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Coral Indicators for the 2023 Gladstone Harbour Report Card: ISP014

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Report Prepared for the Gladstone Healthy Harbour Partnership

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Cover Photo:

A large Turbinaria coral hosting a small Acropora coral while in competition with Sargassum algae and a coral-killing sponge at Seal Rocks. Photographer – Cassy Thompson (AIMS).

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1 EXECUTIVE SUMMARY

This report presents a detailed description of the benthic communities at coral monitoring locations within the Mid and Outer Harbour reporting zones that form the basis of the coral community indicator of the 2023 Gladstone Harbour Report Card.

In May 2023, the Australian Institute of Marine Science (AIMS) resurveyed benthic communities at permanent coral monitoring locations in the Mid Harbour (four locations) and Outer Harbour (two locations). Overall, the condition of these communities remains 'very poor' and received a grade E (Table 1, Figure 2).

Report card grades are based on the assessment of four indicators of coral condition: the proportion of the substrate occupied by living corals (coral cover), the proportion of the substrate occupied by large fleshy species of algae (macroalgae cover), the density of juvenile hard corals (juvenile density) and the rate of change in coral cover relative to the expected change for a given community (cover change).

Observed levels of the indicators were converted to scores based on thresholds developed for the 2018 Gladstone Harbour Report Card. This includes the methodology adopted in 2018 which saw juvenile density updated to be consistent with those used by the Reef Report Card to focus on colonies in the (<5cm) size class.

Table 1 Coral indicator and sub-indicator scores for 2023

Juvenile density	Coral cover	Macroalgae cover	Coral change	Coral indicator	
				Score	**Grade**
0.11	0.08	0.07	0.31	**0.14**	**E**

The 'very poor' grade for coral communities is heavily weighted by the continuing low cover of corals in contrast to high cover of macroalgae on most reefs. Mean coral cover (hard and soft coral combined) across Gladstone Harbour was 6%. This was a slight decrease from the 8% observed in 2022 and continues to be substantially lower than the 39% mean hard coral cover estimated in the Mid Harbour reporting zone in 2009 (BMT WBM 2013). Whilst the BMT WBM (2013) report does not provide a mean estimate for soft coral cover, data presented indicates that soft coral cover ranged between ~4% - 40%. Loss of coral cover coincided with major flooding in 2013. These floods almost certainly exposed corals to lethally low levels of salinity and high turbidity.

The current low cover of corals across the Harbour highlights the lack of recovery of coral communities over the last decade.

The sub-indicators, macroalgae cover, juvenile density, and coral change, are included in the monitoring program to provide measures of the recovery potential of coral communities following the impact of acute events, such as the 2013 floods.

The cover of macroalgae remains high, which translates into the 'very poor' assessment for this sub-indicator. Macroalgae can limit coral recovery through a variety of pathways including direct competition for space and suppression of coral recruitment.

The ‘very poor’ assessment of the juvenile density sub-indicator is likely to reflect both the pressures imposed by high cover of macroalgae and limited availability of coral larvae due to low coral cover within the Harbour.

The score for the coral change sub-indicator remains classified as ‘poor’, further demonstrating the slow recovery of coral cover within the Harbour. High water temperatures in early 2020 resulted in coral bleaching at the two Outer Harbour sites, Seal Rocks North and Seal Rocks South. The stress incurred during this bleaching event likely compounded the ongoing pressure imposed by abundant macroalgae and will have contributed to the observed slow recovery of coral cover in recent years. The prevalence of bio-eroding sponges, that continue to kill corals across the Harbour, further limit the recovery of these reefs.

2 BACKGROUND

Coral communities around the world are under increasing pressure as intensifying land use, urbanisation and industrial development impinge on corals' ability to resist, or recover from, natural disturbances such as floods or storms. Along the Great Barrier Reef (GBR) coast it is well documented that loads of sediments, nutrients and chemical contaminants carried to the sea in catchment runoff have increased since European settlement (Kroon *et al.* 2012, Waters *et al.* 2014).

Within Gladstone Harbour coral communities are subject to the same range of pressures as other inshore coral reefs in the GBR, that compound with potential pressures associated with the operations of the Harbour and associated industries. It is for this reason that AIMS has co-invested with the Gladstone Healthy Harbour Partnership (GHHP) to monitor and report the condition of coral communities within the GHHP reporting area as part of the Gladstone Harbour Report Card.

The indicators, sampling methodology, and scoring system used to derive grades for the Gladstone Harbour Report Card were chosen to be as compatible as practicable to those used for the Great Barrier Reef Report Card (Queensland Government 2015). We note that revisions of the methods used to score coral community condition for the Great Barrier Reef Report Card (Thompson *et al.* 2016) mean that while indicators remain the same, thresholds against which state and regional report card scores are derived now differ from those used for the Gladstone Harbour Report Card for the macroalgae and coral cover indicators. The scoring for the juvenile density indicator for the Gladstone Harbour Report Card was realigned to the State and other Regional report cards in 2018.

This report presents the ninth annual survey of permanent coral monitoring transects constructed in 2015. The purpose of this report is to provide a detailed description of reef communities as observed in 2023 that expands on the necessarily succinct summary of condition presented by the 2023 Gladstone Harbour Report Card.

3 METHODS

3.1 Sampling design

Coral communities are monitored along permanently marked transects. The selection of sites and construction of transects occurred in July 2015 as reported in detail in Thompson *et al.* (2015). In brief, suitable sites were identified at four locations within the Mid Harbour reporting zone and two locations in the Outer Harbour reporting zone (Figure 1). Within each site a series of five 20 metre-long transects, each separated by a space of 5 metres, were constructed along a depth contour identified as the most suitable coral habitat; depths ranged between 0 and 1 metre below lowest astronomic tide (Table A 1) as dictated by the limited depth of hard coral communities within the Harbour. To ensure accurate relocation of sampling, the start of each transect was marked with a steel star-picket, with additional transect markers consisting of lengths of 10 mm steel rod placed at the midpoint and end of each transect. The starting point of the 1st transect was recorded as a GPS location (WGS84 datum) and compass bearings recorded along each transect to aid future relocation (Table A 1). At each transect the following three surveys of the benthic communities are undertaken annually. This report presents data collected on the 21st of May 2023.

3.2 Survey methods

3.2.1 Photo point intercept transects

Estimates of the composition of benthic communities were derived from the identification of organisms on digital photographs taken along the permanently marked transects. The method closely followed Standard Operation Procedure Number 10 of the AIMS Long-Term Monitoring Program (LTMP, Jonker *et al.* 2008) and mirrors that used by the Marine Monitoring Program (MMP). Digital photographs were taken at 50 cm intervals along each transect. Estimations of proportional cover of benthic community components were derived from the identification of the benthos lying beneath five fixed points digitally overlaid onto these images. Benthic cover of any group of interest is estimated as the proportion of all points that were identified and categorised as that group. A total of 32 images were analysed from each transect. Hard and soft corals were identified to genus level. 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 and checking of all identified points.

3.2.2 Juvenile coral surveys

The number of juvenile coral colonies was counted *in situ* along the permanently marked transects. Prior to 2018, corals in the size classes: 0-2 cm, >2-5 cm, and >5-10 cm found within a strip 34 cm wide (data slate length) positioned on the upslope side of the transect line were identified to genus level. Since 2018, reporting of the >5-10 cm size class was discontinued, aligning the methodology used here with that used by the MMP (Thompson *et al.* 2016). Importantly, this method aims 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 excludes small coral colonies considered to have resulted from the fragmentation or partial mortality of larger colonies.

Limiting observations to <5 cm more accurately focuses on juvenile rather than fragmented colonies and helps to exclude small colonies of slow growing corals which do not reflect the recent recruitment and survivorship dynamics which are assessed by this indicator. Further, the realignment of methodology allows direct comparison between Gladstone Harbour coral communities and those of other inshore reefs monitored by the MMP.

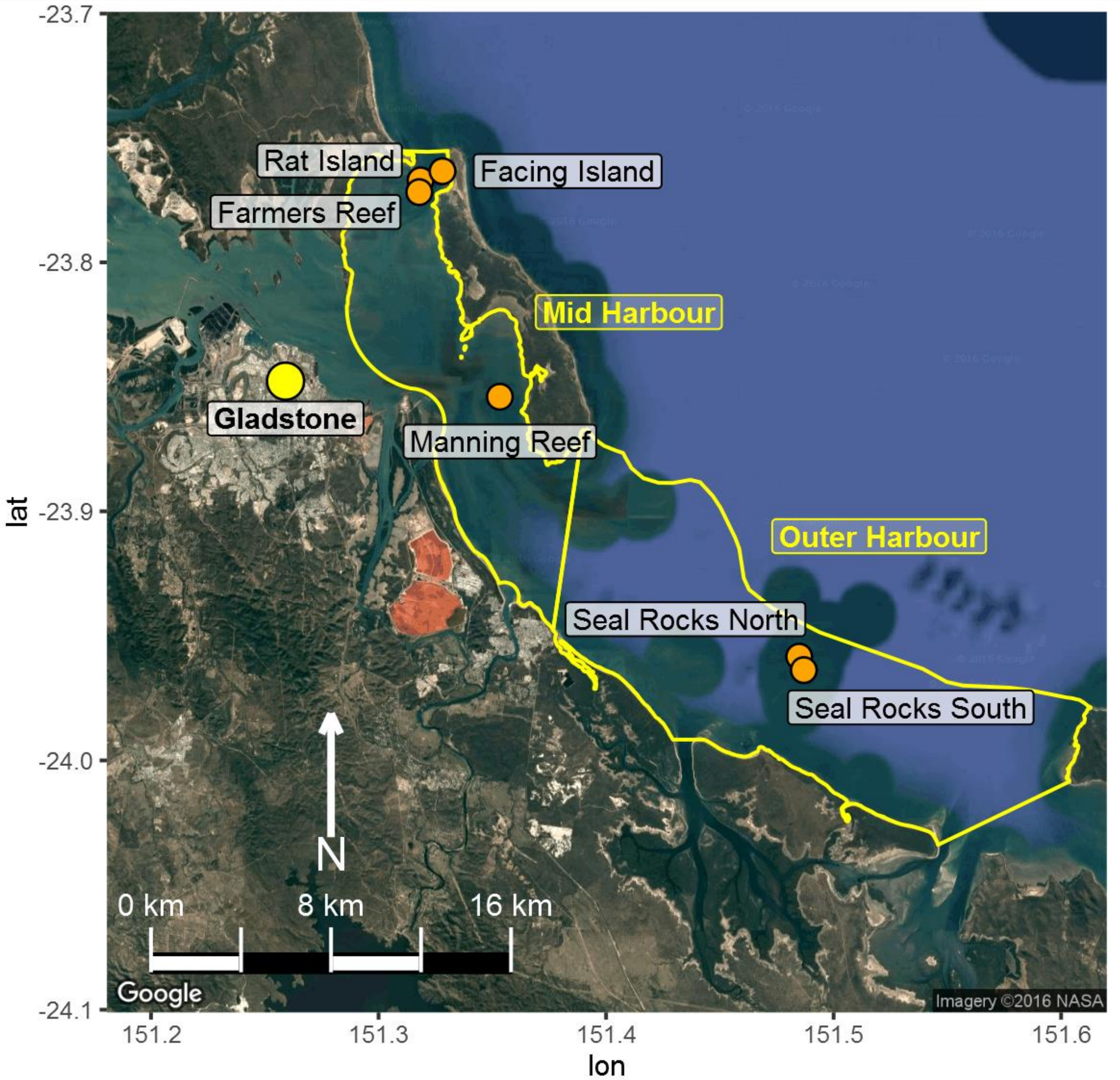

Figure 1 Coral monitoring sites.

3.2.3 Scuba search transects

Scuba search transects document the incidence of disease and other agents of coral mortality observed at the time of survey. This method closely followed 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 declines in coral community condition. For each 20 metre-long transect, a search was conducted within a 2 metre-wide belt 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 coral colonies was recorded.

3.3 Coral community sub-indicators

The coral index is formulated around the concept of community resilience. The underlying assumption is that a 'resilient' community should show clear signs of recovery after inevitable acute disturbances, such as cyclones and coral bleaching events, or, in the absence of disturbance, maintain a high cover of corals and successful recruitment processes. For the Gladstone Harbour Report Card four sub-indicators of coral communities are included, each representing different processes that contribute to coral community resilience.

This section provides an overview of the methods used to estimate and score each sub-indicator that, in combination, capture both the state and resilience of coral communities. A full description for the rationale behind the selection and scoring of each sub-indicator is included in Appendix 2.

3.3.1 Coral cover

The most tangible and desirable indication of a healthy coral community is an abundance of coral. The Coral cover sub-indicator score at each reef is based on the proportional area of substrate covered by either 'Hard' (order Scleractinia) or 'Soft' (subclass Octocorallia) corals.

$$coral\ cover_{ij} = HC_{ij} + SC_{ij}$$

Where, *HC* and *SC* are the proportion of benthos occupied by hard and soft corals respectively,

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, temperature anomalies and, in coastal areas, flooding. The sub-indicators: juvenile density, macroalgae cover, and coral change, 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 in the plankton, 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 sub-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:

$$juvenile\ density_{ij} = J_{ij} / AS_{ij}$$

Where, J=count of juvenile colonies < 5 cm in diameter, AS=area of transect occupied by algae, i=reef and j=time.

3.3.3 Macroalgae cover

High macroalgal abundance may suppress the recovery of coral communities through a variety of mechanisms ranging from competition with surviving colonies through to suppression of the recruitment process (e.g., McCook *et al.* 2001, Hughes *et al.* 2007, Foster *et al.* 2008, Cheal *et al.* 2013, Hauri *et al.* 2010). The values of the macroalgae cover sub-indicator were estimated as the proportion of benthos along point intercept transects identified as macroalgae:

$$macroalgae\ cover_{ij} = MA_{ij}$$

Where, MA is the proportion of the benthos occupied by macroalgae, *i*=reef and *j*=time. Macroalgae is here considered to include all algae larger than the filamentous turf or crustose coralline forms.

3.3.4 Coral change

While high coral cover can justifiably be considered a positive indicator of community condition, the reverse is not necessarily true. Low cover may occur following acute disturbance and, hence, may not be a direct reflection of the community's resilience to underlying environmental conditions. For this reason, in addition to considering the actual level of coral cover we assess the rate at which hard coral cover increases as a measure of recovery potential. The assessment of rates of cover increase is possible as rates of change in hard coral cover on inshore reefs have been modelled (Thompson *et al.* 2016), allowing estimations of expected increases in cover for communities of varying composition to be compared against observed changes.

A Bayesian framework was used to permit propagation of uncertainty through predictions of expected hard coral cover increase from separate models applied to fast growing corals of the family Acroporidae, and the combined cover of all other hard corals. Note that the example presented below for Acroporidae (*Acr*), has the same form as that applied for Other Corals (*OthC*) if these terms are exchanged where they appear in the equations.

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

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

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

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

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

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

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

Where, $Acr_{it}, OthC_{it}$ and Sc_{it} are the cover of Acroporidae coral, other hard coral and soft coral respectively at a given reef at time (t). $eskK$ is the community size at equilibrium (100-proportion of area comprised of unconsolidated substrates) and $rAcr$ is the rate of increase (growth rate) in cover of Acroporidae. Varying effects of Reef (β_j) is also incorporated to account for spatial autocorrelation. Model coefficients associated with the intercept, and Reef (α_i and β_j) all had weakly informative Gaussian priors (the latter two with model standard deviation). The overall rate of coral growth parameters ($rAcr$ or alternatively $rOthC$) constituted the mean of the individual posterior rates of increase ($vAcr_i$ or alternatively $vOthC_i$).

3.3.5 Scoring of sub-indicators

To facilitate the reporting of coral community condition the observed values for each sub-indicator were converted to scores on a common scale of 0 to 1. For each sub-indicator, observed values were scaled against thresholds which were set based on expert opinion and knowledge gained from the time-series of coral community condition collected by the Marine Monitoring Program (MMP) and the AIMS Long Term Monitoring Program (LTMP). Thresholds represent the boundary between report card grades of C and D (score = 0.5) that would indicate the switch between a community in satisfactory condition and one displaying a lack of resilience (Table 2). In addition, 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 cover change indicator, scores are averaged over three years of estimates excluding years during which reefs were categorised as having been impacted by an acute disturbance event. This averaging is done as expected increase in cover from one year to the next can be low, especially when coral cover is low, or the community is dominated by slow growing taxa. This averaging helps to account for sampling error inherent in each annual estimate of coral cover.

Table 2 Thresholds and bounds for scoring coral sub-indicators. Note that the thresholds for the juvenile density were updated in 2018 to account for the change in the methodology described above and are consistent with those used by the MMP on inshore reefs.

Indicator	Threshold (score = 0.5)	Upper bound (score = 1)	Lower bound (score = 0)
Coral cover	40%	90%	0%
Macroalgae cover	14%	5%	20%
Juvenile density	4.6 m^{-2}	13 m^{-2}	0 m^{-2}
Coral change	Lower 95% CI	2* upper 95% CI	Below 2* lower 95% CI

3.3.5.1 Cover change scoring erratum

An error in the script underpinning this indicator was identified and fixed in 2021. The issue related to the three years over which scores were averaged to provide the annual estimate in 2020. Sequential ordering of years within each reef had not occurred and as such the mean value included three years between 2016-2020 rather than explicitly the most recent three valid estimates of change.

3.3.6 Aggregation of sub-indicator scores

The scaling of all scores to the common range of 0 to 1 allows aggregation of scores across sub-indicators at a hierarchy of spatial scales. Within this report scores are presented at the scale of individual sub-indicators at each reef, individual sub-indicators and coral indicator scores for each reporting zone and the whole-of-harbour. Zone scores represent the mean score for each sub-indicator across reefs within that zone, while coral indicator scores represent the mean of the four zone-level sub-indicator scores. Similarly, harbour-wide scores were taken as the mean of the zone-level means for each sub-indicator and the coral indicator score as the mean of these harbour-wide sub-indicator scores. All scores were supplied to AIMS by the Data Integration and Management System based on sub-indicator values uploaded to that system by AIMS.

For the Gladstone Harbour Report Card, indicator scores are derived though the aggregation of bootstrapped distributions of sub-indicator scores, where bootstrapped distributions are produced by repeatedly sampling, with replacement, the observed distribution of sub-indicators. This method of aggregating distributions ensures that each distribution has equal weighting on the aggregation.

In practice, to aggregate sub-indicator scores at each reef to a mean score and estimate of variance for a zone requires that:

1. A bootstrap distribution of 10000 samples is constructed for each sub-indicator within the zone.
2. The resulting bootstrap distributions are added together, and the mean indicator score for the zone along with variance extracted from this combined distribution.

Whole-of-harbour scores were similarly generated by respectively aggregating the sub-indicator distributions within zones, adding the aggregated distributions from each zone together to derive a harbour-level distribution from which mean and variance for sub-indicators at the scale of the Harbour were derived. Finally, adding the whole-of-harbour distributions for each sub-indicator yields the distribution from which the whole-of-harbour coral indicator score, and variance were extracted. Reef level coral indicator scores are simply the arithmetic mean of the scores for each sub-indicator.

Grades for coral community condition were derived from the scores estimated above according to the conversions described in Table 3.

Table 3 Conversion of aggregated indicator scores to report card grades.

A	**Very good** (0.85 – 1.00)
B	**Good** (0.65 – 0.84)
C	**Satisfactory** (0.50 – 0.64)
D	**Poor** (0.25 – 0.49)
E	**Very poor** (0.00 – 0.24)

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 & 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 coral surveys in 2016 AIMS deployed temperature loggers to the pickets marking the first transect at each of Rat Island, Manning Reef, and Seal Rocks North. These loggers are exchanged annually and provide an ongoing record of *in-situ* water temperature and begin the process of developing an accurate climatology for the coral communities in the Harbour. The available times series from deployed loggers were aggregated to weekly mean temperatures for presentation in this report. Once this time-series matures, and baseline climatology of temperatures within the Harbour can be estimated, it is intended that the likelihood of thermal stress will be estimated from these observed data. Until this is possible, the likelihood of thermal stress to corals in the Harbour was interpreted from thermal anomalies estimated as degree heating days (DHD) downloadable from ReefTemp (Garde *et al.* 2014) as published by the Bureau of Meteorology, or degree heating weeks (DHW tutorial) published by the National Oceanic and Atmospheric Association Coral Reef Watch. For this report, annual summaries of DHD from 1 December to 31 March, based on 14 Day IMOS climatology (Garde *et al.* 2014), were downloaded. Maps of estimated maximum annual DHW and DHD for the past five years are provided to help visualise the relative thermal stress in recent years. Thresholds for likely bleaching are between 60-100 DHDs and 4-8 DHWs.

Degree Heating Weeks consider accumulated temperature anomalies that are greater than 1 degree C above the mean of the hottest monthly temperature for a location (Liu *et al*. 2017). The first three years of temperature data recorded by *in-situ* loggers has been used to estimate monthly mean temperatures over the 2017-2019 summers. The means of the hottest month from each year were used to provide a preliminary estimate of mean monthly maximums for each reef against which observed temperature profiles can be considered. Unfortunately, the logger deployed at Rat Island over the 2020-2021 summer was not recovered during surveys in 2021 as the picket to which it was attached had been dislodged and the logger lost.

3.4.2 Runoff

Runoff can impact corals in two ways: by introducing harmful loads of sediment and other contaminants and in extreme cases reducing salinity to levels lethal to corals (van Woesik 1991, Jones & 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 28 parts per thousand (ppt) that begin to cause coral mortality (Berkelmans *et al.* 2012). As a first step in assessing the likelihood that floods exposed corals to salinity-stressor high loads of sediments and nutrients for the adjacent catchments the seasonal discharge of local rivers is compared to long term median flows. Median discharge for the “wet season”, defined here as December-May, is 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 130005A-Fitzroy River at the Gap
- Station 132001A-Calliope River at Castlehope

As the flow of the Boyne River is interrupted by Lake Awoonga Dam, the time and magnitude of overflow of this dam, as reported by the Gladstone Area Water Board, is also considered.

3.5 Plotting of results

3.5.1 Time-series of indicator and sub-indicators

The coral indicator scores over time for the whole-of-harbour and for each reporting zone are plotted as a time series based on data estimates returned to AIMS from the Data Integration and Management System. These plots include the mean and upper and lower confidence intervals based on bootstrapped distributions of the indicator scores. The observed values of the sub-indicators: Coral Cover, Macroalgae and Juvenile Density (from which scores are derived) are plotted using their predicted trend and credible confidence limits estimated from generalised linear models that include random intercepts for each reef sampled.

3.5.2 Comparison of Gladstone Harbour Coral communities to other near-shore reefs

To place the state of the Gladstone Harbour coral communities into broader context, an ordination biplot has been provided. The biplot is based on Bray Curtis dissimilarity of square root transformed genus-level cover of hard and soft corals and higher-level groupings of algae observed on the Gladstone Harbour reefs in 2023, and at reefs sampled by the Marine Monitoring Program (MMP) in 2022. The MMP samples reefs using the same basic sampling methods as applied to the Gladstone Reefs. The MMP data used were the most recent observations available from 2 m depth sites.

To further compare Gladstone Harbour sub-indicator values to those observed on other near-shore reefs, boxplots are provided which show the distribution values for the coral cover, juvenile density and macroalgae cover sub-indicators in Gladstone Harbour and regions monitored by the MMP. For the coral change sub-indicator scores are plotted. As with the ordination above, the MMP data uses only the most recent data available from 2 m depth sites.

4 RESULTS

The Harbour-wide coral indicator score for 2023 is 0.14, grade E, and remains largely unchanged following a period of decline since a peak score of 0.28, grade D, observed in 2017 (Figure 2). The decline in the harbour-wide coral indicator score reflects declines in both the Mid Harbour and Outer Harbour reporting zones. Coral cover across the Harbour remains at very low levels, resulting in the continued ‘very poor’, grade E, assessment for this sub-indicator (Table 1, Table A 2). Macroalgae cover has remained at high levels and assessment for this sub-indicator remains ‘very poor’, grade E. (Table 1, Table A 2). The juvenile density sub-indicator score has continued to decline and remains grade E (Table 1, A 3, Figure 8). The coral change sub-indicator score remained ‘poor’, grade D (Table 1).

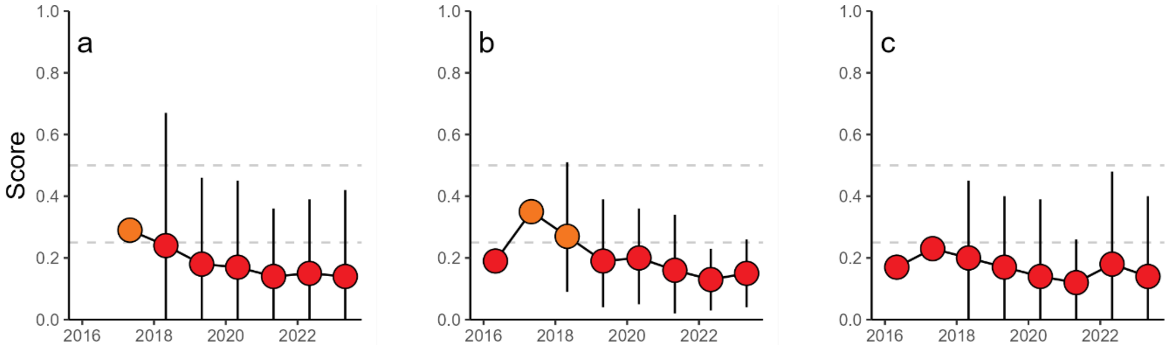

Figure 2 Coral indicator for reporting zones. a) Gladstone Harbour, b) Mid Harbour, c) Outer Harbour. Colours represent the coral report card grade (see Table 3 for details). Dashed lines indicate the thresholds between very poor, poor and satisfactory report card grades.

4.1 Environmental pressures

Seawater temperatures over the 2022/2023 summer were below those likely to have caused severe thermal stress to corals within the Harbour. *In situ* records show summer water temperatures exceeded the short-term mean (2017-2019) but were lower than observed in 2020 when bleaching impacted corals across the harbour (Figure 3). Degree Heating Days (DHD, Garde *et al.* 2014) and Degree Heating Weeks (DHW, Liu *et al*. 2018) provide two summaries of possible temperature related stress to corals. DHD are calculated as the accumulated positive anomaly of summer sea-surface temperature compared with the historical monthly mean climatology of the region. A daily sea surface temperature of one degree higher than the historical mean temperature *for that month* results in one DHD. In contrast DHW estimate accumulated time of exposure of more than 1 degree above the mean of the hottest month from a location’s climatology. Where positive temperature anomalies occur in the hottest summer months both DHW and DHD values will predict similar levels of bleaching probability. However, when anomalies are on the cusp of summer DHD will tend to overpredict likely thermal stress to corals. The mean accumulated DHD estimates for 2022/23 over the summer period (December to March inclusive) were 66 in the Mid Harbour and 62 in the Outer Harbour (Figure 4). These values are at the lower end of the 60-100 DHD threshold for potential severe bleaching. In contrast Degree Heating Week (DWH) estimates were generally below 2 and well below the level likely to cause coral bleaching. Individual colonies of *Porites* at Seal Rocks North and Seal Rocks South were the only bleached corals observed during surveys in early May 2023.

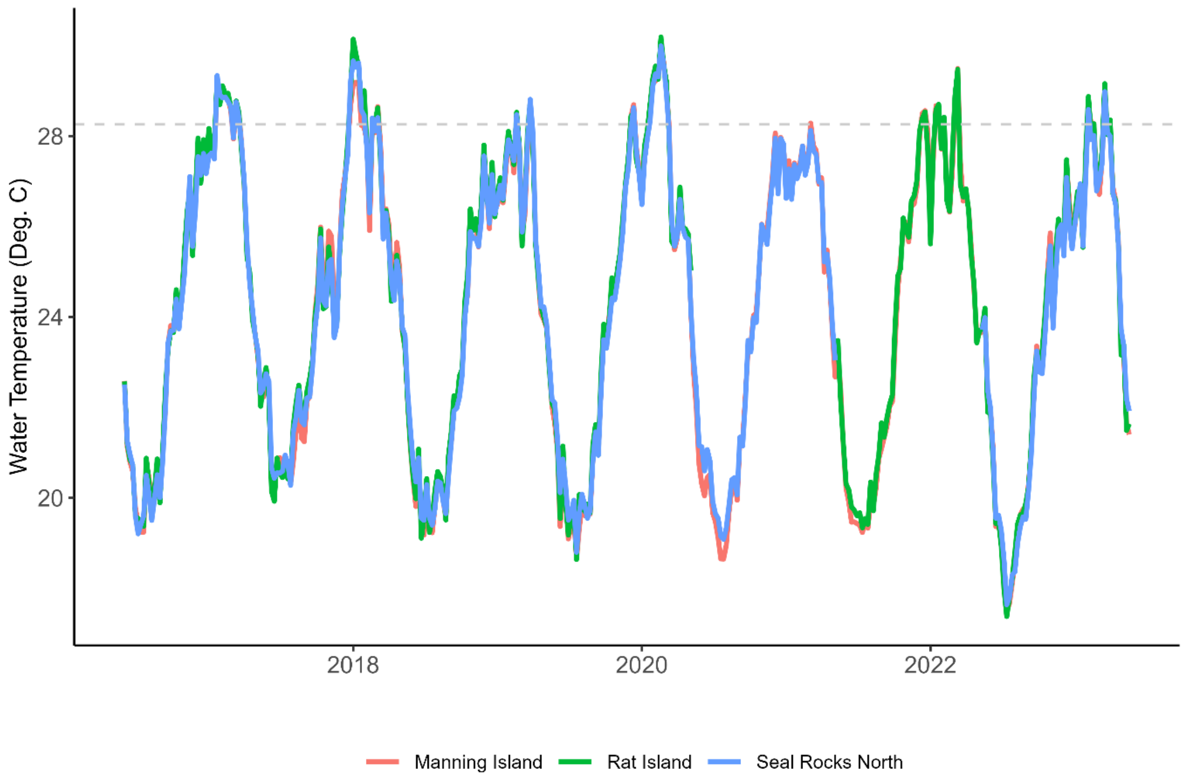

Figure 3. Weekly mean temperature profiles recorded at coral monitoring locations. Horizontal reference line is included as a visual aid and represent the mean of the hottest months in 2017, 2018 and 2019 at each reef. The temperature logger for the 2021-2022 year at Seal Rocks North was lost.

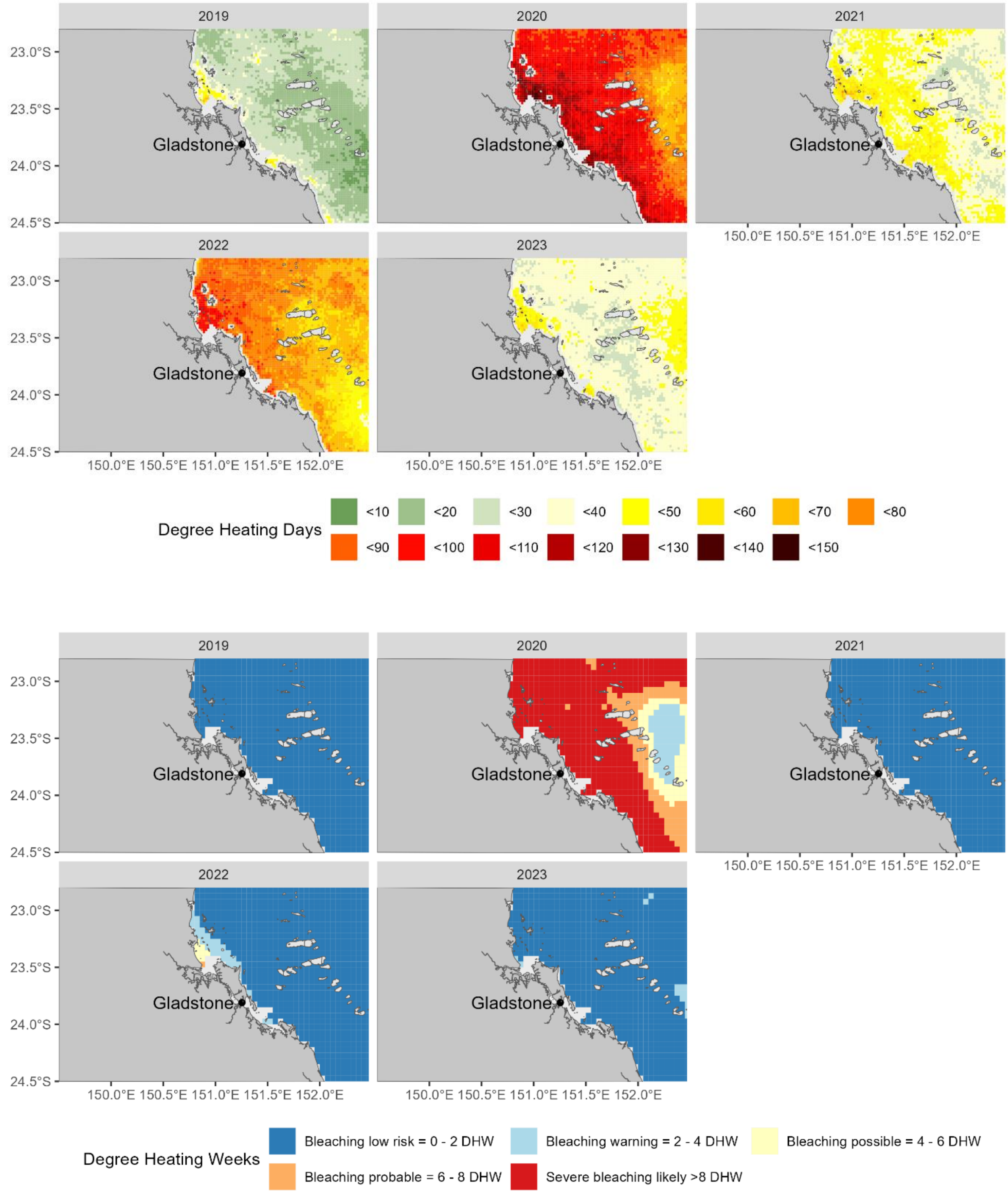

Figure 4 Annual Degree Heating Day and maximum Degree Heating Week estimates.

River discharge for the 2022/2023 wet season was below the baseline median from the Calliope River but 1.3 times the median for the Fitzroy River (Table 4). The Awoonga Dam levels have remained below the spillway since January 2018. These observations negate the potential for low salinity flood impacts and suggest relatively low delivery of nutrients and sediments from the local catchments during the year preceding coral community surveys in 2023. In clear contrast was the extreme flooding of the Calliope River (Table 4) and overflow of the Lake Awoonga Dam in 2013 (as reported in Thompson et al. 2015)

Table 4 River discharge. Values are annual wet season (December to May) discharge as a multiple of the baseline median wet season discharge for the period (1990-2010).

River	Median (ML)	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022	2023
Calliope	53306	17	2.8	5.2	1.2	4.4	0.5	0.0	0.9	0.2	0.7	0.3
Fitzroy	1540100	5.5	1	1.7	1.5	3.9	0.5	0.9	1.6	0.3	1.7	1.3

4.2 Coral cover

In 2023, mean coral cover increased marginally at Farmer, Manning, Seal Rocks North and Seal Rocks South but declined at Rat and Facing (Figure 5; Table A 6). However, the minor fluctuations observed over the six years of monitoring have all remained well within the levels categorised as 'very poor' for all reefs and the trend in this metric remains relatively flat across the Harbour (Figure 5a-c). These results should be considered in terms of the threshold of 40% cover at which this indicator is categorised as 'satisfactory'.

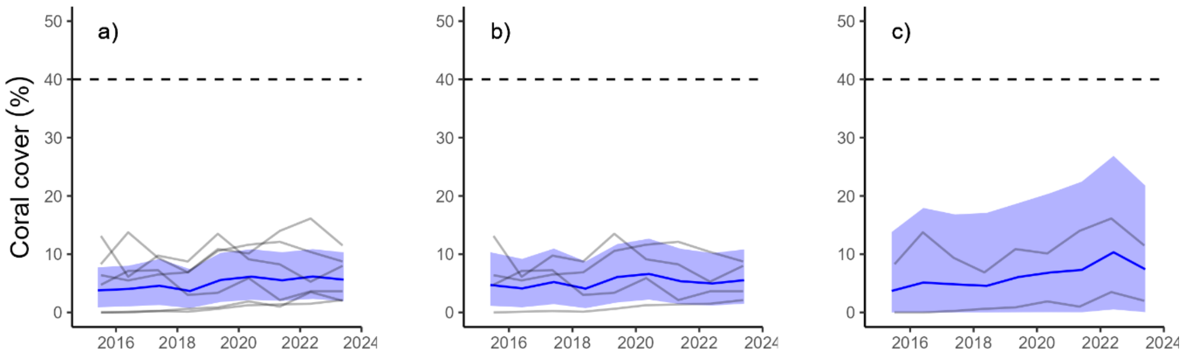

Figure 5 Trends in coral cover for reporting zones. a) Gladstone Harbour, b) Mid Harbour, c) Outer Harbour. Trends shown by blue lines bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles for individual reefs. Dashed lines represent the level at which this sub-indicator transitions between 'poor' and 'satisfactory' report card scores.

This threshold approximates the baseline condition of a mean cover of 39% (maximum 47%) observed at reefs in the North Passage and along the western side of Facing Island in 2009 (BMT WBM 2013).

Extreme flooding of the Boyne River in 2013 caused Lake Awoonga to overflow and, in combination with flows from the Calliope River, will almost certainly have resulted in mortality of corals within the Harbour (Thompson *et al.* 2015, Jones *et al.* 2015). Monitoring of salinity within the Mid Harbour reporting zone by Vision Environment (2013a & b) confirmed modelling results (Jones *et al.* 2015) indicating the presence of water with salinity levels well below the threshold of 22 PSU, lethal to *Acropora* corals (Berkelmans *et al.* 2004), for a period of 3 days.

It is important not to over-interpret the minor changes in coral cover observed since 2015. All sampling incurs some degree of sampling error. The use of fixed transects does minimise this error, however some variability in estimates should be expected. Of note is that large erect species of macroalgae, can overtop corals excluding them from observation. High and variable cover of macroalgae are likely to contribute to small variations in coral cover but also an overall underestimation of coral cover.

Within the Mid Harbour coral cover remains very low (Figure 5b). Rat Island's trend of increase from 2016 to 2021 was interrupted with coral cover declining slightly in 2022 and 2023 as high cover of macroalgae persist (Figure 6). At both Facing Island and Farmers Reef coral cover has been variable with no trends evident. Corals remain very rare at Manning Reef where cover is limited to the few corals that have recruited in recent years (Figure 6).

Coral cover for the Outer Harbour declined slightly at both reefs, although these results remain well within the bounds of likely sampling error (Figure 5c). The slight decline in coral cover estimates in the coincide with notable increase in Macroalgae cover (Figure 6), the over topping of corals by these large erect algae may partly explain the lower estimates of coral cover. Coral cover at Seal Rocks North remains extremely low amongst very high cover of macroalgae. At Seal Rocks South coral cover had steadily increased between 2018 and 2022. The recent appearance of the genus *Acropora* within the coral community at Seal Rocks (Figure 6) is an important step in the recovery of these communities as these corals have the potential to grow rapidly.

Scuba search data indicates that the bio-eroding sponge *Cliona orientalis* continues to impact the coral community across the Harbour and in particular colonies of *Turbinaria* at Seal Rocks South, along with *Porites* and *Cyphastrea* on Mid Harbour reefs (Table A 11). This sponge is almost certainly contributing to a lack of coral cover recovery across the Harbour.

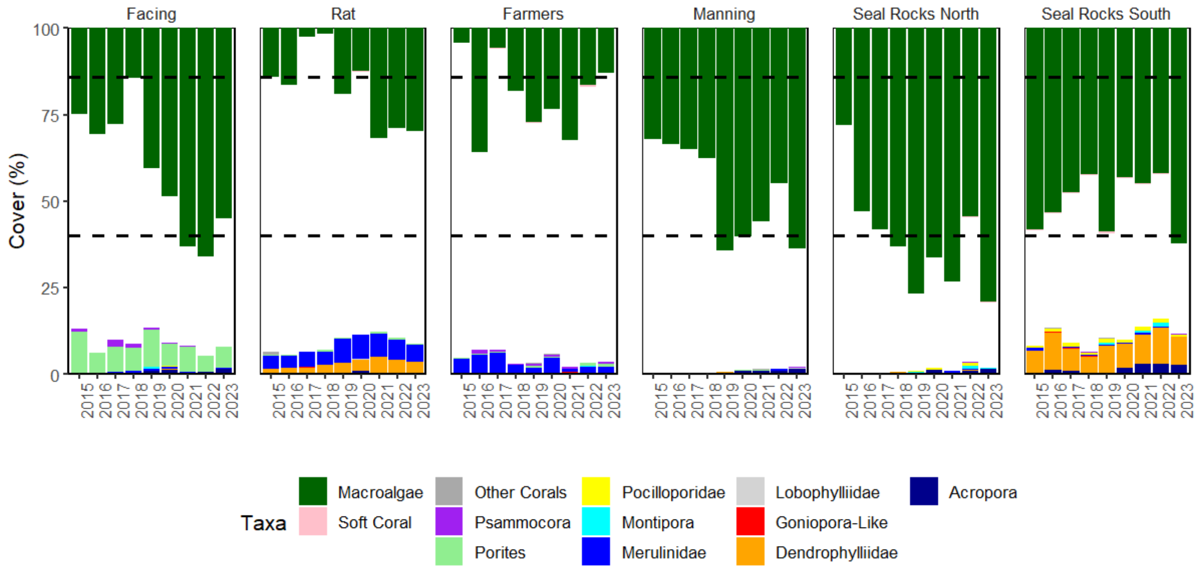

Figure 6 Composition of benthic cover at each location. Rising bars break down coral cover into major taxonomic groups (Families and Genera). Hanging bars represent macroalgae cover and are read in reverse (observed cover is read as 100 – y axis value, i.e. 10% cover will appear as a bar between 100 and 90% on the plot). White space is the remaining cover not occupied by indicators and will include: sand and silt substrate, turfing and crustose coralline algae along with other organisms such as sponges. Dashed reference lines indicate the boundary between the condition categories 'Poor' and 'Satisfactory'. Hanging macroalgae cover bars not extending to the upper reference line would be categorised as 'Satisfactory', or better. Rising bars for coral cover would have to extend to the lower reference line to receive a 'Satisfactory' categorisation.

4.3 Macroalgae cover

The mean cover of macroalgae across the harbour is very high (Figure 7a-c) ensuring the assessment for this sub-indicator remains 'very poor'. Within the Mid Harbour zone, the macroalgae cover declined slightly at Farmers Reef which is the only reef across the harbour to receive a 'satisfactory' categorisation for this indictor. At the remaining Mid harbour reefs the cover of Macroalgae has remained persistently above levels observed in the early years of program (Figure 6, Table A 6). In the Outer Harbour the cover of macroalgae was higher than previously recorded at both reefs (Figure 6, Figure 7c).

The generally high cover across the Harbour suggests that despite water quality being generally within guideline values in both the Mid and Outer Harbour ([2022 | Gladstone Healthy Harbour Partnership (ghhp.org.au)]()), the availability of nutrients within the Harbour is clearly not limiting to the macroalgae communities. Given the persistent high cover of macroalgae, and the performance of the other indicators, it is very likely that these algal communities are contributing to the suppression of coral recovery across the Harbour.

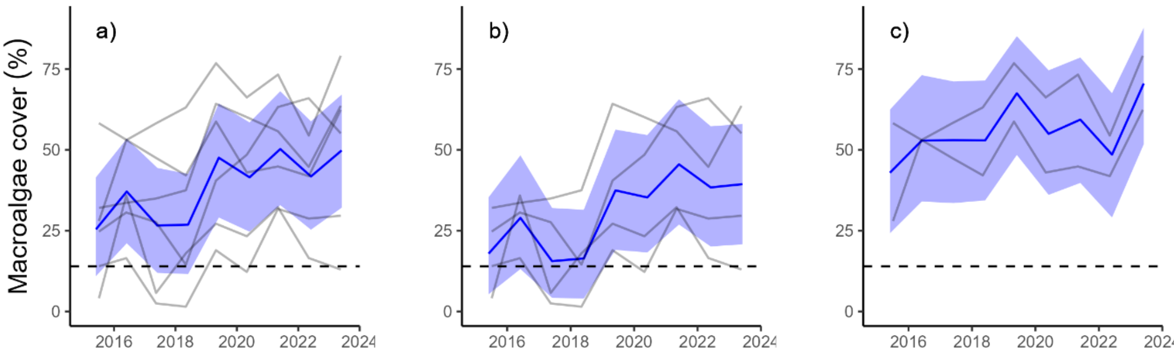

Figure 7 Trends in macroalgae cover for reporting zones. a) Gladstone Harbour, b) Mid Harbour, c) Outer Harbour. Trends shown by blue lines bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles for individual reefs. Dashed lines represent the level at which this sub-indicator transitions between 'poor' and 'satisfactory' report card scores.

As with coral communities (Table A 9) differences in the taxonomic composition of macroalgal communities (Table A 10) suggest fine scale differences in the combined physical and chemical environments at the monitoring locations. Monitoring undertaken by the MMP elsewhere on the GBR demonstrates that at reefs predisposed to high cover of macroalgae, cover is typically variable between years (Thompson *et al.* 2016). Within Gladstone Harbour, variability in macroalgae communities is especially evident at reefs in the Mid Harbour zone where cover and composition vary both from year to year within individual reefs but also between reefs (Figure 7b, Table A 10). In contrast, although there is some variability in the overall cover of macroalgae, the community composition at reefs in the Outer Harbour appear relatively stable, with communities consistently dominated by the two brown macroalgae genera, *Sargassum* and *Lobophora* (Table A 10).

4.4 Juvenile density

The harbour-wide mean density of juvenile corals has continued to decline, and this sub-indicator remains ‘very poor’ (Figure 8a, Table A 3). In 2023 the density of juvenile corals was higher than observed in 2022 at Rat and Farmers but was lower than previously observed at the other reefs monitored (Figure 9, Table A 6). Notably lacking across the Harbour are juveniles of the family Acroporidae. High coral cover recorded at several sites in 2009 included a high representation of this family (BMT WBM 2013). Indeed, dead *Acropora* skeletons are the primary substrate at both Seal Rocks North and Manning Reef (authors pers. obs.). Until juveniles of this family appear and survive, recovery of coral communities at these locations is unlikely.

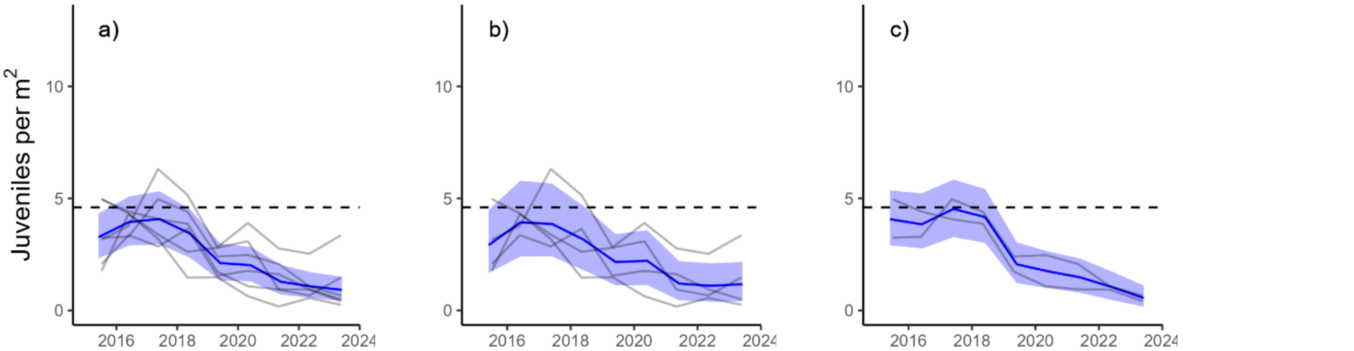

Figure 8 Trends in juvenile density for reporting zones. a) Gladstone Harbour, b) Mid Harbour, c) Outer Harbour. Trends shown by blue lines bound by 95% confidence intervals of those trends (shading), grey lines represent observed profiles for individual reefs. Dashed lines represent the level at which this sub-indicator transitions between ‘poor’ and ‘satisfactory’ report card scores.

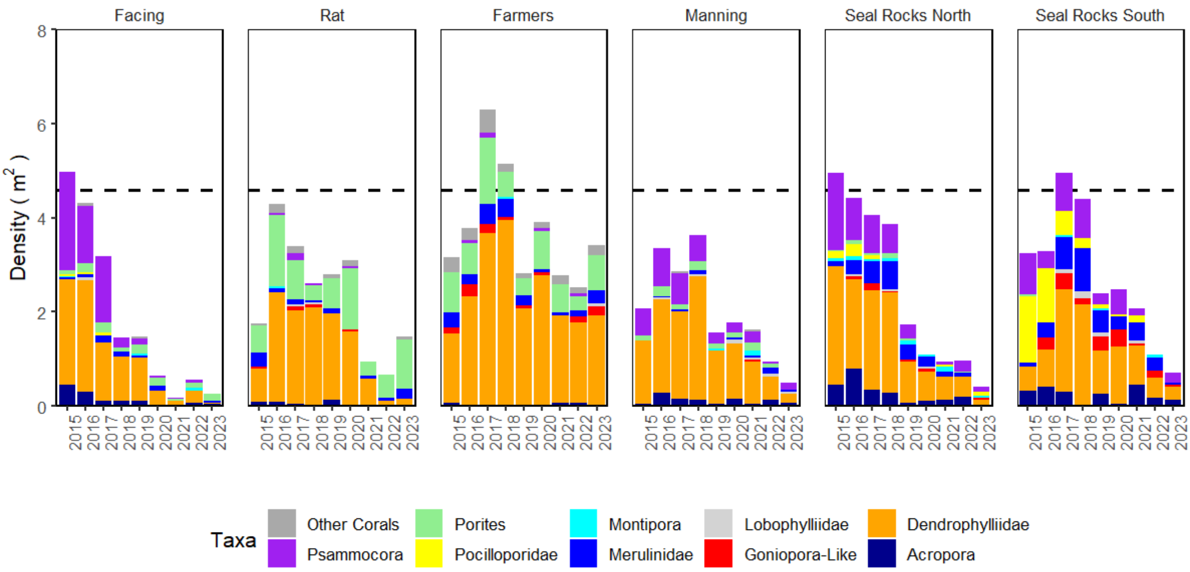

Figure 9 Composition of juvenile coral communities at each location. Bars break down juvenile density into major taxonomic groups (Families and Genera). Dashed reference line indicates the boundary between the condition categories ‘Poor’ and ‘Satisfactory’. Juvenile density would have to extend to the reference line to receive a ‘Satisfactory’ categorisation.

4.5 Change in hard coral cover

The Harbour-level score for coral cover change has remained within the poor range since first estimated in 2017 (Figure 10a). The Mid Harbour score has declined since 2021 to enter the very poor category for the first time in 2023. Very poor scores returned for Manning Reef and Rat Island are most influential in this result (Figure 10b, Table 5). In the Outer Harbour declines in coral cover at Seal Rocks North and Seal Rocks South between 2022 and 2023 have moderated the coral cover change score that had increased in 2022.

In general, the ongoing 'poor' score demonstrates that recovery of coral communities continues to fall short of modelled expectations. Important to note is that the scores for this indicator are averaged over a three-year period during which acute pressures have not been observed, and so coral cover should be in a state of recovery. The observation of bleached corals at Outer Harbour reefs in 2020 meant that changes in coral cover between 2019 and 2020 did not inform the cover-change scores in the Outer Harbour in 2020, 2021 or 2022.

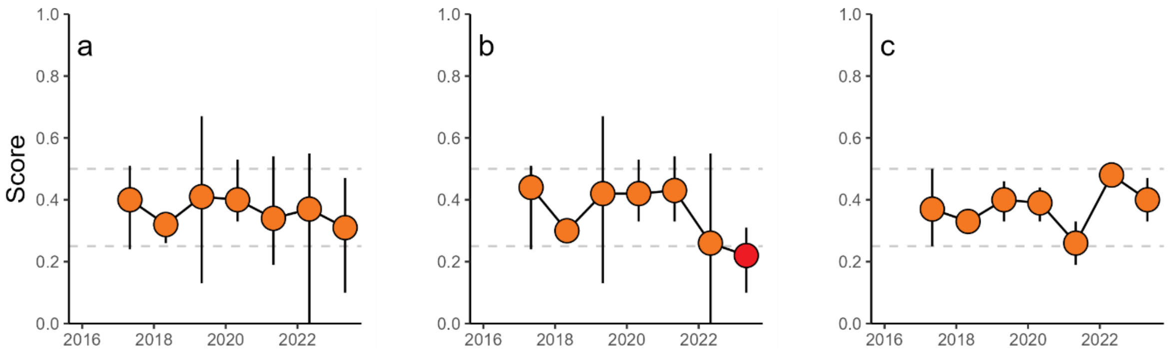

Figure 10 Coral cover change timeseries Coral cover change timeseries. Scores for a) Harbour, b) Mid Harbour and c) Outer harbour. Dashed reference lines included for lower thresholds for 'Satisfactory' and 'Poor' condition categorisations.

Table 5 Coral cover change sub-indicator scores for reefs and zones 2023.

Zone	Reef	Reef-level		Zone-level	
		Score	Condition	Score	Condition
Mid Harbour	Facing Island	0.28	Poor	0.22	Very Poor
	Farmers Reef	0.31	Poor		
	Manning Reef	0.19	Very Poor		
	Rat Island	0.10	Very Poor		
Outer Harbour	Seal Rocks North	0.33	Poor	0.40	Poor
	Seal Rocks South	0.47	Poor		

4.6 Comparisons with other near-shore reefs

The composition of benthic communities within Gladstone Harbour are distinct from communities on most near-shore reefs monitored by the MMP. Ordination analysis highlights it is the combination of low cover of most coral genera and high cover of macroalgae that define the Gladstone Harbour communities (Figure 11). The primary axis of the ordination (MDS1, Figure 11) explains 38% of the variance in community composition and effectively separates reefs with communities dominated by red or brown macroalgae on the right from those with proportionally higher cover of a range of coral genera, to the left. The distinction evident in the ordination is clear when comparing the relatively low coral cover and high macroalgae cover on Gladstone Harbour reefs to the cover of these groups at near-shore reefs in other regions (Figure 12a, c).

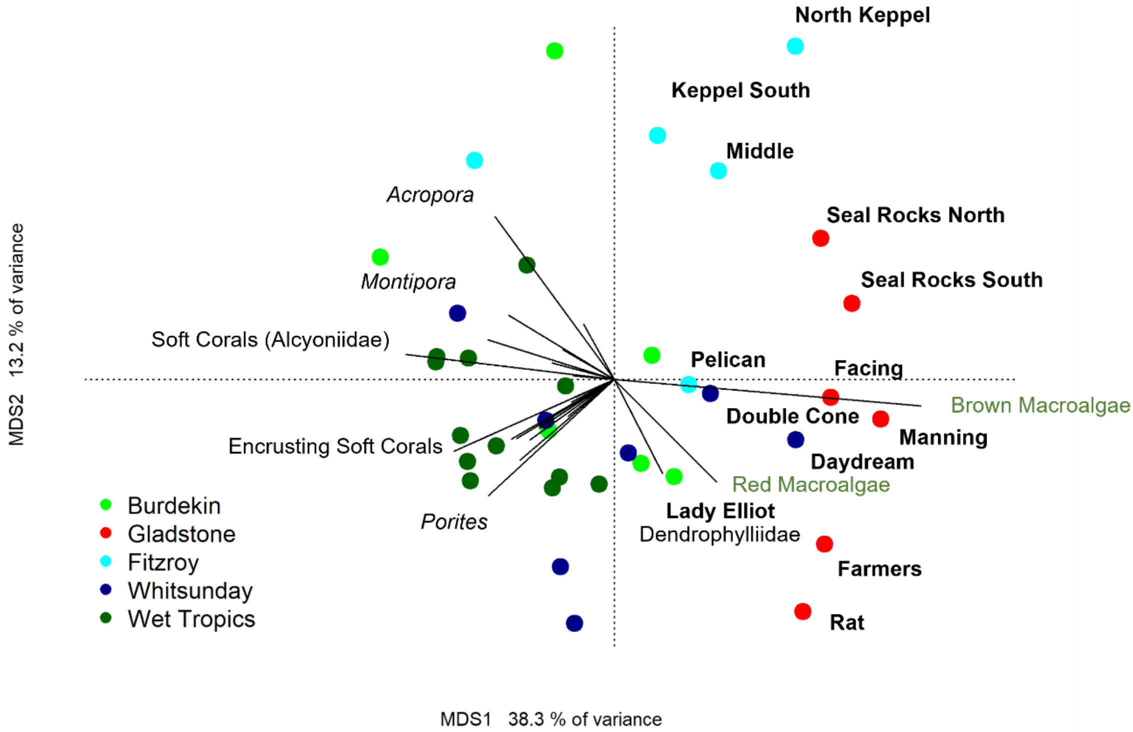

Figure 11 Ordination comparing Gladstone Harbour reef communities with other near-shore reefs. The biplot is based on Bray Curtis dissimilarity of square root transformed cover of coral (genus or family – vectors with black text) and higher order-level cover of macroalgae (vectors with green text) observed on the Gladstone Harbour reefs in 2023 and at 2 m reefs sampled by the Marine Monitoring Program (MMP) in 2022. Labels on the axis indicate how much of the variance in community composition is captured by the first (x-axis, MDS1) and second (y-axis, MDS2) dimensions of the ordination.

As might be expected, the Gladstone Harbour reefs are regionally most similar to those in the Fitzroy Region (Figure 11). Of the reefs monitored in the Fitzroy Region Pelican Island is the most proximal to the mouth of the Fitzroy River and the coral communities were severely impacted by flooding of that river in 2011. Like the reefs in Gladstone Harbour, recovery of coral communities at Pelican Island has been negligible and macroalgae continue to dominate the benthic community. The benthic communities at two reefs in the Whitsunday Region, Daydream Island and Double Cone Island, also group with the Gladstone reefs. Both these reefs were severely impacted by Cyclone Debbie in 2017, with coral being reduced to very low levels. As with the Gladstone Harbour reefs the benthic communities at Daydream Island and Double Cone Island saw a distinct increase in the cover of macroalgae and had shown negligible recovery in 2022.

The reefs of Gladstone Harbour sit in the lower end of the spectrum for most indicators relative to other near-shore reefs monitored by the MMP (Figure 12). Whilst it is evident that these have supported healthier coral communities in the past (BMT WBM 2013) present coral cover is well below that of other inshore areas.

Of concern is the continued low density of juvenile corals and high cover of macroalgae within Gladstone Harbour compared to other inshore reefs as these indicate a bottleneck for recovery (Figure 12 b, c).

While the mean coral change indicator is similar to that recently reported in the Whitsunday region (Figure 12d), the rate of recovery is low. In the Whitsundays several reefs had shown little to no recovery of coral cover since being severely impacted by Cyclone Debbie (Thompson *et al.* 2022). Further at most shallow reef sites in the Whitsunday Region the coral communities include higher representation of relatively fast-growing corals and this influences the expected rate of coral cover at those reefs. Finally, the MMP scoring system is biased toward lower scores compared to that used for the Gladstone scores as changes in cover equivalent to the lower confidence interval of predicted change returns a score of 0.4 within the MMP system, compared to 0.5 for Gladstone. In combination, this means that while scores were similar to or above those observed in the Whitsunday Region the observed rate of recovery remains low within the Harbour.

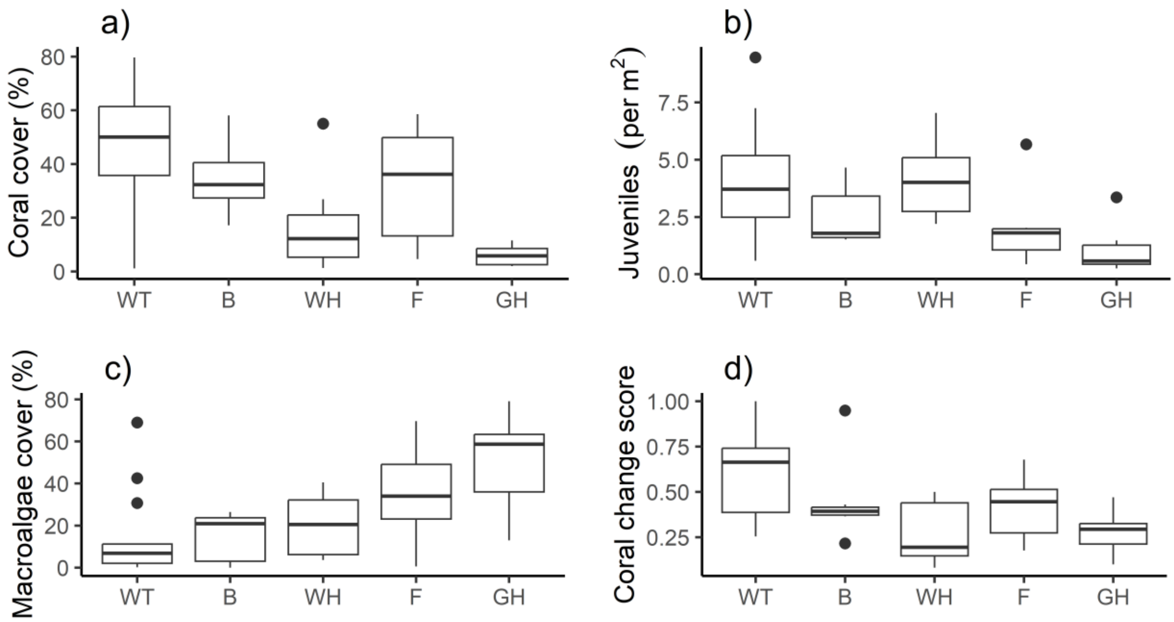

Figure 12 Regional comparisons of sub-indicators. a) coral cover, b) juvenile density, c) macroalgae cover, d) coral change score. WT= Wet Tropics, B=Burdekin, WH=Whitsundays, F= Fitzroy, GH= Gladstone Harbour. Data are based on current indicator values and scores for Gladstone Harbour reefs and the most recent available data for 2 m reefs sampled by the MMP.

5 CONCLUSION

Results from surveys in 2023 demonstrate a further decline in the condition of coral communities in Gladstone Harbour. While coral bleaching, in response to high water temperature in early 2020, is likely to have contributed to the current very poor condition, ongoing monitoring since 2015 demonstrates a clear lack of recovery since the severe loss of coral noted in 2015 (Thompson *et al.* 2015). This loss of coral coincided with the inundation of the Harbour's reefs by floodwaters in 2013. The low salinity within the harbour during these floods was considered the primary cause of coral mortality and precluded the ability to consider any cumulative or prior impact of extensive development of the Harbour through to 2013 (Jones *et al.* 2015, Thompson *et al.* 2015).

The magnitude of coral loss that occurred in 2013 dictated very low scores for the coral cover sub-indicator in 2015. Following such severe disturbance, it is the recovery processes that best describe the coral communities' overall condition. The macroalgae cover, juvenile density, and coral change sub-indicators are all formulated to assess the recovery process and, collectively, demonstrate the limited recovery potential being exhibited by the coral communities within the Harbour.

In combination, the continued poor or very poor scores for each sub-indicator corroborate studies that demonstrate density-dependant feedback mechanisms which promote a persistent shift from coral to macroalgal dominance where conditions allow the proliferation of macroalgae (Mumby *et al.* 2007, Mumby *et al.* 2013). Across the inshore reefs monitored by AIMS the prevalence of macroalgae in shallow waters was positively related to Chl *a* concentration in the surrounding waters implicating nutrient availability as a factor contributing to their prevalence (Thompson *et al.* 2022). Large fleshy macroalgae such as *Sargassum* and *Asparagopsis* and, in particular, the lower matt-forming species such as *Lobophora* and *Dictyota*, all of which are abundant across the Harbour, have been shown to be highly disruptive to coral community recovery (Birrell *et al.* 2008, Diaz-Pulido *et al.* 2010, Hauri *et al*. 2010). Macroalgae such as *Lobophora* and *Dictyota* have been shown to directly impact adult coral colonies leading to tissue loss, declines in coral fitness, and reduced growth rate (Lirman 2001, Vega Thurber *et al.* 2012, Morrow *et al* 2017). Ongoing competition between coral and macroalgae is likely to contribute to the 'poor' score for the coral change sub-indicator at most reefs. High cover of macroalgae is also likely to be affecting coral recruitment processes (Johns et al. 2018) contributing to the 'very poor' score for juvenile density. High cover of macroalgae has been associated with low densities of juvenile corals on several reefs monitored by the MMP. Notable examples include some reefs in the Fitzroy Region where the density of juvenile corals has remained very low since high cover of macroalgae replaced corals killed by floodwaters in 2011 (Berkelmans *et al.* 2012, Thompson *et al.* 2022).

In addition to hampering recruitment, coral-macroalgae interactions are potentially reducing the fecundity of adult corals (Tanner 1995, Foster *et al.* 2008), further limiting the supply of larvae from an already depleted population of adult corals.

Ongoing low density of juvenile corals indicates that recruitment processes are a severe bottleneck for the recovery of these coral communities. Generally, low recruitment of corals may be expected given the low abundance of coral brood-stock within the Harbour that will naturally limit local population fecundity. The Allee Effect - low rates of fertilization due to the low density of spawning corals, will further reduce the supply of locally spawned larvae (Gascoigne & Lupcius 2004). Coral recruitment can also be supressed by a range of water quality constituents, including fine-grained sediments, metals and chemical contaminants (Richmond *et al.* 2018). It is beyond the scope of the available data to tease apart the cumulative influences of population-level and water quality related pressures on coral recruitment within the Harbour. That said, investigating avenues to reduce anthropogenic impacts during the annual period of coral spawning through to settlement would seem prudent (Fraser *et al.* 2017).

The continued decline to very low densities of juvenile corals observed in 2023 is likely influenced by multiple processes. In addition to the ongoing influence of macroalgae, as described above, the high water temperatures in early 2020 are likely to have reduced juvenile densities by increasing the mortality rate of settled corals or potentially limiting the fecundity of adult corals over the late 2020 spawning season (Ward *et al.* 2002). An examination of the composition of juvenile communities highlights that it is the genus *Turbinaria* that has undergone the greatest absolute reduction across the Harbour. High variability in the density of *Turbinaria* juveniles has also been noted at some turbid water reefs monitored elsewhere on the Great Barrier Reef by the MMP (Thompson *et al.* 2022), although the underlying process driving this variability remains unknown. The juvenile communities across the Harbour do, however, continue to include a higher diversity of genera than the adult communities, suggesting in-flow of larvae from beyond the Harbour. Although these juveniles are yet to contribute to increased coral cover, this apparent connectivity to more distant brood-stock is a promising sign for the resilience of these communities. The continued presence of *Acropora* juveniles, although in low densities, remains a positive sign. *Acropora* were a key component of the coral communities at most sites prior to the 2013 floods (BMT WBM 2013), and the reestablishment of these fast-growing species will be fundamental to the recovery of these communities.

The coral change sub-indicator explicitly accounts for an expected low rate of coral cover increase due to both the low existing coral cover and communities dominated by slow growing species. Despite these modest expectations, the poor scores for this metric demonstrate that combined pressures imposed by the environmental conditions within the Harbour are limiting the recovery of coral cover. Further influencing the score for this sub-indicator is the widespread presence of the bio-eroding sponge *Cliona orientalis* which continues to be the most significant contributor to coral mortality within the Harbour.

High water temperatures in early 2020 were a further setback to the recovery of coral communities. The stress caused by high temperatures in early 2020 was particularly evident at the Outer Harbour sites where surveys in late April 2020 revealed that many colonies of the families *Acroporidae* and *Pocilloporidae* were at least partially bleached (Costello *et al.* 2020). These families are comprised of relatively fast-growing species key to the recovery of coral cover. Lower influence of bleaching in the Mid Harbour is likely due to the higher proportion of bleaching-resistant taxa at those sites. Although water temperatures were elevated in early 2022 we saw no evidence that this caused widespread bleaching within Gladstone Harbour.

In the broader context of near-shore reefs on the GBR, comparison of sub-indicator scores with those from reefs in other regions demonstrate Gladstone Harbour reefs perform poorly. In general, the Gladstone Harbour reefs were most similar to reefs which had recently undergone severe disturbances or had not recovered from previous disturbances. These communities exhibit either very low coral cover, very high cover of macroalgae or a combination of both. Of concern is that at Pelican

Island in the Fitzroy Region, where benthic communities are most like the Gladstone reefs, the coral community has shown negligible recovery since 2011 (Thompson *et al.* 2022). What is unclear is whether such long-term hiatus in recovery is natural for coral communities in these marginal conditions or symptomatic of delayed recovery in the face of mounting cumulative pressures associated with climate change and declining water quality.

Overall, the 2023 results further demonstrate the lack of resilience in coral communities within Gladstone Harbour that has been previously reported. Given the current depleted state of coral cover, recovery will depend heavily on the connectivity with reefs beyond the Harbour for larval supply. However, the subsequent settlement and growth of these larvae are likely to remain low until the underlying conditions promoting the continued high cover of macroalgae are identified and mitigated.

6 ACKNOWLEDGMENTS

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8 APPENDICES

8.1 Appendix 1: Data Tables

Table A 1 Site location and transect directions. Minor corrections from those detailed in Thompson *et al*. 2015 are included. Required maintenance of transect markers is indicated (*italic text*). At each transect a steel star picket marks the start point, then there are 10mm diameter sections of reinforcing bar at 10 m and at the end (20 m) of each transect. There is a 5 m gap between consecutive transects within each site.

Reef	Depth	Latitude	Longitude	Transect directions	
Seal Rocks North	1 m	23 57.500	151 29.092	1	295 (*10m rod*)
				2	285 then 310@10 m
				3	300 then 320@10 m
				4	30 then 105@10 m
				5	50 then 60@10 m (*Picket*)
Seal Rocks South	1 m	23 57.825	151 29.215	1	0 then 30@10 m (*Picket*)
				2	30 then 350@10 m (*both rods*)
				3	260 then 250@10 m
				4	190 (*end rod*)
				5	230 (*Picket*)
Rat Island	1 m	23 46.022	151 19.107	1	305 then 300@10 m
				2	300 (*Picket*)
				3	330 then 320@10 m (*Picket*)
				4	330 then 290@10 m
				5	300 then 285@10 m
Facing Island	0-1 m	23 45.801	151 19.687	1	220 then 210@10 m (*10m rod*)
				2	190 then 180@10 m (*10m rod*)
				3	180 then 210@10 m
				4	240 then 230@10 m (*10m rod*)
				5	170 (*Picket*)
Farmers Reef	1 m	23 46.306	151 19.073	1	50
				2	40 then 50@10 m
				3	60
				4	60 then 75@10 m
				5	60 then 40@10 m
Manning Reef	0-0.5	23 51.239	151 21.199	1	30 then 10@10 m, 50 to T2
				2	60 then 0@10 m, 80 to T3
				3	60 then 300@10 m, 300 to T4 (*end rod*)
				4	315 then 20@10 m, 350 to T5 (*end rod*)
				5	320 then 40@10 m

Table A 2 Sub-indicator values for Gladstone Harbour. For the coral change sub-indicator the tabulated values are the mean of the changes in cover from the previous year, scores for this sub-indicator are based on a three-year rolling mean of these changes but only when reefs are not impacted by an acute disturbance event, and also consider the composition of the communities at each reef.

	Year	Juvenile density (m2) *		Coral cover (%)		Cover change (%)		Macroalgae cover (%)	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD
Gladstone Harbour	2015	4.0	1.4	3.1	4.3	NA	NA	23.4	6.6
	2016	3.9	0.1	5.8	1.5	0.8	3.0	41.1	16.9
	2017	4.2	0.4	5.4	0.8	-0.4	2.3	35.3	24.9
	2018	3.7	0.6	4.2	0.7	-1.2	0.0	35.3	24.6
	2019	2.1	0.1	6.5	0.8	2.3	0.1	52.7	21.2
	2020	2.1	0.4	6.5	0.7	0.0	0.2	45.3	13.1
	2021	1.4	0.1	6.7	1.1	0.3	1.7	52.4	9.4
	2022	1.1	0.1	7.5	3.3	0.7	2.2	43.6	6.5
	2023	1.0	0.6	6.2	0.8	-1.2	2.4	55.5	21.5

* Note: values given for juvenile densities are based on the current methodology and have been back calculated for previous years to allow comparison. This applies to all following tables of indicator values.

Table A 3 Indicator and sub-indicator scores for Gladstone Harbour.

	Year	Juvenile density	Coral cover	Coral change	Macroalgae cover	Coral Indicator	
						Score	Grade
Gladstone Harbour	2015	0.28	0.06	NA	0.19	0.18	E
	2016	0.34	0.07	NA	0.03	0.15	E
	2017	0.38	0.07	0.40	0.26	0.28	D
	2018	0.39	0.05	0.32	0.22	0.24	E
	2019	0.23	0.08	0.41	0.01	0.18	E
	2020	0.12	0.08	0.40	0.07	0.17	E
	2021	0.15	0.07	0.34	0.00	0.14	E
	2022	0.12	0.09	0.37	0.04	0.15	E
	2023	0.11	0.08	0.31	0.07	0.14	E

* Note: Juvenile density indicator scores are based on the current methodology and have been back calculated for previous years to allow comparison. Coral Indicator scores for previous years have also been adjusted accordingly. This applies to all following tables of indicator scores.

Table A 4 Sub-indicator values for reporting zones

Zone	Year	Juvenile density (m^{-2})		Combined cover of hard and soft coral (%)		Change in hard coral cover (%)		Macroalgae cover (%)	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD
Mid Harbour	2016	3.9	0.5	4.7	3.1	-1.3	4.1	29.2	8.7
	2017	3.9	1.6	5.9	4.0	1.2	1.7	17.7	16.0
	2018	3.2	1.6	4.7	3.9	-1.2	2.1	17.9	14.9
	2019	2.2	0.8	7.1	6.1	2.4	2.3	37.7	19.8
	2020	2.3	1.4	7.0	4.5	-0.1	2.9	36.0	22.1
	2021	1.4	1.1	6.0	5.1	-0.9	1.9	45.8	16.2
	2022	1.2	0.9	5.2	3.8	-0.9	1.8	39.0	21.4
	2023	1.4	1.4	5.6	3.2	0.5	1.7	40.3	23.3
Outer Harbour	2016	3.9	0.8	6.9	9.7	2.9	4.1	53.1	0.1
	2017	4.5	0.6	4.8	6.5	-2.0	3.2	52.9	7.6
	2018	4.1	0.4	3.8	4.4	-1.1	2.1	52.6	14.8
	2019	2.1	0.5	5.9	7.0	2.2	2.6	67.7	12.7
	2020	1.8	1.0	6.0	5.8	0.1	1.1	54.6	16.4
	2021	1.5	0.8	7.5	9.2	1.5	3.4	59.1	20.1
	2022	1.0	0.1	9.8	8.9	2.2	0.2	48.2	8.9
	2023	0.5	0.2	6.8	6.7	-2.9	2.0	70.7	11.8

Table A 5 Indicator and Sub-indicator scores for reporting zones.

Zone	Year	Juvenile density	Coral cover	Coral change	Macroalgae cover	Coral Indicator Score	Coral Indicator Grade
Mid Harbour	2015	0.23	0.08		0.37	0.23	E
	2016	0.33	0.05		0.10	0.16	E
	2017	0.33	0.08	0.44	0.50	0.33	D
	2018	0.34	0.06	0.30	0.41	0.27	D
	2019	0.24	0.09	0.42	0.02	0.19	E
	2020	0.15	0.09	0.44	0.15	0.20	E
	2021	0.15	0.07	0.43	0.00	0.16	E
	2022	0.13	0.06	0.26	0.07	0.13	E
	2023	0.15	0.07	0.22	0.22	0.14	E
Outer Harbour	2015	0.33	0.05		0.00	0.13	E
	2016	0.33	0.09		0.00	0.14	E
	2017	0.44	0.06	0.37	0.00	0.21	E
	2018	0.45	0.05	0.33	0.00	0.20	E
	2019	0.22	0.07	0.40	0.00	0.17	E
	2020	0.08	0.08	0.39	0.00	0.14	E
	2021	0.15	0.07	0.26	0.00	0.12	E
	2022	0.11	0.12	0.48	0.00	0.18	E
	2023	0;06	0.08	0.40	0.0	0.14	E

Table A 6 Sub-indicator values for individual reefs. Values for change in coral cover are absolute change in hard coral cover between years.

Zone	Reef	Year	Juvenile density (m^2)	Coral cover (%)	Change in hard coral cover (%)	Macroalgae cover (%)
Mid Harbour	Facing Island	2015	5	13.1		24.8
		2016	4.3	6.1	-7	30.6
		2017	3.2	9.8	3.7	27.6
		2018	1.5	8.8	-1	14.5
		2019	1.5	13.5	4.8	40.5
		2020	0.6	9.1	-4.4	48.5
		2021	0.2	8.2	-0.9	63.2
		2022	0.6	5.3	-3	66
		2023	0.3	8	2.6	55.1
	Farmers Reef	2015	3.2	4.8		4.1
		2016	3.8	7.1	2.7	35.9
		2017	6.3	7.2	0	5.8
		2018	5.2	3	-4.1	18
		2019	2.8	3.4	0.3	27.2
		2020	3.9	5.9	2.5	23.3
		2021	2.8	2.1	-3.6	32.2
		2022	2.5	3.6	1.1	16.5
		2023	3.4	3.6	0.4	13
	Manning Reef	2015	2.1	0		32
		2016	3.4	0.1	0.1	33.6
		2017	2.9	0.3	0.1	35
		2018	3.6	0.1	-0.1	37.5
		2019	1.6	0.6	0.5	64.2
		2020	1.8	1.2	0.6	60.1
		2021	1.6	1.4	0.1	55.8
		2022	0.9	1.5	0.1	44.8
		2023	0.5	2.1	0.6	63.6
	Rat Island	2015	1.8	6.4		14
		2016	4.3	5.5	-0.9	16.5
		2017	3.4	6.5	1	2.5
		2018	2.6	6.9	0.4	1.5
		2019	2.8	10.6	3.6	19
		2020	3.1	11.6	0.9	12.2
		2021	0.9	12.1	0.7	31.8
		2022	0.7	10.4	-1.8	28.8
		2023	1.5	8.8	-1.6	29.6

Zone	Reef	Year	Juvenile density (m^2)	Coral cover (%)	Change in hard coral cover (%)	Macroalgae cover (%)
Outer Harbour	Seal Rocks North	2015	5	0		28
		2016	4.4	0	0	53
		2017	4.1	0.2	0.3	58.2
		2018	3.9	0.6	0.4	63.1
		2019	1.7	0.9	0.3	76.8
		2020	1.1	1.9	1	66.2
		2021	0.9	1	-0.9	73.3
		2022	0.9	3.5	2.4	54.4
		2023	0.4	2	-1.5	79.1
	Seal Rocks South	2015	3.2	8.3		58.2
		2016	3.3	13.8	5.8	53.1
		2017	5	9.4	-4.3	47.5
		2018	4.4	6.9	-2.6	42.1
		2019	2.4	10.9	4.1	58.8
		2020	2.5	10.1	-0.6	43
		2021	2.1	14	3.9	44.9
		2022	1.1	16.1	2.1	41.9
		2023	0.7	11.5	-4.4	62.4

Table A 7 Timeseries of indicator and sub-indicator scores for Mid Harbour reefs

Zone	Reef	Year	Scores					Grade
			Juvenile density	Coral cover	Coral change	Macroalgae cover	Coral indicator	
Mid Harbour	Facing Island	2015	0.41	0.16		0.00	0.19	E
		2016	0.46	0.08		0.00	0.18	E
		2017	0.25	0.12	0.50	0.00	0.22	E
		2018	0.16	0.11	0.33	0.46	0.27	D
		2019	0.16	0.17	0.67	0.00	0.25	D
		2020	0.00	0.11	0.33	0.00	0.11	E
		2021	0.02	0.10	033	0.00	0.11	E
		2022	0.06	0.07	0.00	0.00	0.03	E
		2023	0.03	0.10	0.28	0.00	0.10	E
	Farmers Reef	2015	0.26	0.06		1.00	0.44	D
		2016	0.34	0.09		0.00	0.14	E
		2017	0.53	0.09	0.50	0.95	0.52	C
		2018	0.53	0.04	0.33	0.17	0.27	D
		2019	0.31	0.04	0.13	0.00	0.12	E
		2020	0.30	0.07	0.46	0.00	0.21	E
		2021	0.30	0.03	0.46	0.00	0.20	E
		2022	0.27	0.05	0.55	0.29	0.29	D
		2023	0.36	0.05	0.31	0.56	0.32	D
	Manning Reef	2015	0.12	0.00		0.00	0.04	E
		2016	0.25	0.00		0.00	0.08	E
		2017	0.22	0.01	0.51	0.00	0.18	E
		2018	0.40	0.00	0.27	0.00	0.17	E
		2019	0.17	0.01	0.29	0.00	0.12	E
		2020	0.08	0.02	0.34	0.00	0.11	E
		2021	0.18	0.02	0.38	0.00	0.14	E
		2022	0.10	0.02	0.24	0.00	0.09	E
		2023	0.05	0.03	0.19	0.00	0.07	E
	Rat Island	2015	0.11	0.08		0.50	0.23	E
		2016	0.39	0.07		0.29	0.25	D
		2017	0.31	0.08	0.24	1.00	0.41	D
		2018	0.28	0.09	0.26	1.00	0.41	D
		2019	0.31	0.14	0.59	0.09	0.28	D
		2020	0.22	0.15	0.31	0.60	0.32	D
		2021	0.10	0.15	0.54	0.00	0.20	E
		2022	0.07	0.13	0.24	0.00	0.11	E
		2023	0.16	0.11	0.10	0.00	0.09	E

Table A 8 Timeseries of indicator and sub-indicator scores for Outer Harbour reefs.

Zone	Reef	Year	Scores					Grade
			Juvenile density	Coral cover	Coral change	Macroalgae cover	Coral indicator	
Outer Harbour	Seal Rocks North	2015	0.42	0.00		0.00	0.14	E
		2016	0.38	0.00		0.00	0.13	E
		2017	0.36	0.01	0.25	0.00	0.15	E
		2018	0.42	0.01	0.34	0.00	0.19	E
		2019	0.19	0.01	0.46	0.00	0.17	E
		2020	0.01	0.02	0.25	0.00	0.07	E
		2021	0.10	0.01	0.19	0.00	0.08	E
		2022	0.10	0.04	0.50	0.00	0.16	E
		2023	0.05	0.03	0.33	0.00	0.10	E
	Seal Rocks South	2015	0.25	0.10		0.00	0.12	E
		2016	0.28	0.17		0.00	0.15	E
		2017	0.51	0.12	0.50	0.00	0.28	D
		2018	0.48	0.09	0.33	0.00	0.22	E
		2019	0.26	0.14	0.33	0.00	0.18	E
		2020	0.15	0.13	0.00	0.00	0.07	E
		2021	0.20	0.12	0.33	0.00	0.16	E
		2022	0.12	0.20	0.47	0.00	0.20	E
		2023	0.07	0.14	0.47	0.00	0.17	E

Table A 9 Genus level coral cover and abundance of juvenile corals at reefs surveyed in 2023.

Sample type	Location	*Acropora* (Acroporidae)	*Montipora* (Acroporidae)	*Duncanopsammia* (Dendrophylliidae)	*Turbinaria* (Dendrophylliidae)	*Leptastrea* (Leptastreidae)	*Moseleya* (Lobophylliidae)	*Micromussa* (Lobophylliidae)	*Cyphastrea* (Merulinidae)	*Favites* (Merulinidae)	*Paragoniastrea* (Merulinidae)	*Pocillopora* (Pocilloporidae)	*Bernardpora* (Poritidae)	*Goniopora* (Poritidae)	*Porites* (Poritidae)	*Psammocora* (Psammocoridae)	*Pseudosiderastrea* (Rhizangiidae)
Cover (%)	Facing Island	1.38	0.13						0.25						6.13	0.13	
	Farmers Reef		0.13	0.13	0.25				1.38	0.25					0.75	0.75	
	Manning Reef	1.38	0.25		0.13		0.13									0.25	
	Rat Island Reef	0.38			3.00				3.75	1.13				0.13	0.38		
	Seal Rocks Nth	1.38	0.38								0.13						
	Seal Rocks Sth	2.75	0.13		8.00							0.38				0.25	
Juveniles (count)	Facing Island	1			1				1						4		
	Farmers Reef			4	24	2		1	2	1			3		11		1
	Manning Reef	2			5		1			1						4	
	Rat Island Reef				3	1			3	1					21		
	Seal Rocks Nth		1	1	3							3	1			3	
	Seal Rocks Sth	3			7									1		5	

Table A 10 Cover (%) of algae, sponges and sand and silt at reefs surveyed in 2023.

	Red Macroalgae			Brown macroalgae								Green Macroalgae						
	Unidentified	*Asparagopsis*	*Peyssonnelia*	Unidentified	*Dictyota*	*Dictyopteris*	*Lobophora*	*Padina*	*Sargassum*	*Spatoglossum*	*Stypopodium*	*Caulerpa*	Crustose coralline algae	Turf algae		Sand and Silt	*Cliona orientalis* (Sponge)	Other Sponge
Facing Island	0.88	13.88			2.00		8.00		30.38				0.00	22.88		11.00	2.63	0.25
Farmers Reef		9.00			1.13		2.88						0.00	27.63		53.50	1.38	0.88
Manning Reef		27.50			7.25		28.50		0.38				0.50	13.25		20.25	0.00	0.25
Rat island		21.88			3.00		4.75						0.88	24.25		33.25	2.63	0.50
Seal Rocks North	2.13		0.75	0.88	0.13		12.39		62.33		0.25	0.25	0.25	5.88		12.76	0.00	0.00
Seal Rocks South	1.63		0.00	1.50	0.88	0.38	9.25	0.63	47.75	0.13	0.25		0.50	7.50		17.88	0.13	0.13

Table A 11 Causes of coral mortality at time of survey. Area of survey 200 m2 at each reef. Data from 2019-2022 included for comparison. No data are included for Manning Reef where no ongoing mortality was recorded. Bio-eroding sponge is primarily *Cliona orientalis*. Bleaching and physical damage are recorded as the proportion of colonies affected; a range indicates variability among transects.

Reef	Damage	Genus	Colonies affected				
			2019	2020	2021	2022	2023
Facing Island	Bio-eroding sponge	*Porites*	17	22	8	10	17
		Turbinaria			1		
	Atramentous necrosis	*Psammocora*					1
	Bleaching			0-5%			
Farmers Reef	Atramentous necrosis	Cyphastrea			2		
	Bio-eroding sponge	*Cyphastrea*	5	7	4	7	8
		Porites				1	
		Turbinaria	1				
	Bleaching			0-1%			
	Unknown	*Porites*				1	
Rat Island	Atramentous necrosis	Cyphastrea			1	7	1
	Bio-eroding sponge	*Cyphastrea*	6	8	9	5	10
		Plesiastrea	2	1			
		Porites				1	
		Turbinaria	2	4	3	2	4
		Favites		1	1		1
	Black Band Disease	*Turbinaria*		1			
	Bleaching			0-10%			
Seal Rocks North	Bleaching			1-50%			<1%
	Bio-eroding sponge	Favites					1
		Goniopora					1
		Turbinaria					3
Seal Rocks South	Atramentous necrosis	Turbinaria			1		
	Bio-eroding sponge	*Turbinaria*	8	9	7	6	3
		Goniopora					1
		Favites			1	1	1
	Bleaching		0-1%	20-40%			<1%
	Physical			0-1%			
	Unknown	*Acropora*				1	

Table A 12 Size-class distribution of juvenile corals. Values are number of juveniles observed in 100m x 0.34m belt transects (34m2) at each reef. Data from all years of surveys included for comparison.

Reef	Size-class (cm)	2015	2016	2017	2018	2019	2020	2021	2022	2023
Facing Island	0-2	107	67	32	19	13	3	0	4	1
	2-5	28	58	58	20	27	15	5	12	6
Farmers Reef	0-2	32	47	64	56	30	24	21	18	18
	2-5	17	26	39	39	19	39	25	22	31
Manning Reef	0-2	52	55	49	46	18	18	8	7	6
	2-5	6	40	29	45	23	27	28	14	7
Rat Island	0-2	19	48	44	30	17	22	2	5	9
	2-5	23	43	28	26	44	33	14	7	20
Seal Rocks North	0-2	111	80	55	42	17	7	8	10	4
	2-5	31	48	64	69	36	23	19	14	8
Seal Rocks South	0-2	52	27	58	32	13	8	12	7	6
	2-5	30	55	58	64	44	39	30	13	10

8.2 Appendix 2: Rationale for sub-indicator selection and threshold setting.

8.2.1 Coral cover

For coral communities, the underlying assumption for resilience is that recruitment and subsequent growth of colonies is sufficient to compensate for losses resulting from the combination of acute disturbances and chronic adverse environmental conditions. High abundance of coral, expressed as proportional cover of the substratum, can be interpreted as an indication of resilience as the corals are clearly able to survive the ambient environmental conditions. In addition, high cover equates to a large brood-stock, a necessary link to recruitment and an indication of the potential for recovery of communities in the local area. Corals also contribute to the structural complexity of a reef and as such support increased biodiversity and provide important ecosystem services such as the provision of habitat for fishes. Finally, high cover is the most tangible reflection of a healthy coral community and a desirable state from an aesthetic perspective. The consideration of both hard and soft corals in this indicator recognises that all corals have a place on coral reefs and that the cover of an area by any coral is effectively mutually exclusive of another.

The selection of critical values or thresholds for coral cover about which to base assessments of condition is difficult. From MMP observations since 2005 there are no strong indications that either hard or soft coral cover varies substantially along water quality gradients suggesting a common Great Barrier Reef (GBR) wide threshold for coral cover is appropriate. We do, however, acknowledge that differing disturbance histories in space and time are likely to confound any analysis attempting to quantify such a relationship. For the MMP, the setting of a threshold for coral cover is still under discussion, however, is likely to be based on an aspirational target of ~50% cover. This target is informed by two prior assessments of coral cover on nearshore reefs. A broad scale survey of nearshore reefs between Cape Tribulation and the Keppel Islands using the same sampling methods as used in Gladstone Harbour undertaken in 2004 returned a mean cover of hard corals of 33% and of soft coral of 5% (Sweatman *et al*. 2007). This total coral cover mean of 38% was observed following the severe loss of corals that occurred as result of thermal bleaching in 1998 and also 2002 (Berkelmans *et al*. 2004) and so is considered too low as a threshold that would indicate “good condition”. Secondly, a summary of surveys from over 100 sites between Cape Flattery and the Keppel Islands prior to 1996 returned a mean cover of hard corals of 62% (Ayling 1997). In this second study, soft coral cover was not reported, and the surveys were based on a range of video and line intercept techniques. AIMS in-house analysis of coral cover estimates using line intercept (LIT) sampling along the same sites as photo point intercept (PIT) used by the MMP reveal a consistent bias with PIT being ~ 78% of that estimated by LIT ($r^2 = 0.99$). Correcting for technique puts the pre-1996 hard coral cover on inshore reefs at a mean of approximately 48%. Allowing some soft coral cover and rounding to an even percentage, the MMP is looking toward a threshold of 50% for the combined cover of hard and soft coral on inshore reefs. Finally, surveys conducted prior to 2009 in the Mid Harbour reporting zone of Gladstone Harbour had mean hard coral cover of 39% (BMT WBM 2013). Although the BMT WBM (2013) report did not provide a mean estimate for soft coral cover, Figure 4.4 of that report indicates soft coral cover in the Mid Harbour ranged between ~4% - 40%. These figures do not greatly deviate from the 50% combined cover of hard and soft corals likely to be used by the MMP in the future and so we suggest applying a 50% threshold for Gladstone also.

No prior data exist for the Outer Harbour reporting zone and so again we suggest a consistent use of the 50% threshold as this will allow comparison of condition across zones but also other regions of the GBR monitored by the MMP.

8.2.2 Macroalgae cover

Macroalgal (MA) recruitment, growth and biomass are controlled by a number of environmental factors such as the availability of suitable substratum, sufficient nutrients and light, and rates of herbivory (Schaffelke *et al.* 2005). High macroalgal abundance may suppress reef resilience (e.g., Hughes *et al.* 2007, Foster *et al.* 2008, Cheal *et al.* 2013; but see Bruno *et al.* 2009) by increasing competition for space or changing the microenvironment into which corals settle and grow (e.g., McCook *et al.* 2001a, Hauri *et al.* 2010). On the GBR, high macroalgal cover correlates with high concentrations of chlorophyll, a proxy for nutrient availability (De'ath and Fabricius 2010). Once established, macroalgae pre-empt or compete with corals for space that might otherwise be available for coral growth or recruitment (e.g., Box and Mumby 2007, Hughes *et al.* 2007). For the purpose of this indicator, macroalgae are considered as species of the Rhodophyta (Red algae), Phaeophyta (Brown algae) and Chlorophyta (Green algae), excluding the encrusting coralline or short turf like species. The latter two groups are recorded as part of the assessments but are not aggregated into the MA indicator.

The interactions between corals and algae are complex, likely species-specific and, mostly, un-quantified (McCook *et al.* 2001a). Because of this it is difficult to determine realistic thresholds of macroalgal cover from which to infer information about the resilience of coral communities. Recent AIMS analysis of MMP data aimed at determining a threshold for the MA indicator gave a threshold of ~23% for communities in less than 3m depth below lowest astronomic tide (LAT), beyond which the density of juvenile corals declines. This direct influence on coral community replenishment could be used to define an upper bound for macroalgae cover. A further consideration is that within the MMP data set MA cover varies along environmental gradients with highest cover found in turbid areas and where wave or current action precludes the accumulation of fine sediments. As turbidity declines or the proportion of sediments with fine grain sizes increase then the cover of macroalgae also declines. This response to environmental conditions is a further constraint to the expectation of the level of MA cover at many locations. Current thinking within the MMP is to include the threshold mentioned above for an influence of juvenile corals as an upper threshold though reduce this to modelled estimates of cover based on observed relationships between MA cover, turbidity and sediment composition, in cases where these predictions are lower than the threshold for influence on juvenile corals. For the Gladstone Healthy Harbour Partnership monitoring, AIMS has collected sediment samples from each monitoring location and determined sediment grain size composition. The depth of these samples was only 1-2m below LAT and so will not be directly comparable to grain size compositions from MMP reefs that were sampled at the depth of 5m below LAT where wave driven resuspension is generally reduced. The results of the sediment analysis indicate that there is not a substantial accumulation of fine sediments at the coral sampling locations selected in Gladstone Harbour and this along with the limited depth of the reefs suggest turbidity and sedimentation will not be limiting macroalgae cover.

In light of the above considerations an upper bound of 20% cover of macroalgae was adopted for the Gladstone Harbour reefs as this is below the threshold for impacts to juvenile settlement at shallow depths but also recognises that macroalgae cover is a natural component of shallow reef communities in nearshore areas of the southern GBR. The most comparable reef monitored by AIMS to those in Gladstone Harbour is Pelican Island in Keppel Bay. At Pelican Island MA cover declined to ~5% as the coral community at 2m below LAT recovered. The lower bound for cover of MA on Gladstone Harbour reefs was set at 5% as this is in line with cover at Pelican Island during a period that corals were showing strong recovery from past disturbance events but also allowing some natural occurrence of MA. We suggest the threshold for cover for MA be set midway between the lower and upper bounds at 12.5%. We point out that the scoring of this indicator is the inverse to that used for coral cover or juvenile densities as high MA cover is considered a poor indication of coral community condition.

8.2.3 Juvenile density

Common disturbances to inshore reefs include cyclones (often associated with flooding), thermal bleaching, and outbreaks of crown-of-thorns seastar, all of which can result in widespread mortality of corals (e.g., Sweatman *et al.* 2007, Osborne *et al.* 2011). Recovery from such events is reliant on both the recruitment of new colonies and regeneration of existing colonies from remaining tissue fragments (Smith 2008, Diaz-Pulido *et al.* 2009). Previous studies have shown that elevated concentrations of nutrients, agrichemicals, and turbidity can negatively affect reproduction in corals (reviewed by Fabricius 2005, van Dam *et al.* 2011 Erftemeijer *et al.* 2012) and increased organic carbon concentrations can promote coral diseases and mortality (Kline *et al*. 2006, Kuntz *et al*. 2005). Furthermore, high rates of sediment deposition and accumulation on reef surfaces can affect larval settlement (Babcock and Smith 2002, Baird *et al.* 2003, Fabricius *et al.* 2003) and smother juvenile corals (Harrison and Wallace 1990, Rogers 1990, Fabricius and Wolanski 2000). Any of these water quality-related pressures on the early life stages of corals have the potential to suppress the resilience of communities reliant on recruitment for recovery. For these reasons the density of juvenile corals is an important indicator of coral community resilience, especially in periods following severe disturbance events.

The number of juvenile colonies observed along fixed area transects may be biased due to the different proportions of substratum available for coral recruitment. For example, live coral cover effectively reduces the space available for settlement of coral larvae, as do sandy or silty substrata onto which corals are unlikely or unable to settle. To create a comparative estimate of the density of juvenile colonies between reefs and through time, the numbers of recruits observed along fixed transects are converted to densities per area of transect that is 'available' for settlement. This standardisation divides the number of juvenile corals observed along fixed transects by the area of those fixed transects that is not occupied by existing corals or deposits of loose sediments to which corals could not settle.

The setting of a threshold against which to assess observed densities of juvenile corals is problematic as detailed demographic studies that allow the estimation of adequate levels of recruitment that are likely to ensure coral community resilience have not been undertaken for the range of communities present in the turbid nearshore waters of the GBR.

For the MMP selection of thresholds for the scoring of this metric was based on the analysis of recovery outcomes for MMP and LTMP reefs up to 2014 (Thompson *et al*. 2016), which provided a baseline condition from which changes could be inferred as improvements or declines in condition. Changes to the methods for juvenile density estimates outline in this report requires thresholds are also adjusted. Previously, the thresholds for this metric were based on <10cm juvenile size classes, with a mean of 7.5 per m^2 of available substrate being the density at which the indicator score went from ‘poor’ to ‘satisfactory’. For the revised estimates of juvenile corals (<5cm), the mean is 4.6 colonies m^{-2}, with the 10^{th} and 90^{th} percentiles of the distribution being 0 and 13 juveniles per m^2. These observations serve as a guide to the densities of juveniles that can be expected on inshore reefs.

One study that explicitly focused on estimating the density of juvenile corals (<10 cm) required for coral communities to recover rather than shift to an algal dominated state following severe disturbance suggested a threshold of 6.2 juveniles per m^2 (Graham *et al.* 2015). Because this work was undertaken in the Seychelles the relevance to the inshore GBR is unknown. However, considering the similarity between the inshore GBR mean and the threshold of Graham *et al.* 2015, the initial value of 7 juvenile colonies per m^2 of available substrate was adopted for the Gladstone Harbour threshold. As of 2018 a value of 4.6 will set the threshold to account for the reduced size class of <5cm and remains consistent with the threshold of Graham *et al.* 2015.

8.2.4 Cover change

This indicator metric is based on the rate at which hard coral cover increases. While high coral cover can justifiably be considered a positive indicator of community condition, the reverse is not necessarily true. Low cover may occur following acute disturbance and, hence, may not be a direct reflection of the community’s resilience to underlying environmental conditions. For this reason, in addition to considering the actual level of coral cover we also assess the rate at which hard coral cover increases as a direct measure of recovery potential. This indicator reflects the coral growth performance on a per reef basis by comparing observed increase in coral growth (in the absence of acute disturbance) to expected coral growth. Estimates are derived by comparing the observed rate of change in hard coral cover at a given reef to that predicted by a multi-species form of the Gompertz growth equation (Dennis & Taper 1994, Ives *et al.* 2003). The equations used were parameterised from the time-series of coral cover from reefs monitored by the LTMP and the MMP over the period 1987-2007.

The growth models used are parameterised in a Bayesian framework to permit propagation of uncertainty from the two models onto the overall growth expected. For the Gladstone Harbour Report Card, the model is parameterised specifically for 2m depths. Observations of annual change in benthic cover derived from 47 near-shore reefs sampled over the period 1987-2007 were used to parameterise two multi-species Gompertz growth equations. These models returned estimates of growth rates for corals of the family Acroporidae and the combined grouping of all other hard corals. These two groups were modelled separately as the growth rate of Acroporidae is substantially higher than most other corals. Within these model’s growth rate estimates are dependent on the cover of each of these hard-coral groups along with the cover of soft coral which in combination represent space competitors and so limit the area available for coral cover increase.

Model projections of future coral cover on GBR inshore reefs based on the growth rates estimated by these models coupled with the observed disturbance history for inshore reefs of the GBR over the period 1987-2002 indicated a long-term decline in coral cover (Thompson & Dolman 2010).

For this reason, the positive score of 1 was reserved for only those reefs at which the observed rate of change in cover exceeded twice the upper 95% confidence interval of the change predicted. Observations falling within the upper and lower confidence intervals of the change in predicted cover were scored as neutral (indicator score 0.5) and those not meeting the lower confidence interval of the predicted change received an indicator score of 0. The rate of change is averaged over three years of observations. As implemented in 2017 only two years of change were used (2015-2016 and 2017-2017), future applications will be based on a rolling mean of three years of observed changes. Years in which disturbance events occurred at particular reefs were not included as there is no logical expectation for an increase in cover in such situations.

8.2.5 References

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