Southern Inshore Marine Zone Coral Monitoring Program Baseline Report 2019

Johnston Davidson, Angus Thompson, Paul Costello

Report prepared for

Mackay-Whitsunday-Isaac Healthy Rivers to Reef Partnership

Australian Government

Australian Institute of Marine Science

AIMS: Australia's tropical marine research agency

Australian Institute of Marine Science
PMB No 3
Townsville MC Qld. 4810

This report should be cited as:
Davidson J, Thompson A, Costello P (2019) Southern Inshore Marine Zone - Coral Monitoring Program: Baseline Report 2019. Report prepared for Mackay-Whitsunday-Isaac Healthy Rivers to Reef Partnership. Australian Institute of Marine Science, Townsville. (pp)

DISCLAIMER

<table>
<tr><td colspan="5">Vendor shall ensure that documents have been fully checked and approved prior to submittal to client</td></tr>
<tr><td colspan="2">Revision History:</td><td>Name</td><td>Date</td><td>Comments</td></tr>
<tr><td rowspan="2">1</td><td>Prepared by:</td><td>Johnston Davidson, Angus Thompson</td><td>29.10.2019</td><td></td></tr>
<tr><td>Approved by:</td><td>Nicole Webster</td><td>29.10.2019</td><td></td></tr>
<tr><td rowspan="2">2</td><td>Reviewed by</td><td></td><td></td><td></td></tr>
<tr><td>Revised by</td><td></td><td></td><td></td></tr>
</table>

Cover photo: Henderson Island reef slope January 2019

CONTENTS

LIST OF FIGURES

LIST OF TABLES

1 EXECUTIVE SUMMARY

This report details the site selection and baseline surveys that allow estimation of coral scores for the Southern Inshore Marine Zone of the Mackay-Whitsunday-Isaac Report Card. The setup and survey of these reefs represents a 50/50 co-investment between the Mackay-Whitsunday-Isaac Healthy Rivers to Reef Partnership and the Australian Institute of Marine Science that begins to fill a knowledge gap surrounding the state and dynamics of coral communities in this macro-tidal area of the Great Barrier Reef. The survey design captures variability in communities across a steep gradient in water quality from reefs within 3 km of the coast, situated in highly turbid waters, out to those ~80km offshore.

Sampling methods replicated those used by the Marine Monitoring Program to allow estimation of coral condition index scores, consistent with those reported in other inshore zones of regional and Reef level report cards. Surveys were undertaken between the 27th January and 27th May 2019. The results translate into a coral index score of 0.20. In the context of the Reef Report Card, this score translates into a report card grade of E and categorises coral communities as being in very poor condition.

The scores are heavily influenced by the very high proportion of macroalgae amongst the algal community, with scores of 0 returned at all locations. Given macroalgae are known to suppress coral recruitment processes, it is unsurprising that scores for the juvenile coral indicator are also poor or very poor across the zone. However, despite clear limitations to coral resilience indicated by the macroalgae and juvenile scores, coral cover was sufficient to be categorised as grade A (very good) at Henderson Island, grade B (good) at Temple Island, grade 'C' (satisfactory) at Connor and Aquila islands. It was only the more offshore reefs at Pine Peak Island and Pine Islets where coral cover was categorised as grade D (poor) at 5 m depths and grade E (very poor) at 2 m depths.

Sampling of the Henderson Island, Connor Island, Pine Peak Island and Pine Islets occurred in late January. Some macroalgae, including *Sargassum* a genus common at these reefs, are known to be seasonally abundant over summer months. It is possible the timing of sampling has resulted in underestimation of coral index scores. Sampling coinciding with seasonally high macroalgal cover will result in an underestimate of the macroalgae indicator score but also scores for juvenile and coral cover due to reduced detection of both adult and juvenile corals beneath the algal canopy. Future sampling should attempt to avoid the summer period and be targeted toward May, June each year.

This report presents baseline results from these reefs, making it difficult to accurately identify the pressures that contributed to the observed variability in coral community condition. The area is continually exposed to strong tidal currents and resuspended sediment, factors which are likely to have limited reef development in the longer-term. The prevalence of macroalgae suggests water quality favours this group of coral competitors. Recent weather data also suggest that coral bleaching, as a result of high sea surface temperatures in 2017, along with high waves associated with Cyclones Dylan in 2014, Marcia in 2015, Debbie in 2017 and Iris in 2018 may have caused losses of coral cover in recent years. Given the bottleneck imposed by very low densities of juvenile corals, it will be important to assess whether recovery can occur if coral cover is reduced to low levels, as is currently the state at Pine Peak Island and Pine Islets.

Finally, we discuss the potential need to revisit the selection of Aquila Island as a monitoring location based on logistical constraints realised during baseline surveys.

2 BACKGROUND

Coral communities are an iconic component of the marine ecosystems in Northern Australia. Inshore coral reefs of the Great Barrier Reef are impacted by multiple pressures including large scale disturbances such as cyclones, through to more localised issues such as elevated levels of nutrients or suspended sediments that may result from activities in the coastal zone and in adjacent catchments.

Successful management of coral communities requires the ability to identify where and when the resilience of communities is compromised and then the identification of the causative pressures. The Mackay-Whitsunday-Isaac

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 agreed design included stratification of monitoring effort across the steep environmental gradient of water quality that improves with distance from the coast. In scoping documents this design included replication of reefs into zones across the coastal shelf classified as Inner, Mid and Outer shelf positions. These same classifications have been used by AIMS Long-Term Monitoring Program (LTMP) since 1985 and we consider it confusing to propagate use of these terms in the inshore context. Rather we propose to simply consider the design replicate locations across the evident water quality gradient.

The sampling methods were chosen to replicate 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 Reef 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.

Specific objectives of the program were:

- To establish and document long-term monitoring sites at which to assess the condition of coral communities within the Southern Inshore Zone at; Pine Peak Island, Pine Islets, Henderson Island, Connor Island, Temple Island, and Aquila Island.
- To identify key environmental factors influencing coral community condition.
- To provide the Healthy Rivers to Reef Partnership with report card scores for coral communities for the Southern Inshore Zone.

3 METHODS

3.1 Sampling Design

The physical environment experienced by corals at a given location are described by a combination of depth, aspect of a reef relative to prevailing weather conditions, and the location of a reef along the steep gradient in water quality within the inshore Great Barrier Reef. Depth and exposure to wind-driven waves determine the exposure of corals to pressures associated with suspended particles (Wolanski *et al.* 2005). Light, required for coral's autotrophic acquisition of energy, attenuates exponentially with depth at a rate proportional to turbidity (Van Duin *et al.* 2001; Storlazzi *et al.* 2015), while sedimentation increases as a function of suspended sediment concentration, particle size, and turbulence (Storlazzi *et al.* 2015). Locational differences may also influence the risk of exposure to acute disturbances such as cyclones and flooding.

Fringing reef development around islands within the Southern Inshore Zone is limited (Van Woesik and Done 1997, Cheal *et al.* 2001, Sea Research 2018). Kleypas (1996) and Cheal *et al.* (2001) describe three types of reef structure found in the Northumberland Group; fringing reefs (attached to islands with a well-defined emergent reef flat), incipient reefs (short slopes with carbonate accretion but no developed reef flat), and coral communities (individual coral colonies growing on igneous rock usually dominated by macroalgae). Where possible, monitoring sites were selected on fringing or incipient reefs. The initial selection of reefs was guided by surveys undertaken by Cheal *et al.* (2001) and Sea Research (2018). In practice, availability of fringing or incipient reefs was limited at several locations. Minimal carbonate accretion was found at Temple Island, Aquila Island and the 2 m depth at Connor Island and monitoring locations span a mix of incipient reef and coral communities.

To capture the spatial heterogeneity in environmental conditions experienced by corals, monitoring locations were replicated with three positions along the gradient in water quality from the very turbid waters close to the coast through to the clearest waters some 80km offshore (Table 1). At each reef, two replicate sites separated by at least 150m were selected haphazardly from the surface with the only limitations being that they were positioned on areas of substrate suitable for corals. Within each site five transects of 20 metre length were constructed to follow the depth contour of the site in a clockwise direction from the start point. 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, Pine Islets, Henderson Island, and Connor Island as predicted by Navionics electronic charts on the day of site construction. At Temple Island and Aquila Island the reef slope transitioned to sand at 1-1.5m below LAT and as such transects were set at 1m below LAT only. A summary of the sampling design is presented as Table 1. 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.

Figure 1 Map showing islands selected of coral monitoring in the Southern Inshore Zone. More detailed map showing the location of sites at each reefs are found in Figure A 1 to Figure A 6.

Table 1 Sampling design

Island	Type	Aspect	Sites per location	Depths per site	20m transects per site and depth
Pine Peak Island	Fringing	North	2	2 m and 5 m	5
Pine Islets	Incipient	South	2	2 m and 5 m	5
Henderson Island	Incipient	West	2	2 m and 5 m	5
Connor Island	Incipient -coral communities	East	2	2 m and 5 m	5
Temple Island	Incipient -coral communities	East	2	1m only	5
Aquila Island	Incipient -coral communities	South - East	2	1m only	5

3.2 Sampling methods

3.2.1 Photo point intercept transects

Benthic cover was estimated using photo point intercept transects (PPIT, Jonker *et al.* 2008). Along the upslope side of each transect line digital images of the substrate were taken at ~40cm elevation at 50cm intervals. Benthos beneath 5 evenly spaced points on each image was identified to the finest taxonomic resolution possible; typically genus level for corals and larger algae. A total of 32 images were analysed from each transect. Identifications for each point were entered directly into a data entry front-end to an Oracle® database, developed by AIMS. This system allows the recall of stored transect images. For data quality assurance all identified points were checked by a second observer.

3.2.2 Juvenile coral surveys

The number of juvenile coral colonies were counted *in situ* along the permanently marked transects. Corals in the size classes: 0-2cm and >2-5cm found within a strip 34cm wide (data slate length) positioned on the upslope side of the transect line were identified to genus level and recorded. Importantly, this method aimed to record only those small colonies assessed as juveniles, i.e. 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 20m transect a search was conducted within a 2m 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 seastars, 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.* 2018). 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 scored is based on the deviation in community composition beyond baseline condition confidence intervals. The estimation of confidence intervals in 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 develops. This section provides an overview of the rationale for the selection of the three indicators used to assess coral community condition in 2019. A full description of these and the additional indicators can be found in Thompson *et al.* (2018).

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 coral 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 estimated as the proportion of the total cover of algae 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 density 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.

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 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 Proportion 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 GBR. 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 2m	0.2%	3.4%
	Pine Peak 5m	0%	6.3%
	Pine Islets 2m	0.2%	5.4%
	Pine Islets 5m	0%	6.4%
	Henderson 2m	0.2%	3.9%
	Henderson 5m	0%	6.7%
	Connor 2m	0.2%	12.1%
	Connor 5m	0.2%	10.3%
	Temple 1m	0.3%	23%
	Aquila 1m	0.3%	23%
Juvenile density	All	13 m^{-2}	0 m^{-2}

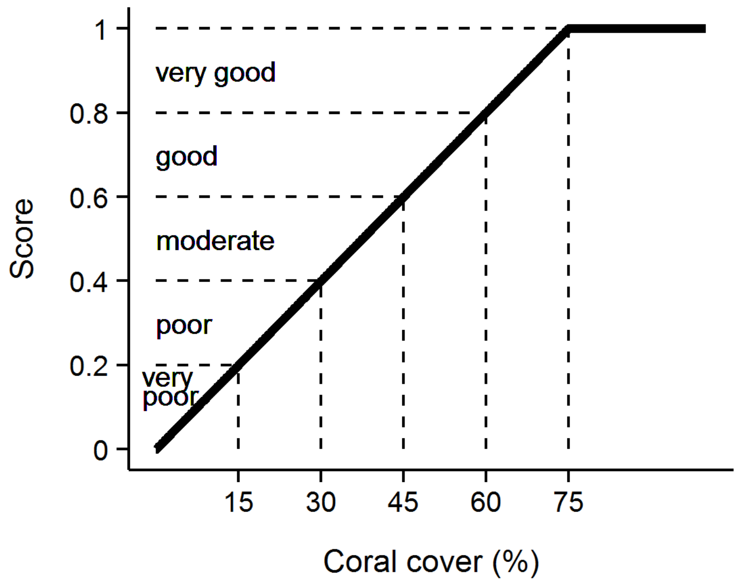


Figure 2 An example of a scoring diagram, here for the Coral Cover metric. Numeric scores and associated condition classifications based on observed coral cover are presented (see also Table 3).

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 classification 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

3.3.6 Data analysis

Differences in benthic community composition between monitoring locations, and other inshore Great Barrier Reefs, were explored using unconstrained principle coordinates analysis (Gower 1966) based on Bray-Curtis dissimilarities (Beals 1984) estimated from square-root transformed percent cover estimates of corals and identifiable algae genera. This analysis was done using R software (Core Team 2016) and the vegan package (Oksanen *et al.* 2016).

3.4 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.

3.4.1 Thermal bleaching

Thermal stress, resulting in coral bleaching is an increasing threat to coral communities in a warming world (Schleussner et al. 2016). During initial survey in January and May 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 (2m and 5m), and Aquila Island (1m). These loggers will be retrieved and replaced annually and have begun the process of recording an accurate in-situ climatology experienced by coral communities at each island. Once in-situ climatology has been described (~ 5 years) deviations from summer mean monthly maximum temperatures (DHW) will be included as an additional measure of thermal stress. In the interim, thermal anomalies expressed as degree hearting days (DHD) downloadable from ReefTemp (Garde *et al.* 2014) as published by the Bureau of Meteorology allow the identification of atypically warm periods likely to lead to coral bleaching. For each island, waypoints were selected in open water approximately 2 km out from the reefs (Table 4). These waypoints served as the central locations for a set of nine pixels from which annual DHD and a time-series of monthly anomalies were downloaded. DHD are the sum of positive daily temperature anomalies across the period 1st December to the 31st March, whereas monthly anomalies are the mean daily anomaly for a given month; both estimates were based on the 14 Day IMOS climatology (Garde *et al.* 2014). Mean values of DHD and monthly anomalies were estimated as the average of the values from the nine pixels at each location. 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), though the pattern of warming and individual tolerances of species will add variability to these thresholds.

Table 4 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
Conner Island	-21.6957	149.67
Temple Island	-21.6239	149.5132
Aquila Island	-21.9428	149.5535

Table 5 Annual degree heating days (DHD) calculated for reefs across the coastal shelf from 'outer' to 'inner'. Comparison with Dunk Island, a nearshore island at which severe bleaching although limited mortality was observed in 2017.

Year	Pine Peak	Pine Islets	Henderson	Connor	Temple	Aquila	Dunk North	Dunk South
2002	0	0	0	0	0	0	0	0
2003	3	0	9	11	20	11	22	23
2004	63	34	66	58	64	72	34	33
2005	21	13	20	38	56	44	20	32
2006	63	64	75	74	65	75	37	38
2007	6	9	8	10	18	21	9	9
2008	13	9	9	14	17	16	33	25
2009	44	56	44	57	62	74	22	27
2010	46	47	48	54	43	23	39	45
2011	24	23	18	24	28	47	37	24
2012	25	23	35	35	36	33	17	12
2013	30	34	34	42	47	43	49	47
2014	5	15	6	7	22	16	2	6
2015	72	80	91	105	115	112	60	67
2016	74	61	77	81	90	84	43	58
2017	120	124	121	118	145	140	115	131
2018	47	54	46	52	81	73	42	45
2019	28	24	28	47	57	50	62	42

Annual DHD estimates indicate 2017 and to a lesser degree 2015 as years during which the corals in the Southern Inshore Zone were likely to have experienced thermal stress (Table 5). In both 2015 and 2017 the reefs closer to the coast, Aquila and Temple were exposed to greater temperature anomalies, presumably due to the shallower water depth and so lower mixing with deeper cooler waters. Included for comparison are annual totals from Dunk Island, an inshore island off the Tully

River catchment in the Wet Tropics region where corals the DHD estimates for 2017 resulted in severe bleaching although limited coral mortality (Thompson *et al.* 2018). Sea Research (2018) visited sites in this area during October 2017, including Pine Peak and Henderson islands, and reported only sparse bleaching among a few coral colonies. By October it is likely that most corals impacted by thermal stress in the previous summer would have either died or recovered.

A comparison with other Natural Resource Management (NRM) (sub) regions shows the Southern Inshore reporting region has experienced a similar DHD regime to Whitsunday Inshore reporting region. Unfortunately, the impact of bleaching during 2017 in the Whitsunday Inshore region was largely obscured by the damage caused by Cyclone Debbie, however, there was little evidence of coral mortality at reefs sheltered from cyclone driven seas (Thompson *et al.* 2018).

Table 6 Comparison of mean annual degree heating days (DHD) among NRM (sub)regions inshore reefs. The Southern Inshore Zone and the year 2017 are highlighted for ease of comparison.

Year	Barron Daintree	Johnstone Russell-Mulgrave	Tully Herbert	Burdekin	Whitsunday	Southern Inshore	Fitzroy
2002	0	0	0	0	0	0	0
2003	13	8	18	20	3	9	3
2004	23	36	38	42	54	60	13
2005	23	18	20	34	21	32	33
2006	24	33	38	40	40	69	17
2007	14	10	9	9	6	12	2
2008	29	19	26	21	14	13	NA
2009	34	26	24	40	38	56	0
2010	25	31	40	32	43	44	31
2011	32	31	37	30	23	27	54
2012	19	27	26	48	38	31	22
2013	44	35	47	48	39	38	30
2014	4	5	4	10	10	12	16
2015	39	35	58	78	76	96	91
2016	35	40	46	87	74	78	61
2017	101	115	121	132	131	128	120
2018	41	30	40	56	43	59	78
2019	41	43	50	52	30	39	43

3.4.2 Runoff

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). As a generalisation, the presence of coral communities can be interpreted as direct evidence that 'typical' salinity levels do not pose a threat to coral communities; it is deviations to levels below 28ppt that begin to cause coral mortality (Berkelmans *et al.* 2012). The level of discharge from local rivers also relates to the loads of sediments and nutrients entering the marine environment to which the recovery rate of coral communities shows a negative relationship in some inshore areas (Thompson *et al.* 2019). As a first step in assessing the likelihood that floods may have directly caused salinity-related stress to corals, or indirectly reduce coral health, the seasonal discharge of local rivers is compared to long term median flows. Median discharge for the "water year" (1st October through to 31st September) are 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.* (in prep). 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). While the Pioneer River is the major contributor among adjoining catchments in this area, models show flows are predominately northwards away from of the Southern Inshore Zone, although the flows from the coastal streams south of Mackay in March 2017 did extend to Temple and Aquila Islands (eReefs model on-line).

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.

<table>
<tr><th>Basin</th><th>Gauge Station_Id</th><th>LT median (ML)</th><th>2011</th><th>2012</th><th>2013</th><th>2014</th><th>2015</th><th>2016</th><th>2017</th><th>2018</th><th>2019</th></tr>
<tr><td>Pioneer</td><td>124001B</td><td>692,342</td><td>5.2</td><td>2.3</td><td>1.7</td><td>0.9</td><td>0.2</td><td>0.9</td><td>2.0</td><td>0.4</td><td>1.7</td></tr>
<tr><td>Plane</td><td>126001A, 126003A</td><td>309,931</td><td>4.1</td><td>2.5</td><td>1.7</td><td>0.7</td><td>0.2</td><td>0.8</td><td>2.5</td><td>0.2</td><td>1.1</td></tr>
<tr><td>Styx</td><td rowspan="3">129001A</td><td rowspan="3">381,986</td><td rowspan="3">4.8</td><td rowspan="3">1.5</td><td rowspan="3">5.2</td><td rowspan="3">2.9</td><td rowspan="3">2.0</td><td rowspan="3">1.8</td><td rowspan="3">2.7</td><td rowspan="3">1.4</td><td rowspan="3">0.7</td></tr>
<tr><td>Shoalwater</td></tr>
<tr><td>Waterpark Creek</td></tr>
</table>

3.4.3 Cyclones and storms

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 (http://www.bom.gov.au/cyclone/history/tracks/index.shtml). In addition, wave height data from the Mackay buoy (Mackay wave buoy page) was accessed to verify exposure to extreme wave heights associated with cyclones. Four of the top five wave heights recorded by the Mackay buoy since 1975 have occurred since 2010 and, in descending order, can be attributed to cyclones Dylan (2014), Ului (2010), Debbie (2017) and Iris (2018). While each of these cyclones are likely to have impacted coral communities in the Southern Inshore Zone, 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, crossing the coast at Shoalwater Bay on February 20th 2015 (Cyclone Marcia track). Although waves attributable to Cyclone Marica do not feature in the 10 highest waves recorded at either the Mackay or Hay Point wave-buoys, this can be explained by the track of the storm being to the south east of the buoys. Higher seas are expected to the south of cyclone tracks. Indeed, the fourth highest waves recorded at the Emu Park buoy can be attributed to this storm. 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.

3.5 Environmental setting of reefs

3.5.1 Ambient Water Quality

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).

Non-algal particulate (NAP) concentration, a proxy for total suspended sediments, and Chlorophyll *a* (Chl *a*) derived from the MODIS aqua satellite mounted sensor were downloaded from the Australian Bureau of Meteorology[2]. For each monitoring location a square of nine 1 km^2 pixels were identified in closely adjacent waters from which daily medians were used to estimate monthly means. Ambient conditions for NAP and Chl a were estimated as the reef level means over the period 2003 – 2019.

Within the Southern Inshore Zone both Chl *a* and NAP diminish seaward from the coast as the influences of riverine inputs and resuspension of sea bed sediments decline (Table 8). At Pine Peak, Pine Islets, and Henderson mean concentrations of Chl *a* and NAP are within the range observed at other inshore reefs monitored by the Marine Monitoring Program (Figure 3). Concentrations at Connor and Temple islands are relatively high, with Aquila Island having the highest Chl *a* and NAP concentrations across all reefs monitored. Maintaining such high NAP values suggests tidal and wave driven resuspension strongly influence the physical environment at these nearshore reefs. Broad Sound has the largest tidal range on Australia's east coast (~9m maximum tidal range at McEwin Islet) and this undoubtedly contributes to the observed high turbidity. The ambient levels of Chl *a* exceed GBRMPA guidelines (Anon 2010) of 0.45 ugL^{-1} at Temple and Aquila islands, with NAP concentrations at Aquila Island also above the guideline of 2 mgL^{-1} set for total suspended solids.

Table 8 Ambient water quality conditions. Values represent the mean concentrations estimated from satellite imagery over the period 2003-2019. Values in bold exceed GBRMPA guidelines.

Reef	**Chl a** (ugL^{-1})	**NAP** (mgL^{-1})
Pine Peak Island	0.30	0.84
Pine Islets	0.31	0.90
Henderson Island	0.35	0.91
Connor Island	0.38	1.96
Temple Island	**0.46**	1.74
Aquila Island	**0.53**	**3.59**

[2] [2] *Marine water quality indices produced by the Australian Bureau of Meteorology as a contribution to eReefs - a collaboration between the Great Barrier Reef Foundation, Australian Government, Bureau of Meteorology, Commonwealth Scientific and Industrial Research Organisation, Australian Institute of Marine Science and the Queensland Government. Data are acquired from NASA spacecraft. http://www.bom.gov.au/marinewaterquality/. Although the confidence in individual estimates of Chl a in turbid inshore waters is low the time averaged conditions do describe gradient that correspond to differences in benthic communities.*

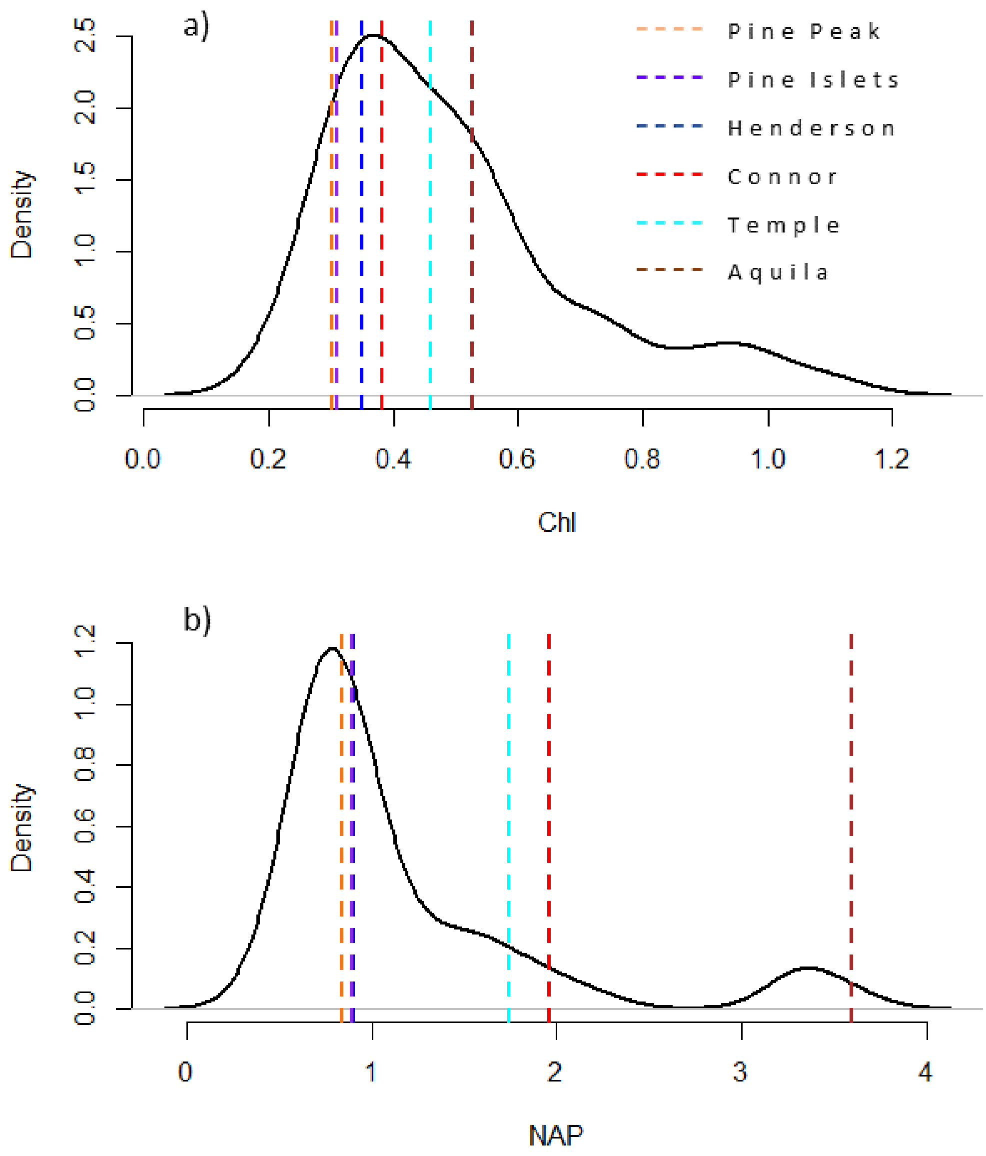


Figure 3 Ambient water quality conditions a) Chlorophyll *a* and b) Non- algal particles (NAP). Black lines show the distribution of satellite derived water quality conditions (mean 2003-2019) for nearshore reefs of the GBR included in AIMS LTMP and MMP. Coloured lines indicate the relative position of target reefs within each distribution as per Table 8.

3.5.2 Sediment characteristics – Hydrodynamic setting

As a proxy for the hydrodynamic setting of a site the composition of sediments is a useful covariate to consider in terms of coral community dynamics (Wolanski et al. 2005). Higher proportions of fine clay and silt sized particles in the sediment identify sites more prone to sediment accumulation than those with course grained sediments. In areas with high levels of suspended sediments, such as Broad Sound, a high proportion of fine grained particles will indicate high rates of sedimentation. It should be noted however, that high rates of sedimentation may not be fully captured by grainsize as the flux of fine sediments may alternate strongly from accumulation of fine material during calm conditions to removal under exposure to wave-driven or tidal flows.

At each site six small samples of surface sediments were collected. Sampling was conducted using a 100mm syringe tube that had had the restricted end removed. The open tube was plunged into deposits of sediment > 20mm deep encountered along the benthic transects and plugged to capture an undisturbed core of sediment. The top 10mm fraction from each core was kept and combined into a single sample from each site. These samples were then sieved through a 63µm sieve and the proportion of clay and silt sized particles in the sample determined by dry weight of the portions retained and passed through the sieve. At four islands samples were collected along the 5 m transect sites, conforming to the sampling design for inshore reefs monitored under the MMP. At Temple and Aquila islands the reef slope did not extend to this depth and samples were taken from the shallower 1m sites.

Table 9 Sediment composition at monitoring locations. Values indicate the percentage of total weight of sample that passed through a 63 um sieve.

Reef	Site	Depth (m below LAT)	Proportion of sample passed through 63 µm sieve (%)	
			2019	Mean
Pine Peak Island	1	5	27.65	18.04
Pine Peak Island	2	5	8.44	
Pine Islets	1	5	9.96	7.84
Pine Islets	2	5	5.71	
Henderson	1	5	21.34	20.82
Henderson	2	5	20.29	
Connor Islet	1	5	10.59	11.85
Connor Islet	2	5	13.1	
Temple Island	1	1	4.42	4.54
Temple Island	2	1	4.65	
Aquila Island	1	1	26.01	19.78
Aquila Island	2	1	13.54	

4 RESULTS

4.1 Coral community condition assessment

The overall coral index score for the Southern Inshore Zone in 2019 was graded as E categorising the coral communities as being in very poor condition (Table 10 & Table 11). This overall index score does however mask the substantial differences in the condition of coral communities between reefs. The index scores were lowest at the 2 m depths of Pine Islets and Pine Peak Island - index scores of 0.04 and 0.05 respectively. The condition of coral communities at these islands contrasted with Henderson Island where coral communities were in moderate condition (score 0.41) at the 2 m depth (Table 12). Averaged over depths, the index scores for all Southern Inshore reefs were below the median score observed across all other inshore reefs monitored by the LTMP and MMP (Table 12). Consistently minimum scores of zero for the macroalgae indicator were highly influential in the low grade for this inshore zone.

Table 10 Indicator values for Southern Inshore Zone. Juvenile densities are corrected for area of algal covered substrate, as a potential area for colonisation.

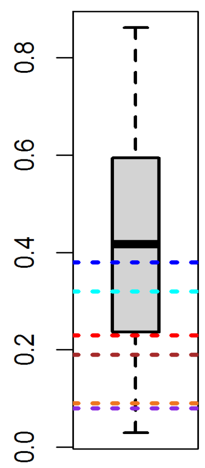
	Year	Juvenile Density (per m^2)		Coral Cover (%)		Macroalgae proportion (%)	
		Mean	SD	Mean	SD	Mean	SD
Regional summary	2019	1.48	0.9	36.39	23.53	62.41	22.96

Table 11 Indicator scores for Southern Inshore Zone.

	Year	Juvenile corals	Coral Cover	Macroalgae	Report Card	
					Score	Grade
Regional Scores	2019	0.13	0.49	0	0.20	E

Table 12 Index grade and scores for each reef and depth combination. Figure to the right indicates reef level indicator scores (mean across depths) from this study (colour coded horizontal lines) relative to the distribution of scores observed on inshore reefs monitored by the MMP and LTMP (Thompson *et al.* in prep).

Reef	Depth	Grade	Index	Indicator lines
Pine Peak	2	E	0.05	
	5	E	0.12	
Pine Islets	2	E	0.04	
	5	E	0.12	
Henderson Island	2	C	0.41	
	5	D	0.36	
Connor Island	2	D	0.21	
	5	D	0.24	
Temple Island	1	D	0.32	
Aquila Island	1	E	0.19	



4.2 Coral Cover

Coral Cover scores are based on the combined cover of hard and soft corals. The reefs furthest from the coast, Pine Peak Island and Pine Islets, had the lowest scores for the Coral Cover indicator (Table 13). At both these locations coral cover was very low at the 2 m depths and, while higher, the coral cover at 5 m depths of both reefs was still within the 15 % to 30 % range classified as "Poor". Averaged over depths the coral cover scores at Pine Peak Island and Pine Islets was in the lower 25th percentile of reefs monitored across the inshore GBR (Table 13). Unsurprisingly given the very low cover, the diversity of hard corals was low at the 2 m sites with the community dominated by *Porites* at Pine Peak Island and a mix of *Acropora*, *Montipora* (family Acroporidae) and *Turbinaria* (family Dendrophylliidae) at Pine Islets (Table A 2, Figure A 7). Other than *Acropora* at Pine Islets, these same genera were also most common among the more diverse communities at the 5 m depths (Table A 2). The reef slope at Pine Peak Island was surveyed in 2000 (Cheal *et al.* 2001) when hard coral cover was 18%, approximately double that observed at the similar depth (5 m) in 2019. The cover of soft corals was very low at Pine Islets (Table 13). At Pine Peak Island (Table 13, Table A 2) the low cover of soft corals was dominated by *Briareum* and to a lesser degree *Sinularia,* similar to that reported by Cheal *et al.* (2001).

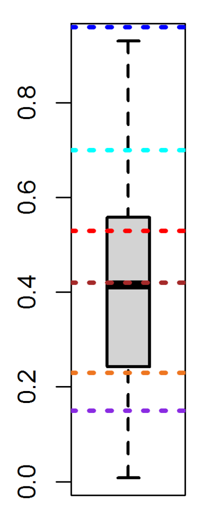
The coral cover at Henderson Island (Table 13) contrasted starkly with that described above for the tow reefs further offshore. Coral Cover scores at Henderson Island were very high, both in comparison with other reefs in the Southern Inshore Zone and the Inshore GBR in general (Table 13, Figure A 8). The depth-averaged hard coral cover at Henderson was over 50%, with the community dominated by *Acropora* that represented 94% of corals at 2 m and 69% at 5 m (Table A 2). In combination with moderate cover of soft corals, predominantly *Klyxum*, the Coral Cover score here was classified as 'Very Good' for both 2 m and 5 m depths. The sites at Henderson Island were established by Sea Research in October 2017 at which time hard coral cover was recorded as approximately 60 % at 2 m and 52 % at 5 m (Sea Research 2018), very similar to that observed in January 2019. Again, *Acropora* was the dominant genera at both depths in 2017.

The Coral Cover scores at Connor Island were 'Satisfactory' at both depths (Table 13). The marked increase in NAP concentration at Connor Island compared to the islands further offshore is reflected in the coral community where large colonies of laminar hard corals *Montipora* and *Turbinaria* and the soft coral genus *Sinularia* dominate the community (Table A 2, Table A 3).

Closer to the coast Temple and Aquila islands have similar hard coral cover estimates (Table 13). The high cover of soft corals at Temple Island elevates the coral cover score into the 'Good' category compared to the 'Satisfactory' score observed at Aquila Island. Both reefs have hard substrate in shallow water with the reef slopes running to the surrounding loose substrate below the ~1 m below LAT depth of the monitoring transects. The very high turbidity at Aquila Island is reflected in the coral community with large colonies of foliose *Montipora* (Table A 2, Figure A 9). The coral community at Temple Island also includes a high proportion of *Montipora* however other hard corals, including *Turbinaria* and *Acropora*, and the soft corals *Sinularia* and *Xenia*, contribute to a more diverse community (Table A 2, Table A 3, Figure A 9).

Table 13 Coral cover and indicator scores for each location. Figure to the right indicates reef-averaged Coral Cover indicator scores from this study (coloured coded horizontal lines) relative to the distribution of scores observed at inshore reefs monitored by the LTMP and MMP (Thompson *et al.* in prep).

Reef	Depth	Hard coral cover (%)	Soft coral cover (%)	Coral cover (%)	Coral cover Score	Indicator lines
Pine Peak	2	3.45	7.15	10.60	0.14	
	5	9.39	14.96	24.35	0.32	
Pine Islets	2	2.69	1.38	4.06	0.05	
	5	14.75	3.94	18.69	0.25	
Henderson Island	2	57.05	17.9	74.96	1.00	
	5	48.75	19.19	67.94	0.91	
Connor Island	2	22.77	12.9	35.66	0.48	
	5	33.06	10.25	43.31	0.58	
Temple Island	1	19.50	33.13	52.63	0.70	
Aquila Island	1	20.75	11.00	31.75	0.42	

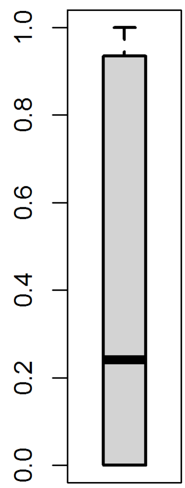


4.3 Macroalgae proportion

The proportion of macroalgae amongst the total cover of algae was above the maximum threshold (Table 2) resulting in the minimum score of zero for this indicator (Table 14). The cover of macroalgae was extremely high at both Pine Peak Island and Pine Islets with the community dominated by large brown algae of the genus *Sargassum* in the shallows with increasing presence of *Lobophora* at the 5 m depths (Table A 4, Figure A 8). *Sargassum* was also the dominant macroalgae at both Temple and Aquila islands where Chl *a* concentration exceeds guideline levels for inshore waters (Table 8). Interestingly, for Aquila Island, while the high NAP level (Table 8) must drastically reduce light levels during spring high tides, the shallow depth of this reef allows sufficient light to maintain these algae. The influence of high turbidity, and so reduced light, likely contributes to the higher representation of red algae at Connor Island (Table 8, Table A 4). Although the total cover of macroalgae is low at Henderson Island due to the high cover of corals, macroalgae do occupy approximately a third of the limited substrate available to coral recruitment (Table 14). As with the reefs further offshore the most common macroalgae at the 5 m depth at Henderson Island was *Lobophora*.

Table 14 Macroalgae cover and indicator scores for each location. Figure to the right indicates the distribution of macroalgae indicator scores observed at inshore reefs monitored by the LTMP and MMP (Thompson *et al.* in prep). As the macroalgae scores from this study were all zeros they were not indicated on this figure.

Reef	Depth	Macroalgae cover	Macroalgae proportion	Macroalgae score
Pine Peak	2	77.86	88.44	0
	5	51.69	74.46	0
Pine Islets	2	89.75	94.69	0
	5	48.69	65.79	0
Henderson Island	2	5.76	23.77	0
	5	8.81	36.48	0
Connor Island	2	37.33	61.38	0
	5	18.75	38.42	0
Temple Island	1	27.19	69.9	0
Aquila Island	1	32.31	70.79	0

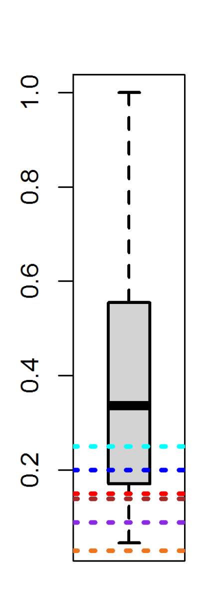


4.4 Juvenile density

Across the Zone the density of juvenile hard corals was classified as 'Very Poor' (Table 11). The density of juveniles at Aquila Island, Connor Island, Pine Islets and Pine Peak Island were below the 25th percentile of the most recent densities observed at other inshore reefs monitored by the LTMP and MMP (Table 15). The highest densities were observed at Temple Island and the 2 m depth at Henderson Island, although these were below median levels observed elsewhere in the inshore Great Barrier Reef and categorised as 'Poor' (Table 15). The lowest densities of juvenile hard corals were observed at Pine Peak Island and Pine Islets where macroalgae cover was at very high levels (Table 14). The lowest diversity of hard coral juveniles, measured as numbers of genus observed, was Pine Peak Island (Table A 5), although this value was limited by the very low abundance of juveniles observed. Corals of the family Acroporidae (Genus Acropora and Montipora) are often fast growing and have the potential to provide for rapid recovery. These fast-growing genera were not common at any of the reefs surveyed (Table A 5, Figure A 7). Bolstering juvenile densities at Temple Island was the genus Turbinaria (Family Dendrophylliidae, Figure A 7), this genus has been observed to recruit in very high densities following disturbances at other inshore reefs, however survival through to adults can be low (Thompson et al. 2019).

Table 15 Juvenile hard coral abundance, density and indicator scores for each location. Density has been adjusted for the area of algal covered substrates. Figure to the right indicates reef-averaged Juvenile indicator scores from this study (coloured coded horizontal lines) relative to the distribution of scores observed at inshore reefs monitored by the LTMP and MMP (Thompson *et al.* in prep).

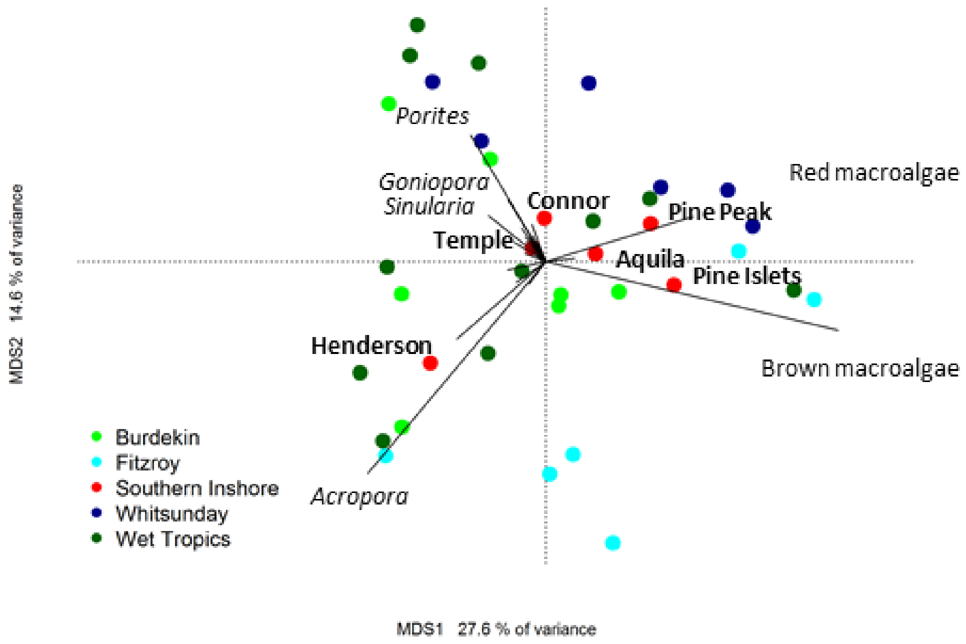
Reef	Depth	Juvenile abundance	Juvenile density (per m^2)	Juvenile score
Pine Peak	2	7.5	0.25	0.02
	5	8	0.33	0.03
Pine Islets	2	27	0.83	0.07
	5	28	1.11	0.1
Henderson Island	2	21	2.49	0.22
	5	14.5	1.95	0.17
Connor Island	2	34	1.69	0.15
	5	27.5	1.67	0.15
Temple Island	1	39	2.85	0.25
Aquila Island	1	24.5	1.57	0.14



4.5 Pressures noted during surveys

Few incidences of recent or ongoing mortality of corals were observed during baseline surveys. The most frequent damage was observed at Henderson Island where disease and small patches of mortality due to 'unknown cause' on *Acropora* colonies was the most common cause of partial mortality (Table A 6). The higher level of damage observed at Henderson Island is primarily due to the far greater number of corals compared to other sites but also the dominance of *Acropora*, a genus typically showing higher incidence of damage than most others. Disease was also noted on *Montipora* colonies especially at Pine Islets (Table A 6) and given the low cover is of more concern than the levels observed at Henderson Island. Sediment deposits on *Montipora* at Aquila Island were common (Figure A 9c, d) however these were not causing obvious damage. In general, the corals were mostly healthy at the time of survey.

As noted by previous studies (Kleypas 1996, van Woesik & Done 1997) the coral communities in the Southern Inshore Zone have clearly been shaped by the high turbidity in the region, however the communities remain within the range of communities observed in other inshore regions (Figure 4).



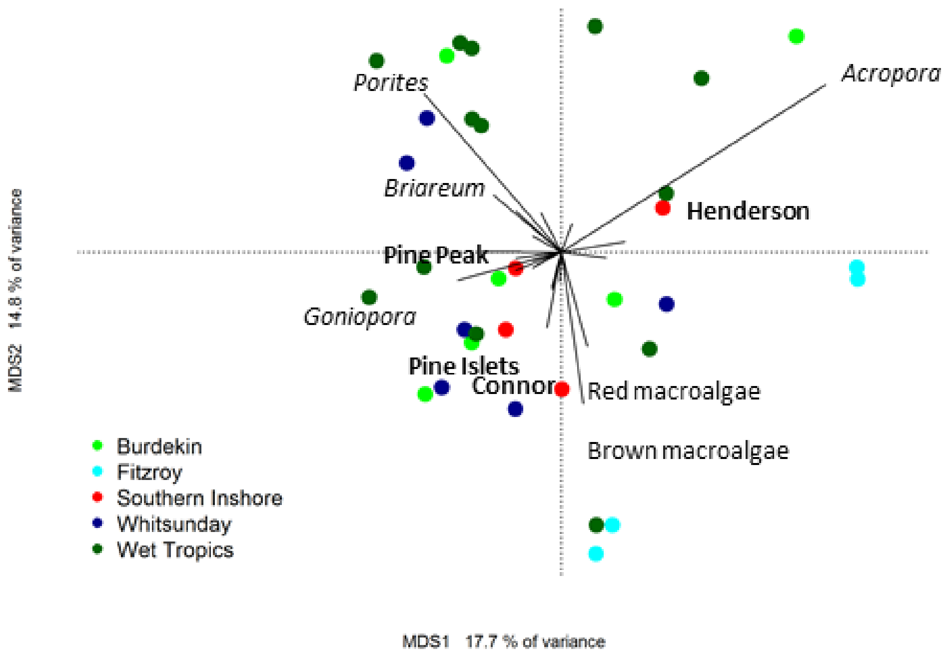


Figure 4 Variability in benthic communities. Ordination biplots of inshore reef communities at 1 or 2 m (top) or 5 m depths (bottom). Plots represent a two-dimensional projection of the variability in communities based the cover of hard and soft coral genera and major algal groups. Data from MMP NRM Regions and the Southern Inshore Zone reefs are categorised by colour. Only Southern Inshore Zone reefs are named. Vectors indicate relative abundance of genera or algal groups with those most influential (longest) named. For example, Henderson Island communities at both depths are distinct from other Southern Inshore Zone reefs due to higher cover of *Acropora*.

4.6 Logistical considerations

The are several environmental constraints that need to be considered when planning the resurvey of the reefs in the Southern Inshore Zone.

The Broad Sound-Shoalwater Bay area has the highest tidal range along the Queensland coast. Surveys should be timed to coincide with neap tide periods to reduce the risk of strong currents and elevated turbidity. The baseline surveys were all undertaken during neap tides (generally < 3m change between high and low tide over the period of survey). It was only at Aquila Island, on a rising tide with a tidal range of 2.6 m, that excessive currents were encountered. Wind driven resuspension can also reduce in-water visibility. Winds in excess of 15 knots at Conner Island and Temple Island, and at 10 knots at Aquila Island, despite coinciding with neap tides, resulted in water visibility levels below 1 m at which point surveys are impossible to complete.

The proximity of the survey locations in relation to coastal access points is a further consideration. Access to Temple, Aquila and Connor Islands is most convenient via Carmila Creek. This requires ~3.5m of tide at McEwen Island (Bureau of Meteorology Tide Predictions). In combination with the need to survey during periods of neap tides and low winds severely restricts the availability of suitable periods within which to undertake sampling. The more offshore reefs are less restricted as during neap tides these reefs were all successfully surveyed with winds in the range of 15 to 20 knots from the East. However, the most accessible launch point for these 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 would be required. Table 16 provides a reference point for the conditions experienced during baseline surveys.

Based on the above considerations and observations during baseline work we recommend the limits specified in Table 16. for coral monitoring activities. The setup and survey of sites at both Temple Island and Aquila Island in April was compromised by very poor (<1m) in water visibility that limited our ability to both locate suitable habitat and undertake surveys, winds were in the range of 10-15 knots, which was higher than forecast the previous day. On returning in May during very light winds 0-5 knots conditions at Temple Island were good, with in-water visibility >4 m allowing repositioning and Resurvey of the site established in April and the setup and survey of a second suitable site. Despite, conditions of minimal tide and wind the in-water visibility at Aquila was again very poor 1-2m. We did manage to establish and survey the current Site 1 on suitable habitat, however Site 2 remains on substandard habitat being comprised of scattered corals interspersed with sandy patches. This site is also limited by depth with corals only extending to < 1m below lowest astronomic tide datum. We could not locate a more suitable site. It is possible that this site could be repositioned during the next survey as the 5th transect did encounter more consolidated coral habitat. Alternatively, it could be argued that a single site be maintained at this reef.

Table 16 Weather conditions during baseline works.

Reef	Date	Wind (knots)	Tide (Range)	Observations
Pine Peak Island	27/01/19	15-20 ESE	Low->Rising (3m)	Visibility 4-5m negligible current. Convoluted steep reef slope boarding extensive reef flat. 5m sites toward base of reef slope.
Pine Islets	28/01/19	15-20 E	Low->Rising (2.6m)	Visibility 5m negligible current. Convoluted steep reef slope habitat with limited reef flat. Some areas of incipient reef. 5m sites toward base of reef slope.
Henderson Island	29/01/19	20-25 ENE	High->Falling (3m)	Visibility 5m negligible current. Shallowly sloping reef slope with very limited or no reef flat development. Site 1 5m variable depth ~4-8m.
Knight Island	30/01/19	15-20 E	Falling (3m)	Searched available hard substrates along Western side of Island. South West suitable for a single 2m site boarding reef flat constructed of micro-atolls, No suitable second 2m site or 5m sites. Unsuitable for monitoring sites.
Connor Island	30/01/19	15-20 E	Low->Rising (3.2m)	Visibility 1-2m, negligible current. Site 1 5m depth incipient reef. Wave and visibility conditions precluded further work.
Connor Island	31/01/19	15 E	High->Falling (4m)	Visibility 2m, negligible current. Incipient reef, 2m sites very little carbonate substrate. Site 1 2m very convoluted as dictated by depth contour.
Temple Island	31/01/19	15-20 E	Low (4m)	Visibility <1m, rough conditions, unworkable.
Temple Island	27/04/19	15 SE	Falling (2.6m)	Visibility<1m, located and setup site 1. Searched for second site but conditions largely unworkable
Aquila Island	27/04/19	10-15 SE	Rising (2m)	Visibility <1m, set up survey a single site, unworkable for second site.
Temple Island	27/05/19	5 variable	Falling-> Low (3m)	Visibility 4-7m, negligible current. Repositioned and surveyed Site 1 initiated Site 2. Transects at ~1m as reef slope did not extent to 2m. Very little carbonate substrate, primarily coral communities on rock
Aquila Island	27/05/19	5 variable	Low->Rising (2.5m)	Visibility 1-2m, current became unworkable as tide rose. Site 1 incipient reef. Transects at 1m as reef slope did not extend to 2m. Site set up in April (now site 2) was repositioned in attempt to improve capture of coral community. Transects remain over a mix of scattered corals and rock on a sandy substrate and patches of incipient reef at 0-1m depth. Strong current precluded further efforts to improve position of transects, although habitat is limited.

5 DISCUSSION

The overall condition of Southern Inshore Zone reefs monitored in 2019 was categorised as 'very poor'. This categorisation reflects scores for three metrics that, in combination, have been formulated to represent not only reef state, but also processes that support reef resilience (Thompson *et al.* 2019). Relative to other inshore reefs monitored in the Great Barrier Reefs the scores for the Zone are low.

Previous studies of reefs in the Southern Inshore Zone have demonstrated limited development of carbonate substrates (coral reefs) compared to those laid down by corals in other areas (Hopley *et al.* 1983, van Woesik 1992, Kleypas 1996, van Woesik & Done1997). The logical conclusion reiterated by these authors was that the environmental conditions of the area are not suitable for sustained coral reef development. Low light levels due to high turbidity and large tidal amplitudes were primarily implicated in the selection of corals with thin, encrusting or foliaceous growth forms, rather than the more robust massive or extensive branching morphologies necessary for reef development (van Woesik & Done 1997). Our results from Connor, Aquila, and Temple islands, show that where coral communities were dominated by laminar growth forms of *Montipora*, and *Turbinaria*, reef development was minimal. Ambient conditions were highly turbid, consistent with previous assessments.

At the more offshore reefs of Henderson Island, Pine Islets and Pine Peak Island, high turbidity alone cannot explain the low scores observed. At Henderson Island hard coral cover was high and included a high proportion of the genus *Acropora* along with a variety of massive corals, the taxa reported as important for reef development. The very high cover of brown macroalgae, that require light to thrive, at the 5 m depths of the most offshore reefs, in combination with lower estimates of non-algal particulates, all suggest turbidity is not the primary driver of poor condition at these reefs. The large reef flat at Pine Peak Island also suggests that corals have flourished in that location historically. Coral communities are, however, naturally dynamic, existing in a cycle of disturbance and recovery (Connell *et al.* 1997). Variability in the coral cover indicator scores may imply that exposure to disturbances has varied among reefs. Conversely, low scores for the macroalgae and juvenile indicators suggest potential bottlenecks in the recovery process.

The lack of detailed historical data from the locations monitored limit the ability to assess the impacts of past disturbances. High wave events measured at the Mackay wave buoy over the last decade, along with the close passage of Cyclone Marcia suggest it is very likely some loss of coral cover at reefs open to storm driven waves will have occurred. Although reefs in the area were exposed to unusually warm waters over the 2016/17 summer, heat stress was similar to that observed in the Whitsunday Islands and Burdekin Region where, although corals did bleach, mortality was not high (Thompson *et al.* 2018). The only location for which we have a solid baseline for comparison is Pine Peak Island. Cheal *et al.* 2001 recorded hard coral cover of ~16% (3 m) and 18% (6 m) in September 2000, considerably higher than 3.5% (2 m) and 9.4% (5 m) recorded in January 2019. This loss of cover does imply exposure to disturbance. That the cover of *Acropora* at Pine Peak Island, a genus both sensitive to thermal stress (Marshal and Baird 2000) and physical damage during cyclones (Fabricius *et al.* 2008), is high at nearby Henderson Island suggests exposure to waves at Pine Peak Island, a site with a more exposed aspect, rather than coral bleaching, was the more likely cause of coral loss.

The very low scores for the macroalgae and juvenile indicators strongly suggest that recovery from disturbance in this area is likely to be limited by low recruitment of corals to replace those killed. Large fleshy macroalgae such as *Sargassum* and particularly matt forming species such as *Lobophora* are known to disrupt coral community recovery (Hauri *et al.* 2010). Macroalgae have been shown to reduce: gamete development in adult corals, and the settlement, survival and growth of juvenile corals (reviewed by Birrell *et al.* 2008, Foster *et al.* 2008, Diaz-Pulido *et al.* 2010). At the levels of macroalgae cover observed in 2019 it is reasonable to conclude that macroalgae have contributed to the poor scores for the juvenile density indicator. Even at Henderson Island where hard coral cover is high, consideration of the high proportion of macroalgae in the algal community helps to explain the low density of juvenile hard corals observed, and suggests, should this location be exposed to an acute disturbance, recovery may be slow.

The January surveys at Pine Peak Island and Pine Islets, and to a lesser extent Connor and Henderson islands, may have resulted in a negative bias in index scores. The biomass of *Sargassum* can vary seasonally with peaks in late summer (Vuki & Price 1994). The timing of surveys is likely to have intersected with higher cover of this algae than had surveys occurred later in the year. However, given the magnitude of exceedance of threshold values for the macroalgae indicator along with the potential for other, more ephemeral, algae to take advantage of reducing biomass of *Sargassum* it is unlikely scores for the macroalgae indicator would have substantially improved. No such seasonality has been described for *Lobophora*. High cover of *Sargassum* will have obscured some live coral from point intercept surveys as the algae can over-top small and encrusting corals. This over-topping will have reduced scores for coral cover although we consider these reductions would be minor as coral cover beneath *Sargassum* was not obvious at the time surveys. High cover of macroalgae also has the potential to reduce the detection of juvenile corals, but as macroalgae tends to limit juvenile coral settlement and survival, and efforts were made to look beneath algae, again we do not consider this will have caused a substantial bias. Comparing our estimate of macroalgae cover at Pine Peak Island with that of Cheal *et al.* 2001 suggests an increase from ~36% (6 m) to 52% (5m); although at this point the increase cannot be separated from seasonality.

Macroalgae are not the only pressure likely to be influencing the observed low density of juvenile hard corals. Direct effects of high concentrations of suspended sediments can reduce fertilisation (Ricardo et al. 2016) whereas the accumulation of sediments on the substrate can preclude larval settlement (Ricardo et al. 2017). Kleypas (1996) showed a strong correlation between suspended sediments and tidal range in Broad Sound. This variable turbidity suggests periodic fluxes of sedimentation and resuspension processes. Indeed, patches of fine sediment overlaying live coral tissue were common at Aquila Island at the time of surveys that coincided with neap tides and low winds, conditions favouring sedimentation rather than resuspension (Wolanski *et al.* 2005). The regional low cover of hard corals is also likely to incur a feedback of lower larval supply.

Accessibility, weather and tidal conditions impose severe constraints on when monitoring can be scheduled. The high tidal range in the area dictates that surveys are planned to coincide with neap-tide conditions. In addition, there is a requirement for light winds. The offshore reefs at Pine Islets and Pine Peak Island are some 80 km from shore dictating wind conditions lower than 15 knots for comfortable access via a trailered vessel. The inshore reefs require winds below 10 knots to limit wave driven resuspension leading to high turbidity. Additionally, access to Aquila, Temple and Connor Islands is most convenient via Carmila Creek, that becomes impassable within approximately 2 hours either side of low tide.

The problem of minimum visibility to conduct the surveys was highlighted at Aquila Island where the turbidity levels experienced, despite neap tides and calm winds, were only barely sufficient to undertake surveys. We are also mindful that the second site constructed at Aquila Island is suboptimal as a monitoring sites as we could not, despite two attempts, locate a suitable 120 m stretch of incipient reef on which to locate the transects. We therefore suggest attempting to relocate the transects at Aquila site 2, and if on a third attempt suitable habitat is not found then reducing surveys to the present site 1 only. Connor Island was also surveyed in conditions of limited in-water visibility. Should better conditions reveal more suitable habitat during the next survey, it may be prudent to relocate transects. The site least representative of an incipient reef at Connor Island was site 1, 2 m depth, where the transects follow a convoluted contour around a predominantly rocky substrate.

6 CONCLUSION

The baseline survey of monitoring locations generally supports the conclusions of previous surveys; that the environmental conditions within the Southern Inshore Zone imposes a selection pressure on coral communities that has limited reef development. This is particularly true of the reefs in more turbid settings. The coral community at Henderson Island demonstrates that where turbidity is lower, coral communities are more diverse and high cover can be realised. However, macroalgae are also supported and the high cover of this group appears to have created a bottleneck for the recovery of coral cover at Pine Peak Island and Pine Peak Islets. With the implantation of permanent monitoring sites, future observations will help to disentangle the pressures influencing the dynamics of coral communities in this unique environment.

7 REFERENCES

Anon (2010) Water quality guidelines for the Great Barrier Reef Marine Park. Revised edition 2010. The Great Barrier Reef Marine Park Authority, Townsville. ISBN 9781921682292 101pp

Beals EW (1984) Bray-Curtis ordination: an effective strategy for analysis of multivariate ecological data. In MacFadyen A, & Ford ED (eds.) Advances in Ecological Research 14, Academic Press, London

Berkelmans R, De'ath G, Kininmonth S, Skirving WJ (2004) A comparison of the 1998 and 2002 coral bleaching events on the Great Barrier Reef: spatial correlation, patterns, and predictions. *Coral reefs* 23:74-83

Berkelmans R, Jones A, Schaffelke B (2012) Salinity thresholds of *Acropora spp*. on the Great Barrier Reef. Coral reefs 31:1103-1110

Birrell CL, McCook LJ, Willis BL, Diaz-Pulido GA (2008a) Effects of benthic algae on the replenishments of corals and the implications for the resilience of coral reefs. Oceanography and Marine Biology: An Annual Review 46:25-64

Cheal A.J., Coleman G., Delean S., Fitzpatrick B., Miller I., Osborne K., Page C. and Sweatman H. 2001. Status of Fringing Reefs and Options for Long-Term Reef Monitoring in the Northumberland Islands, Southern Great Barrier Reef, Australia. AIMS Report Number 33. Australian Institute of Marine Science, Townsville.

Connell, J. H. (1997). Disturbance and recovery of coral assemblages. Coral reefs, *16*(1), S101-S113.

De'ath G, Fabricius KE, Sweatman H, Puotinen M (2012) The 27-year decline of coral cover on the Great Barrier Reef and its causes. Proceedings of the National Academy of Sciences 190(44):17995-17999

Diaz-Pulido G, Harii S, McCook LJ, Hoegh-Guldberg O (2010) The impact of benthic algae on the settlement of a reef-building coral. Coral Reefs 29:203-208

Erftemeijer PLA, Reigl B, Hoeksema BW, Todd PA (2012) Environmental impacts of dredging and other sediment disturbances on corals: A review. Marine Pollution Bulletin 64:1737-1765

Fabricius KE (2005) Effects of terrestrial runoff on the ecology of corals and coral reefs: Review and synthesis. Marine Pollution Bulletin 50:125-146

Fabricius KE, De'ath G, Puotinen MJ, Done T, Cooper TF, Burgess SC (2008) Disturbance gradients on inshore and offshore coral reefs caused by a severe tropical cyclone. Limnology and Oceanography 53(2): 690-704

Foster NL, Box SJ, Mumby PJ (2008) Competitive effects of macroalgae on the fecundity of the reef-building coral *Montastrea annularis*. Marine Ecology Progress Series 367:143-152

Garde LA, Spillman CM, Heron S, Beeden R (2014) ReefTemp Next Generation: A New Operational System for Monitoring Reef Thermal Stress, Journal of Operational Oceanography, 7:21-33

Gower JC (1966) Some distance properties of latent rot and vector methods used in multivariate analysis. Biometrika 53:325-328

Gruber, R., Waterhouse, J., Logan, M., Petus, C., Howley, C., Lewis, S., Tracey, D., Langlois, L., Tonin, H., Skuza, M., Costello, P., Davidson, J., Gunn, K., Lefevre, C., Shanahan, M., Wright, M., Zagorskis, I., Kroon, F., Neilen, A., (in prep), *Marine Monitoring Program: Annual Report for Inshore Water Quality Monitoring 2018-19. Report for the Great Barrier Reef Marine Park Authority*, Great Barrier Reef Marine Park Authority, Townsville

Harmelin-Vivian ML (1994) The effects of storms and cyclones on coral reefs: a review. Journal of Coastal Research Special Issue 12:211-231

Hauri C, Fabricius K, Schaffelke B, Humphrey C (2010) Chemical and physical environmental conditions underneath mat- and canopy-forming macroalgae, and the effects on understory corals. PloS One 5(9):e12685

Hopley D, Slocombe AM, Muir F, Grant C. 1983 Nearshore fringing reefs in North Queensland. Coral Reefs 1:151-160

Hughes TP, Rodrigues MJ, Bellwood DR, Ceccarelli D, Hoegh-Guldberg O, McCook L, Moltschaniwskyj N, Pratchett MS, Steneck RS, Willis B (2007) Phase Shifts, Herbivory, and the Resilience of Coral Reefs to Climate Change. Current Biology 17:360-365

Jones AM, Berkelmans R (2014) Flood impacts in Keppel Bay, Southern Great Barrier Reef in the aftermath of cyclonic rainfall. PLOS one doi:10.1371/journal.pone.0084739

Jonker M, Johns K, Osborne K (2008) Surveys of benthic reef communities using underwater digital photography and counts of juvenile corals. Long-term Monitoring of the Great Barrier Reef, Standard Operational Procedure Number 10, Australian Institute of Marine Science, Townsville

Kleypas JA. 1996 Coral reef development under naturally turbid conditions: fringing reefs near Broad Sound, Australia. Coral Reefs 15:153-167

Marshall PA, Baird AH (2000) Bleaching of corals on the Great Barrier Reef: differential susceptibilities among taxa. Coral Reefs 19(2):155-163

Maynard JA, Turner, PJ, Anthony KRN, Baird A, Berkelmans R, Eakin CM, Johnson J, Marshall PA, Packer GR, Rea A, Willis BL (2008) ReefTemp: An interactive monitoring system for coral bleaching using high-resolution SST and improved stress predictors. Geophysical Research Letters 35: L05603, doi:10.1029/2007GL032175

McCook LJ, Jompa J, Diaz-Pulido G (2001) Competition between corals and algae on coral reefs: a review of evidence and mechanisms. Coral Reefs 19:400-417

Miller IR, Jonker M, Coleman G (2009) Crown-of-thorns seastars and coral surveys using the manta tow and SCUBA Search techniques. Long-term Monitoring of the Great Barrier Reef Standard Operational Procedure Number 9, Edition 3. Australian Institute of Marine Science, Townsville

Oksanen J, Guillaume Blanchet F, Friendly M, Kindt R, Legendre P, McGlinn D, Minchin PR, O'Hara RB, Simpson GL, Solymos P, Stevens HH, Szoecs E, Wagner H (2016) Community Ecology Package, Version 2.4.1. http://127.0.0.1:30877/library/vegan/html/vegan-package.html

Osborne K, Dolman AM, Burgess SC, Jones KA (2011) Disturbance and the dynamics of coral cover on the Great Barrier Reef (1995-2009). PLoS ONE 6:e17516

Ricardo G, Jones, Negri A, Stocker R (2016) That sinking feeling: Suspended sediments can prevent the ascent of coral egg bundles. *Scientific Reports.* doi:10.1038/srep21567

Ricardo GF, Jones RJ, Nordborg M, Negri AP (2017) Settlement patterns of the coral *Acropora millepora* on sediment-laden surfaces. *Science of The Total Environment* doi.org/10.1016/j.scitotenv.2017.07.153

Ridgeway G (2007) Generalized boosted models: a guide to the gbm package, http://www.saedsayad.com/docs/gbm2.pdf

Sea Research (2017) Reconnaissance of Coral Communities in the Southern Inshore Mackay-Whitsunday Region: 2017 with suggested Long-Term Coral Monitoring Design. Report to Healthy Rivers to Reef Partnership.

Schleussner C-F, Lissner TK, Fischer EM, Wohland J, Perrette M, Golly A, Rogelj J, Childers K, Schewe J, Frieler K, Mengel M, Hare W, Schaeffer M (2016) Differential climate impacts for policyrelevant limits to global warming: the case of 1.5°C and 2°C. Earth System Dynamics 7:327-351

Storlazzi CD, Norris BK, Rosenberger KJ (2015) The influence of grain size, grain color, and suspended-sediment concentration on light attenuation: Why fine-grained terrestrial sediment is bad for coral reef ecosystems. *Coral Reefs*, *34*(3), 967-975

Sweatman H, Thompson A, Delean S, Davidson J, Neale S (2007) Status of Near-Shore reefs of the Great Barrier Reef 2004. Marine and Tropical Sciences research Facility Research Report Series. Reef and Rainforest Centre Limited, Cairns, 169p

Thompson A, Lønborg C, Costello P, Davidson J, Logan M, Furnas M, Gunn K, Liddy M, Skuza M, Uthicke S, Wright M and Zagorskis I, Schaffelke B (2014) Marine Monitoring Program. Annual Report of AIMS Activities 2013 to 2014– Inshore water quality and coral reef monitoring. Report for the Great Barrier Reef Marine Park Authority. Australian Institute of Marine Science 146p

Thompson A, Costello P, Davidson J, Logan M, Coleman G, Gunn K, Schaffelke B (2016) Marine Monitoring Program. Annual Report for coral reef monitoring: 2014 to 2015. Report for the Great Barrier Reef Marine Park Authority. Australian Institute of Marine Science 133p

Thompson A, Costello P, Davidson J, Logan M, Coleman G, Gunn K (2018) Marine Monitoring Program Annual Report for inshore coral reef monitoring: 2016 - 2017 Great Barrier Reef Marine Park Authority. Australian Institute of Marine Science 148p

Thompson A, Costello P, Davidson J, Logan M, Coleman G (2019) Marine Monitoring Program Annual Report for Inshore Coral Reef Monitoring: 2017 to 2018. Great Barrier Reef Marine Park Authority. Australian Institute of Marine Science 132p

Wolanski E, Fabricius K, Spagnol S, Brinkman R (2005) Fine sediment budget on an inner-shelf coral-fringed island, Great Barrier Reef of Australia. *Estuarine Coastal and Shelf Science* 65:153-158

van Dam JW, Negri AP, Uthicke S, Muller JF (2011) Chemical pollution on coral reefs: exposure and ecological effects. In: Sanchez-Bayo F, van den Brink PJ, Mann RM (Eds.), Ecological Impact of Toxic Chemicals. Bentham Science Publishers Ltd.

van Duin EHS, Blom G, Johannes Los F, Maffione R, Zimmerman R, Cerco CF, Dortch M, Best EPH (2001) Modelling underwater light climate in relation to sedimentation, resuspension, water quality and autotrophic growth. *Hydrobiologia* 444:25–4

van Woesik R (1991) Immediate impact of the January 1991 floods on the coral assemblages of the Keppel Islands. Research Publication No. 23, GBRMPA

van Woesik R, Done TJ. 1997 Coral communities and reef growth in the southern Great Barrier Reef. Coral Reefs 16: 103-115

Vuki VC, Price IR (1994) Seasonal changes in the Sargassum populations on a fringing coral reef, Magnetic Island, Great barrier reef region, Australia. Aquatic Botany 48(2)153-166

8 APPENDICES

Figure A 1 Pine Peak Island monitoring sites. Red dots show location of sites (S1, S2) for each depth (D2, D5).

Figure A 2 Pine Islets monitoring sites. Red dots show location of sites (S1, S2) for each depth (D2, D5).

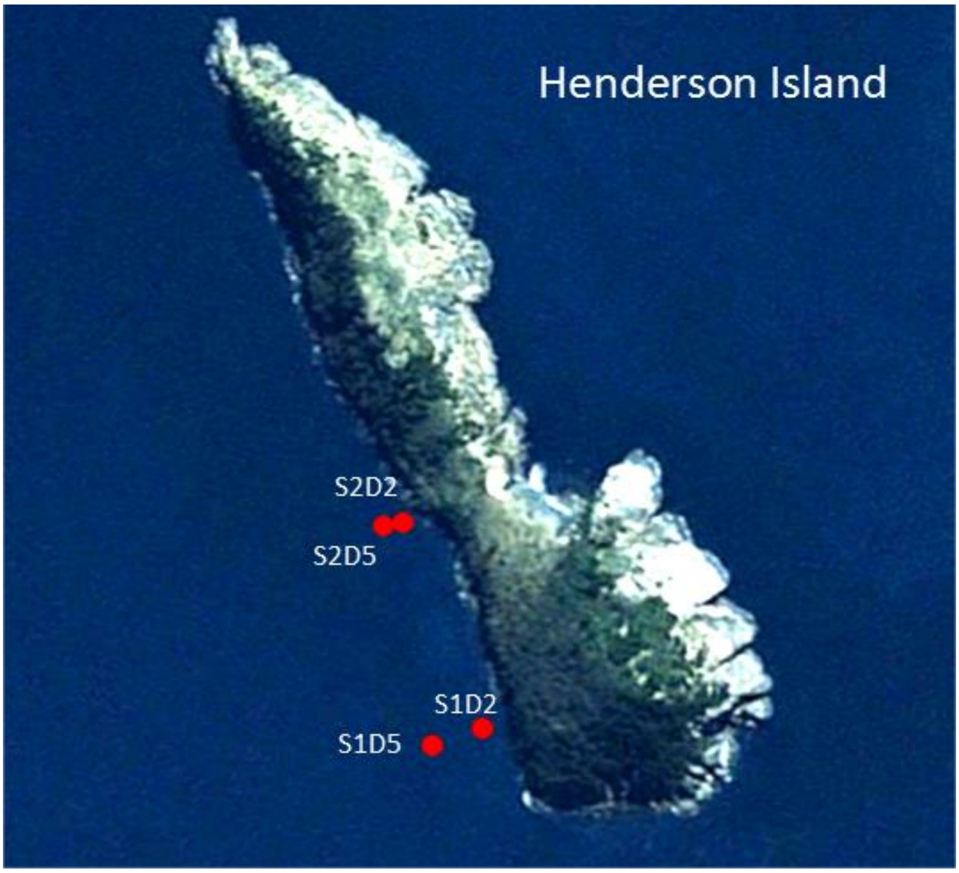


Figure A 3 Henderson Island monitoring sites. Red dots show location of sites (S1, S2) for each depth (D2, D5).

Figure A 4 Connor Island monitoring sites. Red dots show location of sites (S1, S2) for each depth (D2, D5).

Figure A 5 Temple Island monitoring sites. Red dots show location of sites (S1, S2).

Figure A 6 Aquila Island monitoring sites. Red dots show location of sites (S1, S2).

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, note first rod is at 3m, contour
	21.51433	**150.25125**	**5**	**1**	**1**	340 then contour
	Waypoint between transects 3 & 4				2	150, 110@6m, 60@10m rod, 320 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, 180@9m
					4	260, 300@3m
					5	210, 340@4m
	21.51375	**150.25513**	**5**	**2**	**1**	90 330@11m
	Waypoint between transects 3 & 4				2	0, 100@2m, 30@10m rod, 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, 180@10m rod
	Waypoint between transects 3 & 4				2	340
					3	240
					4	50, 90@10m
					5	120
	21.65743	**150.21917**	**5**	**2**	**1**	200
	Waypoint between transects 3 & 4				2	270, 320@10m rod
					3	270, 200@10m rod
					4	30, 120@10m rod
					5	180, 60@10m rod

Table A1 continued

Reef	Latitude S	Longitude E	Depth	Site	Tran	Compass directions
Henderson Island	21.48542	149.90965	2	1	1	340
	Waypoint between transects 3 & 4				2	330
					3	330, 350@10m rod
					4	150
					5	160, start shoreside PM
	21.4856	149.90907	5	1	1	310, 330@10m rod
	Waypoint between transects 3 & 4				2	300 over large Lobophyllia to end
					3	320, ends short of large Porites
					4	130, 120@10m rod
					5	150, 200@10m rod
	21.48313	149.90868	2	2	1	310
	Waypoint between transects 3 & 4				2	320
					3	320, 300@10m rod
					4	120
					5	150
	21.48317	149.90845	5	2	1	0, 350@10m rod
	Waypoint between transects 3 & 4				2	300, 320@10m rod
					3	320, 310@10m rod
					4	180, 150@10m rod
					5	180
Connor Island	21.71732	149.67282	2	1	1	30, 180@10m rod
	Site is convoluted around rocks. Waypoint at transect 1				2	270, 290@10m rod
					3	140, 190@10m rod
					4	190, 90@10m rod
					5	60, 90@10m rod
	21.71725	149.67322	5	1	1	180, 90@10m rod
	Waypoint between transects 3 & 4				2	170, 210@10m rod
					3	170, 150@10m rod
					4	30, 0@10m rod
					5	30
	21.72188	149.67168	2	2	1	150, 110@10m
	Waypoint between transects 3 & 4				2	150, 140@10m
					3	150, 100@10m
					4	300
					5	330, 300@10m
	21.7218	149.6721	5	2	1	150
	Waypoint between transects 3 & 4				2	120
					3	120, 180@6m, 150@10m
					4	280, 330@10m
					5	310, 300@10m

Table A1 continued

Reef	Latitude S	Longitude E	Depth	Site	Tran	Compass directions
Temple Island	**21.59608**	**149.50102**	**1**	**1**	**1**	200, 170@10m
	Waypoint between T1-T4				2	150, 180@10m
					3	190
					4	350
					5	330, 310@10m
	21.60285	**149.49932**	**1**	**2**	**1**	240, 220@10m
	Waypoint between T1-T4				2	190, 200@10m
					3	180, 190@10m
					4	90, 30@10m, 340@12m, 300
					5	30, 50@10m
Aquila Island	**21.95682**	**149.58102**	**1**	**1**	**1**	190, 180@10m, 140 to T2
	Waypoint between T1-T4				2	140
					3	170
					4	320
					5	330, 310@10m
	21.96112	**149.58158**	**1**	**2**	**1**	120
	Waypoint between T1-T4				2	90
					3	110
					4	0
					5	30

Table A 2 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 presented as Genus Richness

Reef	Depth	*Acropora*	*Cyphastrea*	*Favia*	*Galaxea*	*Goniastrea*	*Goniopora*	*Lobophyllia*	*Montipora*	*Mycedium*	*Pachyseris*	*Platygyra*	*Porites*	*Turbinaria*	Other HC	Genus Richness
Pine Peak Island	2	0.25	0.00	0.00	0.00	0.00	0.00	0.00	0.31	0.00	0.00	0.44	2.32	0.00	0.13	5
	5	1.06	0.00	0.19	0.00	0.38	0.44	0.19	0.81	0.50	0.06	0.44	4.01	0.00	1.31	20
Pine Islets	2	0.75	0.00	0.00	0.00	0.00	0.00	0.00	0.63	0.00	0.00	0.06	0.19	0.69	0.38	8
	5	0.06	0.00	0.06	0.06	0.38	0.56	0.19	4.13	1.69	0.19	0.56	2.06	2.25	2.56	25
Henderson Island	2	53.61	0.00	0.13	0.06	0.13	0.06	0.13	1.44	0.00	0.13	0.63	0.06	0.06	0.63	14
	5	33.69	0.13	0.00	1.56	0.00	1.00	4.25	2.38	0.00	0.19	0.19	0.00	0.63	4.75	21
Connor Island	2	3.94	0.25	0.19	0.00	0.44	0.63	0.19	6.06	0.00	0.57	0.31	0.75	7.31	2.13	20
	5	1.81	0.19	1.38	3.94	1.94	0.44	0.00	9.88	0.00	1.44	0.00	0.31	7.63	4.13	20
Temple Island	1	2.81	1.06	0.31	0.00	0.44	0.38	0.00	7.38	0.00	0.88	1.50	0.50	3.25	1.00	18
Aquila Island	1	0.44	0.00	0.06	0.00	0.44	0.06	0.00	17.94	0.00	0.06	0.13	0.13	0.06	1.44	13

Table A 3 Cover of soft coral genera. 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*	*Sarcophyton*	*Sinularia*	*Xenia*	Other SC
Pine Peak Island	2	4.20	0.06	0	0.50	1.63	0.31	0.44
	5	10.83	0.13	0.31	0.31	2.44	0	0.94
Pine Islets	2	0.25	0	0	0	0.38	0	0.75
	5	1.69	0.19	0.75	0	1.25	0	0.06
Henderson Island	2	1.19	1.38	10.08	1.69	3.00	0	0.56
	5	0.56	0.69	10.13	3.00	3.06	0	1.75
Connor Island	2	1.38	0.25	0	0.57	8.95	0.56	1.19
	5	1.19	0.06	0.06	0.31	8.13	0	0.50
Temple Island	1	4.06	0.06	0.06	0.69	14.38	13.19	0.69
Aquila Island	1	0.31	0	0.25	0.75	7.50	1.38	0.81

Table A 4 Cover of Algae. Identified macroalgae genera with a cover of at least 1% at any reef are separated. All less abundant or un-resolved genera and smaller algae are grouped.

Reef	Depth	Brown macroalgae			Red macroalgae				Green macroalgae	Turf algae	Blue-green algae	Coralline algae
		Lobophora	*Sargassum*	Other	*Botryocladia*	*Peyssonnelia*	Calcareous red macroalgae	Other				
Pine Peak Island	2	5.69	63	2.13	0	0	0.13	3.44	0.44	8.52	0.13	2.19
	5	24.41	23.34	2.44	0	0	0.63	1.69	0.19	15.14	0.69	0.75
Pine Islets	2	0.81	84.56	1.38	0	0.06	0	1.63	0	4.44	0	1.31
	5	10.25	35.25	2.5	0	0.06	0.13	1.56	0.06	22.06	0.25	0.81
Henderson Island	2	3.07	2.01	0.31	0	0	0	0.25	0.06	17.26	0	0.13
	5	8.06	0.13	0.13	0	0	0	0.13	0.06	13	0	0.19
Connor Island	2	4	7.38	0.88	0.06	1.5	5.25	19.5	0.19	16.56	0	3.56
	5	2.19	1	1.25	1.19	2.56	3.44	10.31	0.06	20.75	0	2.94
Temple Island	1	1.44	18	0.56	0	1.25	0.88	5.06	0	10.63	0	1.88
Aquila Island	1	1.25	17.69	1.44	0	1	3.94	7.81	0.19	12	0	1.31

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

Reef	Depth	*Acropora*	*Alveopora*	*Cyphastrea*	*Favia*	*Favites*	*Goniastrea*	*Goniopora*	*Lobophyllia*	*Montipora*	*Moseleya*	*Pocillopora*	*Porites*	*Psammocora*	*Turbinaria*	Other genera	Genus Richness	Number	Density
Pine Peak Island	2	0.5	0	0	0.5	0	0	0	0.5	1	0	3	1	0.5	0	0.5	8	7.5	0.25
	5	1.5	1	0	0	0	0	0.5	1	0	0	0.5	2	0	0	1.5	9	8	0.33
Pine Islets	2	1	0	0	2.5	0.5	0.5	1	0.5	6.5	0.5	2	1.5	4.5	1.5	4.5	19	27	0.83
	5	1.5	0.5	0	2	2	0	2.5	3.5	1	1	2.5	1	0	2	8.5	24	28	1.11
Henderson Island	2	3	0	0	2	1	5	0.5	3	0	0	2.5	0	1	0.5	2.5	12	21	2.49
	5	3.5	0	0	0	0	1	0.5	2	3	0.5	1.5	0	0	0	2.5	10	14.5	1.95
Connor Island	2	4.5	0	0	0	0.5	1	1.5	0	3.5	0	14.5	0	1.5	6	1	10	34	1.69
	5	0.5	5.5	0.5	1	1	2	1	0	3	0	2	1	1.5	5.5	3	15	27.5	1.67
Temple Island	1	2.5	0	2.5	2.5	0.5	1.5	2	0	2	0	4.5	5	0.5	13.5	2	15	39	2.85
Aquila Island	1	0.5	1.5	1.5	1	1	0.5	2	0.5	5	2.5	4	0	0.5	2.5	1.5	16	24.5	1.57

.

Table A 6 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 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		Connor		Temple	Aquila
		2m	5m	2m	5m	2m	5m	2m	5m	1m	1m
Disease	*Acropora*					7	1				
	Galaxea						1				
	Goniopora						1				
	Montipora			1	6	2	1	2	2		1
	Turbinaria										1
Unknown cause	*Acropora*					8	2	1		1	
	Echinopora				1						
	Galaxea								2		
	Montipora	1			1				3	2	2
	Pocillopora		2							1	
	Porites	2									
	Turbinaria				1						
Sponge - *Cliona orientalis*	*Cyphastrea*									1	
	Favites							2			
	Goniastrea									1	
	Platygyra					1				1	
	Porites										1
	Turbinaria							1	1	2	
Sediment	*Acropora*						1				
Total number of Colonies		**3**	**2**	**1**	**9**	**18**	**7**	**6**	**8**	**9**	**5**
Anchoring (proportion of colonies)	-							<1%	<1%		
Bleaching (proportion of colonies)	*Goniopora*					<1%	<1%				

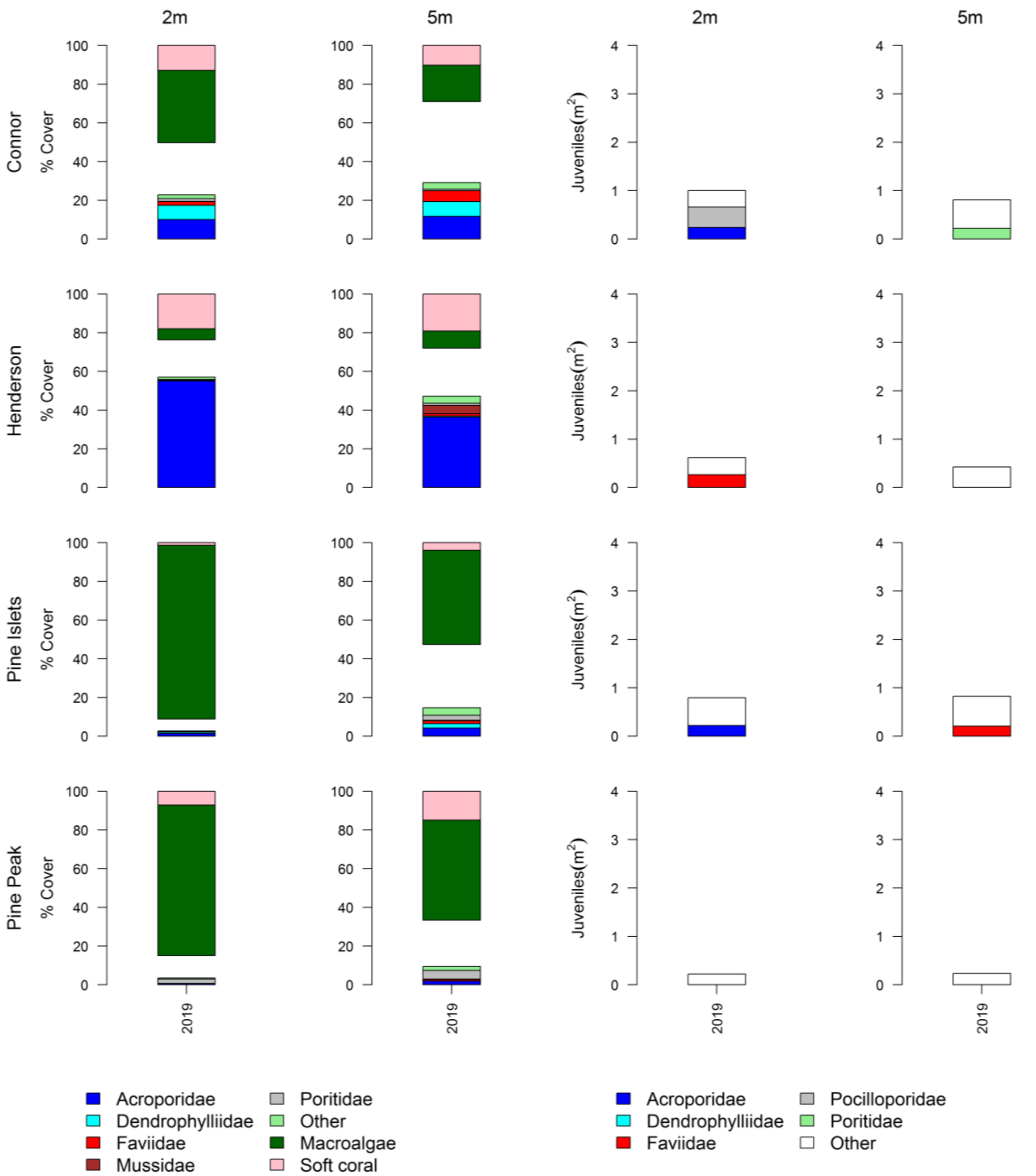


Figure A 7 Composition of benthic cover and hard coral juveniles. The left-hand plots show the breakdown of cover for hard coral families at 2 m and 5 m depths. Families that had a cover of at least 3% at either depth of any reef in the Zone are differentiated cover of all other families are grouped 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.

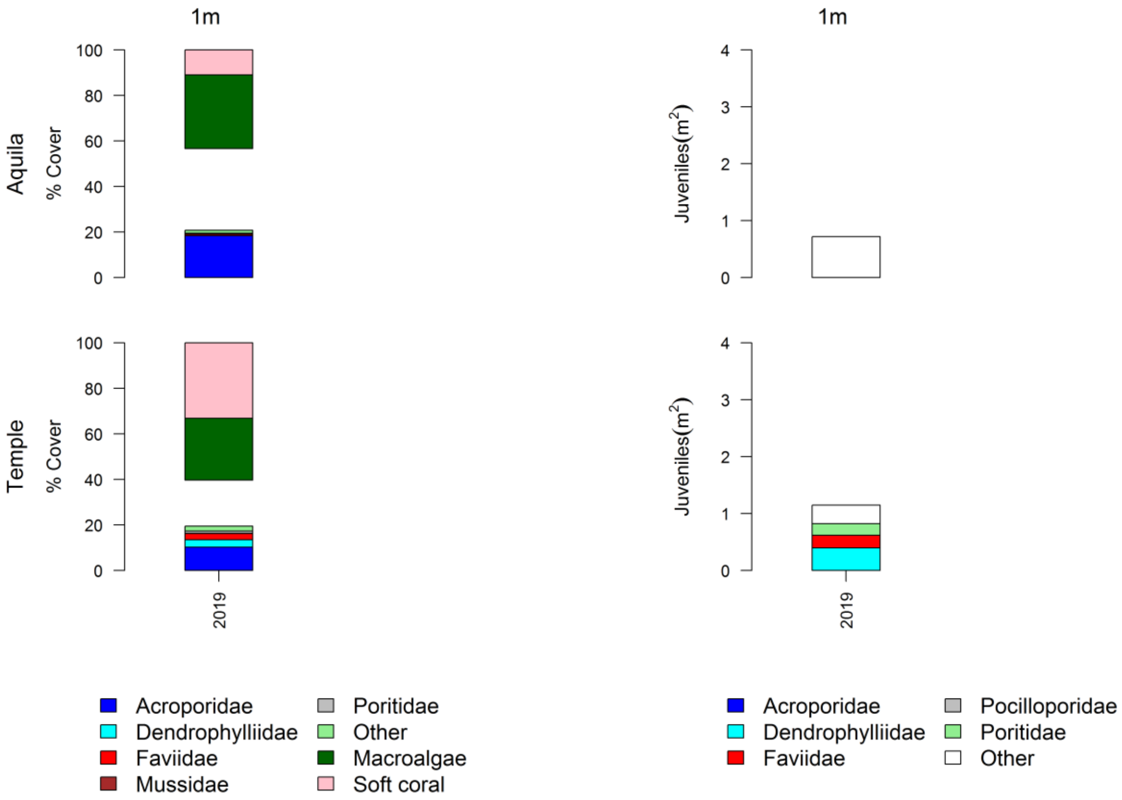


Figure A 7 continued, for the 1 m deep sites at Aquila and Temple Islands.

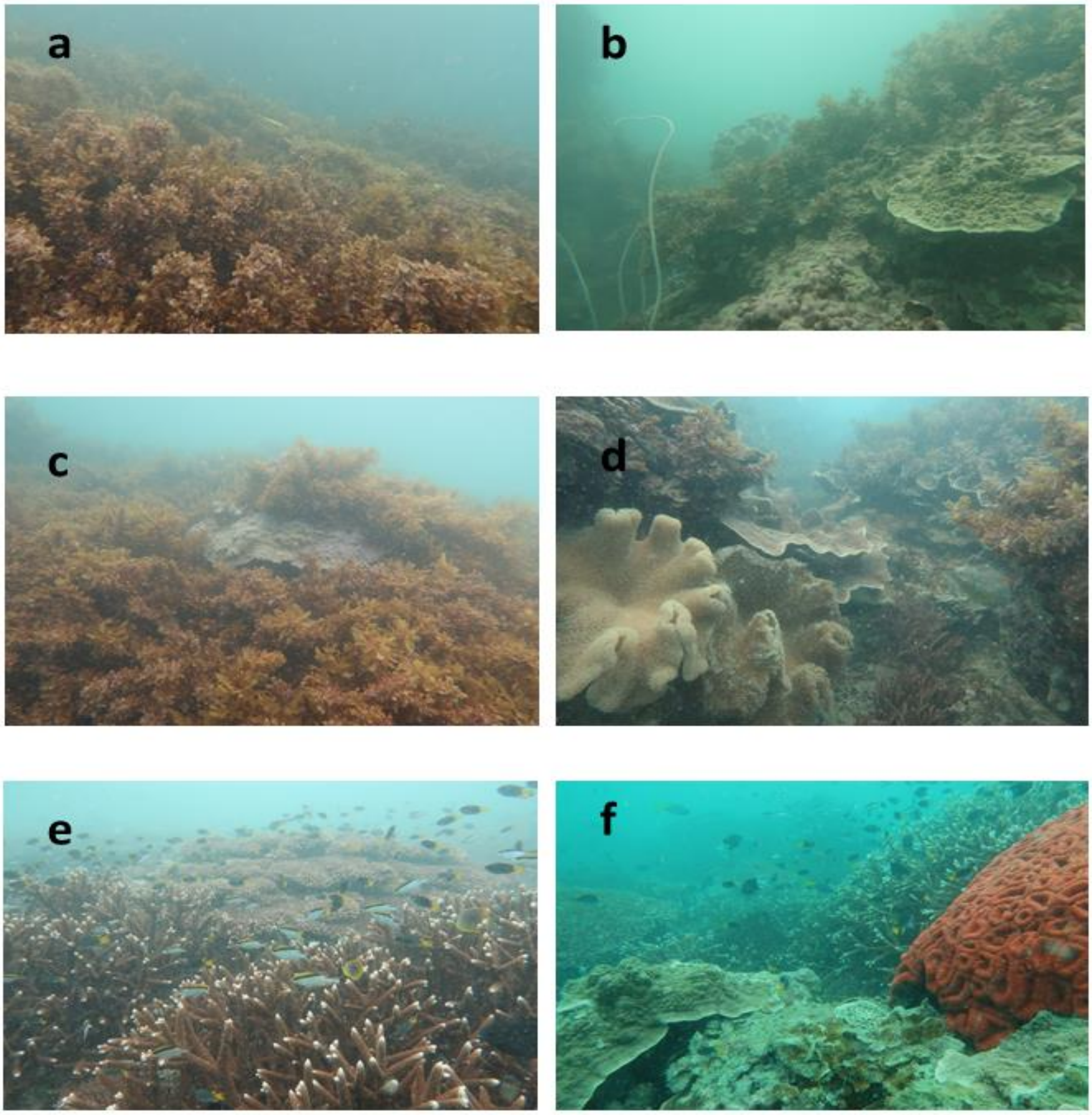


Figure A 8 Benthic community photos at a) Pine Peak 2m b) Pine Peak 5m c) Pine Islets 2m d) Pine Islets 5m e) Henderson 2m f) Henderson 5m. Dominant macroalgae at Pine Peak and Pine Islets 2m compared to abundant *Acropora* corals at Henderson 2m. Mixed hard and soft corals at Pine Peak and Pine Islets 5m compared with diverse hard corals at Henderson 5m.

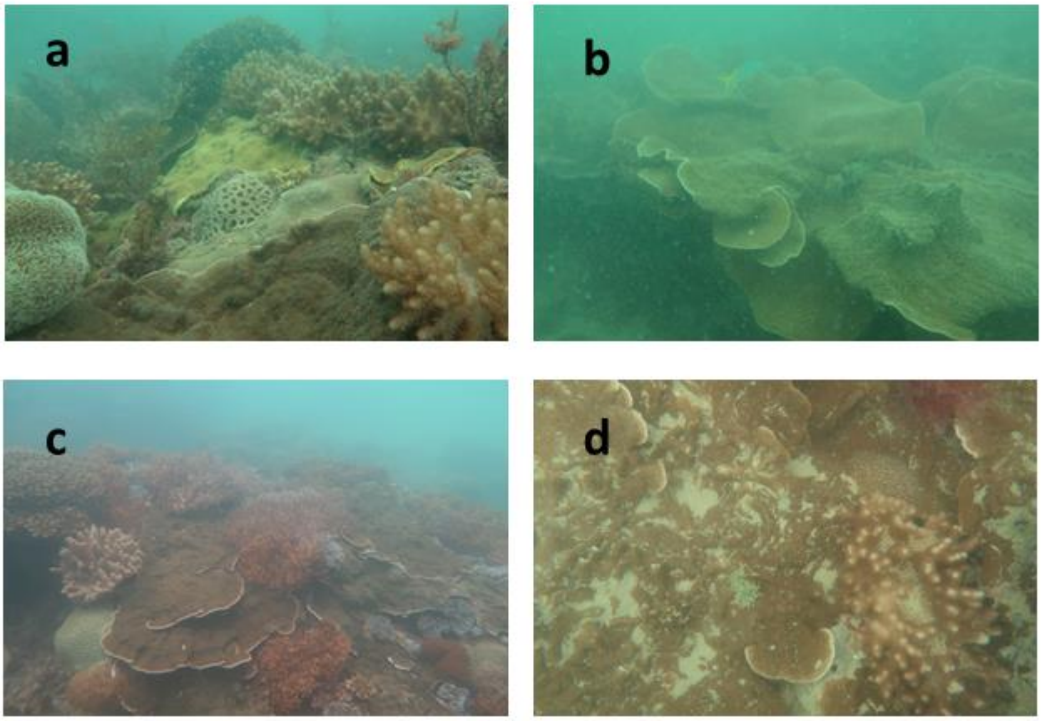


Figure A 9 Coral community photos at a) Connor 2m b) Connor 5m c) Temple 1m d) Aquila 1m. Mixed hard and soft corals at Connor 2m and Temple 1m. Large foliose colonies of *Turbinaria* at Connor 5m and *Montipora* at Aquila 1m.