Southern Inshore Zone: Coral Indicators for the 2022 Mackay-Whitsunday-Isaac Report Card

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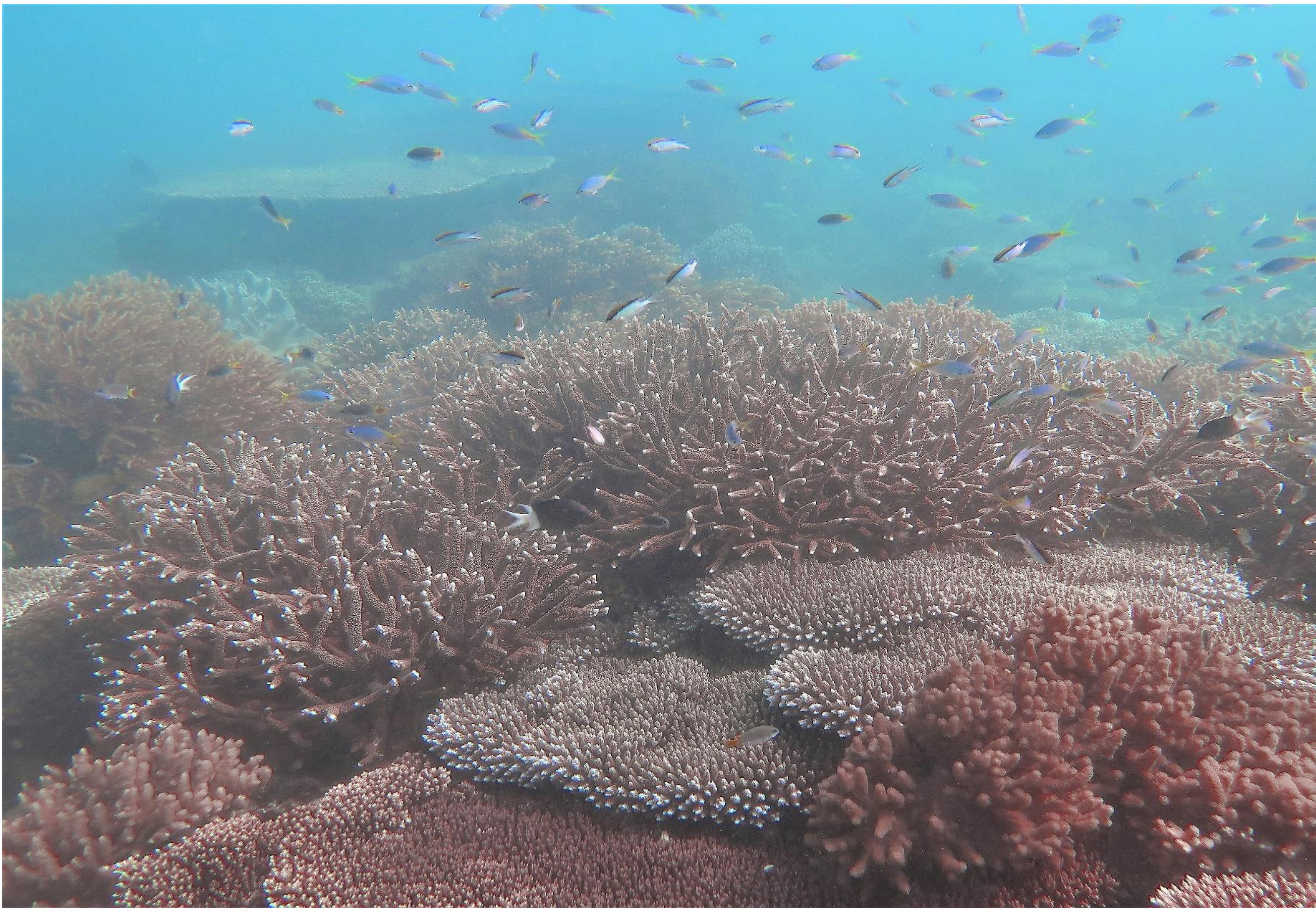

A report prepared for the Mackay-Whitsunday-Isaac Healthy Rivers to Reef Partnership

Australian Government
Australian Institute of Marine Science
AIMS: Australia's tropical marine research agency.
www.aims.gov.au

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This report should be cited as:
Davidson J, Thompson A, Thompson C (2022) Southern Inshore Zone – Coral Indicators for the 2022 Mackay-Whitsunday-Isaac Report Card. Report prepared for Mackay-Whitsunday-Isaac Healthy Rivers to Reef Partnership. Australian Institute of Marine Science, Townsville. (43pp)

The metadata record relating to this report:
Australian Institute of Marine Science (AIMS). (2020). Reef Catchments Southern Inshore Coral Monitoring Program, https://apps.aims.gov.au/metadata/view/0478852b-27ad-48dc-833f-ae93920ce73e

Project Leader shall ensure that documents have been fully checked and approved prior to submittal to client			
Revision History:	*Name*	*Date*	*Comments*
Prepared by:	*Johnston Davidson, Angus Thompson*	*25-10-2022*	
Approved by:	*Britta Schaffelke*	*31-10-2022*	
Reviewed by:	*Brie Sherow*	*22-06-2023*	Subsequent to submission, request for reviewer name to be updated
Revised by:	*Johnston Davidson*	*23-06-2023*	Reviewer name update completed

Cover photo: Henderson Island reef slope showing robust coral community in June 2022
Image: Tane Sinclair-Taylor

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1 EXECUTIVE SUMMARY

This report presents the 2022 results of 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 Southern Inshore Zone in the 2022 Mackay-Whitsunday-Isaac Report Card.

Between June and July 2022, the Australian Institute of Marine Science (AIMS) resurveyed benthic communities at permanent coral monitoring locations at five reefs in the Southern Inshore Zone. The overall report card grade for community condition in 2022 improved to D ('poor'), based on a Coral Index score of 0.25 (Figure 1).

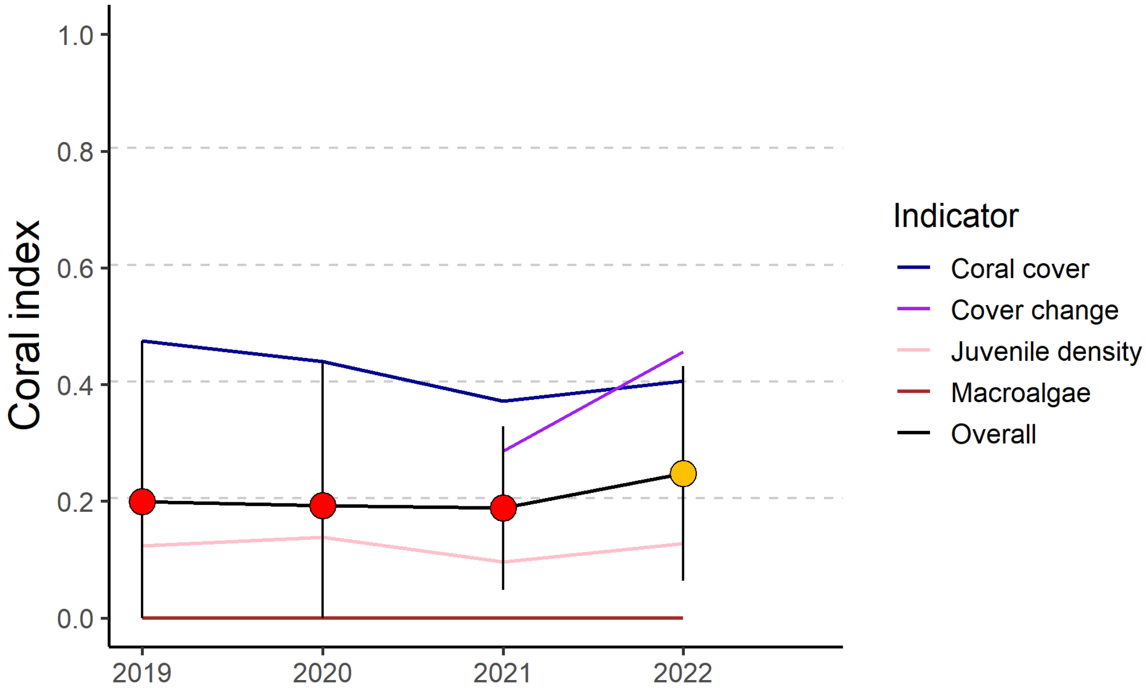

Figure 1 Coral Index and indicator scores. The Coral change indicator was added in 2022 and back calculated for 2021, as such the plotted score for the Coral Index of 0.19 in 2021 is slightly higher than the previously reported value of 0.16 that did not include the Coral change indicator.

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
- **Coral change** -the rate at which hard coral cover increased.

The Coral change indicator was incorporated into the Coral Index for the first time in 2022.

The improvement in the Coral Index from 0.16 in 2021 reflects modest gains in the Coral cover and Juvenile density indicator scores. In addition to contributing to the return of Coral cover scores into the ‘satisfactory’ range, increased cover of hard corals occurred at a rate that ensured the Coral change indicator was also scores as ‘satisfactory’.

The improvement in the Coral Index reverses the declining trend observed in previous years and signals the early recovery of coral cover following the impacts of a marine heat wave that caused severe coral bleaching in 2020 and the subsequent loss of coral cover across the region.

In March 2022 another thermal heat wave occurred, however there were few signs of bleaching at the time of our surveys. Improvement in the Coral cover indicator and ‘satisfactory’ condition reported for the first implementation of the Coral change indicator demonstrate that this event did not severely impact corals in the region.

The Coral cover and Coral change indicators demonstrate the ongoing resilience and growth of adult corals. However, the ongoing high cover of macroalgae at most reefs saw the continued score of 0 for the Macroalgae indicator (Figure 1). While the Juvenile density indicator improved slightly in 2022, the density of juvenile corals remains very low on most reefs. In combination the ‘very poor’ scores for the Macroalgae and Juvenile density indicators put downward pressure on the Coral Index and demonstrate the ongoing limited potential for coral recruitment across the region.

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 (Thompson *et al*. 2020a). 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 fourth 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 2022 that expands on the necessarily succinct summary of overall condition presented by the 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.

At each reef, two replicate sites separated by at least 150m 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 on the day of site construction.

Sites at Henderson Island were setup in 2018 by a third party and the depth of some transects at site 1 are set 1-3 m deeper than the intended 5 m datum. At Temple Island and Aquila Island the reef slope transitioned to sand at 1-1.5m below LAT and as such transects were set at 1m 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 7.

Most reefs were monitored in June 2022, with the exception of Aquila Island that was monitored in July 2022 (Table 1).

Table 1 Dates of coral monitoring.

Island	2019	2020	2021	2022
Pine Peak Island	27th January	26th May	6th March	5th June
Pine Islets	28th January	27th May	6-7th March	4th June
Henderson Island	29th January	25th -26th May	7th March	4th June
Temple Island	27th May	27th-28th May	3rd June	3rd June
Aquila Island	27th May	12th July	3rd June	6th 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.* 2008). Along the upslope side of each transect line, digital images of the substrate were taken at ~40cm 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: 0-2cm and >2-5cm found within a strip 34cm 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.* 2009) 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.* 2020a), the development of which is described in detail in Thompson *et al* 2020b. The indicators, Coral cover, Macroalgae and Juvenile density are estimated on the most recent observation for each reef and have been used since the start of this program. The Coral 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 Coral change in 2021 are supplied, although values 2021 should be treated with caution as relate only to changes at Aquila, Temple and Pine Islets 2 m. AIMS do 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. 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.* (2020b).

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). 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*. 2016). 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 Coral 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 Coral 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.* 2016), 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:

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

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

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

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

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

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

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

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 2m	0.2%	3.4%
	Pine Peak Island 5m	0%	6.3%
	Pine Islets 2m	0.2%	5.4%
	Pine Islets 5m	0%	6.4%
	Henderson Island 2m	0.2%	3.9%
	Henderson Island 5m	0%	6.7%
	Temple Island 1m	0.3%	23%
	Aquila Island 1m	0.3%	23%
Juvenile density	All	13 m^{-2}	0 m^{-2}
Coral change	All	2* upper 95% CI	Hard Coral cover declined

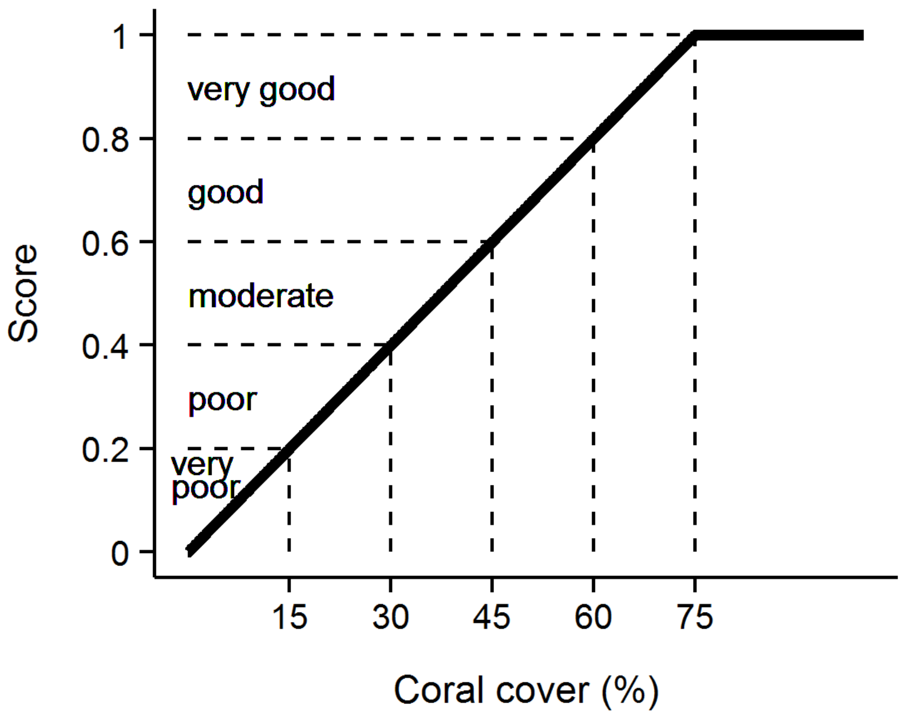

Figure 3 An example of a scoring diagram, here for the Coral Cover metric.
Numeric scores and associated condition classifications based on observed coral cover are presented (see also Table 2).

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. The Coral change indicator trend is not plotted due to limited data as of 2022.

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 a number of 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 Acroporiidae, this is split further into genus groups *Acropora* and *Monitpora*. Photos representative of coral communities at each reef and depth in 2022 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), and anomalously high summer temperatures resulting in coral bleaching (Berkelmans *et al.* 2004; Sweatman *et al.* 2007). 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 (2m and 5m), Henderson Island (2 m and 5 m), and Aquila Island (1 m). These loggers were retrieved during our resurveys in 2020, 2021, and 2022. 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 (updated from previous reports to be calculated from available data 1990 – 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. The factors were supplied by James Cook University and reflect those.

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. 2016). 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

The in-situ temperature pattern across all monitored sites in the region describes a marine heat wave in early 2022. Temperatures peaked in March but the duration and intensity of this event were reduced compared to the heat wave in 2020 (Figure 4).

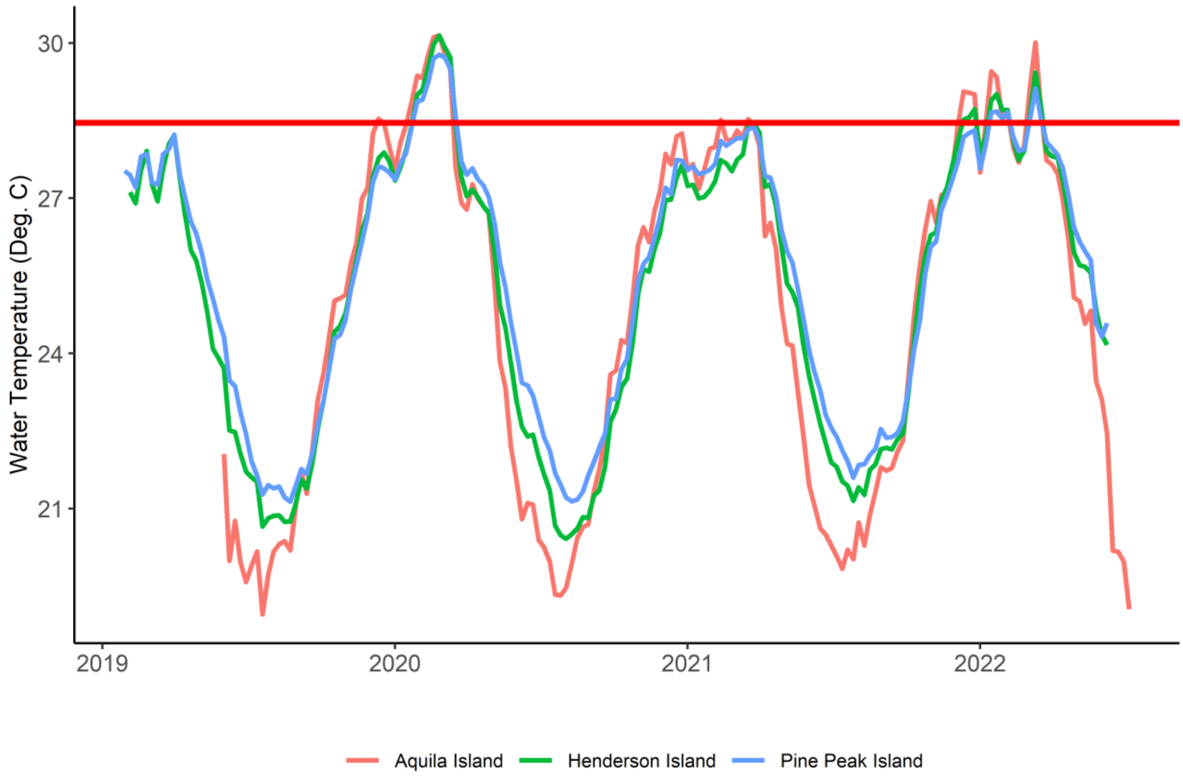

Figure 4 Temperature profiles recorded by in-situ loggers. Temperatures recorded during the hottest months of 2020, 2021 and 2022. The horizontal reference line was derived from the mean of the means of the hottest month each year observed over timeseries of in-situ temperature data available from reefs in Whitsunday Islands. This baseline excluded years in which bleaching was observed.

The observed temperatures in 2022 were below that which caused widespread bleaching and subsequent loss of coral cover in 2020. This was supported by the estimates of degree heating weeks (DHW) from satellite observations processed by the NOAA Coral Reef Watch (CRW) daily global 5km resolution coral bleaching heat stress monitoring program (Figure 5). The five panels show the thermal signature for non-bleaching years (2018, 2019, 20021) and the contrast with years 2020 and 2022 when a marine heat wave passed through the region. In comparison with the heat signature for 2020, the DHW estimates for 2022 indicated potential for moderate bleaching on the more offshore reefs but potential for severe bleaching close to the coast (Figure 5).

As an explanatory note, and to place the 2022 summer in perspective, the degree heating week estimates for 2020 were the highest recorded over the last four years of the survey (Figure 5). 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.

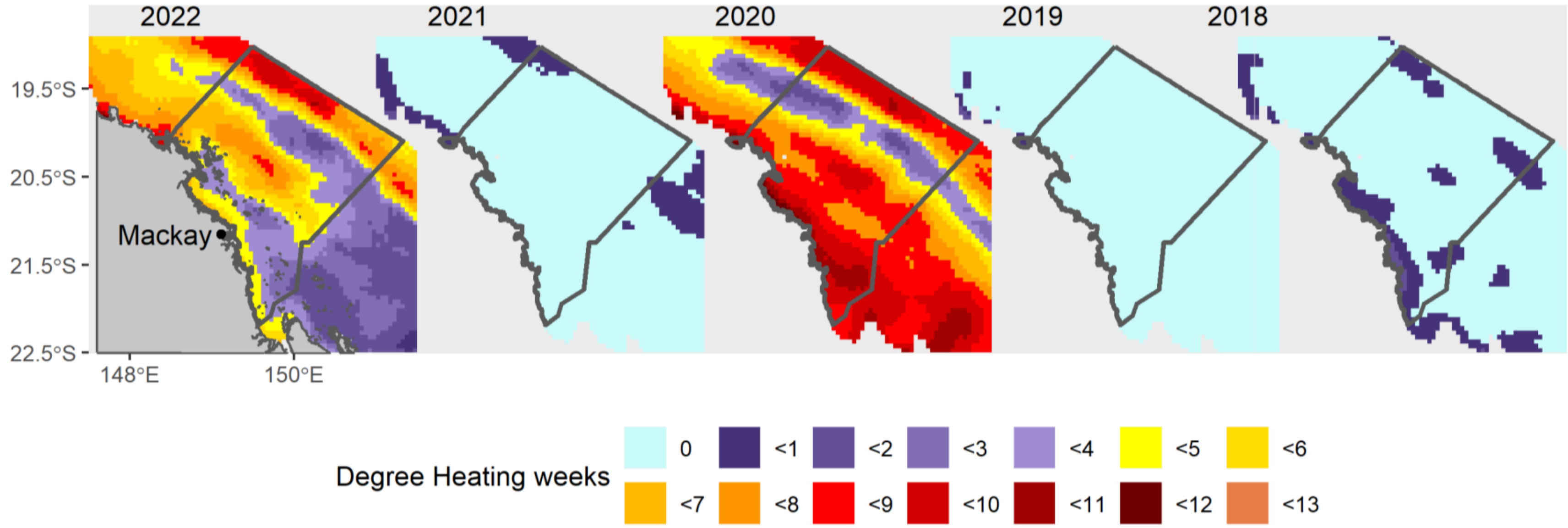

Figure 5 Annual estimates of thermal stress to corals. Degree Heating Week estimates downloaded from NOAA coral reef watch. DHW values as indicators of thermal stress on the Great Barrier Reef are interpreted as follows: DHW values from 0 - 2: no bleaching (i.e., normal summer conditions), 2 - 4: possible bleaching, 5 - 8: severe bleaching and some coral mortality, >8: extreme bleaching and widespread mortality likely (Cantin *et al* 2021).

4.1.2 Runoff

River discharge 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 2021-2022 water-year (October to September) increased from below median levels in the north to 2.2 times median levels for Water Park 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; Thompson *et al.* 2016), 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 (1986-2016). 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
Pioneer	125007A	558,735	5.9	2.5	1.9	1	0.2	1	2.3	0.4	1.9	0.6	0.4	0.5
Plane	126001A, 126003A	848,985	4.6	2.8	1.9	0.8	0.2	0.9	2.8	0.3	1.3	1.1	0.5	0.3
Waterpark Creek	129001A	349,614	4.8	1.5	5.2	2.9	2.2	1.8	2.6	1.4	0.7	1.5	1.8	2.2

4.1.3 Cyclones and Storms

There were no cyclones likely to have impacted reefs in the Southern Inshore Zone during the 2021-2022 cyclone season. 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 buoy 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). Cyclone Marica, a category 5 system, came closest to the reefs reported here, tracking southwards past Middle Percy with winds in excess of 80 knots before crossing the coast at Shoalwater Bay on February 20th 2015 (Figure 6). Waves from TC Marcia were the fourth highest waves recorded at the Emu Park buoy. 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 Cyclone Marcia.

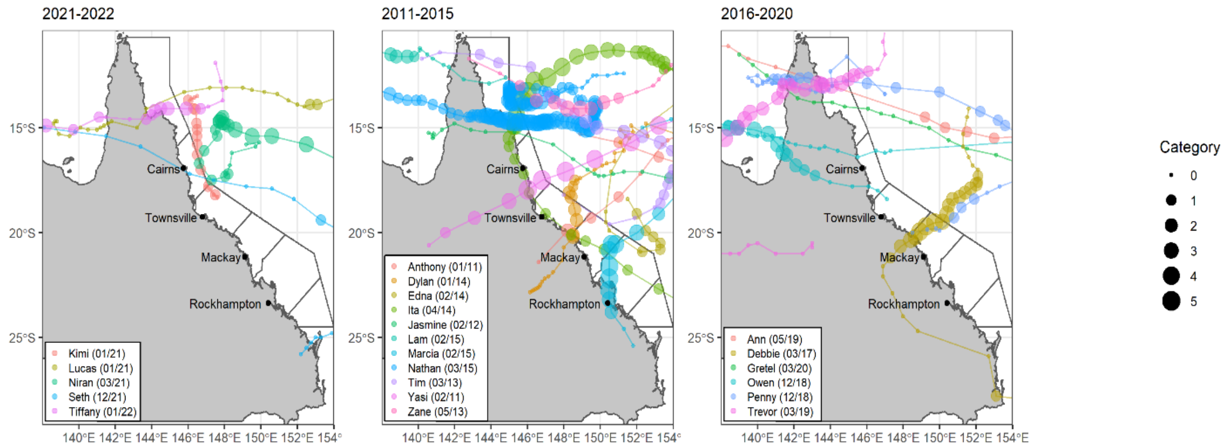

Figure 6 Tracks of tropical cyclones passing through the region.
All cyclones crossing through the Mackay Whitsunday Isaac regional report card reporting area over the last 15 years are displayed**.**

4.1.4 Biological Damage

A total of 13 colonies were identified with disease across four of the five reefs. Diseased colonies ranged from branching *Acropora* to massive *Porites* (Table A 5). In addition, there were a total of 13 colonies for which recent mortality was unknown. In combination, these 26 colonies spanned 4 genera, down from last year's observations (31 colonies, 10 genera), and substantially lower than the 55 colonies across 12 genera observed in 2019 (Figure A 1). Given the relatively high level of *Acropora*-dominated coral cover, it's no surprise that the reef community at Henderson Island had the greatest number of affected colonies (17 *Acropora* spp). Brown band, black band, and white syndrome diseases were represented at both depths among a minor number of affected colonies.

The number of colonies being overrun by the encrusting sponge *Cliona orientalis* have dropped from 15 colonies and six genera in 2021 to six colonies and four genera in 2022. As previously noted, most observations of *C. orientalis* continue to occur among those inshore reefs with higher turbidity; in 2022 this was Temple Island. Afflicted colonies represent a range of genera; Cyphastrea, *Hydnophora, Platygyra, and Turbinaria* (Table A 5).

No crown-of-thorns seastars have been observed in this study. Another coral predator, the gastropod *Drupella,* made a notable appearance at Henderson Island (5m) where 20 individuals were observed feeding on branching *Acropora* colonies (Table A 5). Within the dominant coral community at this location the density of the corallivore is relatively low, however its continued presence may result in increased observations of patchy coral mortality.

Partial bleaching, due to exposure to thermal stress, was observed on a few individual *Acropora* colonies on the shallow sites at Henderson Island and also among *Goniopora* and *Monitpora* colonies on the reef slopes (Table A 5). A similar mix of partially bleached colonies was observed on the reef slope of Pine Islets. There were no bleached corals recorded by photo-transects, only by the wider survey of the scuba-search, re-affirming the conclusion that the influence of the marine heat wave in 2022 was considerably less than 2020.

With no significant storms over the 2021-2022 season, physical damage was restricted to a few occurrences at the outer reefs of Pine Peak Island, Henderson Island, and Pine Islets (Table A 5). Given their increased level of exposure compared to Temple and Aquila islands, the dislodgement of substrate is likely due to normal attrition by wave action, anchor damage, or both.

4.2 Coral Community Condition Assessment

The overall coral index score for the Southern Inshore Zone in 2022 was graded as D, categorising the coral communities as being in 'poor' condition (Table 5Table 5**Error! Reference source not found.**. This improvement in grade from previous years reflects minor improvement in scores of the Juvenile coral and Coral cover indicators but also the 'Satisfactory' condition of the Coral change indicator.

The Coral change indicator was included for the first time in 2022. The score for 2021 (Table 5) is back calculated for this report and was not included in the 2021 report card. Valid estimates of Coral change were only available for changes between 2020 and 2021 at Temple Island, Aquila Island and the 2 m depth of Pine Islets due to the ongoing influence of the 2020 marine heat wave on coral cover changes at the remaining reefs. The score for this indicator is estimated as the mean of changes over the previous four years but excludes changes that were considered to have been influenced by an acute event. The marked increase between 2021 and 2022 scores show that coral cover increases between 2021 and 2022 surveys were, on average, meeting expectations based on prior monitoring of inshore coral communities.

The increase in observed hard coral cover, that resulted in a 'Satisfactory' score for the Coral change indicator, logically contributed to the slight improvement in the Coral cover indicator (Table 5). Across the reporting zone the mean cover of hard and soft corals increased to 30% (Table 6). There was a slight improvement in the score for Juvenile corals, however the density of juveniles remains low and within the range previously observed (Table 5, Table 6). Although, the proportion of macroalgae has been reduced (Table 6), it remains at high levels across all reefs and the score for this indicator remains zero (Table 5).

Table 5 Coral Index and indicator scores for 2022. The Coral change indicator was added in 2022 and back calculated for 2021. Report Card scores and grades calculated on the original three indicators, as reported in 2021, are given in parentheses. Scores are coloured as per Table 3.

	Year	Juvenile corals	Coral cover	Macroalgae	Coral 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 (0.16)	E (E)
	2022	0.13	0.40	0	0.46	0.25 (0.18)	D (E)

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 m^2)		Coral cover (%)		Macroalgae proportion (%)	
		Mean	SD	Mean	SD	Mean	SD
Zone summary	2019	1.42	0.96	35.62	23.78	65.79	20.91
	2020	1.59	0.85	32.96	20.14	60.52	25.45
	2021	1.11	0.86	27.87	14.34	65.78	19.88
	2022	1.52	1.15	30.37	17.10	58.91	20.41

The overall index score continues to mask the substantial differences in the condition of coral communities between reefs (Table 7). Across the region, most reef-level index scores have risen since 2021. The 2 m depths of Pine Peak Island and Pine Islets continue to have the lowest scores of 0.08 and 0.09 respectively. During this study very low coral cover and juvenile densities at these sites has kept index scores consistently low compared with the deeper 5 m community at the same reefs, and markedly lower than the other reefs across the region. Index figures for the deeper communities of Pine Peak Island and Pine Islets have improved due to increased coral cover, and at Aquila Island and Temple Island due to increased juvenile density. Consistently minimum scores of zero for the macroalgae indicator are highly influential in the low grade among reefs in the Southern Inshore Zone (Table 5).

Table 7 Index grade and scores for each reef and depth combination. Comparison of the 2022, 2021, 2020, and 2019 Index figures. Scores in parentheses in 2022 exclude Coral change scores to allow comparison across the time-series. Scores are coloured as per Table 3.

Reef	Depth	Index 2019	Index 2020	Index 2021	Index 2022	Grade
Pine Peak Island	2	0.05	0.09	0.08	0.14 (0.08)	E
	5	0.12	0.14	0.12	0.36 (0.15)	E
Pine Islets	2	0.04	0.06	0.06	0.24 (0.09)	E
	5	0.12	0.20	0.15	0.26 (0.17)	E
Henderson Island	2	0.41	0.34	0.19	0.27 (0.22)	D
	5	0.36	0.33	0.28	0.36 (0.31)	D
Temple Island	1	0.32	0.21	0.23	0.18 (0.24)	E
Aquila Island	1	0.19	0.16	0.14	0.14 (0.15)	E

4.3 Coral Cover

Coral Cover scores are based on the combined cover of hard and soft corals. Coral Cover scores improved at five of the eight reef-depth locations in 2022 (Table 8, Figure 7a). These improvements were led by increases in hard coral cover observed at seven of the eight reef-depth locations. Changes in soft coral cover between surveys have been minor and variable (Table 8).

Hard coral cover has increased across the region from 16.7% in 2021 to 19.3% in 2022. Signs of recovery can be seen particularly at Henderson Island where the marine heat wave of 2020 caused hard cover to decline to through to 2021 (Table 8). The improvement in hard coral cover observed in 2022 were due mostly to growth of surviving Acropora colonies at 2 m depth (22.8% cover in 2021 up to 29.5% cover in 2022) (Table A 1). *Acropora* cover at 5 m remains steady while an increase in

Lobophyllia and a doubling of the soft coral *Klyxum* further contributes to the community recovery (Table A 2), promoting the Coral cover score into the 'very good' range (Table 8).

Minor increases in quick-growing foliose *Montipora* coral cover at both depths at Pine Islets (Figure A 2) contributed to the improvement of the Coral cover score, as did minor increases in the robust *Porites* and *Turbinaria* at Pine Peaks Island 5 m (Figure A 2). While the soft coral *Sinularia* was reduced by half at Pine Peaks Island 5 m (Table A 2), the overall influence of changes in soft coral cover at these three reef-depth locations was minimal (Table 8, Figure A 2). Declines in the Coral cover score observed at Aquila Island and at the 2 m reef flat of Pine Peak Island were due primarily to slight reductions in soft coral cover (Table 8). Minor declines in both hard and soft coral cover at Temple Island demoted the Coral cover score to 'poor' (Table 8). Previously moderately common at this reef, both *Platygyra and Pocillopora* were reduced below 1% cover in 2022 (Table A 1).

Notably, there was an increase in genus richness at many locations (Table 2 A). In tandem with an increase in cover, this suggests an upward pressure of both diversity and abundance among the reef communities suggests a certain resilience to recent bleaching events.

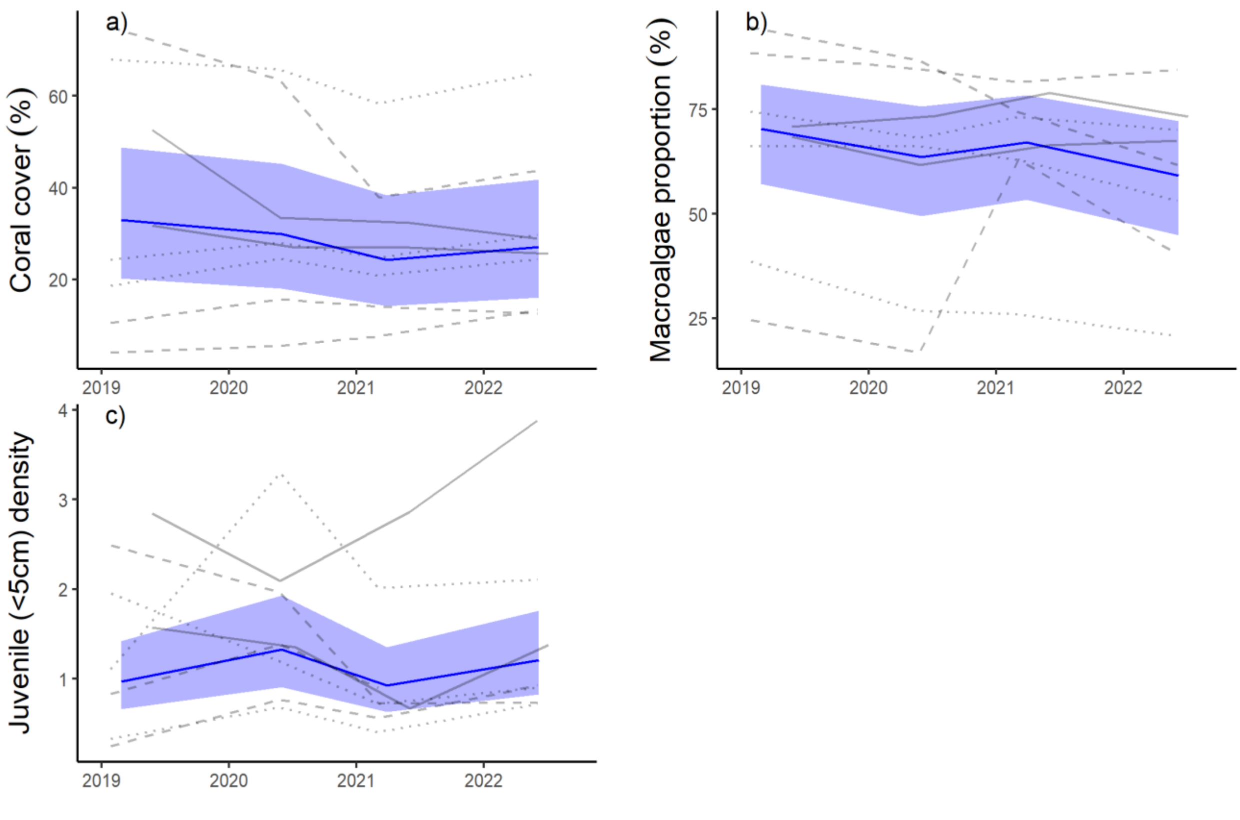

Figure 7 Indicator trends for Southern Inshore Zone.
Blue lines represent trends in: a) coral cover, b) macroalgae proportion, c) juvenile density. Trends are bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5m (dotted lines), 2m (dashed lines), and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

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

Reef	Depth	Year	Hard coral cover (%)	Soft coral cover (%)	Coral cover (%)	Coral cover Score
Pine Peak Island	2 m	2021	3.69	10.44	14.14	0.19
		2022	4.38	8.19	12.57	0.17
	5 m	2021	7.5	17.31	24.81	0.33
		2022	12.34	17.29	29.63	0.40
Pine Islets	2 m	2021	6.19	1.44	7.63	0.1
		2022	9.5	3.94	13.44	0.18
	5 m	2021	13.75	7.15	20.85	0.28
		2022	16.42	7.92	24.35	0.32
Henderson Island	2 m	2021	24.94	12.94	37.88	0.51
		2022	31.88	11.88	43.76	0.58
	5 m	2021	44.31	14	58.31	0.78
		2022	47.69	17.31	65.00	0.87
Temple Island	1 m	2021	16.72	15.59	32.3	0.43
		2022	14.94	14.00	28.94	0.39
Aquila Island	1 m	2021	16.44	10.56	27	0.36
		2022	17.00	8.82	25.82	0.34

4.5 Macroalgae Proportion

While most reefs showed a general decline in macroalgae cover (Figure 7b), the proportion of macroalgae continued to exceed thresholds across all reefs (Table 2), and so the continuation of the grade of E ('very poor', Table 9) for this indicator. The mean proportional macroalgae cover for the region in 2022 was 58.9%, the lowest in the series of annual surveys by this program.

Declines in the brown macroalgae *Lobophora* were common across all reefs in 2022. The largest declines of this genus occurred at Pine Islets with cover of this genus (Table A 3) almost halving compared to 2021 levels (Davidson *et al.* 2021). The brown macroalgae *Sargassum* also decreased at Pine Islets 2 m but gained cover at 5 m. At Henderson Island (2 m) *Lobophora* declined from 32% in to 2021 to 21% in 2022 (Table A 3) and *Sargassum* cover, while less abundant in 2021 (3%), had all but disappeared in 2022. By contrast, Pine Peak 2 m, Temple and Aquila islands had relatively large increases in *Sargassum* (10.1%, 8.5%, 2.5%, cover respectively). Red macroalgae has a modest but important presence in the region (Table A 3), with small gains made at all reefs except at Temple and Aquila islands where small losses occurred.

Table 9 Macroalgae cover and indicator scores for each location.
Scores are coloured as per Table 3.

Reef	Depth	Year	Macroalgae cover (%)	Macroalgae proportion (%)	Macroalgae score
Pine Peak Island	2 m	2021	67.41	81.48	0
		2022	70.85	84.42	0
	5 m	2021	50.81	73.18	0
		2022	44.44	70.14	0
Pine Islets	2 m	2021	65.19	74.39	0
		2022	48.06	61.67	0
	5 m	2021	44.75	62.85	0
		2022	34.58	53.25	0
Henderson Island	2 m	2021	38.38	63.04	0
		2022	21.94	40.25	0
	5 m	2021	8.31	25.88	0
		2022	5.38	20.62	0
Temple Island	1 m	2021	39.02	66.5	0
		2022	39.69	67.55	0
Aquila Island	1 m	2021	43.13	78.95	0
		2022	38.90	73.40	0

4.6 Juvenile Density

The density of juvenile corals across the region continues to be low at all reefs. The juvenile score has been 'poor' or 'very poor' through the four years of this monitoring program (Table 10). However, with the relative respite from disturbances over the last two years, there has been a modest increase in juvenile abundance at most reefs that led to a slight improvement in Juvenile coral indicator score (Figure 7c, Table 10).

Increases in juvenile abundance and density this year have yet to promote a regional Juvenile coral score beyond the grade of E, 'very poor' (Table 5). The only reef with sufficient density of juvenile corals to achieve a 'poor' score was Temple Island (Table 10) where the juvenile density is bolstered by moderate numbers of *Turbinaria* and *Pocillopora* recruits (Figure A 2, Table A 4).

Table 10 Juvenile hard coral abundance, density and indicator scores for each location.
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	2021	32	0.57	0.05
		2022	54	0.94	0.08
	5 m	2021	19	0.4	0.04
		2022	31	0.72	0.06
Pine Islets	2 m	2021	54	0.9	0.08
		2022	48	0.88	0.08
	5 m	2021	97	2.01	0.17
		2022	93	2.13	0.19
Henderson Island	2 m	2021	30	0.73	0.06
		2022	27	0.73	0.06
	5 m	2021	15	0.72	0.06
		2022	17	0.90	0.08
Temple Island	1 m	2021	109	2.87	0.25
		2022	149	4.02	0.35
Aquila Island	1 m	2021	15	0.67	0.06
		2022	49	1.36	0.12

4.7 Coral change indicator

The Coral change indicator was included for the first time in 2022 as this was the first year that all reefs included a disturbance free period. Across the region the score for this indicator was 0.46 (Table 5) demonstrating that , on average, the increase in cover between 2021 and 2022 surveys, and 2020 and 2021 surveys for Temple, Aquila and Pine Islets 2 m (Table 8) was within the bounds of modelled expectations.

The scores did vary among reefs (Table 11). At both Temple Island and Aquila Island hard coral cover declined between 2020 and 2021 resulting in scores of zero. At Temple Island cover declined again in 2022 ensuring the continued zero score for the Coral change indicator. Although hard coral cover increased from 2021 to 2022 at Aquila, this increase was modest and, when combined with the zero score from the previous year, the Coral change score was again 'very poor'.

In contrast, hard coral cover increased at all other reefs between 2021 and 2022 (Table 11). Most of these increases were at or above modelled expectations and Coral change indicators ranged from 'satisfactory' to 'very good' (Table 11). It is noteworthy that of the offshore reefs it was only at 2 m depth at Pine Peak Island, where the cover of macroalgae is extremely high (Table 9), that Coral change score was 'poor'.

Table 11 Reef level Coral change scores.

Only years for which Coral change was estimated are included. Annual scores for each reef are a running mean over up to four years. In 2022 a max of two observations were available for any reef.

Reef	Depth	Year	Change in percent cover of hard coral cover from previous year	Coral change score
Pine Peak Island	2 m	2022	0.7	0.30
	5 m	2022	4.8	1
Pine Islets	2 m	2021	1.9	0.86
	5m	2022	3.3	0.69
	5 m	2022	2.7	0.51
Henderson Island	2 m	2022	6.9	0.54
	5 m	2022	3.4	0.50
Temple Island	1 m	2021	-3.1	0
		2022	-1.8	0
Aquila	1 m	2021	-2.1	0
		2022	0.6	0.11

5 DISCUSSION

The overall condition of Southern Inshore Zone reefs in 2022 was categorised as ‘poor’ and graded ‘D’ based on a Coral Index score of 0.25. This score includes the Coral change indicator for the first time and, as such, is not directly comparable to previously reported scores. Excluding the Coral change indicator returns a Coral Index score of 0.19, a slight improvement on the reported score for 2021 of 0.16. Either way, the 2022 score capture the first improvement in coral community condition at the regional scale since the program began in 2019.

Most influential in previous declines was the reduction in coral cover following the bleaching event of 2020 that caused declines in coral cover in in both the 2020 and 2021 surveys (Davidson *et al.* 2020, 2021). None of the reefs in the region escaped bleaching in 2020. The brunt of the impact was seen at Henderson Island which had significantly more coral cover, principally *Acropora*, than the other reefs in the region. At the time of the 2020 survey, 76% of the shallow corals and 58% of the slope corals had bleached. Re-surveys in 2021 established mortality in the shallows of over 50% and 13% on the slopes. Water temperatures in 2022 were again above median levels and peaked in early March. The extent of the seawater temperature anomaly was lower than that observed in 2020, although temperatures close to the coast were in the range where, based on degree-heating-week estimates, server bleaching was predicted (Cantin et al. 2021). While partial bleaching was observed on individual coral colonies on some transects at Henderson Island and Pine Peak Islands, the timing of surveys in June and July 2022 means that some bleached corals may have either died or recovered by this time. However, that the corals had survived the more extreme bleaching event of 2020 makes it unlikely that the lower stress observed in 2022 will have resulted in coral mortality.

The limited impact of the 2022 marine heat wave on the corals in the region was demonstrated by the increase in hard coral cover at all reefs, Temple Island being the only exception. It was these increases in hard coral cover that are primarily responsible for the improved Coral Index score in 2022. In addition to contributing to modest improvement in the Coral cover indicator score, the rate of change in hard coral since 2021 ensured the Coral change score was in the ‘satisfactory range’.

The Cover change indicator assess the rate of change in hard coral cover, by comparing observed coral cover to the cover predicted by applying a growth function to the cover observed the previous year. A score of 0.5 is achieved when hard coral cover has increased by the predicted amount. In 2022 there were, marked differences in the scores or the Cover change indicator among reefs. Of the three reefs furthest from the coast, it was only at Pine Peak Island 2 m depth that the Coral change score fell into the ‘poor’ range. Of note is that the cover of macroalgae is extremely high at this location and will have been in direct competition with corals for light and space. Scores for the Coral change indicator were ‘very poor’ at both Temple and Aquila Islands. Both these reefs were classified as having no bleaching impact in 2021 as there was limited evidence of ongoing bleaching-related mortality during surveys in 2020 (Davidson *et al*. 2020). This means that the Cover change indicator considered the average scores for changes in hard coral cover between 2020 to 2021 and 2021 to 2022. As hard coral cover declined at both reefs between 2020 and 2021 it seems likely that longer term stress associated with the 2020 bleaching event were realised. However, at neither reef did hard coral cover changes between 2021 and 2022 reach modelled expectations, suggesting additional pressures influenced the growth of corals at these more coastal reefs.

Of ongoing concern for the resilience of coral communities in this region are high levels of macroalgae and low density of juvenile hard corals. Although the scores for these indicators improved in 2022 both remain 'very poor'. The cover of macroalgae across the region declined to the lowest level recorded over the last four years, however, at 60%, this value is still very high. Further, 2022 surveys of the more offshore reefs were undertaken in early June. While this was similar to the late May surveys in 2020, the timing of these surveys was substantially later in the year than the March surveys of 2021 or January surveys of 2019. As cover of macroalgae can vary seasonally, we cannot discount seasonality as an explanation for the reduced cover of macroalgae at this stage.

In combination, persistently high cover of macroalgae and low density of juvenile hard corals point to a bottleneck in hard coral recruitment processes. High cover of macroalgae can negatively impact the recruitment of hard corals and so suppress the recovery or resilience of coral communities (Birrell 2008b, Foster et al. 2008, Diaz-Pulido *et al*. 2010, Mumby *et al*. 2013, Leong *et al*. 2018). Of particular interest is the high cover of *Lobophora* macroalgae at several reefs. The persistent cover of this algae has been attributed to low recruitment of hard corals and lack of recovery of coral cover on an inshore reef off Townsville (Johns *et al.* 2018).

The abundance of juvenile corals continued to be low and highly variable across reefs and depths. With the exception of Henderson Island, where coral cover includes a high proportion of relatively fast-growing corals of the genus *Acropora*, both the juvenile and communities share a mix of resilient but slow growing taxa. The environmental conditions of the Southern Inshore Zone have been identified by previous studies as a challenging environment for corals (Hopley *et al.* 1983, van Woesik 1992, Kleypas 1996, van Woesik & Done1997). The location has few well-developed reef structures, with most formation over the last 6000 years being in the form of incipient reefs derived from accumulated detritus rather than consolidated carbonate substrate. 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, a large tidal range causing strong tidal currents, and proximity to the shallow, silt-laden Broad Sound, resulting in environmental conditions that challenge the resilience of coral communities. Indeed, examining inshore reef structures and coral reef communities between the Whitsundays and Keppel Island groups, Kleypas (1996) and van Woesik & Done (1997) interpret reduced reef development, abundance, and diversity of hard corals as reflecting environmental conditions that are less favourable for coral reef development.

When considering the scores for coral communities in this zone it is important to remember that the scoring thresholds used were devised for coral communities in less challenging environmental settings. While the scoring system used was parameterised for inshore reefs of the Great Barrier Reef, the underlying data were for the most part derived from reefs with better developed carbonate structures than those of the Southern Inshore Zone; Pine Islets, where there is a broad reef flat, being a clear exception. As such, while the scoring system allow comparison of these communities to those in other inshore zones, they may impose unrealistic expectations on the condition of coral communities subject to the ambient environmental conditions of the area. AIMS are currently leading a project under the Reef Integrated Monitoring and Reporting Project to develop an indicator framework for assessing coral community across the Reef. Central to this revision is the provision for thresholds used for scoring that explicitly account for variation in communities across environmental gradients. It is hoped that this revised indicator framework will be adopted across the various reporting platforms and may provide more realistic thresholds for the scoring for reef condition in this region.

6 ACKNOWLEDGEMENTS

We acknowledge the valuable field assistance given by Andrew Giandomenico, and Tane Sinclair-Taylor from AIMS during the 2022 surveys. We'd like to acknowledge the First Nations people, the Yuwibara, Barada Barna, Darumbal people who are the custodians and owners of the sea country we work in, and we pay our respects to their elders past, present and emerging.

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8 APPENDICES

8.1 Reef Level data summaries

Table A 1 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.

Reef	Depth	*Acropora*	*Galaxea*	*Goniopora*	*Lobophyllia*	*Montipora*	*Pachyseris*	*Porites*	*Turbinaria*	Other HC	Genus Richness
Pine Peak Island	2	0.06	0	0.06	0.06	0.25	0	3.00	0	0.94	10
	5	0.19	0.25	0.13	0.06	0.63	0.19	5.95	1.94	3	22
Pine Islets	2	0.31	0.06	0.12	0.06	4.25	0	1.69	1.31	1.69	16
	5	0.31	0.06	1.44	0.19	6.28	1.12	2.07	0.88	4.07	26
Henderson Island	2	29.5	0.06	0.44	0.25	0.44	0	0.06	0.06	1.06	12
	5	32.81	2.06	0.25	5.88	2.69	0	0.12	0.62	3.25	19
Temple Island	1	2.88	0	0.06	0	6.00	0	0.50	2.00	3.50	18
Aquila Island	1	0.81	0	0.06	0.31	13.88	0	0.44	0	1.50	14

Table A 2 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

Reef	Depth	*Briareum*	*Klyxum*	*Lobophyton*	*Sarcophyton*	*Sinularia*	*Xenia*	Other SC
Pine Peak Island	2	4.88	0	0.50	0.62	1.88	0.12	0.19
	5	12.71	0.63	0.57	0.69	2.19	0	0.50
Pine Islets	2	0.56	0.00	2.12	0.19	0.69	0	0.38
	5	3.84	0.75	0.19	1.02	1.94	0.06	0.13
Henderson Island	2	1.88	4.44	1.19	2.00	1.94	0	0.44
	5	0.69	11.06	0.06	3.00	1.50	0	1.00
Temple Island	1	3.50	0	0.44	0.19	9.81	0	0.06
Aquila Island	1	0.06	0.31	1.56	0.88	4.63	1.19	0.19

Table A 3 Cover of algae.
Identified macroalgae genera with a cover of at least 1% at any reef are separated. All less abundant or un-resolved genera and smaller algae are grouped.

Reef	Depth	Brown macroalgae			Red macroalgae	Green macroalgae	Turf algae	Coralline algae
		Lobophora	*Sargassum*	Other				
Pine Peak Island	2	7.57	56.09	0.81	6.32	0.06	8.69	4.38
	5	24.88	14.61	1.13	3.07	0.75	11.14	7.77
Pine Islets	2	7.62	29.69	3.31	7.25	0.19	26.62	3.25
	5	14.49	15.70	1.56	1.83	1.00	25.44	4.92
Henderson Island	2	21.25	0.06	0.06	0.56	0.00	31.75	0.81
	5	5.12	0.00	0.06	0.19	0.00	20.63	0.06
Temple Island	1	2.38	31.5	0.06	5.50	0.25	16.31	2.75
Aquila Island	1	0.88	32.02	0.19	6.13	0.06	11.94	2.31

Table A 4 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.

Reef	Depth	*Acanthastrea*	*Acropora*	*Coelastrea*	*Cyphastrea*	*Dipsastraea*	*Duncansomnia*	*Favites*	*Goniopora*	*Homophyllia*	*Leptastrea*	*Lobophyllia*	*Montipora*	*Moseleya*	*Paragoniastrea*	*Platygyra*	*Pocillopora*	*Porites*	*Psammocora*	*Stylocoeniella*	*Turbinaria*	Other genera	Genus Richness	Number
Pine Peak Island	2	1	3	0	0	0	0	0	1	0	0	1	1	0	0	0	16	21	4	4	0	2	11	54
	5	2	0	0	0	0	0	1	1	0	1	1	0	0	0	0	9	8	1	0	1	6	14	31
Pine Islets	2	5	4	1	0	5	0	0	2	4	0	0	0	0	0	0	10	1	2	5	8	1	12	48
	5	7	4	0	1	2	0	7	4	2	1	13	6	0	1	4	4	7	4	0	13	13	26	93
Henderson Island	2	2	1	5	0	0	0	0	0	0	5	0	4	0	0	1	5	0	0	0	0	4	9	27
	5	0	4	0	0	1	0	1	0	0	0	1	3	0	0	0	3	1	0	0	0	3	10	17
Temple Island	1	0	9	1	4	9	6	6	0	0	0	0	6	4	5	2	40	5	0	0	50	2	15	149
Aquila Island	1	0	0	0	3	2	0	4	5	0	0	1	11	4	0	0	6	0	2	0	5	6	13	49

Table A 5 Coral health survey results.

Number of colonies along the ten 20 m long and 2 m wide transects searched at each reef and depth combination in 2022 having recently lost tissue (patches of bare white skeleton) attributed to a range of causes. Anchor or physical damage and bleached corals are recorded as a proportion of coral cover at the site effected: 0 = absent, 0+ = individual colonies, -1 = 1-5%, +1 = 6-10%, 2 = 11-30%, 3 = 31-50%, 4 = 51-75%, 5 = 76-100%.

Cause	**Genus**	**Pine Peak**		**Pine Islets**		**Henderson**		**Temple**	**Aquila**
		2m	**5m**	**2m**	**5m**	**2m**	**5m**	**1m**	**1m**
Disease	*Acropora*					5	3		
	Montipora			1					2
	Porites		1						
	Psammocora		1						
Unknown cause	*Acropora*		1			5	4		
	Montipora								3
Sponge - *Cliona orientalis*	*Cyphastrea*							1	
	Hydnophora							1	
	Platygyra							1	
	Turbinaria					1		2	
Drupella	*Acropora*						20		
Total number of Colonies			3	1		11	27	5	5
Bleaching (proportion of colonies)					0+	0+	0+		
Physical (proportion of colonies)			0+		0+	0+	0+		

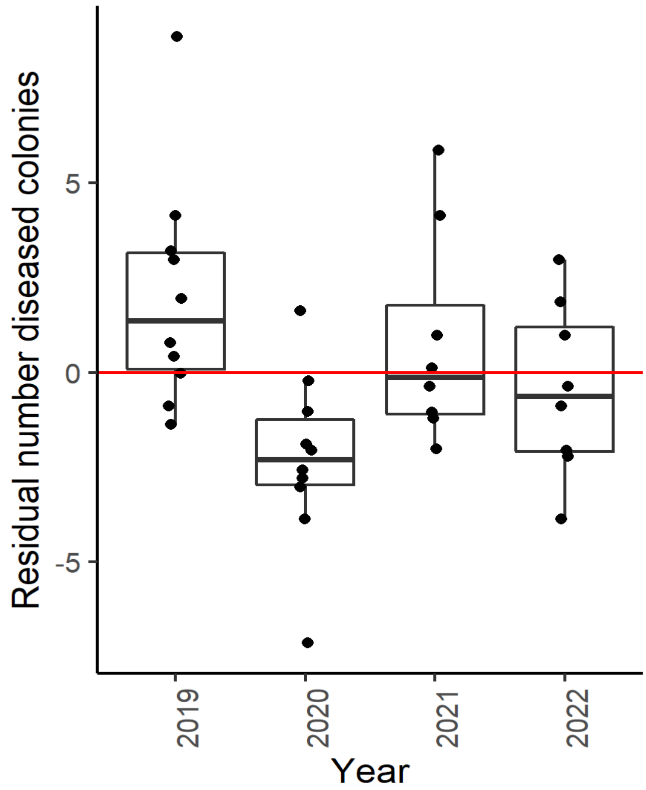

Figure A 1 Relative coral disease by year.
Data are standardised to the reef and depth mean across years. Boxplots show the median (bold horizontal line), 25th to 75th quartiles (box), and 1.5 times the inter-quartile range (whiskers). Solid dots are the relative number of coral colonies suffering ongoing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

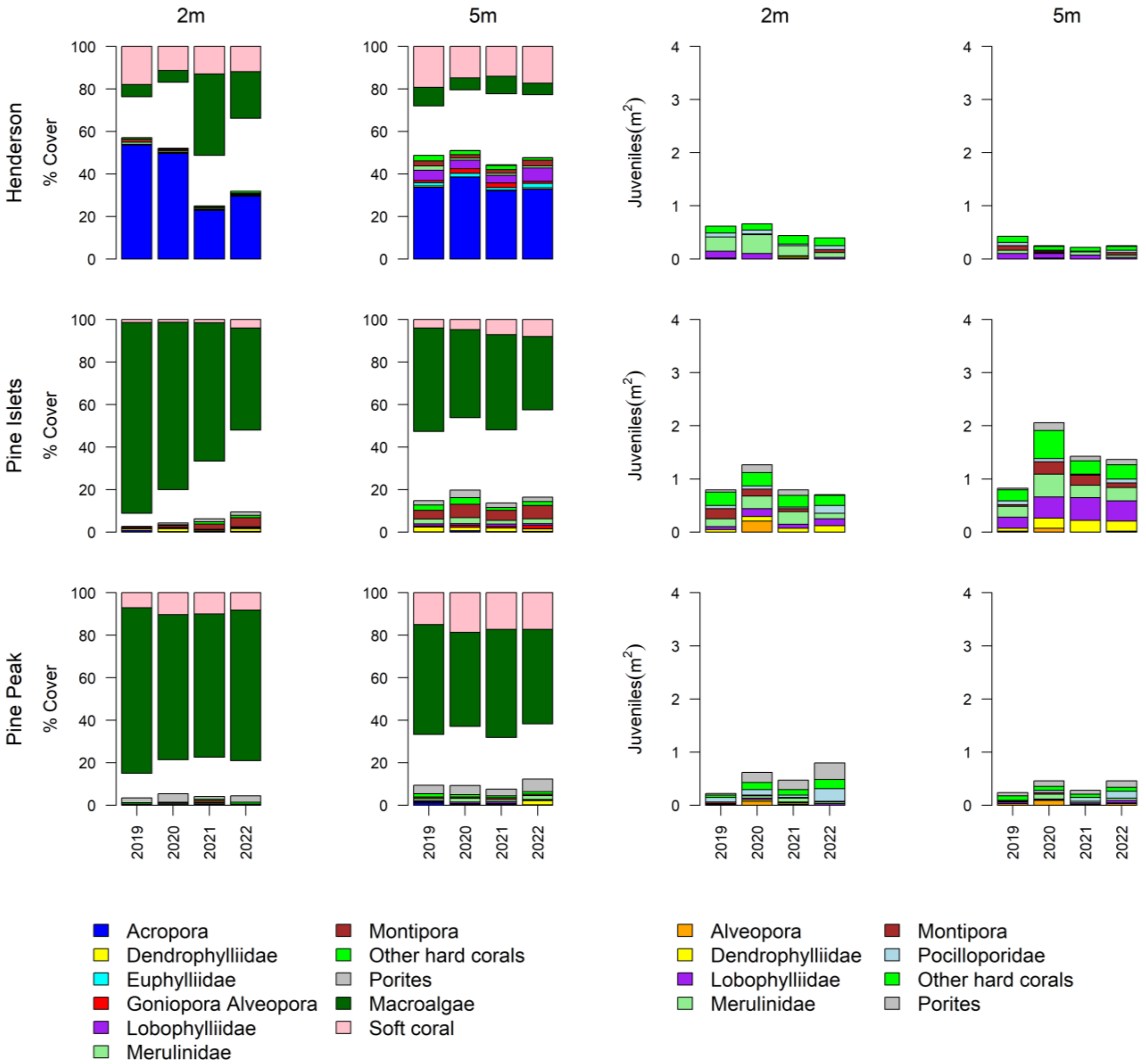

Figure A 2 Composition of benthic cover and hard coral juveniles.
The left-hand plots show the breakdown of cover for hard coral families at 2 m and 5 m depths. Families that had a cover of at least 3% at either depth of any reef in the Zone are differentiated. Cover of all other families are grouped 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 by family at 2 m and 5 m depths.

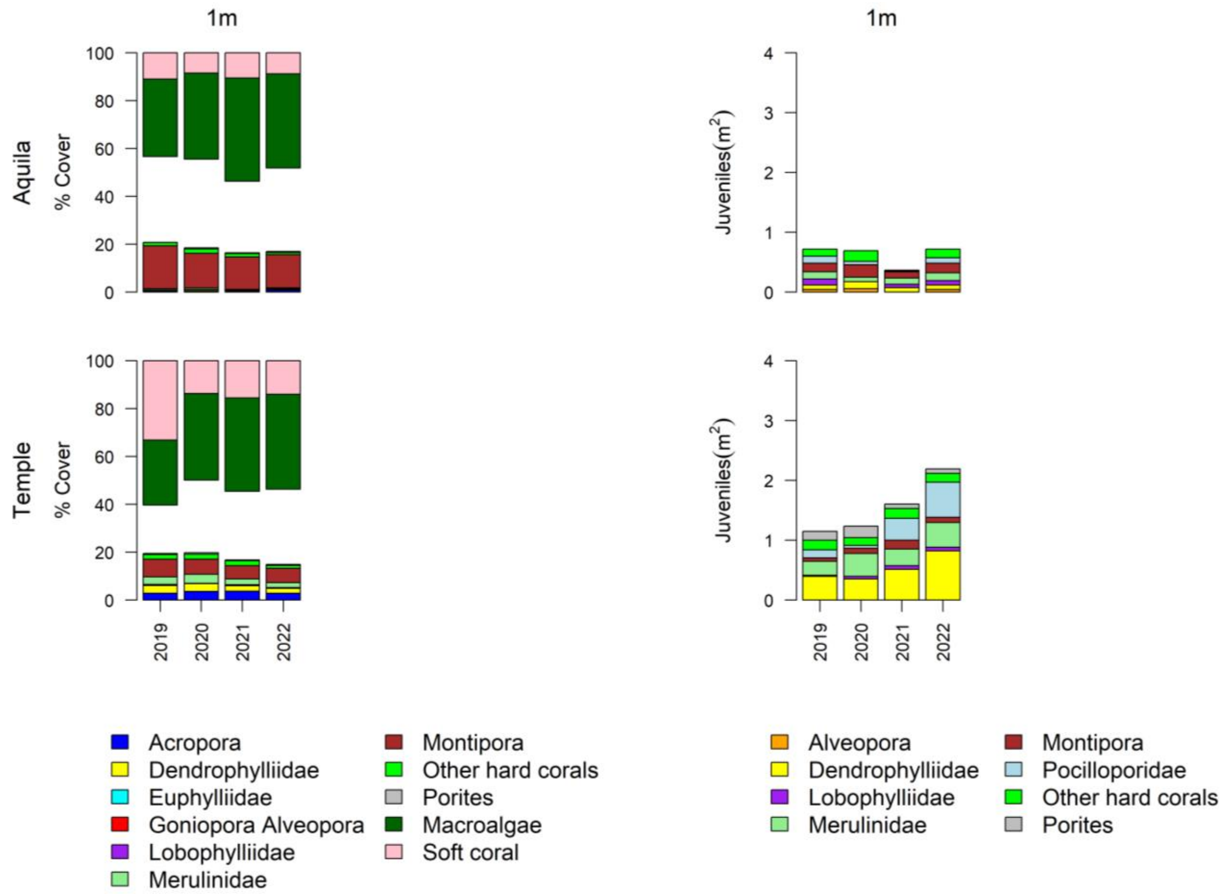

Figure A 2 continued, for the 1 m deep sites at Aquila and Temple Islands.

8.2 Images of benthic communities

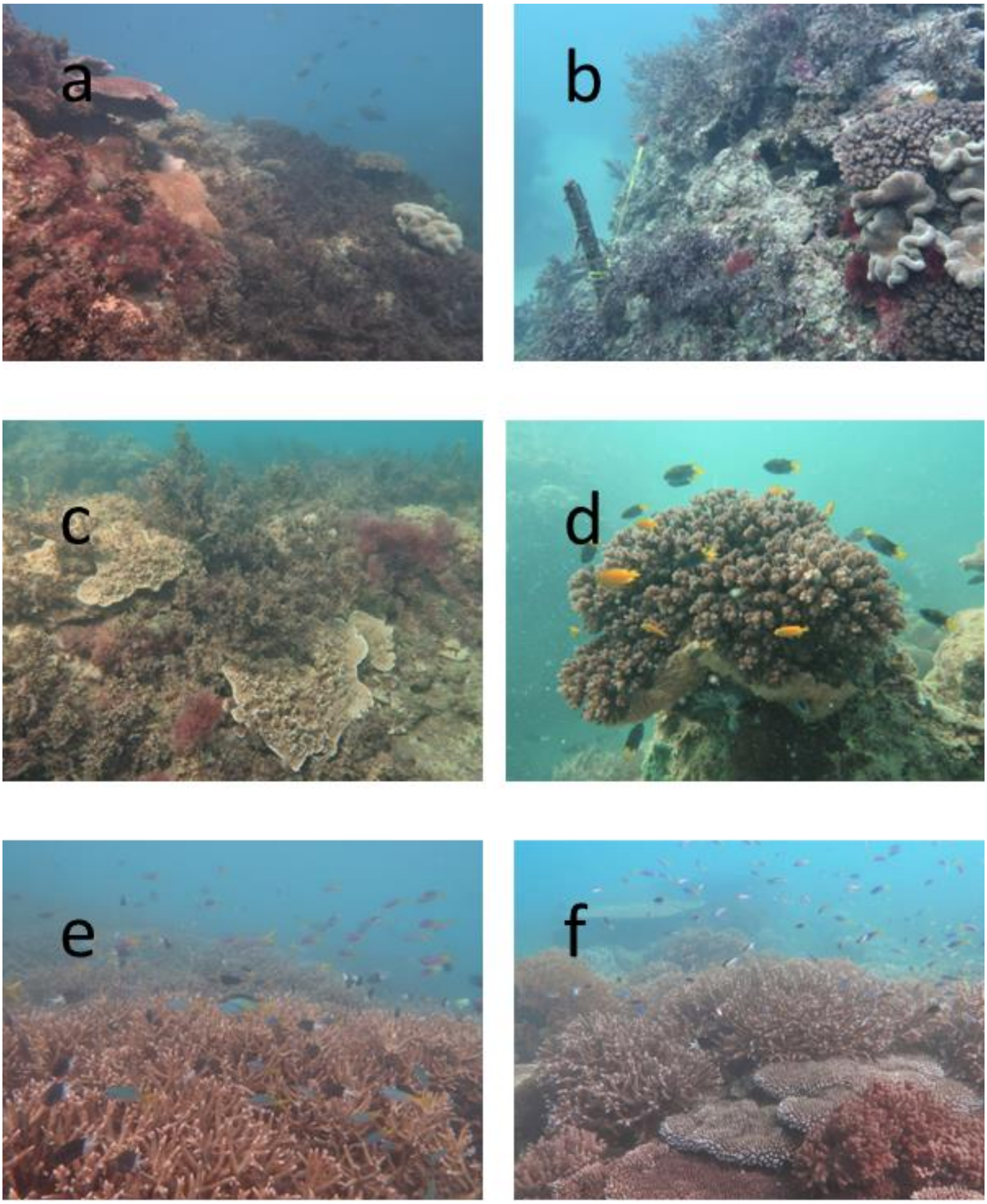

Figure A 3 Benthic community photos at outer reefs.

a) Pine Peak Island 2m b) Pine Peak Island 5m c) Pine Islets 2m d) Pine Islets 5m e) Henderson Island 2m f) Henderson Island 5m.

Despite the dominance of macroalgae at Pine Peak Island and Pine Islets at 2m and 5m, growth of scattered colonies such as *Acropora*, *Montipora,* and *Pocillopora* indicates continued recovery of coral cover. By contrast, fields of *Acropora* corals at Henderson Island thrive at both 2m and 5m during this respite from widespread disturbance.

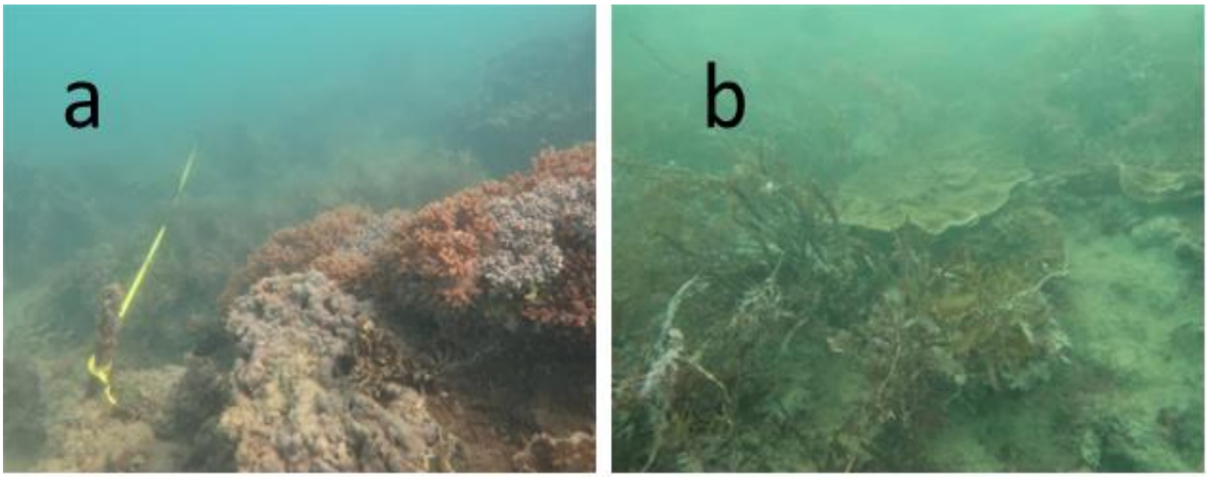

Figure A 4 Benthic community photos at inner reefs
a) Temple Island 1m b) Aquila Island 1m.
A mix of soft corals and encrusting *Montipora* at Temple Island 1m. At Aquila Island 1m, colonies of encrusting *Montipora* among withered stalks of *Sargasum* macroalgae.

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 < 3m 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. This requires ~3.5m 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 <15 knots are required. These reefs can be successfully resurveyed with winds in this range. The 2022 resurvey was fortunate to have a rare opportunity when neap tides and good weather coincided in June to allow safe survey of these outer reefs. Suitable conditions to survey Aquila Island did not occur until July. Table A 7 provides a reference point for the conditions experienced during 2022 re-surveys.

Table A 6 Weather conditions and tide heights experienced during 2022 works.
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
Henderson Island Site 2	4/6/2022	6 W	Mid rise (1.9m)	Visibility 6 -7m
Pine Islets	4/6/2022	5 E	Either site of high (1.9 m)	Visibility 9m
Henderson Island Site 1	5/6/2022	10 E	Mid rise (1.8 m)	Visibility 12m
Pine Peak Island	5/6/2022	5-10 E	Either side of high (1.8 m)	Visibility 10m
Temple Island	3/06/2022	7 E	Mid rise (2.4m)	Visibility 4-5m
Aquila Island	6/7/2022	0-2 W	Rising to high (3.4 m)	Visibility 1-2m Moderate tidal current

Table A 7 Waypoints and compass directions for transects for monitoring sites.

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

Table A 8 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@10m rod
					4	150
					5	160, start shoreside PM
	21.4856	**149.90907**	**5**	**1**	**1**	310, 330@10m rod
	Waypoint between transects 3 & 4				2	300 over large Lobophyllia
					3	320, ends short of large
					4	130, 120@10m rod
					5	150, 200@10m rod
	21.48313	**149.90868**	**2**	**2**	**1**	310
	Waypoint between transects 3 & 4				2	320
					3	320, 300@10m rod
					4	120
					5	150
	21.48317	**149.90845**	**5**	**2**	**1**	0, 350@10m rod
	Waypoint between transects 3 & 4				2	300, 320@10m rod
					3	320, 310@10m rod
					4	180, 150@10m rod
					5	180
Temple Island	**21.59608**	**149.50102**	**1**	**1**	**1**	200, 170@10m
	Waypoint between T1-T4				2	150, 180@10m
					3	190
					4	350
					5	330, 310@10m
	21.60285	**149.49932**	**1**	**2**	**1**	240, 220@10m
	Waypoint between T1-T4				2	190, 200@10m
					3	180, 190@10m
					4	90, 30@10m, 340@12m, 300
					5	30, 50@10m
Aquila Island	**21.95682**	**149.58102**	**1**	**1**	**1**	190, 180@10m, 140 to T2
	Waypoint between T1-T4				2	140
					3	170
					4	320
					5	330, 310@10m
	21.96112	**149.58158**	**1**	**2**	**1**	120
	Waypoint between T1-T4				2	90
					3	110
					4	0
					5	30