Southern Inshore Zone: Coral Indicators for the 2025 Mackay-Whitsunday-Isaac Report Card

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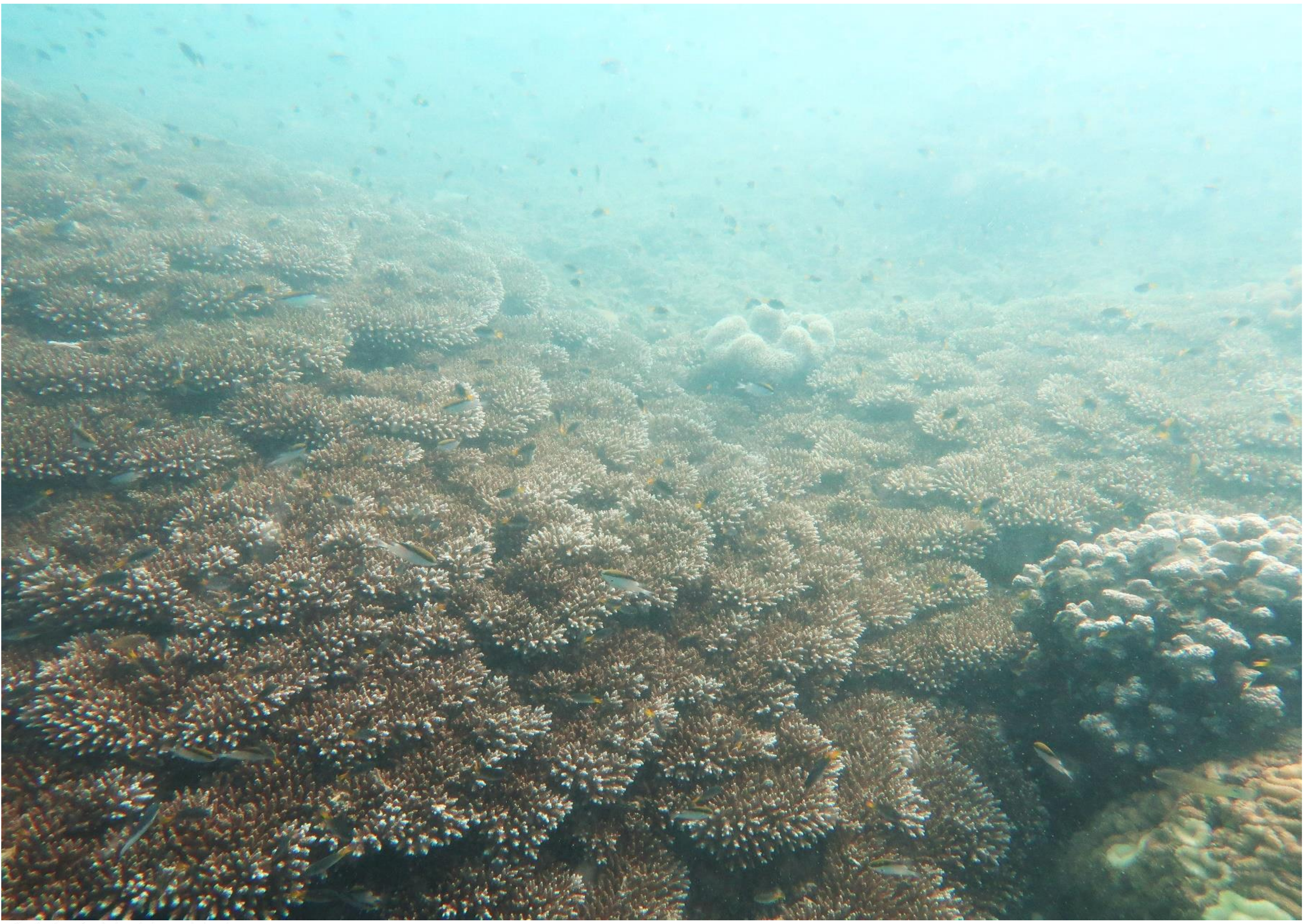

A report prepared for the Mackay-Whitsunday-Isaac Healthy Rivers to Reef Partnership

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Cover photo: Henderson Island reef slope showing extensive *Acropora* hard coral in July 2024.
Image: Johnston Davidson

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1 EXECUTIVE SUMMARY

This report presents the results of monitoring undertaken in 2024 for the coral component of the Mackay-Whitsunday-Isaac Healthy Rivers to Reef Partnership's Southern Inshore Monitoring Program. Coral communities were monitored by the Australian Institute of Marine Science under a 50/50 co-investment arrangement. These results form the basis of the coral indicator scores for the Southern Inshore Zone that inform the 2025 Mackay-Whitsunday-Isaac Report Card.

Between June and July 2024, the Australian Institute of Marine Science (AIMS) resurveyed benthic communities at permanent coral monitoring locations at five reefs in the Southern Inshore Zone of the Great Barrier Reef (GBR). The overall report card grade for community condition in 2024 remained at D ('poor'), based on a Coral Index score of 0.24 (Figure 1).

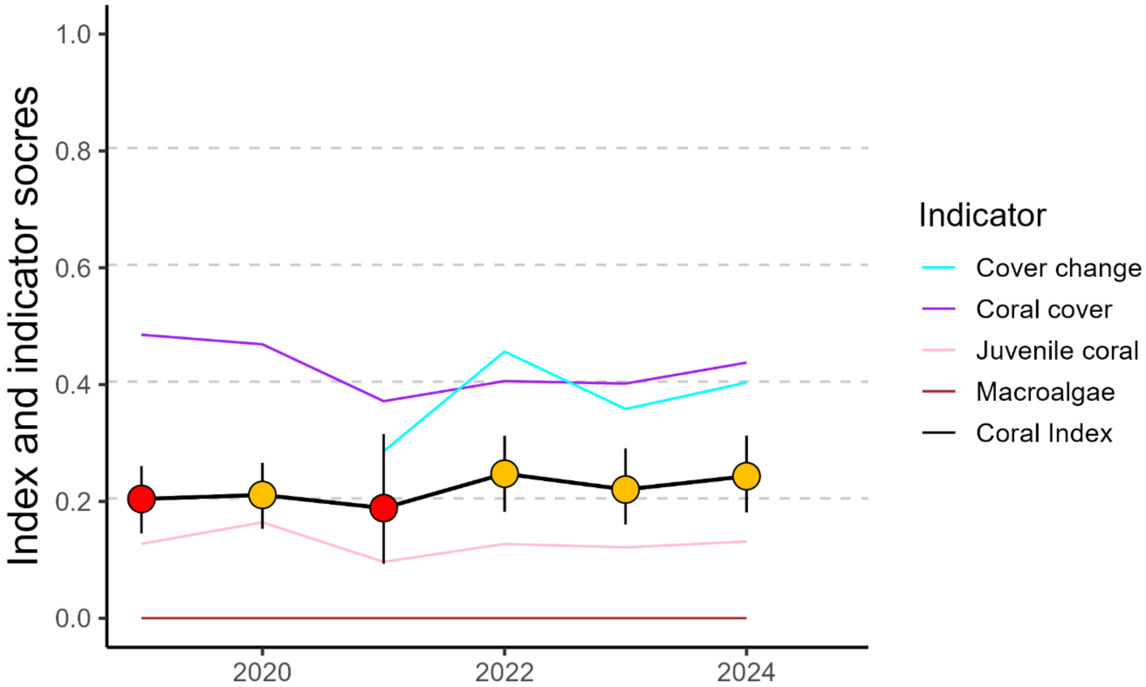

Figure 1 Coral Index and indicator scores. The Cover change indicator was added in 2022 and back calculated for 2021. Score categories are: >0.8 'very good' (A), > 0.60 ≤ 0.80 'good' (B), > 0.40 ≤ 0.60 'satisfactory' (C), > 0.20 ≤ 0.40 'poor' (D), 0 ≤ 0.20 'very poor' (E).

The Coral Index scores are based on the assessment of four indicators of coral condition:

- **Coral cover** - the proportion of the substrate occupied by living corals,
- **Macroalgae** - the proportion of the benthic algae cover comprised of large fleshy species,
- **Juvenile density** - the density of juvenile hard corals, and
- **Cover change** -the rate at which hard coral cover increased.

The marginal improvement in the Coral Index from 0.22 in 2023 reflects gains in the Coral cover and Cover change indicator scores and, to a lesser extent, the Juvenile density indicator score. The general increase in coral cover among reefs returned the Coral cover indicator into the 'satisfactory' grade (Figure 1)

A severe marine heat wave swept across the GBR in early 2024. While the level of heat-stress, measured as Degree Heating Weeks (DHW), was similar to that observed during a marine heat wave in 2020 the impacts to coral communities were less severe. During surveys in 2020 a high proportion (41%) of corals were bleached and loss of coral cover recorded in 2020 and 2021 were attributed to this bleaching event. In contrast very few corals were bleached during surveys in 2024 (3%), and coral cover had increased.

The ongoing high cover of macroalgae at most reefs saw the continued score of 0 for the Macroalgae indicator (Figure 1). The Juvenile density indicator was largely unchanged in 2024 as the density of juvenile corals remains very low on most reefs. In combination the 'very poor' scores for the Macroalgae and Juvenile density indicators continue to put downward pressure on the Coral Index. In contrast, the improved coral cover score, led by increased cover of hard corals at Henderson Island, signals a resumption of the gradual recovery following the impact of the 2020 marine heat wave. However, at most reefs both coral cover and juvenile density continue to be severely constrained by the sustained high levels of competitive macroalgae.

2 BACKGROUND

Inshore coral reefs of the Great Barrier Reef are impacted by multiple pressures including large scale disturbances such as cyclones and coral bleaching, through to more localised issues such as elevated levels of nutrients or suspended sediments that may result from activities in the coastal zone and adjacent catchments (Waterhouse *et al*. 2024). Successful management of coral communities requires the ability to identify where and when the resilience of communities is compromised and then identify and remediate causative pressures.

The Healthy Rivers to Reef Partnership (HR2RP) was created in October 2014 with the objective of using a collaborative, community-led approach to inform long-term management of the region's waterways and marine environments. In October 2015, the pilot report card was released which provided a snapshot of waterway health in the region.

The HR2RP identified a knowledge gap in the Southern Inshore Zone of the report card and, following an initial scoping study in October 2017 by Sea Research (2018), co-invested with the Australian Institute of Marine Science (AIMS) to establish a long-term monitoring project of corals in the area. The design spans a gradient in water quality from the coast out to the Percy Island group some 80 km offshore.

The sampling methods used are consistent with those used more broadly by AIMS under the Marine Monitoring Program (MMP). The MMP has strongly invested in the development of indicator metrics that focus on coral community resilience as a tool for synthesising coral monitoring. The coral Index, which is based on a series of indicators, is central to reporting of coral community condition across regional and state level report cards. There are considerable efficiencies in terms of indicator development, quality control and reporting in following the standards for sampling and analysis developed by the MMP.

This report presents the sixth annual survey of five permanent coral monitoring locations in the Southern Inshore Zone reported by the Mackay-Whitsunday-Isaac HR2RP Report Card. The purpose of this report is to provide a description of reef communities observed in 2024 that expands on the necessarily succinct summary of overall condition presented by the 2025 Mackay-Whitsunday-Isaac Report Card.

3 METHODS

3.1 Sampling Design

Coral communities are monitored along permanently marked transects. The selection of sites and construction of transects occurred in January and May of 2019, as reported in detail in Davidson *et al.* (2019).

In brief, suitable sites were identified at five fringing reefs located along the gradient in water quality from the very turbid waters close to the coast through to the clearest waters some 80km offshore (Figure 2).

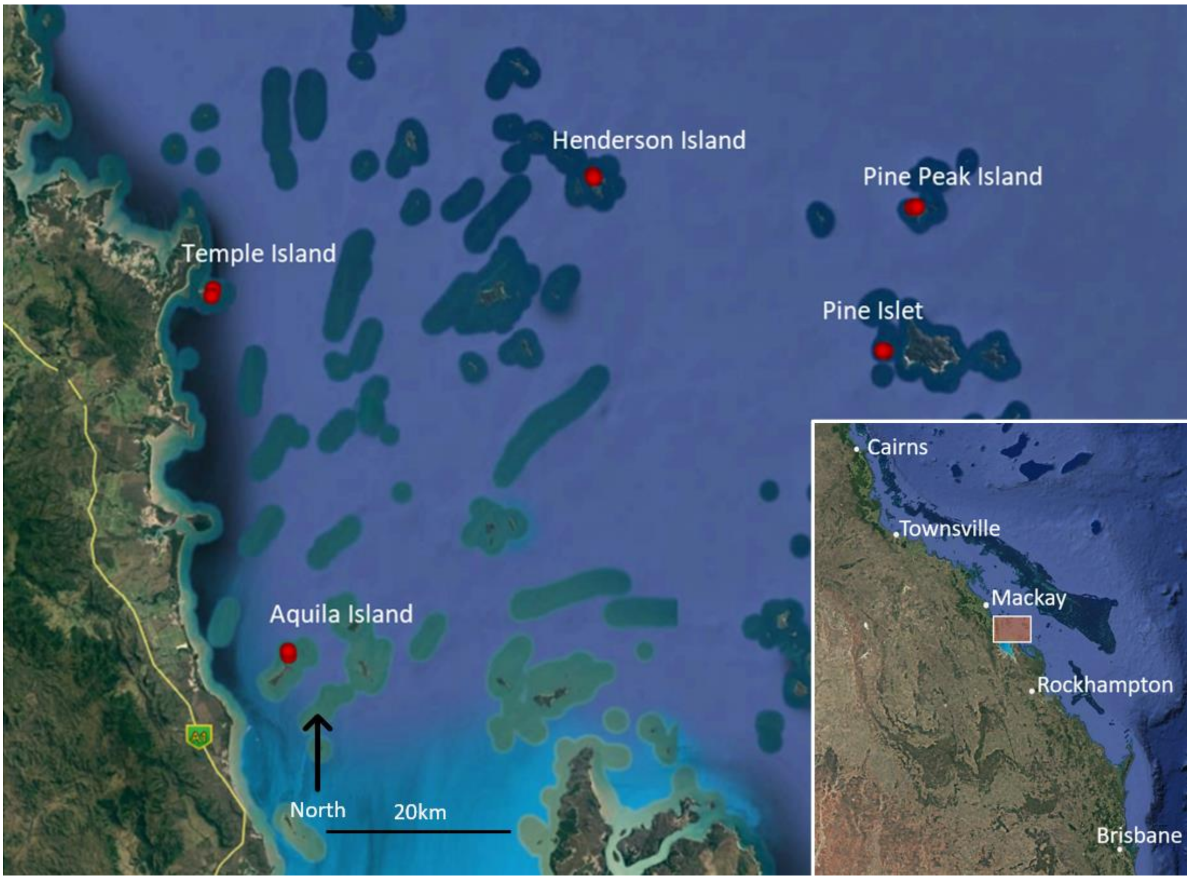

Figure 2 Map showing islands selected of coral monitoring in the Southern Inshore Zone. Insert shows location of monitoring area in relation to the Queensland coastline.

At each reef, two replicate sites separated by at least 150 m were selected haphazardly from the surface with the only limitations being that they were positioned on areas of substrate suitable for corals. Within each site, five transects of 20 metre length were constructed to follow the depth contour of the site. Each transect was separated from the previous by a gap of 5 m and marked with a steel fence post ‘star-picket’ at the start and a section of 10 mm steel rod at both the 10 m and end marks. In recognition of the importance of depth as a determinant of coral community composition (e.g., Thompson *et al.* 2014), transects were replicated at both 2 m and 5 m depths below lowest astronomic tide datum (LAT) at Pine Peak Island and Pine Islets as predicted by Navionics electronic charts at the time of site construction.

Sites at Henderson Island were setup in 2018 by a third party and parts of some transects at site 1 were set 1-3 m deeper than the intended 5 m datum. In 2022 the last rod at Transect 3 of the 5 m depth at Site 1 was relocated to better follow the depth contour. In 2024 the same was done for the last rod at Transect 2. In addition to keeping transects at a consistent depth these slight amendments improve dive logistics based on the use of DCIEM dive tables as per Australian Scientific Dive Standards. This is an importance consideration given the accessibility of these sites.

At Temple Island and Aquila Island the reef slope transitioned to sand at 1-1.5 m below LAT and as such transects were set at 1 m below LAT only. Additional details including the GPS waypoints marking the start of each site and depth combination along with compass directions along each transect are provided in Table A 9.

Reefs were monitored on visits in June and July 2024 (Table 1).

Table 1 Dates of coral monitoring.

Island	2019	2020	2021	2022	2023	2024
Pine Peak Island	27th January	26th May	6th March	5th June	23rd June	12th June
Pine Islets	28th January	27th May	6-7th March	4th June	22nd-23rd June	11-12th June
Henderson Island	29th January	25th -26th May	7th March	4th June	22nd June	12th June + 13th July
Temple Island	27th May	27th-28th May	3rd June	3rd June	24th June	12th July
Aquila Island	27th May	12th July	3rd June	6th July	24th June	12th July

3.2 Sampling Methods

3.2.1 Photo Point Intercept Transects

Benthic cover was estimated using photo point intercept transects (PPIT, Jonker *et al.* 2020). Along the upslope side of each transect line, digital images of the substrate were taken at ~40 cm elevation at 50cm intervals. Benthos beneath five evenly spaced points on each image was identified to the finest taxonomic resolution possible, typically genus level for corals and larger algae. In addition, the state of bleaching observed at each point was recorded as one of three levels: fully bleached, partially bleached, and non-bleached. A total of 32 images were analysed from each transect. 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 stored transect images. For data quality assurance all identified points were checked by a second observer.

3.2.2 Juvenile Coral Surveys

The number of juvenile coral colonies were counted *in situ* along the permanently marked transects. Corals in the size classes: 0-2 cm and >2-5 cm found within a strip 34 cm wide (data slate length) positioned on the upslope side of the transect line were identified to genus level and recorded. Importantly, this method aimed to record only those small colonies assessed as juveniles, i.e., having resulted from the settlement and subsequent survival and growth of coral larvae, and so did not include small coral colonies considered to have resulted from the fragmentation or partial mortality of larger colonies.

3.2.3 Scuba Search Transects

Scuba search transects documented the incidence of disease and other agents of coral mortality and stress observed at the time of survey. This method followed closely the Standard Operation Procedure Number 9 of the AIMS Long-Term Monitoring Program (Miller *et al.* 2020) and serves to help identify probable causes of any declines in coral community condition.

For each 20 m transect a search was conducted within a 2 m wide belt transect centred on the marked transect line and the incidence of: coral disease, coral bleaching, coral predation by *Drupella* or crown-of-thorns sea stars, overgrowth by sponges, smothering by sediments, or physical damage to colonies was recorded.

3.3 Coral Community Indicators

The indicators and methods used to derive report card scores for coral communities are a subset of those used for the Reef Report Card (Thompson *et al.* 2022), the development of which are described in detail in Thompson *et al*. (2020). The indicators, Coral cover, Macroalgae and Juvenile density have been used since the start of this program. The Cover change indicator requires repeated observations that span a period during which the coral communities were not subjected to an acute pressure, such as a marine heatwave or tropical cyclone. As most reefs were impacted by coral bleaching in 2020, with flow on effects evident in 2021, this indicator was first implemented in 2022. Back calculated scores for Cover change in 2021 are supplied, although values from 2021 should be treated with caution as they relate only to changes at Aquila, Temple and Pine Islets 2 m. AIMS does not support the inclusion of the Community composition indicator in this region based on analysis by Thompson *et al.* (2022) that demonstrates this indicator primarily varies in response to changes in coral cover, which is captured by the Coral cover indicator. In addition, the low cover of corals at several reefs are not considered an aspirational condition on which to set a baseline. This section provides an overview of the rationale for the selection of the four indicators used to assess coral community condition and how they are scored. A full description of these indicators can be found in Thompson *et al.* (2020).

3.3.1 Coral cover

The most tangible and desirable indication of a healthy coral community is an abundance of coral. The coral cover indicator scored reefs based on the proportional area of substrate covered by both 'Hard' (order Scleractinia) and 'Soft' (subclass *Octocorallia*) corals.

$Coral\ cover_{ij} = hard\ coral\ cover_{ij} + soft\ coral\ cover_{ij}$ where i = reef and j = time.

While high coral cover provides a good indication that environmental conditions are supportive of the growth and survival of corals, low cover does not necessarily indicate the opposite. Coral communities are naturally dynamic, being impacted by acute disturbance events such as cyclones (Harmelin-Vivian 1994; Osborne *et al.* 2011), temperature anomalies (Berkelmans *et al.* 2004) and, in coastal areas, flooding (van Woesik 1991; Jones and Berkelmans 2014). The Juvenile and Macroalgae indicators were included as they represent the potential for coral communities to recover from disturbances.

3.3.2 Macroalgae

Macroalgae may suppress the recovery of coral communities through a variety of mechanisms ranging from direct competition with surviving colonies through to physical and chemical suppression of the recruitment process (McCook *et al.* 2001; Hughes *et al.* 2007; Foster *et al.* 2008; Hauri *et al.* 2010, Clements *et al.* 2020). To ensure that the assessment of macroalgae cover was independent of the cover of corals, and that differences in available space for algal colonisation were considered, the indicator for macroalgae was defined as the proportion of the total algae cover that is made up of large fleshy species, collectively macroalgae.

$Macroalgae\ proportion_{ij}$ = $Macroalgae\ cover_{ij}$ /$Total\ algae\ cover_{ij}$ where i = reef and j = time.

3.3.3 Juvenile density

The density of juvenile corals is an indicator of the successful completion of early life history stages of corals from gametogenesis through fertilisation, larval survival, settlement to the substrate and then early post settlement survival, all of which may be impacted by poor water quality (reviewed by Fabricius 2005; van Dam *et al.* 2011; Erftemeijer *et al.* 2012). The juvenile indicator was derived from counts of juvenile hard corals along belt transects and converted to a density per area of potentially colonisable hard substrate, estimated as the proportion of benthos identified as algae along the co-located point intercept transects.

$Juvenile\ density_{ij} = J_{ij}\ /\ A_{ij}$

Where J= count of juvenile colonies < 5cm in diameter, A = area of transect occupied by algae (m^2), i = reef and j = time.

Selection of thresholds for the scoring of this metric was based on the analysis of recovery outcomes for MMP and AIMS' Long-Term Monitoring Program (LTMP) reefs up to 2014 (Thompson *et al*. 2020). From these time series a binomial model was fitted 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. Consequently, a juvenile density of 4.6 m^{-2} was considered to be the threshold at which the indicator score improves from 'poor' to 'satisfactory'. The upper threshold density, at which the probability was > 80% for coral cover to recover at predicted rates, was calculated at 13 m^{-2}, the indicator score improving again from 'good' to 'very good'.

3.3.4 Cover change

While high coral cover can justifiably be considered a positive indicator of community condition, the reverse is not necessarily true. Low cover may occur following acute disturbance and, hence, may not be a direct reflection of the community's resilience to underlying environmental conditions. For this reason, in addition to considering the actual level of coral cover, we assess the rate at which hard coral cover increases as a measure of recovery potential. The assessment of rates of cover increase is possible as rates of change in hard coral cover on inshore reefs have been modelled (Thompson *et al.* 2020), allowing estimations of expected increases in cover for communities of varying composition to be compared against observed changes.

A Bayesian framework was used to permit propagation of uncertainty through predictions of expected hard coral cover increase from separate models applied to fast growing corals of the family

Acroporidae, and the combined cover of all other hard corals. Note that the example presented below for Acroporidae (*Acr*), has the same form as that applied for Other Corals (*OthC*) if these terms are exchanged where they appear in the equations:

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

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

$$vAcr_i = \alpha + \sum_{j=0}^{J} \beta_j Reef_i$$

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

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

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

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

Where, $Acr_{it}, OthC_{it}$ and Sc_{it} are the cover of Acroporidae coral, other hard coral and soft coral respectively at a given reef at time (t). $eskK$ is the community size at equilibrium (100-proportion of area comprised of unconsolidated substrates) and $rAcr$ is the rate of increase (growth rate) in cover of Acroporidae. Varying effects of Reef (β_j) is also incorporated to account for spatial autocorrelation. Model coefficients associated with the intercept, and Reef (α_i and β_j) all had weakly informative Gaussian priors (the latter two with model standard deviation). The overall rate of coral growth parameters ($rAcr$ or alternatively $rOthC$) constituted the mean of the individual posterior rates of increase ($vAcr_i$ or alternatively $vOthC_i$).

3.3.5 Scoring of Indicators

To facilitate the reporting of coral community condition, the observed values for each indicator were converted to scores on a common scale of 0 to 1. For each indicator, observed levels were scaled against thresholds used by the MMP. These thresholds were set based on expert opinion and knowledge gained from the time-series of coral community condition collected by the MMP and LTMP. Upper bounds were set that represent values of indicators that were considered to represent communities in as good a condition as could be expected in the local environment (Figure 3 uses coral cover as an example). Conversely, lower bounds were set to represent minimal resilience (Table 2). While observations may exceed these limits, any such values will be capped at the minimum or maximum score (0 or 1 respectively). For the macroalgae indicator upper and lower bounds were set individually for each reef and depth to account for natural variation in macroalgal abundance across the steep gradient in water quality that exists in the inshore Great Barrier Reef. Selection of the reef-level thresholds were based on predictions of macroalgae proportion based on gradient boosted models (Ridgeway 2007). The models predict macroalgae proportion based on mean chlorophyll *a* and non-algal particulate (turbidity) concentrations for each reef derived from MODIS Aqua data sourced from the Bureau of Meteorology[1].

[1] *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 by the Bureau, Australian Institute of Marine Science, and the Commonwealth Scientific and Industrial Research Organisation.*

Table 2 Indicator score thresholds.

Indicator	Location	Upper bound (score=1)	Lower bound (score=0)
Coral cover	All	75%	0%
Macroalgae	Pine Peak Island 2 m	0.2%	3.4%
	Pine Peak Island 5 m	0%	6.3%
	Pine Islets 2 m	0.2%	5.4%
	Pine Islets 5 m	0%	6.4%
	Henderson Island 2 m	0.2%	3.9%
	Henderson Island 5 m	0%	6.7%
	Temple Island 1 m	0.3%	23%
	Aquila Island 1 m	0.3%	23%
Juvenile density	All	13 m^{-2}	0 m^{-2}
Cover change	All	2* upper 95% CI	Hard Coral cover declined

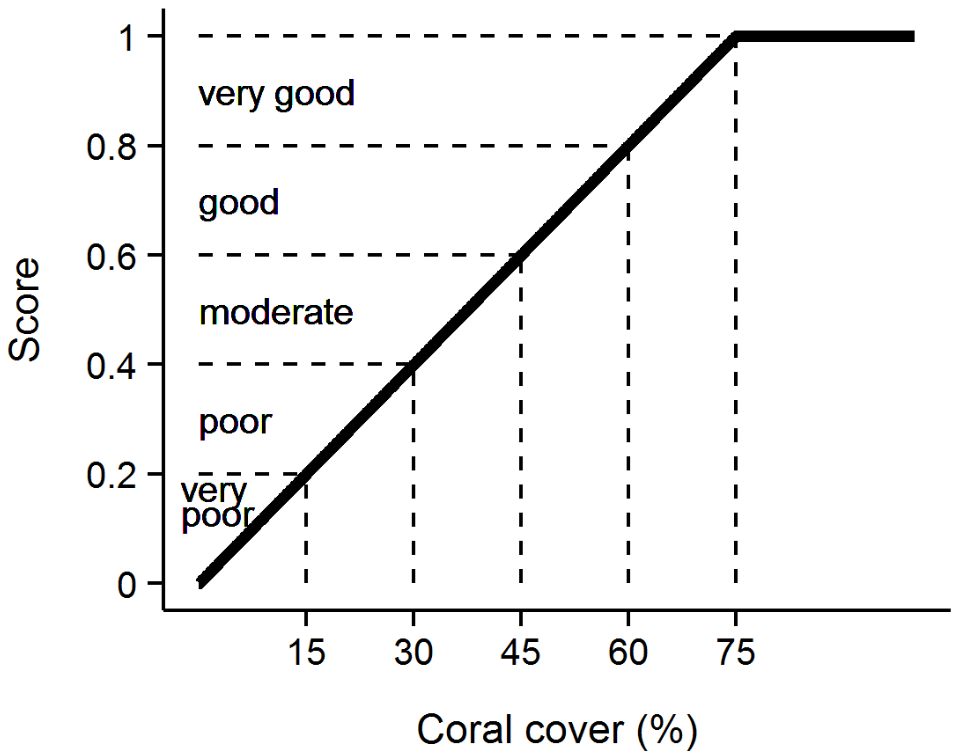

Figure 3 An example of a scoring diagram, here for the Coral Cover metric. Numeric scores and associated condition classifications based on observed coral cover are presented (see also Table 2).

3.3.6 Aggregation of Indicator Scores

The scaling of all scores to the common range of 0 to 1 allows the aggregation of scores across indicators at a hierarchy of spatial scales. At any given spatial scale, the mean of the individual indicator scores provides the Coral Index score. Within this report, indicator and index scores are presented at the scale of individual indicators at each reef and depth, and for the Southern Inshore Zone. Grades and associated condition classifications for coral communities were derived from the index scores, according to the conversions described in Table 3.

Table 3 Indicator scores, condition descriptions and report card grade conversions. Scores are rounded to the nearest single decimal place.

Score	Condition description	Grade
> 0.80	very good	A
> 0.60 ≤ 0.80	good	B
> 0.40 ≤ 0.60	satisfactory	C
> 0.20 ≤ 0.40	poor	D
0 ≤ 0.20	very poor	E

3.3.7 Data Analysis

A panel of plots provide temporal trends in the Coral Index and the indicators on which the index is based.

For each of the 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. Observed trends for individual reef and depth combinations (averaged over sites) are provided as grey lines. Annual Coral Index scores are the arithmetic mean of the three indicator scores; associated confidence intervals are derived from bootstrapped distributions of reef and depth level scores.

Genus level cover data for the current year are included in Appendix Table A 1, Table A 2, Table A 3 and Table A 4. In 2022 AIMS adopted an updated taxonomic classification scheme for hard corals based primarily on molecular studies that altered the accepted taxonomy of several coral species. The taxonomy adopted aligned with the World Register of Marine Species. This change means that it is not appropriate to compare values for genus richness of hard coral cover or juvenile hard corals with those presented in previous reports.

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 2. These additional plots breakdown cover and density of corals to the taxonomic level of Family. Due to the overall abundance of the family Acroporidae, this is split further into genus groups *Acropora* and *Montipora*. Photos representative of coral communities at each reef and depth in 2024 are at Appendix Figure A 3 (a-f) and Figure A 4 (a-b).

3.3.8 Key Pressures

Coral communities are susceptible to a range of pressures. Identifying these pressures and the associated drivers is essential in determining the likely cause of impacts to coral community condition. For inshore reefs of the GBR common disturbances to coral communities include physical damage caused by tropical cyclones (Osborne *et al.* 2011; De'ath *et al.* 2012), exposure to low salinity waters during flood events (van Woesik 1991; Jones and Berkelmans 2014), predation by corallivorous crown-of-thorn seastars (Pratchett *et al.* 2017), and anomalously high summer temperatures resulting in coral bleaching (Berkelmans *et al.* 2004; Sweatman *et al.* 2007), most recently summarised by Thompson *et al.* 2024. It is only once the influences of acute pressures have been accounted for that the potential impacts of chronic pressures such as elevated turbidity and nutrient levels can be inferred.

3.3.9 Thermal Stress

Thermal stress, resulting in coral bleaching, is an increasing threat to coral communities in a warming world (Schleussner *et al.* 2016). In 2019 temperature loggers (Vemco Minilog-II-T) were deployed to star pickets marking site 1, transect 1 at each of Pine Peak Island (2 m and 5 m), Henderson Island (2 m and 5 m), and Aquila Island (1 m). These loggers were retrieved annually. In 2024 the temperature loggers were upgraded to the RBR*solo*3T. As this time-series develops, an accurate temperature climatology for each location will be developed enabling the estimation of site-specific temperature stress metrics. In the interim, the mean of maximum summer temperatures from time-series of temperatures recorded by the MMP at Whitsunday Islands reefs has been adopted as a visual reference for temperatures recorded in the Southern Inshore Zone.

Satellite-based estimates of thermal stress resulting in coral bleaching were accessed to allow spatial and inter-annual comparisons of thermal stress across the Mackay Whitsunday Isaac reporting region. Thermal anomalies expressed as Degree Heating Weeks (DHW) were sourced from NOAA coral reef watch . Thresholds at which severe coral bleaching is likely are DHW values greater than eight (Lui *et al*. 2014). Realised severity of bleaching will depend on the pattern of warming and differences in the tolerances of coral species.

3.3.10 Runoff

Median discharge for the water-years 1990-1991 through to 2019-2020 are compared to the current year. Discharge data were sourced from the Queensland Government water monitoring portal. Correction factors to account for un-gauged portions of the catchment were applied to gauged discharge. These data were supplied by Dr Stephen Lewis from TropWater at James Cook University and represent those reported by the Great Barrier Reef Marine Monitoring Program.

3.3.11 Cyclones

Significant impacts to coral reefs in the GBR have been attributed to cyclone and storm damage (Osborne *et al.* 2011; De'ath *et al.* 2012). Due to the physical nature of damage associated with cyclones, impacts are readily identifiable by surveys the following winter. In addition, cyclones are well publicised and highly unlikely to go unnoticed. Verification of the potential impacts of past cyclones was assessed based on viewing seasonal cyclone tracks published online by the Australian Bureau of Meteorology.

3.3.12 Environmental Settings of Reefs.

Turbidity and nutrient levels are critical components of the aquatic environment and are fundamental determinants of benthic community composition and condition. For the reporting of coral community condition in inshore areas, nutrient availability determines the level of macroalgae cover that can be expected, influencing the thresholds set for scoring macroalgae on a site-specific basis (Thompson *et* *al.* 2020). In addition, the composition of sediments, as a proxy for the hydrodynamic setting of a site, is a useful covariate to consider in terms of coral community dynamics (Wolanski *et al.* 2005). For a detailed appraisal of both nutrient and sediment regimes in the local environment of the Southern Inshore Zone, see our baseline report, Davidson *et al.* (2019).

4 RESULTS

4.1 Pressures

4.1.1 Thermal Stress

Over the 2023-24 period in-situ temperature records showed temperatures at the monitored sites peaked on 5th - 6th February 2024, with the highest temperature at Aquila Island (30.7°C) marginally exceeding those recorded during the marine heatwave of 2020 (Figure 4). Temperatures at all three sites exceeded the baseline used, derived from long-term temperature records of reefs in the Whitsunday Islands.

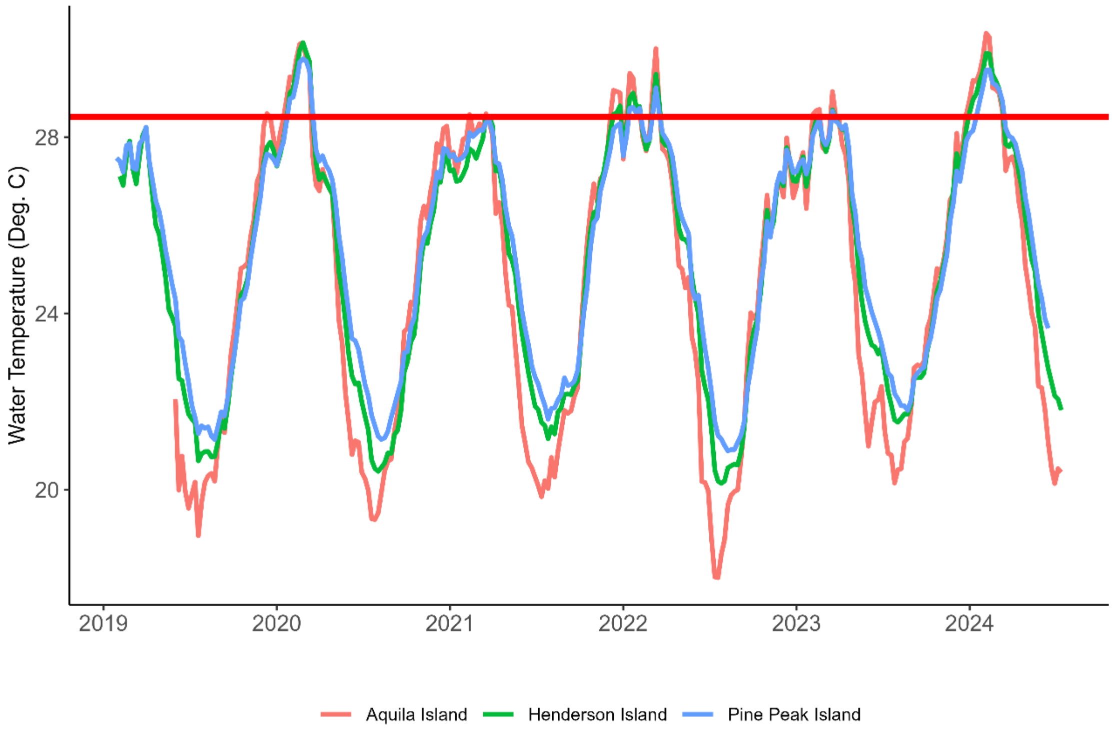

Figure 4 Temperature profiles recorded by in-situ loggers. The horizontal reference line was derived from the mean of the means of the hottest month each year observed over timeseries of *in-situ* temperature data available from reefs in Whitsunday Islands. This baseline excluded years in which bleaching was observed.

The *in situ* recorded temperature anomalies were reflected in estimates of Degree Heating Weeks (DHW) that show levels of heat stress across the region were the highest recorded over the past eight years, with DHW levels greater than ten predicting extreme bleaching (Figure 5). Indeed, thermal anomalies on the GBR in 2024 were the highest recorded to date (Cantin *et al.* 2024, Henley *et al.* 2024). It is notable that the impacts to coral communities in this region were much less severe than those caused by the 2020 event.

As an explanatory note, DHW estimates represent the sum of weekly mean temperatures that exceed the mean temperature of the hottest month in a location's climatology by at least one degree. DHW values aggregate over a rolling twelve-week period (Liu *et al.* 2014).

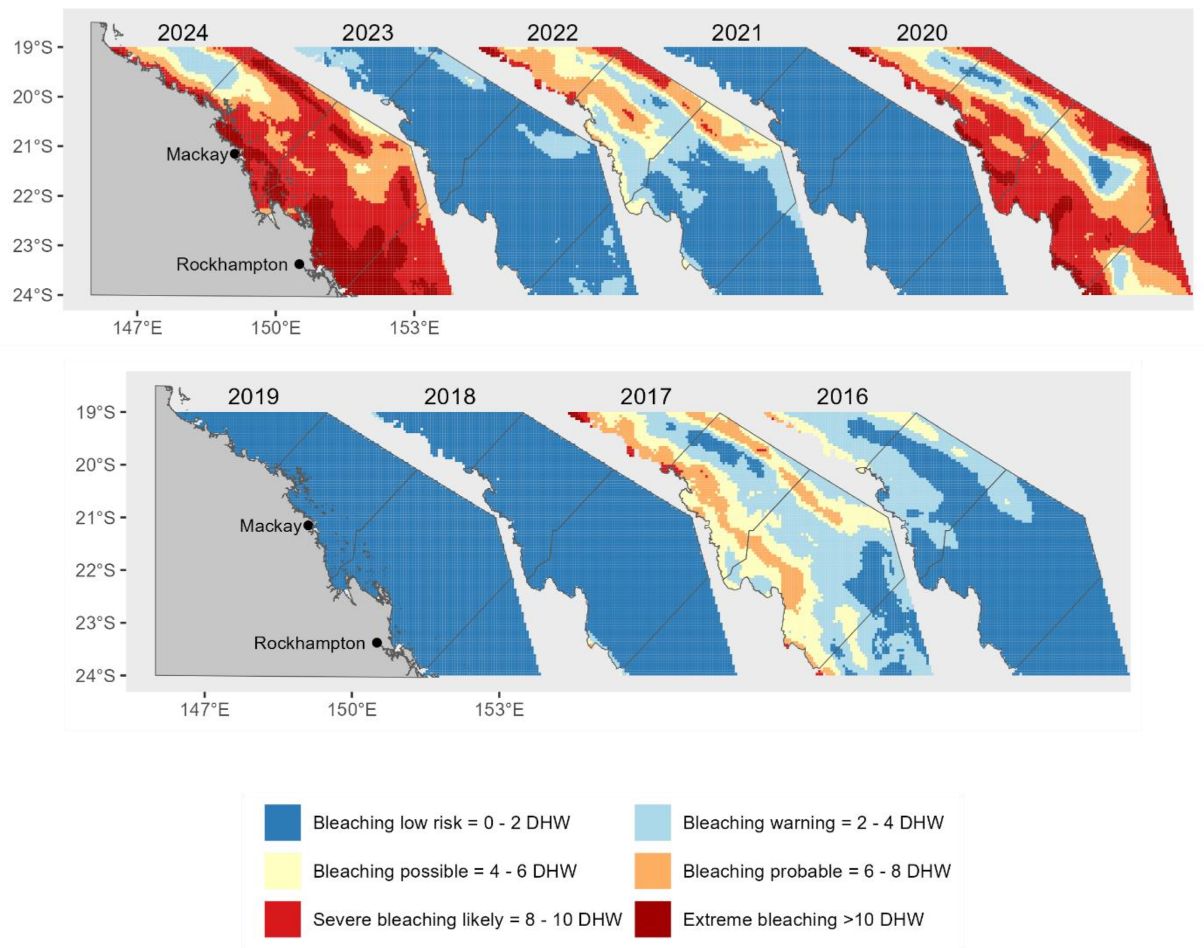

Figure 5 Annual estimates of thermal stress to corals. Data are the annual maximum degree heating week (DHW) estimates for each ~25 km^2 pixel. Data were sourced from NOAA coral reef watch. DHW values as indicators of thermal stress on the Great Barrier Reef are interpreted as follows: DHW values from 0 - 2: low risk of bleaching (i.e., normal summer conditions), 2 - 4: coral bleaching warning, 4 - 6: coral bleaching possible, 6 - 8: coral bleaching probable,8 – 10: severe coral bleaching likely, >10: extreme coral bleaching

4.1.2 Runoff

River flow data highlights a period of very high discharge in 2011 and again in 2013, with the amplitude of exceedance reduced in later years (Table 4). Discharge from the region's catchments over the 2023-2024 water-year (October to September) were at or below median levels in the Pioneer and Plane basins respectively but 2.0 times the median level for Waterpark Creek (Table 4). Although exposure to reduced salinity has proven lethal to coral communities in the inshore GBR (van Woesik 1991; Jones and Berkelmans 2014), the levels of discharge observed in this region since 2019 do not appear to have resulted in direct impacts to the coral communities monitored.

Table 4 Annual freshwater discharge for the catchment basins bordering the Southern Inshore Zone. Values represented as proportional to the long-term median (1991-2020). Flows are corrected for ungauged area of catchments. 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.

Basin	Gauge Station_Id	LT median (ML)	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024
Pioneer	125016A	616216	5.9	2.5	1.9	1	0.2	1	2.3	0.4	1.9	0.6	0.4	0.5	1.2	1.0
Plane	126001A, 126003A	1058985	3.9	2.4	1.8	0.8	0.4	0.9	2.4	0.4	1.2	1.1	0.6	0.5	1.4	0.6
Waterpark Creek	129001A	392614	4.4	1.4	4.7	2.7	1.9	1.7	2.5	1.4	0.7	1.4	1.7	2.1	1.5	2.0

4.1.3 Cyclones and Storms

There were no cyclones likely to have impacted reefs in the Southern Inshore Zone during the 2023-2024 cyclone season. Cyclone Jasper crossed the Queensland coast as a category 2 cyclone north of Cooktown in December 2023, while Cyclone Kirrily crossed as a category 1 just north of Townsville in February 2024 (the tracks of these cyclones have not been included in Figure 6). However, it should be noted that recovery from severe disturbance caused by cyclones can be slow, and exposure to high waves during past cyclones likely continues to influence coral cover.

Of the top six wave heights recorded by the Mackay wave buoy (Queensland Government 2024a) since 1975 four have occurred since 2010 and, in descending order, can be attributed to cyclones Dylan (2014), Ului (2010), Debbie (2017) and Iris (2018). Tropical Cyclone Marcia, a category 5 system, came closest to the reefs reported here, tracking southwards past Middle Percy Island with winds in excess of 80 knots before crossing the coast at Shoalwater Bay on February 20th 2015 (Figure 6). Waves from TC Marcia (maximum height 7m) were the fourth highest waves recorded at the Emu Park buoy (Queensland Government 2024b). Of note is that the orientation of the monitoring sites at Henderson and Temple islands, along with protection offered by surrounding islands, will have afforded some protection from damaging seas produced by TC Marcia.

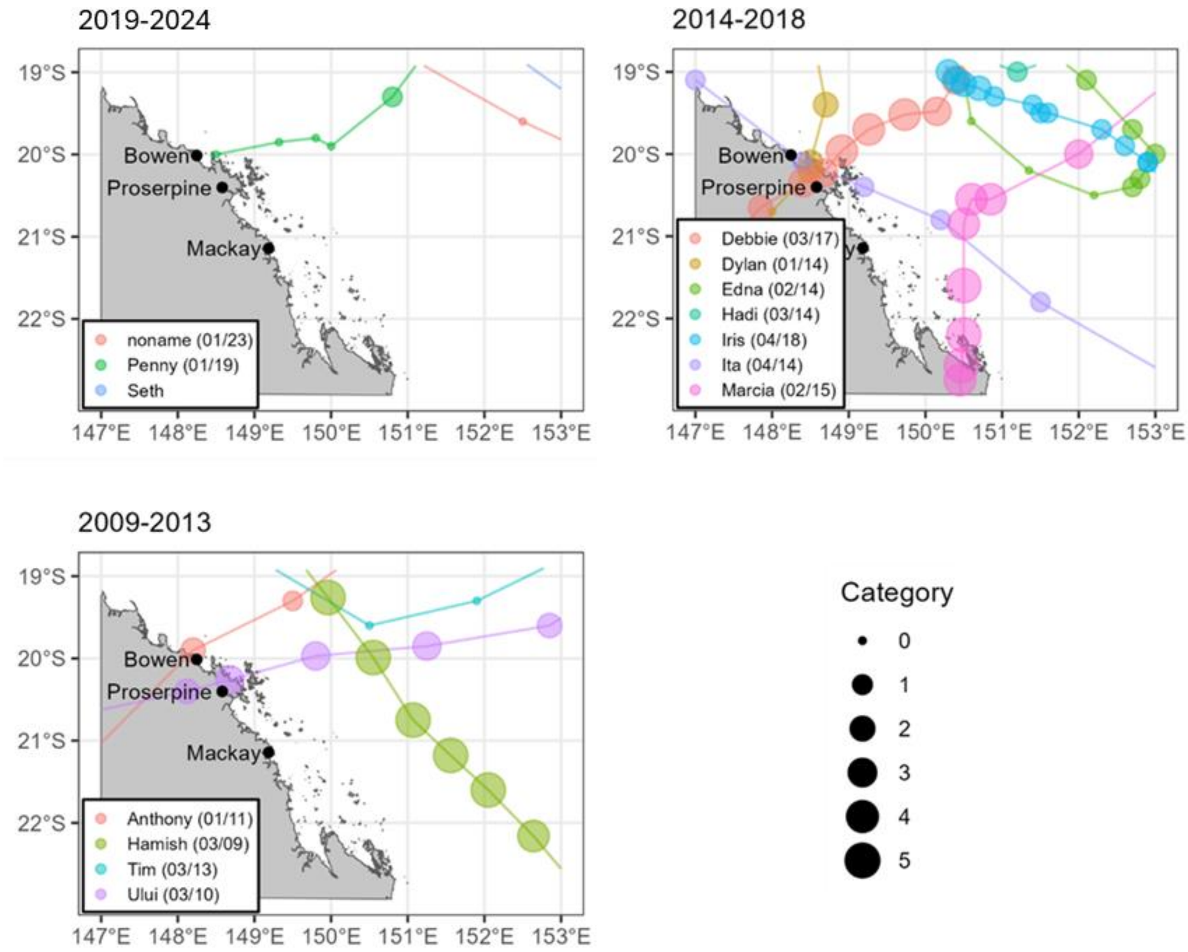

Figure 6 Tracks of tropical cyclones passing through the region. All cyclones crossing through the Mackay Whitsunday Isaac regional report card reporting area over the last 15 years are displayed. Tracks sourced from the Bureau of Meteorology

4.1.4 Biological Damage

During surveys in 2024, four to five months after temperatures peaks, 3% of corals were either bleached or partially bleached, although the prevalence of bleaching was variable among reefs (Table A 1, Table A 2). Although temperatures during the 2024 marine heat wave were comparable to those observed in 2020 (Figure 4) a far higher proportion of corals (41%) were still bleached during surveys in 2020 (Davidson *et al.* 2020).

A total of 20 colonies were identified with disease across all five reefs, a doubling in number from 2023. Diseased colonies ranged from branching *Acropora* to massive *Porites* (Table A 3). In addition, there were a total of 23 colonies for which recent mortality was unknown. In combination, these 43 colonies spanned seven genera, a significant increase from last year's observations (25 colonies and five genera, Davidson *et al.* 2023) but more comparable with observations from 2021 (31 colonies, 10 genera) and 2019 (55 colonies, 12 genera, Figure A 1). Given the relatively high level of coral cover, it's no surprise that the reef community at Henderson Island had the greatest number of affected colonies (14 *Acropora* corals). However, Pine Peak Island, with relatively low coral cover, has a similarly high number of affected colonies (12), with brown band disease, and white syndrome disease observed at both depths.

The number of colonies being overrun by the encrusting sponge *Cliona orientalis* has increased from four colonies and two genera in 2023 to nine colonies and six genera in 2024 (Table A 3). Observations of *C. orientalis* in 2024 were confined to the shallow waters at Aquila and Temple islands. Afflicted genera were *Cyphastrea, Dipsastraea, Goniastrea, Goniopora, Montipora,* and *Turbinaria.*

No crown-of-thorns seastars have been observed in this study. Another coral predator, the gastropod *Drupella,* had made a notable appearance at Henderson Island (5 m) in 2022 where 20 individuals were observed feeding on branching *Acropora* colonies. In 2024 there were no *Drupella* observed at Henderson Island or at the other locations in the study.

With no significant storms over the Southern Inshore Zone during the 2023-2024 season, there were no observations of recent physical damage at any of the visited reefs (Table A 3).

4.2 Coral Community Condition Assessment

The overall Coral Index score for the Southern Inshore Zone in 2024 was graded as D, categorising the coral communities as being in ‘poor’ condition (Table 5). While the Report Card category remains unchanged from 2023, the Report Card score has improved due primarily to an increase in the Coral cover indicator that crossed from a category ‘D’ (poor) in 2023 to category ‘C’ (satisfactory) in 2024 (Table 5). Growth in coral cover also improved the Cover change indicator, contributing further to the Report Card score.

Across the reporting zone there has been a 10% increase in mean cover of hard and soft corals to 33% (Table 6). There was a slight improvement in the density of juveniles (Table 6) though the score for Juvenile density remains ‘very poor’ (Table 5). The proportion of macroalgae across the region remains high at 69% (Table 6) and for all reefs the score for this indicator remains zero (Table 5).

Table 5 Coral Index and indicator scores for 2024. The Cover change indicator was added in 2022 and back calculated for 2021. Scores are coloured as per Table 3.

	Year	Juvenile density	Coral cover	Macroalgae	Cover change	Report Card	
						Score	Grade
Zone Scores	2019	0.12	0.47	0	NA	0.20*	E*
	2020	0.14	0.44	0	NA	0.19*	E*
	2021	0.10	0.37	0	0.29	0.19	E
	2022	0.13	0.40	0	0.46	0.25	D
	2023	0.12	0.40	0	0.36	0.22	D
	2024	0.13	0.44	0	0.40	0.24	D

* Report card scores not directly comparable to latter years as do not include the Cover change indicator

Table 6 Indicator values for Southern Inshore Zone. Juvenile densities are corrected for area of algal covered substrate, as a potential area for colonisation.

Year	Juvenile density (per m²)		Coral cover (%)				Macroalgae proportion (%)	
			Total (Hard + Soft)		Hard only			
	Mean	SD	Mean	SD	Mean	SD	Mean	SD
2019	1.42	0.96	35.6	26.5	22	20.3	65.8	23.6
2020	1.59	0.84	33	21.3	22.5	19	60.5	25.6
2021	1.11	0.86	27.9	15.6	16.7	13.1	65.8	17.6
2022	1.46	1.13	30.4	17.1	19.3	14	58.9	20.4
2023	1.39	1.08	30.1	21.9	19.4	16.9	69.7	19.4
2024	1.51	1.44	33.3	24.3	21.6	19.6	69.3	18.8

The overall Index score continues to mask the substantial differences in the condition of coral communities among reefs (Table 7). Scores among most reefs in 2024 have improved or remained at 2023 levels. Higher Index scores at Henderson Island contrast with those at Pine Islets where, at 2 m depth, the index score continues a decline. There have been no changes to the Index grade at any of the sites in 2024.

Table 7 Index grade and scores for each reef and depth combination. Comparison of Index figures from 2019 to 2024. * indicate scores prior to the inclusion of the Cover change indicator and are not directly comparable to later years. Scores are coloured as per Table 3.

Reef	Depth	Index 2019	Index 2020	Index 2021	Index 2022	Index 2023	Index 2024	Grade
Pine Peak Island	2	0.05*	0.09*	0.08*	0.14	0.09	0.12	E
	5	0.12*	0.14*	0.12*	0.36	0.23	0.23	D
Pine Islets	2	0.04*	0.06*	0.06*	0.24	0.16	0.12	E
	5	0.12*	0.20*	0.15*	0.26	0.17	0.19	E
Henderson Island	2	0.41*	0.34*	0.19*	0.27	0.32	0.40	D
	5	0.36*	0.33*	0.28*	0.36	0.45	0.50	C
Temple Island	1	0.32*	0.21*	0.23	0.18	0.22	0.27	D
Aquila Island	1	0.19*	0.16*	0.14	0.14	0.12	0.12	E

4.3 Coral Cover

Coral Cover scores are based on the combined cover of hard and soft corals. Coral cover and the related scores improved at seven of the eight reef-depth locations in 2024 (Table 8, Figure 7a). These improvements were led by increases in hard coral cover, supported by increases in soft coral cover at five of the eight reef-depth locations (Table 8).

Across the region mean hard coral cover was 21.6%, up from 19.4% observed in 2023 (Table 8). Henderson Island was clearly contributing most to this regional increase with a rise from 37.1% to 44% at 2 m depth and 52.9% to 59% at 5 m depth (Table 8). These gains were principally due to increased cover of *Acropora,* the most common genus at this reef (Table A 4, Figure A 2). At Pine Islets (2 m) minor losses (5.8% to 4.9%), mainly in *Porites* and *Montipora* cover, resulted in the only decline in the Coral cover score. Among the soft corals there were modest losses and gains across all reef-depth locations (Table 8). In combination with hard coral, these losses had little influence on the Coral cover scores, however, gains in *Briareum* and *Rhystima* at Pine Peak (5 m), and in *Sclerophytum* at Temple (Table A 5), contributed to transitioning the Coral cover grade from 'poor' to 'satisfactory' (Table 8).

While changes in hard and soft coral between 2023 and 2024 have been generally minor, undoubtedly the main impact to the region has been the 2024 marine heatwave and resulting coral bleaching. Across the region photo-transect analysis revealed 3% of hard corals were still bleached at the time of survey well after peak summer heat, (Table A 1). However, only two reefs were moderately impacted, Pine Peak Island (2 m) and Aquila Island where 13% and 12% of corals were bleached (Table A 1). Acroporidae was the most frequently affected hard coral family, however despite two genera, *Acropora* and *Montipora,* being abundant and ubiquitous, the proportion of bleached corals at any given reef was generally low. Notable levels of bleaching (12%) were recorded within Acroporidae at Aquila Island (Table A 2) while 23% of Poritidae were bleached at Pine Peak Island (2 m).

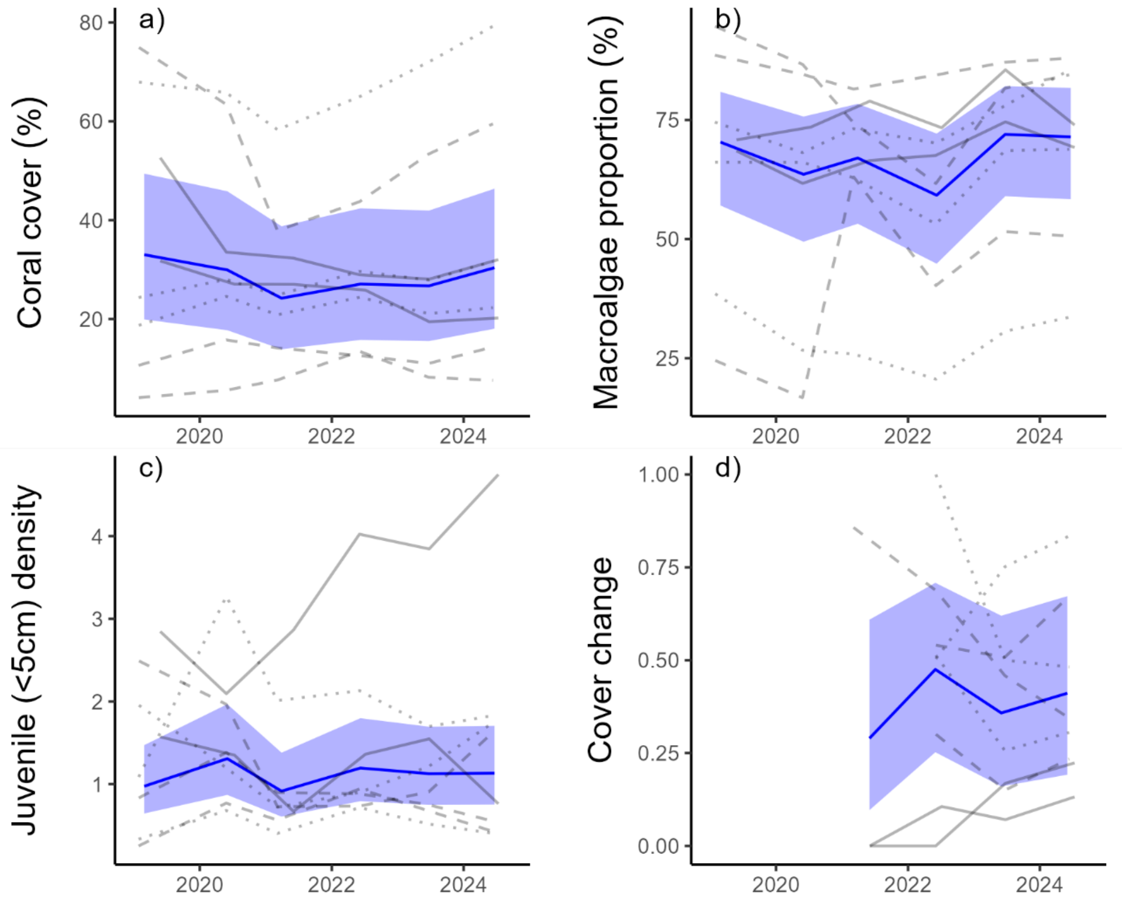

Figure 7 Indicator trends for Southern Inshore Zone. Blue lines represent trends in: a) coral cover, b) macroalgae proportion of total algae cover, c) juvenile density per m^2 of available substrate, d) cover change. Trends are bound by 95% confidence intervals of those trends (blue shading), grey lines represent observed profiles at 5 m (dotted lines), 2 m (dashed lines), and 1 m (solid lines) for individual reefs.

Table 8 Coral cover and indicator scores for each location. Comparing percent cover and scores for 2024 and 2023. Scores are coloured as per Table 3.

Reef	Depth (m)	Year	Hard coral cover (%)	Soft coral cover (%)	Coral cover (%)	Coral cover Score
Pine Peak Island	2	2023	4.1	6.9	11.0	0.15
		2024	4.8	9.6	14.3	0.19
	5	2023	10.3	17.6	27.9	0.37
		2024	10.9	20.8	31.7	0.42
Pine Islets	2	2023	5.8	2.4	8.2	0.11
		2024	4.9	2.6	7.6	0.10
	5	2023	15.2	5.9	21.1	0.28
		2024	16.5	5.8	22.3	0.30
Henderson Island	2	2023	37.1	16.3	53.4	0.71
		2024	44.2	15.3	59.5	0.79
	5	2023	52.9	19.1	72.0	0.96
		2024	59.0	20.3	79.3	1.00
Temple Island	1	2023	17.1	10.9	28.1	0.37
		2024	18.6	13.4	32.0	0.43
Aquila Island	1	2023	13.0	6.4	19.4	0.26
		2024	14.0	6.2	20.2	0.27

4.4 Macroalgae Proportion

In 2024 the proportion of algae classified as macroalgae continues to exceed thresholds deemed to result in negative impacts to coral community resilience (Table 5, Table 6). The grade of 'E' ('very poor', Table 9) continues across all reef-depth locations for this indicator. The mean proportional macroalgae cover for the region in 2024 was 69% and, while brown macroalgae cover remains dominant at most reefs within the region, the previous rise in macroalgae proportion from 2022-2023 has stalled across the region (Figure 7b). Indeed, at many reef-depth locations there has seen a slight decline in 2024 (Figure 7b).

The largest contributor to macroalgae cover remains the family Sargassacae, clearly dominant at Pine Peak Island (2 m), Pine Islets, and Aquila Island (Table A 6). However, several shifts in the brown macroalgae community were noted (using Table A 3 Davidson *et al.* 2023 for reference):

- At Pine Peak Island (2 m) cover of *Lobophora* increased from 10.4% to 22.3% and *Stypopodium* from 1.4% to 8.8% contrasting a reduction in cover of the Sargassacae from 55.2% to 34.9%. Similar although less pronounced changes occurred at 5 m depth.

- At Temple Island Sargassacae declined from 36.7% to 21% contrasting a slight increase in *Lobophora* from 3.1% to 5%.
- At Henderson Island *Lobophora* is the only taxa with cover greater than 1%, though this declined from slightly from 22.9% to 19.2% at 2 m. Cover also declined slightly at 5 m but was already relatively low at 6.2%. Further, Henderson Island is the only location in this study where corals have greater cover than macroalgae.

Red macroalgae retains a modest but important presence in the region. Highest covers of this group were recorded at Aquila Island and Temple Island, where red algae cover increased by 57% relative to that observed in 2023 (Table A 3 Davidson *et al.* 2023). The cover of turf algae has doubled at Aquila Island, perhaps partly in response to bleaching mortality.

Table 9 Macroalgae cover and indicator scores for each location. Comparison of 2024 and 2023 data. Scores are coloured as per Table 3.

Reef	Depth	Year	Macroalgae cover (%)	Macroalgae proportion (%)	Macroalgae score
Pine Peak Island	2 m	2023	75.7	87.1	0
		2024	72.7	88.0	0
	5 m	2023	53.4	78.0	0
		2024	55.6	85.5	0
Pine Islets	2 m	2023	70.4	81.7	0
		2024	73.7	84.5	0
	5 m	2023	49.5	68.6	0
		2024	47.6	68.8	0
Henderson Island	2 m	2023	23.4	51.6	0
		2024	20.1	50.6	0
	5 m	2023	6.6	30.6	0
		2024	5.6	33.6	0
Temple Island	1 m	2023	49.8	74.6	0
		2024	41.9	69.2	0
Aquila Island	1 m	2023	51.7	85.5	0
		2024	46.8	73.9	0

4.5 Juvenile Density

At the regional level, the overall density of juvenile hard corals, corrected for area of transects occupied by algae, has had a minor increase since 2023 but the category for the Juvenile score has remained ‘very poor’ over the six year period of this study (Table 5, Table 6). At most reef-depth locations the category for the Juvenile score remained ‘very poor’ with changes over the last year being minor and variable (Table 10). In 2024 the abundance of juvenile hard corals declined in the shallow 2 m depths at Pine Peak Island and Pine Islets, and at Aquila Island (Table 10, Figure A 2). There was a marginal increase at Pine Islets (5 m) and at Henderson Island. However, a more substantial increase in juvenile density was recorded at Temple Island where the Juvenile indicator score improved to ‘satisfactory’ (Table 10, Figure 7c).

Only Temple Island has maintained an increasing abundance of juvenile hard corals since 2022 (Figure 7c), with major contributions from *Pocillopora* (family Pocilloporidae), *Turbinaria* (family Dendrophylliidae), *Acropora*, and *Porites* (Figure A 2). In 2024, a large decline in *Pocillopora* at Temple Island was offset by a rise in *Dipsastraea* (family Merulinidae) and *Porites* (Table A 7, Figure A 2). Pine Islets (5 m) continues to have the highest genus richness among the sites but, typical of the juvenile community, the genus richness is not as well represented in the adult community (Table A 7, Figure A 2).

Table 10 Juvenile hard coral abundance, density and indicator scores for each location. Comparison of 2024 and 2023 data. Density has been adjusted for the area of algal covered substrates. Scores are coloured as per Table 3.

Reef	Depth	Year	Juvenile abundance	Juvenile density (per m^2)	Juvenile score
Pine Peak Island	2 m	2023	40	0.68	0.06
		2024	24	0.43	0.04
	5 m	2023	24	0.52	0.04
		2024	21	0.41	0.04
Pine Islets	2 m	2023	45	0.75	0.07
		2024	33	0.55	0.05
	5 m	2023	83	1.70	0.15
		2024	85	1.83	0.16
Henderson Island	2 m	2023	28	0.91	0.08
		2024	40	1.63	0.14
	5 m	2023	18	1.21	0.11
		2024	20	1.74	0.15
Temple Island	1 m	2023	171	3.84	0.33
		2024	189	4.75	0.41
Aquila Island	1 m	2023	63	1.55	0.13
		2024	33	0.76	0.07

4.6 Cover change indicator

The Cover change indicator score across the region in 2024 was 0.40 (Table 5), a minor increase from 2023 though still within the 'poor' category as increases in hard coral cover in recent years have been below modelled expectation. Combined with increased indicator scores for Coral cover and Juvenile density, the improvement in the Coral change indicator supported a small improvement in the annual Report Card score (Table 5). A plot of the Cover change indicator scores shows an increase at most reef-depth locations in 2024, though the large error bars indicate large variation among sites (Figure 7d, Table 11).

At Henderson Island the Cover change score increased to 'good' at 2 m and 'very good' at 5 m (Table 11), driven by continued recovery in the cover of *Acropora* corals (Figure A 2). Minor increases in hard coral cover at Pine Peak Island (2 m) and Temple Island lifted the Cover change score from 'very poor' to 'poor' (Table 11). Both Pine Islets (5 m) and Temple Island had minor gains but retained their respective grades. Notably, the downward trajectory of the Coral change score has continued for Pine Peak Island (5 m) declining from 'very good' to 'satisfactory'. Similarly, Pine Islets (2 m) has declined from 'very good' to 'poor' (Table 11, Figure 7d) as annual coral growth rates continue to be below modelled expectations.

Table 11 Reef level Cover change scores. Only years for which Cover change was estimated are included. Annual scores for each reef are a running mean over up to four years as indicated by the Over period.

Reef	Depth	Year	Period	Change in percent cover of hard coral cover		Cover change score
				Over period	From previous year	
Pine Peak Island	2 m	2022	2022		0.7	0.3
		2023	2022-2023		-0.3	0.15
		2024	2022-2024	1.1	0.7	0.23
	5 m	2022	2022		4.8	1
		2023	2022-2023		-2	0.5
		2024	2022-2024	3.4	0.6	0.48
Pine Islets	2 m	2021	2021	1.9	1.9	0.86
		2022	2021-2022	5.2	3.3	0.69
		2023	2021-2023	1.6	-3.7	0.46
		2024	2021-2024	0.6	-0.9	0.34
	5 m	2022	2022	2.6	2.6	0.51
		2023	2022-2023	1.4	-1.2	0.26
		2024	2022-2024	2.7	1.3	0.31
Henderson Island	2 m	2022	2022	6.9	6.9	0.54
		2023	2022-2023	12.1	5.2	0.51
		2024	2022-2024	19.3	7.1	0.67
	5 m	2022	2022	3.4	3.4	0.5
		2023	2022-2023	8.6	5.2	0.75
		2024	2022-2024	14.7	6.1	0.83
Temple Island	1 m	2021	2021	-3.1	-3.1	0
		2022	2021-2022	-4.9	-1.8	0
		2023	2021-2023	-2.7	2.2	0.17
		2024	2021-2024	-1.2	1.5	0.22
Aquila	1 m	2021	2021	-2.1	-2.1	0
		2022	2021-2022	-1.5	0.6	0.11
		2023	2021-2023	-5.5	-4	0.07
		2024	2021-2024	-4.5	1	0.13

5 DISCUSSION

The overall condition of Southern Inshore Zone reefs in 2024 was categorised as ‘poor’ and graded ‘D’. The Coral Index score improved slightly to 0.24 compared to the 0.22 reported for 2023. Contributing to the improvement were increased scores for Coral cover, Cover change and, to a lesser degree, Juvenile coral indicators. The relatively low Coral Index score remains influenced by the high prevalence of macroalgae across the zone. At the reef level the coral community has changed little since 2023. Differences in coral cover, macroalgae, and juvenile density metrics among reefs and depths were relatively minor and variable.

A marine heat wave in early 2024 caused widespread bleaching throughout the GBR (Cantin *et al.* 2024). Thermal anomalies on the GBR in 2024 were the highest recorded to date (Cantin *et al.* 2024, Henley *et al.* 2024). While temperatures in 2024 have exceeded long-term averages at large spatial scales, the accumulation of elevated temperatures on the GBR had a strong latitudinal component, being greatest for reefs of the Central and Southern GBR, including the Southern Inshore Zone (Figure 4). Temperature anomalies for this region were greatest on February 5-6th, with the maximum temperature for the year recorded at Aquila Island (30.70^{O}C), the highest temperature recorded in this program. However, the influence of thermal stress on the coral communities in 2024 was highly variable among reefs and depths, and much less severe than observed following the 2020 event.

Following the severe bleaching event of 2020, surveys in late May revealed a minor reduction in coral cover but also that a high proportion (41%) of the remaining hard coral cover was still bleached (Davidson *et al.* 2020). This ongoing stress was attributed to the further loss of hard coral cover observed during surveys in 2021. In contrast, surveys in 2024 document a slight increase in hard coral cover with minimal ongoing bleaching, with just 3% of hard coral cover across the zone being fully or partially bleached during surveys in June and July. The adult coral community in 2024 are survivors of the severe bleaching event in 2020 and the loss of susceptible corals in 2020/21 may partly explain the reduced impact this year. In combination, the observations of increased hard coral cover and limited ongoing bleaching suggest there is unlikely to be a similar knock-on impact as was observed in 2021 following the 2020 event. However, following marine heat waves, increased coral mortality has been linked to coral disease as pathogens overcome corals weakened by thermal stress, exacerbating the impact of post-bleaching mortality (Bruno *et al.* 2007, Brodnicke *et al.* 2019, Howells *et al.* 2020). Within the region there was a notable 44% spike in observed coral disease from 2023 levels (mostly brown band and white syndrome), primarily among genus groups made vulnerable by thermal stress; *Acropora* at Henderson Island and *Pocillopora* at Pine Peak Island (Table A 3). However, with an average of only 3% ongoing bleaching observed in 2024 it is unlikely that further post-bleaching mortality will adversely influence coral cover growth figures in 2025.

Despite the exposure to high levels of heat stress, the average coral cover in 2024 had almost recovered to the 35.6% observed at the start of the program in 2019. Indeed, the regional hard coral cover for 2024 was almost level with that of the pre-bleaching year, 2019. The high and increasing cover of *Acropora* at Henderson Island disproportionately influences the improved Coral cover and Cover change scores. Mean hard coral cover across both depths is 52%. By comparison, the mean hard coral cover among the other reefs is 12%, comprised mainly of a range of resilient but slower-growing taxa such as *Monitpora, Porites, Turbinaria*, and *Lobophyllia*. Soft coral cover also declined as a result of the 2020 bleaching. In 2024 recovery of soft coral includes modest gains in *Sclerophytum* at Temple, Aquila, and Henderson islands, and *Briarium* at Pine Peak Island, which all contribute to the improved Coral cover indicator score. Recovery of coral cover to pre-disturbance levels is of prime interest to management agencies, but, with the frequency and spatial extent of disturbance events increasing, there is a concern that the time and growth-rate available between disturbances will limit 'full recovery' to those reefs with 25% hard coral cover or less (Emslie *et al.* 2024).

Thermal stress has been linked to a reduction in the reproductive health of corals that may have a long-lasting effect on the resilience of communities dependent on successful spawning, settlement, and recruitment (Ward *et al.* 2002, Hughes *et al.* 2019). This is of particular concern in context of the Southern Inshore Zone reefs where low Juvenile scores indicate coral recruitment is already limiting coral community resilience. Across the region in 2024 there was a 6% reduction in the abundance of juvenile corals compared to 2023, but relative changes were highly variable among reefs. Small increases at Henderson Island (2 m) and Temple Island were tempered by declines at other reefs, particularly Aquila Island, where thermal stress may have been responsible for some of the loss. Interestingly, there continued to be large number of *Acropora* juveniles recorded at Temple Island. This cohort would have recruited following the bleaching event of 2020 and suggests either intra or inter-reefal supply from an active and fecund source. Temple Island also received increased numbers of *Dipsastraea, Porites, Cyphastrea,* and *Favites* juveniles, and consistently supports large numbers of juvenile *Turbinaria*. Notably, Henderson Island continues to record very few juvenile *Acropora* corals compared to Temple Island despite *Acropora* cover being 10-fold greater, demonstrating the variable connectivity to broodstock among these reefs, or suggesting the influence of density dependence limits the space for recruitment as the adult population grows (Edmunds *et al.* 2018).

Of ongoing concern for the resilience of coral communities in this region are the high levels of macroalgae. Dense canopies of macroalgae, that compete with corals, dominate the benthic communities at most of the reefs in this study. There are a number of pathways by which macroalgae competition occurs; from limiting the space or light available to corals (Tanner 1995, Hauri *et al.* 2010), physically damaging corals via abrasion (Clements *et al.* 2018, Clements *et al.* 2020), chemically interfering with coral recruitment process (Foster *et al.* 2008, Monteil *et al.* 2020), or promoting bacterial communities pathogenic to corals (Smith *et al.* 2006, but see Clements and Hay 2023). At the regional level the proportion of algal cover comprised of macroalgae species has remained generally consistent since 2019, with a mean of 65% and a range of 59% to 70% (Figure 7b). This masks the varied, and at times considerable, changes in macroalgae cover at the reef level. For example, on the shallow reef flat of Henderson Island the cover of macroalgae increased seven-fold from 5.8% in 2019 to 38.4% in 2021, driven primarily by a rapid increase in *Lobophora* following the 2020 bleaching event. The cover of *Lobophora* has since moderated with macroalgae cover down to 20.1% in 2024. Only Henderson Island has a benthic community with more coral than macroalgae cover. In contrast, brown macroalgae of the family Sargassaceae continue to dominate the reefs at Pine Peak Island and Pine Islets with a combined average of 37% cover in 2024 (compared to the average hard coral cover of 9%). The high density of macroalgae puts downward pressure on both the growth and replenishment of coral communities contributing to a state of persistent low coral cover.

The region is unique in geophysical terms, with an extensive continental shelf isolating the region from the more offshore reef matrix of the Great Barrier Reef. The large tidal range, causing strong tidal currents, and proximity to the shallow, silt-laden Broad Sound, result in environmental conditions that challenge the resilience and match the primary variables described by Fabricius *et al.* 2023 as supporting sustained abundance of macroalgae cover. Indeed, examining inshore reef structures and coral reef communities between the Whitsundays and Keppel Island groups, Kleypas (1996), van Woesik (1992), and van Woesik & Done (1997) interpreted reduced reef development, abundance, and diversity of hard corals as reflecting environmental conditions that are less favourable for coral reef development. These physical environmental conditions need to be kept in mind when considering the condition of these communities as summarised by the Coral Index, a scoring tool developed for reefs where historical reef development has occurred.

Results from the 2024 survey suggest that the marine heat wave had much less impact among reefs in the region than that of 2020, and that coral communities are recovering, albeit slowly. We are reminded that slow growth and replenishment, which is likely typical of coral communities in this area, makes these communities particularly vulnerable to increased frequency of disturbance such as marine heatwaves, cyclones, and flood plumes. Across the region communities with low coral cover continue to be shaped by abundant macroalgae. Henderson Island, clear of abundant macroalgae, is the only reef demonstrating clear recovery from the 2020 bleaching event. The conclusion from the 2024 survey is that the reef communities appeared resistant to the extreme heat wave conditions this year but that the conditions which sustain a macroalgae-dominant environment continue to challenge the resilience of coral communities more generally.

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8 APPENDICES

8.1 Reef Level data summaries

Table A 1 Overall bleaching at each reef and depth in 2024

Numbers are accumulated data points from photo transects of live hard coral in three bleached states; totally bleached (B), partially bleached (PB), not bleached (NB). Overall proportion of live hard coral exhibiting bleaching is given.

Reef	Depth	B	PB	NB	% bleached
Pine Peak Island	2	4	6	66	13.16
Pine Peak Island	5	3	12	159	8.62
Pine Islets	2	0	0	79	0.00
Pine Islets	5	4	3	256	2.66
Henderson Island	2	1	5	699	0.85
Henderson Island	5	2	6	935	0.85
Temple Island	1	0	1	296	0.34
Aquila Island	1	13	15	196	12.50

Table A 2 Proportion of ongoing bleaching among hard coral families in 2024

Values represent the number of points identified as hard coral and categorised as being bleached white (B), partially bleached (PB) or note bleached (NB). Data are aggregated by taxonomic family. Proportion of bleaching within family, and overall proportion of hard coral bleaching are given for each reef. Combined depths.

<table>
<tr><th>Reef</th><th>Family</th><th>B</th><th>PB</th><th>NB</th><th>Proportion bleached (%)</th><th>Total hard coral (%)</th><th>Overall proportion bleached (%)</th></tr>
<tr><td rowspan="10">Pine Peak</td><td>Acroporidae</td><td>0</td><td>0</td><td>28</td><td>0</td><td rowspan="10">7.81</td><td rowspan="10">10.00</td></tr>
<tr><td>Agariciidae</td><td>0</td><td>0</td><td>1</td><td>0</td></tr>
<tr><td>Dendrophylliidae</td><td>0</td><td>0</td><td>13</td><td>0</td></tr>
<tr><td>Euphyllidae</td><td>0</td><td>0</td><td>3</td><td>0</td></tr>
<tr><td>Lobophylliidae</td><td>0</td><td>0</td><td>6</td><td>0</td></tr>
<tr><td>Merulinidae</td><td>1</td><td>1</td><td>30</td><td>6</td></tr>
<tr><td>Pachyseridae</td><td>0</td><td>1</td><td>5</td><td>17</td></tr>
<tr><td>Pocilloporidae</td><td>0</td><td>0</td><td>15</td><td>0</td></tr>
<tr><td>Poritidae</td><td>6</td><td>16</td><td>119</td><td>16</td></tr>
<tr><td>Psammocoridae</td><td>0</td><td>0</td><td>5</td><td>0</td></tr>
<tr><td rowspan="11">Pine Islets</td><td>Acroporidae</td><td>4</td><td>3</td><td>141</td><td>5</td><td rowspan="11">10.7</td><td rowspan="11">2.06</td></tr>
<tr><td>Dendrophylliidae</td><td>0</td><td>0</td><td>37</td><td>0</td></tr>
<tr><td>Euphylliidae</td><td>0</td><td>0</td><td>2</td><td>0</td></tr>
<tr><td>Fungiidae</td><td>0</td><td>0</td><td>7</td><td>0</td></tr>
<tr><td>Leptastreidae</td><td>0</td><td>0</td><td>1</td><td>0</td></tr>
<tr><td>Lobophylliidae</td><td>0</td><td>0</td><td>19</td><td>0</td></tr>
<tr><td>Merulinidae</td><td>0</td><td>0</td><td>58</td><td>0</td></tr>
<tr><td>Pachyseridae</td><td>0</td><td>0</td><td>10</td><td>0</td></tr>
<tr><td>Plesiastreidae</td><td>0</td><td>0</td><td>1</td><td>0</td></tr>
<tr><td>Pocilloporidae</td><td>0</td><td>0</td><td>7</td><td>0</td></tr>
<tr><td>Poritidae</td><td>0</td><td>0</td><td>43</td><td>0</td></tr>
</table>

Reef	Family	B	PB	NB	Proportion bleached (%)	Total hard coral (%)	Overall proportion bleached (%)
	Psammocoridae	0	0	7	0		
	Unknown	0	0	2	0		
Henderson	Acroporidae	3	10	1390	1	51.61	0.91
	Dendrophylliidae	0	0	10	0		
	Euphylliidae	0	0	33	0		
	Fungiidae	0	0	13	0		
	Lobophylliidae	0	1	96	1		
	Merulinidae	0	0	21	0		
	Pocilloporidae	0	0	14	0		
	Poritidae	0	0	57	0		
Temple	Acroporidae	0	1	204	0.5	18.55	0.34
	Dendrophylliidae	0	0	34	0		
	Merulinidae	0	0	42	0		
	Pocilloporidae	0	0	4	0		
	Poritidae	0	0	10	0		
	Psammocoridae	0	0	2	0		
Aquila	Acroporidae	11	11	162	12	14.00	12.50
	Dendrophylliidae	0	1	3	25		
	Merulinidae	0	0	7	0		
	Plesiastreidae	0	0	1	0		
	Pocilloporidae	0	3	0	100		
	Poritidae	1	0	9	10		
	Psammocoridae	1	0	13	7		
	Unknown	0	0	2	0		

Table A 3 Coral health survey results.
The number of colonies along the ten 20 m long and 2 m wide transects searched at each reef and depth combination in 2024 having recently lost tissue (patches of bare white skeleton) attributed to a range of causes. Anchor or physical damage to corals is recorded as a proportion of coral cover at the site effected: 0 = absent, 0+ = individual colonies, 1- = 1-5%, 1+ = 6-10%, 2- = 11-20%, 2+ = 21-30%, 3- = 31-40%, 3+ = 41-50%, 4- = 51-62%, 4+ = 63-75%, 5- = 76-82%, 5+ = 83-100%. Bleached hard corals are reported as a percentage of the total hard corals from phototransect analysis.

Cause	Genus	Pine Peak		Pine Islets		Henderson		Temple	Aquila
		2 m	5 m	2 m	5 m	2 m	5 m	1 m	1 m
Disease	*Acropora*	1	1			1	2		1
	Lobophyllia		1						
	Merulina				1				
	Montipora	1			3			1	4
	Pocillopora		2						
	Porites			1					
Unknown cause	*Acropora*					6	6	1	
	Montipora							1	1
	Pocillopora	1	4		1				
	Porites	1							
	Turbinaria				1				
Sponge - *Cliona orientalis*	*Cyphastrea*								1
	Dipsastraea								1
	Goniastrea								1
	Goniopora								1
	Montipora							2	1
	Turbinaria							2	
Total number of Colonies		4	8	1	6	7	8	9	11
Bleaching (% of hard corals)		13.16	8.62	0	2.66	0.85	0.85	0.34	12.5
Physical (proportion of colonies)		0	0	0	0	0	0	0	0

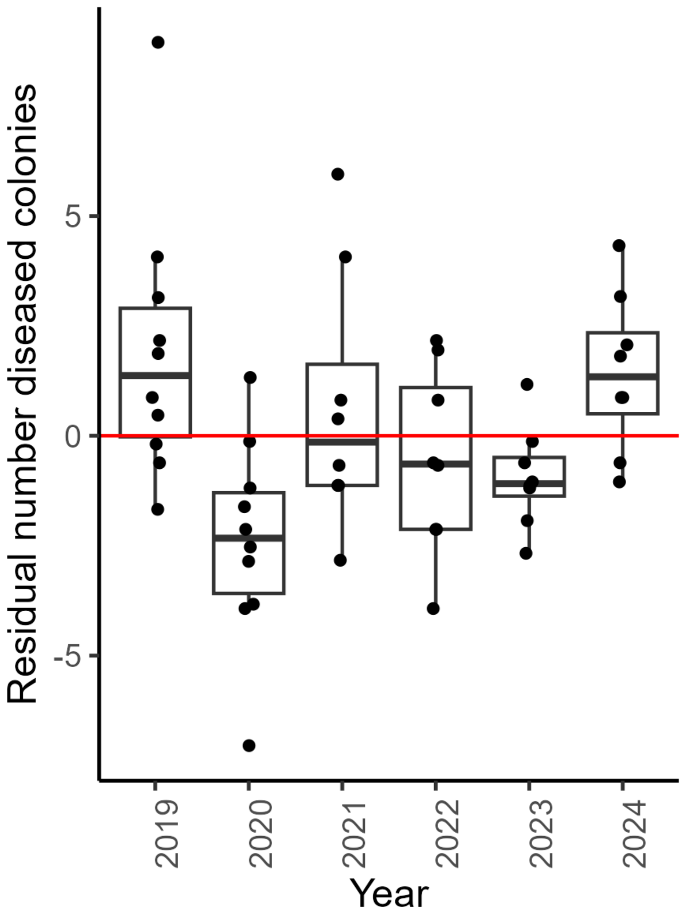

Figure A 1 Relative coral disease by year. Data are standardised to the reef and depth mean across years. Boxplots show the median (bold horizontal line), 25th to 75th quartiles (box), and 1.5 times the inter-quartile range (whiskers). Solid dots are the relative number of coral colonies suffering ongoing mortality attributed to disease for each reef, depth and year

Table A 4 Cover of hard coral genera. Genus with a minimum cover of 1% at any reef are included. All less abundant genera are grouped as Other HC. Total number of genus observed is presented as Genus Richness. Data from phototransect analysis.

Reef	Depth	*Acropora*	*Alveopora*	*Galaxea*	*Goniopora*	*Lobophyllia*	*Merulina*	*Montipora*	*Platygyra*	*Porites*	*Turbinaria*	Other HC	Genus Richness
Pine Peak Island	2	0.06	0	0.06	0.12	0	0.06	0.62	0.19	2.88	0	0.75	13
	5	0.31	0.06	0	0.19	0.19	0.06	0.75	0.12	5.57	0.81	2.81	21
Pine Islets	2	0.5	0	0	0.06	0	0	1.94	0.12	0.31	1.31	0.69	15
	5	0.19	0	0.13	0.94	0.44	1.69	6.64	0.06	1.38	1	4	25
Henderson Island	2	42.29	0.19	0	0.38	0.38	0	0.69	0	0.06	0	0.25	8
	5	43.19	1.62	2.06	1.06	5.44	0.06	1.75	0.31	0.25	0.62	2.62	19
Temple Island	1	4.81	0.12	0	0.19	0	0	8	1.62	0.31	2.06	1.44	15
Aquila Island	1	0.12	0	0	0.25	0	0	11.38	0.06	0.38	0.25	1.56	12

Table A 5 Cover of soft coral genera. Genus with a cover of at least 1% at any reef are included. All less abundant genera are grouped as Other SC. Data from phototransect analysis.

Reef	Depth	*Briareum*	*Cladiella*	*Klyxum*	*Lobophyton*	*Rhytisma*	*Sarcophyton*	*Sclerophytum*	Other SC
Pine Peak Island	2	4.88	0.19	0	0.31	0.38	0.5	3.12	0.19
	5	14.32	0.06	0.25	0.25	1.5	1.32	3.07	0
Pine Islets	2	0.25	0.06	0	1.75	0	0	0.5	0.06
	5	2.5	0	0.69	0.19	0	0.88	1.56	0
Henderson Island	2	0.88	1.13	5.64	0.06	0	2.57	5	0
	5	0.25	0.06	13.69	0	0	3.12	3.12	0
Temple Island	1	3	0.25	0	0.5	0	0.81	8.81	0.06
Aquila Island	1	0	0.19	0.12	1.12	0	0.31	4.44	0

Table A 6 Cover of algae. Identified macroalgae genera with a cover of at least 1% at any reef are separated. All less abundant or un-resolved brown macroalgae are grouped as 'Other'algae are grouped. Data from phototransect analysis.

Reef	Depth	Brown macroalgae				Red macroalgae	Green macroalgae	Turf algae	Coralline algae
		Lobophora	Family Sargassaceae	Stypopodium	Other				
Pine Peak Island	2	22.33	34.9	8.82	0.88	5.38	0.38	8.07	1.88
	5	36.43	13.89	2.56	0.19	2.38	0.12	7.32	2.13
Pine Islets	2	4.31	65	0.56	0.69	3.06	0.06	12.12	1.44
	5	10.58	32.23	1.13	0.69	2.82	0.13	18.64	2.88
Henderson Island	2	19.24	0	0	0	0.88	0	19.11	0.5
	5	5.44	0.06	0	0	0.12	0	10.94	0.19
Temple Island	1	5	21	0.12	1.62	14	0.12	16.56	2.06
Aquila Island	1	1.44	34.94	0	0.38	10	0.06	15.31	1.19

Table A 7 Abundance of juvenile hard corals by genus. Total number observed per Reef and Depth, genera with at least 4 corals observed on any reef separated. All less abundant genus grouped as Other genera. Data from visual census of juveniles along transect.

Reef	Depth	*Acropora*	*Acanthastrea*	*Cyphastrea*	*Coelastrea*	*Dipsastraea*	*Echinophyllia*	*Favites*	*Goniopora*	*Leptastrea*	*Lobophyllia*	*Montipora*	*Paragoniastrea*	*Pocillopora*	*Porites*	*Psammocora*	*Turbinaria*	Other genera	Genus Richness	Number
Pine Peak Island	2	1	0	0	0	0	1	0	0	0	4	2	0	4	4	1	0	7	13	24
	5	0	1	0	0	1	0	1	0	0	2	1	0	4	3	1	0	7	13	21
Pine Islets	2	1	0	0	0	0	0	2	1	0	4	1	2	3	5	4	3	7	15	33
	5	7	5	1	0	5	5	6	5	0	6	3	4	4	5	0	10	19	24	85
Henderson Island	2	8	2	0	5	4	0	4	1	5	2	1	0	3	0	0	0	5	14	40
	5	5	2	0	0	0	0	3	0	0	2	3	0	2	0	0	0	3	9	20
Temple Island	1	40	0	15	0	22	0	18	1	1	1	5	3	2	20	1	47	13	20	189
Aquila Island	1	0	0	0	0	5	0	1	0	0	0	9	2	2	0	4	1	9	12	33

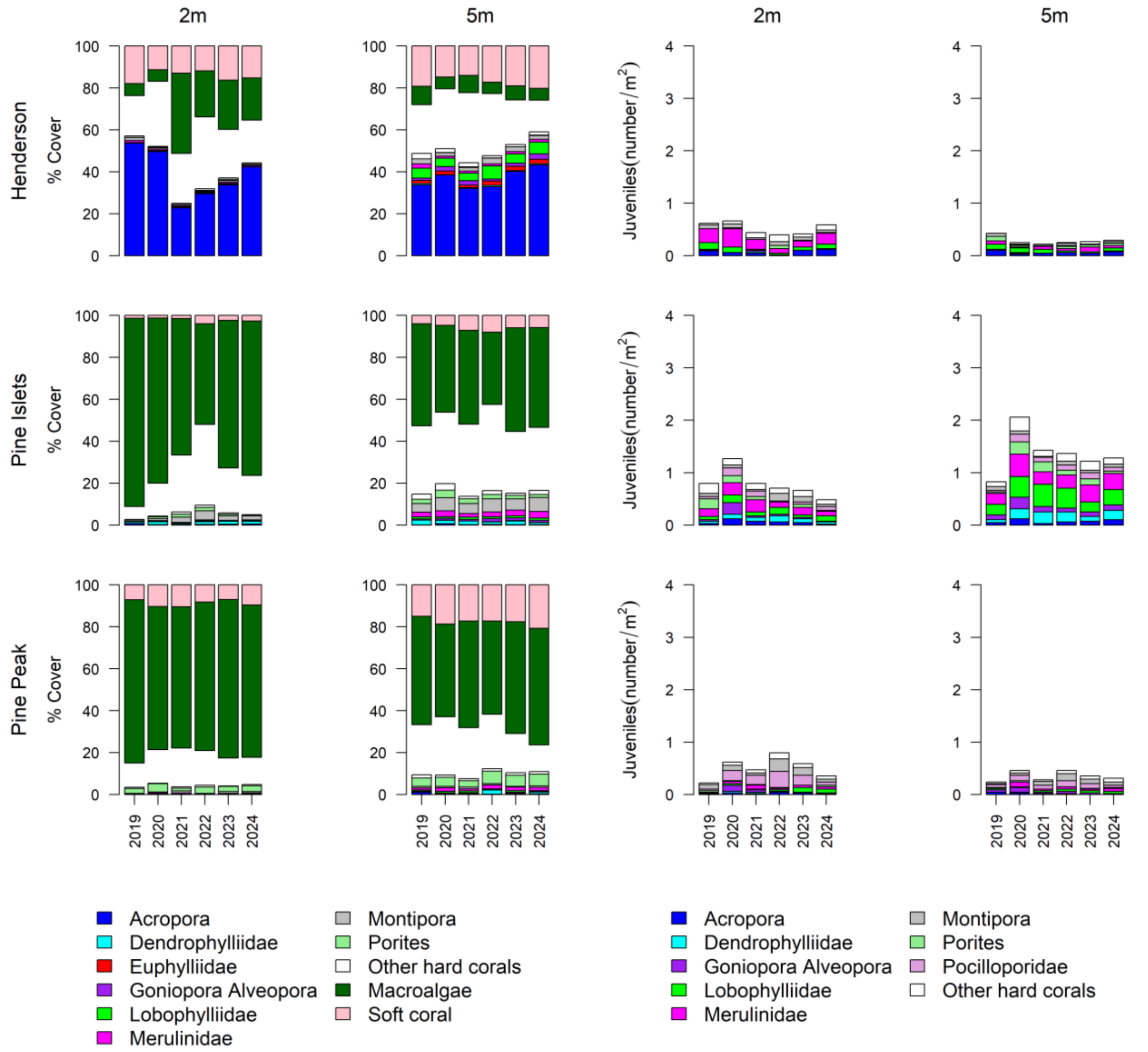

Figure A 2 Composition of benthic cover and hard coral juveniles. The left-hand plots show the breakdown of cover for hard coral families at 2 m and 5 m depths. Families that had a cover of at least 3% at either depth of any reef in the Zone are differentiated. Cover of all other families are grouped as Other. The cover of Macroalgae and soft corals are also included (hanging). The right-hand plots show the density of juvenile (< 5 cm) hard corals per m^2 of transect area by family at 2 m and 5 m depths.

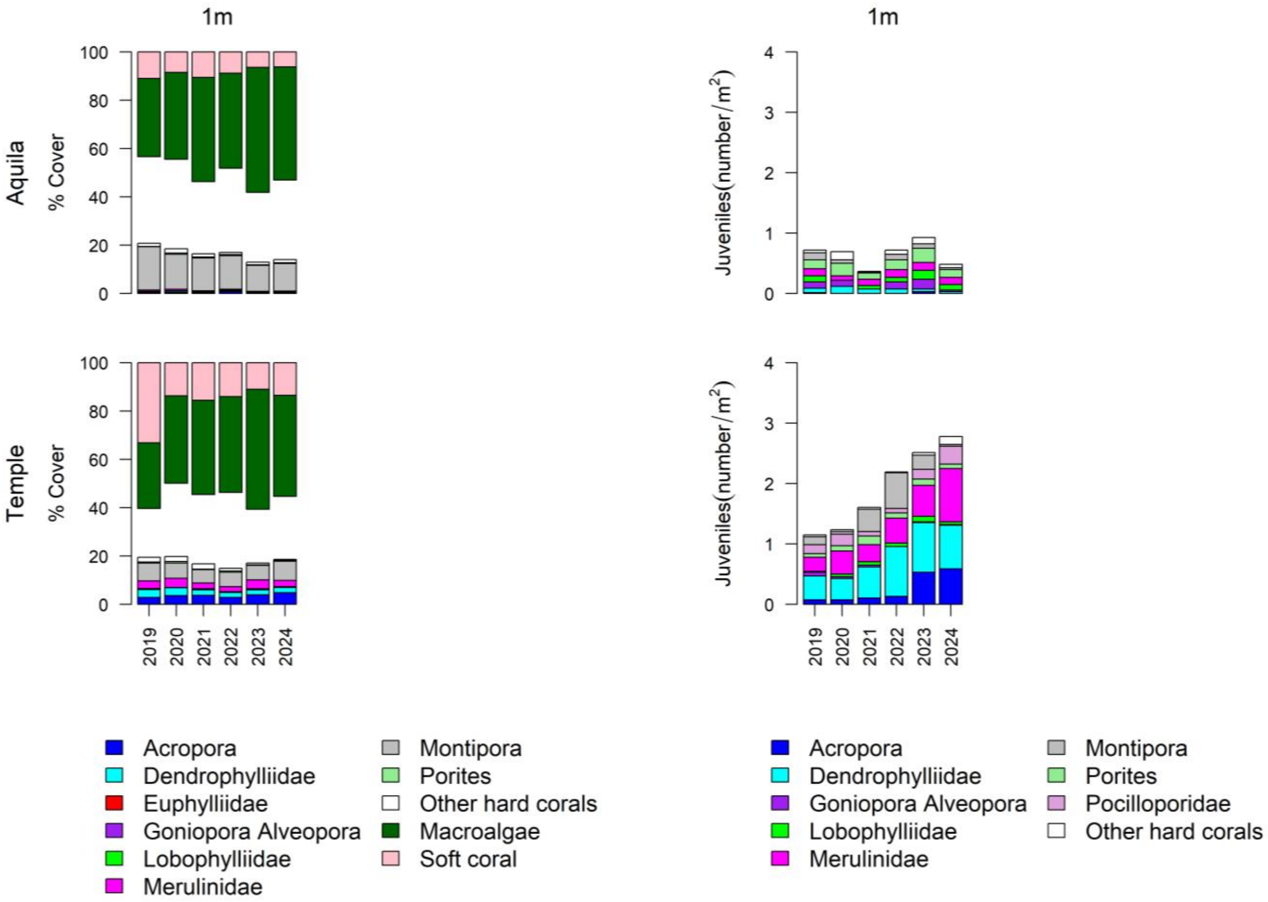

Figure A 2 continued, for the 1 m deep sites at Aquila and Temple Islands.

8.2 Images of benthic communities

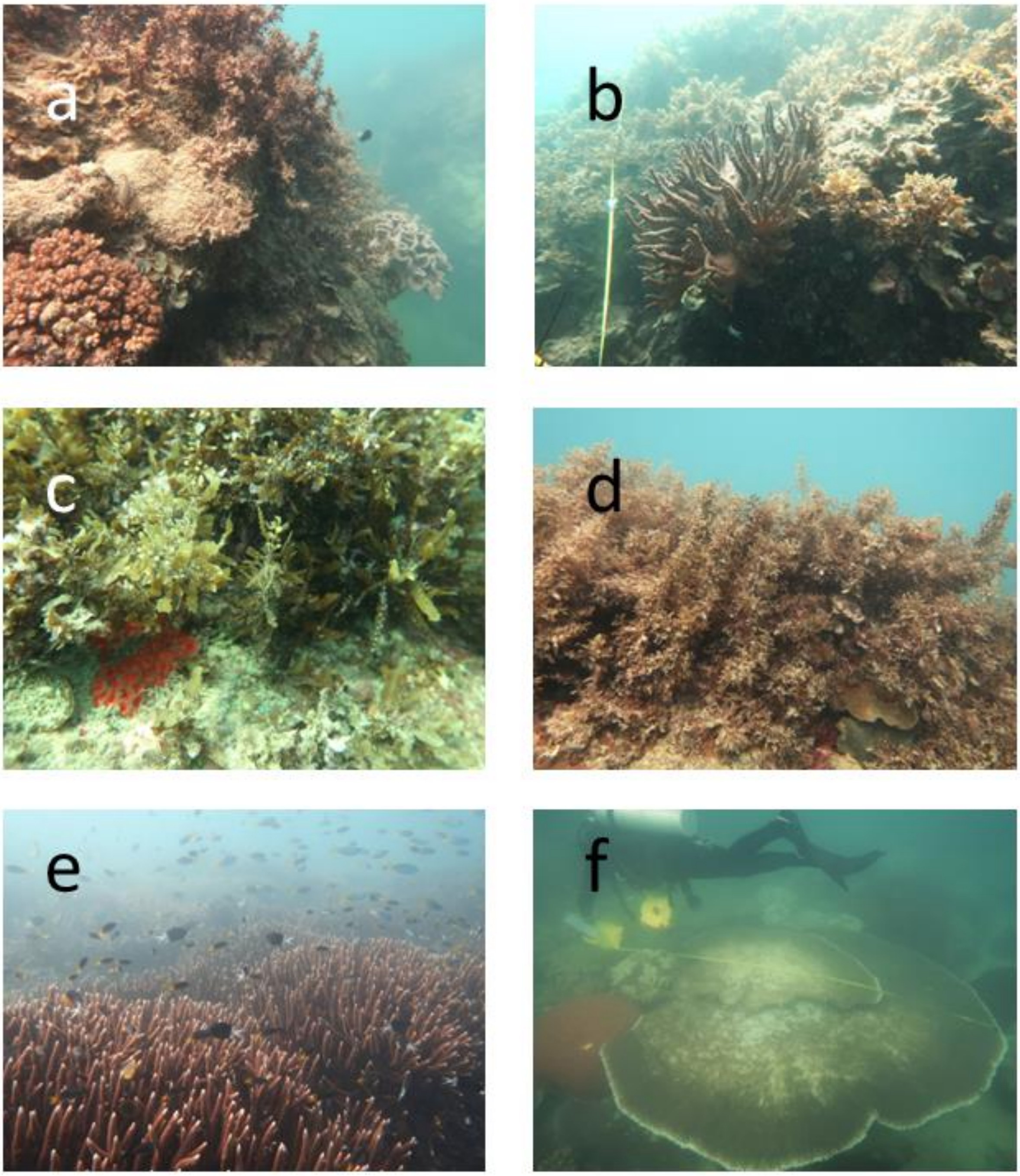

Figure A 3 Benthic community photos at outer reefs. Communities dominated by macroalgae at :a) Pine Peak Island 2 m, b) Pine Peak Island 5 m, c) Pine Islets 2 m, d) Pine Islets 5 m, contrast with the fields and large colonies of *Acropora* at e) Henderson Island 2 m and f) Henderson Island 5 m.

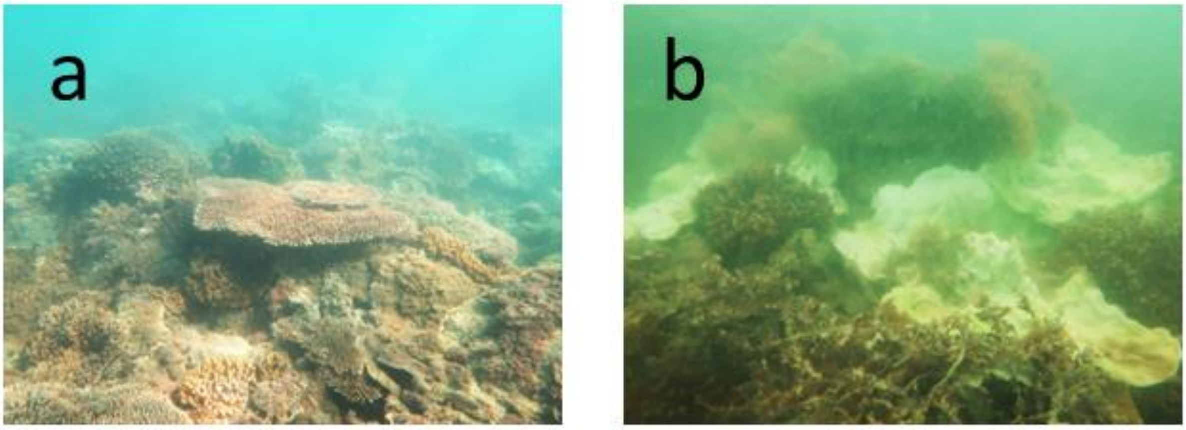

Figure A 4 Benthic community photos at inner reefs a) mixed colonies of hard corals and soft corals at Temple Island, b) *Montipora* bleaching among brown macroalgae at Aquila Island.

8.3 Logistical Considerations

The are several environmental constraints that need to be considered for the future monitoring of the Southern Inshore Zone coral communities.

The Broad Sound-Shoalwater Bay area has the highest tidal range along the Queensland coast. Surveys must be timed to coincide with neap tide periods to reduce the risk of strong currents and elevated turbidity. The resurveys were all undertaken during neap tides (generally < 3 m change between high and low tide over the period of survey). Wind driven resuspension can also reduce in-water visibility, and periods of wind speeds above 15kts require a following day or two of calm weather to allow settlement of suspended particles before surveys can begin.

The proximity of the survey locations in relation to coastal access points is a further consideration. In combination with the need to survey during periods of neap tides and low winds, the availability of suitable periods within which to undertake sampling is severely restricted. Access to Aquila Island is most convenient via Carmila Creek boat ramp. This requires ~3.5 m of tide at McEwen Island (Bureau of Meteorology Tide Predictions). Surveying Aquila Island from Carmila Creek meets the demand for quick access to the site and egress from falling tide. However, the most accessible launch point for Temple Island and the more offshore reefs is Sarina Beach, some 80 km from Pine Islets and Pine Peak Island. Given the distance to be travelled on the open waters, predicted winds below 15 knots are required. These reefs can be successfully resurveyed with winds in this range. Table A 8 provides a reference point for the conditions experienced during 2024 re-surveys and notes on the state required to avoid strong currents.

Table A 8 Weather conditions and tide heights experienced during 2024 works, with additional information on local currents. Tidal range taken from Percy Island for Pine Peak, Pine Islets and Henderson Island, Hay point for Temple Island and McEwen Islet for Aquila Island.

Reef	Date	Wind (knots)	Tide State during survey and range between nearest high and low water ()	Observations	Local currents typically experienced during tide cycle that may limit surveys
Henderson Island Site 2	12/6/2024	NW 7	High (2 m)	Visibility 7-8 m, no current	Current on falling tide. Much less current from mid-rise to high tide.
Henderson Island Site 1	13/7/2024	NW 8	Mid-rise (2.5 m)	Visibility 7-8 m, no current	Current on falling tide. Much less current from mid-rise to high tide.
Pine Islets Site 1	11/6/2024	SE 10	High and falling (2.1 m)	Visibility 6 m, no current	No limitation from local currents at time of survey.
Pine Islets Site 2	12/6/2024	NW 2	Low and rising (2 m)	Visibility 9 m, no current	No limitation from local currents at time of survey.
Pine Peak Island	12/6/2024	NW 2	Mid rise (2 m)	Visibility 8-11 m, no current.	Site 1: Current on falling tide. Both sites: Less current on rising tide.
Temple Island	12/7/2024	SW 6	Mid rise (2.4 m)	Visibility 7-8 m, current began to increase as tide rose at site 2	Both sites: Manageable current on rising tide. No limitation from local current on falling tide.
Aquila Island	12/7/2024	SW 5	Mid to high (3.2 m)	Visibility 1 m, strong current on rising tide at Site 1.	Site 1: Strong current on rising tide – much less around high tide. Site 2: Current workable on last hour of rising tide.

Table A 9 Waypoints and compass directions for transects and monitoring sites

Reef	Latitude S	Longitude E	Depth	Site	Tran	Compass directions
Pine Peak Island	21.51447	150.25145	2	1	1	350, 90@10 m rod
	Waypoint between transects 3 & 4				2	140, 120@10 m rod, 30@15 m, 340 to T3
					3	0, 120@12 m
					4	210, 300@4 m
					5	150, note first rod is at 3 m, contour
	21.51433	150.25125	5	1	1	340 then contour, 90 @ 10m
	Waypoint between transects 3 & 4				2	150, 110@6 m, 60@10 m rod, 320 to T3
					3	320 then contour
					4	240, 180@14 m
					5	contour
	21.51392	150.25532	2	2	1	190, 90@ 10 m rod
	Waypoint between transects 3 & 4				2	10, 50@10 m rod
					3	80, 200@10 m
					4	260, 300@3 m
					5	210, 340@4 m
	21.51375	150.25513	5	2	1	90 330@11 m
	Waypoint between transects 3 & 4				2	0, 100@2 m, 30@10 m rod, 120@15 m
					3	150, 90@10 m rod
					4	330, 260@7 m
					5	270, 190@9 m
Pine Islets	21.65762	150.22165	2	1	1	20, 0@5 m
	Waypoint between transects 3 & 4				2	300, 230 to T3
					3	240
					4	120
					5	50, 180@10 m
	21.65782	150.22162	5	1	1	280
	Waypoint between transects 3 & 4				2	350
					3	270, 240@10 m rod, 300@13 m
					4	120
					5	60, 100@10 m
	21.65717	150.21898	2	2	1	230, 190@10 m rod
	Waypoint between transects 3 & 4				2	340, 350@10 m
					3	240
					4	50, 90@10 m
					5	130
	21.65743	150.21917	5	2	1	200
	Waypoint between transects 3 & 4				2	270, 320@10 m rod
					3	270, 200@10 m rod
					4	30, 120@10 m rod
					5	180, 60@10 m rod

Table A 9 continued.

Reef	Latitude S	Longitude E	Depth	Site	Tran	Compass directions
Henderson Island	21.48542	149.90965	2	1	1	340
	Waypoint between transects 3 & 4				2	330
					3	330, 350@10 m rod
					4	150
					5	160, start shoreside PM
	21.4856	149.90907	5	1	1	310, 330@10 m rod
	Waypoint between transects 3 & 4				2	300 over large Lobophyllia end
					3	320, 20@10 m
					4	130, 100@10 m rod
					5	150, 200@10 m rod
	21.48313	149.90868	2	2	1	310
	Waypoint between transects 3 & 4				2	300
					3	320, 300@10 m rod
					4	120
					5	150
	21.48317	149.90845	5	2	1	0, 350@10 m rod
	Waypoint between transects 3 & 4				2	300, 320@10 m rod
					3	330, 310@10 m rod
					4	180, 170@10 m rod
					5	180
Temple Island	21.59608	149.50102	1	1	1	200, 170@10 m
	Waypoint between T1-T4				2	150, 180@10 m
					3	190
					4	350
					5	330, 310@10 m
	21.60285	149.49932	1	2	1	240, 220@10 m
	Waypoint between T1-T4				2	190, 200@10 m
					3	180, 190@10 m
					4	90, 30@10 m, 340@12 m, 300 to T5
					5	30, 50@10 m
Aquila Island	21.95682	149.58102	1	1	1	190, 180@10 m, 140 to T2
	Waypoint between T1-T4				2	140
					3	170
					4	320
					5	330, 310@10 m
	21.96112	149.58158	1	2	1	120
	Waypoint between T1-T4				2	90
					3	110
					4	0
					5	30