Southern Inshore Zone: Coral Indicators for the 2021 Mackay-Whitsunday-Isaac Report Card

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Cover photo: Henderson Island reef flat in March 2021, showing corals that survived high water temperatures in 2020 amongst others that were killed.
Image: Daniela Ceccarelli

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1 EXECUTIVE SUMMARY

This report presents the 2021 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 2021 Mackay-Whitsunday-Isaac Report Card.

Between March and June 2021, the Australian Institute of Marine Science (AIMS) resurveyed benthic communities at permanent coral monitoring locations at five reefs[1] in the Southern Inshore Zone.

The overall report card grade for coral community condition in 2021 was E ('very poor'), based on a coral index score of 0.16 (Table 1). This is a decline from the 2020 score of 0.19 and reflects a decrease in both the coral cover and juvenile density indicators. Most influential in this decrease was the reduction in coral cover following the bleaching event of 2020, the impact of which was still in progress during the previous 2020 survey. The 2021 survey provides an assessment of the surviving coral communities, with particularly large losses of hard coral cover observed at Henderson Island and loss of juvenile density at both Henderson Island and Pine Islets.

The report card grades are based on the assessment of three 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, and
- Juvenile corals - the density of juvenile hard corals

Prior to this year's surveys, relatively high coral cover at Henderson Island had kept the regional Coral cover indicator score in the mid-range ('Satisfactory'). The impact of high water temperatures in 2020, that caused coral bleaching and ensuant loss of coral cover, reduced the regional Coral cover indicator score into the ('Poor') range in 2021. The regional Coral Index score continues to be heavily influenced by the high cover of macroalgae found across all reefs with the score of 0 maintained for this indicator (Table 1). The ongoing presence of a high proportion of macroalgae at all reefs is likely to be limiting the replenishment of coral communities and contributing to the continued, 'very poor' grade for juvenile corals (Table 1).

Table 1. Coral index and indicator scores for 2021, with comparison for 2019 and 2020..
To facilitate this comparison, scores for Connor Island have been excluded from 2019 and 2020 calculations, resulting in an overall score drop for 2019 and 2020, and a grade drop for 2020 from 'D' to 'E'.

	Year	Juvenile corals score	Coral cover score	Macroalgae score	Report Card	
					Coral Index Score	Grade
Regional Scores	2019	0.12	0.47	0	0.20	E
	2020	0.14	0.44	0	0.19	E
	2021	0.10	0.37	0	0.16	E

[1] Note: Connor Island, which had been surveyed in 2019 and 2020, has been excluded from further monitoring due to the high risk of conditions being unsuitable for annual survey work.

No major disturbance events affected reefs in the Southern Inshore Zone between 2020 and 2021 surveys: there was no widespread bleaching as water temperatures remained within long-term averages, no cyclones entered the area, and there were no periods of extensive flood plumes.

However, as reported last year (Davidson *et al.* 2020), a high percentage of corals (67% at Henderson Island) were bleached during surveys in May 2020. It is highly likely that this ongoing stress contributed to the further loss of coral cover observed in 2021.

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 third 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 2021 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 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 1). A sixth site at Connor Island was included in 2019 and 2020 however proved extremely difficult to survey due to the very low under water visibility and it was agreed this site be removed from the program.

Figure 1. 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 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 1-1.5 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 1.

The more offshore reefs were monitored in March 2021, Aquila Island and Temple Island were monitored in June 2021 (Table 2).

Table 2. Dates of coral monitoring.

Island	**2019**	**2020**	**2021**
Pine Peak Island	27th January	26th May	6th March
Pine Islets	28th January	27th May	6-7th March
Henderson Island	29th January	25th -26th May	7th March
Temple Island	27th May	27th-28th May	3rd June
Aquila Island	27th May	12th July	3rd 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 50 cm 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 a curated 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., which result 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

This section provides an overview of the rationale for the selection of the three indicators used to assess coral community condition in 2021.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. Of the five indicators included in the Reef report card two require multiple annual observations for estimation and as such were not estimated here. The rate of coral cover change indicator requires at least three annual visits. The change in community composition indicator score is based on the deviation in community composition from a baseline. The estimation of confidence intervals about the baseline community composition requires five observations. It is envisaged that both indicators for the rate of coral cover increase and changes in community composition will be incorporated as the time-series of this program extends.

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 though 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 LTMP reefs up to 2014 (Thompson *et al*. 2016). From these time series a binomial model was fit to juvenile densities observed at times when coral cover was below 10% and categorised based on recovery rate as being either below or above the predicted lower estimate of hard coral cover increase as estimated by the cover change indicator described below. This analysis identified a threshold of 4.6 juveniles per m^2 beyond which the probability that coral cover would subsequently increase at predicted rates outweighed the probability of lower than predicted rates of recovery.

3.3.4 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 2 uses coral cover as an example). Conversely, lower bounds were set to represent minimal resilience (Table 3). 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[2]

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

Table 3. 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}

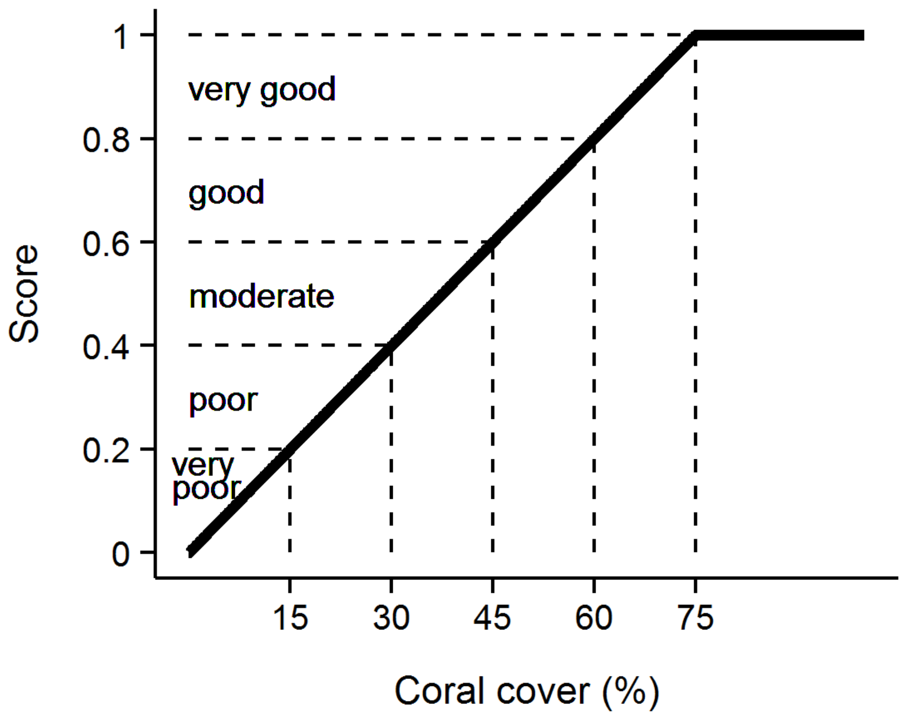

Figure 2. An example of a scoring diagram, here for the Coral Cover metric. Numeric scores and associated condition classifications based on observed coral cover are presented (see also Table 2).

3.3.5 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 4.

Table 4. 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

3.3.6 Data Analysis

A panel of plots provide temporal trends in the coral condition index and the three indicators on which the index for the Southern Inshore Zone is based.

For each of the three 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 2, Table A 3, Table A 4, and Table A 5.

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. Photos representative of coral communities at each reef and depth in 2021 are at Appendix Figure A 3 (a-f) and Figure A 4 (a-d).

3.3.7 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.

3.3.8 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 and 2021.

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.

Two sources of 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 days (DHD) were sourced from the Bureau of Meteorology's ReefTemp (Garde *et al.* 2014) and Degree Heating Weeks (DHW) sourced from NOAA coral reef watch. Thresholds at which moderate and severe bleaching are expected have been approximated as 60 and 100 DHD respectively (Maynard *et al.* 2008; Garde *et al.* 2014) and severe coral bleaching is likely at DHW values greater than eight (Lui *et al*. 2014). For both DHD and DHW estimates, realised severity of bleaching will depend on the pattern of warming and differences in the tolerances of coral species.

DHD are the sum of positive daily temperature anomalies from the seasonal climatology of a location across the period 1st December to the 31st March. A single degree heating day results from a temperature of one degree higher than the climatology mean for that day. The climatology used for this report was the IMOS 14 Day mosaic. In contrast 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.

In addition to annual maps of DHD estimates, location specific estimates were extracted for a set of nine 1 km^2 pixels centred on waypoints selected in open water approximately 2 km out from the monitored reefs (Table 5).

Table 5. Location of satellite derived environmental information.

Location	Latitude	Longitude
Pine Peak Island	-21.5467	150.2599
Pine Islet	-21.6656	150.1978
Henderson Island	-21.5291	149.9218
Temple Island	-21.6239	149.5132
Aquila Island	-21.9428	149.5535

3.3.9 Runoff

Median discharge for the water-years were calculated from available data 1986 – 2016 and 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 reported in Gruber *et al.* (2020).

3.3.10 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.

3.3.11 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* sea temperature records demonstrate temperatures across the 2020-2021 summer were well below those observed the previous year and similar to baseline conditions as estimated from historical records form the Whitsunday Islands (Figure 3). Observations during 2021 confirm that corals have either fully recovered normal colour or have perished due to thermal stress over the 2019-2020 summer.

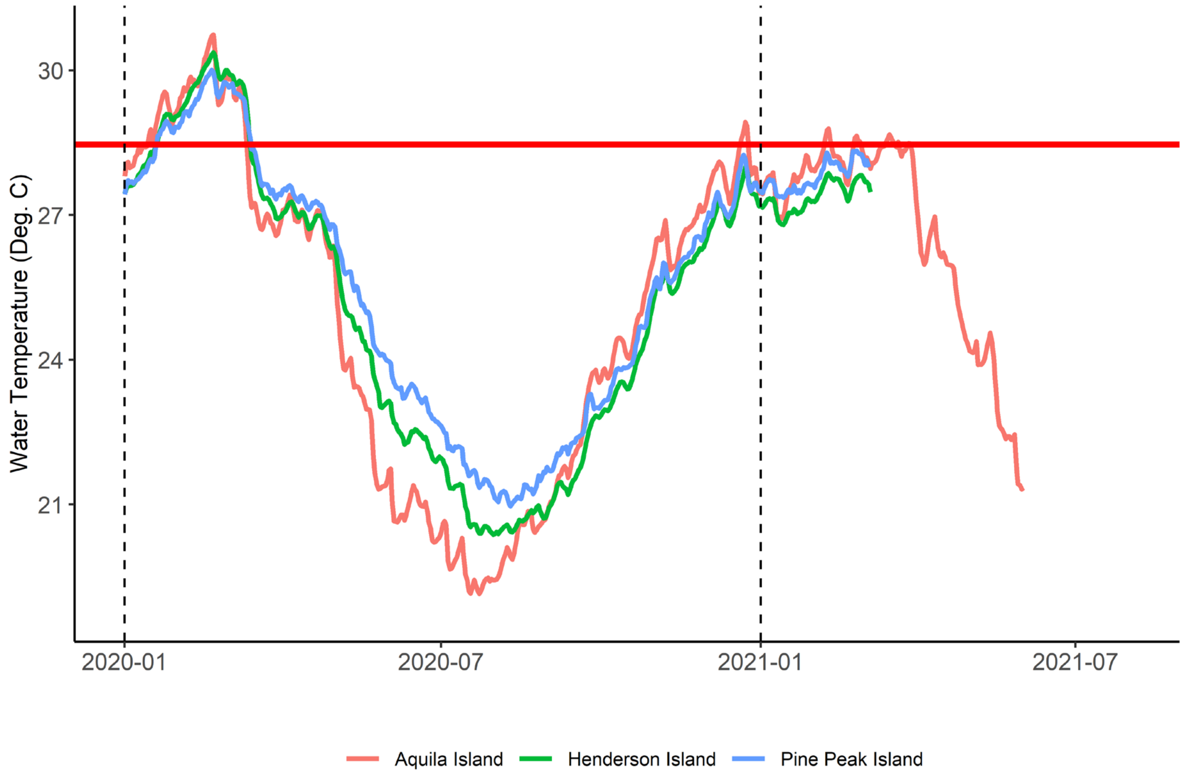

Figure 3. Temperature profiles recorded by in-situ loggers. The horizontal reference line was derived from the mean of the means of the hottest month each year observed over timeseries of in situ temperature data available from reefs in Whitsunday Islands. This baseline excluded years in which bleaching was observed.

Limited thermal stress over the 2020-2021 summer was also predicted by satellite-based temperature models. In 2021 neither degree heating day (DHD) or degree heating week (DHW) estimates exceeded those expected to cause severe coral bleaching (Figure 4, Garde *et al.* 2014, Lui *et al.* 2014), though DHDs of around 60 have been associated with mild bleaching responses (Maynard *et al.* 2008). The shallow inshore corals of Temple and Aquila islands (DHDs 62, Table 6) may have been exposed to mild temperature stress however this was not evident at the time of our surveys in June.

As an explanatory note, and to place the 2021 summer in perspective, the degree heating week estimates for 2020 were the highest recorded over the last five years, DHD estimates were even higher in 2017 (Table 6). DHD are the sum of positive daily temperature anomalies from the seasonal climatology of a location across the period 1st December to the 31st March. In contrast 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. This difference explains the slight discrepancy in estimated stress in 2017 relative to 2020 between the two indices. Higher DHD values in 2017 compared to 2020, in contrast to the relative values for DHW, suggest anomalies in 2020 occurred during the hotter period of the summer window.

Reef level DHD estimates suggest consistently higher temperatures at Aquila and Temple islands compared to the those further offshore (Table 6), a result captured in marginally higher observed temperatures at Aquila Island than those at either Henderson Island or Pine Peak Island (Figure 3).

Table 6. Annual degree heating days (DHD).. Calculated for reefs across the coastal shelf from 'outer' to 'inner'. Values above 100 are highlighted as a visual cue to where and when there was a high likelihood of thermal stress.

Year	Pine Peak	Pine Islets	Henderson	Temple	Aquila
2003	3	0	9	20	11
2004	63	34	66	64	72
2005	21	13	20	56	44
2006	63	64	75	65	75
2007	6	9	8	18	21
2008	13	9	9	17	16
2009	44	56	44	62	74
2010	46	47	48	43	23
2011	24	23	18	28	47
2012	25	23	35	36	33
2013	30	34	34	47	43
2014	5	15	6	22	16
2015	72	80	91	115	112
2016	74	61	77	90	84
2017	120	124	121	145	140
2018	47	54	46	81	73
2019	28	24	28	57	50
2020	97	106	116	131	120
2021	50	50	54	62	62

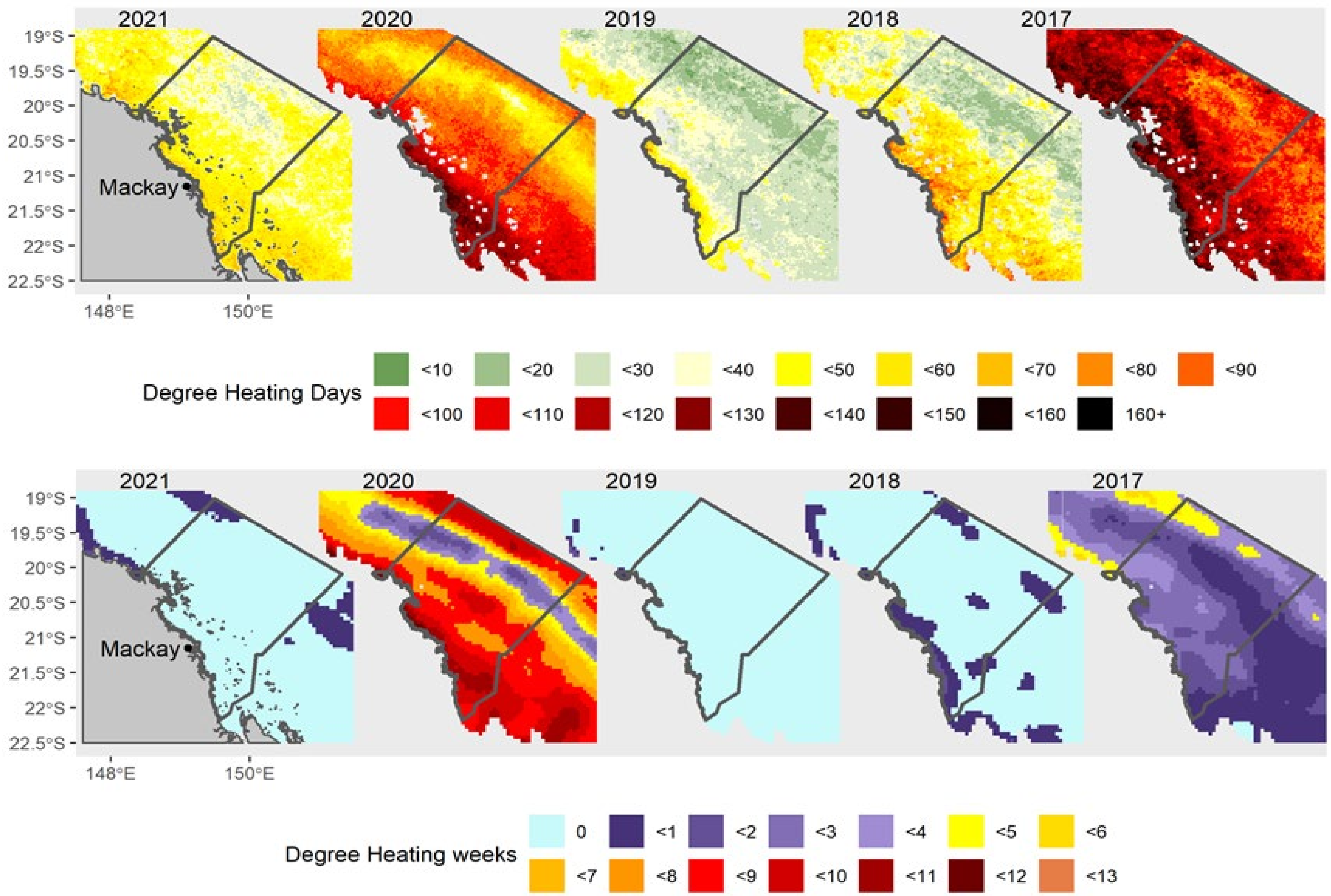

Figure 4. Annual estimates of thermal stress to corals.Top panel provides Degree Heating Day estimates downloaded from the Australian Bureau of Meteorology ReefTemp. The bottom panel provides Degree heating Week estimates downloaded from NOAA coral reef watch.

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 7). Discharge from the region's catchments over the 2020-2021 water-year (October to September) increased slightly from below median levels in the north to 1.8 times median levels for Water Park Creek (Table 7). 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 moderate flows in recent years are unlikely to have resulted in direct impacts to the coral communities monitored.

Table 7. 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
Pioneer	124001B	692,342	5.2	2.3	1.7	0.9	0.2	0.9	2.0	0.4	1.7	0.6	0.3
Plane	126001A, 126003A	309,931	4.1	2.5	1.7	0.7	0.2	0.8	2.5	0.2	1.1	1.0	0.4
Styx Shoalwater Waterpark Creek	129001A	381,986	4.8	1.5	5.2	2.9	2.0	1.8	2.7	1.4	0.7	1.5	1.8

4.1.3 Cyclones and Storms

There were no cyclones likely to have impacted reefs in the Southern Inshore Zone during the 2020-2021 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 five 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, that came closest to the reefs reported here, tracking southwards past Middle Percy with winds in excess of 80 knots, crossing the coast at Shoalwater Bay on February 20th 2015 (Figure 5). While waves from TC Marcia were not recorded, the fourth highest waves recorded at the Emu Park buoy can be attributed to this cyclone. 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.

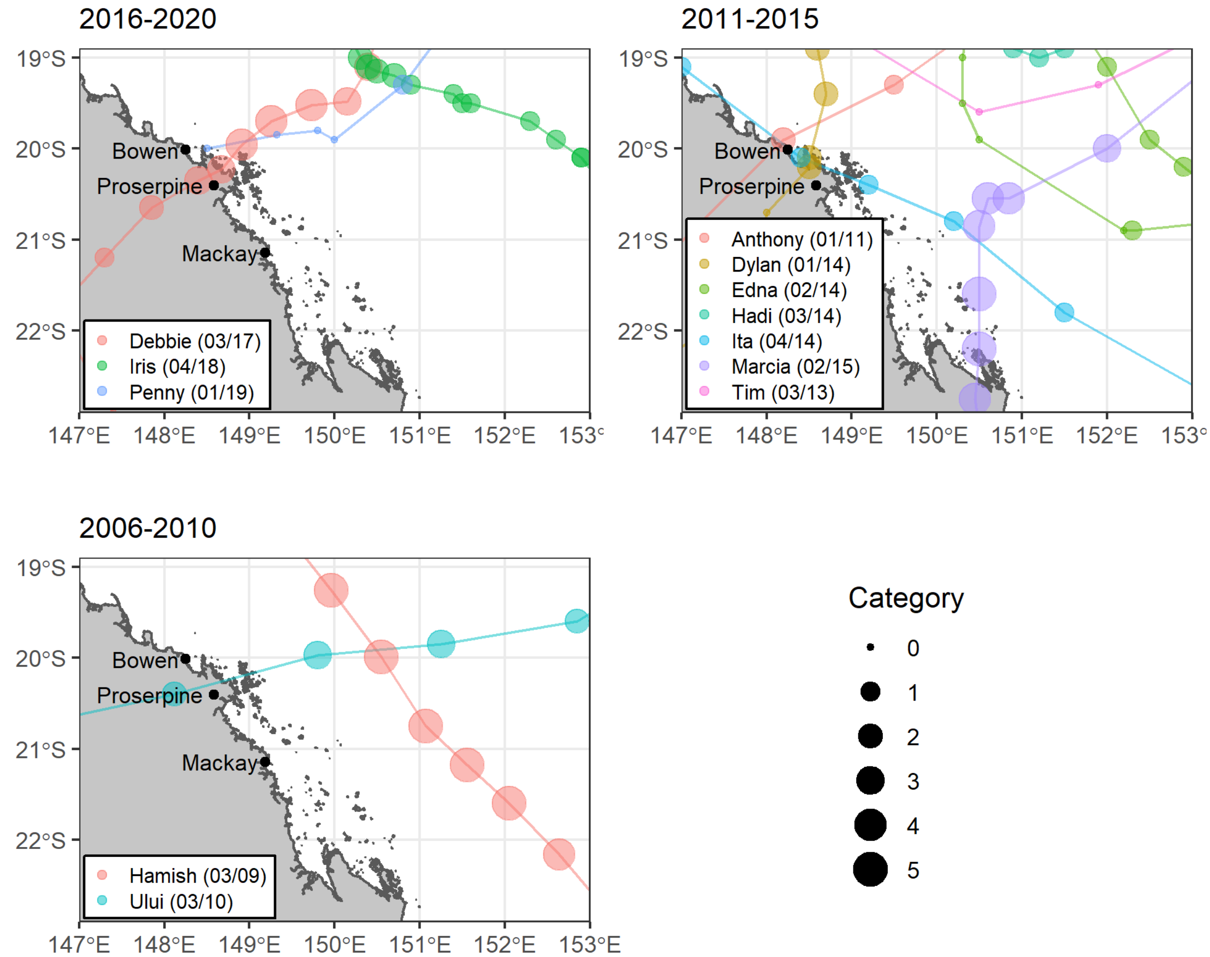

Figure 5. Tracks of tropical cyclones passing through the region.All cyclones crossing through the Mackay Whitsunday Isaac regional report card reporting area over the last 15 years are displayed.

4.1.4 Biological Damage

With colour returned to coral colonies, the distribution of disease was more easily observed than during last year's bleaching event. A total of 11 colonies were identified with disease across four of the five reefs. Diseased colonies ranged from branching Acropora to massive Porites (Table A6). In addition, there were a total of 21 colonies for which recent mortality was unknown. In combination, these 31 colonies spanned 10 genera, up from 2020 observations (14 colonies, 7 genera), but reduced from 2019 (55 colonies, 12 genera, Figure A 1, Table A 6). The greatest number of affected colonies (14 Acroporidae) were recorded in the shallows at Henderson Island (2 m) where in bleaching 2020 was most severe.

The number of colonies being overrun by the encrusting sponge *Cliona orientalis* has dropped from 15 colonies in 2020 to seven colonies in 2021. As noted last year, most observations of *C. orientalis* continue to occur among those inshore reefs with higher turbidity: Temple and Aquila islands. Afflicted colonies represent a range of genera including *Goniastrea, Turbinaria*, and *Favites spp* (Table A 6).

4.2 Coral Community Condition Assessment

The overall coral index score for the Southern Inshore Zone in 2021 was graded as E, categorising the coral communities as being in 'very poor' condition (Table 8). This represents a continuation of the grade from previous years. Connor Island was removed from the sampling design in 2021 and back-calculating zone-level scores for 2019 and 2020 indicate that this change degraded the zone level score in 2020 from 0.21 (grade D) when data from Connor Island was included to 0.19 (grade E) under the reduced design (Table 8).

Consistently minimum scores of zero for the macroalgae indicator (Table 8), due to a high proportion of macroalgae amongst the algal cover across the region (Figure 6), are highly influential in the low grade for this zone. Within this grade, scores have consistently declined since 2019 (Table 8). Most influential in this decline have been reductions in mean coral cover (Figure 6).

The indicator scores continue to vary among the reefs monitored. Scores at Pine Peak Island, Pine Islets and Aquila Island declined slightly from those recorded in 2020 and continued to sit within the range categorised as grade E (Table 9). The lowest scores again recorded at 2 m depths of Pine Peak Island and Pine Islets (Table 9). The indicator score for Henderson Island has continued to decline; at 2 m depth where scores have declined from 0.41 (grade C) in 2019 to the current 0.19 (grade E), at 5 m depth scores remain in the range of grade D (Table 9). There was a marginal increase in score at Temple Island were the community remains graded as D (Table 9).

Table 8. Coral indicator scores for 2019-2021.Values for 2019 and 2020 that include Connor Island data are included for comparitave purposes.

	Zone Scores and Grades Including Connor Island		Zone Scores and Grades		
	2019	2020	2019	2020	2021
Juvenile corals	0.13	0.16	0.12	0.14	0.10
Coral Cover	0.49	0.47	0.47	0.44	0.37
Macroalgae	0	0	0	0	0
Coral Indicator Score	0.20	0.21	0.20	0.19	0.16
Coral Indicator Grade	E	D	E	E	E

Table 9. Index scores and grade for each reef and depth combination. Comparison of the 2021, 2020 and 2019 Index scores.

Reef	Depth	Index 2019	Index 2020	Index 2021	Grade
Pine Peak Island	2	0.05	0.09	0.08	**E**
	5	0.12	0.14	0.12	**E**
Pine Islets	2	0.04	0.06	0.06	**E**
	5	0.12	0.20	0.15	**E**
Henderson Island	2	0.41	0.34	0.19	**E**
	5	0.36	0.33	0.28	**D**
Temple Island	1	0.32	0.21	0.23	**D**
Aquila Island	1	0.19	0.16	0.14	**E**

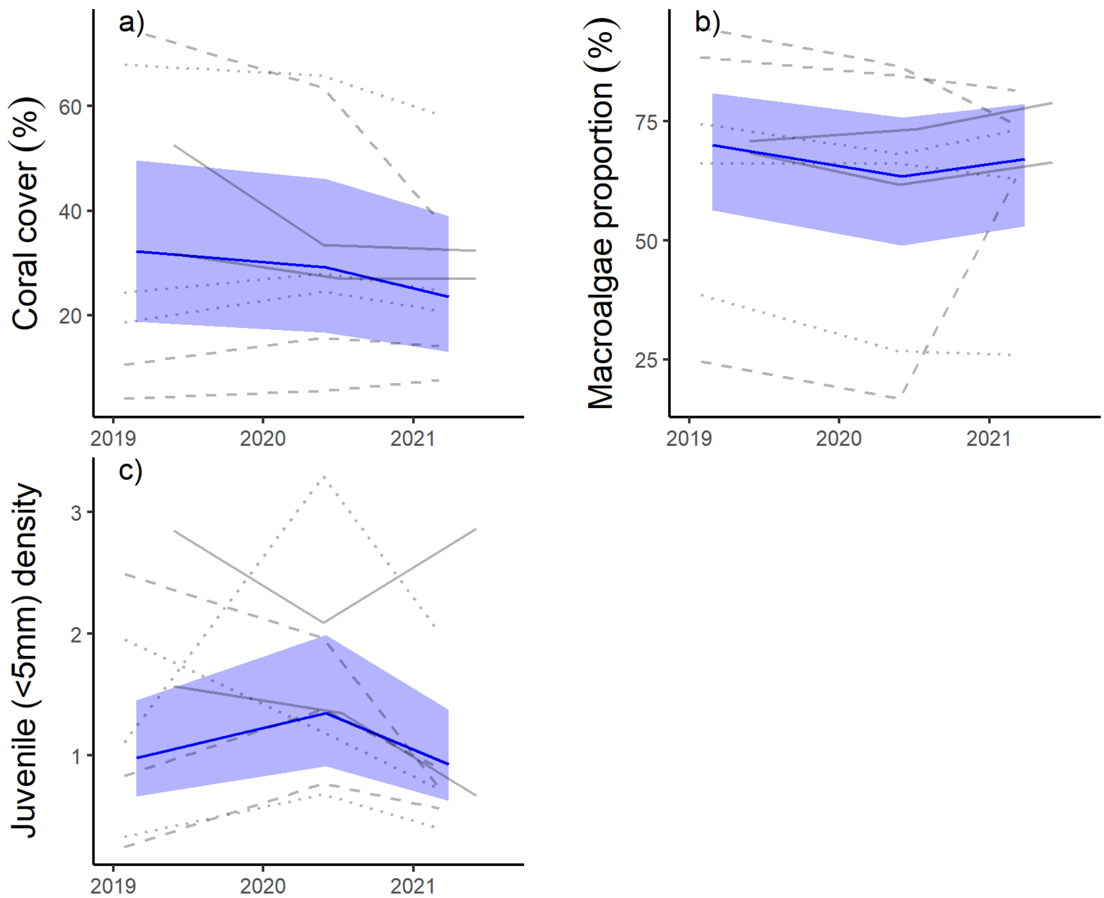

Figure 6. Indicator trends for Southern Inshore Zone. Blue lines represent trends in: a) coral cover, b) macroalgae proportion, c) juvenile density. Trends are bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles at 5m (dotted lines), 2m (dashed lines), and 1m (solid lines) for individual reefs.

4.3 Coral Cover

Coral Cover scores are based on the combined cover of hard and soft corals. Zone level mean coral cover declined further in 2021 (Figure 6a). Most influential in this decline was the continued loss of hard coral at Henderson Island (Table 10). Based on our observations that a high proportion of corals were severely bleached and ongoing mortality was occurring during surveys in 2020, the cause of further loss of coral cover at Henderson Island has been ascribed to the 2020 coral bleaching event.

Despite the further loss of coral cover recorded in 2021, Henderson Island continues to have the highest values for hard coral cover across the reefs monitored in the Southern Inshore Zone (Table 10). In 2020 hard coral cover at both 2 m and 5 m depths was above 50%; however, a high proportion of this cover (76% at 2 m and 58% at 5 m) was exhibiting signs of bleaching (Davidson *et al.* 2020). Subsequent coral mortality occurred and by 2021 more than half the cover of hard corals and a quarter of the cover of soft corals at 2 m depth had died (Table 10) as a result of the high water temperatures experienced in early 2020. This impact reduced scores for the coral cover indicator from 'very good' to 'satisfactory'. Corals at 5 m depth were less impacted and the coral cover score dropped to a grade of 'good' (Table 10). Although susceptible to coral bleaching, the genus *Acropora* remains the dominant hard coral at Henderson Island (Table A 2, Figure A 2). The only bleaching recorded from the region in 2021 was a single large corymbose *Acropora* colony on the reef flat at Henderson Island, and a single encrusting colony of *Coscinaraea* on the slopes of Pine Peak Island. There was no indication that these were remnants of the 2020 event, or that there was an obvious reason for colour loss.

A minor downward trend in both hard and soft coral was recorded at Pine Peak Island for both 2 m and 5 m depths. For the shallow 2 m depth the coral cover score was already on the boundary of 'poor' (0.21), and the slight decline in coral cover changed the grade to 'very poor' (Table 10). At 5 m the hard coral cover was marginally higher and, combined with the abundant *Briareum* soft coral community (Table A 3), kept the overall coral cover grade as 'poor' (Table 10).

Pine Islets continues to have a very low cover of both hard and soft coral at 2 m depth and retains the lowest coral cover score in the zone (Table 10). On the reef slope, hard coral cover is marginally higher and dominated by large colonies of *Monitpora* (Table A 2). The encrusting soft coral *Briareum* was resilient to high water temperatures and contributes strongly to maintenance of the coral cover grade of 'poor' (Table 10, Table A 3).

Further inshore, both Temple and Aquila islands experienced only minor fluctuations in coral cover that had no influence on the grades in 2021 compared to previous years that remain 'satisfactory' and 'poor' respectively (Table 10); slight reductions across hard coral genera were met with minor increases in soft coral.

Table 10. Coral cover and indicator scores for each location 2019 - 2021. Coral cover scores are coloured as per Table 3.

Reef	Depth	Year	Hard coral	Soft coral cover	Coral cover (%)	Coral cover Score
Pine Peak Island	2	2019	3.45	7.15	10.6	0.14
		2020	5.44	10.31	15.75	0.21
		2021	4.13	10.01	14.14	0.19
	5	2019	9.39	14.96	24.35	0.32
		2020	9.31	18.69	28	0.37
		2021	7.5	17.31	24.81	0.33
Pine Islets	2	2019	2.69	1.38	4.06	0.05
		2020	4.25	1.31	5.56	0.07
		2021	6.19	1.44	7.63	0.1
	5	2019	14.75	3.94	18.69	0.25
		2020	19.8	4.76	24.56	0.33
		2021	13.75	7.1	20.85	0.28
Henderson Island	2	2019	57.05	17.9	74.96	1
		2020	52.13	11.31	63.44	0.85
		2021	24.94	12.94	37.88	0.51
	5	2019	48.75	19.19	67.94	0.91
		2020	51	14.82	65.82	0.88
		2021	44.31	14	58.31	0.78
Temple Island	1	2019	19.5	33.13	52.63	0.7
		2020	19.82	13.69	33.51	0.45
		2021	16.72	15.59	32.3	0.43
Aquila Island	1	2019	20.75	11	31.75	0.42
		2020	18.53	8.51	27.04	0.36
		2021	16.44	10.56	27	0.36

4.4 Macroalgae Proportion

The proportion of macroalgae in the algal community continued to exceed thresholds across all reefs (Table 3), resulting in a macroalgae grade of E (‘very poor’, Table 11).

At all reefs the macroalgae community is dominated by large brown algae of the genus *Sargassum* or *Lobophora*, *Sargassum* tending to be more abundant at the shallower sites with *Lobophora* taking over at 5 m depths (Table A 4). An exception is Henderson Island where *Lobophora* has increased from 2% to 34% at 2 m depth following the 2020 bleaching event (Table A 4), a phenomenon reported previously in the Keppel Islands (Diaz-Pulido *et al.,* 2009).

In general, the cover and proportion of macroalgae in the algal communities on each reef has been variable among years (Table 11). In 2021 the greatest decline occurred at 2 m depth at Pine Islets, where cover declined from 79% in 2020 to 65% in 2021 (Table 11). Elsewhere cover and proportional representation of macroalgae in the algal communities has remained stable since 2020 (Table 11, Figure 6b).

Table 11. Macroalgae cover and indicator scores for each location, depth, and year.

Reef	Depth	Year	Macroalgae cover (%)	Macroalgae proportion (%)	Macroalgae score
Pine Peak Island	2	2019	77.86	88.58	0
		2020	68.3	84.59	0
		2021	67.41	81.48	0
	5	2019	51.69	74.55	0
		2020	44.25	68.14	0
		2021	50.81	73.18	0
Pine Islets	2	2019	89.75	94.66	0
		2020	78.75	86.66	0
		2021	65.19	74.39	0
	5	2019	48.69	66.13	0
		2020	41.43	66.1	0
		2021	44.75	62.85	0
Henderson Island	2	2019	5.76	24.52	0
		2020	5.56	16.73	0
		2021	38.38	63.04	0
	5	2019	8.81	38.52	0
		2020	5.64	26.73	0
		2021	8.31	25.88	0
Temple Island	1	2019	27.19	68.5	0
		2020	36.16	61.69	0
		2021	39.02	66.5	0
Aquila Island	1	2019	32.31	70.82	0
		2020	35.92	73.49	0
		2021	43.13	78.95	0

4.5 Juvenile Density

The density of juvenile corals declined and continues to be categorised as 'very poor' (Figure 6, Table 8). The only reef to show an increase in density and abundance of juveniles was Temple Island where the score increased from 'very poor' to 'poor'(Table 12). This rise was due to the moderate increase in abundance of *Pocillopora* (family Pocilloporidae), and *Turbinaria* (family Dendrophylliidae)(Figure A 2). In contrast the score for this indicator declined 'poor' to 'very poor' at Pine Island 5 m (Table 12) where the number of genera represented by juvenile colonies also declined, from 31 in 2020 to 23 in 2021.

Table 12. Juvenile hard coral abundance, density and indicator scores for each location. Density has been adjusted for the area of algal covered substrates.

Reef	Depth	Year	Juvenile abundance	Juvenile density (per m^2)	Juvenile score
Pine Peak	2	2019	7.5	0.25	0.02
		2020	21	0.77	0.07
		2021	16	0.57	0.05
	5	2019	8	0.33	0.03
		2020	15	0.68	0.06
		2021	9.5	0.4	0.04
Pine Islets	2	2019	27	0.83	0.07
		2020	43	1.38	0.12
		2021	27	0.9	0.08
	5	2019	28	1.11	0.1
		2020	69.5	3.3	0.29
		2021	48.5	2.01	0.17
Henderson Island	2	2019	21	2.49	0.22
		2020	22.5	1.97	0.17
		2021	15	0.73	0.06
	5	2019	14.5	1.95	0.17
		2020	8.5	1.2	0.1
		2021	7.5	0.72	0.06
Temple Island	1	2019	39	2.85	0.25
		2020	42	2.09	0.18
		2021	54.5	2.87	0.25
Aquila Island	1	2019	24.5	1.57	0.14
		2020	23.5	1.35	0.12
		2021	12.5	0.67	0.06

4.6 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). Only once was work postponed due to poor visibility – at Aquila Island (Table 13).

Wind impacts ability to complete surveys both from a comfort and safety perspective in travelling to and from dive sites but also causing wave driven resuspension that reduces in-water visibility.

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, launch sites are tidally constrained. Access to Aquila Island is most convenient via Carmila Creek. This requires ~3.5m of tide at McEwen Island (Bureau of Meteorology Tide Predictions). 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 and the open waters, winds <15 knots are required. The 2021 resurvey was fortunate to have a rare opportunity to utilise the AIMS vessel RV Cape Ferguson in March to survey these outer reefs; suitable conditions to survey Aquila and Temple islands did not occur until June. Table 13 summarises the timing of, and conditions experienced, for 2021 surveys. All field work activities, including time on boat, meal-times and overnight accommodation, were carried out under a comprehensive risk assessment protocol with strict compliance to COVID-19 controls.

Planning for future resurveys of Henderson Island, Pine Islets, and Pine Peak Island should consider the use of a suitable vessel for a limited overnight charter of 3 nights. This would give the monitoring team access to work sites without the need for a daily four-hour journey, with reduced time on water adding an extra element of safety. It would also allow greater control over trip planning as access to these outer reefs as access would be less wind dependant, and the need for sufficient tide to launch from Sarina Beach alleviated.

Table 13. Weather conditions and tide heights experienced during 2021 works.

Reef	Date	Wind (knots)	Tide (Range)	Observations
Pine Peak Island Pine Islets (1.5 sites)	6/03/2021	10 ESE	Falling->Low (2.4m) Rising->High (4.3m)	Visibility 7-10m negligible current.
Pine Islets (0.5) Henderson Island	7/03/2021	5-10 E	Falling->Low (2.1m) Rising->High (4.4m)	Visibility 5-7m slight Visibility 8-9m
Temple Island Aquila Island	3/06/2021	5 E	Falling->Low (2.1m) Rising->High (6.1m)	Visibility 1m no current at Temple Is, current increase on turn of tide at Aquila Is

5 DISCUSSION

The overall condition of Southern Inshore Zone reefs in 2021 was categorised as 'E' and graded 'very poor' based on a coral index score of 0.16. This represents an ongoing decline from the initial 2019 index score of 0.20.

Most influential in this decline has been the loss of coral cover as the result of a marine heat wave in early 2020 that caused widespread coral bleaching. The cumulative impact of this event continued to reduce coral cover over the period between surveys in 2020 and 2021. There were no other noted disturbances to reefs in the region in 2020-2021.

The extent of bleaching among reefs monitored in the Southern Inshore Zone was detailed in the 2020 report (Davidson *et al.* 2020). None of the reefs in our study escaped bleaching, but the level of bleaching did vary among reefs. Unique among the reefs in this monitoring program, Henderson Island has the lowest cover of macroalgae and highest cover of hard corals, principally *Acropora.* This site was also the most severely impacted by the 2020 coral bleaching event. At the time of the 2020 survey, several months after the marine heat wave had passed, 76% of the shallow corals, and 58% of the slope corals at Henderson Island were still bleached. When re-surveyed in 2021, these corals had either survived and regained normal colour or died allowing the full impact of the bleaching event to be estimated. Compared to cover observed in 2019, over 50% of hard coral cover was lost from the 2 m depths. The impact on hard corals was substantially lower at 5 m depth where just under 10% of hard coral cover was lost. Soft coral cover was also impacted with approximately a quarter of soft coral cover lost between 2019 and 2021. Sully and van Woesik (2020) suggest that a turbid environment can moderate the impact of bleaching, that those coral communities in 'marginal habitats' are better able to deal with higher temperatures and periods of reduced sustenance during bleaching as they are more likely to switch to heterotrophy to fulfil their energy requirements. This concept may explain the reduced impacts observed at 5 m depth at Henderson Island, but also the minimal impacts of the marine heat wave on the more inshore reefs, situated in often highly turbid waters.

Delayed coral mortality following marine heat waves has been be linked to coral disease as pathogens overcome corals weakened by thermal stress (Bruno *et al.* 2007, Brodnicke *et al.* 2019, Howells *et al.* 2020). In 2020 assessing the prevalence of disease was made difficult by the bleached state of the coral community, and in 2021 there was no knowing how many corals had been infected by, then succumbed to, disease in the intervening period. The level of disease among these reefs was similar in 2019 before and 2021 after the bleaching event suggesting ongoing stress was unlikely. However, thermal stress may have a longer-lasting effect on the resilience of these communities as thermal stress has been shown to reduce spawning, settlement, and recruitment (Ward *et al.* 2002, Hughes *et al.* 2019) in subsequent years.

Any reduction in coral recruitment in this area is a concern. 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 (Diaz-Pulido *et al*. 2010, Mumby *et al*. 2013, Leong *et al*. 2018). Of particular interest is the increase in cover of *Lobophora* macroalgae in the shallows of Henderson Island following the recent loss of coral cover due to coral bleaching. The persistent cover of this algae has been attributed to low recruitment of hard corals and lack of recovery of coral cover; the conclusion of a study of an inshore reef off Townsville (Johns *et al.* 2019). However, Diaz-Pulido *et al.* (2009) describe a similar situation where a similar rapid increase in *Lobophora* occurred when branching *Acropora* were partially killed following a beaching event in the Keppel Islands. In that study the cover of *Lobophora* did not persist and rapid growth of surviving hard corals occurred, dominating the reef community, following a bleaching event in the Keppel Islands region. The state of the algal community over the next few years is expected to be a strong factor influencing the trajectory of coral cover at this reef.

The environmental conditions of the Southern Inshore Zone have been identified by previous studies as a marginal 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 the harsh ambient conditions of the area.

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 on 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.

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8 APPENDICES

Table A 1. 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, contour to end
	21.51433	**150.25125**	**5**	**1**	**1**	340 then contour
	Waypoint between transects 3 & 4				2	150, 110@6m, 60@10m rod, 320 to T3
					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, 200@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, 120@15m
					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, 220@10m
	Waypoint between transects 3 & 4				2	340, 350@10m
					3	240
					4	50, 90@10m
					5	130
	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

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
					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, 20@10m
					4	130, 100@10m
					5	150, 200@10m
	21.48313	**149.90868**	**2**	**2**	**1**	310
	Waypoint between transects 3 & 4				2	320
					3	310, 300@10m
					4	120
					5	150
	21.48317	**149.90845**	**5**	**2**	**1**	0, 350@10m
	Waypoint between transects 3 & 4				2	300, 320@10m
					3	330, 310@10m
					4	180, 170@10m
					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
					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	310
					5	330, 310@10m
	21.96112	**149.58158**	**1**	**2**	**1**	120
	Waypoint between T1-T4				2	90
					3	110
					4	0
					5	30

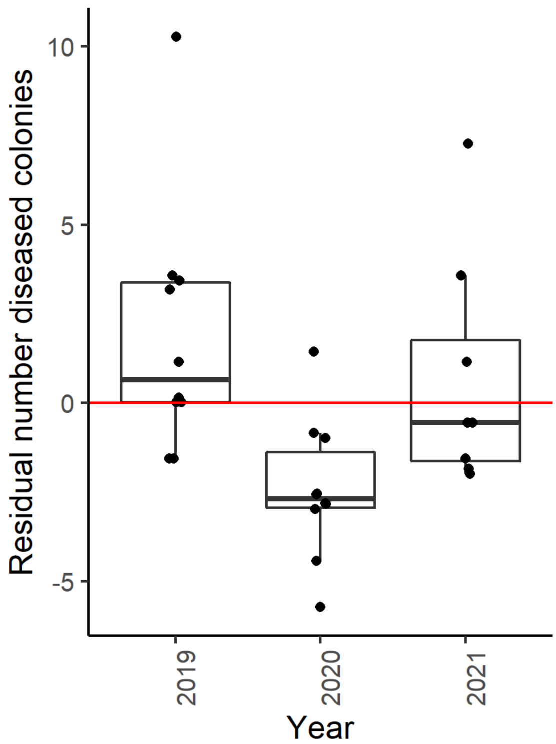

Figure A 1.Relative coral disease by year. Data are standardised to the reef and depth mean across years. Boxplots show the median (bold horizontal line), 25th to 75th quartiles (box), and 1.5 times the inter-quartile range (whiskers). Solid dots are the relative number of coral colonies suffering ongoing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

Table A 2. Cover of hard coral genera 2021. 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 presented as Genus Richness

Reef	Depth	*Acropora*	*Galaxea*	*Goniopora*	*Lobophyllia*	*Montipora*	*Platygyra*	*Pocillopora*	*Porites*	*Turbinaria*	Other HC	Genus Richness
Pine Peak Island	2	0.00	0.00	0.44	0.00	1.00	0.06	0.25	1.31	0.00	1.06	13
	5	0.06	0.25	0.06	0.00	0.81	0.06	0.44	3.06	0.50	2.25	19
Pine Islets	2	0.31	0.00	0.13	0.06	2.44	0.00	0.06	1.38	0.50	1.31	13
	5	0.19	0.00	0.75	0.06	4.81	0.00	0.06	2.25	1.69	3.94	22
Henderson Island	2	22.81	0.19	0.13	0.19	0.56	0.19	0.31	0.00	0.00	0.56	13
	5	32.13	1.13	1.63	3.38	1.75	0.06	0.44	0.31	0.31	3.19	23
Temple Island	1	3.76	0.00	0.44	0.00	5.51	1.25	1.81	0.25	2.19	1.50	14
Aquila Island	1	0.13	0.00	0.06	0.00	13.63	0.00	0.06	0.38	0.44	1.75	11

Table A 3. Cover of soft coral genera 2021. Genus with a cover of at least1% at any reef are included. All less abundant genera are grouped as Other SC

Reef	Depth	*Briareum*	*Cladiella*	*Klyxum*	*Lobophyton*	*Sarcophyton*	*Sinularia*	*Xenia*	Other SC
Pine Peak Island	2	5.82	0.00	0.00	0.56	1.06	2.07	0.19	0.31
	5	12.19	0.00	0.31	0.13	0.31	4.06	0.00	0.31
Pine Islets	2	0.56	0.06	0.00	0.56	0.00	0.25	0.00	0.00
	5	3.08	0.00	1.32	0.31	0.63	1.77	0.00	0.00
Henderson Island	2	1.88	3.44	2.69	0.38	1.00	3.56	0.00	0.00
	5	0.31	3.63	5.06	0.00	3.31	1.69	0.00	0.00
Temple Island	1	3.01	0.31	0.06	1.25	0.44	10.45	0.00	0.06
Aquila Island	1	0.13	0.06	0.00	0.38	0.56	8.19	0.88	0.38

Table A 4. Cover of algae groups 2021. 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	Blue-green algae	Coralline algae
		Dictyota	*Lobophora*	*Padina*	*Sargassum*	*Stypopodium*	Other	Calcareous red macroalgae	Other	*Caulerpa*	Other			
Pine Peak Island	2	0.38	12.70	0.06	46.03	1.25	0.75	0.06	4.94	1.19	0.06	13.45	0.19	1.69
	5	0.25	31.88	0.06	15.06	0.38	1.13	0.06	1.56	0.06	0.38	14.50	0.06	4.06
Pine Islets	2	1.00	14.06	2.06	41.13	0.25	1.75	0.63	4.31	0.00	0.00	20.44	0.06	1.94
	5	0.19	26.73	1.07	14.32	0.13	0.94	0.06	1.13	0.00	0.19	23.64	0.00	2.82
Henderson Island	2	0.06	34.19	0.25	3.81	0.00	0.00	0.00	0.06	0.00	0.00	22.44	0.06	0.00
	5	0.13	7.31	0.06	0.31	0.00	0.25	0.00	0.19	0.00	0.06	23.63	0.00	0.19
Temple Island	1	0.00	4.07	0.00	22.99	0.00	0.75	2.32	7.89	0.88	0.13	18.28	0.00	1.38
Aquila Island	1	0.00	1.63	0.00	29.25	0.00	0.19	1.94	9.50	0.31	0.31	10.56	0.00	0.94

Table A 5. Abundance of juvenile hard coral genera 2021. Mean abundance per site for genera with at least 2 corals per site at any reef separated. All less abundant genus grouped as Other genera.

Reef	Depth	*Acropora*	*Cyphastrea*	*Favia*	*Favites*	*Goniastrea*	*Goniopora*	*Lobophyllia*	*Montipora*	*Moseleya*	*Pocillopora*	*Porites*	*Pseudosiderastrea*	*Scolymia*	*Turbinaria*	Other genera	Genus Richness	Number	Density
Pine Peak Island	2	0.5	0	0	1	0	1	0	0.5	0.5	1.5	6	0	0	1	2	15	16	0.57
	5	0.5	0	0.5	0.5	0	0.5	1	0	0	2.5	2.5	0	0	0	0.5	10	9.5	0.41
Pine Islets	2	2.5	0	3	2.5	1.5	1	1.5	2	0	1	3.5	0	0.5	2.5	3	18	27	0.90
	5	1	1	2	1.5	0	3.5	9	6.5	0.5	0.5	3	1.5	3	7.5	5.5	23	48.5	2.01
Henderson Island	2	1.5	0	0.5	0	5	0.5	0	0	0	1	0	2	0	0.5	2.5	13	15	0.73
	5	1.5	0	0	0.5	0.5	0	2.5	0.5	0	0	0	0	0	0	1.5	9	7.5	0.72
Temple Island	1	3.5	2.5	3	2.5	1.5	1	0	5	2	12.5	2.5	0	0	17.5	0	12	54.5	2.87
Aquila Island	1	0	0.5	2	0.5	0.5	0	0.5	3.5	1.5	0	0.5	0.5	0	2.5	0	10	12.5	0.67

Table A 6. Coral health survey results 2021. Number of colonies along the ten 20 m long and 2 m wide transects searched at each reef and depth combination 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.

Cause	Genus	Pine Peak		Pine Islets		Henderson		Temple	Aquila
		2m	5m	2m	5m	2m	5m	1m	1m
Disease	*Acropora*					3			
	Coscinaraea								1
	Montipora				4	1			
	Porites		1		1				
Unknown cause	*Acropora*			1		10	2		1
	Mycedium	1							
	Oxypora		1						
	Pachyseris				1				
	Pectinia		1						
	Pocillopora				1				
	Turbinaria			1					
Sponge - *Cliona orientalis*	*Cyphastrea*							1	
	Favites				1			1	
	Goniastrea							2	
	Montipora								1
	Platygyra							1	
	Turbinaria					1			1
Hyperplasia	*Montipora*				1				
Sediment	*Montipora*								3
Total number of Colonies		**1**	**3**	**2**	**9**	**15**	**2**	**5**	**7**

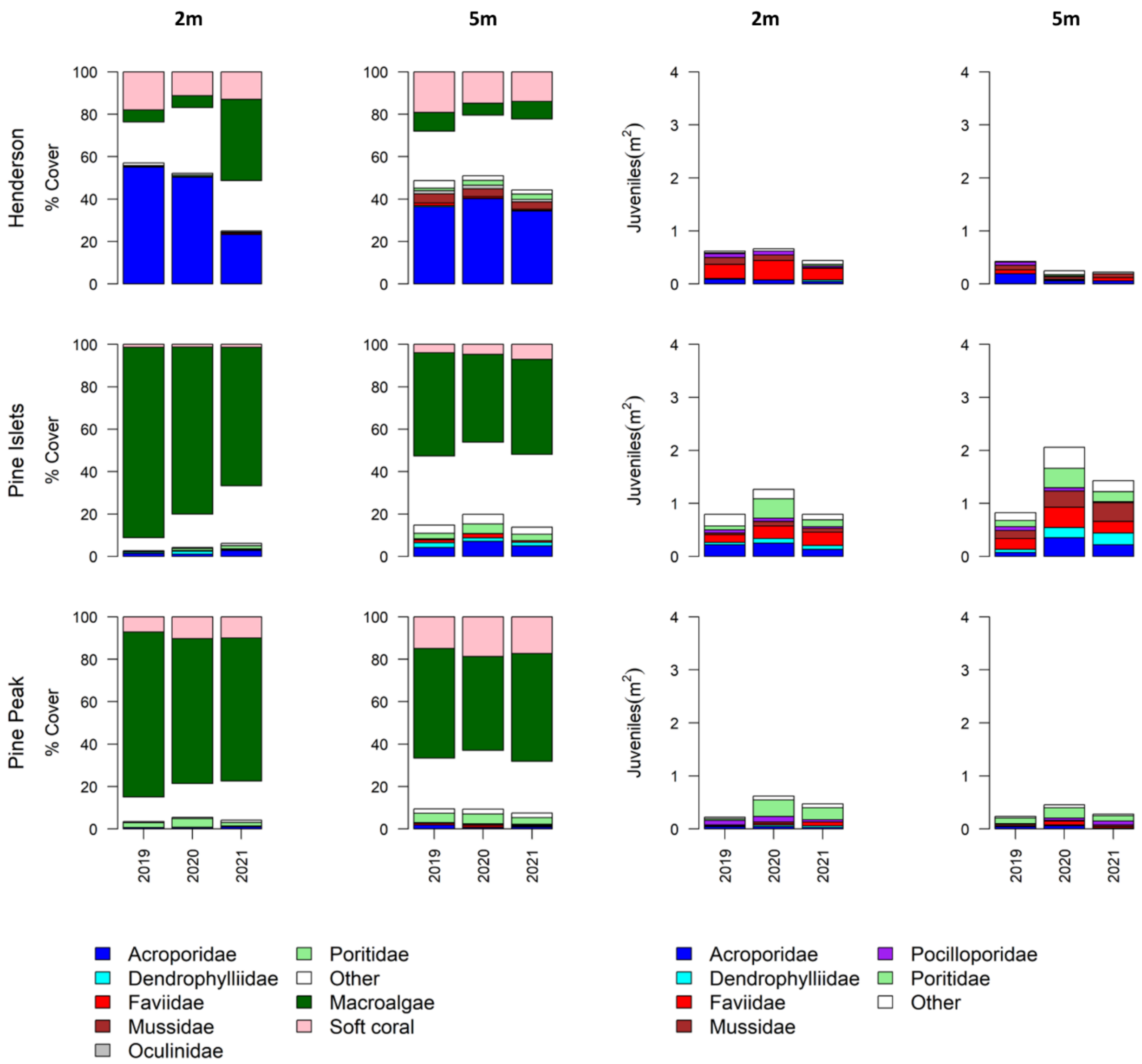

Figure A 2. Composition of benthic cover and hard coral juveniles. The left-hand plots show the breakdown of cover for hard coral families at 2 m and 5 m depths. Families that had a cover of at least 3% at either depth of any reef in the Zone are differentiated. Cover of all other families are grouped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

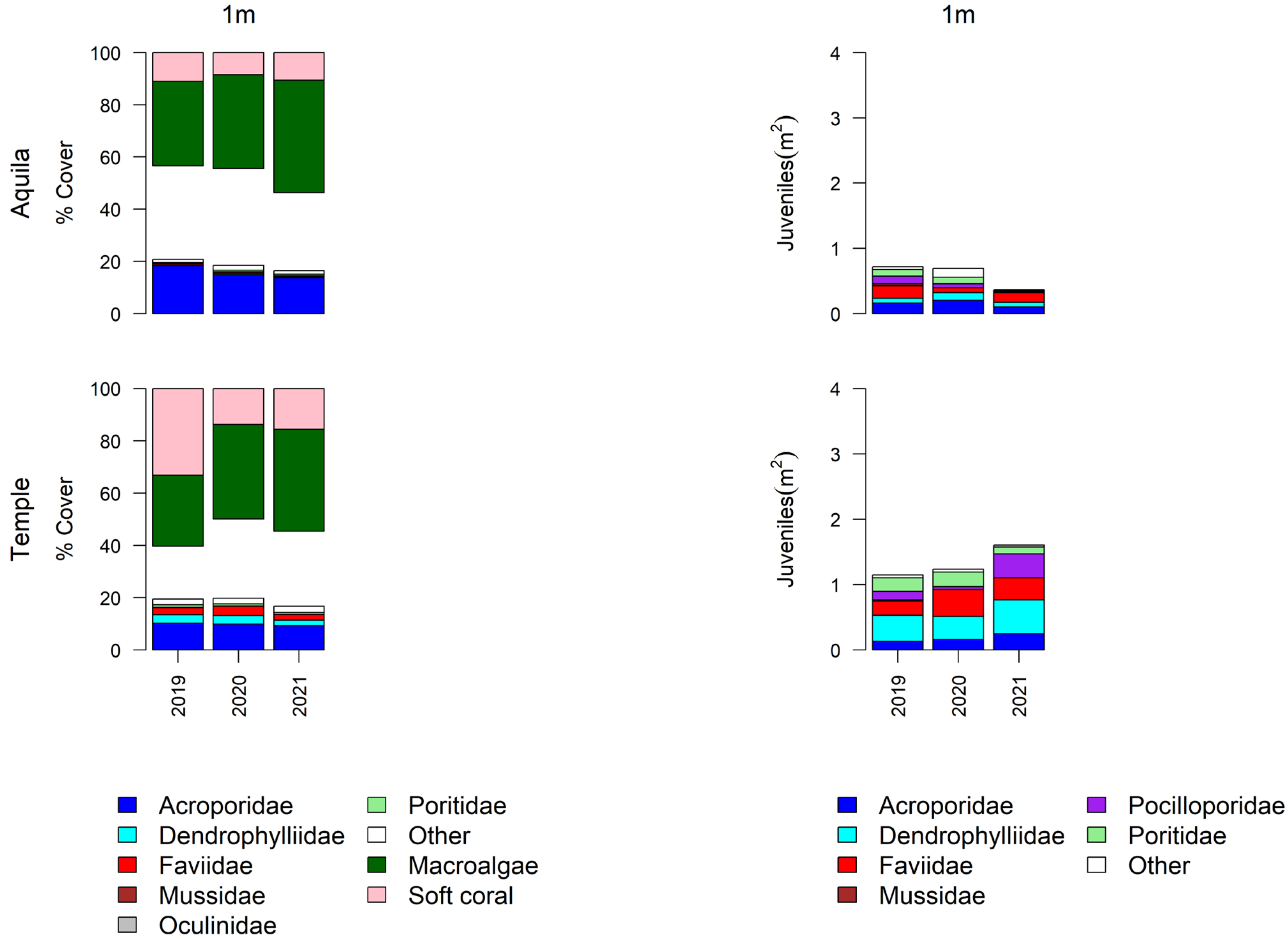

Figure A 2 continued, for the 1 m deep sites at Aquila and Temple islands.

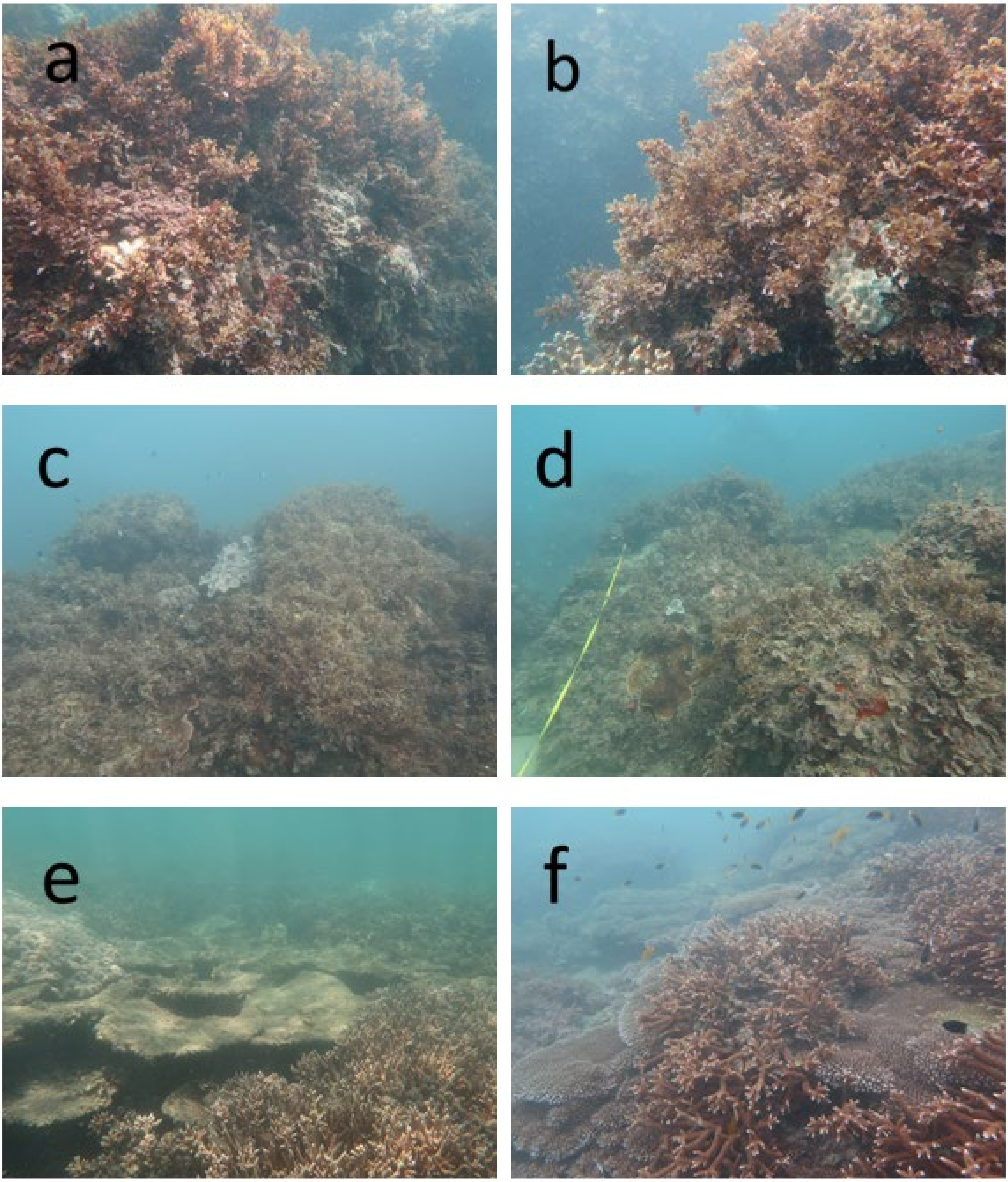

Figure A 3. Benthic community photos at outer reefs a) Pine Peak Island 2m b) Pine Peak Island 5m c) Pine Islets 2m d) Pine Islets 5m e) Henderson Island 2m f) Henderson Island 5m. Macroalgae dominate Pine Peak Island and Pine Islets at 2m and 5m, where scattered colonies of *Acropora*, *Montipora,* and *Porites* colonies have survived the bleaching. By contrast, fields of *Acropora* corals at Henderson Island have survived despite the heavy mortality from bleaching at both 2m and 5m.

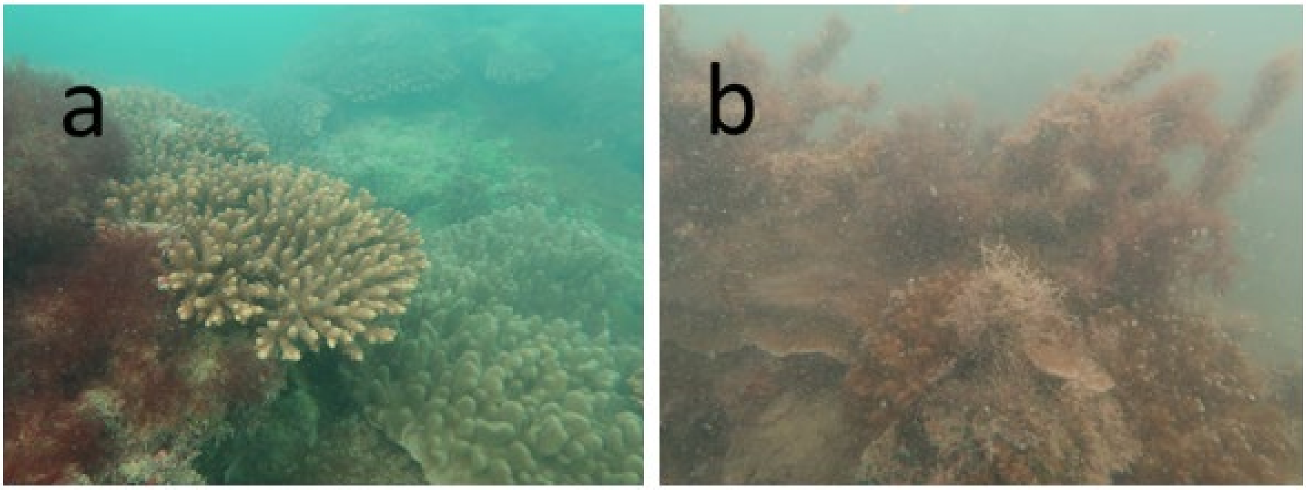

Figure A 4. Benthic community photos at inner reefs a) Temple Island 1m b) Aquila Island 1m. Mixed hard and soft corals at Temple Island 1m show complete recovery from bleaching. At Aquila Island 1m, mixed colonies of foliose *Montipora* and digitate *Porites* continue to thrive.