Port of Abbot Point Ambient Coral Monitoring Program: Report 2017

NQ Bulk Ports

Angus Thompson, Johnston Davidson, Paul Costello

AIMS: Australia's tropical marine research agency

Townsville
2018

Australian Institute of Marine Science

PMB No 3 Townsville MC Qld 4810	PO Box 41775 Casuarina NT 0811	Indian Ocean Marine Research Centre University of Western Australia, M096 Crawley WA 6009

This report should be cited as:
Thompson A, Costello P, Davidson J (2018) *Port of Abbot Point Ambient Monitoring Program: Report 2017*. Report prepared for North Queensland Bulk Ports. Australian Institute of Marine Science, Townsville. (39 pp)

DISCLAIMER

Vendor shall ensure that documents have been fully checked and approved prior to submittal to client				
Revision History:		*Name*	*Date*	*Comments*
1	Prepared by:	*Angus Thompson*	*24/01/2018*	
	Approved by:	*Britta Schaffelke*	*24/01/2018*	
2				

Cover photo:
Corals at Camp West in May 2017

CONTENTS

LIST OF FIGURES

LIST OF TABLES

1 EXECUTIVE SUMMARY

The Port of Abbot Point Ambient Coral Monitoring Program (the program) was initiated in May 2016. This report outlines the results of the second year of monitoring surveys conducted in May/June 2017 and October/November 2017. The sampling methods used were selected to allow the estimation of coral condition index scores as used by the Reef Plan Reef Report Card. As at November 2017 the overall condition index score was 0.25, a marked decline from the score of 0.36 in October 2016. The decline in condition reflects the cumulative impact of Tropical Cyclone (TC) Debbie and high summer water temperatures that led to coral bleaching. In the context of the Reef Report Card these scores translate into a report card grade of D which is “poor”. The decline in scores observed at the reefs reported here broadly reflect declines in adjacent regions. Coral Index scores declined in 2017 in the Mackay Whitsunday Region, in response to damage incurred during TC Debbie, and in the Burdekin Region as a result of coral bleaching.

At Camp Island the coral index score in October 2016 was low, 0.18 (grade E), reflecting the very high cover of large fleshy seaweeds (macroalgae), low cover of corals and low densities of juvenile corals. In November 2017 the coral index score at Camp remained similar, at 0.16, with little change in the density of juvenile corals (mean density declined from 3.8 m^{-2} to 3.3 m^{-2}), and a small increase in coral cover at eastern sites but a decline at western sites (mean cover declined from 16% to 13.2%) . Despite a significant decrease (75.3% down to 62.8%), the proportion of macroalgae in the algal community remained well above threshold values and the index score for macroalgae remained unchanged.

In contrast, the index score for Holbourne Island declined substantially from 0.53 in October 2016, representing a grade of C, to 0.33 and a grade of D in October 2017. This decline at Holbourne reflects the severe impact associated with TC Debbie that reduced the cover of corals and density of juvenile corals to very low levels (mean coral cover 36.2% down to 6.7%, mean juvenile density declined from 2.0 m^{-2} to 0.65 m^{-2}), and saw a slight increase in the proportion of macroalgae in algal communities at 5 m depths (1.7% to 3.4%).

The lack of impact from TC Debbie at Camp Island can be explained by the direction of storm-driven waves. Wave records from a location adjacent to Holbourne Island documented two peaks in wave energy. Late in the afternoon and evening of the 27th March the mean height of set waves exceeded 7.5m from a north-westerly direction. As the cyclone passed, a second peak in wave height occurred in the morning of the 28th March, with waves in excess of 5m running from the south-east. Although waves were not monitored at Camp Island the location of the island relative to the mainland coast limits exposure to waves from either of these directions. At Camp Island East, loss of coral cover was attributed to a combination of anomalously high sea surface temperatures over the 2016/17 summer and coral disease.

Future surveys will provide information about changes in the overall coral index score and the component indicator conditions. Especially important will be the determination of coral recovery, or lack thereof, to draw conclusions about the ecosystem condition and exposure to ongoing pressures. Potentially limiting recovery potential at Holbourne Island are low densities of juvenile crown-of-thorns starfish that were observed consuming fragmented corals that had survived TC Debbie. This observation is of concern as, historically, recovery of coral communities following disturbances has been slow. Low densities of juvenile corals suggest that recovery may be limited by larval supply, which in the face of regional declines in coral cover, puts additional importance on survival of remnant corals for the recovery of coral communities. Persistently high cover of macroalgae and high levels of disease at Camp Island indicate ongoing environmental pressures associated with high nutrient levels, which are likely to limit coral recovery.

Seasonal increase in the cover of macroalgae between late wet and late dry season samples in both 2016 and 2017 were observed at Camp Island. This seasonality should be considered for any future changes to the sampling design as changes in cover of macroalgae have the potential to bias estimates of juvenile densities and coral cover. It is recommended that the timing of future surveys is seasonally consistent with the late wet season sampling undertaken in April/May as a way of limiting bias associated with high cover of macroalgae. The added benefit of this timing would be earlier access to data required for reporting, and earlier assessment of any summer disturbance events. We see little value in continued biannual sampling as disturbance events are typically restricted to the summer months and recovery rates slow enough that real changes between biannual samples are unlikely to exceed observational error.

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 nutrient or suspended sediments as a result of activities in the coastal zone and in adjacent catchments.

The successful management of activities potentially harmful to coral communities requires the ability to disentangle the impacts of local (manageable), pressures from larger scale processes. To achieve this goal baseline information relating to the dynamics of communities exposed to ambient environmental conditions is an essential precursor to assessing the response of communities exposed to potentially damaging conditions.

The Port of Abbot Point Ambient Coral Monitoring Program (the program) was initiated by the Australian Institute of Marine Science (AIMS) under contract to North Queensland Bulk Ports (NQBP). The overarching goal of this program is to develop an understanding of the condition of coral communities on fringing reefs in the vicinity of Abbot Point and of the key environmental factors influencing that condition. This understanding will benefit port master-planning and inform the design and interpretation of future impact assessments.

Specific objectives of the program are:

- To assess and report the condition of coral communities at Holbourne Island and Camp Island
- To identify key environmental factors influencing coral community condition

This report provides a summary of the condition of coral communities observed during late wet season and late dry season samples (May/June and October/November) in 2017 with reference to changes that have occurred since 2016. Differences in communities between sites, seasons, and years are discussed in terms of likely drivers of community condition.

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 both acute disturbances such as cyclones or flooding as well as any pressures relating to port activities that may have either directional or depth stratified impacts.

In recognition of the importance of aspect and depth as determinants of coral community composition (e.g. Thompson et al. 2014), and exposure to potential acute and chronic pressures, sampling locations were selected on both the windward "Eastern" and leeward "Western" aspects of both Camp Island and Holbourne Island (Figure 1). At all locations at Holbourne Island transects were replicated at both 2m and 5m depths below lowest astronomic tide datum (LAT) as predicted by Navionics electronic charts on the day of site construction. At Camp Island East the reef slope transitioned to sand supporting seagrass at 1.5 to 2m below LAT and as such transects were set at 1.5m below LAT only. At Camp Island West the reef slope extended to 2-3m below LAT and transects were set at 2m LAT only (Table A1. 1).

As coral communities are spatially heterogeneous two levels of replication were included within each combination of aspect and depth. 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. At Holbourne West the sites are slightly closer together due to the limited extent of reef slope and a desire to keep the transect markers away from the beach, not only to avoid, as much as possible, potential damage to markers caused by anchoring but also to limit disruption of visual amenity at this popular recreational area. Within each site, five 20 metre long transects 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. 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 A1. 1.

Table 1Sampling design

Island	Zones	Sites per Zone	Depths per site	20m Transects per site and depth
Holbourne	East and West	2	2m and 5m	5
Camp	East and West	2	1.5-2m only	5

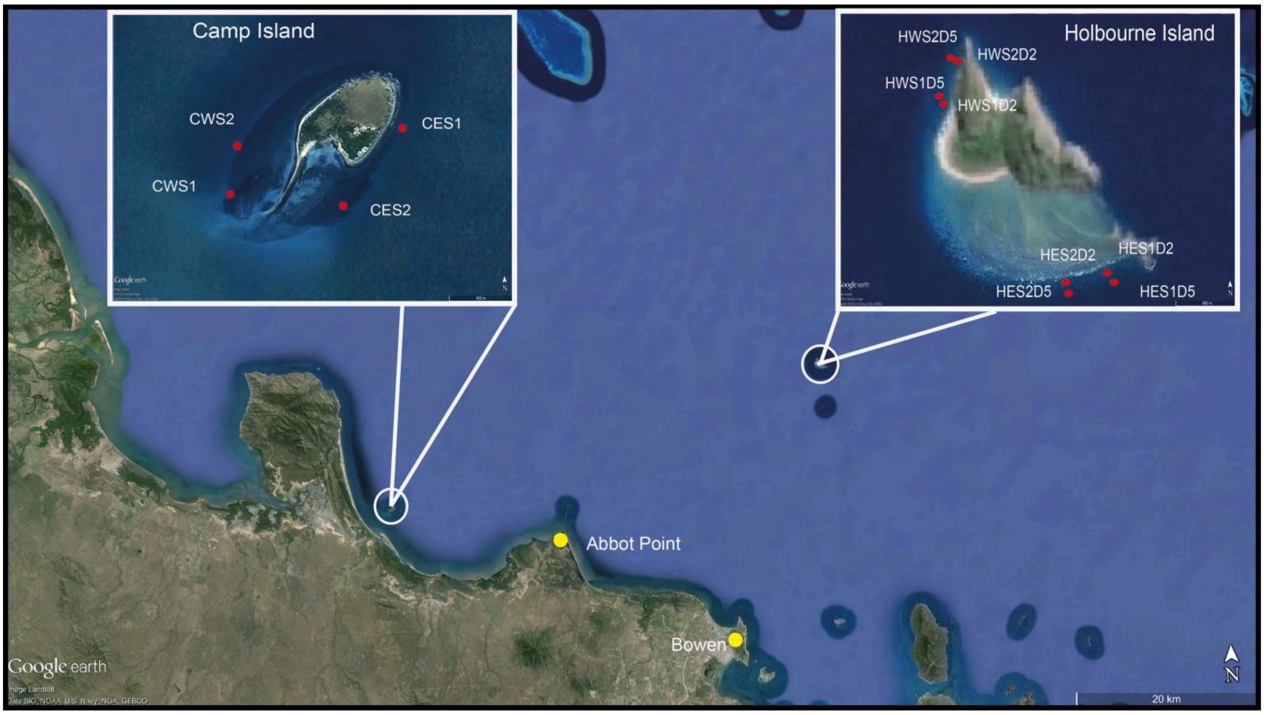

Figure 1 Coral monitoring locations. Site labels are abbreviated in the form; Reef –"H" for Holbourne Island and "C" for Camp Island, Aspect – "W" for West and "E" for East, Site number - S1 or S2 and Depth below

low tide datum – "D2" for 2m and "D5" for 5m. No depths included at Camp Island as only 1 depth was sampled at each site.

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, >2-5cm, and >5-10cm 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 were a subset of those used for the 2015 Reef Report Card (Queensland Government 2016; Thompson et al. 2016). 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 is scored on the basis of 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, a full description of these and the additional indicators con be found in Thompson et al. (2016).

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 either 'Hard' (order Scleractinia) or '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 indicators Juvenile Density and Proportion of Macroalgae Cover were included as they represent the potential for coral communities to recover from disturbances.

3.3.2 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 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).

3.3.3 Macroalgae Proportion

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; Cheal et al. 2013). 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.

$Macroalgae\ Proportion_{ij}$ = $Macroalgae\ cover_{ij}$ /$Total\ algae\ cover_{ij}$

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. 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].

Table 2 Indicator score thresholds

Indicator	**Location**	**Upper bound (score=1)**	**Lower bound (score=0)**
Coral Cover	All	75%	0%
Macroalgae Proportion	Camp East	9%	23%
	Camp West	6%	23%
	Holbourne East 2m	5.9%	10.2%
	Holbourne East 5m	0.6%	9.9%
	Holbourne West 2m	3.7%	8.8%
	Holbourne West 5m	0.5%	9.8%
Juvenile Density		13 m^{-2}	0 m^{-2}

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 surveys in May 2016 temperature loggers were deployed to star pickets marking site 1, transect 1 at each of Holbourne East 2m, Holbourne East 5m and Camp East. These loggers were retrieved and replaced during October 2016, and again in May and October 2017, beginning the process of recording an accurate in-situ climatology at each island. Until this time-series matures the likelihood of thermal stress to corals can be interpreted from thermal anomalies downloadable from ReefTemp (Garde et al. 2014) as published by the Bureau of Meteorology. For each combination of island and aspect, waypoints were selected in open water approximately 2 km

---

[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.*

out from the reefs (Table 3). These waypoints serve as the central locations for a set of nine pixels from which the 2015/16 and 2016/17 annual summary of degree heating days (DHD) and a time-series of monthly anomalies were downloaded. DHD are the sum of daily temperature anomalies from long term monthly averages across the period 1st December to the 31st March, whereas monthly anomalies are the mean daily anomaly for a given month; both estimated 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 3 Location of satellite derived environmental information

Location	Latitude	Longitude
Camp East	-19.829	147.896
Camp West	-19.829	147.896
Holbourne East	-19.751	148.371
Holbourne West	-19.711	148.34

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).

As a first step in assessing the likelihood that floods may have led to a direct salinity-related stress to corals, the seasonal discharge of local rivers is compared to long term median flows. Median discharge for the “wet season” defined here as December-May are calculated from available data 1990-2010 and compared to the current year. Discharge data were sourced from the Queensland Government water monitoring portal for:

- Station 121001A-Don River at Ida Creek
- Station 121002A – Elliot River at Guthalungra
- Station 120006B-Burdekin River at Clare

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 cyclones was assessed based on: published cyclone tracks (http://cimss.ssec.wisc.edu/tropic2/#), wave height data from the Abbot Point buoy (https://www.qld.gov.au/environment/coasts-waterways/beach/waves-sites/abbot-point/), and modelled wind and wave exposures for reefs (see Puotinen et al. 2016 for a detailed explanation of methods).

3.5 Spatial and temporal analysis

A series of generalised linear mixed effects models (Bolker 2009; McCulloch and Neuhaus 2013; Bates et al. 2015) were applied to the coral community response variables: juvenile density, macroalgae proportion, hard coral cover and soft coral cover. These models allowed the accommodation of non-normal response distributions including the binomial distribution of cover-based responses and the Poisson distribution for juvenile density. Further, juvenile density was modelled using juvenile abundance as the response and including an offset term of the log of the area available to juveniles within the sampled transects. Random components were included in each model to account for the correlations included by the repeated measures sampling design. Finally for each response linear contrasts were computed based on the estimated models to explicitly inform the questions:

- Did the response differ between Islands?
- Did the response differ between the eastern and western aspects of each combination of Island and depth?
- Did the response differ between 2m and 5m depths at either Holbourne Island East or West?
- Did the response differ between 2016 and 2017 within each combination of Island, depth, and aspect

Differences in benthic community composition 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.

All statistical analyses were done using R software (Core Team 2016) and the packages lme4 (Bates et al. 2015), multicomp (Hothorn et al. 2008) and vegan (Oksanen et al. 2016).

3.6 Environmental setting of reefs

As part of the baseline report for this program (Thompson et al. 2017), the underlying ambient water quality and hydrodynamic setting of the monitoring locations was determined. The methods used and subsequent results are replicated here.

3.6.1 Ambient Water Quality

Turbidity and nutrient levels are critical components of the aquatic environment and are fundamental determinants of benthic community composition. For the reporting of coral community condition in inshore areas, nutrient availability in particular 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). Relative to other reefs of the inshore GBR, levels of chlorophyll *a* (Chl) and Total Suspended Solids (TSS) at Holbourne Island show the ambient conditions here to be some of the highest quality for reefs monitored by the Marine Monitoring Program (Figure 2) reflecting the distance of this reef from the coast. In contrast Chl levels at Camp Island were double that at Holbourne Island whilst TSS was ~70% greater (Table 4, Figure 2). At higher than 0.45ugL^{-1}, the concentration of Chl at Camp Island predisposes that location to a high proportion of macroalgae and subsequent detrimental impacts to coral communities.

That the ambient water quality conditions at the two monitoring locations are within the range observed at other reefs monitored by AIMS (Figure 2) ensures the dynamics of communities observed can be validly compared to those occurring elsewhere in the inshore GBR and monitored as part of the AIMS LTMP and Reef Plan MMP.

Table 4 Ambient water quality conditions. Values represent the 10 year mean for each water quality parameter derived from satellite data; CDOM (m-1): Spatial distribution of mean of observed absorption due to coloured dissolved organic matter at 440 nm; Chl (mg/ m3): observed chlorophyll-*a* concentration, TSS (g/m^3): Spatial distribution of mean of observed total suspended solids.

Reef	**CDOM**	**Chl**	**TSS**
Camp Island	0.048	0.475	0.903
Holbourne Island	0.031	0.236	0.507

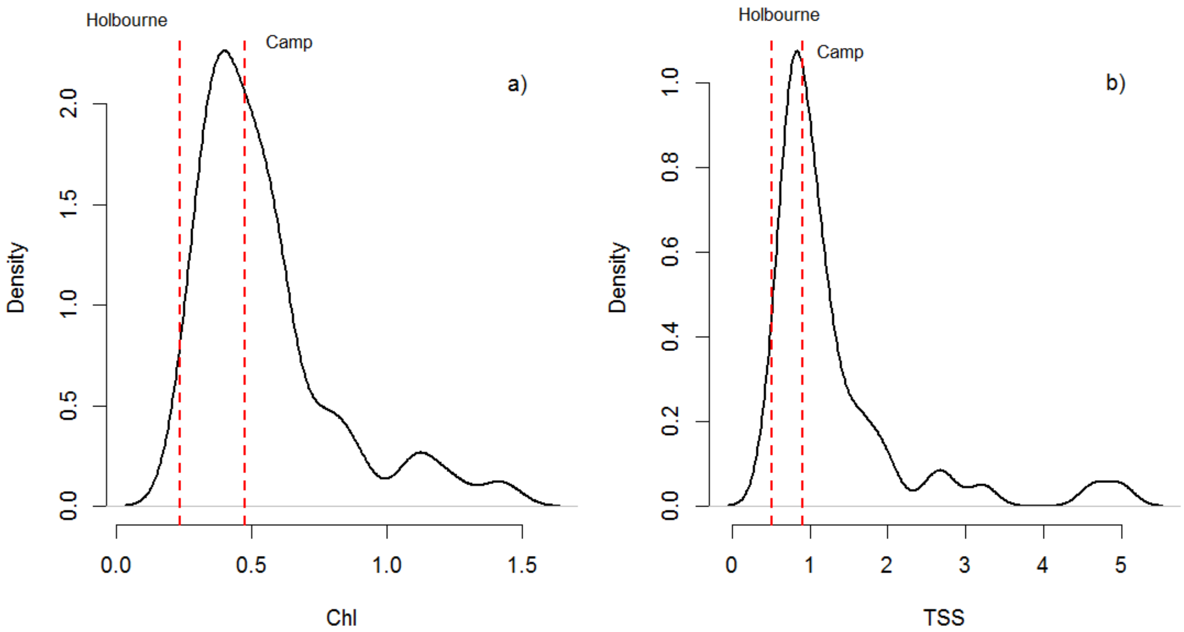

Figure 2 Ambient water quality conditions. a) Chlorophyll *a* and b) Total Suspended Solids. Black lines show the distribution of satellite derived water quality conditions averaged over 10 years for nearshore reefs of the GBR included in AIMS LTMP and MMP. Red lines indicate the relative position of Camp and Holbourne Island's within each distribution.

3.6.2 Sediment characteristics – Hydrodynamic setting

As a proxy for the hydrodynamic setting of a site (Wolanski et al. 2005) the composition of sediments is a useful covariate to consider in terms of coral community dynamics. Higher proportions of fine clay and silt sized particles in the sediment identify sites more prone to sediment accumulation that those with course grained sediments.

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 Holbourne Island samples were collected at the 5m sites, conforming to the sampling design for inshore reefs monitored under the MMP. As 5m sites were not present at Camp Island samples were taken from the shallower sites. Samples from the two islands are not directly comparable; those from Camp Island are assumed to have a lower proportion of clay and silt sized particles than would have been sampled at 5m should depth have permitted. As such, although the grainsize distributions are similar between the two islands (Table 5), accounting for the shallower depth of sampling at Camp, the results should be interpreted as the coral communities at Camp being more sheltered from wave exposure than those at Holbourne.

Table 5 Sediment composition at monitoring locations. Values indicate the percentage of total weight of sample with grainsizes <64 microns.

Reef	Site	Depth	% < 64um		
			May 2016	October 2016	Mean
Holbourne East	1	5	1.7	3.6	2.5
Holbourne East	2	5	2.6	2.3	
Holbourne West	1	5	9.5	10.6	9.0
Holbourne West	2	5	8.8	7.0	
Camp Island East	1	2	2.3	3.0	2.7
Camp Island East	2	2	3.5		
Camp Island West	1	2	9.3	6.7	8.0
Camp Island West	2	2	6.1		

3.6.3 Wave height data

Wave height and direction during the passage of Tropical Cyclone Debbie was sourced from a NORTEK AWAC acoustic wave and current profiler maintained by AIMS. This sensor was located on the seabed approximately 2.5 km to the south of Holbourne Island.

3.6.4 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. Within this report scores are presented at the scale of individual indicators at each reef and depth, individual indicators and Report Card scores for each island, and the whole of area. For island-level scores each indicator score was estimated as the mean of indicator scores for each combination of aspect and depth, and Report Card scores as the mean of the three individual indicator mean scores. Similarly the whole of region scores were taken as the mean of island level means for each indicator and the Report Card score as the mean of these region-wide individual indicator scores. Grades for coral community condition were derived from the scores estimated above, according to the conversions described in Table 6.

Table 6 Indicator scores, condition descriptions and report card grade conversions.

Score	Condition description	Grade
> 0.8	Very good	**A**
> 0.6 ≤ 0.8	Good	**B**
> 0.4 ≤ 0.6	Satisfactory	**C**
> 0.2 ≤ 0.4	Poor	**D**
0 ≤ 0.2	Very poor	**E**

4 RESULTS

4.1 Environmental pressures

4.1.1 Coral bleaching

Over the summer of 2016/17 sea surface temperatures were anomalously high, with the mean accumulated degree heating days for the study sites estimated at 127.7 (Figure 3) . This value exceeds the threshold of 100 DHD at which severe coral bleaching is expected (Garde et al. 2014). At the time of the post summer surveys in mid-May and early June there was evidence of bleaching with individual colonies still bleached at most sites, in particular Holbourne West 5m where 5-10% of colonies where bleached. These estimates do not capture the full extent of the bleaching impact as corals that had died over the preceding months could not be included in these estimates. At site 1 Camp West it was clear that mortality had occurred with numerous colonies observed as being dead or partially dead at the time of survey. Substantially confounding the estimate of bleaching related impacts was exposure to the high seas generated by TC Debbie.

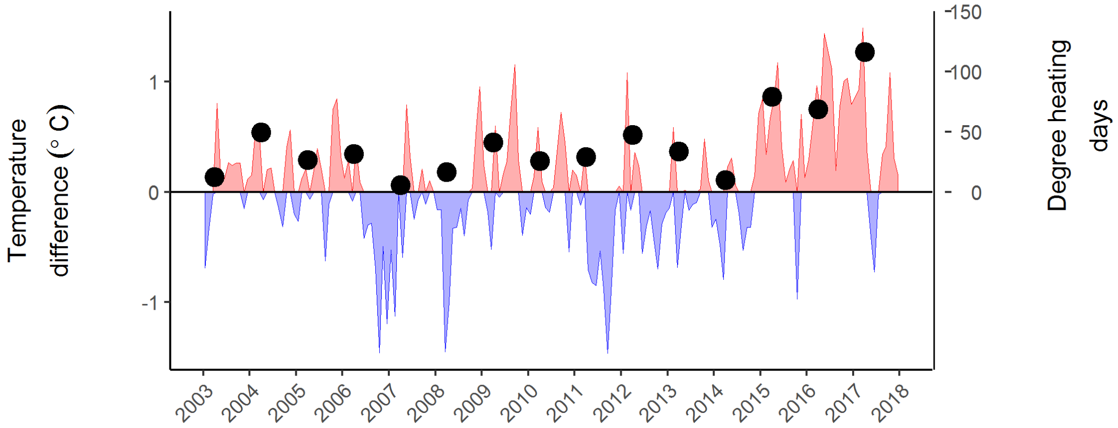

Figure 3 Temperature anomalies. Red and blue bars indicate mean monthly deviations. Black dots indicate annual degree heating days accumulated December 01 – March 31. All values are means among the points listed in Table 4 and are relative to IMOS 14 day climatology.

4.1.2 Tropical Cyclone Debbie

TC Debbie crossed the coastline on 28th March near Airlie Beach as a category 4 cyclone (Figure 4a) with wind speeds in the order of 150 + km/h. The slow movement of TC Debbie, averaging 7km/hr, caused a persistent exposure to storm waves over large areas of the reef, including Holbourne Is (Figure 4b). Wave monitoring adjacent to Holbourne Island indicated mean set waves in excess of 7.5 m form a north westerly direction between 2pm and 8pm on the 27th of March. From 8pm to 11pm higher set waves with a mean height of 8 m and a maximum height of 11.26 m were recorded however the direction was not captured by the monitoring equipment. Based on the track of the cyclone it is likely these highest waves were also from a northwest to northerly direction. As the storm passed the direction of highest waves swung to the southeast with the mean height of set waves exceeding 5m between 2am and 8am on the 28th of March (Figure 5). This swing in direction explains the severe damage to coral communities at Holbourne Island. The directions of maximum wave energy also explain the limited damage at Camp Island that is sheltered by the mainland to both the northwest and southeast. Historically, the highest recorded wave height measured by the Abbot Point wave buoy (deployed in 2012) was 6.5 m that occurred on the 13th April 2014 when

Tropical Cyclone Ita passed within 5km of Camp Island as a category 1 cyclone. This will have almost certainly caused substantial damage to coral communities both study locations.

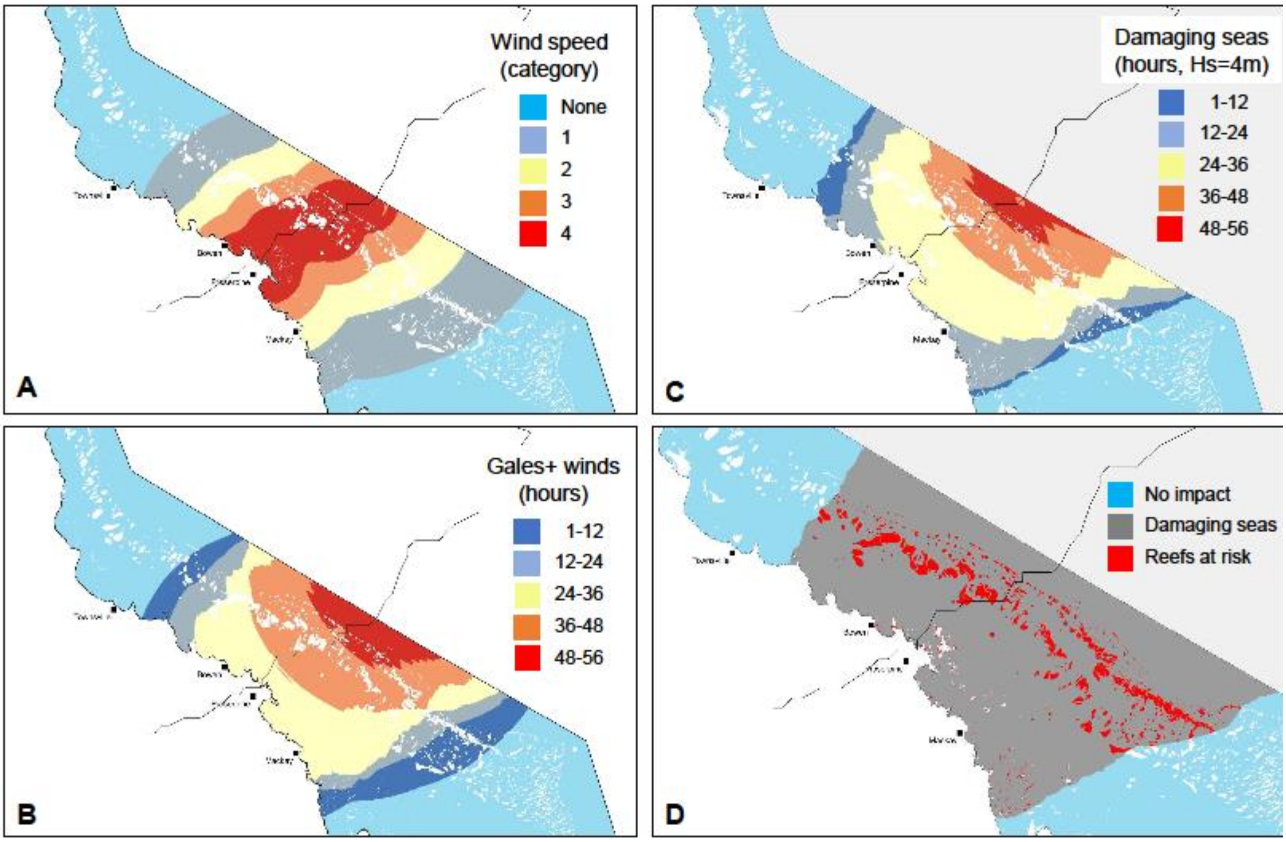

Figure 4 Tropical Cyclone Debbie track. Modelled exposure of GBR reefs to: A- maximum wind speeds of various strengths, B- gale force or higher winds (17 m/s), and C- the potential for very rough seas (characterised by 4m significant wave heights). Vulnerable colonies on reefs located in the very rough sea state zone (D) are likely to have been catastrophically damaged. Source: Marji Puotinen, AIMS-Perth. See Puotinen et al. (2016) for a detailed explanation of methods.

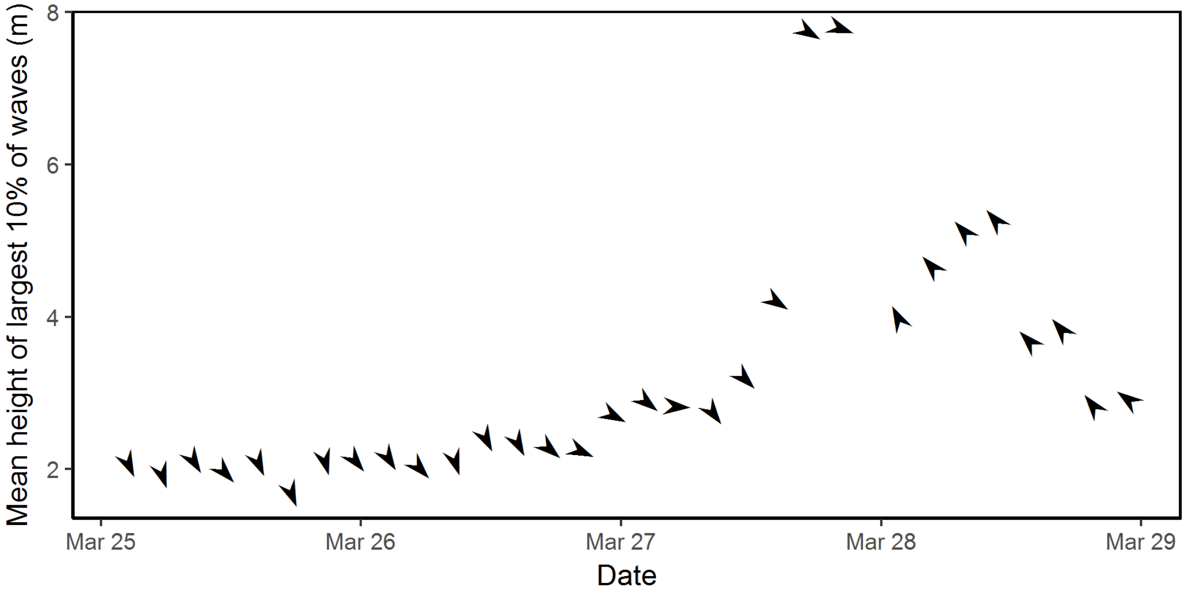

Figure 5 Wave record from Holbourne Island. Data sourced from Australiana Institute of Marine Science. Plotted are mean height of the largest 10% of waves (set waves) over each three hour period. Symbols represent the direction from which the waves were travelling. The highest record of 11.26m occurred at 11pm on the 27th-March and is not included as no direction was captured.

4.1.3 Runoff

An additional influence of TC Debbie was the resulting discharge from local rivers. River discharge data highlights a period of well above median flows including 2008, 2009 and then very high discharge in 2011 (Table 7). In 2017 discharge was again above median levels suggesting increased nutrient and sediment inputs, but also potentially exposing corals on the shallow reefs of Camp Island to low salinity waters. Obscured in annual flow summaries are the magnitude of individual flow peaks. Since record flooding in 1991, the highest single day discharges from both the Don and Elliot Rivers occurred on the 12th Feb 2008 with single day discharge exceeding the annual medians presented in Table 5. Single day exceedance of the Elliot River annual median also occurred in each of 2012, 2013, 2014. The proximity of Camp within 3km of the mouth of the Elliot River along with the shallow nature of the sites may expose the coral communities here to low salinity waters during such flood events. In 2017 single day exceedance of annual median discharge for both the Elliot and Don Rivers occurred on the 29th March, during the passage of TC Debbie, adding a further confounding pressure influencing the corals at Camp Island in 2017.

Table 7 Annual discharge form local rivers. Values expressed as proportion of median discharge over the period 1980-2010. Colour heat used to define level of exceedance of the median: yellow=2-3 times median, orange=3-5 times median, red=5-10 times median and dark red >10 time median.

River	Median discharge (ML)	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
Burdekin	4593112	6.0	6.4	1.7	7.6	3.4	0.8	0.3	0.2	0.4	0.9
Don	39986	7.2	3.9	2.4	14.5	2.0	2.0	0.7	0.7	0.3	2.9
Elliot	23568	6.2	6.0	2.2	8.7	4.1	2.3	1.3	0.4	0.5	2.9

4.1.4 Coral predators

In addition to the pressures imposed by the physical environment the abundance of coralivorous snails of the genus *Drupella* were higher than previously recorded at both Camp East and West as well as at the 5m depth at Holbourne East. Also observed at Holbourne East were three juvenile crown-of-thorns starfish (Table A1. 7). At Holbourne East, in particular, the predation of surviving coral fragments is likely to have contributed to the slight reduction of coral cover between surveys in May and October 2017 (Table 12).

4.2 Coral community condition assessment

The overall coral index score for coral communities at Camp and Holbourne Islands declined from 0.36 in October 2016 to 0.26 in October 2017 remaining in grade D, categorised as poor condition (Table 8 & Table 9). This decline is primarily due to the impact of TC Debbie that caused clear damage to the coral communities at Holbourne; where the index declined from 0.53 in 2016 to 0.33 in 2017. In contrast, whilst there was a minor decline in the index at Camp from 0.18 to 0.16, direct impact from the cyclone was limited with thermal stress and possibly localised flooding more likely implicated in the observed decline. To provide a fuller understanding of the changes in these reef communities, and the index, we consider each of the three indicators separately.

Table 8 Indicator values for Abbot Point Region.

	Year	Season	Juvenile Density (m^2)		Coral Cover (%)		Proportion Macroalgae (%)	
			Mean	SD	Mean	SD	Mean	SD
Regional summary	2016	May	3.66	1.68	27.69	10.17	28.58	40.17
		October	2.88	1.30	26.12	14.31	38.14	52.50
	2017	May	1.71	1.50	12.13	7.47	25.10	35.33
		October	1.97	1.86	9.96	4.65	32.41	42.94

Table 9 Indicator scores for Abbot Point Region.

	Year	Season	Juvenile Density	Coral Cover	Proportion of Macroalgae	Report Card	
						Score	Grade
Regional Scores	2016	May	0.32	0.37	0.50	0.40	D
		Octobe	0.26	0.35	0.47	0.36	D
	2017	May	0.16	0.16	0.50	0.27	D
		Octobe	0.18	0.13	0.42	0.25	D

4.3 Coral Cover

Tropical Cyclone Debbie significantly reduced the cover of both hard and soft corals at Holbourne Island (Table 10). Between October 2016 and October 2017, 78% of the coral cover was lost from Holbourne East and 86% from Holbourne West. The level of coral cover remaining at each site was below 15% at all Holbourne Island aspect and depth combinations resulting in Coral Cover scores within the very poor category (Table 10). During surveys in May 2017 the influence of TC Debbie was obvious. The majority of soft corals present in 2016 had disappeared, as had much of the hard coral (, Figure A1. 1, Figure A2. 1 – A2. 5). Predictably, fragile branching *Acropora* and plate-like *Montipora* corals, that dominated these sites in 2016, were heavily impacted (collectively Acroporidae, Figure A1. 1). Remaining hard corals showed clear evidence of physical damage with many being either toppled, fragmented or abraded (physical damage, Table A1. 7). The predation occurring at Holbourne East, in particular that attributed to crown-of-thorns starfish, is likely compounding the effects of the cyclone and suppressing the early phase of recovery, as if not eaten, the remaining fragments of corals on which predation was occurring would otherwise be expected to grow and contribute to increased cover. *Porites* corals provided a mainstay during the cyclone, with cover maintained between surveys. The impact of the cyclone decreased with depth with remaining coral cover higher at 5m than at 2m depths at both Holbourne East and West (Table 11).

Table 10 Coral cover and indicator scores for late dry season observations in 2016 and 2017. Coral cover scores are coloured as per Figure 4. Statistical probability of no change in cover between years are indicated as > 5% (ns), < 5% (*), less than 1% (**) and less than 0.1% (***) against 2017 observations.

	Aspect	Depth	Month	Year	Hard Coral cover	Soft Coral cover	Coral cover	Coral cover Score
Camp	East	2m	October	2016	4.2	0	4.2	0.06
			November	2017	11.8***	0 (n/a)	11.8***	0.16
	West	2m	October	2016	27.6	0.2	27.8	0.37
			November	2017	14.2***	0.6 (ns)	14.8***	0.20
Holbourne	East	2m	October	2016	11.0	10.1	21.1	0.28
			October	2017	3.8***	1.3***	5.1***	0.07
		5m	October	2016	38.0	19.1	57.1	0.76
			October	2017	7.6***	4.8***	12.4***	0.17
	West	2m	October	2016	16.8	14.4	31.1	0.42
			October	2017	0.9***	0***	0.9***	0.01
		5m	October	2016	20.3	15.4	35.6	0.48
			October	2017	7.5***	0.8***	8.3***	0.11

Table 11Spatial differences in coral cover among monitoring locations. Results of contrasts based on generalised linear mixed models applied to hard coral cover and soft coral cover observed in October/November 2017. Statistical probability of no difference are indicated as > 5% (ns), < 5% (*), less than 1% (**) and less than 0.1% (***).

	Depth	Reef	Aspect
Hard Coral	Higher at 5m at both Holbourne East (***) and West (***)	No difference between Camp and Holbourne	No difference 5m depths at Holbourne 2m depth – Holbourne East higher than Holbourne West (**), No difference at Camp (ns)
Soft Coral	Higher at 5m at Holbourne East (***) No difference Holbourne West (*)	Higher at Camp (***)	5m depth higher at Holbourne East (**). 2m depth do difference at Camp (ns). Insufficient cover to model at Holbourne
Coral Cover	Higher at 5m at both Holbourne East (***) and West (***)	No difference between Camp and Holbourne	Higher at Holbourne East 2m (***) and 5m (**). No difference Camp (ns)

In addition to the loss of corals was evidence of substantial relocation of sand and rubble, toppling of corals and removal of consolidated reef framework. This was particularly evident at Holbourne West where this shifting of material along with the distortion of steel transect markers (Figure A2. 5) allows some appreciation of the forces to which corals were exposure during the passage of TC Debbie.

In contrast, there was little evidence for damage associated with TC Debbie at Camp Island. At Camp East coral cover increased from 4.2% in October 2016 to 11.8% in November 2017 (Table 10) with much of this increase occurring between October 2016 and June 2017, when cover was 10.4%. As the observed increase in coral cover coincided with a decrease in the cover of macroalgae (Table 13) there is likely some over-estimate in the level of increase between surveys.

When using the photo point intercept technique the detection of corals is reduced in the presence of tall macro algae species, such as *Sargassum*, that can over-top corals and so obscure them from detection. At Camp West, coral cover in November 2017 was significantly lower than that observed in October the previous year (Table 10). While coral cover at Camp West did decline between October 2016 and June 2017, the majority of the decline occurred between June and November 2017 coinciding with very high levels of the disease, atramentous necrosis amongst the *Montipora* community noted in November (Figure 6, Table A1. 7). The seasonal increase in this disease was also noted in 2016 suggesting ongoing mortality post surveys in October 2016 was likely. During surveys of Camp West, site 1, in May 2017 a number of recently dead corals of the genus *Acropora* and also some of the large *Pavona* colonies, unique to this site, were noted. There was some ongoing mortality to the *Acropora* documented as disease (Table A1. 7). That these colonies were in situ indicated that the mortality was not the result of exposure to physical damage during TC Debbie. A more likely explanation is that the mortality was associated with coral bleaching as a result of high summer temperatures; although, flash flooding of the Elliot River during the passage of TC Debbie may have added reduced salinity to the stresses imposed on these corals. By November 2017 the remaining *Acropora* were in better health with only four colonies showing signs of disease or ongoing partial mortality of unknown cause compared to the 15 colonies so classified in June. While elevated levels of 'atramentous necrosis' disease has being linked to exposure to high water temperatures (Jones et al. 2004). Anthony et al. (2008) caution the black lesions that lead to the 'atramentous necrosis' classification of the disease are likely caused by a secondary infection and that the cause of the primary infection remains unknown and not necessarily linked to high water temperature. That we saw the full range of phases of this disease from small patches of bleached tissue though to almost complete colony mortality in November 2017 (Figure 6) and a presentation of this disease in each survey since May 2016 demonstrates the ongoing susceptibility of the *Montipora* to disease at Camp Island.

Despite the severity of impact to coral communities at Holbourne that sees coral cover in 2017 being lower than that observed at Camp (Table 11), the benthic communities remain distinct. The relative magnitude of impacts to communities at each location can be visualised in biplot of community composition (Figure 7). The x-axis on this plot explains the most variability in community composition and clearly separates locations at Camp from those at Holbourne, due largely to higher proportions of brown macro algae (Phaeophyta) and *Montipora* within the benthic communities at Camp compared to the higher representation of soft corals (*Cladiella*, *Nepthea*, *Klyxum*) and branching and bottlebrush forms of *Acropora* at Holbourne. It should be noted that less common algal groups and coral genera were included in the ordination represented by Figure (7) though, for clarity were not included on the figure. In contrast the y-axis of Figure (7) captures the change in abundance within these groups. Marked shifts in the vertical plane at Holbourne demonstrate that although the magnitude of cover changed dramatically this did not result in marked change in in the composition of the benthic communities. An exception was Holbourne West 5m where the significant increase in macroalgae (Table 13) contributes to the shift to the left between 2016 and 2017 (Figure 7). Conversely, the relatively minor impacts observed at Camp are illustrated by relatively minor movement of locations within the plot (Figure 7). A slight shift to the right at Camp East predominantly reflects reduced cover of macroalgae (Table 13).

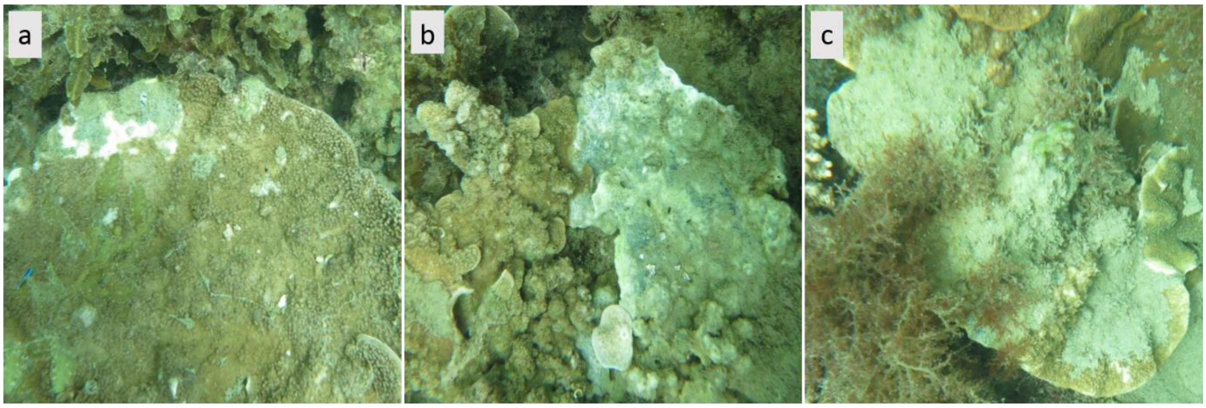

Figure 6 Sequence of ‘atramentous necrosis’ infection stages observed at Camp East in November 2017. a) small patches of bleached tissue and onset of secondary infection producing black bacterial community, b) classic presentation of secondary infection, c) colony mostly dead and covered with sediment.

Table 12 Coral cover and indicator scores observed in 2017. Scores are coloured as per Figure 4. Statistical probability of no change in Coral cover between May/June and October/November observations are indicated as > 5% (ns), < 5% (*), less than 1% (**), less than 0.1% (***) and (n/a) when cover was insufficient to analyse.

	Aspect	Depth	Month	Hard Coral cover	Soft Coral cover	Coral cover	Coral cover Score
Camp	East	2m	June	10.4	0	10.4	0.14
			November	11.8 (ns)	0 (n/a)	11.8 (ns)	0.16
	West	2m	June	23.9	0.5	24.4	0.33
			November	14.2***	0.6 (n/a)	14.8 ***	0.2
Holbourne	East	2m	May	3.1	0.5	3.6	0.05
			October	3.8 (ns)	1.3 (n/a)	5.1 (ns)	0.07
		5m	May	8.1	6.1	14.2	0.19
			October	7.6 (ns)	4.8 (ns)	12.4 (ns)	0.17
	West	2m	May	0.8	0	0.8	0.01
			October	0.9 (ns)	0 (n/a)	0.9 (ns)	0.01
		5m	May	7.5	1.3	8.8	0.12
			October	7.5 (ns)	0.8 (ns)	8.3 (ns)	0.11

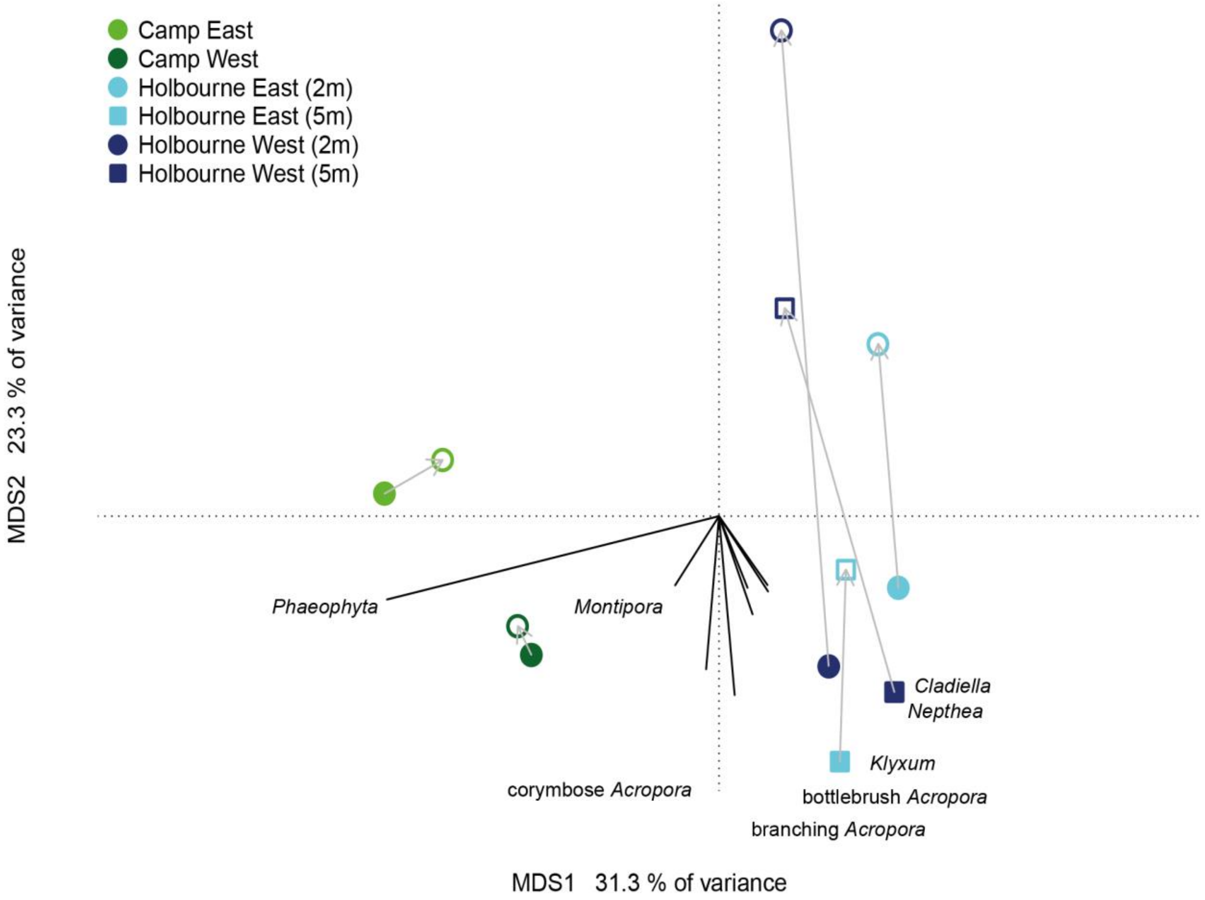

Figure 7 Benthic community composition biplot. PCoA biplot of hard coral, soft coral, and macroalgal community composition based on proportional cover of these groups at each location (combination of island, aspect and depth) and year in the monitoring design. Locations are defined by a combination of Symbol and colour (see legend). Filled symbols indicate observations from October 2016 and open symbols observations from October/November 2017. Position relative to the first two dimensions of Euclidian space represent the ecological distance (dissimilarity) between communities. Arrows are included to highlight the observed change in community composition for each location. Lines (eigenvectors) represent the relative abundance of genera. Eigenvectors begin at the plot origin and extend in the direction of increasing abundance of a given genus, for example, Camp West has higher cover of the brown algae family (Phaeophyta) than all locations to the right of the plot. The length of the eigenvectors indicates the relative magnitude of cover differences among locations. For clarity, only a subset of eigenvectors that represent the genera that vary the most among locations are included on the plot.

4.4 Macroalgae proportion

Observations from May 2017 at Holbourne Island revealed very low cover of macroalgae with levels at or below those observed in October 2016 (Table 13,). Between May and October 2017 macroalgae cover increased slightly resulting in significant increases in the Macroalgae proportion indicator at the 5 m depths on both the eastern and western aspects of the Island (Table 13). This increase also marked a significant increase when compared to observations from October 2016 (Table 14). The statistical significance of minor increase in cover observed at 2m depths could not be analysed due to the zero estimates of cover from May. Comparing October 2017 observations to those from October 2016 suggested a significant increase at the 2m depth (Table 14). In terms of indicator scores, despite these increases the resulting cover was still very low and it was only at

Holbourne West 5m that the score changed appreciably, declining from 1 and a categorisation of 'very good' in 2016 to 0.66 and a categorisation of 'good' in 2017 (Table 14). The algae that colonised Holbourne West were predominantly species of calcified red algae (Figure 8). In contrast, despite a reduction in cover of macroalgae from 2016 levels at both Camp East and West, cover remained well above the maximum threshold of 23% for this indicator ensuring scores remain at the minimum level of zero and a 'very poor' condition categorisation (Table 14).

Seasonality in the cover of macroalgae at Camp observed in 2016 (Thompson et al. 2017) was again observed in 2017 with a significant increase in cover between June and November at both eastern and western sites (Table 13). The increase in cover primarily reflects increased cover of the genus *Sargassum* between June and November (Table A1. 5). In contrast, the cover of *Dictyota,* a smaller, more ephemeral brown algae, declined from moderate levels in June to be at low levels or absent in November (Table A1. 5). Seasonality in the Red macroalgae was less consistent between the two aspects of Camp, with cover more than doubling from 4.38% to 10.13% at Camp West and declining slightly from 2.26% at Camp East (Table A1. 5).

Despite increases in macroalgae at Holbourne and declines at Camp in 2017, Camp continues to support significantly higher cover of macroalgae (Table 15).

Table 13 Macroalgae cover, proportional representation in the algal community, and indicator scores for observations in 2017. Macroalgae scores are coloured as per Figure 4. Statistical probability of no change in Macroalgae proportion cover between May/June and October/November observations are indicated as > 5% (ns), < 5% (*), less than 1% (**), less than 0.1% (***) and (n/a) when cover was insufficient to analyse.

	Aspect	Depth	Month	Macroalgae cover	Macroalgae proportion	Macroalgae Score
Camp	East	2m	June	58.6	68.2	0.00
			November	62.8	76.7***	0.00
	West	2m	June	24.1	32.0	0.00
			November	41.7	48.8***	0.00
Holbourne	East	2m	May	0	0	1.00
			October	0.1	0.1 (n/a)	1.00
		5m	May	0.3	0.4	1.00
			October	2.5	3.0***	0.74
	West	2m	May	0	0	1.00
			October	1.1	1.4 (n/a)	1.00
		5m	May	0.1	0.1	1.00
			October	2.9	3.7***	0.66

Table 14 Macroalgae cover, proportional representation in the algal community, and indicator scores for late dry season observations in 2016 and 2017. Macroalgae scores are coloured as per Figure 4. Statistical probability of no change in Macroalgae proportion cover between years are indicated as > 5% (ns), < 5% (*), less than 1% (**) and less than 0.1% (***) against 2017 observations.

	Aspect	Depth	Month		Year	Macroalgae cover	Macroalgae proportion	Macroalgae Score
Camp	East	2m	October		2016	87.9	93.8	0.00
			November		2017	62.8	76.7***	0.00
	West	2m	October		2016	42.3	56.7	0.00
			November		2017	41.7	48.8 (***)	0.00
Holbourne	East	2m	October		2016	0.1	0.1	1.00
			October		2017	0.1	0.1 (ns)	1.00
		5m	October		2016	1.2	3.2	0.72
			October		2017	2.5	3.0 (ns)	0.74
	West	2m	October		2016	0.3	0.5	1.00
			October		2017	1.1	1.4*	1.00
		5m	October		2016	0.1	0.2	1.00
			October		2017	2.9	3.7***	0.66

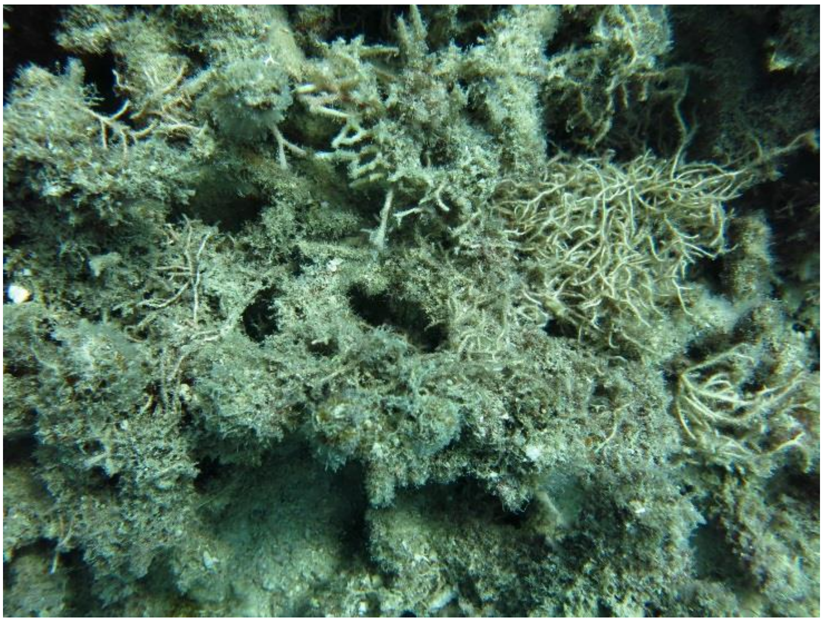

Figure 8 Example of the calcareous red algae present at Holbourne West in November 2017.

Table 15 Spatial differences in proportion of macroalgae in the algal community October/November 2017. Statistical support for differences in cover based on generalised linear mixed models. Statistical probability of no difference are indicated as >5% (ns), <5% (*), less than 1% (**) and less than 0.1% (***).

	Depth	Reef	Aspect
MA proportion	Higher at 5m at both Holbourne East (***) and West (**)	Higher at Camp (***)	5m – no difference with aspect (ns) 2m – no difference with aspect at either Holbourne (ns) or Camp (ns)

4.5 Juvenile Density

The impact of TC Debbie at Holbourne resulted in a significant decline in the density of juvenile hard corals at all depth and aspect combinations (Table 16). The severity of the impact of TC Debbie varied with depth, with juvenile densities in October 2017 significantly lower at 2m than at 5m depths (Table 17). At 2m depth juvenile densities where lower at Holbourne West than at Holbourne East mirroring the post cyclone distribution of coral cover (Table 17). In contrast, juvenile densities did not significantly decline at Camp Island (Table 16) although the lower density observed in 2017 at Camp West was sufficient to down grade the categorisation of the indicator score from 'moderate' to 'poor' and remove the previously observed difference in densities between Camp East and Camp West (Table 17). Despite densities at Camp being higher than at Holbourne (Table 17), it should be noted that the densities at Camp are in-themselves low, likely due the pressures the persistently high cover of macroalgae imposes on the early life history stages of corals (McCook et al. 2001, Foster et al. 2008, Hauri et al. 2010). Bolstering the density on both aspects of Camp was a high representation of the genus *Turbinaria* (Family Dendrophylliidae, Table A1. 5, Figure A1. 2). Within the inshore GBR high densities of *Turbinaria* are commonly observed on close-inshore reefs (Thompson et al. 2016). As with other inshore reefs monitored by the AIMS, a high proportion of *Turbinaria* in the juvenile community is not reflected in the composition of coral cover (compare family Dendrophylliidae in Figure A1. 1 and Figure A1. 2) suggesting that these juveniles may suffer high rates of mortality reducing their contribution to recovery of coral cover. Excluding *Turbinaria,* the density of juvenile hard corals at Camp East would be very low (Figure A1. 2). Comparing the 2017 density of juvenile corals observed in May/June to those in October/November identifies an increase at Camp East supported by increased abundance of *Montipora* and *Turbinaria* juveniles (Table 18, Table A1. 6). Given the timing of these surveys relative to settlement periods over summer this increase most likely indicates the growth of juveniles to a size making them more available to survey.

Table 16 Juvenile hard coral abundance, density (m^{-2}) and indicator scores for late dry season samples in 2016 and 2017. Juvenile scores are coloured as per Figure 4. Statistical probability of no change in Juvenile density between years are indicated as > 5% (ns), < 5% (*), less than 1% (**) and less than 0.1% (***) against 2017 observations.

	Aspect	Depth	Month	Year	Juvenile abundance	Juvenile density	Juvenile Score
Camp	East	2m	October	2016	90	2.8	0.25
			November	2017	88.5	3.1 (ns)	0.28
	West	2m	October	2016	71	4.8	0.43
			November	2017	84.5	3.4 (ns)	0.31
Holbourne	East	2m	October	2016	20	1.0	0.09
			October	2017	5	0.2***	0.02
		5m	October	2016	28.5	2.3	0.21
			October	2017	20	0.8***	0.07
	West	2m	October	2016	38.5	2.0	0.18
			October	2017	9.5	0.4***	0.04
		5m	October	2016	46.5	2.5	0.23
			October	2017	33.5	1.2***	0.11

Table 17 Spatial and differences in density of juvenile hard corals October/November 2017. Statistical support for differences in cover based on generalised linear mixed models. Statistical probability of no difference are indicated as >5% (ns), <5% *, less than 1% ** and less than 0.1% ***.

	Depth	Reef	Aspect
Juvenile density	Higher at 5m at both Holbourne East (***) and West (**)	Higher at Camp (***)	No effect of aspect at Camp (ns) or Holbourne 5m (ns), Higher at Holbourne East 2m (*)

Table 18 Juvenile hard coral abundance, density (m^{-2}) and indicator scores for 2017 observations. Juvenile scores are coloured as per Figure 4. Statistical probability of no change in Juvenile density between May/June and October/November observations are indicated as > 5% (ns), < 5% (*), less than 1% (**) and less than 0.1% (***) against October/November observations.

	Aspect	Depth	Month	Juvenile abundance	Juvenile density	Juvenile Score
Camp	East	2m	June	64	2.2	0.20
			November	88.5	3.1 **	0.28
	West	2m	June	78	3.4	0.31
			November	84.5	3.4	0.31
Holbourne	East	2m	May	9.5	0.4	0.04
			October	5	0.2 (ns)	0.02
		5m	May	17	0.7	0.06
			October	20	0.8 (ns)	0.07
	West	2m	May	8.5	0.3	0.03
			October	9.5	0.4 (ns)	0.04
		5m	May	31	1.2	0.11
			October	33.5	1.2 (ns)	0.11

5 CONCLUSIONS

Tropical Cyclone Debbie severely impacted the coral community at Holbourne Island, drastically reducing both coral cover and the density of juvenile corals. Although this event clearly impacted the current state of the communities, as reflected in reduced index scores, coral communities are naturally exposed to acute disturbances and it is how they recover in subsequent years that provides a more balanced assessment of the 'health' of the system. In the longer term, the time-series of coral cover derived from this monitoring program will allow the direct assessment of recovery rate during periods free from acute disturbance events. Until such observations are available the potential for recovery can be inferred from the indicators included in this report.

Recovery from severe disturbance necessarily relies on a combination of the ongoing survival and growth of both the remaining corals and new recruits, both of which may be supressed by ongoing environmental pressures or limiting processes. Positive attributes that suggest potential for recovery at Holbourne are the low representation of macroalgae, the high representation of rapidly growing coral species observed prior to TC Debbie, in particular coral of the family the 'Acroporidae', and low levels of coral disease. In combination these attributes all suggest limited influence of chronic pressures associated with the biogeochemical setting of the reef. In contrast, the low density of juvenile corals observed in 2016 and then further reduced by the cyclone potentially indicate limited larval supply at this reef. Holbourne Island is relatively isolated compared to the more densely arranged and large reefs further offshore, or the fringing reefs surrounding islands at headlands closer to the coast. This geographic setting raises the potential that isolation from brood-stock may be limiting larval supply; a situation that can limit the recovery potential of coral communities (Underwood et al. 2009).

As at the end of 2017 region coral cover was low due to the combined impacts of TC Debbie, that occurred on reefs to the south, and coral bleaching and crown-of-thorns starfish outbreaks that have resulted in loss of coral cover to the north (AIMS unpublished data). The large geographic footprint of these impacts, in combination with local losses, are likely to have reduced the population of corals that have the potential to supply larvae to Holbourne Island. Against low recruitment of corals the survival and growth of colonies or fragments of coral surviving TC Debbie becomes increasingly important. The presence of juvenile crown-of-thorns starfish observed in 2017 are, as such, an added concern for the recovery of the coral community at Holbourne Island. Holbourne Island has been surveyed periodically by AIMS since 1987, when an outbreak of crown-of-thorns starfish was noted. Recovery following this outbreak was slow with reef-wide coral cover remaining below 10% for at least a decade (http://apps.aims.gov.au/reef-monitoring/reef/19103S). A caveat on interpreting this time series is that additional disturbances, such as coral bleaching in 1998 and 2002 and cyclone Aivu in 1989 are likely to have further impacted coral communities and so obscured any intervening recovery. Although the density crown-of-thorns starfish observed in 2017 was relatively low compared to that observed on other reefs, the accessibility of Holbourne Island to Bowen elevates the societal value of this reef. This value, coupled with the small area (that would make control efforts more successful) may elevate this reef as a priority for crown-of-thorns population control.

While there was no evidence that TC Debbie impacted the coral communities at Camp, mortality noted in June 2017 among the *Acropora* at Camp West did suggest the acute impact of high water temperatures over the preceding summer. In addition, there is strong evidence that the underlying environmental conditions of the location contribute to the poor condition of coral communities.

The high cover of both brown (Phaeophyta) and red (Rhodophyta) macroalgae clearly differentiate the benthic communities at Camp Island from those at Holbourne Island, and point to the additional pressure imposed by the availability of nutrients, at Camp Island; within the inshore GBR persistently high macroalgae cover is limited to areas with mean chlorophyll *a* concentrations exceeding 0.4ugL^{-1} (Thompson et al. 2016). A secondary influence of high nutrient availability is the detrimental influence that macroalgae have on the early life history stages of corals (McCook et al. 2001; Hauri et al. 2010). Any influence of macroalgae on juvenile settlement and survival will compound with potential regional limitation to larval supply mentioned previously. Finally the availability of nutrients has been reported as a chronic pressure linked to increased susceptibility of corals to disease (Vega Thurber et al. 2013), a point supported by the persistence of disease amongst coral communities at Camp, and in particular the *Montipora*.

The biannual sampling incorporated in this program is beginning to show some consistent seasonality in the responses of benthic communities at Camp Island. Seasonality in macroalgae has been documented for *Sargassum* on coral reefs at Magnetic Island where plants have been describes as growing through the warmer summer months before shedding their thallus and persisting as holdfasts through winter (Vuki and Price 1994). In both 2016 and 2017 we observed clear increase in the cover of macroalgae, predominantly due to increased cover of Sargassum, between surveys in May/early June and October/November. These observations suggest an earlier regrowth of Sargassum beginning in spring rather than summer, certainly the plants in October/November were noticeably less fouled with epiphytic organisms supporting the interpretation that this was fresh growth. The limited time Sargassum exists in the holdfast only state has also been noted anecdotally on other reefs monitored by AIMS. In contrast *Dictyota* was observed to decline in cover between the autumn and spring surveys. Seasonality has also emerged in the incidence of diseased colonies amongst the *Montipora*. Considering the observed seasonality, and the potential that changes in the tall macroalgae species occupying Camp Island appear to also bias coral cover estimates, any future inter-annual comparisons need to be made with consideration of this seasonality. There is much less seasonality evident at Holbourne Island although slight increase in red macroalgae cover did occur between samples in May and October 2017. Despite limited seasonality it will be prudent to compare trends in the recovery at this reef from samples taken at the same time of year in order to reduce any possible seasonal influences not detected to date. Given that most disturbances to coral communities occur over summer months and the reduced abundance of macroalgae observed in the late wet season samples we suggest sampling in April/May should annual rather than biannual sampling be considered in the future. Such a reduction in sampling would not greatly reduce the ability of the program to detect recovery of coral communities as the rate of increase in coral cover is typically sufficiently slow that real changes within sub-annual timeframes are likely to be within the sampling error of cover estimates.

6 REFERENCES

Akaike H (1974) A new look at the statistical model identification, Automatic Control, IEEE Transactions on Automatic Control, 19(6):716-723

Anthony SL, Page CA, Bourne DG, Willis (2008) Newly characterized distinct phases of the coral disease 'atramentous necrosis' on the Great Barrier Reef. Diseases of Aquatic Organisms, 81:255-259

Bates D, Maechler M, Bolker B, Walker S (2015) Fitting Linear Mixed-Effects Models Using lme4. Journal of Statistical Software, 67(1):1-48

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

Bolker BM, Brooks ME, Clark CJ, Geange SW, Poulsen JR, Stevens MHH, White JSS (2009) Generalized linear mixed models: a practical guide for ecology and evolution. Trends in Ecology Evolution, 24(3):127-135

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

Cheal AJ, , Emslie M, MacNeil MA, Miller I, Sweatman H (2013) Spatial variation in the functional characteristics of herbivorous fish communities and the resilience of coral reefs Ecological Applications, 23:174–188

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

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

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

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 RJ, Bower J, Hoegh-Guldberg O, Blackall LL (2004) Dynamics of a temperature-related coral disease outbreak. *Marine Ecology Progress Series* 281:66-77

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

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

McCulloch CE, Neuhaus JM (2013) Generalized linear mixed models. John Wiley Sons, Ltd.

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

Queensland Government (2016) GBR 2015 report card detailed results. http://www.reefplan.qld.gov.au/measuring-success/report-cards/2015/assets/gbr-2015report-card-detailed-results.pdf

Puotinen M, Maynard JA, Beeden R, Radford B, Williams GJ (2016) A robust model for predicting where tropical cyclone waves damage coral reefs. Scientific Reports 6:26009. doi: 10.1038/srep26009.

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

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 policy-relevant 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 (2017) Port of Abbot Point Ambient Monitoring Program: Baseline Report. Report prepared for North Queensland Bulk Ports. Australian Institute of Marine Science, Townsville 35p

Underwood JN, Smith LD, van Oppen MJH, Gilmour JP (2009) Ecologically relevant dispersal of corals on isolated reefs: implications for managing resilience. *Ecological Applications*, 19: 18–29. doi:10.1890/07-1461.1

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 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

Vega Thurber RL, Burkepile DE, Fuchs C, Shantz A, McMinds R, Zaneveld JR (2013) Chronic nutrient enrichment increases prevalence and severity of coral disease and bleaching. Global Change Biology, doi:10.1111/gcb.12450

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

7 APPENDIX 1

Table A1. 1 Waypoints and detailed compass directions for monitoring sites, updated after repair of sites following Cyclone Debbie.

Reef	**Latitude**	**Longitude**	**Depth**	**Site**	**Tran**	**Compass directions**
Holbourne East	19.7332 E	148.3644 S	2	1	1	300
					2	205, 270@10m
					3	220, 280@10m
					4	280, 240@12m
					5	270, 230@7m
Holbourne East	19.7338 E	148.3647 S	5	1	1	215, 150@13m
					2	240, 210@15m
					3	230, 260@10m
					4	270, 315@6m
					5	300
Holbourne East	19.7336 E	148.3618 S	2	2	1	265, 300@5m, 230@15m
					2	260
					3	260, 250@reo
					4	280, 220@15m
					5	240, 330@4m, 280@10m
Holbourne East	19.734 E	148.3618 S	5	2	1	210, 200@10m
					2	270, 255@10m
					3	240, 290@2m
					4	320, 0 toT5
					5	320, 290@10m
Holbourne West	19.7252 E	148.3547 S	2	1	1	10, 20@6m
					2	10, 20@10m
					3	340
					4	20
					5	10, 340@5m
Holbourne West	19.7249 E	148.3545 S	5	1	1	40, 340@reo, 50 to T2
					2	350, 0@8m, 300@15m
					3	30, 330@5m, 40@10m
					4	60, 40@3m, 300@5m, continue around bommie
					5	330, 25@10m
Holbourne West	19.7233 E	148.3556 S	2	2	1	345, 0@10m
					2	25, 75@10m
					3	10, 20@10m
					4	330, 20@5m, 100@15m
					5	50, 340@8m

Table A1. 1 continued.

Reef	**Latitude**	**Longitude**	**Depth**	**Site**	**Tran**	**Compass directions**
Holbourne West	19.7232 E	148.3553 S	5	2	1	30
					2	350, 300@10m
					3	30, 20@7m, backtrack 230 to T4
					4	0, 340@10m
					5	100, 110@8m, 170@11m
Camp East	19.8508 E	147.9052 S	1.5	1	1	170
					2	270
					3	225, 230@10m
					4	210, 220@10m
					5	200
Camp East	19.8541 E	147.9012 S	1.5	2	1	230
					2	270
					3	235
					4	240, 200@10m
					5	170, 155@7m
Camp West	19.8533 E	147.8942 S	2	1	1	340, 0@10m
					2	0
					3	0, 35@11m
					4	345, 300@8m, 330@11m
					5	20, 10@7m
Camp West	19.8512 E	147.8950 S	2	2	1	40, 30@10m
					2	345, 30@8m
					3	30
					4	130
					5	20

Table A1. 2 Indicator values and scores at each reef and depth combination.

REEF	Month	Coral cover	Macroalgae proportion	Juvenile density	Coral cover Score	Juvenile score	Macroalgae score	Index
Camp East	June	10.38	68.18	2.16	0.14	0.2	0	0.11
	November	11.75	76.72	3.12	0.16	0.28	0	0.15
Camp West	June	24.44	31.98	3.37	0.33	0.31	0	0.21
	November	14.75	48.83	3.44	0.2	0.31	0	0.17
Holbourne East 2m	May	3.56	0	0.39	0.05	0.04	1	0.36
	October	5.06	0.08	0.2	0.07	0.02	1	0.36
Holbourne East 5m	May	14.25	0.39	0.66	0.19	0.06	1	0.42
	October	12.38	3.04	0.75	0.17	0.07	0.74	0.32
Holbourne West 2m	May	0.81	0	0.35	0.01	0.03	1	0.35
	October	0.94	1.4	0.41	0.01	0.04	1	0.35
Holbourne West 5m	May	8.75	0.08	1.2	0.12	0.11	1	0.41
	October	8.31	3.68	1.23	0.11	0.11	0.66	0.29

Table A1. 3 Cover of hard coral genera. Genus with a maximum cover of at least 1% at any reef are included. All less abundant genera are grouped as Other HC.

Reef	Month	*Acropora*	*Coscinaraea*	*Montipora*	*Pavona*	*Porites*	*Symphyllia*	Other HC
Camp East	June	1.19	0.06	7.94		0.38		0.81
	November	1.19		8.94		0.31		1.31
Camp West	June	8.00		8.69	3.75	1.38	0.06	2.06
	November	6.75		3.44	1.69	1.00		1.31
Holbourne East 2m	May	0.19	0.38	0.69		0.06	1.06	0.69
	October		1.31	1.75		0.25	0.31	0.13
Holbourne East 5m	May	4.63		0.69		1.00	0.06	1.75
	October	3.94	0.06	0.25		1.44	0.06	1.81
Holbourne West 2m	May			0.25		0.19		0.38
	October			0.06		0.31		0.56
Holbourne West 5m	May	0.06		3.75	0.31	1.06	0.06	2.25
	October	0.06		3.06		1.56	0.25	2.56

Table A1. 4 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	Month	*Briareum*	*Cladiella*	*Nephthea*	*Sinularia*	Other SC
Camp East	June					
	November					
Camp West	June	0.25			0.25	0.25
	November	0.31	0.06		0.19	0.31
Holbourne East 2m	May	0.31	0.19			0.31
	October	0.69	0.50	0.06	0.06	0.69
Holbourne East 5m	May	2.81	0.44	1.31	1.25	2.81
	October	2.06	1.00	0.56	1.00	2.06
Holbourne West 2m	May					
	October					
Holbourne West 5m	May		0.06	0.06	1.00	
	October		0.13		0.50	

Table A1. 5 Cover of Algae. Identified macroalgae genera with a cover of at least 1% at any reef are separated. All less abundant genera and smaller algae are grouped.

Reef	Month	Brown macroalgae					Red macroalgae		Green macroalgae	Coralline algae	Cyanobacteria	Turf algae
		Dictyota	*Lobophora*	*Sargassum*	*Spatoglossum*	Other	*Hypnea*	Other				
Camp East	June	19.38	2.06	29.69	0.50	4.69	0.63	1.63		0.38		26.88
	November		0.44	59.13	1.06	1.00	0.50	0.63		1.25		17.69
Camp West	June	10.50	0.94	7.63		0.69	1.19	3.19		1.75		41.38
	November	0.56	0.19	30.13		0.69	2.00	8.13		0.63		33.25
Holbourne East 2m	May									0.69	0.06	71.13
	October					0.06				0.19		74.38
Holbourne East 5m	May		0.06					0.06	0.12	0.81		73.00
	October		1.44			0.06		0.94	0.06	0.63	0.13	74.19
Holbourne West 2m	May											62.19
	October							1.06	0.06		0.50	62.75
Holbourne West 5m	May							0.06		0.94		71.44
	October		0.06					2.88		0.19	1.50	75.00

Table A1. 6 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 abundance genus grouped as “Other genera”.

Reef	Month	*Acropora*	*Cyphastrea*	*Favia*	*Favites*	*Fungia*	*Goniopora*	*Lobophyllia*	*Montipora*	*Moseleya*	*Pocillopora*	*Porites*	*Psammocora*	*Stylophora*	*Turbinaria*	Other genera
Camp East	June	13.5	2	2	1.5	15.5	0.5		4.5	0.5	0.5	0.5	0.5		34.5	3.5
	November	11.5	2	1.5	1.5	13.5	2		14	0.5	0.5	0.5	2		48	4.5
Camp West	June	30	5	1	1.5	23.5			14	3	2.5	4.5	1		9.5	6
	November	30.5	0.5	3	1.5	13			17	2.5	4	5.5	3		11.5	5.5
Holbourne East 2m	May	5.5		1				0.5			0.5	0.5				1.5
	October	3		0.5	0.5							0.5				0.5
Holbourne East 5m	May	2.5	0.5		1			0.5	2.5		0.5	5		1.5		3
	October	6.5						2	4		0.5	2		2		3
Holbourne West 2m	May	2.5		0.5	0.5	0.5			1			1.5				2.5
	October	0.5		0.5					2		0.5	3.5				2.5
Holbourne West 5m	May	3	2	2.5	2		0.5	3	9.5		0.5	2.5				5.5
	October	3		2	0.5			5.5	10		1	3	1.5			7

.

Table A1. 7 Coral health survey results for 2017.

		Camp East		Camp West		Holbourne East				Holbourne West			
						2m		5m		2m		5m	
DAMAGE	GENUS	Jun	Nov	Jun	Nov	May	Oct	May	Oct	May	Oct	May	Oct
Disease (colonies)	*Acropora*			12	1			8	3			1	
	Goniastrea											2	
	Lobophyllia								1				
	Montipora	2	30	4	36			1	2		1		
	Pavona			2									
	Pocillopora			3				1					
Unknown cause (colonies)	*Acropora*	1		3	3								
	Favites											1	
	Hydnophora											1	
	Montipora	4	1										
	Pocillopora	3							1				
	Stylophora									1			
crown-of-thorns (number of seastars)									3				
crown-of-thorns scar (colonies)	*Acropora*								7				
	Montipora								1				
Drupella (number of snails)	*Acropora*				1			2	105				
	Montipora	1	3				2		1				
	Pocillopora								7				
Sponge - *Cliona orientalis* (on coral colonies)	*Favites*			1									
	Montipora		1					1	1				
	Porites							1			1		1
Cyclone/Physical (proportion of colonies)		0.1%	0.1%	2.4%		94%		79%	0.05%	94%		62%	
Bleaching (proportion of colonies)		0.05%	0.05%	0.05%				0.8%		0.05%		6.9%	

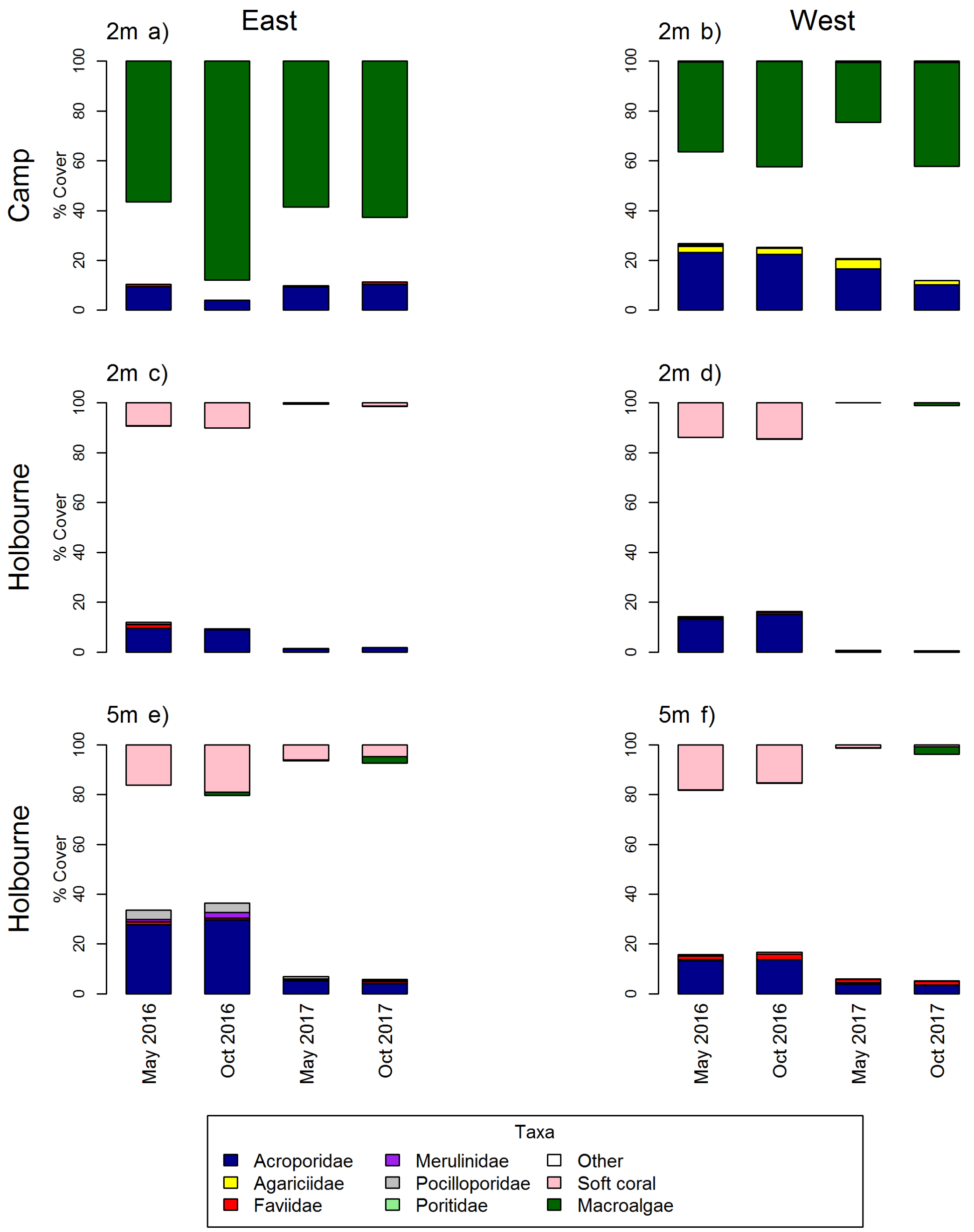

Figure A1. 1 Cover hard coral families, soft coral (hanging) and macroalgae (hanging).

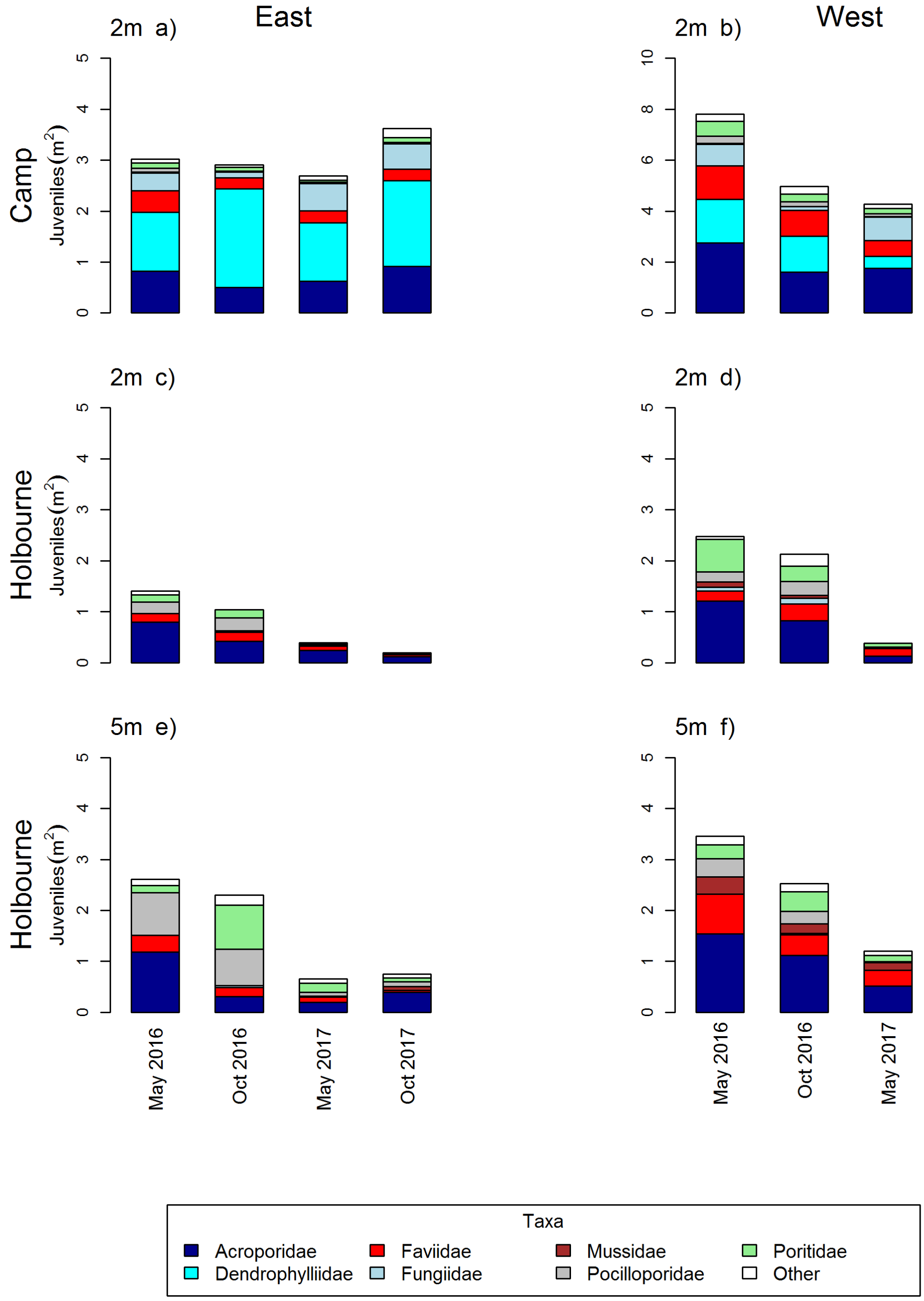

Figure A1. 2 Density of hard coral juveniles by family.

8 APPENDIX 2
ILLUSTRATION OF IMPACTS OF TC DEBBIE

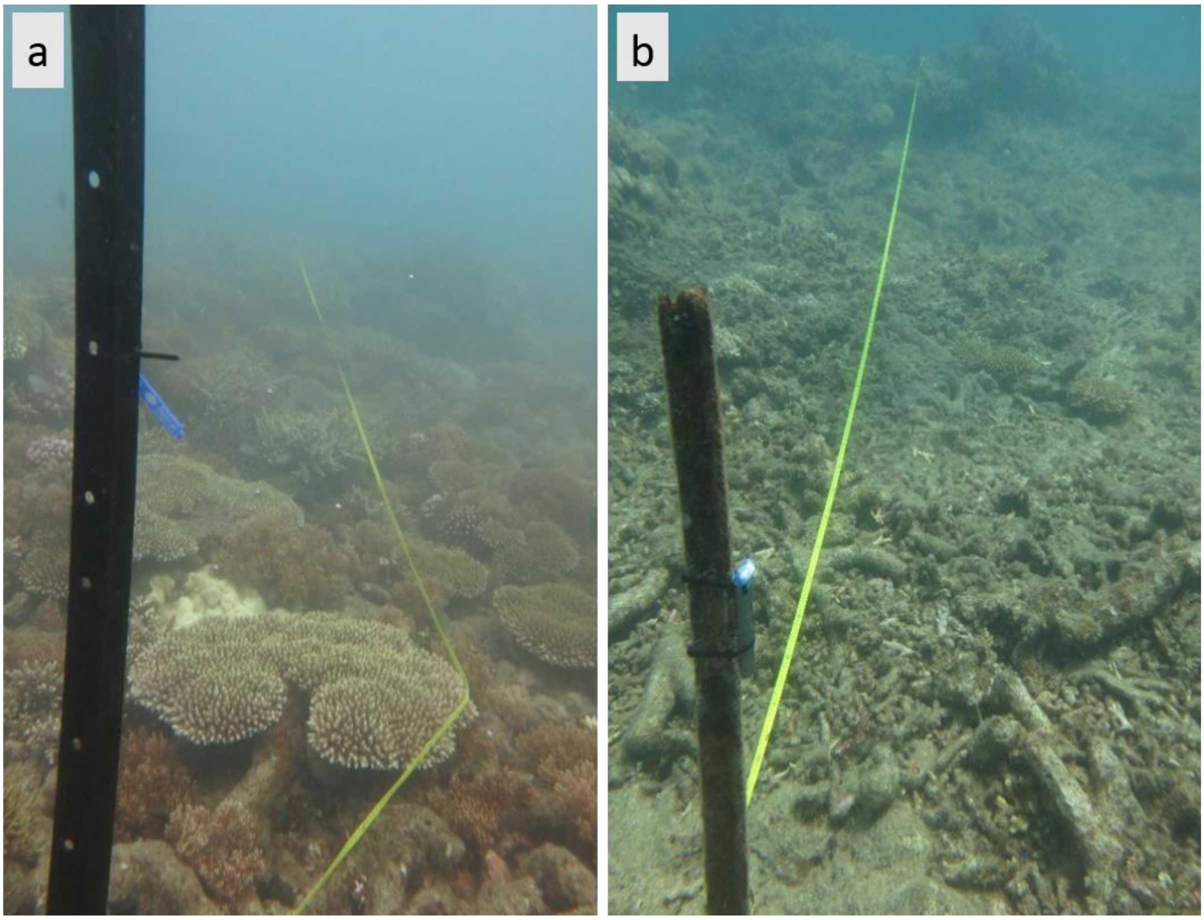

Figure A2. 1 Change in benthic community at Holbourne East, Site 1, 5 m depth. Photos are of the first few meters of the permanent monitoring sites and illustrate the mixed community of soft corals and *Acropora* present in 2016 (a) that was reduced to rubble in 2017 (b).

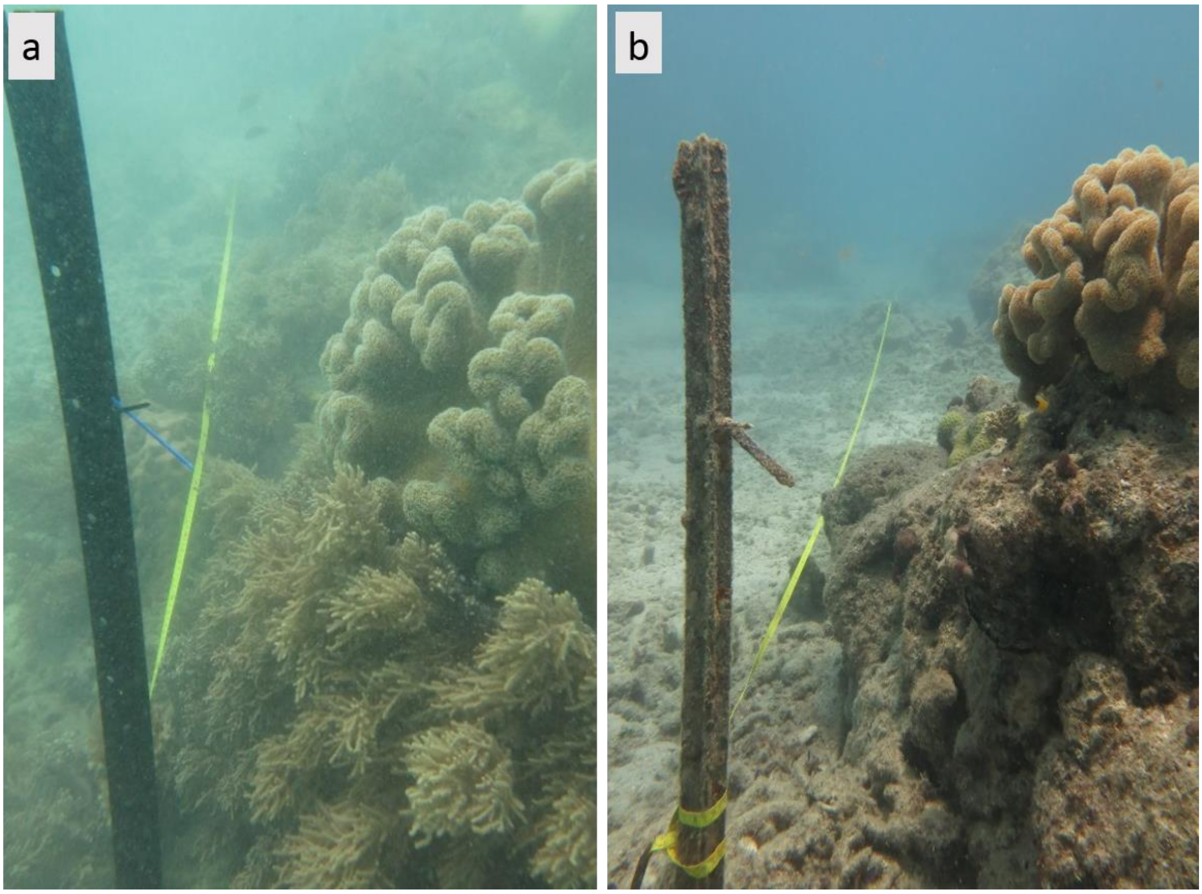

Figure A2. 2 Change in benthic community at Holbourne West, Site 1, 5 m depth. Photos illustrate the loss of soft corals that where abundant in 2016 (a) and to be replaced by turf algae in 2017 (b). The photos capture the same section of the permanent transect. A surviving soft coral (*Sarcophyton*) appears to the right in both images.

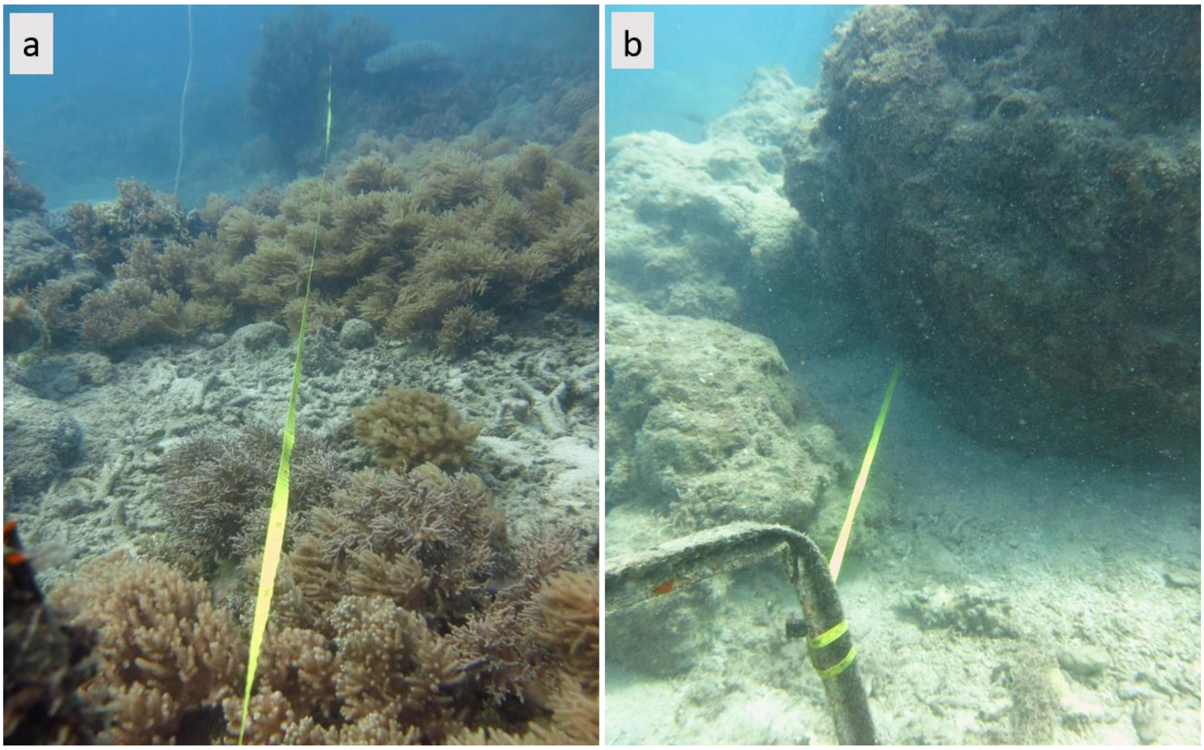

Figure A2. 3 Change in benthic community at Holbourne West, Site 2, 5 m depth. Image a) shows the start of the permanent monitoring transect in 2016 as a rubble substrate colonies by soft corals (Sinularia), the picket marking the site is visible in the lower left corner of the image. Image b) shows the same picket, now bent over, in 2017. The soft corals and much of the rubble substrate have been scoured from the site and a large dead coral head, seen filling the upper right of the image, rolled onto the site.

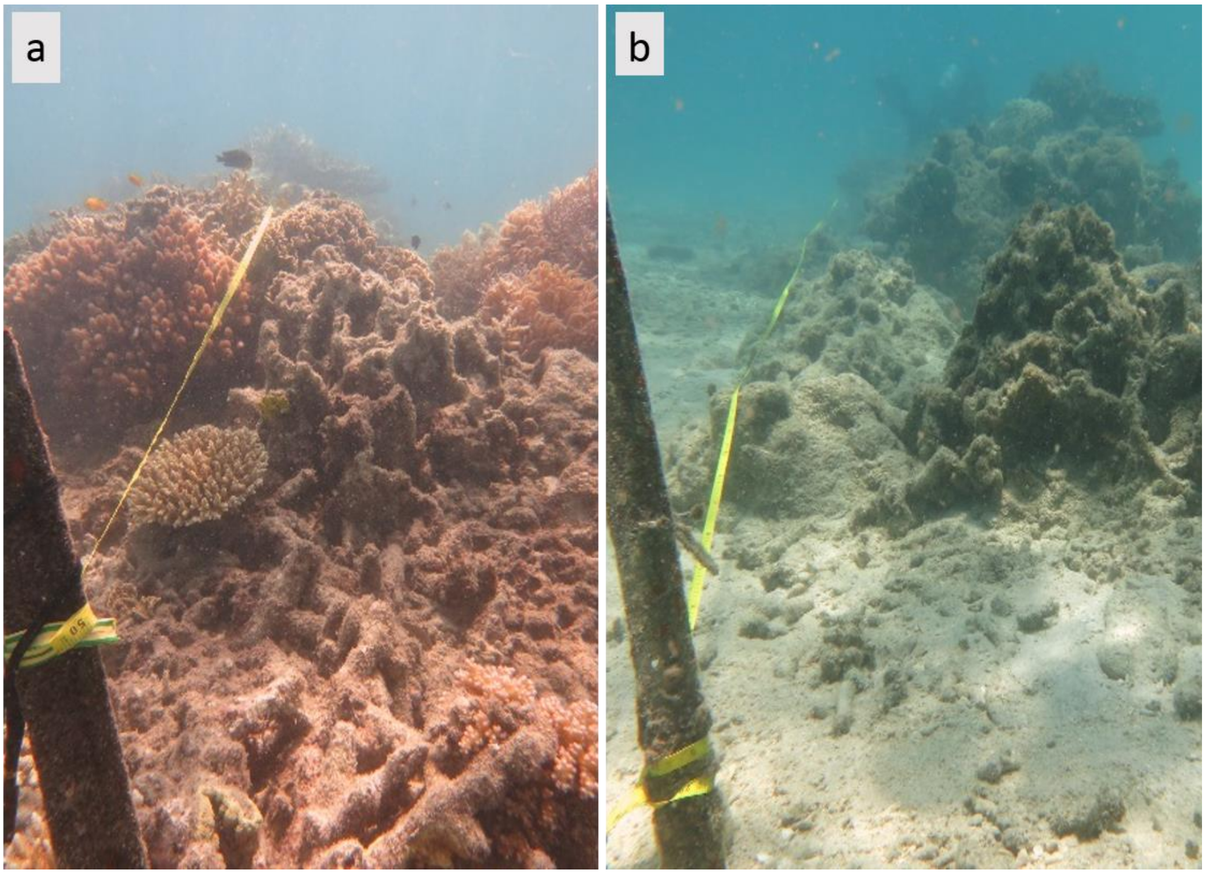

Figure A2. 4 Change in benthic community at Holbourne West, Site 1, 2 m depth. Images demonstrate the removal of consolidated rubble substrate and colonising corals visible in image a) and replaced by sand and loose rubble in image b).

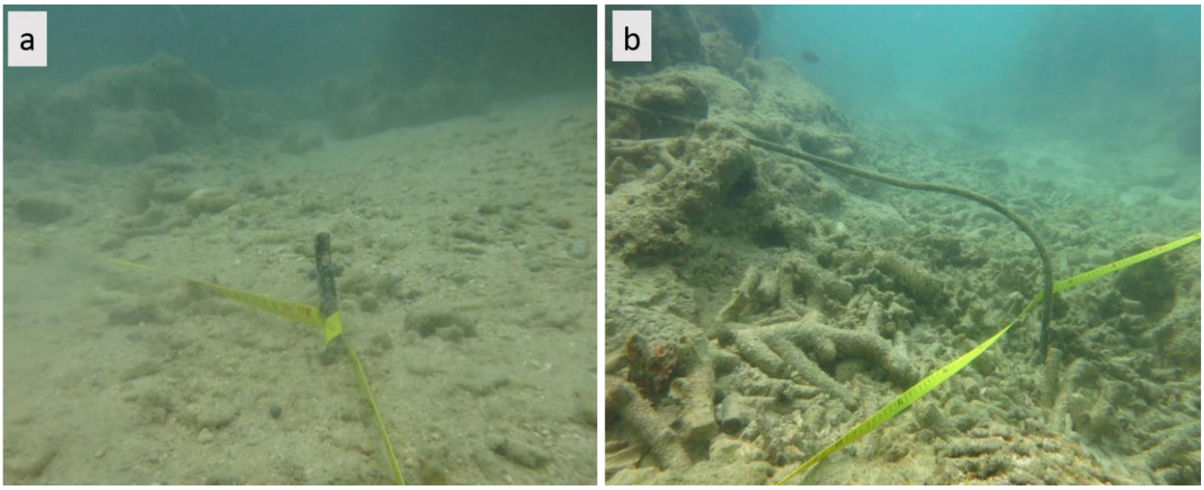

Figure A2. 5 Deposition of sand and rubble partially burying a transect marker at Holbourne West, site 1, 2m depth, a). Heavily scoured substrate and deformed transect marker (10 mm steel bar) at Holbourne East S2 2m depth, b).