Coral reef survey methods for assessing important community elements associated with coral reefs

A comparison of underwater visual survey methods used by Reef Life Survey and James Cook University at Elizabeth & Middleton Reefs

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Important disclaimer

Photos

Rick Stuart-Smith

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Middleton and Elizabeth Reefs are platform reefs associated with a seamount chain that extends northward from Lord Howe Island, managed by the Australian Government as part of the Lord Howe Marine Park. The reefs have been subject to a series of ecological surveys by different teams of scientists between the 1990s and 2018. This report explicitly compares results from two independent surveys conducted simultaneously or successively at specific sites, using established protocols developed by researchers from the ARC Centre of Excellence for Coral Reef Studies, based at James Cook University (JCU), and the University of Tasmania and Reef Life Survey (RLS).

JCU surveyed four 50 m long transects in each of two depth strata (2-4 m and 6-10 m) at each of 19 sites (152 transects, covering 38,000 m^2 of reef), recording: i) benthic composition using point intercept counts with corals and macroalgae identified to genera, ii) coral health (e.g., incidence of coral bleaching, disease, and predation) and coral recruitment within 10 x 1 m belt transects, iii) counts of select macro-invertebrates (e.g., holothurians and urchins) within 2m-wide belts, and iv) size and abundance of select fish species along belt transects with varying width depending on species. Larger and more mobile fishes were surveyed in a 5 m-wide belt whilst deploying the transect tape, to minimise disturbance. Smaller fishes were surveyed by a second diver in a 2m-wide belt. In addition, Galapagos sharks and black cod were surveyed along 1-2 long (250-500 x 20 m) transects per site. RLS surveyed a single 50 m transect at each of two different depth contours at each of 42 sites (84 transects covering 42,000 m^2 of reef; often at ~8 and 12 m depth outside the reefs, and 3 and 5 m inside the reefs), recording i) all fishes, including Galapagos sharks and black cod, within 5 m either side of transect, while assigning fishes to specific size-classes, ii) benthic composition using photo-quadrats, where photographs were taken every 2.5 m and then analysed using point counts, iii) small (cryptic) fishes and macroinvertebrates within 1 m either side of the transect. RLS transects were deployed before counting fishes and invertebrates.

A number of metrics recorded during coral reef surveys are comparable between the two teams, most notably live coral cover (%), macroalgae cover (%), benthic composition, the abundance and biomass of key fish species (black cod, Galapagos shark) and functional groups (herbivorous fishes), the abundance of butterflyfishes, damselfishes and *Coris bulbifrons* (doubleheader wrasse), and the abundance of corallivorous invertebrates (Crown-of-Thorns Starfish [CoTS], *Drupella*); these include metrics used globally as indicators of the health and condition of coral reef ecosystems. Other important metrics, such as the abundance of juvenile / recruit corals or a more complete suite of species (e.g. invertebrates) are only recorded by one team or the other. This reflects the fact that the two teams have different objectives for their survey methods.

Of the 72 fish species covered by the survey methods of both teams on a subset of surveys specifically done along the same transects, most had similar density and biomass values. At the community level, however, sampling by the two teams resulted in significantly different benthic and fish assemblages. JCU survey results included a greater proportion of *Acropora* and *Pocillopora* spp. corals, green and red macroalgae, planktivorous damselfishes and some herbivorous fish species; RLS recorded more sponges, soft corals,

massive, foliose and encrusting corals and brown macroalgae, and higher densities of some *Kyphosus* spp. and black cod *Epinephelus daemelii.* Live coral and turf cover was significantly correlated between the two methods at the local transect scale; total macroalgal cover, the density of grazing fishes, and fish species richness showed no correlation.

When each team used their own methods, the overall conclusions at the reef scale were very similar. When assessing patterns on individual matched transects, community level patterns and some indicators differed considerably, despite a general agreement at the species level. This variability at the transect scale could be due to inherent patchiness in the communities, and/or the differences in methods; this remains one of the unknown factors in the analysis. A comprehensive assessment of the two methods would require a much greater number of matched transects across a wide range of habitats and depths. Unfortunately, time restrictions and different objectives precluded such a detailed quantitative comparison. The primary differences most likely relate to the scale at which each team seeks the greatest accuracy; there is a trade-off between replication at the reef level and at the site level. Metrics that vary most at the local site scale (e.g. black cod, sharks, rare species of corals) are likely to be better captured by the JCU approach, while metrics that would be better characterised by conducting surveys at more sites and a synoptic approach (e.g. overall fish community composition, average live coral cover across the full reef system) are likely to be better captured by RLS.

Differences in the methods used to sample coral reef assemblages invariably lead to differences in the results and sometimes conclusions. The RLS method (42 sites, 168 transect blocks) is designed to capture variation between sites and relies on surveys of more sites to detect changes through time at the scale of reefs; the JCU method (19 sites, 152 transects) is designed to capture more variation within sites and provide a more accurate record of change within those individual sites. Thus, the methods reflect different points along a trade-off of understanding change at small (site) to large (reef/region) scales. Ultimately, the design of any monitoring program needs to be driven by specific questions relating to management priorities. In cases where change needs to be evaluated over scales of entire reefs or MPA zones, site replication and understanding net reef- or zone-level responses (increases or decreases in indicator values) may be prioritised over within-site replication and measurement accuracy for the magnitude of change in indicator values at those sites. On the other hand, where change needs to be evaluated at finer scales, such as for management of tourism impacts or CoTS eradication programs, for example, or how different sites/habitats respond to disturbance, accuracy of change in indicators at the site level may be more critical to determine threshold values for management success. As such, the best approaches for informing management decisions should always be guided by clearly articulated management goals that define the expected scale of impacts and management responses.

5 Introduction

The condition of coral reef ecosystems is declining worldwide (Gardner et al. 2003; Bruno and Selig 2007) owing to global climate change (Hughes et al. 2017; Hughes et al. 2018) superimposed upon perennial pressures associated with coastal development and direct exploitation of marine species (Jackson et al. 2001; Pandolfi et al. 2003). Even before recent effects of global climate change, which has caused successive years of severe coral bleaching in many coral reef regions (Hughes et al. 2018), 20% of the world's coral reefs had been effectively destroyed (Wilkinson 2008). Increasing and cumulative pressures are leading to the transformation of complex habitats with diverse communities of habitat-forming corals to depauperate habitats with limited coral cover, low coral diversity and declining structural complexity (Heery et al. 2018). Even isolated coral reef systems, which are far removed from urban centres and relatively unaffected by coastal development and land-based sources of nutrients, sediments and pollutants, are impacted by pervasive effects of global fisheries (Graham et al. 2006) and climate change (Hughes et al. 2003; Hoegh-Guldberg et al. 2007). In fact, these isolated reefs may be more vulnerable to these pressures due to their distance from sources of larvae to replenish them after disturbance events or periods of heavy exploitation.

Coral reef environments are naturally subject to high levels of disturbances, reflected in the dynamic nature of benthic assemblages (Pratchett et al. 2011a) and associated fish and invertebrate communities (Stuart-Smith et al. 2018). Detecting long-term trends is, therefore, conditional on successive sampling over an extended period (Fabricius and De'ath 2004). Similarly, when testing the effectiveness of specific management actions (e.g. no-take marine reserves, land use changes to ameliorate water quality, protection of individual species), it is only through regular monitoring that biological outcomes of management actions can be understood (Houk and van Woekik 2013). For assessing impacts and detecting the effects of management, monitoring programs must be capable of detecting environmental and ecological change at relevant spatial and temporal scales (Marsh and Trenham 2008).

5.1 Established survey methods

Marine benthic assemblages are surveyed using a large variety of sampling methods, including *in situ* surveys using quadrats and transects, video and photographs later analysed in the laboratory, GIS mapping and remote sensing technology (Goodell et al. 2018). In one comparative analysis, video surveys and digital photoquadrats were considered the most efficient means of estimating habitat cover; the line-intercept-transect method was deemed the least efficient (Molloy et al. 2013). However, for temporal sampling of individual sites, photoquadrats were found to be relatively insensitive to changes in coral cover and condition, unless they are permanently fixed (Molloy et al. 2013). There are also greater constraints on taxonomic discrimination when using photographs or videos, rather than documenting benthic composition *in situ*. Moreover, there are inevitable trade-offs between scale, resolution, representation and replication, that must be considered when implementing monitoring programs; most established monitoring programs

focus on pre-determined habitats (e.g., AIMS-LTMP) to minimise variation due to 'environmental' factors such as depth, aspect and distance from shore.

Non-destructive techniques such as underwater visual surveys (UVS), also referred to as underwater visual census (UVC), are among the most common techniques for quantifying abundance and biomass of fishes (Cheal and Thompson 1997). UVS is relatively inexpensive and, with adequately trained observers, can provide quick estimates of diversity, abundance and length frequency distributions of any chosen suite of fish species. Transect surveys, one of the most popular techniques, involve one or more observers swimming for a defined distance or a set time and counting individual fishes within a strip of pre-determined width (Thanopoulou et al. 2018), ideally tailored to the taxonomic group (Cheal and Thompson 1997). Comparisons of the accuracy, efficiency and biases of different methods have found that density estimates for some reef fish species can be influenced by deploying the transect before or during the fish count (Emslie et al. 2018), the level of experience of the observer (correct identification and accurate length estimation), their swimming speed, ability to exclude individuals that have already been counted, accuracy of estimates of transect width, behaviour of target fishes towards a diver (attraction or evasion) and detectability of target species (Caldwell et al. 2016; Emslie et al. 2018 and references therein). The low detectability of many cryptic, nocturnal and pelagic fishes, for example, means that these entire groups are often excluded from surveys because of the low probability of obtaining a reliable estimate during routine daytime assessments (Ackerman and Bellwood 2000).

5.2 Study locations

Middleton and Elizabeth Reefs are platform reefs associated with a seamount chain that extends northward from Lord Howe Island (Woodroffe et al. 2004). Both reefs consist of an extensive lagoon surrounded by a well-defined reef crest with characteristic spur and groove formations, broken only by channels on the northern edges (Choat et al. 2006). Early research was conducted in the 1980s, with subsequent Australian Museum surveys (Australian Museum 1992). This was followed by a series of ecological surveys conducted by the Australian Institute of Marine Science (AIMS), James Cook University (JCU) and Reef Life Survey (RLS) (Oxley et al. 2004; Choat et al. 2006; Pratchett et al. 2011b; Hoey et al. 2014; Edgar et al. 2016). The 2013 and 2018 surveys by RLS and UTAS used identical methods and similar sites. The last three JCU surveys (Pratchett et al. 2011b; Hoey et al. 2014; Hoey et al. 2018) also used identical methods that are very similar, if not the same, as those of Choat et al. (2006). Despite the sequential nature of these surveys, variation in the methods, taxonomic resolution and sites used raises the question of the extent that results are comparable and can be effectively combined to represent temporal trends in the status and condition of these reef systems.

This report explicitly compares results from two independent surveys conducted simultaneously or successively in 2018, using established protocols developed by researchers from the ARC Centre of Excellence for Coral Reef Studies, based at James Cook University (JCU) versus Reef Life Survey (RLS) methods employed by RLS and University of Tasmania (UTAS) divers. It should be stated from the outset that the different approaches used by JCU versus RLS were never intended to be directly comparable and vary due to differences in the intended purpose and specific focus of their respective surveys. Most notably, RLS developed survey protocols that encompassed all large macroscopic taxa and could be used

across a broad range of habitats (including both tropical and temperate marine systems) to assess broad-scale patterns of biodiversity and community structure. The RLS focus was thus on the scale of reefs and management zones, as well as regions, countries and global patterns, rather than variation within individual sites. In contrast, the surveys methods used by JCU were developed and implemented specifically to assess the health and condition of functionally-important components of coral reef ecosystems (and even more specifically, shallow, coral-dominated environments) at the site and habitat type scales. Moreover, surveys are conducted within standardised habitats and with site-level repetition of transects, explicitly intended to enable statistical comparisons of temporal trends within and among fixed sites.

6 UVC Survey Methods

6.1 Reef Life Survey

6.1.1 Sampling design and survey protocol

These survey methods are designed to maximise the collection of ecological information related to all conspicuous taxa during a single dive (https://reeflifesurvey.com/wp-content/uploads/2015/07/NEW-Methods-Manual_150815.pdf) for two divers. The basic unit monitored is a 50-m long transect line. Along this transect line three survey methods are applied, each focusing on different major taxonomic groupings (Method 1 for fishes, Method 2 for macro-invertebrates and cryptic fishes, and Method 3 for coral and other sessile invertebrates and algae). The width of survey area is different for Method 1 (5 m), Method 2 (1 m) and Method 3 (0.5 m). Most frequently, two depths are surveyed at each site (typically 6-8 m and 10-15 m for sites outside the reef, and 1-3 and 3-6 m for sites inside the reefs); one 50-m transect with two adjoining blocks is surveyed at each depth.

6.1.2 Fish surveys (Method 1)

Fish census protocols involve a diver laying out a 50 m transect line along a depth contour on reef. The number and estimated size-category of all fishes sighted within 5 m “blocks” either side of the transect line are recorded to species level as the diver swims along slowly up and down each side (Figure 1). Large, diver shy fishes that leave the survey area while the fish survey is being conducted are recorded as ‘Method 0’ records, that is, they are not included in quantitative analyses, but are recorded as being present at that site on that date. The transect line is laid immediately prior to commencement of counts. Size-classes of total fish length (from snout to tip of tail) used are 25, 50, 75, 100, 125, 150, 200, 250, 300, 350, 400, 500, 625 mm, and above. Lengths of fishes larger than 500 mm are estimated to the nearest 12.5 cm and individually recorded. Fish abundance counts and size estimates were converted to biomass estimates using length–weight relationships for each species (in some cases genus and family) from Fishbase (www.fishbase.org). In cases where length–weight relationships were described in Fishbase in terms of standard length or fork length rather than total length, additional equations provided in Fishbase allowed conversion to total length, as estimated by divers. For improved accuracy in biomass assessments, the bias in divers’ perception of fish size underwater was additionally corrected using relationships presented in Edgar et al. (2004). Note that estimates of fish abundance made by divers can be greatly affected by fish behaviour for many species (Edgar et al. 2004); consequently, biomass determinations, like abundance estimates, can reliably be compared only in a relative sense (i.e. for comparisons with data collected using the same methods) rather than providing an accurate absolute estimate of fish biomass for a patch of reef.

6.1.3 Macroinvertebrate and cryptic fish surveys (Method 2)

Larger macro-invertebrates (molluscs, echinoderms and crustaceans > 2.5 cm) and cryptic fishes (i.e. inconspicuous fish species closely associated with the seabed that are likely to be overlooked during general fish surveys) are censused along the same transect lines set for fish surveys (Figure 1). Divers swim along the bottom, up then down each side of the transect line, recording all mobile macroinvertebrates and cryptic fishes on exposed surfaces of the reef within 1 m of the line.

6.1.4 Macroalgal and sessile invertebrate surveys (Method 3)

Information on the percentage cover of sessile animals and algae along the transect lines set for fish and invertebrate censuses is recorded using photo-quadrats taken sequentially each 2.5 m along the 50 m transect (Figure 1). Digital photo-quadrats are taken vertically-downward from a height sufficient to encompass an area of at least 0.3 m x 0.3 m. The percentage cover of different macroalgal, coral, sponge and other attached invertebrate species in photo-quadrats is digitally quantified in the laboratory using the Coral Point Count with Excel extensions (CPCe) software (Kohler and Gill 2006) or an equivalent grid-based method for annotation. For general habitat cover characterisation, a grid of 5 points is overlaid on each image and the taxon lying directly below each point recorded, providing counts of composition under 100 points per transect. Identification is either to the lowest possible taxonomic resolution, with taxa for which identification is uncertain grouped with congeners or other members of the family or order, or following morphologically-distinctive functional groups that can be mapped to the CATAMI system.

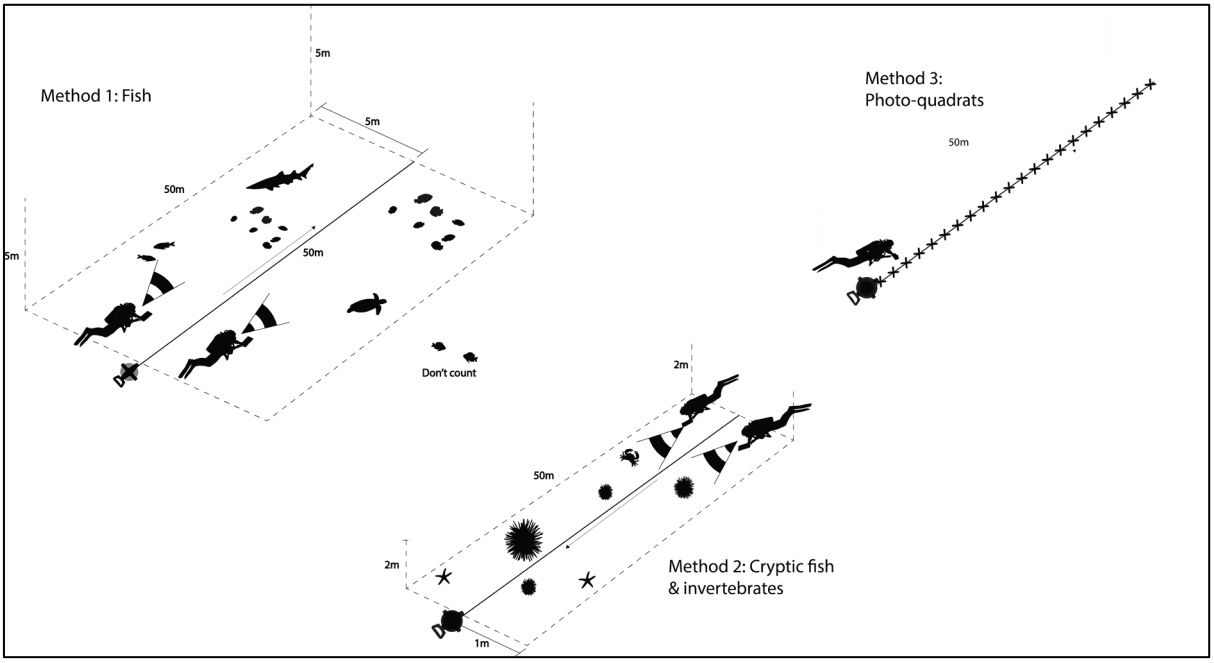

Figure 1. Schematic diagram of survey methods used by RLS.

6.2 ARC Centre of Excellence for Coral Reef Studies, James Cook University

6.2.1 Sampling design

At each site, surveys are conducted within each of two different depth zones, i) the reef crest at 2-4 m and ii) the reef slope at 6-10 m. In each depth zone at each site, four replicate 50 m transects are laid end-to-end and parallel to the depth contour, with a minimum of 10 m between adjacent transects. Along each transect (a) benthic cover and composition, (b) reef topographic complexity, (c) juvenile coral densities (as a proxy for coral replenishment), (d) coral condition/health, (e) herbivorous fish assemblages, (f) endemic and site-attached fish densities, (g) apex predator densities, and (h) unitary invertebrates (or mobile macro-invertebrates) are quantified by a four diver team, with one dive at each site.

6.2.2 Coral reef fishes

Herbivorous reef fishes – Functionally important (mostly, roving herbivorous) coral reef fishes are surveyed using underwater visual census within a 5 m wide belt along each 50-m transect at every study site (Figure 2). Each transect consists of a diver swimming along the depth contour and recording all fishes from the Acanthuridae (surgeonfishes), Scarinae (parrotfishes), Kyphosidae (drummers), and Siganidae (rabbitfishes) within a 5-m belt while simultaneously deploying the transect tape. This procedure minimises disturbance prior to censusing and allows a specified area to be surveyed. Individual fishes are identified to species and placed into 5 cm size categories. Care is taken not to census fish that leave and subsequently re-enter the transect area. Fish densities are converted to biomass using published length-weight relationships for each species, following Hoey and Bellwood (2009).

Endemic and site-attached reef fishes – Smaller more site-attached species from the families Chaetodontidae (butterflyfishes) and Pomacentridae (damselfishes), and juvenile double-header wrasse (*Coris bulbifrons*) are counted by a second diver (following the first) along a 2 m wide path along each transect (i.e., 50 x 2 m) at every study site (Figure 2). Adult double-header wrasse (*C. bulbifrons*) are surveyed together with the herbivorous fishes in a 5 m wide belt along each transect (i.e., 50 x 5 m) at each study site.

Apex predators - To effectively survey larger and highly mobile reef-associated fishes (namely black cod and sharks), 250 – 500 m long underwater visual transects are conducted along the reef at each study site, following Robbins et al. (2006). Transects are run approximately parallel with the reef crest and cover a depth range of approximately 4 – 20 m (Figure 2). All black cod (*Epinephelus daemelii*) and sharks (primarily the Galapagos shark, *Carcharinus galapagensis*) are recorded within 10-m either side of the transect path (i.e., 20 m wide belt), giving a total sample area of 10,000 m^2 (1 hectare) for a 500 m transect, and 5,000 m^2 (0.5 hectare) for a 250 m transect. Two transects are conducted at each site, one 500 m in length and the other 250 m in length.

6.2.3 Habitat structure

Benthic cover and composition - To quantify percentage cover of corals, macroalgae and other sessile organisms the point-intercept method is used, recording the specific organisms or habitat types underlying each of 100 uniformly spaced points (50 cm apart) along each 50-m transect (Figure 2). All hard (scleractinian) and soft (alcyonarian) corals are identified to genus. For survey points that do not intersect live coral, the underlying habitat is categorised as macroalgae (> 5 mm in height; identified to genera), turf algae (< 5 mm in height), crustose coralline algae (CCA), rubble or sand. In doing so, these surveys provide data on cover of major habitat forming and other sessile taxa, as a proportion of available substrata.

Topographic complexity - Topographic complexity is estimated visually at the start of each transect, using the six-point scale formalised by Wilson et al. (2007), where 0 = no vertical relief (essentially flat homogenous habitat), 1 = low and sparse relief, 2 = low but widespread relief, 3 = moderately complex, 4 = very complex with numerous fissures and caves, 5 = exceptionally complex with numerous caves and overhangs.

Coral replenishment - In addition to adult cover (quantified as % cover of different taxa) the density (number per 10 m^2) of juvenile corals is quantified at each site, as a means for assessing rates of replenishment and population turnover among coral populations. The density of juvenile coral colonies (<5 cm maximum diameter, following Rylaarsdam (1983)) are recorded within a 10 x 1 m belt at the start of each transect (Figure 2). The density of juvenile corals does not necessarily provide a good proxy for spatial variation in settlement rates or larval supply (Penin et al. 2010), but is ecologically relevant for assessing population replenishment because it encapsulates variation in both settlement rates of coral larvae and early post-settlement mortality, to effectively measure the number of new coral colonies likely to become established in each habitat and location. Spatial variation in densities of juvenile corals is particularly important in assessing the recovery potential of coral assemblages following acute episodic disturbances, such as coral bleaching (Hoey et al. 2011).

Coral health - To assess local impacts of the corallivorous crown-of-thorns starfish (*Acanthaster* cf. *solaris*), pin-cushion starfish (*Culcita novaeguineae*) and the corallivorous gastropod *Drupella* spp., as well as other large and potentially destructive coral predators such as excavating parrotfishes (Choat et al. 2006; Hoey and Bellwood 2008), any evidence of feeding scars are recorded 1-metre either side of the 50-m transect path (i.e., 50 x 2 m belt). Large parrotfishes, pufferfishes, and triggerfishes are all known to feed on live corals (Hoey and Bellwood 2008; Bonaldo et al. 2014), but unlike other corallivores such as the butterflyfishes that 'pick' at individual polyps when feeding, these larger fishes remove both the coral tissue and the underlying coral skeleton when feeding. When in sufficiently high densities, these fishes can have a large impact on coral populations (Hoey and Bellwood 2008). In addition to signs of corallivory, any evidence of adverse coral health, such as coral bleaching, and coral disease are also recorded at the colony level within the 50 x 2 m belt transects.

6.2.4 Unitary invertebrates

Unitary invertebrates - Counts of unitary invertebrates (mainly holuthurians, urchins, *Acanthaster planci* and *Drupella* spp.) are conducted within a 4-m wide belt along each transect (Figure 2). Densities of all functionally important motile macro-invertebrates are recorded 2-m either side of the 50-m transect line (i.e. 50 x 4m), giving a sample area of 200 m^2 following Pratchett et al. (2011b).

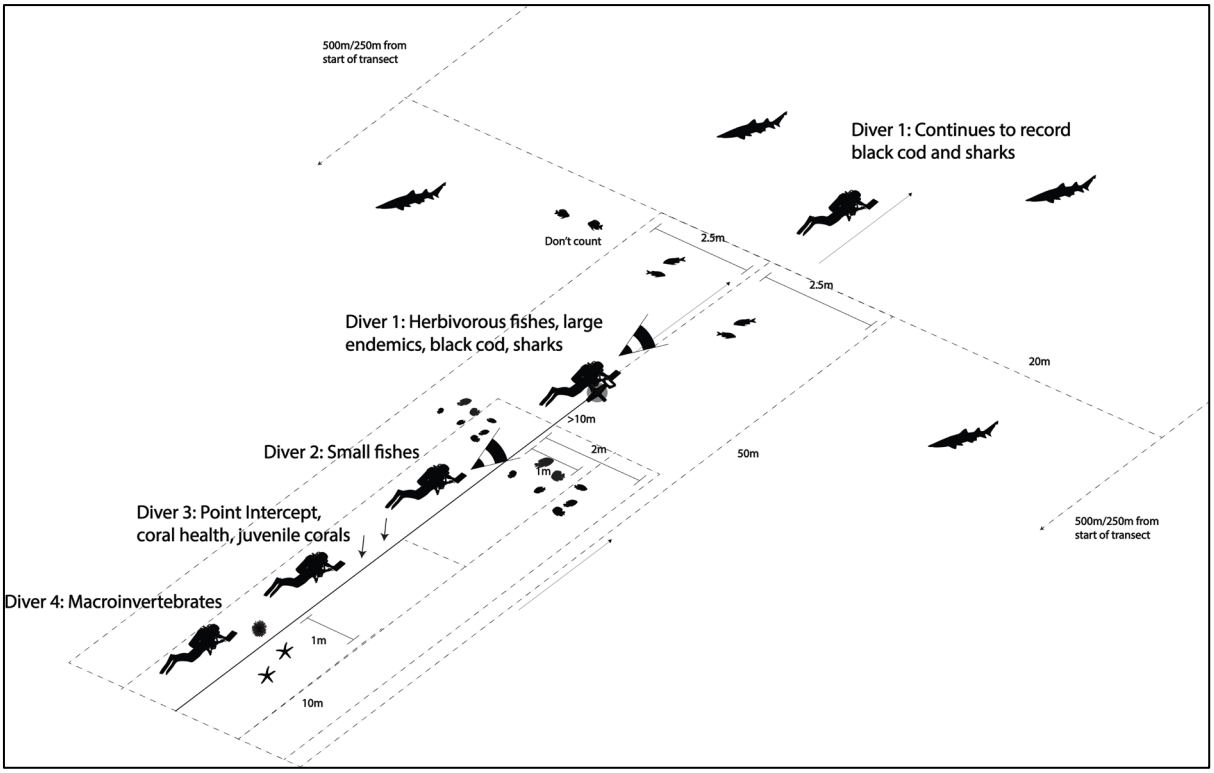

Figure 2. Schematic diagram of survey methods used by the ARC CoE / JCU.

7 Elizabeth and Middleton Reefs: Summary of results

The JCU team surveyed 152 transects at 19 sites, whilst the RLS team surveyed 84 transects at 42 sites. A number of metrics recorded during coral reef surveys can be calculated by using both RLS and JCU approaches (Table 1), most notably live coral cover (%), macroalgae cover (%), benthic composition, the abundance and biomass of key fish species (black cod, Galapagos shark) and functional groups (herbivorous fishes), the abundances of butterflyfishes, damselfishes and *Coris bulbifrons* (doubleheader wrasse), and the abundance of corallivorous invertebrates (CoTS, *Drupella*); these are used globally as some of the indicators of the health and condition of coral reef ecosystems. Other important metrics, such as the abundance of juvenile corals or a more complete suite of species (e.g. invertebrates) are only recorded by one team or the other.

Table 1. Key metrics of coral reef ecological condition that can be calculated using the methodologies reported here (not all possible metrics are listed).

Metric	JCU	RLS
Benthic community structure	✓	✓
Live hard coral cover (%)	✓	✓
Coral composition (genera)	✓	in part
Bleaching	✓	✓
Algal cover (%)	✓	✓
Coral recruit density	✓	in part
Reef fish community structure / species richness	in part	✓
Herbivorous fish biomass	✓	✓
Biomass of fishes >20 cm (B20)	in part	✓
Black cod abundance/size structure	✓	✓
Galapagos shark abundance/size structure	✓	✓
Community Temperature Index (CTI)	in part	✓
Fish functional group richness	no	✓
Cryptic fish densities and richness	no	✓
Mobile invertebrate community structure	no	✓
Mobile invertebrate species richness	no	✓
CoTs densities	✓	✓
Drupella densities	✓	✓
Sea urchin densities	✓	✓
Holothurian densities	✓	✓

7.1 Reef Life Survey 2018 key findings

The primary changes noted in the time between 2013 (January) and 2018 (February-March) were a sharp increase in cryptic fishes, a minor increase in large herbivorous fishes and a decline in turf cover (For detailed results see Table 2). Fishes of conservation interest and endemic species recorded in high densities included the regional endemic doubleheader wrasse *Coris bulbifrons*, black cod *Epinephelus daemelii* and Galapagos shark *Carcharhinus galapagensis*. These species remained relatively stable between 2013 and 2018, although a significant decline in black cod occurred at Elizabeth Reef.

The increase in cryptic and herbivorous fishes, which was also recorded on the southern Great Barrier Reef and Coral Sea during the same period, was hypothesised to be related to warmer seas across the broader region in recent years, and especially the heatwave that caused the 2016 mass coral bleaching event along the GBR and Coral Sea. The consistency in the observations of herbivorous fishes and populations of small cryptic fishes with patterns observed much further north along the GBR and Coral Sea suggest the same large-scale drivers, and point to elevated water temperature as most likely responsible.

The cover of living benthos on both reefs was still dominated by low-lying turf growing on a dead coral base in 2018. This appears to be typical of highly exposed reef fronts, but also of reefs that have suffered past disturbances and coral mortality. A decline in turf cover observed at both reefs could be related to the increases in other sessile invertebrates. Live hard coral cover was higher at both reefs than that recorded on RLS surveys in 2013 (>30% increase at Middleton Reef), and by other researchers in 2011 and 2014, suggesting that corals have either been recovering from earlier disturbances or at least remained stable over the last decade. Slow recovery is expected on isolated sub-tropical reefs such as the EMR, where connectivity to source reefs and growth rates of corals are naturally low.

Illegal fishing at Elizabeth Reef may be partly responsible for an observed decline in black cod. During this most recent survey, black cod biomass was similar to that recorded in 2013 at Middleton Reef, but was significantly lower than previously recorded at Elizabeth Reef, and lower than at Middleton Reef in 2018 surveys.

7.2 James Cook University 2018 key findings

The main findings in 2018 include:

- Mean cover of hard (Scleractinian) corals was high in 2018 (Elizabeth Reef: 39.95% ±1.72 SE; Middleton Reef: 26.62% ±1.87 SE) and has increased at both reefs by >40% since 2014. There was however, low incidence of coral bleaching in 2018, which may have worsened after surveys were completed due to the significant accumulation of heat stress that occurred in 2017/18.

- Aside from bleaching, there was very low incidence of coral predation, disease, and or other unexplained coral injuries. Very few crown-of-thorns starfish or *Drupella* spp. were recorded in 2018, though there is evidence of a persistent low-density population of crown-of-thorns starfish at Elizabeth Reef. Given their considerable reproductive potential and low visitation to these locations, it might be prudent to remove these starfish so as to minimise the likelihood of future population outbreaks.

- Cover of macroalgae (conspicuous seaweeds >5 mm in height) was high in back reef and lagoonal habitats, but mostly very low in exposed reef habitats. Both the cover and the predominant type of macroalgae found at each site was largely unchanged since 2014. Similarly, the physical structure (topographic complexity) of reef habitats was very consistent among sites and largely unchanged from 2011 to 2018.

- The abundance of herbivorous fishes recorded at Elizabeth and Middleton Reefs in 2018 were similar to those recorded in 2011 and 2014 (Elizabeth: 2011: 67.3 fishes/250m^2, 2014: 54.8 fishes/250m^2, 2018: 58.8 fishes/250m^2; Middleton: 2011: 72.8 fishes/250m^2, 2014: 59.6 fishes/250m^2, 2018: 60.0 fishes/250m^2).

- The abundances of endemic fishes (*Amphiprion mccullochi*, *Chaetodon tricinctus*, and *Coris bulbifrons*) varied greatly among study sites, but have changed very little from 2011 to 2018. However, the abundance of *C. tricinctus* at Elizabeth Reef has declined from 1.89 fish/100m^2 in 2014 to 1.28 fish/100m^2 in 2018, despite increases in coral cover during this period.

- The abundance of Galapagos sharks (*Carcharinus galapagensis*) at Elizabeth Reef in 2018 (10.2 individuals/ha ± 4.0 SE) was higher than recorded in 2014 (6.6 individuals/ha ± 1.7 SE). In contrast, the abundance of *C. galapagensis* at Middleton in 2018 (8.9 individuals/ha ± 2.4 SE) was lower than recorded in 2014 (17.1 individuals/ha ± 3.0 SE).

- The abundance of black cod (*Epinephelus daemelii*) at Elizabeth (1.5 fish/ha ± 0.4 SE) and Middleton Reefs (1.4 fish/ha ± 0.6 SE) in 2018, was lower than recorded in 2014 (2.5 fish/ha ± 0.6 SE and 2.2 fish/ha ± 0.5 SE, respectively).

8 Comparison of 2018 results – reef scale

Comparing key results obtained by the two teams, using all replicates, shows whether the scale of replication combined with sampling protocol, can yield similar results. Many variables were similar, including important metrics like live coral cover, where both teams obtained estimates of ~35-40% at Elizabeth Reef and 21-27% at Middleton Reef (Table 2). Turf cover, the abundance and biomass of some key species of fishes and the density of corallivorous CoTs were also similar. JCU recorded higher cover of macroalgae, which could be due to the difference between photoquadrat and *in-situ* recording, or to differences in classification. RLS recorded higher abundance and biomass of herbivorous fishes, black cod and Galapagos sharks, especially at Middleton Reef. This could be due to the greater number of sites, and therefore also habitats, surveyed by RLS (34; 24 sites, not all with two depths more than 4m apart) compared with JCU (18; 9 sites x 2 depths). For areas that are zoned at the reef scale, such as the EMR (where each reef is a different management zone), these conclusions are useful for management, as they suggest that at least for some key metrics the survey methods are interchangeable.

Table 2. Reef-wide average values for key variables measured by each team in 2018, using all transects and sites. Standard errors are shown in brackets.

	JCU		RLS	
Metric	**Elizabeth**	**Middleton**	**Elizabeth**	**Middleton**
Live hard coral cover (%)	39.95 (1.72)	26.62 (1.87)	37.96 (3.47)	21.48 (2.17)
Macroalgae cover (%)	6.57 (0.94)	9.56 (1.38)	1.22 (0.68)	1.36 (0.67)
Turf cover (%)	34.21 (1.97)	42.21 (2.76)	35.04 (4.27)	48.55 (4.32)
Herbivorous fish abundance (individuals/250m^2)	58.8 (6.01)	60.0 (5.38)	83.34 (8.32)	141.51 (26.43)
Herbivorous fish biomass (kg/250m^2)	17.45 (1.92)	24.50 (3.79)	36.98 (10.55)	64.44 (17.01)
Black cod abundance (individuals/ha)	1.50 (0.35)	1.41 (0.55)	2.22 (0.82)	2.33 (0.86)
Black cod biomass (kg/ha)	16.05 (4.16)	19.41 (5.77)	19.17 (7.21)	39.49 (15.17)
Galapagos shark abundance (individuals/ha)	10.2 (4.0)	8.9 (2.4)	12.44 (2.98)	23.0 (5.56)
Galapagos shark biomass (kg/ha)	379.64 (204.39)	208.02 (102.43)	192.50 (43.37)	395.16 (99.53)
Amphiprion mccullochi abundance (individuals/250m^2)	0.38 (0.26)	0.03 (0.02)	0.22 (0.13)	0.98 (0.48)
Chaetodon tricinctus abundance (individuals/250m^2)	4.70 (0.72)	4.32 (0.92)	2.47 (0.47)	1.77 (0.43)
Coris bulbifrons abundance (individuals/250m^2)	1.10 (0.17)	1.60 (0.23)	3.44 (0.85)	2.48 (0.43)
CoTs densities (individuals/100m^2)	0.01 (0.01)	0	0.08 (0.08)	0

9 Comparison of 2018 results – transect scale

Comparing the results obtained by the two teams using different sampling protocols at the scale of individual transects allows a more targeted investigation of methodological differences without the effects of different replication at two different scales. We compared the abundance and biomass of fish species surveyed by both teams at 16 transects at 10 sites (four at Elizabeth Reef, six at Middleton Reef) at which both methods were applied. MDS, Permanova and regression analyses were used for analysis of the overlapping metrics. Firstly, we selected sites that had been surveyed by both teams, with all replicates, and standardised the categories and nomenclature of recorded taxa and the area covered by each team (to % cover for benthos and to individuals or kg 100 m^{-2}, the smallest unit, for fishes). We conducted an MDS of the Bray-Curtis similarity matrix of the log (x+1) transformed data.

Thirteen benthic categories common to both teams were compared along the matched transects. The JCU team recorded higher percent cover of macroalgae and *Pocillopora* spp. (Table 3).

Of the 72 fish species surveyed by both teams on matched transects, 23 species were only recorded by one team; of the remaining 49, 44 species (90%) had similar density values (Table 4). RLS recorded higher densities of *Carcharhinus galapagensis*, *Coris bulbifrons* and *Scarus ghobban*, while JCU recorded higher densities of *Parma polylepis*. Of the 23 species for which biomass was recorded by JCU, all but two had similar biomass; RLS recorded higher biomass of *Carcharhinus galapagensis* and *Chlorurus spilurus*.

Table 3. Percent cover of key benthic categories recorded by both teams along matched transects at 10 sites. Comparability and *p* values determined by Pearson correlation.

Category	JCU	RLS	Comparison
Algal turf	38.88	39.13	Comparable
Live coral cover	42.00	27.44	Comparable
Macroalgae	7.31	2.44	JCU>RLS ($p<0.05$)
Pocillopora spp.	1.81	0.38	JCU>RLS ($p<0.01$)
Massive coral	6.44	5.06	Comparable
Foliose coral	0.31	0.31	Comparable
Soft coral	1.19	1.50	Comparable
Caulerpa spp.	0.50	0.19	Comparable
Halimeda spp.	0.06	0.13	Comparable
Crustose coralline algae	12.50	21.69	Comparable
Rubble	4.81	3.13	Comparable
Sand	2.81	1.44	Comparable

Table 4. Average abundance and biomass, standardised to individuals per 100m^2, of fish species surveyed by both teams. All transects are used for each site surveyed by both teams.

	N. per 100m^2		Kg per 100m^2		**Density**	Biomass
Species	JCU	RLS	JCU	RLS	Comparison	Comparison
Abudefduf vaigiensis	5.00	0.67		0.04	Comparable	
Acanthurus albipectoralis	2.80	7.20	0.64	3.55	Comparable	Comparable
Acanthurus dussumieri	1.40	1.45	0.73	1.18	Comparable	Comparable
Acanthurus nigrofuscus	7.92	4.40	0.24	0.39	Comparable	Comparable
Acanthurus nigroris		0.80		0.06		
Acanthurus olivaceus		0.40		0.02		
Acanthurus triostegus	16.00		0.35			
Amphiprion mccullochi	2.00	0.80		0.03	Comparable	
Carcharhinus galapagensis	0.07	1.36	0.83	18.12	RLS>JCU ($p<0.001$)	RLS>JCU ($p<0.001$)
Centropyge bispinosa	1.00	0.60		0.02	Comparable	
Centropyge tibicen	1.33	1.20		0.05	Comparable	
Chaetodon auriga	1.17	1.37		0.15	Comparable	
Chaetodon citrinellus	1.25	1.11		0.09	Comparable	
Chaetodon ephippium		0.60		0.09		
Chaetodon flavirostris	0.61	1.32		0.27	Comparable	
Chaetodon lineolatus	0.67	1.52		0.10	Comparable	
Chaetodon lunulatus		1.30		0.14		
Chaetodon melannotus	0.50	0.40		0.07	Comparable	
Chaetodon mertensii	0.73	0.80		0.06	Comparable	
Chaetodon pelewensis	0.67	0.87		0.07	Comparable	
Chaetodon plebeius	0.83	0.55		0.03	Comparable	
Chaetodon speculum	0.67					
Chaetodon tricinctus	1.42	1.18		0.17	Comparable	
Chaetodon trifascialis	1.50	0.86		0.09	Comparable	
Chaetodon ulietensis		1.00		0.12		
Chaetodon unimaculatus		0.80		0.08		
Chaetodon vagabundus	0.67	0.80		0.12	Comparable	
Chaetodontoplus conspicillatus		0.40		0.07		
Chlorurus microrhinos	2.00	1.14	1.86	1.30	Comparable	Comparable
Chlorurus spilurus	6.22	13.77	0.69	2.15	Comparable	RLS>JCU ($p<0.05$)
Chromis flavomaculata		13.40		0.14		
Chromis hypsilepis	71.57	130.62		4.01	Comparable	
Chromis margaritifer	5.00	2.96		0.02	Comparable	

Species	N. per 100m^2 JCU	N. per 100m^2 RLS	Kg per 100m^2 JCU	Kg per 100m^2 RLS	**Density** Comparison	Biomass Comparison
Chromis vanderbilti	8.00	24.40		0.09	Comparable	
Chromis viridis		52.00		0.97		
Chrysiptera notialis	6.64	10.00		0.08	Comparable	
Coris bulbifrons	0.63	1.24		0.98	RLS>JCU (p<0.01)	
Ctenochaetus striatus	1.60	0.40	0.20	0.15	Comparable	Comparable
Dascyllus aruanus	68.00	21.20		0.22	Comparable	
Dascyllus reticulatus	3.00	0.67		0.02	Comparable	
Dascyllus trimaculatus		3.40		0.14		
Epinephelus daemelii	0.02		0.08			
Forcipiger flavissimus	0.83	0.80		0.05	Comparable	
Genicanthus semicinctus		0.80		0.20		
Girella cyanea		1.27		1.00		
Kyphosus bigibbus		0.60		0.76		
Kyphosus cinerascens		0.80		0.33		
Kyphosus sectatrix	12.95	54.48	7.01	26.47	Comparable	Comparable
Kyphosus sydneyanus	1.00					
Naso maculatus	2.00		2.17			
Naso unicornis	1.03	0.57	0.85	0.37	Comparable	Comparable
Neoglyphidodon polyacanthus	9.20	2.88		0.09	Comparable	
Parma polylepis	2.33	1.00		0.25	JCU>RLS (p<0.01)	
Plectroglyphidodon dickii	6.00	1.72		0.05	Comparable	
Plectroglyphidodon johnstonianus	7.80	1.40		0.04	Comparable	
Plectroglyphidodon lacrymatus		1.60		0.03		
Prionurus maculatus	4.00	7.96	2.03	8.02	Comparable	Comparable
Scarus altipinnis	1.30	1.09	1.18	1.02	Comparable	Comparable
Scarus chameleon	1.47	0.80	0.05	0.37	Comparable	Comparable
Scarus frenatus	1.02	1.00	0.25	1.11	Comparable	Comparable
Scarus ghobban	0.80	4.60	0.74	0.54	RLS>JCU (p<0.05)	Comparable
Scarus globiceps	4.00		0.06			
Scarus niger	2.80		0.06			
Scarus oviceps	1.00		0.12			
Scarus psittacus	4.90	4.27	0.15	0.66	Comparable	Comparable
Scarus schlegeli	3.27	0.90	0.35	0.66	Comparable	Comparable
Stegastes apicalis	2.00	0.40		0.01	Comparable	

Species	N. per 100m^2 JCU	N. per 100m^2 RLS	Kg per 100m^2 JCU	Kg per 100m^2 RLS	**Density** Comparison	Biomass Comparison
Stegastes fasciolatus	10.25	9.54		0.62	Comparable	
Stegastes gascoynei	20.92	19.44		0.85	Comparable	
Zanclus cornutus	0.67	0.67		0.13	Comparable	
Zebrasoma scopas	0.60	1.60	0.08	0.22	Comparable	Comparable
Zebrasoma veliferum		0.53		0.19		

Sampling by the two teams resulted in significant differences at the community level, for benthic and fish survey data. JCU survey results included a greater proportion of *Acropora* and *Pocillopora* spp., green and red macroalgae; RLS recorded more sponges, soft corals, massive, foliose and encrusting corals and brown macroalgae (Figure 3). There was some overlap, however; one JCU site was grouped with RLS sites that had high cover of turf, *Halimeda* and sand. There was more overlap between the two teams' fish surveys than for the benthic community, but with JCU data tending to fall towards the left of the plot (Figure 4).

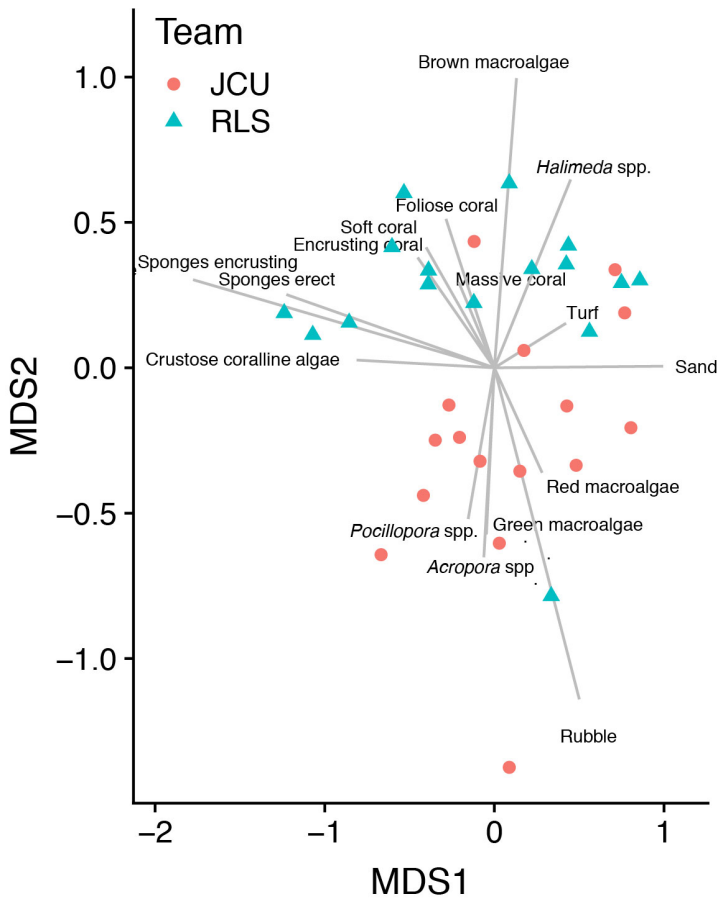

Figure 3. Non-metric MDS of benthic categories recorded by both teams at the EMR on the same transects. Permanova Team MS = 11703, Pseudo-F = 5.61, p = 0.008; Site (Team) MS = 3105.3, F = 5.53, p < 0.001.

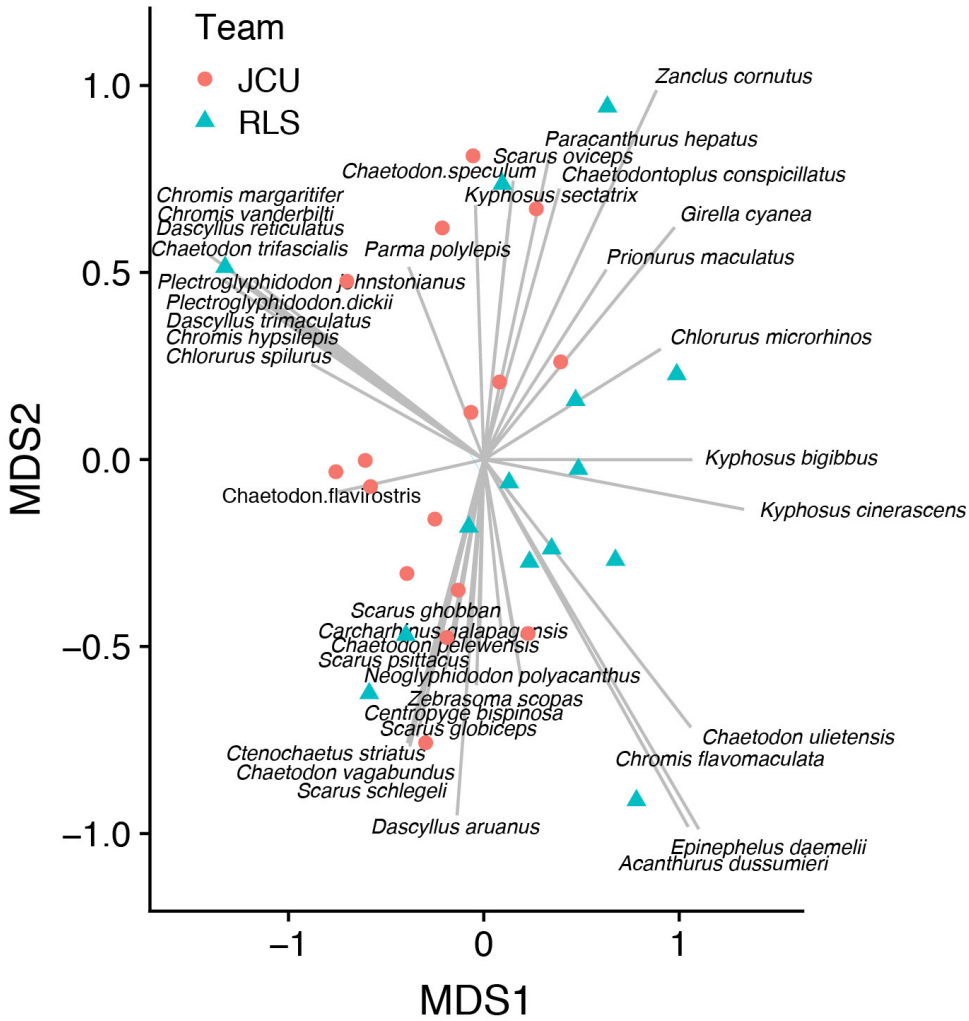

Figure 4. Non-metric MDS of fish species recorded at the EMR by the two teams. Vectors are shown for variables with a correlation of at least 0.5. Permanova Team MS = 24213, Pseudo-F = 4.57, p < 0.001; Site (Team) MS = 7254, F = 2.01, p < 0.001.

The following indicators were tested at the transect scale to assess the compatibility of the two methods: live coral (% cover), turf (% cover), macroalgae (% cover), grazing fishes (individuals 100 m^{-2}) and fish species richness (using only the suite of species surveyed by both teams). Live coral and turf cover estimates from both methods were positively correlated ($R^2 = 0.59$, $p < 0.001$ and $R^2 = 0.36$, $p < 0.05$) (Figure 5). Macroalgae cover, grazer density and fish species richness showed no correlation (Figure 5, Figure 6).

Larger-scale spatial patterns were also assessed using only the subsets of surveys by each team that consisted of the matched transects to calculate the mean values for each reef. Differences between Elizabeth and Middleton Reefs in hard coral, macroalgae, turf and soft coral cover were largely consistent between teams, with higher coral cover at Elizabeth Reef and higher macroalgae cover at Middleton Reef (Figure 7). RLS recorded higher biomass and species richness of fishes than JCU at both reefs; grazer biomass was higher for the RLS team only at Middleton Reef. Overall fish density was similar between teams, except for Elizabeth Reef, where RLS recorded very high densities of reef fishes, but only at one site, resulting in high variability (Figure 8).

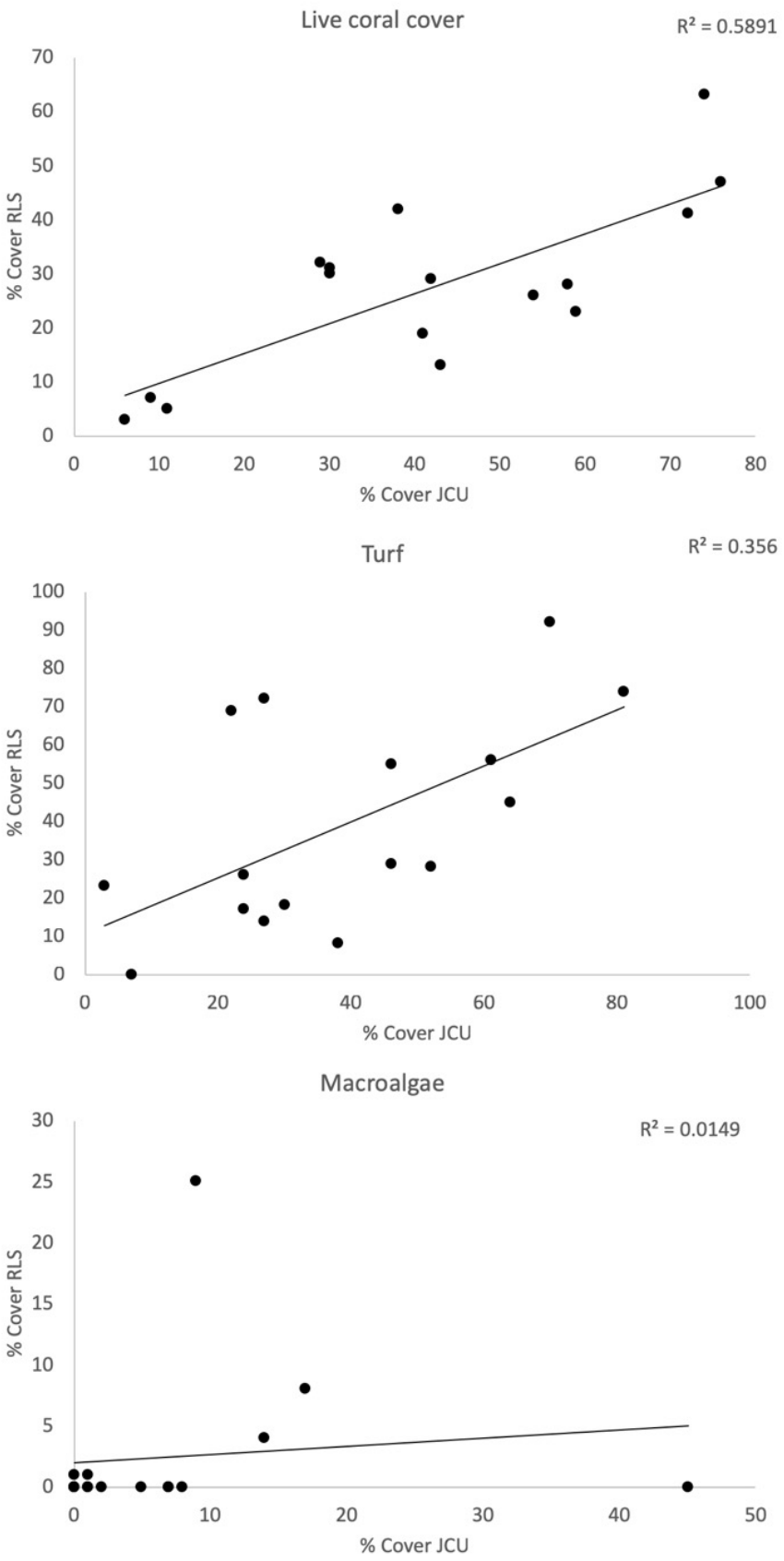

Figure 5. Coral, turf and macroalgae cover recorded on the same transects by JCU and RLS teams.

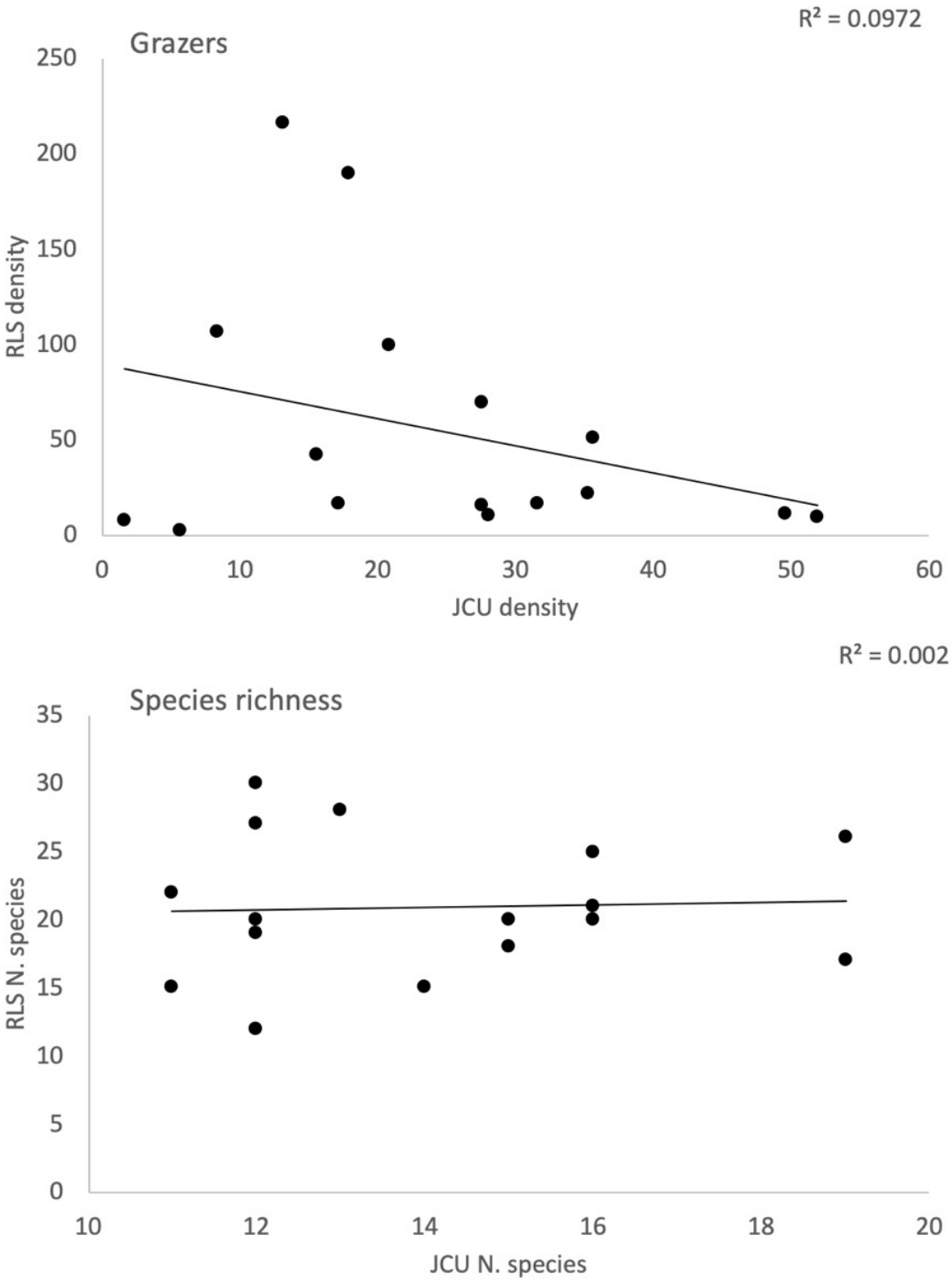

Figure 6. Grazer density (standardised abundance per 100 m^2) and fish species richness recorded on matched transects by JCU and RLS. As it was transects that were matched and not block, the two RLS block estimates were averaged for each transect.

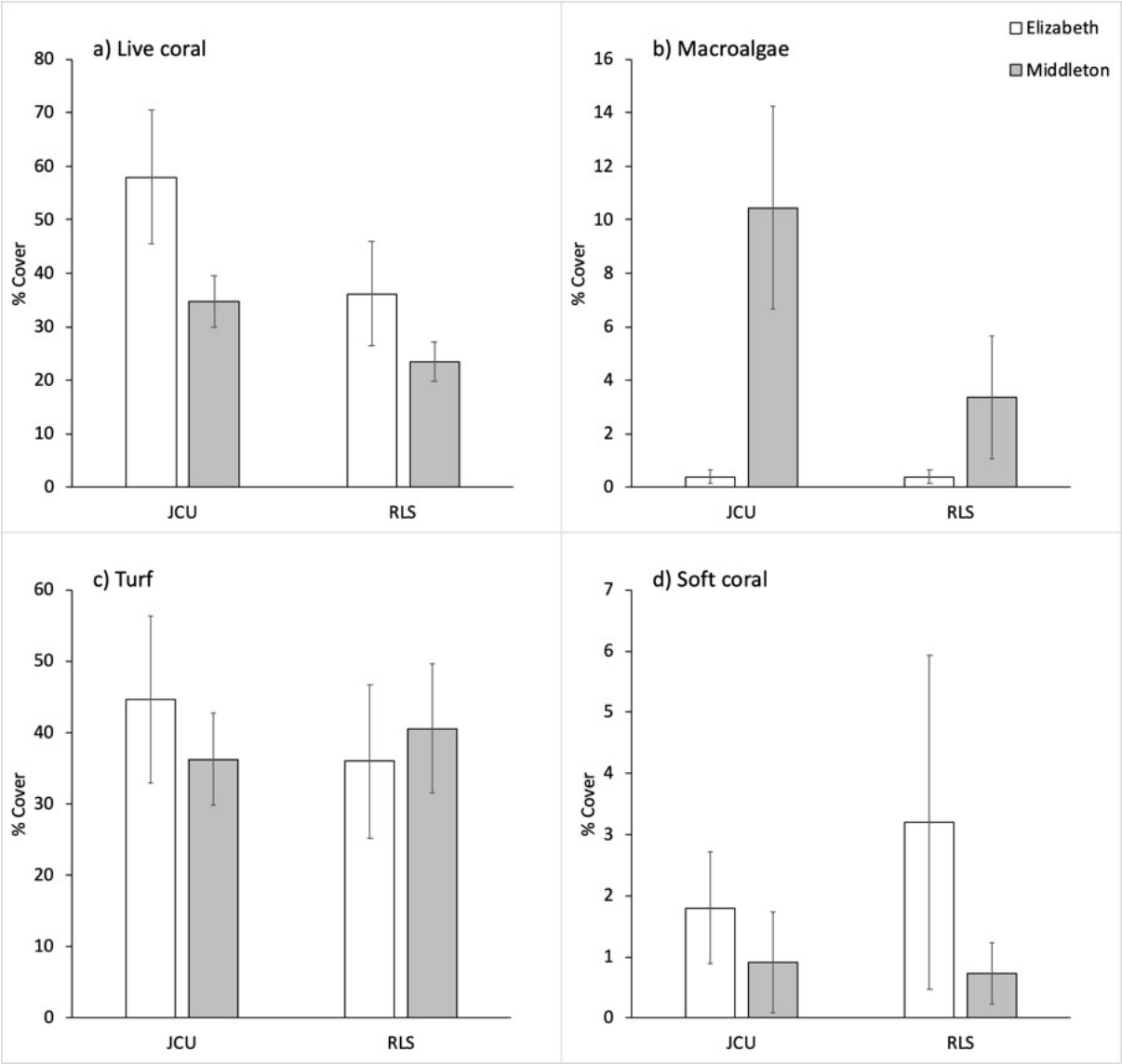

Figure 7. Between-reef variability in a) coral cover, b) macroalgae cover, c) turf cover and d) soft coral cover recorded by JCU and RLS at Elizabeth Reef and Middleton Reef in 2018. Means (+/- 1 SE) presented here were calculated only from 16 matched transects surveyed by both teams.

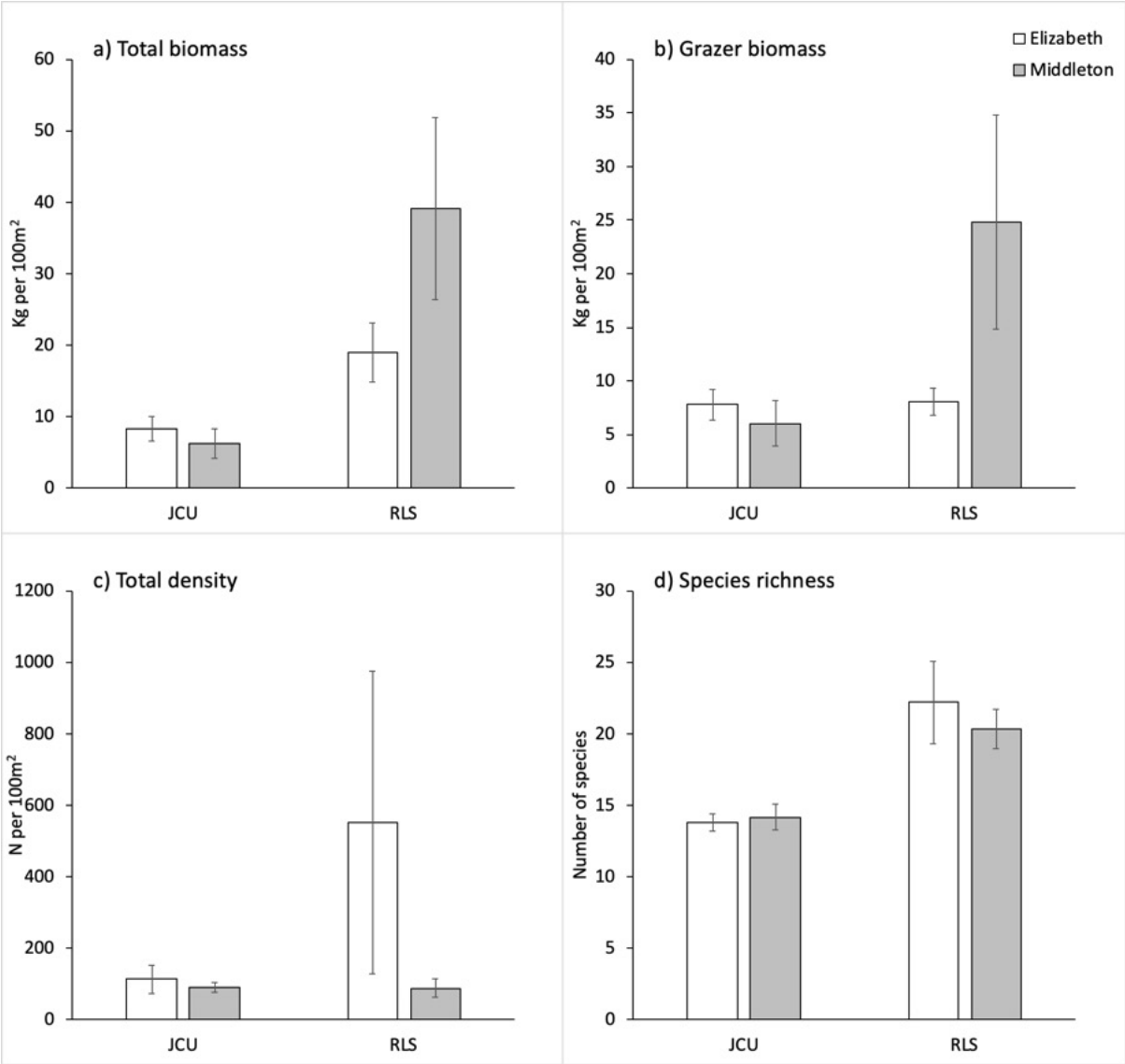

Figure 8. Between-reef variability in a) total fish biomass, b) grazer biomass, c) total fish density and d) species richness of selected fish species recorded by both JCU and RLS at Elizabeth Reef and Middleton Reef in 2018. Means (+/- 1 SE) presented here were calculated only from 16 matched transects surveyed by both teams.

Differences in transect dimensions (for fishes) and survey methods (for benthos) produced different results for the two teams. The two teams also set out to survey the EMR with a focus on different scales. Also, JCU traded off some detail (e.g., not trying to be comprehensive in counting all reef fishes, but with a focus on specific groups that are ecologically or economically important) to cover a greater area within each site, while considering within-site variability. Taxa were specifically chosen as representatives of a functioning coral reef community. In contrast, RLS surveys were designed to be taxonomically comprehensive, by including indicator groups, but also conspicuous reef fishes, cryptic fishes and macroinvertebrates, which is rare among reef surveys (Brock 1982). They trade off replication within the site level to survey more sites, allowing region-wide comparisons of synoptic change.

The method for setting transects with a small gap between along a depth contour (Hodgson 1999), rather than setting transects continuously end-on-end, increases distance covered by surveys (and representation of habitats, and patchily distributed organisms) and allows the calculation of a mean value with a variance measure at the level of sites. It is debatable whether transects surveyed within the same depth strata (as per the JCU method) are sufficiently statistically independent to allow estimation of variance at the level of individual sites (independent of depth) which can be statistically compared to estimates of variation between depths, among sites, or through time. But regardless of this, coverage of a greater total area by more transects within sites by the JCU approach (i.e. 200 m x 5 m at each of two depths for JCU = 2,000 m^2 *cf*. an average of 50 m x 10 m at each of two depths for RLS = 1,000 m^2), improves the accuracy of estimates of the typical densities of species in the groups of fishes that are surveyed at that site. Thus, a key point of difference between the JCU and RLS approaches at the site level is the greater coverage of microhabitat and small-scale patchiness within sites by the JCU approach, thus facilitating a more accurate picture of the reef community at this scale.

Beyond the dimensions and positioning of the sampling units or replicates, the method used by surveyors in the water can also affect results. Recording a subset of taxa, especially for reef fishes, can reduce biases in an individual observer's data (Watson et al. 1995), but can impose restrictions on some of the indicators possible (see Table 1). It may potentially also miss 'ecological surprises', such as arrival at a site of a new species that subsequently becomes dominant. Laying the transect tape before conducting the fish count, as in the RLS method, can also affect the species composition in the data as fish that are easily spooked can take up to 17 hours to return to the area, and will therefore be recorded less frequently in the dataset (Emslie et al. 2018). For higher density estimates of the suite of timid reef fish species (and thus less variability between counts and greater power), it is advantageous for the observer to deploy the tape while counting the fishes, for another diver to deploy the tape behind the fish surveyor, or to use a timed swim (Emslie et al. 2018).

The ability to detect temporal changes requires repeated surveys at the same location, using the same methods. To maximise detection of temporal trends when few sites are surveyed, it would be preferable to use permanently marked transects or plots (thereby reducing variability due to inherent patchiness at the level of individual sites); but this requires considerable dive time to set and relocate markers, and subsequent findings cannot be extrapolated beyond the specific linear areas sampled. Most coral reef monitoring programs use fixed GPS locations to locate specific sites and lay their replicates haphazardly at a specified depth or habitat. This allows comparisons at the level of depths and habitat within sites rather than specific fixed sampling units.

Comparisons of variance components for live coral cover in our study region showed that equal or greater variation occurs among transects within a site than between sites (especially when accounting for aspect). This highlights the inherent patchiness in benthic assemblages and no doubt explains many of the differences in the comparisons of benthic communities estimated by JCU and RLS approaches along the same transects as in this report. Increasing the area surveyed by using multiple transects assists in 'smoothing' out this finer scale variability, and may provide a statistical opportunity to partition variability within sites. In terms of assessing temporal trends, the larger area surveyed within each site (JCU) therefore adds greater confidence to the results at the site level. In contrast, when surveying only 2 x 50 m transects at individual sites, assessment of change through time using the RLS approach will be less accurate for

particular sites with high small-scale patchiness. Thus, accuracy of trends through time will vary considerably between sites as a result of differences in patchiness. On the other hand, the increased number of sites possible to survey at each reef using the RLS approach due to less overall diver time at each site (2 divers rather than 4 divers operating), results in a greater characterisation of between-site patchiness.

The power in temporal comparison thus differs between JCU and RLS approaches; JCU approach more accurately characterises change at individual sites, with a comparatively greater risk of changes on unsurveyed parts of the reef remaining unnoticed, while the RLS approach relies on more sites to allow a temporal comparison of overall reef condition at one time point to overall reef condition at the next time point. If insufficient sites are surveyed to adequately characterise the general reef condition, there is a comparatively greater risk of not picking up change between the two periods. Both JCU and RLS methods can scale up to cover the relative weaknesses of each approach through greater survey effort and therefore cost, with JCU expanding the number of sites covered to reduce the risk of missing important change at unsurveyed sites, and RLS surveying more transects at each site (or more points in time) to reduce the risk of inadequately detecting change.

9.1.1 Potential for integration

When each team used their own methods, the overall conclusions at the reef scale were generally similar. When assessing patterns on individual matched transects, community level patterns and some indicators differed considerably, despite a general agreement at the species level. This variability at the transect scale could be either due to inherent patchiness in the communities or differences between the methods. The key to the differences is the scale at which each team seeks the greatest accuracy; there is a trade-off between replication at the reef level and at the site level. Values that vary most at the fine scale (e.g. black cod, sharks, rare species of corals) are likely to be better captured by the JCU approach. Values which would be better characterised by conducting surveys at more sites and a synoptic approach (e.g. overall fish community composition, average live coral cover) are likely to be better captured by RLS. Some differences at the transect scale also undoubtedly relate to different areas of reef surveyed, with the RLS footprint extending 5 m each side of the transect line (one block on either side) for fishes and the JCU footprint extending 2.5 m each side. Some stochasticity was also added by large passing schools that had moved on (or not arrived) when the second team was undertaking censuses. For example, the large difference between methods in counts of convict tang (*Acanthurus triostegus*) likely relates to a single large mobile school of this species.

The different methods used by the two teams are not directly compatible without calibration. Integrated data analysis is possible, but with multiple caveats. The two methods can, however produce many of the same indicators, which provide broadly comparable information at scales larger than individual transects. Ultimately, more data collected at each reef, even by different teams, provides a basis for better-informed management decisions and, if teams were to survey the same location at different times, there is potential for more frequent updates on the condition of the reef. This can translate to more rapid responses should the changes detected require shifts in management.

The most effective way to design the ideal survey protocol or monitoring program is for the questions that need answering to be clearly framed (Houk and van Woekik 2013). For ongoing monitoring of Elizabeth and Middleton Reefs, some specific management objectives relating to particular values may be better answered by one method or another, whether due to more accurate coverage of particular taxa or indicators, or whether due to the patchiness in the distribution of the value being greater at small scales (e.g. within sites) or large scales (e.g. between sites). Providing specific recommendation on methods requires knowledge of these specific values and management objectives. Some examples may be tracking populations of Galapagos sharks and black cod versus overall assessment of management zone performance for biodiversity protection. Greater emphasis on more area and specific methods for large fishes within sites would make JCU methods preferable, while greater coverage of reef sites, habitats and species would make RLS methods preferable.

9.1.2 Recommendations

Recommendations for future shallow-reef monitoring at Elizabeth and Middleton Reefs are as follows:

- Working from clearly defined research questions and associated data needs, identify the most appropriate monitoring protocols. Clearly framed questions will result in methodologies designed to provide specific answers.

- Undertake recurrent monitoring to test for temporal trends in the health and condition of the shallow coral reef habitats and organisms, ideally on an annual basis to allow tracking of changing conditions associated with climate fluctuations. However, given the high cost of annual surveys, we recommend recurrent ecological monitoring at least every 2-4 years, a period that matches data needs for State of the Environment reporting. Reactive surveys should additionally be undertaken in the aftermath of known disturbances (e.g., following anomalously high temperatures that could result in significant bleaching, or ship groundings). Future sampling programs should also recognise the information content of prior surveys and, wherever possible, use comparable methods to allow for accurate temporal comparisons.

- Explore the possibility of adding additional surveys to assess the population status of priority marine species (e.g. threatened, keystone, apex predators, locally/regionally endemic).

- Explore the possibility of monitoring trends in key pressures and threats (e.g. accurately logging ship visits, temperature variation between habitats and depths, fishing pressure).

10 Conclusions

EMR surveys by JCU and RLS dive teams differ in multiple ways that primarily relate to the scale at which reef communities are characterised. Comparative analyses indicate that the two approaches overlap with some differences at the transect scale, while generating largely similar conclusions at the reef scale. The largest differences were evident in turf cover at all scales, probably due to different ways that 'turf' is defined using the RLS photo-quadrat and JCU line intercept methods.

Overall, JCU and RLS monitoring protocols were found to possess complementary strengths and weaknesses, generating different sets of benefits that vary with specific research priorities and management needs. Question-driven research allows the pros and cons of the methods used by the two teams to be balanced appropriately, with the ultimate aim of reducing uncertainties in adaptive management.

11 Acknowledgements

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