Southern Inshore Zone: Coral Indicators for the 2024 Mackay-Whitsunday-Isaac Report Card

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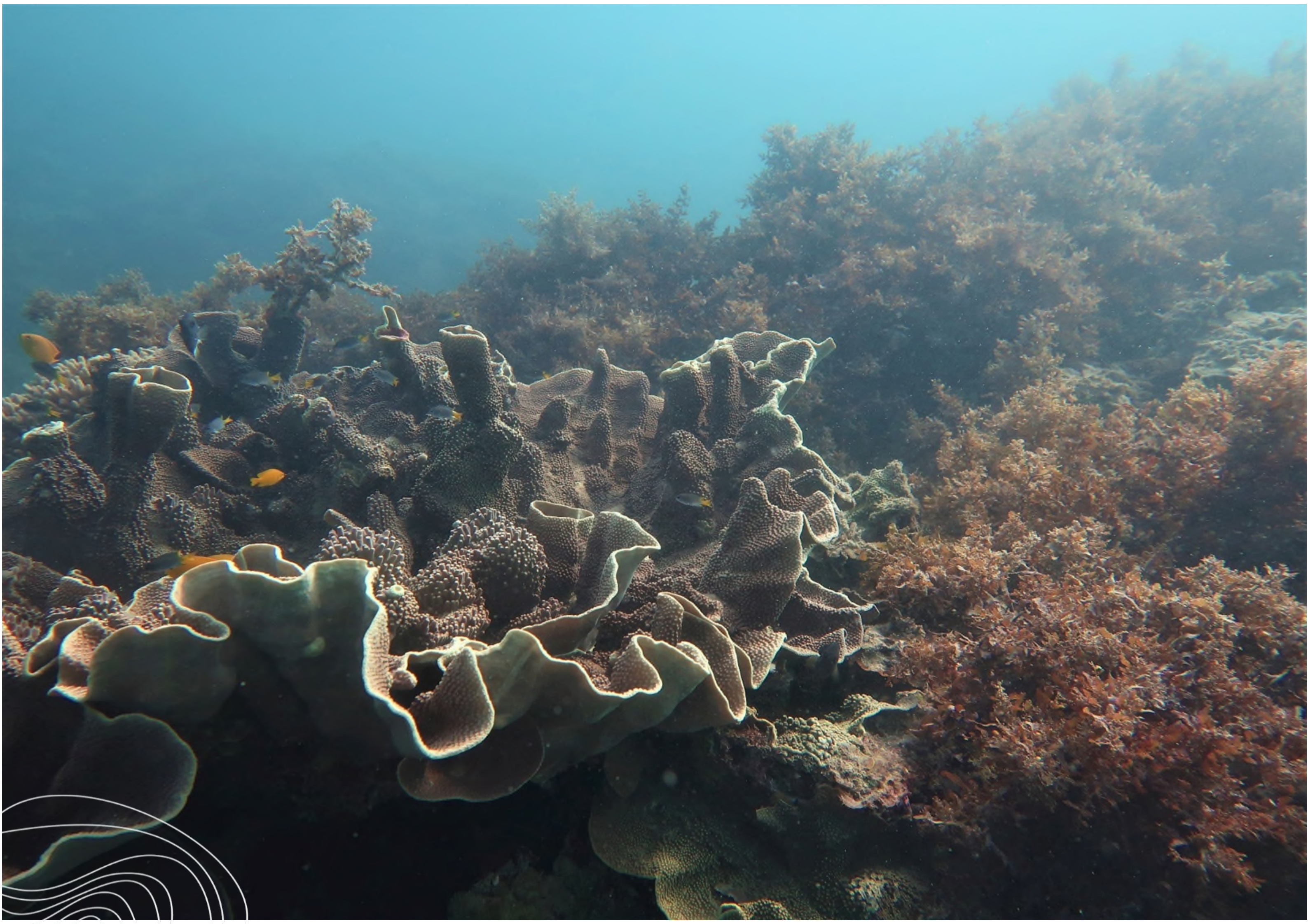

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Cover photo: Pine Islets reef slope showing large foliose *Turbinaria* surrounded by *Sargassum* in June 2023
Image: Cassandra Thompson

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1 EXECUTIVE SUMMARY

This report presents the results of monitoring undertaken in 2023 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 from 2023 form the basis of the coral indicator scores for the Southern Inshore Zone that inform the 2024 Mackay-Whitsunday-Isaac Report Card.

In June 2023, 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 2023 remained at D ('poor'), based on a Coral Index score of 0.22 (Figure 1).

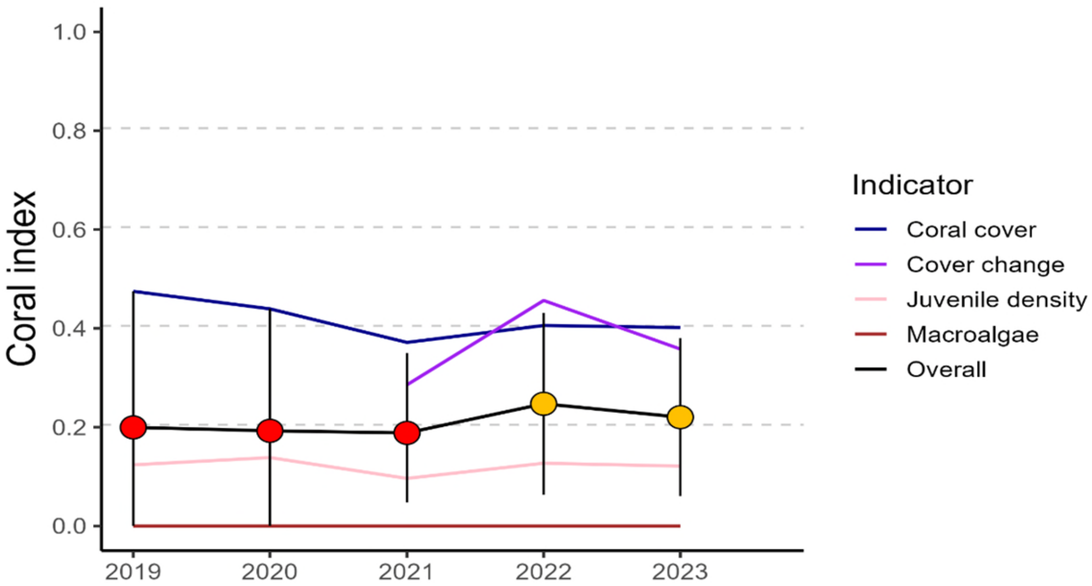

Figure 1 Coral Index and indicator scores. The Cover change indicator was added in 2022 and back calculated for 2021, as such the plotted score for the Coral Index of 0.19 in 2021 is slightly higher than the previously reported value of 0.16 that did not include the Cover change indicator.

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

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

The slight decline of the Coral Index from 0.25 in 2022 reflects the slight decline in Juvenile density indicator scores and the decline in the Cover change indicator from 'Satisfactory' to 'Poor'. In addition, there was no overall improvement to the Coral cover score, which has remained 'Poor' since 2021.

Over the 2022-2023 period there were no incidences of widespread disturbance to the region. Peak water temperatures were below those of 2022 and 2020 and there were no observations of bleaching.

There was no direct impact of river discharge on the coral communities and there were no cyclones that would have impacted the region.

Despite 2022-2023 being a period free of disturbance, there was no appreciable improvement in overall coral cover across the region. With the exception of Henderson Island, where recovery of the dominant *Acropora* coral continues, the post-bleaching trajectory of coral communities has plateaued at relatively low levels. Growth in both coral cover and juvenile density continues to be severely restricted by local environmental conditions that favour 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 (Thompson *et al*. 2020). 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 fifth 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 2023 that expands on the necessarily succinct summary of overall condition presented by the 2024 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.

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 on the day 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. We suggest doing the same 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 7.

All reefs were monitored in June 2023 (Table 1).

Table 1 Dates of coral monitoring.

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

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 ~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.* 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.* 2022), the development of which is 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*. 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 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.* 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:

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

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

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

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

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

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

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

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

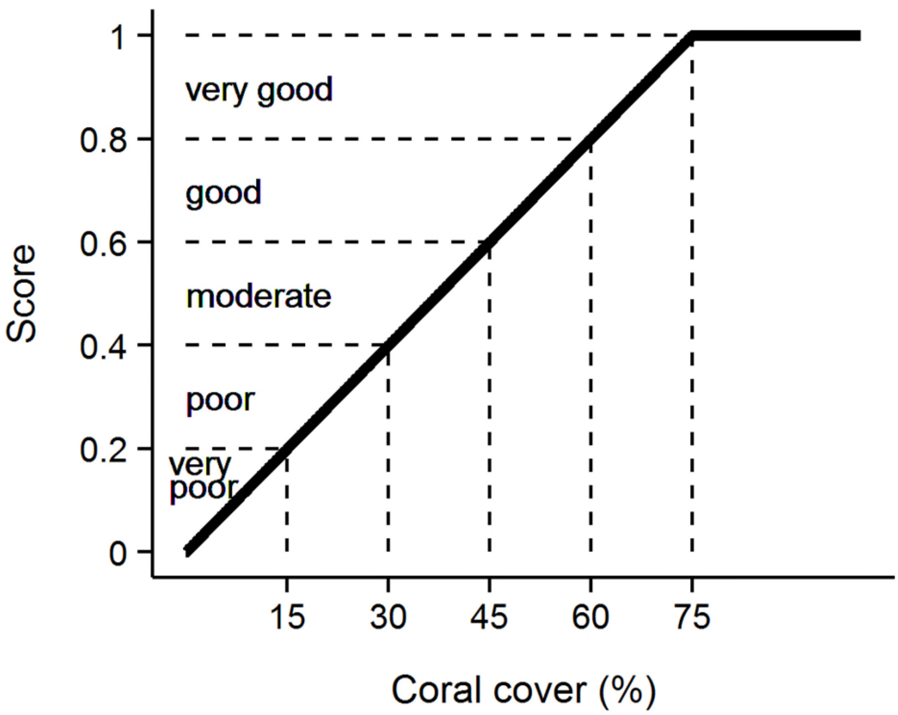

Figure 3 An example of a scoring diagram, here for the Coral Cover metric. Numeric scores and associated condition classifications based on observed coral cover are presented (see also Table 2).

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 *Monitpora*. Photos representative of coral communities at each reef and depth in 2023 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 (2 m and 5 m), Henderson Island (2 m and 5 m), and Aquila Island (1 m). These loggers were retrieved during our resurveys in 2020, 2021, 2022, and 2023. 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.* 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

Over the 2022-23 period in-situ temperature records showed temperature at the monitored sites peaked in March but remained below those recorded during 2022 and the marine heatwave in 2020 (Figure 4).

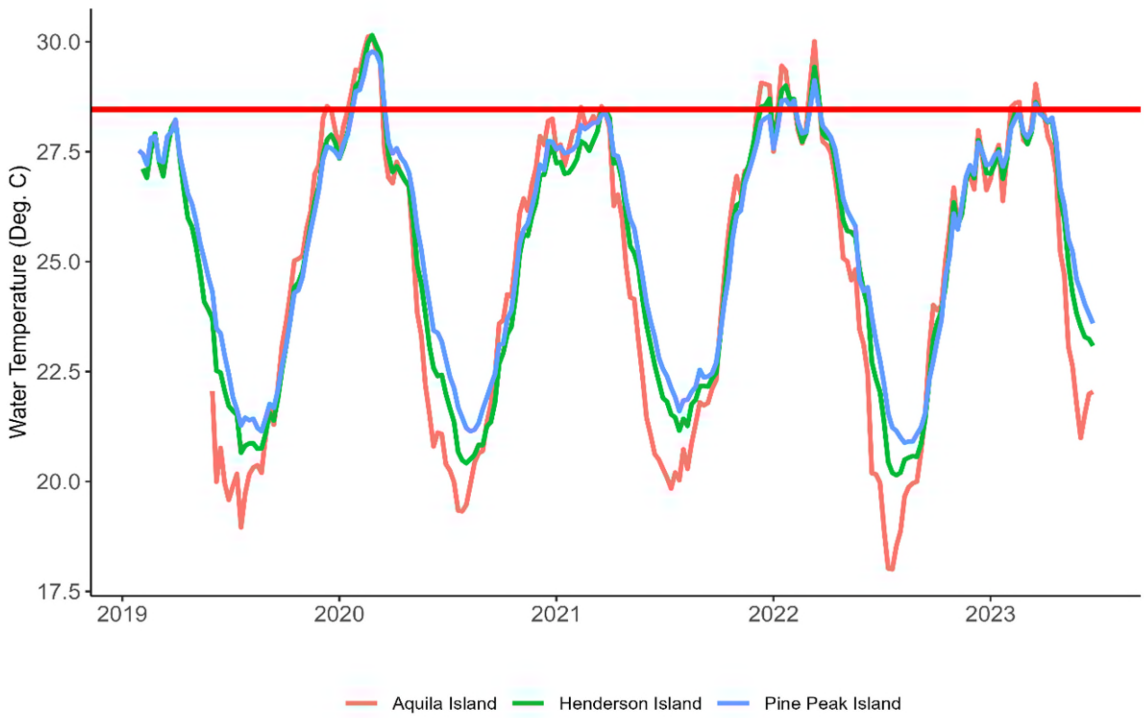

Figure 4 Temperature profiles recorded by in-situ loggers. The horizontal reference line was derived from the mean of the means of the hottest month each year observed over timeseries of in-situ temperature data available from reefs in Whitsunday Islands. This baseline excluded years in which bleaching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

The observed temperatures in 2023 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) that show levels of heat stress in 2023 were low compared with those in 2022 and, in particular, 2020 (Figure 5). 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).

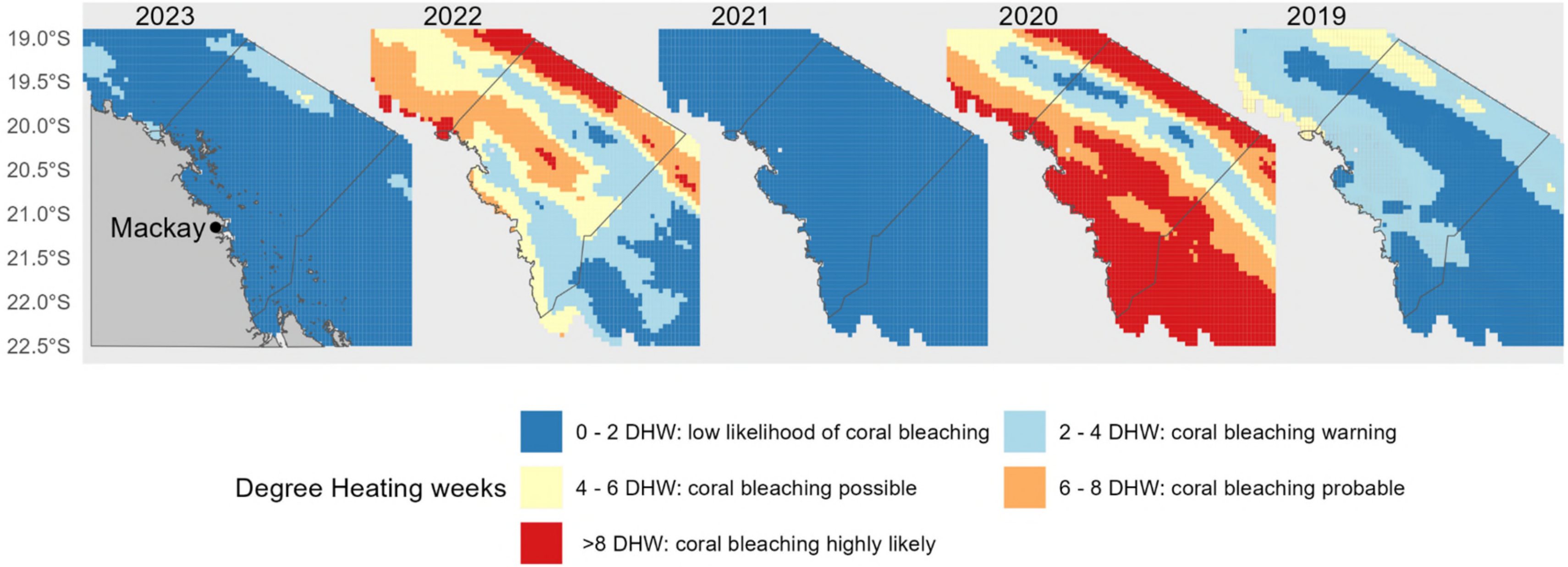

Figure 5 Annual estimates of thermal stress to corals. Data are the annual maximum degree heating week (DHW) estimates for each ~25 km^2 pixel. Data were sourced from NOAA coral reef watch. DHW values as indicators of thermal stress on the Great Barrier Reef are interpreted as follows: DHW values from 0 - 2: low likelihood of bleaching (i.e., normal summer conditions), 2 - 4: coral bleaching warning, 4 - 6: coral bleaching possible, 6 - 8: coral bleaching probable, >8: coral bleaching highly likely (after Cantin *et al.* 2021)

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 2022-2023 water-year (October to September) increased from 1.2 and 1.4 times median levels in the Pioneer and Plane basins respectively to 1.5 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; 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 (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
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
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
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

4.1.3 Cyclones and Storms

There were no cyclones likely to have impacted reefs in the Southern Inshore Zone during the 2022-2023 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.

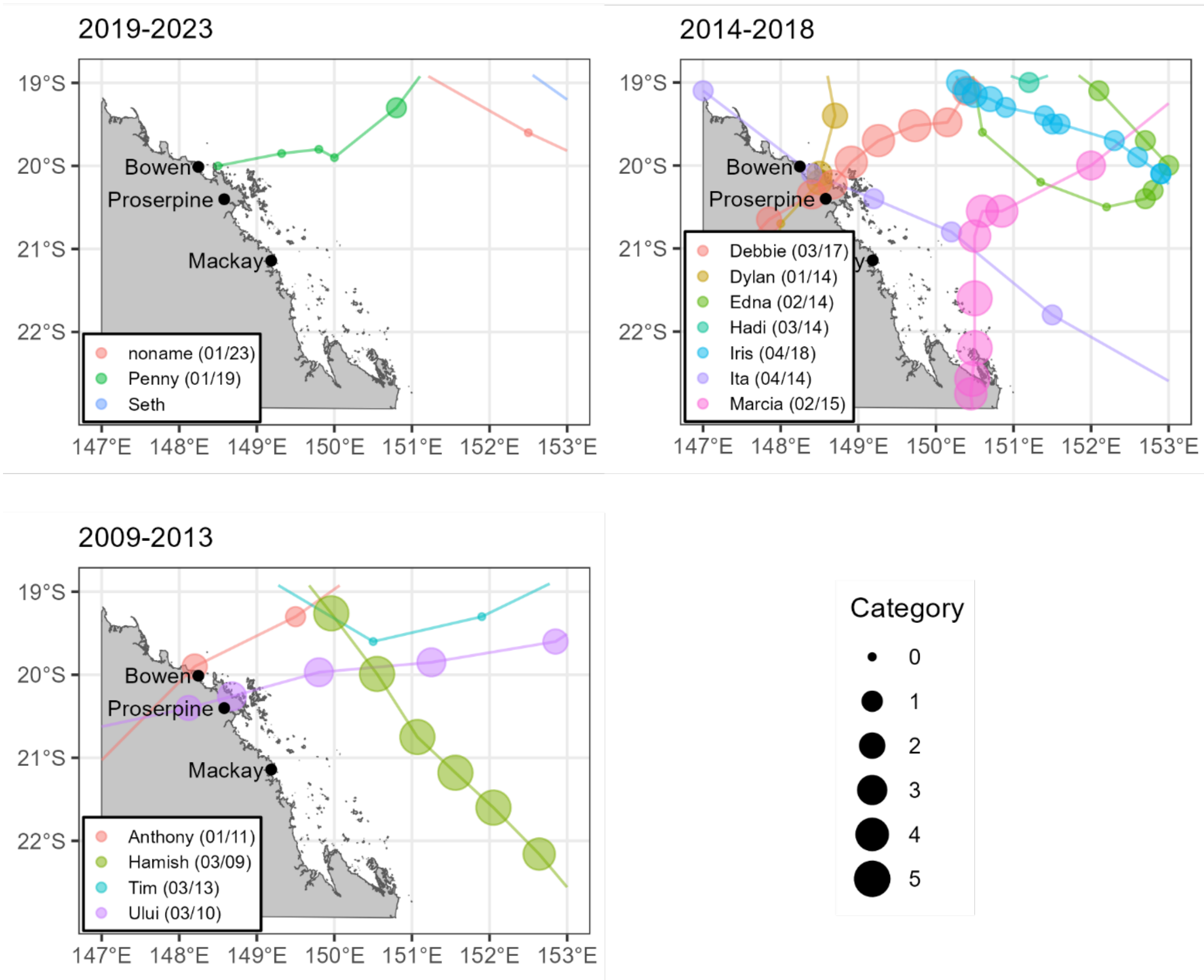

Figure 6 Tracks of tropical cyclones passing through the region. All cyclones crossing through the Mackay Whitsunday Isaac regional report card reporting area over the last 15 years are displayed. Tracks sourced from the Bureau of Meteorology

4.1.4 Biological Damage

A total of 10 colonies were identified with disease across four of the five reefs. Diseased colonies ranged from fragile branching *Acropora* to heavy foliose *Turbinaria* (Table A 5). In addition, there were a total of 15 colonies for which recent mortality was unknown. In combination, these 25 colonies spanned five genera, similar to last year's observations (26 colonies and 4 genera) and substantially lower than 2021 (31 colonies, 10 genera) and 2019 (55 colonies across 12 genera) (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 (13 *Acropora* corals). Brown band and white syndrome diseases were represented at both depths.

The number of colonies being overrun by the encrusting sponge *Cliona orientalis* have dropped from six colonies and four genera in 2022 to four colonies and two genera in 2023. Observations of *C. orientalis* in 2023 were confined to shallow water at Henderson and Temples islands. Afflicted genera were *Cyphastrea* and *Turbinaria* (Table A 5).

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 2023 there were no *Drupella* observed at Henderson Island or at the other locations in the study.

There were no bleached or partially bleached corals recorded by photo-transects or by the wider survey of the scuba-search (Table A 5), re-affirming the conclusion that, compared with the marine heat waves of 2020 and 2022, the summer temperatures of 2023 remained within the tolerance limits of coral communities at all five locations.

With no significant storms over the 2022-2023 season, there were no observations of recent physical damage at any of the visited reefs (Table A 5).

4.2 Coral Community Condition Assessment

The overall Coral Index score for the Southern Inshore Zone in 2023 was graded as D, categorising the coral communities as being in 'poor' condition (Table 5). While the Report Card category remains unchanged from 2022, the Report Card score has declined due primarily to a decline in the Cover change indicator that was reduced from a category of 'C' in 2022 to 'D' in 2023 (Table 5).

Across the reporting zone the mean cover of hard and soft corals remained at 30% (Table 6). There was a very slight decline in the density of juveniles and the score for Juvenile density remains 'very poor' (Table 5, Table 6). The proportion of macroalgae across the region has increased to 70%, the highest level recorded (Table 6) and for all reefs the score for this indicator remains zero (Table 5).

Table 5 Coral Index and indicator scores for 2023. 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

* 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 m^2)		Coral cover (%)		Macroalgae proportion (%)	
		Mean	SD	Mean	SD	Mean	SD
Zone summary	2019	1.42	0.96	35.6	23.8	65.8	20.9
	2020	1.59	0.85	33.0	20.1	60.5	25.4
	2021	1.11	0.86	27.9	14.3	65.8	19.9
	2022	1.52	1.15	30.4	17.1	58.9	20.4
	2023	1.39	1.08	30.1	21.9	69.7	19.4

The overall Index score continues to mask the substantial differences in the condition of coral communities between reefs (Table 7). At Henderson Island Index scores increased in 2023 and at 5 m depth this increase was sufficient to elevate the coral community grade to ‘C’. The Index score also increased at Temple Island where the grade improved from ‘E’ to ‘D’. At all other locations Index scores declined in 2023 and grades remained either ‘E’ or ‘D’.

Table 7 Index grade and scores for each reef and depth combination. Comparison of Index figures from 2019 to 2023. * 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	Grade
Pine Peak Island	2	0.05*	0.09*	0.08*	0.14	0.09	E
	5	0.12*	0.14*	0.12*	0.36	0.23	D
Pine Islets	2	0.04*	0.06*	0.06*	0.24	0.16	E
	5	0.12*	0.20*	0.15*	0.26	0.17	E
Henderson Island	2	0.41*	0.34*	0.19*	0.27	0.32	D
	5	0.36*	0.33*	0.28*	0.36	0.45	C
Temple Island	1	0.32*	0.21*	0.23	0.18	0.22	D
Aquila Island	1	0.19*	0.16*	0.14	0.14	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 declined at six of the eight reef-depth locations in 2023 (Table 8, Figure 7a). These declines were led by decreases in hard coral cover observed at five of the eight reef-depth locations. Changes in soft coral cover between surveys have been minor and variable (Table 8).

Across the region mean hard coral cover was 19.4%, remaining similar to the 19.3% observed in 2022 (Table 8). Henderson Island was a notable exception where hard coral cover increased to 37% at 2 m depth and 53% at 5 m depth (Table 8). These increases were principally due to increased cover of *Acropora,* the most common genus at this reef (Table A 1, Figure A 3). There was also an increase in soft coral cover (Table 8), particularly *Klyxum* and *Sinularia*, the latter more than doubling in cover at 2 m (Table A 2). In combination these increases have promoted the Coral cover score for Henderson into the ‘good’ category at 2 m and maintained the ‘very good’ category at 5 m (Table 8). The only other location to have recorded an increase in hard coral cover was Temple Island (Table 8), however, this was offset by a reduction in soft coral cover, and in particular the genus *Sinularia* (Table 8, Table A 2). For soft corals, there was a very minor increase observed at 5 m depth at Pine Peak Island.

At all other reefs across the region the combined cover of hard and soft corals declined resulting in reduced Coral cover indicator scores (Table 8, Figure 7a). The largest decline occurred at Aquila Island where reduced cover of the hard coral genus *Montipora* and soft coral genus *Xenia* were influential. A notable decline was also recorded at 2 m depth at Pine Islets where the cover of *Montipora* and the soft corals *Sinularia*, and *Briareum* was reduced.

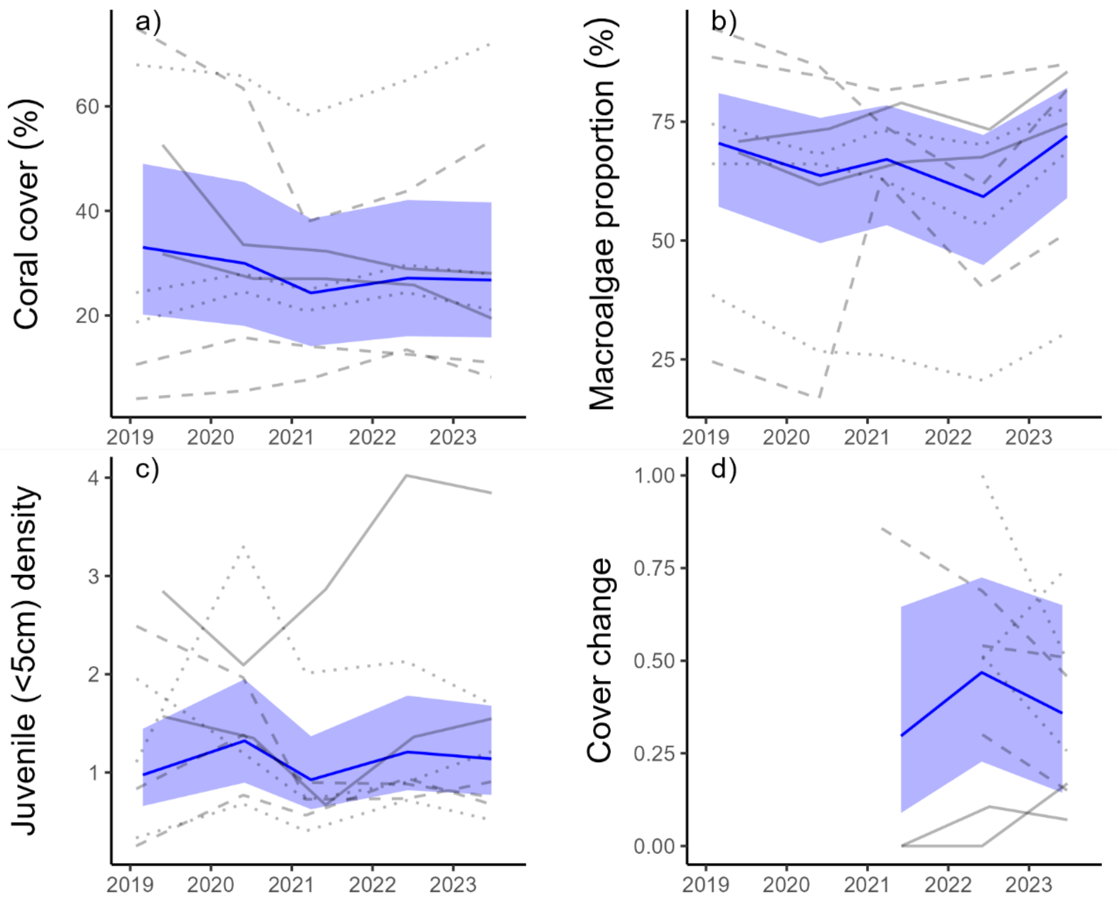

Figure 7 Indicator trends for Southern Inshore Zone. Blue lines represent trends in: a) coral cover, b) macroalgae proportion of total algae cover, c) juvenile density per m^2 of available substrate, d) cover change. Trends are bound by 95% confidence intervals of those trends (shading), grey lines represent 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 2023 and 2022. 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	2022	4.4	8.2	12.6	0.17
		2023	4.1	6.9	11	0.15
	5	2022	12.3	17.3	29.6	0.4
		2023	10.3	17.6	27.9	0.37
Pine Islets	2	2022	9.5	3.9	13.4	0.18
		2023	5.8	2.4	8.2	0.11
	5	2022	16.4	7.9	24.3	0.32
		2023	15.2	5.9	21.1	0.28
Henderson Island	2	2022	31.9	11.9	43.8	0.58
		2023	37.1	16.3	53.4	0.71
	5	2022	47.7	17.3	65.0	0.87
		2023	52.9	19.1	72.0	0.96
Temple Island	1	2022	14.9	14.0	28.9	0.39
		2023	17.1	10.9	28.1	0.37
Aquila Island	1	2022	17.0	8.8	25.8	0.34
		2023	13.0	6.4	19.4	0.26

4.5 Macroalgae Proportion

Macroalgae continue to dominate algal communities. The proportion of algae classified as macroalgae has increased at all reefs in this region (Figure 7b, Table 9), and continues to exceed thresholds deemed to result in negative impacts to coral community resilience (Table 2). The grade of 'E' ('very poor', Table 9) continues for this indicator. The mean proportional macroalgae cover for the region in 2023 was 69.7%, the highest in the series of annual surveys by this program (Table 6, above Figure 7b). This contrasts with the 2022 figure of 58.9%, the lowest in the series.

Changes in cover were variable among macroalgae groups for different reef communities (Figure A 2):

- At Pine Islets (2 m) the brown macroalgae *Sargassum* more than doubled cover from 30% (Table A 3 Davidson *et al.,* 2022) to 62% (Table A 3). This was countered by declines in red macroalgae and the brown macroalgae *Lobophora*, resulting in an increase in overall macroalgae cover from 48% to 70% (representing a rise in Macroalgae proportion from 62% to 82%, Figure 7 b).
- A similar pattern was observed at Pine Islets (5 m) where the overall increase in macroalgae cover from 35% to 50%.
- *Lobophora* had gains at Pine Peak Island (2 m, 7% to 10%; 5 m, 25% to 29%) but only minor increases at Henderson Island.
- Henderson Island has a particularly depauperate macroalgae taxa, with *Lobophora* the only taxa with cover greater than 1%. 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, Temple Island and at 2 m depth at Pine Peak (Table A 3), where cover increased relative to that observed in 2022 (Table A 3 Davidson *et al.* 2022).

Table 9 Macroalgae cover and indicator scores for each location. Comparison of 2023 and 2022 data. Scores are coloured as per Table 3.

Reef	Depth	Year	Macroalgae cover (%)	Macroalgae proportion (%)	Macroalgae score
Pine Peak Island	2 m	2022	70.9	84.4	0
		2023	75.7	87.1	0
	5 m	2022	44.4	70.1	0
		2023	53.4	78.0	0
Pine Islets	2 m	2022	48.1	61.7	0
		2023	70.4	81.7	0
	5 m	2022	34.6	53.3	0
		2023	49.5	68.6	0
Henderson Island	2 m	2022	21.9	40.3	0
		2023	23.4	51.6	0
	5 m	2022	5.4	20.6	0
		2023	6.6	30.6	0
Temple Island	1 m	2022	39.7	67.6	0
		2023	49.8	74.6	0
Aquila Island	1 m	2022	39.3	73.4	0
		2023	51.7	85.5	0

4.6 Juvenile Density

The density of juvenile corals across the region continues to be low at all reefs, and the category for the Juvenile score has been 'poor' or 'very poor' through the five years of this monitoring program (Table 5, Table 6). In 2023 the abundance of juvenile hard corals recorded declined at Pine Peak Island and Pine Islets, showed a very marginal increase at Henderson Island and more substantial increases at Temple Island and Aquila Island (Table 10, Figure A 2). These changes in abundance of juveniles are generally reflected in the changes to Juvenile indicator scores (Table 10). The exception being at Temple Island where the increased number of juvenile corals was offset by the increased cover of algae resulting in a slight decline the density of juvenile corals that are the basis of the indicator score (Table 10). At the regional level, the overall density of juvenile corals corrected for area of transects occupied by algae declined (Figure 7c, Table 6) leading to a decline in the overall Juvenile indicator score (Table 5).

Only Temple Island has maintained a Juvenile density in the 'poor' category for the past three years due to an abundance of juvenile *Pocillopora* (family Pocilloporidae) and *Turbinaria* (family Dendrophylliidae) corals (Figure A 2). In 2023 the abundance of juvenile *Pocillopora* decreased by 40%, however, a fourfold increase in *Acropora* abundance (from 9 to 36 juveniles), together with moderate increases in *Favites, Cyphastrea* (family Merulinidae), and *Porites* (Table A 4, Figure A 2), resulted in a total of 171 juveniles at Temple Island, the highest abundance recorded during this survey

Table 10 Juvenile hard coral abundance, density and indicator scores for each location. Comparison of 2023 and 2022 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 m²)	Juvenile score
Pine Peak Island	2 m	2022	54	0.94	0.08
		2023	40	0.68	0.06
	5 m	2022	31	0.72	0.06
		2023	24	0.52	0.04
Pine Islets	2 m	2022	48	0.88	0.08
		2023	45	0.75	0.07
	5 m	2022	93	2.13	0.19
		2023	83	1.70	0.15
Henderson Island	2 m	2022	27	0.73	0.06
		2023	28	0.91	0.08
	5 m	2022	17	0.90	0.08
		2023	18	1.21	0.11
Temple Island	1 m	2022	149	4.02	0.35
		2023	171	3.84	0.33
Aquila Island	1 m	2022	49	1.36	0.12
		2023	63	1.55	0.13

4.7 Cover change indicator

The Cover change indicator score for 2023 was 0.36, a decline from 2022 that transitions the category boundary from 'satisfactory' to 'poor' (Table 5). This result is driven by declines in hard coral cover at most reefs between 2022 and 2023 (Table 11). A plot of the Cover change indicator scores in 2023 shows the distinct decline from 2022 levels, though the large error bars indicate large variation among sites (Figure 7d).

As no acute disturbance events impacted the area between 2022 and 2023 surveys, coral communities should be in a state of recovery with coral cover expected to increase. When hard coral cover declines between observations during a recovery period, the annual Cover change score for the reef is zero. Including these zeros for 2023 into the rolling mean over the period for which 2023 Cover change scores are estimated resulted in scores transitioning the boundary between categories to the next lower level, or remaining 'very poor' at: Aquila, Pine Peak and Pine Islets (Table 11).

At Henderson Island the Cover change score remained 'moderate' at 2 m and rose to 'good' at 5 m (Table 11), driven by continued recovery in the cover of *Acropora* corals (Figure A 2). At Temple Island an increase in hard coral cover in 2023 lifted the Cover change score but it remains in the 'very poor' category due to declines in previous years (Table 11).

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
	5 m	2022	2022		4.8	1
		2023	2022-2023		-2	0.5
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
	5 m	2022	2022	2.7	2.7	0.51
		2023	2022-2023	1.4	-1.2	0.26
Henderson Island	2 m	2022	2022	6.9	6.9	0.54
		2023	2022-2023	12.1	5.2	0.51
	5 m	2022	2022	3.4	3.4	0.5
		2023	2022-2023	8.6	5.2	0.75
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
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

5 DISCUSSION

The overall condition of Southern Inshore Zone reefs in 2023 was categorised as 'poor' and graded 'D' based on a Coral Index score of 0.22 and indicates a slight decline from 2022. Contributing to the decline were reduced scores for the Cover change and to a lesser degree the Juvenile coral indicators. Both these reductions and the continued 'poor' condition of coral communities across the region are influenced by the high prevalence of macroalgae amongst the algal communities.

On coral reefs, macroalgae compete with corals and in so doing reduce the resilience of coral communities. There are a number of pathways by which this competition occurs; from simply limiting the space or light available to corals (Tanner 1995, Hauri *et al.* 2010), physically damaging corals via abrasion (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). Low Coral Index scores are strongly influenced by high levels of macroalgae at all reefs that ensure the scores for the Macroalgae indicator have remained at the minimum value of zero across the region. The lower threshold for the Macroalgae indicator (above which macroalgae scores are set to zero) was derived based on observed relationships between macroalgae, coral cover, the density of juvenile corals and the rate of recovery of coral cover at other inshore reefs (Thompson *et al.* 2020). The generally poor or very poor scores for these indicators observed in this region since 2019 add support to the inference that persistently high levels of macroalgae are limiting the resilience of these reefs.

What is not obvious in the Macroalgae indicator results is that macroalgae cover increased in 2023 and the ramifications for these increases on the interpretation of changes in the other indicators. The Cover change indicator compares the observed cover of hard corals from one year to the next to a modelled expectation of this change. The expected change in coral cover from one year to the next is often small, especially when coral cover is low and the coral communities are dominated by slower growing taxa, as they are at most reefs in this region. It is an unavoidable artifact of the sampling methods that marked changes in the cover of macroalgae can bias estimates of hard coral cover as the algae variably obscure or reveal underlying corals. Given the likelihood that observed coral cover in 2023 will have been biased toward lower estimates compared to those from 2022 due to the increase in macroalgae cover, we caution against over-interpretation of the decline in Cover change scores observed at Pine Peak Island, Pine Islets and Aquila Island in 2023.

In contrast, Cover changes scores at Henderson Island remained moderate (2 m) or increased to good (5 m) as the coral community continues to recover towards pre-bleaching levels (Figure 7, Figure A 2) at or above rates expected for inshore reefs. Recovery is driven by the rapid growth of *Acropora* colonies. Increase in hard coral at Temple Island in 2023 was a positive sign and while the Cover change score remained in the 'very poor' category, this was due to scores of zero in previous years weighing heavily on the rolling mean over up to four years on which this indicator is based.

Similarly, increased cover of macroalgae will result in an increase in the total cover of algae that is used to adjust counts of juvenile hard corals based on the area of the transects deemed 'available' to coral recruitment. The intent of adjusting the transect area to the proportion occupied by algae is that under ideal conditions these algae communities would be dominated by short turf and coralline algae that offer attractive substrates for coral recruitment. Increased cover of macroalgae canopy will not only have projected over substrates occupied by other algae, but also portions of the substrate occupied by other organisms such as corals and sponges as well as some areas of silt and sand. The result being that increased macroalgae cover in 2023 will have had a negative influence on the Juvenile indicator score due to both real limitations to coral recruitment processes and as an artifact of overestimating the area of 'available substrate' used to convert counts of juveniles to the densities on which the score is based.

Across the region there were no appreciable improvements in juvenile abundance. Gains made at the inshore reefs of Temple Island and Aquila Island are tempered by the decline at other reefs. An example to note is the unusually large number of *Acropora* juveniles observed at Temple Island in 2023. 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. This reef was also the recipient of increased numbers of *Cyphastrea* and *Favites* juveniles. Interestingly, Temple Island also supports a consistently large number of the resilient *Turbinaria* juveniles and can boast the largest total number of juveniles observed in this study (171).

Juvenile abundance remains highly variable across the region. It is highly likely that coral replenishment in the region is influenced by interactions between physical factors (such as high tidal driven currents) and biological factors (such as the abundance of macroalgae, or encrusting benthic invertebrates other than corals), along with regional population sizes of coral broodstock, that combine to limit the supply or successful settlement of coral larvae. That Henderson Island continues to record very few juvenile *Acropora* corals compared to Temple Island despite *Acropora* cover being 10-fold greater demonstrates the variable connectivity to broodstock among these reefs.

Finally, in 2023 there was no appreciable improvement in overall coral cover across the region. Ongoing increase in Coral cover scores at Henderson Island where improved cover of both hard and soft corals has occurred contrast the declines at all other reefs. Again, slight declines in coral cover scores will be confounded by the bias associated with increased cover of macroalgae. Where the canopy of macroalgae was dense the hard coral communities were comprised of encrusting and massive genera such as *Porites* (Pine Peak Island), and *Montipora* (Pine Islets) that would be underestimated due to over topping by macroalgae. However, at Pine Peak Island (5 m), where the lower substrate-hugging *Lobophora* is the most abundant brown macroalgae, coral cover also declined, suggesting observed declines were not only due to observation bias.

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. The environmental conditions of large tidal range, inshore location, and low secchi depth (a proxy for high turbidity) match those described by Fabricius *et al.* 2023 as supporting sustained abundance of macroalgae cover. Sustained high densities of macroalgae, particularly the canopy-forming *Sargassum*, are likely to be inhibiting coral growth (Clements *et al.* 2018) and thus reinforcing their dominance on reefs in the region.

Results from the 2023 survey suggest that across the region the growth and replenishment of coral communities continues to be shaped by abundant macroalgae. In the span of this study, low coral cover continues to be the normal state. Slow growth and replenishment make the survivorship of these ecologically isolated coral communities vulnerable to increased frequency of disturbance such as marine heatwaves, cyclones, and flood plumes. Henderson Island, clear of abundant macroalgae, is the only reef demonstrating continued recovery from the 2020 bleaching event and has the only population of *Acropora* corals potentially capable of supplying propagules to local reefs. The conclusion from the 2023 survey is that growth and replenishment of corals in the region has plateaued, and improvement will depend on a positive shift in the balance between coral cover and macroalgae cover.

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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*	*Alveopora*	*Galaxea*	*Goniopora*	*Lobophyllia*	*Merulina*	*Montipora*	*Platygyra*	*Porites*	*Turbinaria*	Other HC	Genus Richness
Pine Peak Island	2	0.06	0	0	0.06	0.06	0.06	0.81	0.06	2.5	0	0.44	12
	5	0.44	0.06	0	0.5	0.25	0.31	0.31	0.38	5.31	0	2.75	19
Pine Islets	2	0.38	0	0	0.06	0	0	2.25	0.25	0.69	1.44	0.75	10
	5	0.19	0	0.12	1.25	0.44	1.19	5.5	0.19	1.62	1.75	2.94	22
Henderson Island	2	33.69	0	0.19	0.38	0.19	0.31	0.69	0.12	0.62	0.06	0.81	14
	5	40.19	1.25	2	0.31	3.88	0	2.31	0.06	0	0.38	2.56	20
Temple Island	1	3.88	0.25	0	0.19	0	0	6	1.94	0.19	2.12	2.56	15
Aquila Island	1	0	0.12	0	0.12	0.06	0	10.81	0	0.19	0.06	1.62	13

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*	*Cladiella*	*Klyxum*	*Lobophyton*	*Sarcophyton*	*Sinularia*	Other SC
Pine Peak Island	2	3.38	0.19	0	1	0.31	1.62	0.44
	5	12.56	0.06	0.62	1.31	0.44	1.94	0.62
Pine Islets	2	0.25	0.06	0.06	1.94	0	0.06	0
	5	2.5	0	1.31	0.62	0.5	0.94	0
Henderson Island	2	1.19	0.88	7.31	0.38	2	4.56	0
	5	1	0.5	11.56	0.31	3.69	1.75	0.25
Temple Island	1	2.81	1.25	0	0.31	0.06	6.5	0
Aquila Island	1	0.06	0.69	0.62	1.06	0.56	3.12	0.31

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 brown macroalgae are grouped as ‘Other’algae are grouped.

Reef	Depth	Brown macroalgae				Red macroalgae	Green macroalgae	Turf algae	Coralline algae
		Lobophora	*Sargassum*	Stypopodium	Other				
Pine Peak Island	2	10.38	55.25	1.38	0.38	8.31	0	7.56	3.62
	5	28.88	20.12	1	0.69	2.31	0.38	11.81	3.25
Pine Islets	2	3.69	61.62	0.5	1.31	3.25	0	12.94	2.88
	5	9.94	36.62	1.12	0.31	1.38	0.12	20	2.69
Henderson Island	2	22.88	0.06	0	0	0.44	0.06	21.81	0.19
	5	6.25	0.06	0	0	0.12	0.19	14.81	0.19
Temple Island	1	3.12	32.69	0.06	4.81	8.94	0.12	15.75	1.19
Aquila Island	1	1.44	30.69	0	5.75	13.38	0.44	8.06	0.69

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	*Acropora*	*Alveopora*	*Cyphastrea*	*Dipsastraea*	*Duncansomnia*	*Favites*	*Goniopora*	*Homophyllia*	*Lobophyllia*	*Montipora*	*Moseleya*	*Paragoniastrea*	*Platygyra*	*Pocillopora*	*Porites*	*Psammocora*	*Turbinaria*	Other genera	Genus Richness	Number
Pine Peak Island	2	2	0	0	0	0	2	0	0	4	0	1	0	0	10	13	3	1	4	11	40
	5	0	0	0	0	0	2	2	0	2	1	0	0	0	6	6	1	0	4	10	24
Pine Islets	2	3	0	0	2	0	1	2	0	3	4	0	1	2	7	3	4	5	8	16	45
	5	5	3	2	3	0	5	3	4	5	8	1	4	5	3	8	4	6	14	26	83
Henderson Island	2	7	0	0	2	0	3	0	0	3	1	0	0	0	4	0	1	0	7	11	28
	5	3	0	0	0	0	3	0	0	2	3	0	0	1	1	0	0	0	5	11	18
Temple Island	1	36	0	12	6	4	14	1	0	3	7	2	1	2	16	11	0	52	4	18	171
Aquila Island	1	2	5	4	2	0	1	6	0	0	16	9	2	0	5	0	5	3	3	14	63

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 2023 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**
		2 m	**5 m**	**2 m**	**5 m**	**2 m**	**5 m**	**1 m**	**1 m**
Disease	*Acropora*		1			3	2		
	Montipora			1	1				1
	Turbinaria				1				
Unknown cause	*Acropora*					4	4	1	
	Favites							1	
	Montipora								1
	Pocillopora	1		1	1				1
Sponge - *Cliona orientalis*	*Cyphastrea*							1	
	Turbinaria					1		2	
Total number of Colonies		1	1	2	3	8	6	5	3
Bleaching (proportion of colonies)		0	0	0	0	0	0	0	0
Physical (proportion of colonies)		0	0	0	0	0	0	0	0

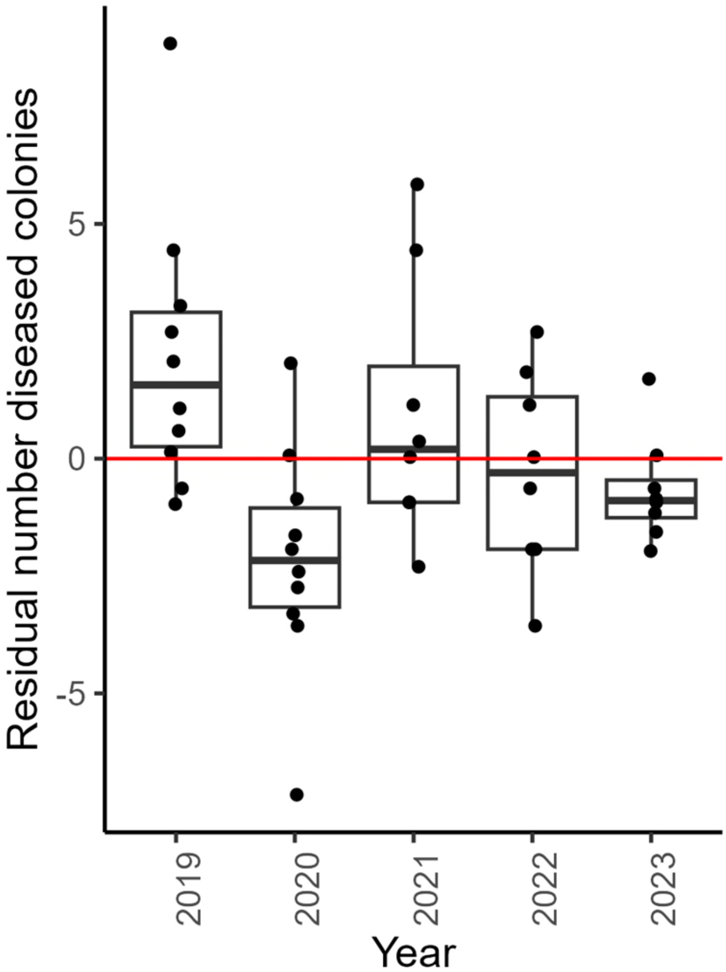

Figure A 1 Relative coral disease by year. Data are standardised to the reef and depth mean across years. Boxplots show the median (bold horizontal line), 25th to 75th quartiles (box), and 1.5 times the inter-quartile range (whiskers). Solid dots are the relative number of coral colonies suffering ongoing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

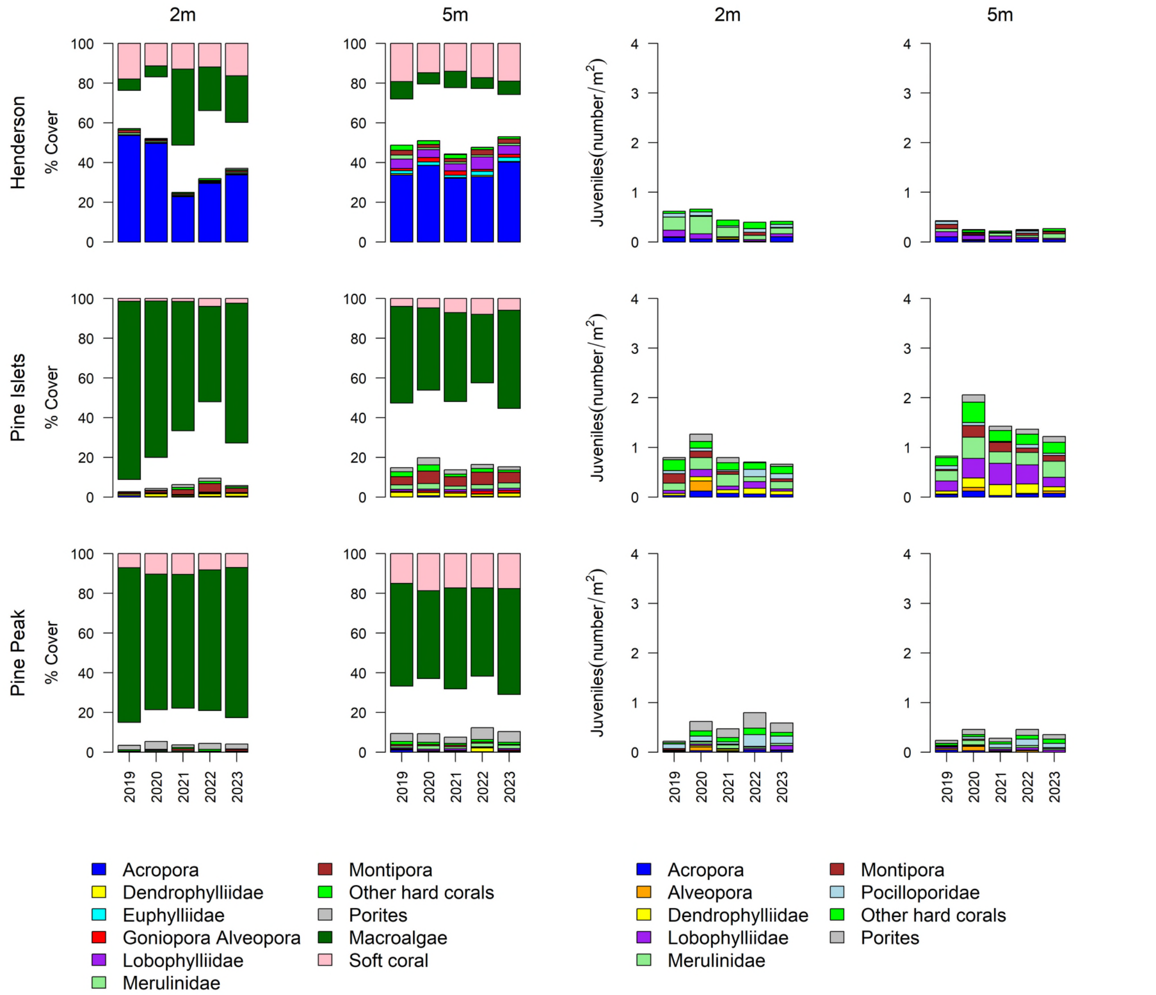

Figure A 2 Composition of benthic cover and hard coral juveniles. The left-hand plots show the breakdown of cover for hard coral families at 2 m and 5 m depths. Families that had a cover of at least 3% at either depth of any reef in the Zone are differentiated. Cover of all other families are grouped 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 m² of transect area by family at 2 m and 5 m depths.

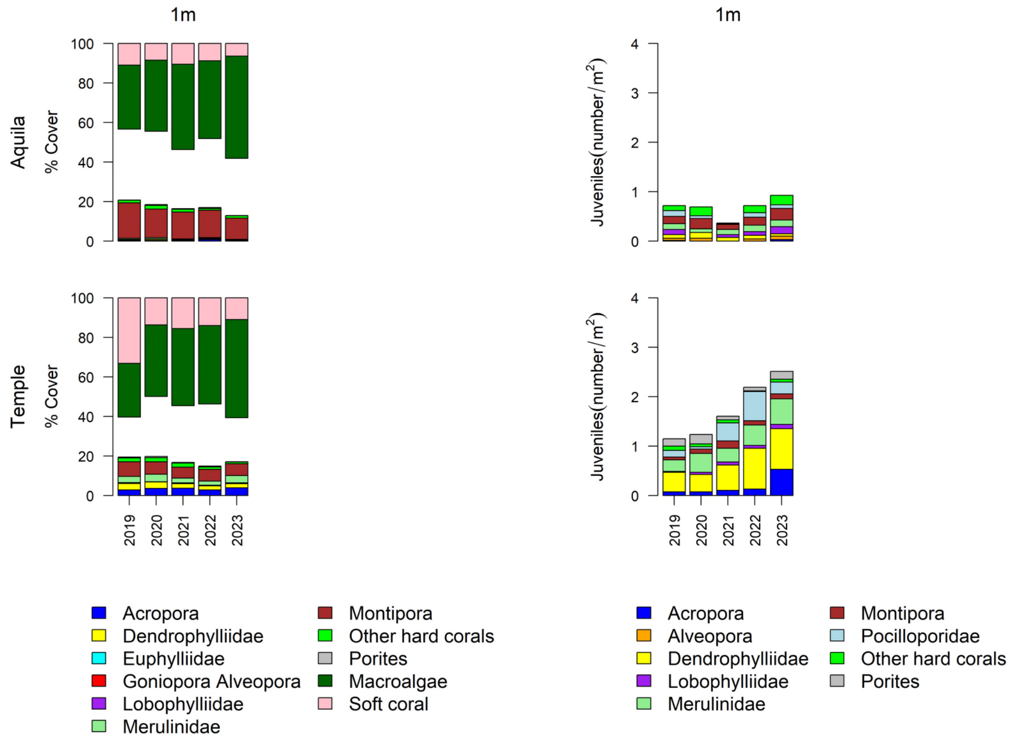

Figure A 2 continued, for the 1 m deep sites at Aquila and Temple Islands.

8.2 Images of benthic communities

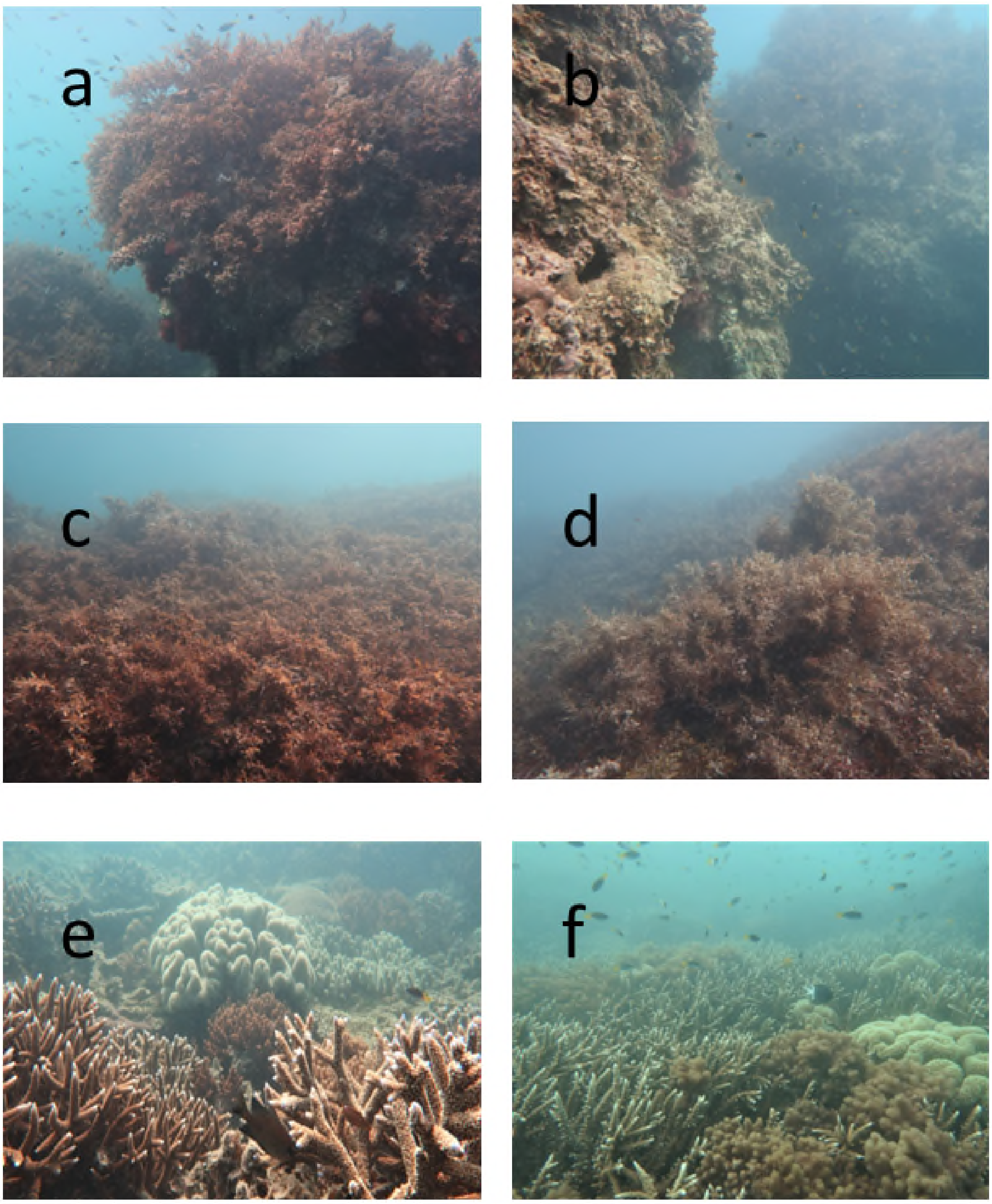

Figure A 3 Benthic community photos at outer reefs. Communities dominated by macroalgae at :a) Pine Peak Island 2 m, b) Pine Peak Island 5 m, c) Pine Islets 2 m, d) Pine Islets 5 m, contrast the fields of *Acropora* and soft corals at e) Henderson Island 2 m and f) Henderson Island 5 m.

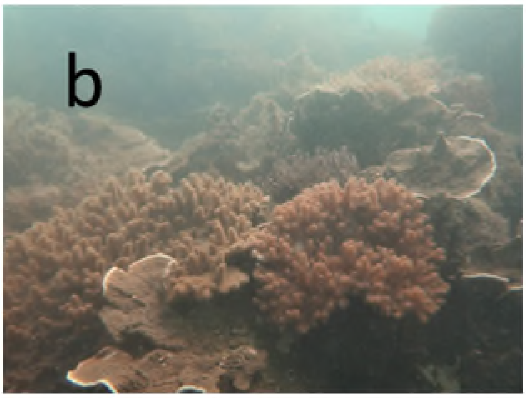

Figure A 4 Benthic community photos at inner reefs. a) mix of hard and soft corals among Sargassum macroalgae at Temple Island, b) colonies of encrusting *Montipora* among soft corals at Aquila Island.

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. 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 <15 knots are required. These reefs can be successfully resurveyed with winds in this range. The 2023 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. Table A 6 provides a reference point for the conditions experienced during 2023 re-surveys.

Table A 6 Weather conditions and tide heights experienced during 2023 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	22/6/2023	6 SE	Mid rise (3 m)	Visibility 5-6 m, no current
Pine Islets Site 1	22/6/2023	6 E	Either site of high (4.2 m)	Visibility 6 m , no current
Pine Islets Site 2	23/6/2023	8 N	Mid rise (3 m)	Visibility 8-9 m, no current
Pine Peak Island	23/6/2023	8-10 N	Either side of high (4 m)	Visibility 7-9 m, no current
Temple Island	24/6/2023	5 N	Mid rise (3.1 m)	Visibility 9 m, current began to increase as tide rose at site 2
Aquila Island	24/6/2023	0-5 NE	Rising to high (5.0 m)	Visibility 2 m, strong current as soon as top of tide reached at site 2

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@10 m rod
	Waypoint between transects 3 & 4				2	210, 120@10 m rod, 30@15 m
					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
	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
					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 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@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
					5	30, 50@10 m
Aquila Island	**21.95682**	**149.58102**	**1**	**1**	**1**	190, 180@10 m
	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