.19		0.94		0.31	0.13	0.13		0.19		0.31		4.88	0.06			1.00
	Havannah North	5	13.06			0.10		0.17	0.16		0.20	0.30	0.06	0.10		0.03	0.13	2.63	0.03	0.03	0.10		0.07	0.10		0.03		0.66	0.03		0.23	0.59
	Havannah	2	12.01					0.81	0.06		0.13	0.75	0.13	0.19	2.00		0.13	6.82				0.13	0.06	0.94		0.44		3.06			0.44	0.31
		5	6.13		0.13	0.75	1.56	0.88	0.38	0.44	2.63	1.06	0.06	0.50	0.13	0.38	2.69	3.69	0.31	1.06	0.88	0.38	0.75	0.38		0.13	0.19	0.75			2.69	1.88
	Pandora	5	2.06			0.25				0.06			0.06	0.13				1.63							0.81			3.13			0.06	0.50
		2	2.44			0.31	3.00		0.63	0.25	0.38	0.38		0.13			0.19	0.69			0.13			0.69		0.13		0.06			0.19	0.69
	Pandora North	5	1.37	1.10	0.07	0.17	0.03	0.96	0.57	0.17	1.70	1.37	0.07	11.05	0.03	0.63	1.00	1.77	1.00	0.67	4.55	0.10	0.47	0.23		0.10	0.27	3.54	0.03		6.62	2.40
	Lady Elliot	2	9.25			0.19			0.13		3.63	1.44		0.13		0.06	0.06	5.63				1.88						0.69			1.88	0.13
		5	1.19	1.19		0.25			0.81	0.56		14.69	0.13	3.69		2.13	0.56	0.44	1.19	1.88	2.75		0.44	0.44			0.88	2.31	0.19		3.38	1.31

(sub)-region	Reef	Depth	*Acropora*	*Alveopora*	*Caulastrea*	*Cyphastrea*	*Diploastrea*	*Echinopora*	*Favia*	*Favites*	*Fungia*	*Galaxea*	*Goniastrea*	*Goniopora*	*Isopora*	*Lobophyllia*	*Merulina*	*Montipora*	*Mycedium*	*Oxypora*	*Pachyseris*	*Pavona*	*Pectinia*	*Platygyra*	*Plesiastrea*	*Pocillopora*	*Podobacia*	*Porites*	*Psammocora*	*Symphyllia*	*Turbinaria*	*Other*
	Magnetic	2	3.13			0.81		0.44	0.06	0.19		0.75	0.06	0.56			0.06	10.94			0.56	0.31			0.56	0.44		1.44	0.13		1.38	1.13
		2	1.75			0.19		0.31	2.75	0.63	0.31	0.81		5.13		0.13	3.69	3.13	0.06	0.88	3.88		0.75	1.50	0.19	0.56	2.19	2.13	0.06		1.00	3.25
	Middle Rf LTMP	5	1.47			0.07		0.07	0.91	0.10	0.03	0.13	0.07	15.38	0.23	0.24	0.47	5.54	0.07	0.20	4.27	0.13	0.41	0.24		0.13	0.23	1.31		0.03	0.37	1.21
Mackay-Whitsunday	Hayman	5				0.03	0.93	0.24	0.53	0.20	0.13		0.23	0.30		0.17		1.00		0.03	0.66	0.10		0.43				0.30				0.30
	Langford	5	0.03			0.07	0.20	0.17	1.09	0.27	0.03	0.03	0.49	3.86		0.46		0.59			0.33	0.46		0.03		0.07		1.27			0.03	1.09
	Border	5	0.63	0.10	0.03	0.03	0.50	0.33	1.20	0.33	0.03		0.33	9.35		0.70	0.10	0.60	0.27	0.03	0.53	0.20	0.30	0.10		0.10	0.17	3.16		0.10	0.03	0.83
	Hook	2				0.13			0.13	0.13			0.06	0.13				1.63			0.25	0.06		0.06				0.50			0.25	0.44
		5	0.13			0.13		0.06	0.81	0.38				1.19		0.06	0.06	2.31			0.38	0.13		0.19				11.00	0.13		0.19	0.50
	Double Cone	2						0.31	0.06	0.06				0.06		0.06	0.06	0.06						0.06				0.38			0.19	
		5	0.06									0.88	0.31	14.19		0.38	0.06	0.19		0.06	0.06		0.06	0.13			0.06	1.00			0.06	0.38
	Daydream	2								0.06								0.13		0.06								0.25				
		5	0.06					0.44	0.06	0.06			0.25					1.31			0.06							0.63				0.38
	Dent	2	3.75			0.06			0.06			0.25	0.13	5.00		1.38	0.69	0.13		0.25	0.13	0.56	0.63	0.06		0.06		5.38			0.75	0.63
		5	4.50	0.13				0.25	0.06	0.19		2.19		10.38		0.63	0.63	0.38	0.25	0.94	2.94	0.38	0.44	0.13		0.25	0.13	1.81			0.19	1.44
	Shute Harbour	2	25.00		0.13			0.06	0.25	0.13		0.13	0.06	3.75		0.75	0.25	3.38	0.13	0.25	0.44	0.63	0.38	0.06		0.44		0.50				0.94
		5	4.00	0.06	0.06			0.06		0.06	0.06	0.38		2.13	0.31	1.50	0.19	2.44		0.56	0.75	0.25	0.19	0.25		0.31		1.00				0.63
	Pine	2	0.63									5.38		0.06				0.50	0.13	0.06	0.13		0.13				0.06	0.50				0.38
		5	0.06			0.06			0.19	0.06	0.25	2.50		1.06		0.81	0.06	0.69	0.69	1.19	1.44		1.13	0.31			0.63	0.06			0.06	1.06
	Seaforth	2	0.69					0.13	0.75	0.56		0.06	0.06	0.13		0.38	0.13				0.44	2.88				0.19		5.57			0.63	0.75
		5	0.31		0.19	0.13	0.56		0.31	0.19	0.19	0.06	0.13	5.50		0.19		0.19			0.06	0.69	0.06					1.38				0.69

(sub)-region	Reef	Depth	*Acropora*	*Alveopora*	*Caulastrea*	*Cyphastrea*	*Diploastrea*	*Echinopora*	*Favia*	*Favites*	*Fungia*	*Galaxea*	*Goniastrea*	*Goniopora*	*Isopora*	*Lobophyllia*	*Merulina*	*Montipora*	*Mycedium*	*Oxypora*	*Pachyseris*	*Pavona*	*Pectinia*	*Platygyra*	*Plesiastrea*	*Pocillopora*	*Podobacia*	*Porites*	*Psammocora*	*Symphyllia*	*Turbinaria*	*Other*
Fitzroy	Barren	2	23.03			0.44			0.06	0.13			0.06		1.82	0.25		8.81				0.06				0.50		0.38	0.13			0.58
		5	59.97								0.06							3.94								0.06						
	North Keppel	2	36.88								0.25							0.75								0.06						
		5	18.00								0.25					0.25		2.81		0.06				0.06		0.19			0.25		0.06	0.06
	Middle	2	29.00								0.31		0.06					1.88						0.31		0.81		0.63				
		5	11.13								0.50							2.25						0.13		1.25		0.13	0.13			
	Keppels South	2	17.24			0.19					0.06							4.89						0.06		2.88		0.13				0.06
		5	28.31								0.13							1.44								0.31		0.31			0.06	0.13
	Pelican	2				0.13				0.06								0.13							0.19				0.13			
		5	0.06	3.00		0.56			0.06	2.81			1.13	0.44		0.31		0.19						0.63	0.19				0.81		1.69	0.63
	Peak	2	0.13	0.13		0.94			0.38	0.13														0.06	2.94			0.56	2.38			0.31
		5		0.25		2.63				2.06			0.88	1.44		0.06		0.50						0.06	0.75	0.06		0.44	7.94		1.38	1.25

Table A 11 Percent cover of soft coral families 2019. 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	Ellisellidae	Heliporidae	Neptheidae	Xeniidae	Gorgoniidae like	Other
Barron Daintree	Low Isles	5	0.47		6.35		0.10	0.26		0.01		
Barron Daintree	Snapper North	2	0.06		3.29	2.17						
Barron Daintree	Snapper North	5	0.02		0.81	0.03				1.02		
Barron Daintree	Snapper South	2	0.15		0.13	0.08		3.08				
Barron Daintree	Snapper South	5	0.01		6.94		0.06	5.25				
Johnstone Russell-Mulgrave	Green	5	0.35		0.13			0.13	0.01			0.06
Johnstone Russell-Mulgrave	Fitzroy East	2	0.28		1.13	0.28			0.01			
Johnstone Russell-Mulgrave	Fitzroy East	5	0.99		6.44	0.13			0.10			
Johnstone Russell-Mulgrave	Fitzroy West	2	3.81		0.06							
Johnstone Russell-Mulgrave	Fitzroy West	5	3.13		0.13	0.03						
Johnstone Russell-Mulgrave	Fitzroy West LTMP	5	2.34		0.07	0.02	0.03					0.12
Johnstone Russell-Mulgrave	Franklands East	2	0.09		0.19	0.38		0.13		0.14		
Johnstone Russell-Mulgrave	Franklands East	5	0.23		2.06	0.06				0.02		
Johnstone Russell-Mulgrave	Franklands West	2	0.52			3.53		0.06	0.01			
Johnstone Russell-Mulgrave	Franklands West	5	0.14			0.13						
Johnstone Russell-Mulgrave	High East	2	0.82		7.06	0.06						
Johnstone Russell-Mulgrave	High East	5	0.10		9.56							
Johnstone Russell-Mulgrave	High West	2	0.59		0.06			3.75				
Johnstone Russell-Mulgrave	High West	5	0.13		1.25			2.13				
Herbert Tully	Barnards	2	0.20	0.06	1.06					0.79		
Herbert Tully	Barnards	5	0.18	0.13	3.31					0.48		
Herbert Tully	Dunk North	2	0.29		0.19	0.03				0.10		
Herbert Tully	Dunk North	5	0.10		0.19					0.79	1.06	
Herbert Tully	Dunk South	2	0.14		0.88	0.22						
Herbert Tully	Dunk South	5	0.13		2.13							
Herbert Tully	Bedarra	2	0.02		0.19							
Herbert Tully	Bedarra	5	0.13	1.00	4.25				0.02			

(sub)-region	Reef	Depth	Alcyoniidae	Anthothelidae	Briareidae	Clavulariinae	Ellisellidae	Heliporidae	Neptheidae	Xeniidae	Gorgoniidae like	Other
Burdekin	Palms East	2	0.17									
		5	0.37							0.02		
	Palms West	2	2.03		0.13	0.84			0.90		0.06	
		5	2.02		3.25	0.09	0.13		0.25		0.19	
	Havannah North	5	0.09		0.90	0.23		0.03		0.02		0.03
	Havannah	2	0.03		2.25				0.03			
		5	0.06		7.94		0.13					
	Pandora	2	0.05									
		5	0.05			0.28						
	Pandora North	5	0.29		9.63	1.86						
	Lady Elliot	2	0.06									
		5	0.12		0.06		0.06				0.06	
	Magnetic	2	0.04									
		5	0.23	0.13	0.19							
	Hayman	5	0.39		1.12				0.02			0.07
Proserpine	Langford	5	0.54		0.16							0.07
	Border	5	1.80		0.17				0.01	0.03	0.20	0.03
	Hook	2	0.34		0.31							
		5	0.58		0.56							
	Double Cone	2										
		5	0.02		0.19							
	Daydream	2							0.01			
		5										
	Dent	2	0.30		4.00				0.03			
		5	0.18		0.31							
	Shute Harbour	2	0.71		0.25				0.03			
		5	0.43						0.01			
	Pine	2	0.09		0.13							

(sub)-region	Reef	Depth	Alcyoniidae	Anthothelidae	Briareidae	Clavulariinae	Ellisellidae	Heliporidae	Neptheidae	Xeniidae	Gorgoniidae like	Other
	Pine	5	0.02		0.63							
	Seaforth	2	0.43	0.06	1.25						0.06	
		5	0.09	0.94	1.56					0.04		
	Barren	2	0.29		0.19			0.13		0.72		
Fitzroy		5	0.02							0.80		
	North Keppel	2										
		5	0.06									
	Middle	2	0.04							0.01		
		5	0.08									
	Keppels South	2	0.06							0.11		
		5	0.01							0.05		
	Pelican	2	0.02							0.01		
		5	0.66	1.06		0.06	0.25		0.01	0.03	1.19	
	Peak	2	0.07	0.19							0.06	
		5	0.10						0.03	0.03	1.31	

Table A 12 Percent cover of Macroalgae groups 2019. Genera for which cover exceeded 0.5% on at least one reef are included, rare or unidentified genera are grouped to 'Other within major classes of Macroalgae'.

(sub)-region	Reef	Depth	Rhodophyta (red algae)							Chlorophyta (green algae)			Phaeophyta (brown algae)						
			Hypnea	Peyssonnelia	Calcareous	Liagora	Amphiroa	Asparagopsis	Other	Caulerpa	Halimeda	Other	Spatoglossum	Padina	Dictyota	Lobophora	Sargassum	Stypopodium	Other
Barron Daintree	Low Isles	5			0.23				0.23	0.03	0.73	0.23		0.17					0.20
	Snapper North	2	0.04	0.38	0.71			3.88	35.92	0.46	2.54	0.25		0.13	16.38	0.04			0.75
		5		0.19	0.06				0.38		0.13	0.06			4.38				0.13
	Snapper South	2		0.04	0.08				0.13			0.25							
		5	0.56	0.44	1.75				5.50						0.75	0.06			
Johnstone Russell-Mulgrave	Green	5			0.14				1.03		6.77	0.15							0.19
	Fitzroy East	2		0.06	0.13				0.31										
		5		0.38	0.06				1.50			0.06							
	Fitzroy West	2	0.13	0.13	0.31				0.44		0.06	0.25							
		5		0.31					0.31										0.06
	Fitzroy West LTMP	5							0.29		0.20								0.09
	Franklands East	2	0.25	0.06	0.13				0.38			0.06							
		5	0.19		1.31				0.50										0.63
	Franklands West	2	0.63	0.19	2.25				3.06		0.06				0.50	0.06			0.19
		5	0.19		3.19	3.06			19.38		0.13								
	High East	2	2.38		0.06				2.75						0.06				
		5	0.63	0.19	0.13		0.06		0.31			0.06							
	High West	2	0.75	0.13					4.38										
		5		0.06					1.06										

(sub)-region	Reef	Depth	Rhodophyta (red algae)							Chlorophyta (green algae)			Phaeophyta (brown algae)						
			Hypnea	Peyssonnelia	Calcareous	Liagora	Amphiroa	Asparagopsis	Other	Caulerpa	Halimeda	Other	Spatoglossum	Padina	Dictyota	Lobophora	Sargassum	Stypopodium	Other
Herbert Tully	Barnards	2	0.19	0.31					0.13						0.31				0.06
		5		0.38	0.06				0.19	0.31					0.38				0.06
	Dunk North	2		0.25	0.31				1.75						0.13	0.81	7.44		0.56
		5		0.56	0.06				0.19						0.44	0.50	0.38		
	Dunk South	2		0.31	1.44				1.63	0.06		0.19			0.50	4.25	10.75	0.13	0.94
		5		0.94	0.06				0.63	0.06					0.06	5.00	0.06		0.25
	Bedarra	2	1.38	0.25	0.13				1.50			0.31		0.06	3.31	0.94	15.25		2.50
		5		0.31	0.06										1.00	0.19	0.19		0.31
Burdekin	Palms East	2								1.31		0.06							0.13
		5								1.44		0.06							
	Palms West	2																	
		5							0.13										
	Havannah North	5							1.78	0.30		0.15		0.30	0.84	10.80	10.43		3.16
	Havannah	2		0.13					0.06			0.06			0.69	0.88			
		5		0.19					0.13							6.13	0.31		0.19
	Pandora	2		0.25	0.06			0.06	0.63	0.06		0.06			3.25	5.44	14.69		1.88
		5		0.19	0.06										1.56	5.50	3.00		0.13
	Pandora North	5							1.80					0.10	0.90	11.30	3.41		1.40
	Lady Elliot	2	10.63	2.50	0.31				3.13					0.44	3.56	0.13	2.31		1.13

(sub)-region	Reef	Depth	Rhodophyta (red algae)							Chlorophyta (green algae)			Phaeophyta (brown algae)						
			Hypnea	Peyssonnelia	Calcareous	Liagora	Amphiroa	Asparagopsis	Other	Caulerpa	Halimeda	Other	Spatoglossum	Padina	Dictyota	Lobophora	Sargassum	Stypopodium	Other
	Lady Elliot	5	0.06	0.94	0.56				1.56						2.44				0.31
	Magnetic	2		0.63	0.06				0.31						4.94	8.75	15.13		0.75
	Magnetic	5		1.13	0.44				0.88						2.13	1.69	4.75		0.50
	Hayman	5							1.88			0.10				0.10			1.29
Proserpine	Langford	5							0.13										0.03
	Border	5			0.05				0.15										0.10
	Hook	2		0.06	0.88				1.75	0.13		0.31				0.13			
	Hook	5			0.06				1.13			0.19							
	Double Cone	2	0.31		4.13		8.69	2.44	7.25			0.44		0.63	0.50	3.06	1.56		1.56
	Double Cone	5	0.50	0.38	1.38		0.38	0.44	5.19	0.06		0.19		0.25	0.38	0.81	1.38		3.38
	Daydream	2			3.06			4.19	5.50			0.25		0.63	1.94	0.94	0.06		0.63
	Daydream	5			0.56			0.81	2.69	0.06				0.19	0.06	0.19			
	Dent	2		0.06	0.06				0.50			0.13			0.06	0.56			0.06
	Dent	5		0.44					0.06							0.06			
	Shute Harbour	2		0.06				0.13	0.50							0.38			0.13
	Shute Harbour	5		0.19	0.06				0.31			0.13		0.06		0.13	0.06		0.31
	Pine	2	1.25	0.44	0.88		0.25		8.00	0.06	0.63	0.50			0.88	4.63	2.00		2.94
	Pine	5		0.44	0.13				0.38		0.69				0.13	4.50	0.13		0.13
	Seaforth	2	0.94	0.06	1.94				7.50	0.06	0.06	0.13		0.25	0.19	3.13	4.06		4.25
	Seaforth	5			2.31				0.25	0.06		0.19		0.25	0.06	3.69	0.31		1.31

(sub)-region	Reef	Depth	Rhodophyta (red algae)							Chlorophyta (green algae)			Phaeophyta (brown algae)						
			Hypnea	Peyssonnelia	Calcareous	Liagora	Amphiroa	Asparagopsis	Other	Caulerpa	Halimeda	Other	Spatoglossum	Padina	Dictyota	Lobophora	Sargassum	Stypopodium	Other
Fitzroy	Barren	2							1.20										0.07
		5		0.06					7.01							0.31			
	North Keppel	2		1.06					0.44							25.56			
		5		1.44					0.75						0.50	17.69			0.13
	Middle	2		2.31				0.06	0.75	2.75		0.06				15.50	1.81		0.31
		5		1.50					1.38	0.88					0.50	7.00	0.13		0.63
	Keppels South	2		4.33					0.94						0.19	7.65			0.06
		5		3.63					0.25						0.06	5.69			
	Pelican	2		0.25	2.75				10.19			0.13	0.88	0.06	1.00	5.50	34.25	0.19	3.31
		5		0.94	0.63				5.38			0.44			0.13	7.56	1.63		1.00
	Peak	2		0.75	3.38			1.25	33.75	0.06	2.94	0.06	3.50		0.06	5.06	16.56	1.19	1.06
		5		1.75	2.56				24.56	0.19	2.63					0.63			0.13

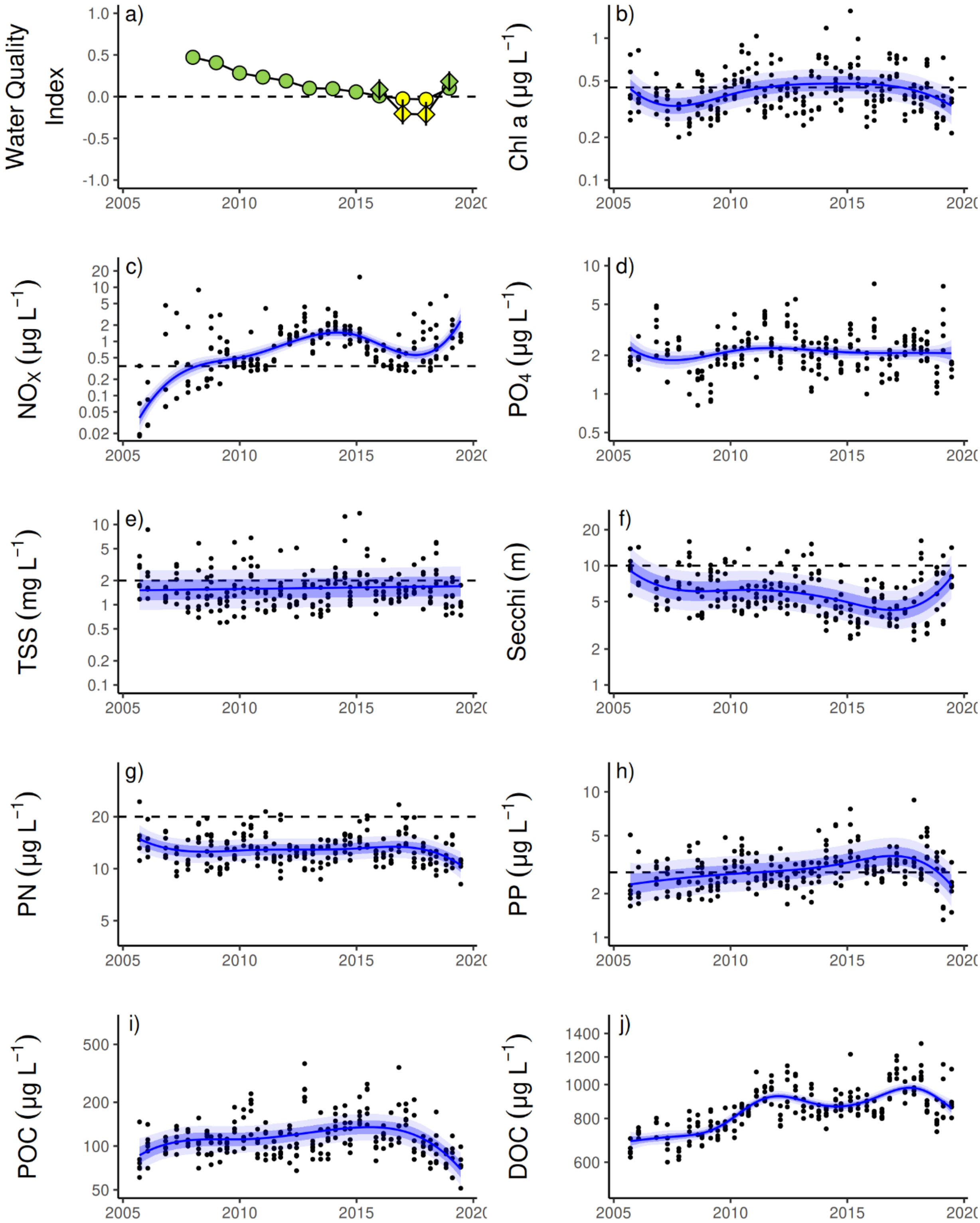


Figure A 10 Temporal trends in water quality: Barron Daintree sub-region. a) water quality index, b) chlorophyll a, c) nitrate/nitrite, d) Phosphate, e) total suspended solids, f) secchi depth, g) particulate nitrogen, h) particulate phosphorus, i) particulate organic carbon and j) dissolved organic 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 Gruber *et al*. (2020).

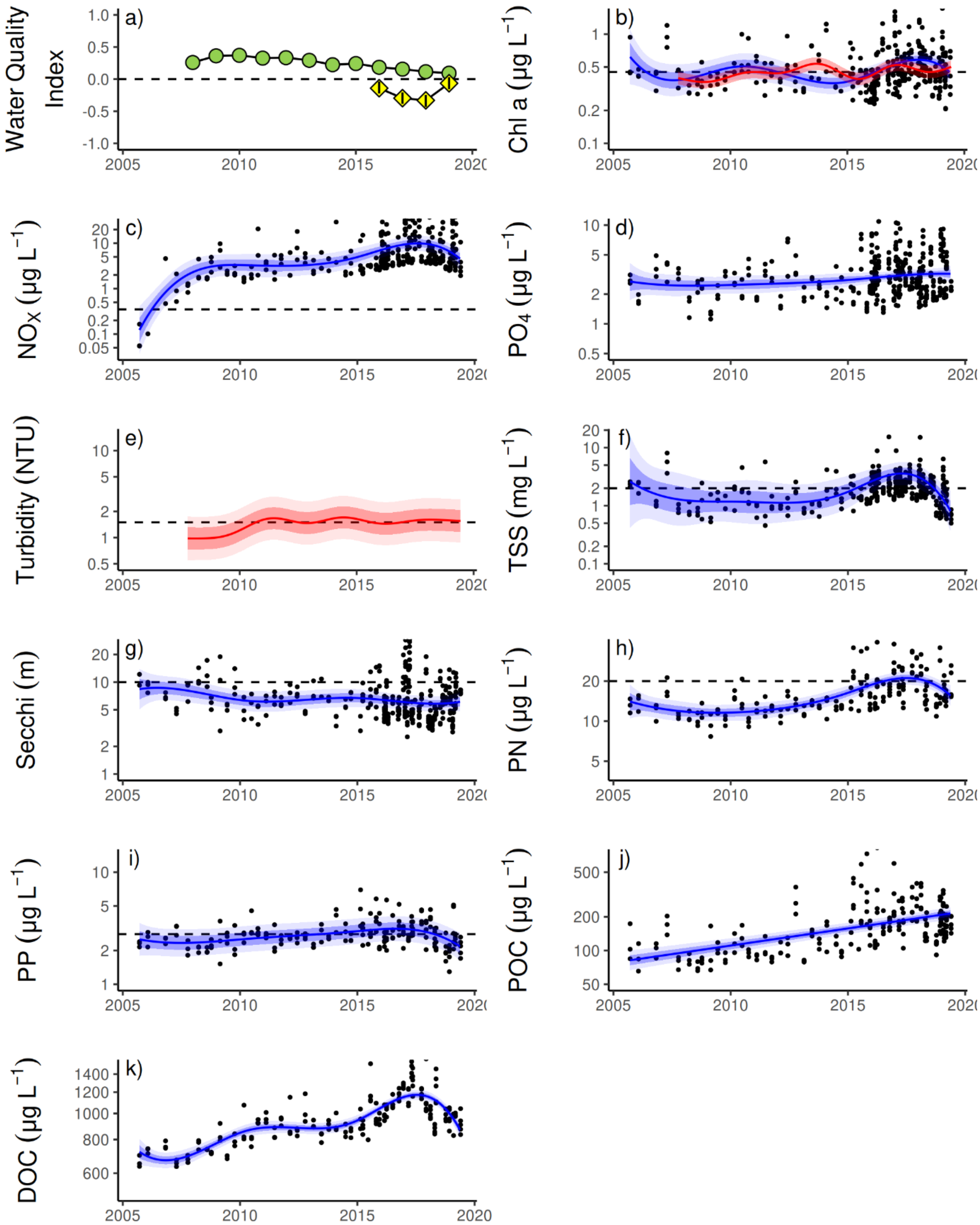


Figure A 11 Temporal trends in water quality. Johnstone Russell-Mulgrave sub-region. a) water quality index, b) chlorophyll a, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus j), particulate organic carbon and 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 Gruber *et al.* (2020).

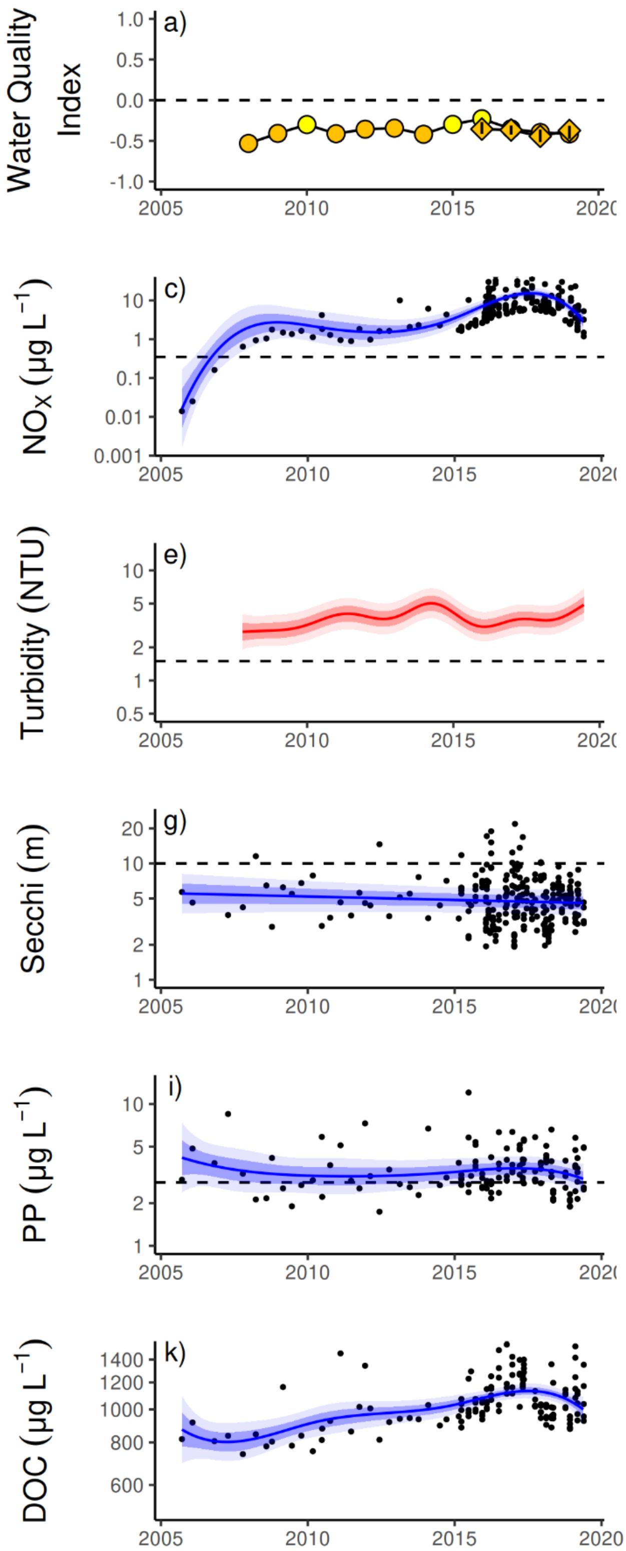


Figure A 12 Temporal trends in water quality: Herbert Tully sub-region. a) water quality index, b) chlorophyll a, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved 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 Gruber *et al*. (2020).

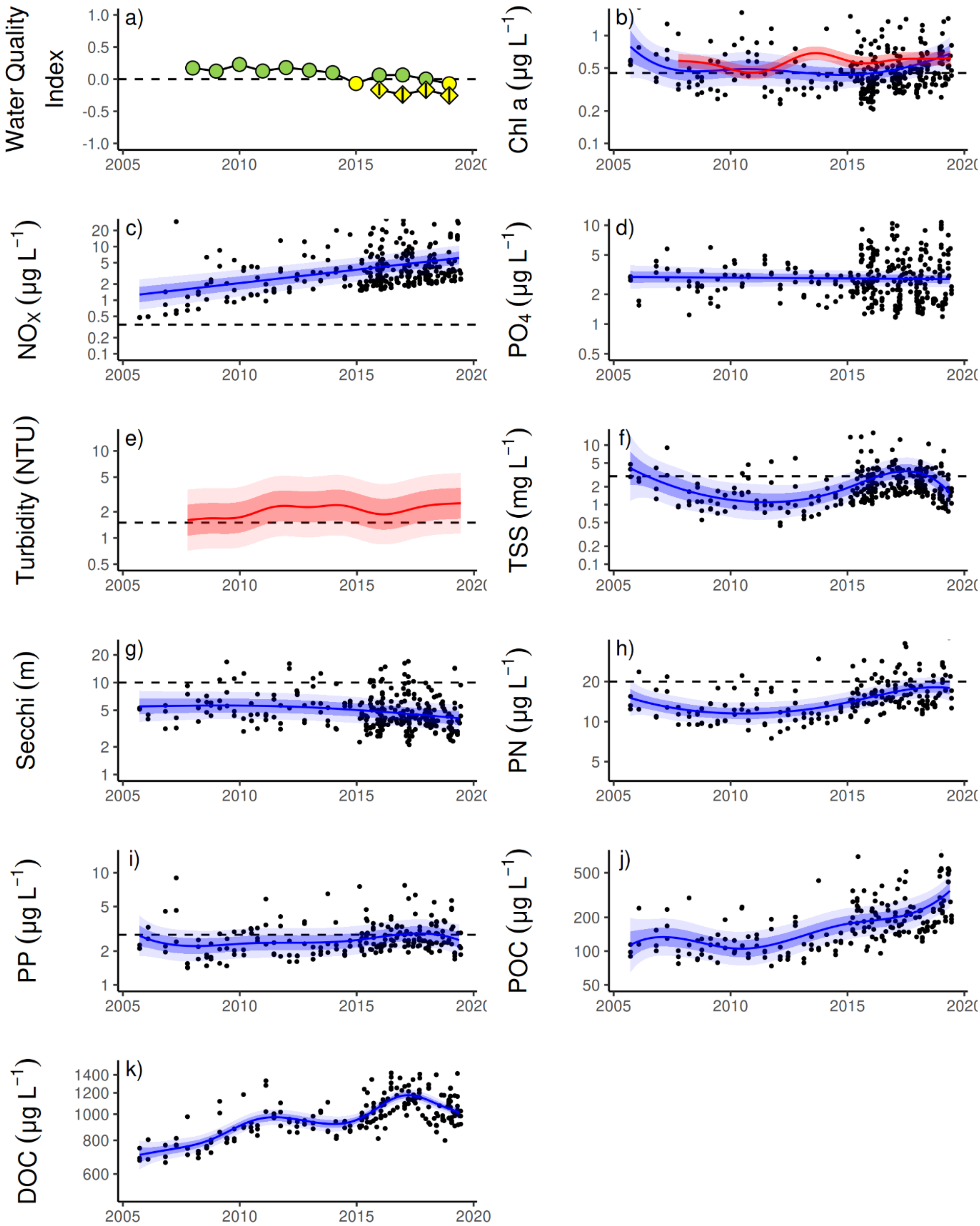


Figure A 13 Temporal trends in water quality: Burdekin region. a) water quality index, b) chlorophyll a, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved organic 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 Gruber *et al.* (2020).

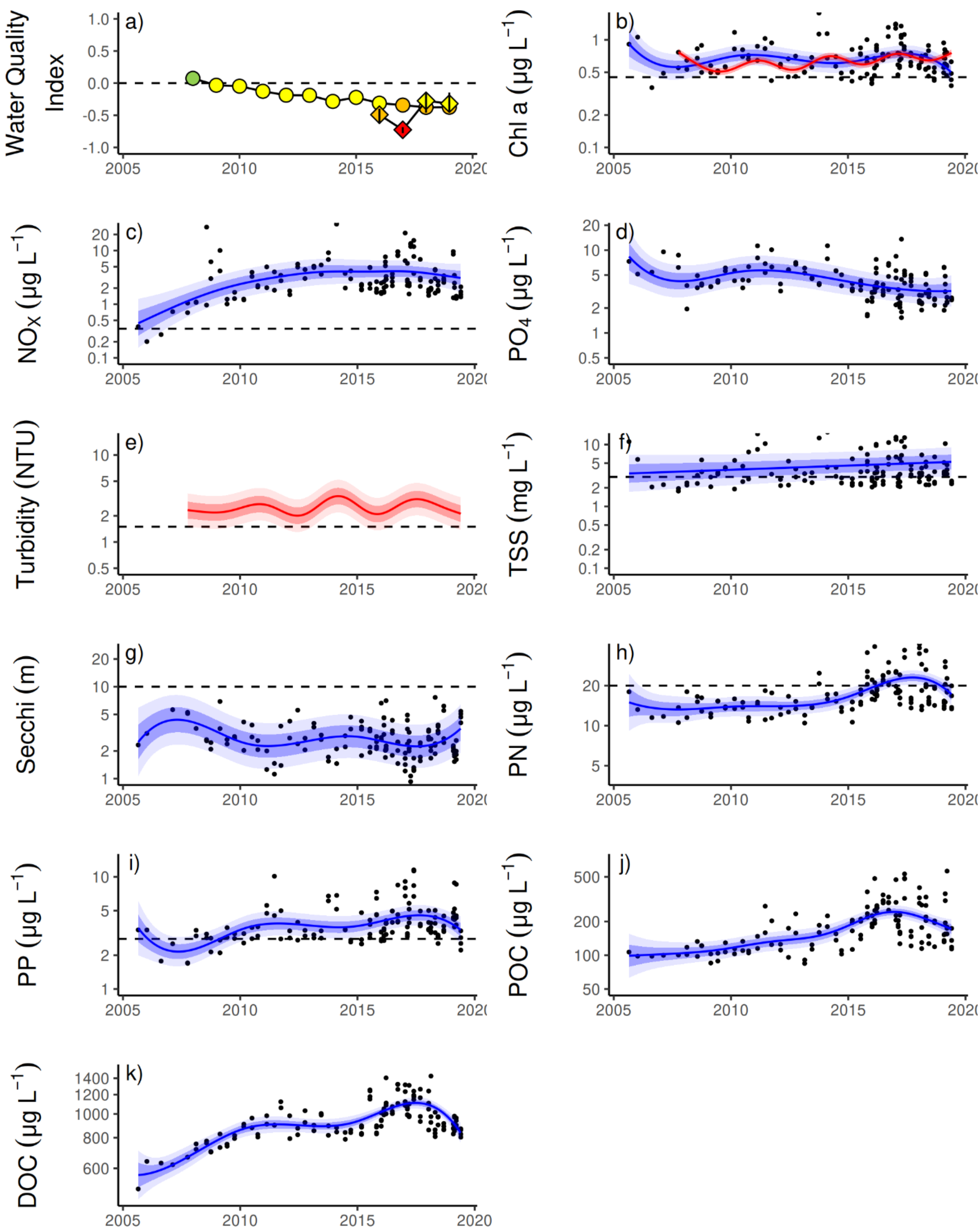


Figure A 14 Temporal trends in water quality. Mackay-Whitsundays region. a) water quality index, b) chlorophyll a, c) nitrate/nitrite, d) Phosphate e) turbidity, f) total suspended solids, g) secchi depth, h) particulate nitrogen, i) particulate phosphorus, j) particulate organic carbon and k) dissolved 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 Gruber *et al*. (2020).

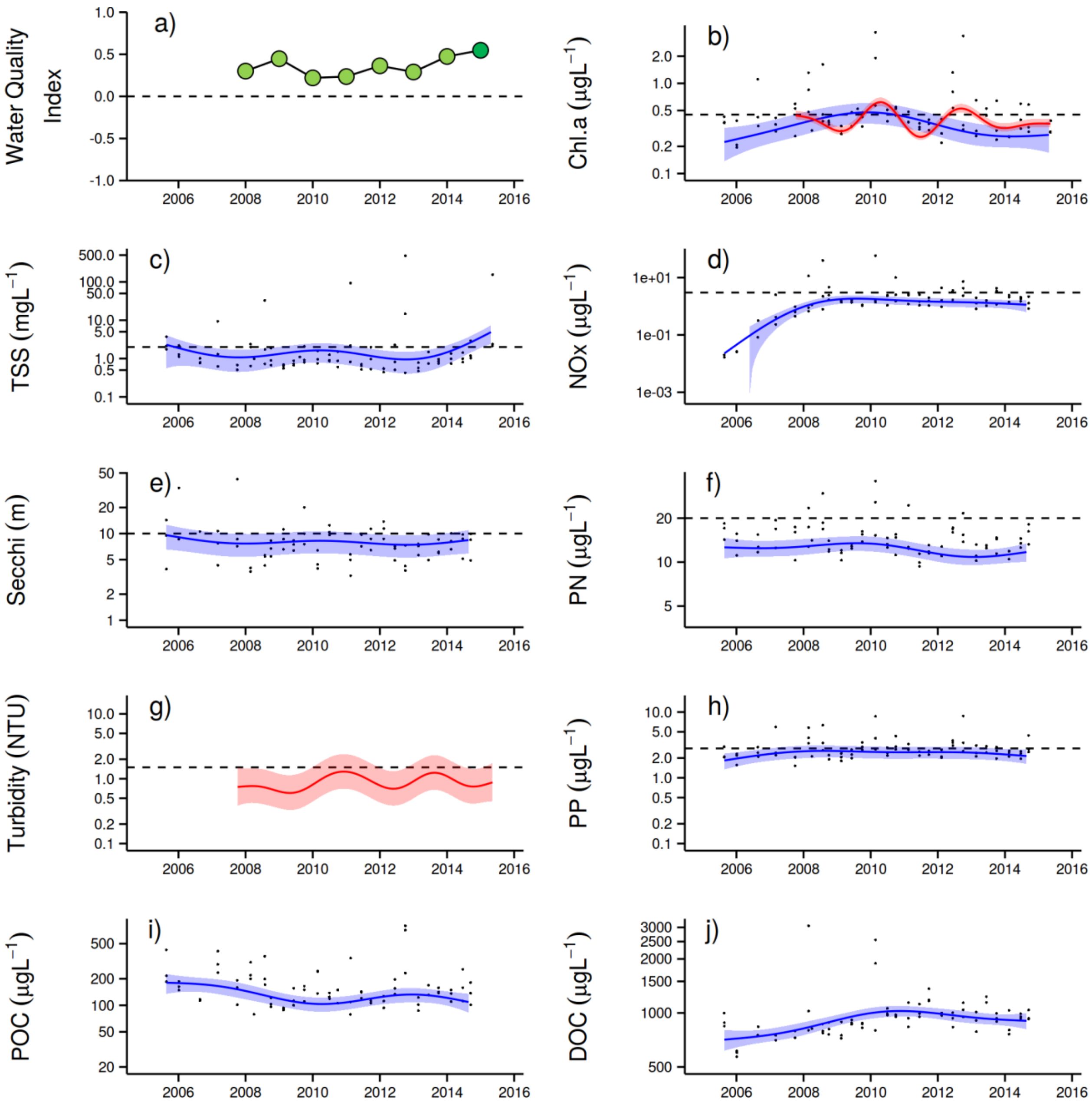


Figure A 15 Temporal trends in water quality: Fitzroy region. a) water quality index, b) chlorophyll a, c) total suspended solids, d) nitrate/nitrite, e) secchi depth, f) particulate nitrogen, g) turbidity, h) particulate phosphorus, i) particulate organic carbon and j) dissolved organic carbon. Water 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 - h and calculated as described in Gruber *et al.* (2020). Trends in POC and DOC values are plotted here (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. Trends of records from ECO FLNTUSB instruments (b, g) are represented in red, individual records are not displayed. Dashed reference lines indicate guideline values (GBRMPA 2010). Extract from Gruber *et al.* (2020).

9 Appendix 2: Publications and presentations 2018–2019

Davidson J, Thompson A, Logan M, Schaffelke B (2019) High spatio-temporal variability in Acroporidae settlement to inshore reefs of the Great Barrier Reef. PLoS ONE 14(1):e0209771. https://doi.org/10.1371/journal.pone.0209771

Jonker MJ, Thompson A, Menéndez P, Osborne K. (2019) Cross-Shelf Variation Among Juvenile and Adult Coral Assemblages on Australia's Great Barrier Reef. *Diversity*, *11*, 85.

Mellin C, Matthews S, Anthony KRN, Brown SC, Caley JM, Johns KA, Osborne K, Puotinen M, Thompson A, Wolff NH, Fordham DA, MacNeil MA (2019) Spatial resilience of the Great Barrier Reef under cumulative disturbance impacts. Global Change Biology. 2019;00:1-15 DOI:10.1111/gcb.14625