CORAL REEF HEALTH IN THE CORAL SEA MARINE PARK
Report on reef surveys
April 2018 to March 2020

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In responding to a tender from Parks Australia, a team of researchers representing the ARC Centre of Excellence for Coral Reef Studies at James Cook University (JCU) completed surveys of 20 reefs in the Coral Sea Marine Park.

On the cover – Extensive bleaching of corals across shallow habitat on Holmes Reefs, central Coral Sea. Photograph taken by Dani Ceccarelli

We acknowledge the traditional owners of the sea country in which this research and monitoring was conducted, and in particular the Meriam people, and pay our respects to their elders, past, present and emerging.

Two traditional owners of the Meriam people joined our team during surveys of Ashmore and Boot Reefs in October 2018, and can be seen here snorkelling over Ashmore Reef.
Image credit: Martin Russell

Acknowledgements

1   *Executive Summary*

The Coral Sea is a critically important and significant ecosystem, which (like coral reefs globally) is increasingly threatened by changing environmental conditions, particularly ocean warming. James Cook University was commissioned by Parks Australia to undertake a multi-year research and monitoring project (2018-2020) to understand the health of coral reefs in the Coral Sea Marine Park (CSMP). The project undertook detailed annual surveys of coral bleaching, fish and invertebrates and associated reef health. In the absence of bleaching, surveys were intended to enable comparisons against historical baselines where bleaching has previously occurred (e.g., 2016 and 2017), and/or set new baselines for areas that may bleach in the future. Surveys were to provide rigorous quantitative information on spatial (i.e., among reefs and regions) and temporal patterns in (i) cover and composition of corals and macroalgae; (ii) structural complexity of reef habitats; (iii) regional patterns of biodiversity; (iv) coral health and injury; (v) coral recruitment; and (vi) abundance and composition of reef fish and ecologically or economically important invertebrates.

The project surveyed 120 sites and over 64 km of reef habitat across 20 reefs in the CSMP, spanning 13 degrees of latitude (~1,800 km) from Boot Reef in the north (9.9ºS) to Cato Reef in the south (23.2ºS). This represents the most extensive assessment of coral reef health and marine biodiversity ever undertaken in the CSMP. Importantly, these surveys included several reefs for which there had been very limited prior research or monitoring, namely Ashmore and Boot Reefs in the north and Mellish Reef in the east of the CSMP. These highly isolated reefs were notable standouts in terms of reef health, supporting a greater diversity of coral and reef fish species, higher coral cover, and/or higher densities and biomass of sharks and reef fish compared to other reefs throughout the CSMP.

Overall, the surveys revealed that reefs within the CSMP support unique coral and reef fish communities that are distinct from those of the adjacent Great Barrier Reef Marine Park (GBRMP), and show some affinities to reefs in the Tasman Sea to the south, and nations on the eastern border of the CSMP (New Caledonia, Vanuatu and the Solomon Islands). Many fish and coral species that are abundant within the CSMP are absent or rare in the GBRMP. There was clear structuring of biological

communities within the CSMP, with northern (Boot, Ashmore, Osprey and Bougainville Reefs), central (Diane Bank to Marion Reef) and southern (Frederick to Cato Reefs) reefs supporting distinct coral and fish communities.

We recorded a total of 259 species of coral and 621 species of reef-associated fishes (461 species of conspicuous (i.e., non-cryptic) reef fishes and 160 species of cryptobenthic fishes) across the CSMP, representing substantial increases from previous assessments. These increases likely reflect the extensive spatial coverage of these assessments and the dedicated sampling of corals and cryptobenthic fishes, rather than temporal changes in the number of species within the CSMP. Importantly, our detailed taxonomic assessments identified 11 coral species and 6 fish species that are likely to be new to science. Many more species are likely to represent new records for Australia, and range extensions within Australian waters.

The general condition of shallow coral reef environments in the CSMP varied considerably among regions, with differences in coral cover and coral species composition likely reflecting the impacts of recent major disturbances, namely coral bleaching events and severe tropical cyclones. Importantly, there was a marked shift in the composition of coral assemblages following the 2016 and 2017 bleaching events, with the cover of thermally sensitive tabular and staghorn *Acropora* decreasing markedly on central, and to a lesser extent southern, CSMP reefs between 2016-17 and 2018-2020. These changes in coral composition were accompanied by localised declines of up to 50% in coral cover at some sites in the central CSMP following the 2016/17 bleaching events. However, there was considerable variation among sites with coral cover increasing at some sites over the same period. Despite these recent changes, coral cover was the highest recorded on three central CSMP reefs (Chilcott Islet, Herald Cays and Lihou Reef) across six surveys since the mid-1980's.

Most coral colonies surveyed across the CSMP in 2018 and 2019 were healthy, and coral assemblages were showing signs of recovery from the 2016 and 2017 bleaching events. In 2020, however, there was extensive and severe coral bleaching across the 16 CSMP reefs surveyed due to high sea surface temperatures. Overall, 63% of coral colonies surveyed across the 16 CSMP reefs showed signs of heat stress (i.e., were bleached to varying degrees, including a small proportion of colonies that were recently dead). The extent of bleaching varied among regions

(from 40% in the southern CSMP to 70-72% in the central and northern CSMP) and among reefs (from 23% at Cato Reef to 89% at Willis Islets). This high incidence of bleaching occurred against a shifted baseline of coral assemblages (due to reductions in bleaching sensitive coral taxa following the 2016/17 bleaching event), indicating the 2020 bleaching event was more severe and widespread than the 2016/17 events. The impact of this bleaching event on coral assemblages is currently unknown, but given the extent and severity of bleaching recorded it is likely to result in considerable mortality of shallow water corals. Repeat surveys before April 2021 are essential to assess the effects of this bleaching event on the cover and composition of coral communities within the CSMP.

The surveys revealed:

- The 2016 and 2017 bleaching events have led to a marked shift in coral species composition on reefs in the central CSMP. There were also reductions in coral cover of up to 50% at some sites, but the response was highly variable among sites with coral cover at some sites increasing.

- Disturbances from severe tropical cyclones and elevated sea surface temperatures appear to be major drivers of coral cover within the CSMP. Average cover of hard corals recorded across the CSMP in 2018-2020 was 24.8% (±0.5 SE). Coral cover was notably lower in the central CSMP (21.7%), compared to the northern (30.6%) and southern CSMP (27.0%), and broadly corresponded with our estimates of the frequency and intensity of damaging waves. Moreover, coral cover on reefs in the northern and central CSMP was generally greater or equal on the reef slope (8-10m depth) than the reef crest (1-3m); indicative of recent disturbances (most likely temperature-induced bleaching) that disproportionately impacted shallow habitats.

- The intensity and occurrence of marine heatwaves has increased steadily in all sectors of the CSMP over the past four decades and are expected to increase further under current predictions.

- There were marked regional differences in the composition, abundance and diversity of hard (Scleractinian) corals and reef fishes across the CSMP (Figure 1).

- A total of 259 species of coral and 621 species of reef-associated fishes across the CSMP, with the diversity of both corals and fish being greatest in the northern CSMP and decreased with increasing latitude.
- 11 coral species and 6 fish species that are likely to be new to science.
- Many fish and coral species that are abundant within the CSMP are absent or rare in the GBRMP.
- There was a general latitudinal trend in the abundance and biomass of coral reef fishes, whereby reefs in the northern CSMP had higher abundance and biomass of fishes than reefs in the central and southern CSMP (Figure 1).
- The biomass of reef fishes (a key indicator of reef health, together with coral cover) on CSMP reefs (500 – 4,000 kg per hectare) was comparable to or greater than that on reefs in the GBRMP and high relative to coral reefs globally. These estimates of reef fish biomass are exceptional given the relatively low levels of coral cover, and seemingly altered composition of coral assemblages on many reefs in the CSMP, and likely reflects their isolation and limited fishing pressure.
- Reef fish assemblages with the CSMP were characterised by high biomass of piscivores (e.g., groupers and snappers), whereas those in the GBRMP had comparatively higher biomass of herbivorous fishes, likely reflecting differences in fishing pressure.
- There were four stand out or 'bright spot' reefs (Ashmore and Boot Reefs in the northern CSMP; and Moore and Mellish Reefs in the central CSMP) that supported substantially greater biodiversity, coral cover, and/or reef fish biomass than other reefs in their respective regions.
- Ashmore and Boot Reefs in the far northern CSMP were not only 'bright spots' in terms of the biological communities they supported, but were unique among CSMP reefs in that they also offered a valuable opportunity to engage with, and work alongside the Meriam people, the traditional owners of this sea country.
- There was a marked delineation in the geographic distribution of sea snakes (mainly, *Aipysurus laevis*), which were abundant on all southern CSMP reefs and Marion Reef (the southernmost reef in the central CSMP), but absent from all other reefs in the central and northern CSMP.

- The abundances of giant clams, trochus, sea cucumbers, and sea urchins were highly variable among reefs within each region of the CSMP, suggesting that local, as opposed to regional, processes are important determinants of their populations. Notably, there was a complete absence of trochus at Ashmore and Boot Reefs, and the densities of sea urchins (*Diadema* spp) and giant clams were 6 to 13-fold greater on Kenn Reef than other CSMP reefs. The causes of these substantial greater densities on Kenn Reef are unknown and warrant further investigation, especially as high densities of sea urchins can destabilise the reef framework and result in net erosion of reef carbonates.

- The density of juvenile corals (an indicator of the recovery potential of coral populations) was generally low (< 2 juveniles per m^2) across the CSMP, and approximately half that recorded on adjacent offshore reefs in the GBRMP. The lower densities of juvenile corals within the CSMP likely reflects the isolated nature of these reefs, and will likely prolong the recovery of coral populations following disturbances.

- Biophysical modelling of larval connectivity and analysis of genetic connectivity for two coral reef fish species indicate that connectivity within the CSMP is low, with many reefs demographically isolated from each other. These results indicate that unlike the large, interconnected network of reefs and inter-reefal habitats of the GBRMP, the CSMP reefs need to be viewed more as isolated and independent reef systems, that depend largely on self-recruitment for replenishment and resilience.

- Preliminary analysis revealed temperature data derived from the Ereef GBR4 model demonstrated a bias toward temperatures that were 1 to 1.5ºC cooler than those recorded by in-water current meters.

- Comparisons of demographic rates of four ecologically distinct fish species revealed limited variation in growth or mortality rates across the CSMP. This is surprising given the ~2ºC difference in water temperature between northern and southern CSMP reefs and suggests that local-scale (e.g., reef or habitat) conditions may be more important in shaping demographic rates than regional-scale variation.

- In addition to the research and monitoring undertaken, several additional projects were leveraged from this collaboration between James Cook University and Parks Australia and capitalised on available space during the voyages. These leveraged projects involved 23 researchers from 7 institutions and represent an in-kind contribution of over 310 person days. Collectively these projects will increase our understanding of regional patterns in biodiversity of a range of reef taxa, how the 3D structure of reef habitats and corals relates to patterns of biodiversity, the spatial distribution of shallow water habitats on CSMP reefs, and the tolerance of CSMP corals to acute thermal stress.
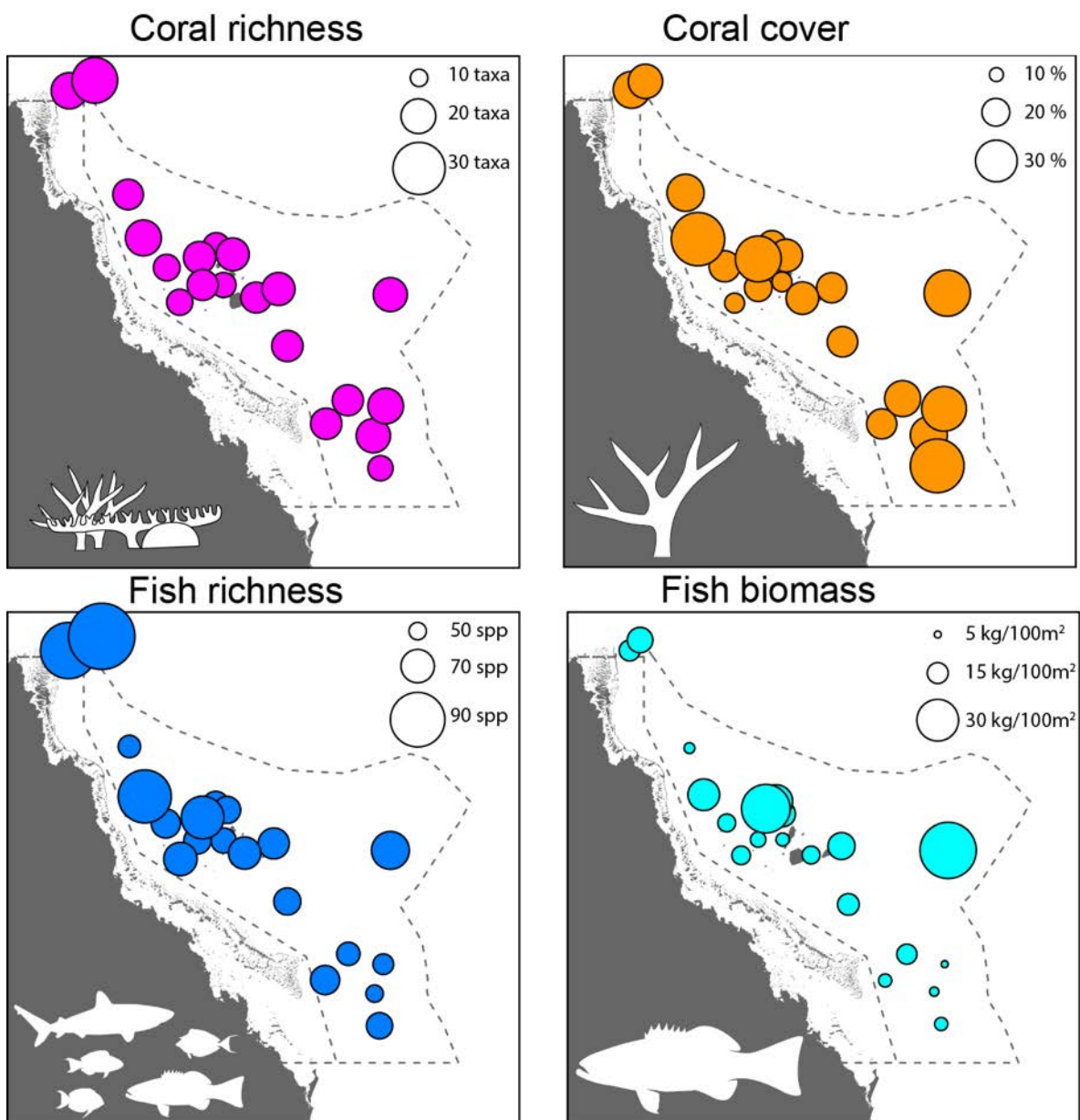
Figure 1. Summary of four key indicators of coral reef health on 20 reefs within the Coral Sea Marine Park: coral diversity, live coral cover, reef fish diversity and reef fish biomass. The size of the symbols in each panel is scaled to the magnitude of that metric on each reef. Values are averaged across habitats and sites on each reef, and based on all surveys conducted during the 3-year project (2018-20).

In conclusion, in 2018-2019 coral reef habitats in the CSMP were showing signs of recovery from recent or sustained disturbances. However, the health of these reefs is likely to change as the outcomes of the extensive and severe bleaching recorded across all CSMP reefs in 2020 are realised. Like in the GBRMP, this is the third major bleaching event in the CSMP in the last 5 years (previous events in 2016 and 2017), and reflective of the increasing frequency and intensity of marine heatwaves (which cause severe and widespread coral bleaching) that are affecting coral reefs globally. Continued surveys of CSMP reefs will be critical to assess the impact of the 2020 bleaching event, and the potential resilience of these isolated reef systems in the absence of other stressors.

Recommendations for future research and monitoring:

- Re-surveying the sites surveyed in 2020 prior to April 2021 is critically important to determine the fate of corals that bleached in 2020, before further disturbances potentially augment patterns of coral abundance.
- Annual monitoring of coral, fish, sea snake and other reef taxa communities on 9-12 reefs (2-4 representative reefs in each of the southern, central and northern CSMP), with all 20 CSMP reefs to be surveyed once every 3-5 years.
- A greater amount of time should be spent at each of the representative reefs (i.e., 3-4 days compared to only 1 day in the present surveys) to allow for surveys of additional habitats and targeted research and monitoring.
- Flexibility should be built into future research and monitoring voyages so that when conditions allow, more wave exposed habitats can be surveyed. This will require the use of timed swims to alleviate the need to deploy and retrieve transect tapes in areas of high and turbulent water flow.
- Comprehensive monitoring of fish and coral assemblages and dedicated research on key demographic and ecological processes at each of the four 'stand out' reefs (i.e., Ashmore, Boot, Mellish and Moore Reefs) should be a priority to understand what makes these reefs unique.

- Regular monitoring (every 1-2 years) of sea urchin and giant clam populations on Kenn Reef to gain a better understanding of the causes for the high densities of these invertebrates at this reef.

- Expansion of connectivity modelling and sampling within all regions bordering the CSMP, with focus towards eastern regions (New Caledonia, Vanuatu, Solomon Islands) to establish their influence on biogeographic patterns in the CSMP.

- A dedicated case study of the degree of isolation and reliance on self-replenishment for fish and coral populations on isolated reefs such as Mellish Reef to establish their susceptibility to disturbances and exploitation.

- Comprehensive and targeted monitoring of giant clam, sea cucumber, trochus and fish assemblages at Boot and Ashmore Reefs should be prioritised every 2-3 years, and conducted in collaboration with the Meriam people over an extended period (6-8 days). This engagement with traditional owners would increase understanding of traditional knowledge and cultural values; establish new, and/or build on existing collaborations between traditional owners, researchers and Parks Australia; and build the capacity of the Meriam people to participate and take an active role in the management of the CSMP.

- Dedicated surveys of deeper reef habitats (30-100m) and soft sediment habitat should be incorporated into future monitoring projects and management planning to better characterise the biodiversity and benthic fish assemblages of CSMP reefs.

- Increased communication and collaboration among government and non-government organisations (e.g., dive tourism and fishing charter operators) should be prioritised to facilitate more extensive sampling and re-visitation across the CSMP using their voyages.

- Comparable research and monitoring in all regions within and bordering the CSMP (i.e., GBRMP, Temperate East Marine Parks Network, New Caledonia, Vanuatu, Solomon Islands and Papua New Guinea) to establish the biogeographical significance and connectivity of the CSMP.

2   Authors

Lead Authors

Professor Andrew Hoey is leading authority on the resilience of reef communities, the functional importance of fishes and invertebrates to ecosystem processes, and the responses of reef communities to human and natural stressors. He has worked on coral reef systems throughout the Indian, Pacific and Atlantic Oceans, and has published >150 scientific papers. Andrew regularly provides scientific advice to government and non-government organizations, at state, national and international levels, on science-based management of marine ecosystems, and has led and/or contributed to the successful completion of several research contracts.

Professor Morgan Pratchett is an internationally acclaimed marine ecologist, who studies responses of coral reef organisms and systems to large-scale disturbances, including mass coral bleaching and outbreaks of crown-of-thorns starfish. He has published >300 scientific papers and has worked throughout the Indo-Pacific, from the Red Sea to Hawaii. As a Research Leader within the ARC Centre of Excellence for Coral Reef Studies, he has led and overseen the successful completion of many research contracts, including recurrent surveys at Elizabeth and Middleton Reefs as well as Ashmore and Cartier National Marine Reserves.

Dr Hugo Harrison is a marine ecologist whose research addresses critical knowledge gaps for the effective management and conservation of marine resources, including seascape connectivity, fisheries science, and ecosystem recovery following large scale disturbances such as coral bleaching. As a Senior Research Fellow within the ARC Centre of Excellence for Coral Reef Studies, Hugo's research aims to identify critical regions that maintain seascape connectivity and identify the adaptive capacity of coral reef ecosystems. Hugo has been leading surveys and expeditions to the CSMP since 2016.

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3   Background

The Coral Sea is a critically important and environmentally significant ecosystem owing to i) the extent and diversity of habitats (including many unique habitats), ii) the unique fauna these habitats support, iii) the provision of habitats for species of conservation significance and, iv) connectivity with Australia's Great Barrier Reef (GBR) and other western Pacific provinces (Ceccarelli et al. 2013; Hoey and Pratchett 2017). These habitats, and the potential connectivity between them, are shaped by both oceanographic and geomorphic features. The major oceanographic features affecting the Coral Sea are west-flowing jets of the Southern Equatorial Current (SEC), which strengthen during the summer months and bifurcate on the Australian continental shelf to form the south-flowing East Australian Current (EAC) and its eddies, and the Hiri Gyre in the Gulf of Papua to the north (Ridgway et al. 2018, Rousselet et al. 2016). Key geomorphic features of the Coral Sea include a deep (4,000m) basin, trenches, two large plateau and seamounts, some of which culminate in emergent reefs and cays (Davies et al.1989, Collot et al. 2011).

The Coral Sea Marine Park (CSMP) is among the largest and most isolated marine parks in the world, encompassing 989,836km^2 and is managed by the Australian Government, Director of National Parks. The CSMP extends from the eastward margin of the Great Barrier Reef Marine Park (GBRMP) to the outer extent of Australia's Exclusive Economic Zone (Figure 3.1). Within the CSMP there are approximately 56 islets and cays and 34 widely separated reef systems with a combined reef area of approximately 15,024 km^2 (DNP 2018).
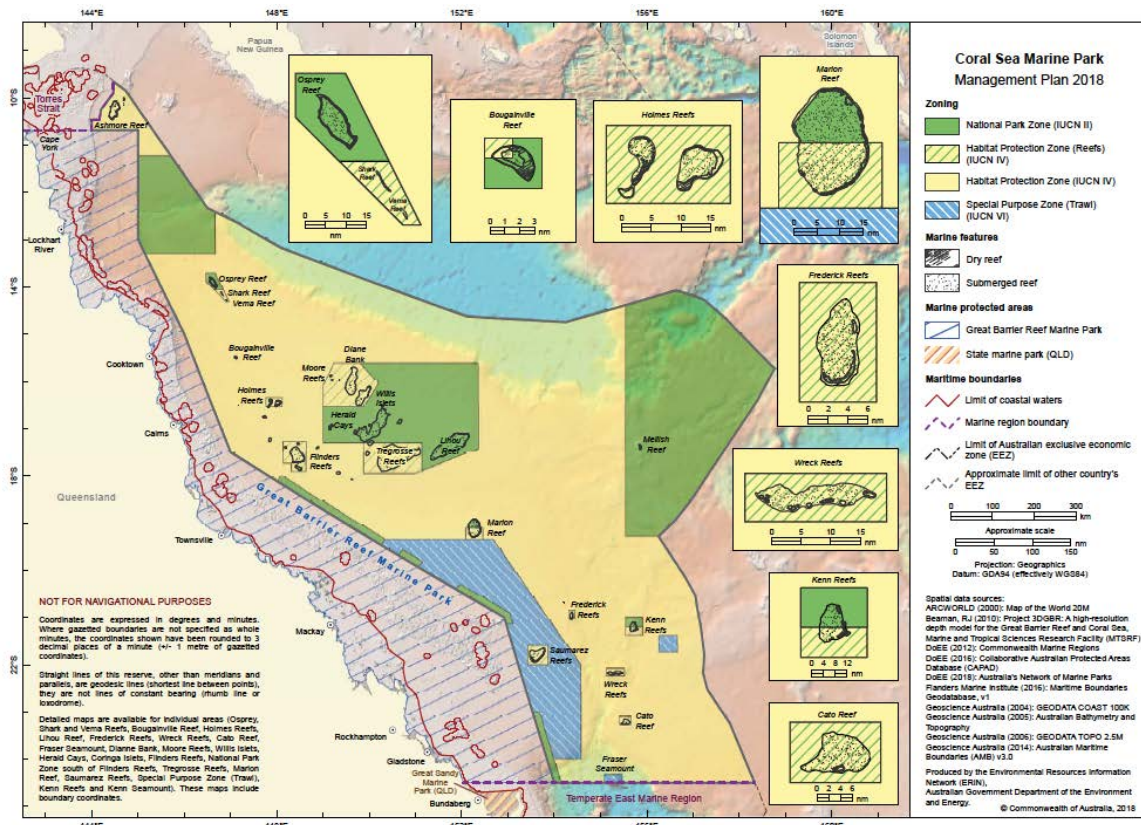
Figure 3.1. Map of the Coral Sea Marine Park, showing management zones implemented in July 2018. (Source: parksaustralia.gov.au)

Previous surveys of shallow coral reef environments within the CSMP have highlighted similarities in species composition and links to the GBRMP and western Pacific Ocean, but with oceanic characteristics and key differences that merit recognition of the Coral Sea as a separate biogeographic province (Ayling and Ayling 1985, Oxley et al. 2004, Veron et al. 2011). More recent surveys (e.g., Stuart-Smith et al. 2013, Harrison et al. 2018) have revealed marked regional differences in coral health and reef condition across the CSMP. Notably, reefs in the southern region of the CSMP (e.g., Wreck and Cato Reef) were reported to have high coral cover and an abundance of fragile branching corals (e.g., *Acropora*) even in the aftermath of coral bleaching that occurred in 2016 (Harrison et al. 2018, 2019). However, coral cover has been reported to be very low across many systems and habitats in the central CSMP (Ayling and Ayling 1985, Oxley et al. 2003, Ceccarelli et al. 2008), and instead were dominated by low-lying algal turfs, and a high cover of *Halimeda* and other calcifying algae (Edgar et al. 2015). In areas where there have been recurrent surveys of benthic communities over several decades (mainly Coringa-Herald Islets and Cays and Lihou Reef), it is

apparent that low coral cover is a persistent feature of these reefs, possibly due to recurrent past disturbances and/or low recovery potential of these reefs (Oxley et al. 2003, 2004, Ceccarelli et al. 2008).

The Coral Sea supports one of the richest faunas of reef fishes anywhere in the world (Ceccarelli et al. 2013), representing a mix of Pacific oceanic and GBR fish faunas. Within the CSMP, Edgar et al. (2015) showed that reef fish diversity was highest in the northern CSMP, in accordance with general latitudinal gradients in reef fish biodiversity (e.g., Bellwood and Hughes 2001) and the proximity of the Coral Triangle biodiversity hotspot (Allen 2008), though coral-dependent fishes (e.g., butterflyfishes) were most abundant on reefs in the southern CSMP, where coral cover was highest. The CSMP also supports relatively high abundance of reef sharks (mainly the grey reef shark, *Carcharhinus amblyrhynchos,* and the silvertip shark, *C. albimarginatus*) compared to other Indo-Pacific localities (Stuart-Smith et al. 2013, Ceccarelli et al. 2013).

3.1   Historical heat stress

Coral reefs globally are increasingly subject to marine heatwaves, which cause mass-bleaching and mass-mortality of scleractinian corals (Heron et al. 2016, Hughes et al. 2017, 2018). The duration and frequency of marine heatwaves have increased globally over the past century with concomitant impacts on biodiversity across a range of ecosystems (Hughes et al. 2018, Oliver et al. 2018, Smale et al. 2019). Prior to this study, four major bleaching events have been reported in the CSMP in 2002 and 2004 (Oxley et al. 2004) and again in 2016 and 2017 (Harrison et al. 2018, 2019). Other bleaching events may have also affected the CSMP but went undetected due to its isolation and infrequent scientific surveys. Furthermore, the intensity of marine heatwaves, measured here as the annual maximum Degree Heating Weeks (DHW, combines both the intensity and duration of heat stress into a single number) averaged across reefs, has increased steadily in all sectors of the CSMP over the past four decades (Figure 3.2). Since 1985, 12-year means of the average maximum DHW have more than doubled in the southern, central and

northern CSMP, and are expected to increase further under current predictions (van Hooindonk et al. 2013).
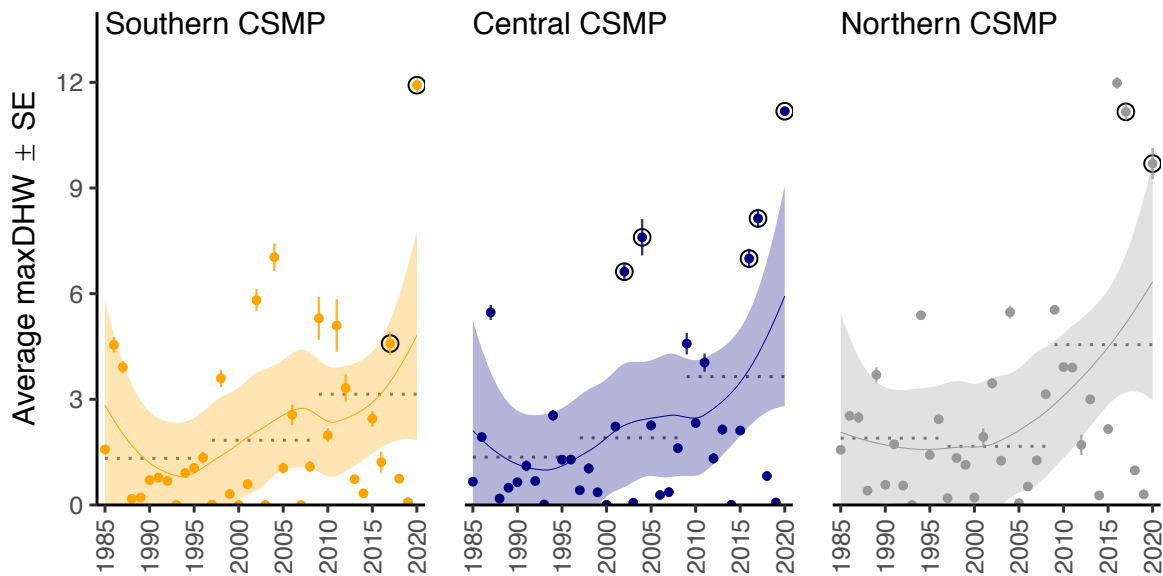
Figure 3.2 Maximum Degree Heating Weeks (DHW) averaged across reefs in the southern, central and northern Coral Sea Marine Park between 1985 and 2020. Documented bleaching events are indicated by larger open circles and 12-years means are represented by dotted lines.

A recent study from the GBRMP suggests that cumulative heat stress above 3 DHW can lead to bleaching of shallow water corals with widespread mortality occurring above 6 DHW (Hughes et al. 2018). Importantly, repeated exposure to damaging marine heatwaves can lead to irreversible changes in coral reef assemblages depending on the intensity (maxDHW) and time between successive thermal stress events (Hughes et al. 2018, 2019). The number and intensity of such events in the CSMP has increased 1.5 to 3.5-fold since the early 1990s (Figure 3.3 a,b), with a concomitant decrease in the return time between events (where DHW> 3) to less than 2 years (Figure 3.3 c). These events have undoubtedly shaped present, and will continue to shape future, coral reef communities in the CSMP.
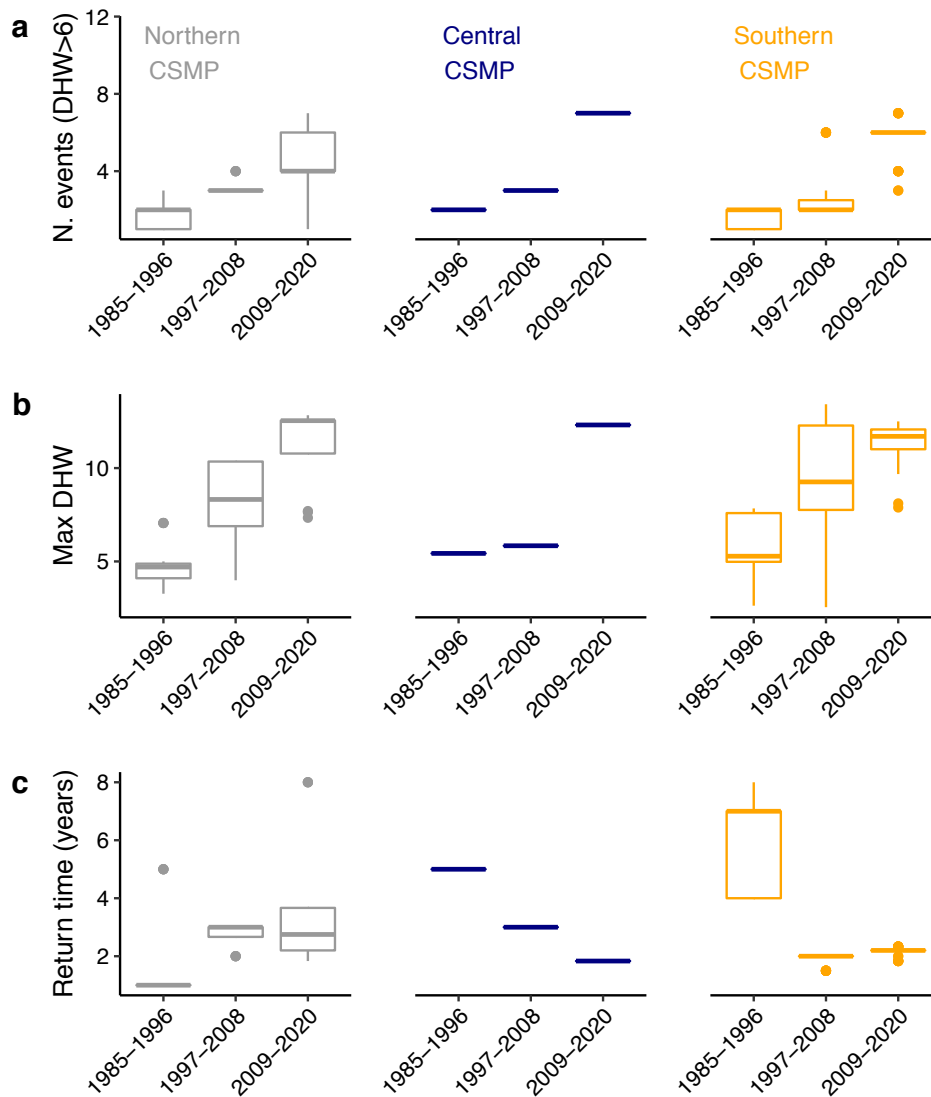
Figure 3.3 The (a) number, (b) intensity and (c) return time of marine heatwaves that are likely to induce coral bleaching (Degree Heating Weeks; DHW > 3) in the Coral Sea Marine Park between 1985 and 2020.

3.2 Objectives and scope

The purpose of this project was to undertake detailed annual *surveys of coral bleaching and associated reef health in the Coral Sea Marine Park.* In the absence of bleaching, surveys are intended to *enable comparisons against historical baselines where bleaching has previously occurred, and/or set new baselines for areas* that may bleach in the future.

Surveys conducted over the 3-years of the project (from April 2018 to March 2020) provide rigorous quantitative information on spatial (i.e., among reefs and regions) and temporal patterns in:

i) benthic cover and composition, including the percentage cover for hard (Scleractinian) and soft (Alcyonarian) corals, macroalgae, and other sessile organisms,

ii) structural complexity of reef habitats,

iii) regional patterns of biodiversity, based on species lists for scleractinian corals and reef fishes,

iv) coral health and injuries caused by coral bleaching, disease, or coral predators (e.g., *Acanthaster* spp and *Drupella* spp)*,*

v) abundance of small/ juvenile corals (<5cm diameter), as a proxy of coral recruitment and population replenishment,

vi) size, abundance and composition of reef fish assemblages, and

vii) abundance of holothurians, urchins and other ecologically or economically important reef-associated invertebrates.

Dedicated sampling was also conducted to address specific research objectives to better understand the key ecological processes that contribute to health of reefs within the CSMP. These were:

i) to investigate fine scale water movement around reefs in the CSMP using water current meters;

ii) to explore connectivity patterns of reefs fishes within the CSMP using genetic sampling of bluespotted coral trout (*Plectropomus laevis*), and a small-bodied pygmy angelfish (*Centropyge bispinosa*);

iii) to investigate the potential effects of previous disturbances (primarily tropical cyclones) on the structure and function of coral reef systems within the CSMP;

iv) to conduct detailed taxonomic assessments of scleractinian corals and reef-associated fishes to better understand regional patterns in biodiversity and identify any previously undescribed species or new species records for the CSMP;

v) to sample fishes (the common coral trout *Plectopomus leopardus,* bluespotted coral trout *Plectropomus laevis,* twospined pygmy angelfish *Centropyge bispinosa,* and the lined bristletooth *Ctenochaetus striatus*) to explore regional differences in demographic rates; and

vi) coring of massive *Porites* corals to explore regional differences in coral growth.

3.3 Leveraged Projects

As well as the projects listed above, several projects were leveraged from this collaboration between James Cook University and Parks Australia and capitalised on available vessel space during the voyages. These leveraged projects include:

i) sampling of cryptobenthic reef fishes to explore regional patterns in biodiversity and community composition;

ii) documenting the diversity of cryptic coral reef invertebrates;

iii) documenting the diversity of shallow water sponges;

iv) 3D modelling of reef slope habitats to explore relationships with the biodiversity and assemblage structure of reef fishes;

v) 3D photogrammetry of coral colonies to investigate variation in structural complexity and the provision of shelter for reef fishes;

vi) mapping of key shallow water reef habitats within the CSMP;

vii) measuring the tolerance of corals to acute thermal stress within the CSMP; and

viii) investigating the reliability of aerial photos for detecting coral bleaching across the CSMP.

Further details of these projects are provided in Appendix 1

Figure 3.4 Team of 13 researchers, 2 Parks Australia staff, and crew on the back deck of the 'Iron Joy' during surveys of the southern Coral Sea Marine Park in February 2020. Image credit: Emma Kennedy

4 Methods

Nine survey voyages were conducted between 12 April 2018 and 22 March 2020. Surveys were generally conducted between mid-February and April of each year to coincide with the likely timing of any coral bleaching due to the accumulation of heat stress over summer. The only exception to this were the surveys of reefs in the northern and southern CSMP in 2018 that were surveyed in October and December, respectively, due to the unavailability of a suitable vessel earlier in the year. Sampling was undertaken at 124 sites across 20 reef systems within the CSMP (Figure 4.1; Appendix 2). The majority of CSMP reefs were surveyed in each year, the only exceptions being Mellish Reef in the far east, and Boot and Ashmore in the far north of the CSMP (Figure 4.1) which were only surveyed in 2018. During our voyage to the far northern CSMP we met and discussed our research with the traditional owners of Ashmore and Boot Reefs (the Meriam people), and invited two traditional owners to join us for our surveys. Where possible, we re-visited sites that were surveyed in 2016-17 by Harrison et al. (2018) so that potential impacts of the 2016/2017 bleaching events could be assessed. An additional 39 sites across 18 outer-shelf reefs of the GBRMP were surveyed using identical methodologies. The surveys of reefs GBRMP reefs were part of, and funded by, other projects but included here for comparative purposes.

9 voyages - 112 days
38 reefs - 163 sites
64.2 km of UVC surveys
>1,500 diver hours

4.1 Sampling design

At each site, surveys were conducted within each of two different habitats, i) the reef crest (approximately 1-3m depth) and ii) the reef slope (9-10m depth, where possible). In shallow reef environments (mainly inside lagoons or in back reef environments), where maximum depths were less than 9m, the reef slope transects were run along the deepest margin of contiguous reef habitats, avoiding extensive areas of sand or rubble. Similarly, it was not always possible to survey the reef crest, due to low tides, limited water depth, and/ or large swells, and in

those cases the reef crest transects were often run just below the outermost edge of the reef crest (2-4m).
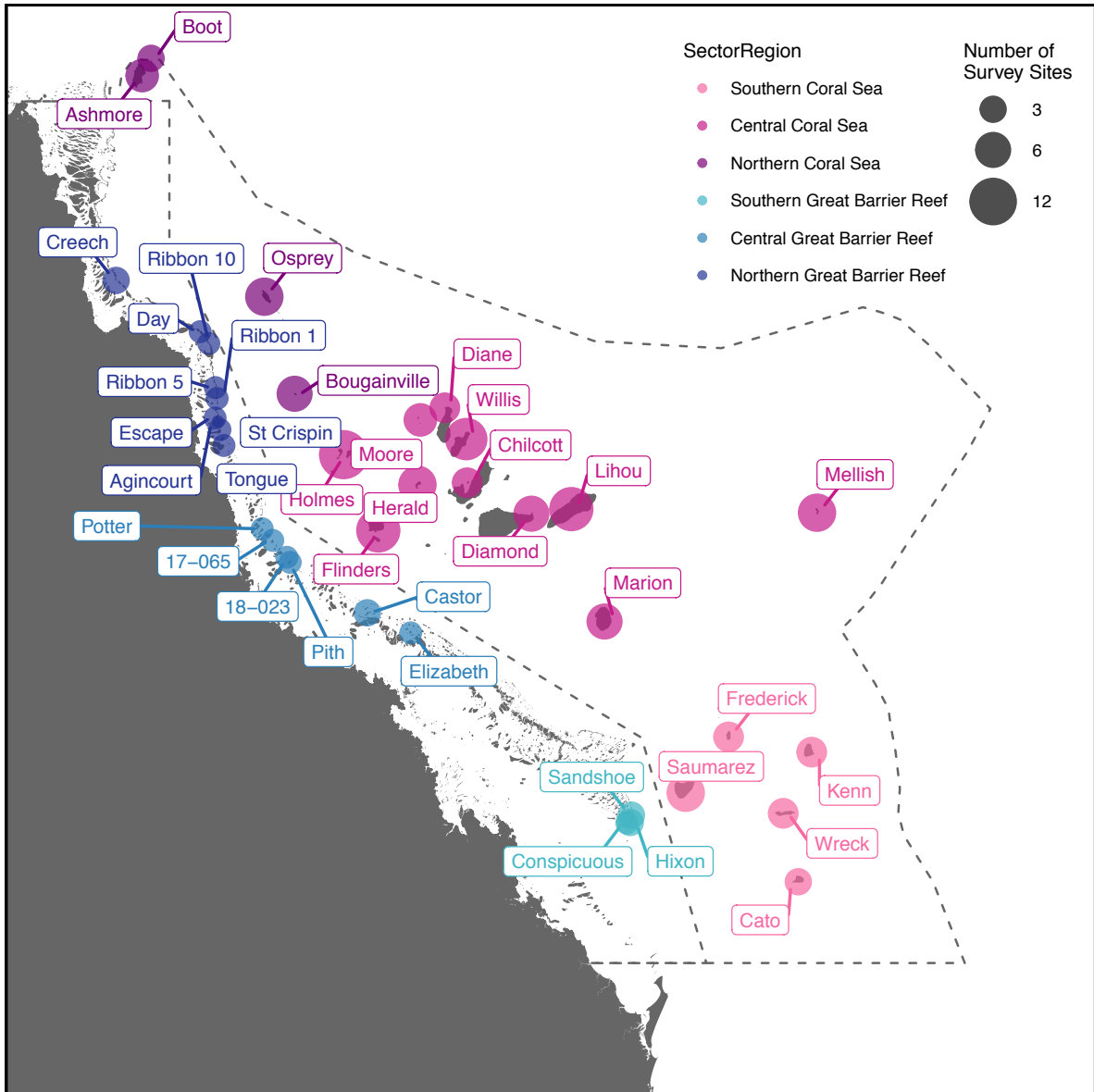
Figure 4.1 Map of the surveyed reefs in the Coral Sea Marine Park and Great Barrier Reef Marine Park. The size of circles indicates the number of independent sites surveyed at each reef during the 3-year (2018-20) project. Colours relate to the regional allocation of reefs in the southern, central, and northern Coral Sea Marine Park and Great Barrier Reef Marine Park, which are used throughout the report. Regional allocation is based on our current understanding of coral and fish communities. Note: reefs within the GBRMP were surveyed as part of, and funded by, other projects but included here for comparative purposes

In each depth zone at each site, three replicate 50m transects were run parallel to the depth contour, with up to 10m between successive transects. Surveys were conducted by a 4-person dive team, whereby the lead diver deployed the transect tape while simultaneously recording all larger (>10 cm total length, TL) or motile fish species, within a 5m wide belt (following Hoey et al. 2014, 2018). Deploying the transect while simultaneously recording fishes minimises disturbance prior to censusing, thereby avoiding any bias due to mobile fishes avoiding (or in some cases being attracted to) divers. The second diver along the transect recorded the size and identity of smaller, site-attached species within a 2m wide belt (e.g. Pomacentridae), while species with larger home ranges were recorded within a 4m wide belt (e.g. Chaetodontidae; Appendix 3). The third diver conducted a point intercept survey, providing important information on coral cover and benthic composition, by recording the sessile organisms or substrate underlying evenly spaced (50cm apart) points along the entire length of the transect. The final (fourth) diver began by measuring coral health and recruitment within a 10m x 1m belt, using a 1m bar to accurately determine the boundaries of the survey area. On the return swim along the transects one diver quantified the abundance of non-coral invertebrates (e.g., sea cucumbers, giant clams, *Tectus* (formerly *Trochus*), and crown-of-thorns starfish) within a 2m wide belt along the full length of each transect.

4.2   Coral and reef habitats

Benthic cover and composition - Point-intercept transects (PIT) were used to quantify benthic composition, recording the specific organisms or substratum types underlying each of 100 uniformly spaced points (50cm apart) along each transect (following Pratchett et al. 2011a). Corals were mostly identified to genus (using contemporary, molecular-based classifications for scleractinian corals), though we pooled data to family for some of the less common genera (e.g., Merulinidae and Lobophyllidae). We also distinguished major growth forms for *Acropora* (tabular, staghorn, and other) and *Porites* (massive versus columnar or branching). Macroalgae were identified to genus. For survey points that did not intersect corals or macroalgae, the underlying substratum was categorised as either sponge, sand/ rubble or carbonate pavement. Further, the proportional cover of crustose coralline

algae (CCA) versus turf algae across all consolidated carbonate substrates (pavement and rubble) was recorded.

Topographic complexity – Topographic complexity was 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.

Figure 4.2 Bougainville Reef, December 2017. The high complexity of coral reefs in the Coral Sea Marine Park support a high density of reef associated fishes. Image credit: Tane Sinclair-Taylor

Coral health – In anticipation of potential coral bleaching, we recorded coral health of all colonies contained within a 10m x 1m belt on each transect (n = 3 per depth zone per site), following protocols developed by the Australian Coral Bleaching Taskforce (Hughes et al. 2017). The 10 x 1 m belt transects were generally run at the start of each 50m transect, but were relocated as required to avoid areas of sand or rubble substrata. For each colony contained wholly or mostly (>50%) within the transect area, we recorded the taxonomic identity, colony size and health. Corals were classified to genera and growth form (as described for PIT above), and then assigned to one of 5 size classes based on their maximum

diameter (<5cm representing juveniles as discussed below, 5-20cm, 20-40cm, 40-60cm and >60cm). The health of each coral colony was then assigned to one of 8 categories (Table 4.1), to document the extent and severity of bleaching, as well as any other recent injuries, such as evidence of recent predation. Where possible, the cause of conspicuous injuries was also recorded, be it due to coral predators (e.g., *Drupella* spp., crown-of-thorns starfish or some parrotfish) observed within or nearby the injured colony, or coral disease.

4.3 Coral reef fishes

Size (body length) and abundance of reef-associated fishes (e.g., Acanthuridae, Chaetodontidae, Labridae, Lethrinidae, Scarinae, Serranidae, and Pomacentridae) was quantified using standard underwater visual census (UVC) along replicate 50m transects (n = 3 per depth zone) at all sites. Various transect dimensions were used to account for differences in the body size, mobility, and detectability of different fishes, as well as making data more comparable to other surveys conducted within the GBRMP (e.g., Emslie et al. 2010) and other Australian Marine Parks (e.g., Hoey et al. 2018). Smaller site-attached species (Pomacentridae and Labridae) were counted in a 2m wide belt ($100m^2$ per transect). Slightly larger bodied, site-attached species (e.g., Chaetodontidae) were surveyed in a 4m wide belt ($200m^2$ per transect), while all larger and more mobile species were counted in a 5m wide belt ($250m^2$ per transect). Body size (total length) was recorded for each individual fish, and converted to biomass using published length-weight relationships for each species. Data were standardised as abundance and biomass per $100m^2$. See Appendix 3 for a comprehensive list of species surveyed.

Table 4.1 Coral health categories distinguishing the condition of individual coral colonies

Coral Health	
H - Healthy (<5% Recent Mortality) 	C - 100% Bleached
P – Pale 	D - 5-50% Recent mortality
A - <50% Bleached 	E - 50-99% Recent Mortality
B - 50-99% Bleached 	F - 100% Recent Mortality

Figure 4.3: Diver surveying reef fish while also deploying transect tape in the shallow reef habitat on Flinders Reefs in March 2020. Image credit: Dani Ceccarelli

4.4   Other reef taxa

Sea snakes - The abundance and size of sea snakes (including Olive sea snakes, *Aipysurus laevis*; Dubois' sea snakes, *Aipysurus duboisii*; Spiny headed or Horned sea snakes, *Acolyptophis peronii*; Turtle-headed sea snakes, *Emydocephalus annulatus*) were quantified in 2019 and 2020 following observations of marked differences in their abundances among reefs. Sea snakes were surveyed within the same 50 x 5m belt transects used to survey large, mobile reef fishes. All sea snakes observed within the transect area were identified to species and their length estimated.

Non-coral invertebrates – Non-coral invertebrates, including potential coral predators (e.g., crown-of-thorns starfish *Acanthaster* cf. *solaris*, pin-cushion starfish *Culcita novaeguineae*, and coral snails *Drupella* spp) as well as ecologically and economically important species, namely long-spined sea urchins (*Diadema* spp) sea cucumbers (holothurians), giant clams (*Tridacna* spp) and trochus (*Tectus* sp, formerly *Trochus* spp), were surveyed in a 2m wide belt along each transect, giving a sample area of 100m^2. For all crown-of-thorns starfish (*Acanthaster* cf. *solaris*) and giant clams (*Tridacna* spp.) observed, the size (diameter and length, respectively) was also recorded (to the nearest 10cm).

Coral predators are potentially important contributors to coral reef health and habitat structure, especially during periods of elevated densities of these coral

predators (Pratchett et al. 2014). Population irruptions of crown-of-thorns starfish (*Acanthaster* cf. *solaris*) are a major contributor to coral loss on the Great Barrier Reef (De'ath et al. 2012) and are thought to have caused considerable coral loss on Elizabeth and Middleton Reefs in the 1980's (Hoey et al. 2018), though it is not known whether there have been outbreaks in the CSMP. Sea urchins, especially long-spined sea urchins of the genus *Diadema*, can also have a major influence on the habitat structure of coral reef environments (e.g., McClanahan and Shafir 1990; Eakin 1996). Like herbivorous fishes, larger urchin species such as *Diadema* spp. may be important in removing algae that would otherwise inhibit coral growth and/or settlement (Edmunds and Carpenter 2001). At high densities, however, intensive grazing by sea urchins may have negative effects on reef habitats, causing significant mortality of juvenile corals, loss of coral cover, thereby reducing topographic complexity of reef habitats (McClanahan and Shafir 1990), and ultimately can lead to a net erosion of the reef carbonates (Glynn et al. 1979; Eakin 1996).

4.5 Water current models

While there are oceanographic and water circulation models available for the Coral Sea (see Section 4.6), the resolution is generally too coarse to resolve reef-scale processes and are not calibrated for the CSMP. Such fine scale variation is particularly relevant for understanding the potential transport and connectivity between reefs and the retention of water masses in the vicinity of reefs. To address this gap, we deployed 50 Marotte HS (high sampling rate) drag and tilt current meters (Figure 4.4) at 18 key locations across 11 reefs (Cato, Wreck, Saumarez, Kenn, Frederick, Flinders, Holmes, Bougainville, and Osprey Reefs in the CSMP, and No Name and Yonge Reefs on the outer-shelf of the GBRMP; Figure 4.5). At each of these reefs the current meters were deployed at two depths (shallow: 3-8m; and deep: 8-15m) at each of 1-2 sites. Additional loggers were also deployed in areas of interest, such as the entrance to lagoons where the results can be used to calculate the residence time. The current meters were deployed over 5 research voyages: December 2018 (Cato, Wreck, Saumarez, and Kenn Reefs), February 2019 (Saumarez, Frederick, and Flinders Reefs), March 2019 (Holmes and Bougainville Reefs), October 2019 (GBRMP: No Name and Yonge Reefs) and

December 2019 (Bougainville and Osprey Reefs). The current meters recorded water flow velocity and direction, and temperature every 1 second for an average of 3 months (the longevity determined by battery life and accumulating fouling on the instrument).

Figure 4.4 A Marriot HS (high sampling rate) drag and tilt current meter deployed at approximately 10m on a reef in the southern Coral Sea Marine Park. This current meter was deployed in February 2019 and the photo taken immediately prior to retrieval in February 2020. Image credit: Martin Russell

The current meters were retrieved and replaced during site re-visits (often after 12 months), though only data recorded during the first 3-months immediately after deployment was used. Five of the shallow current meters were lost, likely due to severe weather and/or wave action (the stakes used for attachment were located but the loggers were missing). Following retrieval, the data was downloaded from each current meter then processed, cleaned and averaged over 10-minute intervals using the Marotte HS software.
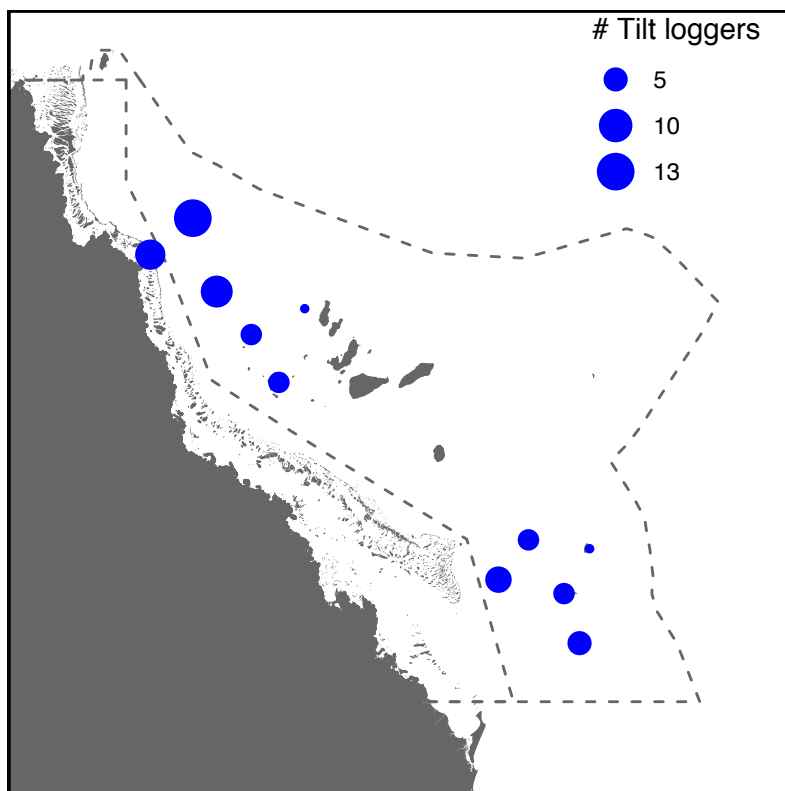
Figure 4.5 Map showing the locations and number of Marriot HS (high sampling rate) drag and tilt current meters deployed throughout the Coral Sea Marine Park and northern Great Barrier Reef Marine Park. The size of the symbols shows the number of current meters deployed at each reef.

4.6 Connectivity of fish populations

Dispersal and connectivity are fundamental ecological processes, driving population dynamics and defining the structure and persistence of populations across ecological and evolutionary timescales (Clobert *et al.* 2001). Since coral reefs are inherently variable environments (at a range of different scales) with reefs separated by varying distances of open water, connectivity between seemingly isolated reefs underpins their resilience to disturbance (Hughes *et al.* 2005) and is central to the management of these ecosystems (Sale et al. 2005; Cowen and Sponaugle 2009). Few population genetic studies of marine organisms span the entire breadth of the GBRMP (Doherty *et al.* 1994; Benzie and Williams 1997; Ayre and Hughes 2004; van Oppen *et al.* 2011) and fewer still have considered the degree of connectivity between reefs of the GBRMP and CSMP. Those that have focus on the giant clam (*Tridacna maxima*) and the foraminifer (*Marginopora vertibralis*) (Benzie 1991, 1998; Benzie and Williams 1992). While early evidence suggested genetic homogeneity across the GBRMP, it has also been suggested that reefs in the Queensland Plateau may act first, as a genetic stepping-stone

between the broader Coral Sea and the GBRMP (Benzie 1998) and second, as a lasting refuge for genetic rescue of populations in the GBRMP.

Genetic methods have traditionally been used to infer the magnitude of connectivity between populations or in this case, distinct coral reefs. They rely on estimates of genetic diversity within populations to estimate the degree of historical gene flow between them. Genetic connectivity is thus the degree to which gene flow affects evolutionary processes within populations (Lowe and Allendorf 2010). Spatial patterns of genetic variation have traditionally been analysed through summary statistics between pairs of populations, therefore missing the simultaneous influence of all populations. A network approach proposes to overcome these limitations by isolating the spatial distribution of genetic variance (Dyer and Nason 2004; Garroway et al. 2008), thus isolating the fraction of genetic diversity that is *unique* to each population from the fraction that is *shared* amongst populations. This approach allows us to deconstruct spatial patterns of genetic variation and quantify the exchange of individuals between habitat patches, thus providing an estimate of demographic connectivity between populations (Rozenfeld et al. 2008).

Tissue samples for 474 bluespotted coral trout (*Plectropomus laevis*) and 476 twospined pygmy angelfish (*Centropyge bispinosa*) were collected from 19 and 17 coral reef atolls in the CSMP and GBRMP, respectively. Samples were collected on SCUBA using spears and custom-made biopsy probes (Pneu-Dart, USA). All samples were collected between March 2016 and December 2017 and preserved in 95% ethanol for genetic analyses. Restriction site-associated DNA sequencing libraries were prepared using a modified double-digest RadSeq protocol described by Peterson et al. (2012) and sequenced on in a single lane on an Illumina Hi-Seq 2500 or Hi-Seq 4000 at KAUST Genomics Core facilities. SNP data were obtained *denovo* following pipelines in STACKS v1.44 (Catchen et al. 2013) with filtering for quality control.

Oceanographic dispersal distance was measured using a Lagrangian model of particles dispersing in the GBR region and Queensland Plateau (Connie2 online tool, available at http://www.csiro.au/connie2/; Condie et al. 2005). The model covers the full extent of the sampling region and includes all major driving forces

such as tides, winds and offshore forcing, has a spatial resolution of 4 km and a temporal range of 8 years (2010-2017). The model parameters incorporate the known pelagic larval duration for *Plectropomus* spp (24 - 29 days) and particle release events are timed with lunar phases year-round. Particles were released from 20 reefs in the CSMP. The number of particles that 'settle' within 4 km of any focal site across simulations will represent the potential connectivity among reefs in the seascape. All model simulations were performed in collaboration with CSIRO Marine and Atmospheric Research laboratories in Hobart, Tasmania. Genetic and oceanographic connectivity matrices were compared using network graphs and multiple regressions to measure the influence of oceanography on the spatial patterns of genetic diversity of reef fish in the CSMP.

4.7 Disturbance history

We used historic tropical cyclone (TC) track data with models that reconstruct cyclone activity (Puotinen et al. 2016) to estimate and measure the potential for damaging wave activity from past cyclones near reefs in the CSMP. We then mapped damage zones for all TCs likely to generate damaging waves within the boundaries of the CSMP from 1985 to 2019 by predicting where wind generated wave action would be strong enough to cause damage to coral reefs (Puotinen et al. 2016). We defined 'damaging seas' where significant wave height (H_s) reached or exceeded 4m for at least one hour. This measure has been shown to correspond with field observations of severe cyclone damage in the GBRMP (Puotinen et al. 2016) and Western Australia (Puotinen et al. 2020). Significant wave height is a measure of the intensity of sea state. It gives the average height of the top one-third highest waves in an area of ocean. Values of H_s equal to 4 indicate wave heights that can be up to twice this. This preliminary analysis assumes adequate fetch, and thus neglects variations in wave exposure likely between sites based on their position with respect to nearby reefs and islands that can block or reduce incoming wave energy.

4.8 Taxonomic assessments

Corals - The biodiversity and biogeography of the Scleractinia (hard corals) of the CSMP is very poorly known. Collections are limited, with the bulk of CSMP specimens in the Museum of Tropical Queensland (MTQ) dating back to the

1980's. Since this time there has been no comprehensive attempt to document and describe the corals of the CSMP. Nonetheless, several species lists have been compiled following previous monitoring expeditions to discrete areas of the CSMP (Oxley et al. 2003, Oxley et al. 2004, Ceccarelli et al. 2008, Ceccarelli et al. 2009). When combined with a list of species held at MTQ there is a total of 227 hermatypic coral species recorded in the CSMP (Appendix 4). However, these lists predate recent major changes in coral taxonomy (Kitahara et al. 2016). Current and future changes, particularly at species level in abundant groups, such as the *Acropora*, *Montipora* and *Porites*, will make most species lists of little value, highlighting the importance of collecting voucher specimens that allow identification to be revisited.

In the absence of robust data on species occurrences, analyses of the biogeography of the Coral Sea have mostly relied on the hypothesized distributions of species presented in Veron (2000). Using these data, Keith et al. (2013) concluded that the CSMP reefs were part of a large Australian province that included reefs in Western Australia, the GBRMP and the arc of countries on the eastern boarder of the CSMP (New Caledonia, Vanuatu and the Solomon Islands) but excluding reefs in the Tasman Sea (Elizabeth and Middleton Reef and Lord Howe Island). Similarly, Oxley *et al.* (2003) concluded that the CSMP reefs were essential depauperate GBRMP reefs. In contrast, Veron et al. (2015) grouped all the reefs of Coral Sea, including the Chesterfield Reefs, into a distinct Coral Sea Ecoregion. Veron et al. (2015) further suggested that the Coral Sea Ecoregion had closer affinities to the GBR, than ecoregions to the east or south. These contracting ideas of the affinities of the Coral Sea coral fauna suggest the need for further collections to substantiate species occurrences throughout the broader Coral Sea region.

Here, we outline the preliminary findings of collections during this 3-year project plus those of a trip to the central CSMP in 2016. The only targeted coral collections occurred in the southern CSMP in December 2019. Collections of corals in the northern (October 2018) and central CSMP (December 2016) were opportunistic. Most of the species records are supported by voucher specimens that will be lodged at MTQ in the near future, however, a few are supported only by images.

Voucher specimens were identified by comparison to the relevant type material and, for a small number of samples to date, molecular analysis.

Fish - Detailed taxonomic assessments of conspicuous (i.e., non-cryptic) reef-associated fishes were compiled based on individuals recorded during our visual surveys and supplemented with underwater observations and photographs taken outside of surveys. Taxonomic assessments of smaller cryptic fish species (hereafter cryptobenthic fishes) that are not effectively captured using visual surveys (Ackerman and Bellwood 2000) were sampled using a series of enclosed $4m^2$ clove oil stations. At each sampling station, a small coral outcrop (or bommie) was fully enclosed by a fine mesh net ($4m^2$ with weighted edges). An impermeable membrane (a waterproof tent fly with weighted edges) was then placed over the net and ~2.5 L of anaesthetic clove oil solution (a fish anaesthetic) was released into the enclosure using spray bottles (Figure 4.6 a). The impermeable membrane retained the clove oil in place, ensuring its efficacy in anaesthetising the cryptobenthic fishes and prevented adverse effects on any organisms outside the enclosure. After a short period (2-5 mins) the net was slowly and progressively removed and divers thoroughly searched the underlying area, collecting all anaesthetised fish with forceps (Figure 4.6 b).
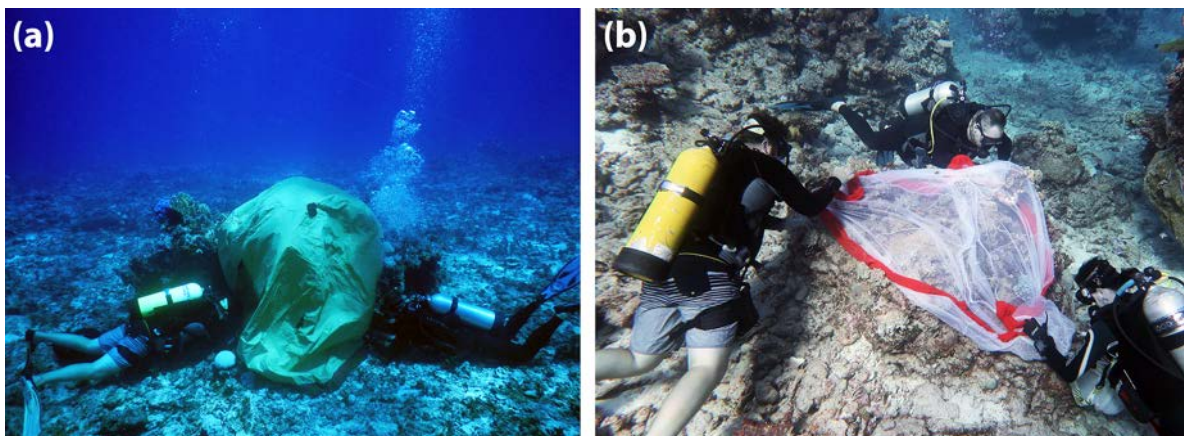
Figure 4.6 Collection of cryptobenthic reef fishes using enclosed clove oil stations; (a) divers releasing clove oil solution under the impermeable membrane; (b) divers searching for and collecting cryptobenthic fishes as the fine mesh net is slowly removed. Image credits: (a) Amanda Hay, (b) Martin Russell.

4.9 Demographic rates of corals and reef fish

Coral recruitment - Densities of juvenile corals (≤5 cm maximum diameter, following Rylaarsdam 1983) are increasingly used to quantify coral recruitment and hence the replenishment of coral populations as opposed to settlement studies that deploy experimental settlement substrates (e.g., tiles) and quantify the number of coral larvae that settle to these substrates. Comprehensive counts of all juvenile colonies, including the smallest colonies that are detectable with the naked eye (approximately 1 cm diameter), enable effective comparisons of coral recruitment among habitats, sites and reefs across the CSMP. All juvenile corals within the 10 x 1m coral health transect were recorded to genus level (Figure 4.7).
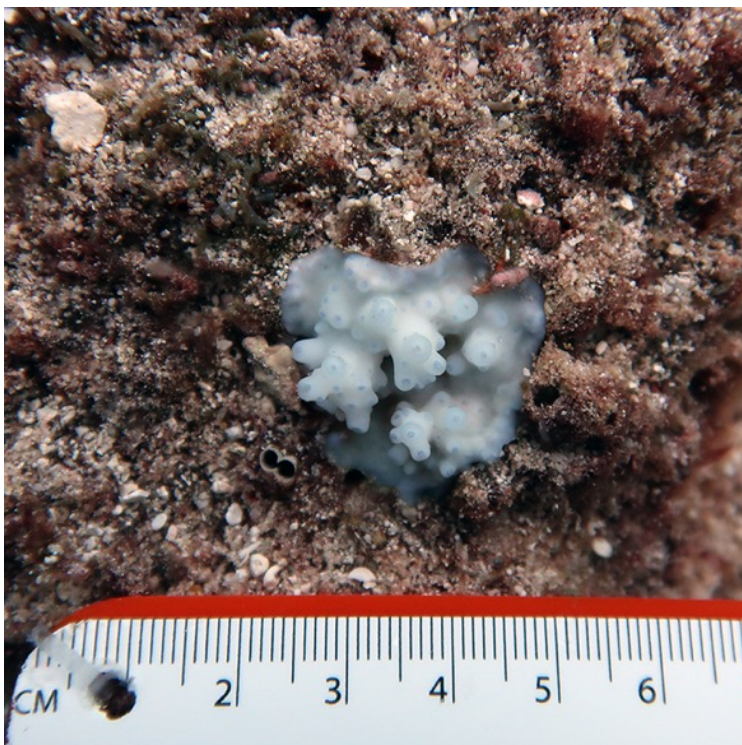
Figure 4.7 Juvenile *Acropora* coral approximately 2cm in diameter surveyed on Diamond Banks, central Coral Sea Marine Park in March 2020. Image credit: Deborah Burn

Coral growth - Cores of massive reef building corals such as *Porites* spp. can provide long-term records of coral growth and, through geochemical and isotopic analyses, regional and global changes in climate and oceanographic conditions. Massive *Porites* corals grow by precipitating aragonite onto their skeletal matrix and deposit annual density bands that allow for chronological studies of skeletal density, linear extension and calcification rates. Isolated coral reef atolls in the CSMP are removed from human influence and provide a record of regional and global environmental constraints on the growth of reef-building corals (Figure 4.8). A

collection of 48 coral cores (each up to 1.5m in length) from 11 isolated atolls in the CSMP contain growth records for massive *Porites* corals for the past 187 years (1832-2018), and thereby provide both a long-term historical and regional perspective on the elevated risk of climate change and ocean acidification on the growth of these, and similar reef-building corals in the CSMP. These cores are permanently available for future work and complement the archival cores at the Australian Institute of Marine Science which span the GBRMP and reefs from the North-west Marine Parks Network.

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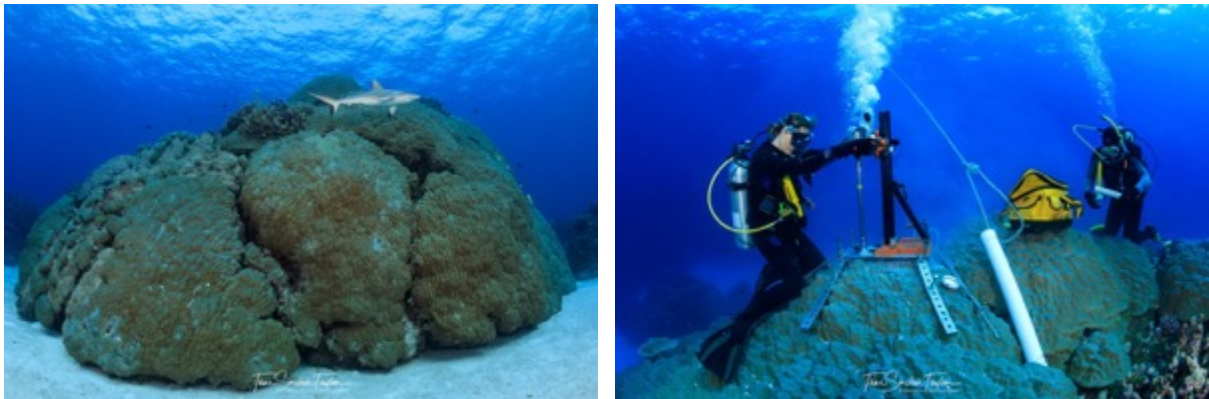
Figure 4.8 'Big Mel' stands 8m tall in the heart of the Coral Sea at Mellish Reef. Aspects of the bommie (*Porites lobata*) show signs of recent and past disturbances, notably sand-blasting from storms, and bleaching. However, 1.5m cores from December 2018 show uninterrupted growth for over 100 years (i.e., 1910-2018). Left: Big Mel dominates over a sandy bottom with a grey reef shark in the foreground. Right: A research scientist from the Australian Institute of Marine Science use pneumatic tools to core through the top of Big Mel. Image credit: Tane Sinclair-Taylor.

Coral skeletal cores were extracted from massive (i.e., dome- or hemispherical-shaped) *Porites* colonies using underwater pneumatic drills operated by SCUBA divers. Cores from the CSMP (total = 48) were sampled from colonies living between 1 and 20 m depth in November and December 2017 and December 2018 (Figure 4.9). Cores were drilled with 5 cm diameter bits, and core holes were filled with cement plugs to prevent infestation by bio-eroding organisms and to provide the coral tissue with a hard surface to grow over during recovery. Annual data for three growth parameters were obtained from 36 cores with the use of standard x-ray and gamma densitometry techniques (Lough and Barnes 2000). These include

average annual skeletal density (measured as the average density between adjacent density minima; g.cm^{-3}), annual extension rate (measured as the linear distance between adjacent density minima; cm.year^{-1}), and annual calcification rate which is the product of skeletal density and annual extension (g.cm^{-3}.year^{-1}).
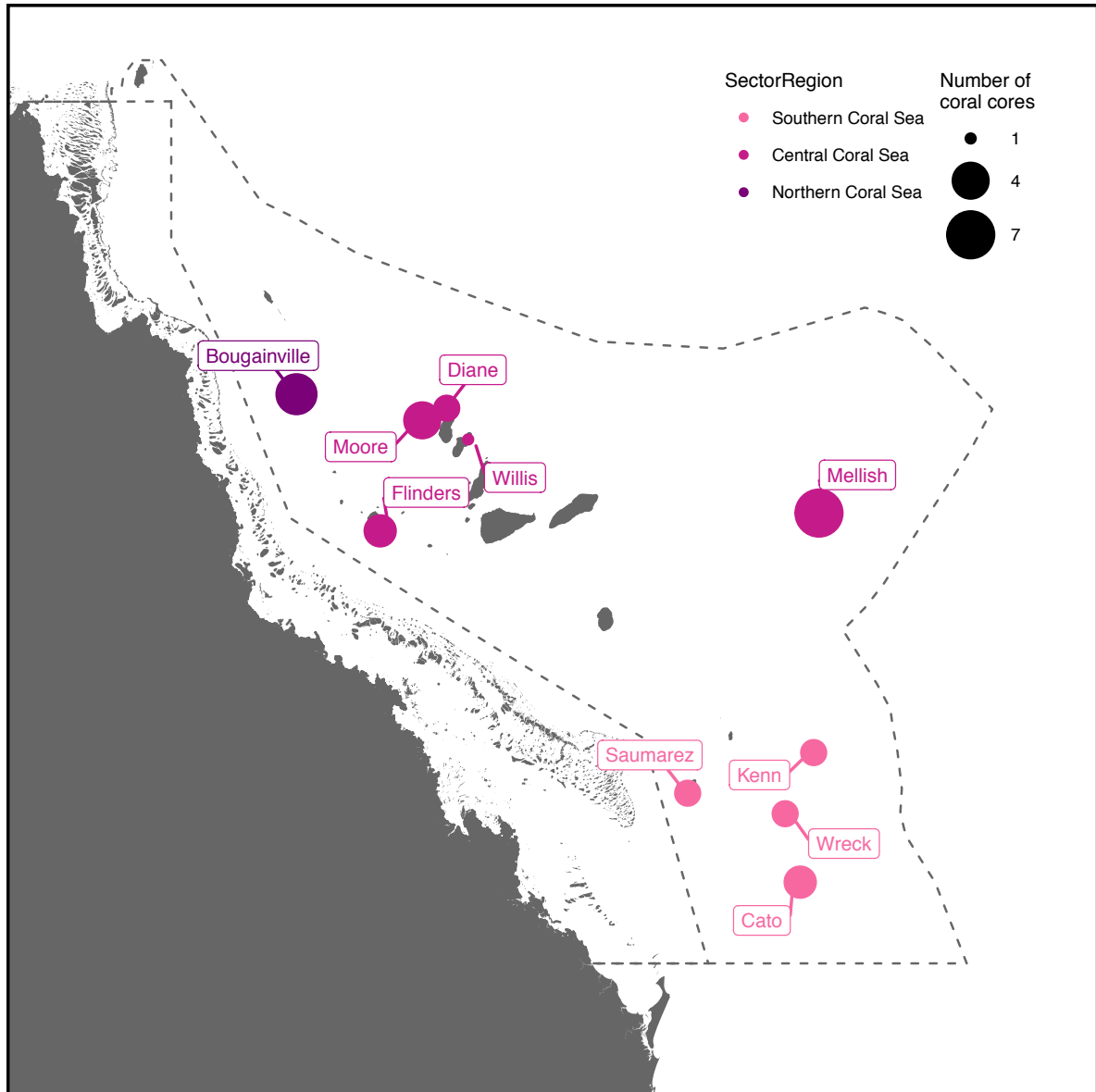
Figure 4.9 Map showing the approximate locations of *Porites* corals cored in the Coral Sea Marine Park between 2017 and 2018. The size of points indicates the number of cores collected from each reef.

High-density 'stress bands', abrupt declines in annual linear extension, within the skeletal growth record reveal signatures of stress events that have been attributed to mass bleaching events caused by thermal stress. These discrete high- density

'stress bands', visible in micro-computed tomography (mirco-CT) scans, can accurately date historical disturbance events (Carilli et al. 2009; Lough and Cooper 2011; DeCarlo and Cohen 2017; Barkley et al. 2018). Cores collected from the CSMP (n = 30) in November and December 2017 at Bougainville Reef (n = 9), Moore Reefs (n = 5), Diane Bank (n = 3), Willis Islets (n = 3), Magdalene Cays (n = 7), and Flinders Reefs (n = 3) were micro-CT scanned to visualize stress bands, partial mortality scars, and annual density bands.

Fish growth and mortality - Variation in the demographic rates (e.g., growth and mortality) is common within reef fishes, particularly for species with broad geographic distributions that span large gradients in environmental conditions. Numerous studies have reported positive relationships between water temperature and the growth and/or mortality of reef fishes (e.g., Choat and Robertson 2002, Trip et al. 2008). Understanding the spatial variation and associated environmental drivers of demographic processes of reef fishes is fundamental for establishing effective management and conservation strategies.

We examined the sagittal otolith (ear bone) microstructure of four species of common reef fish (the common coral trout *Plectopomus leopardus,* bluespotted coral trout *Plectropomus laevis,* twospined pygmy angelfish *Centropyge bispinosa,* and the lined bristletooth *Ctenochaetus striatus*) to compare their demographic rates among regions within the CSMP, and between the CSMP and adjacent GBRMP. The species were selected to span a range of taxonomic and trophic groups, and maximum body sizes (*P. leopardus*: 75 cm total length, TL; *P. laevis*: 100cm TL; *C. bispinosa*: 10cm TL; *C. striatus*: 26cm TL; Randall et al. 1997). Individuals from each species were collected from reefs throughout the CSMP and GBRMP, and supplemented with historical collections where possible (Figure 4.10). The length and weight of each individual was recorded, their sagittal otoliths removed and sectioned, and the data modelled using von-Bertalanffy growth functions (following Taylor et al. 2019).
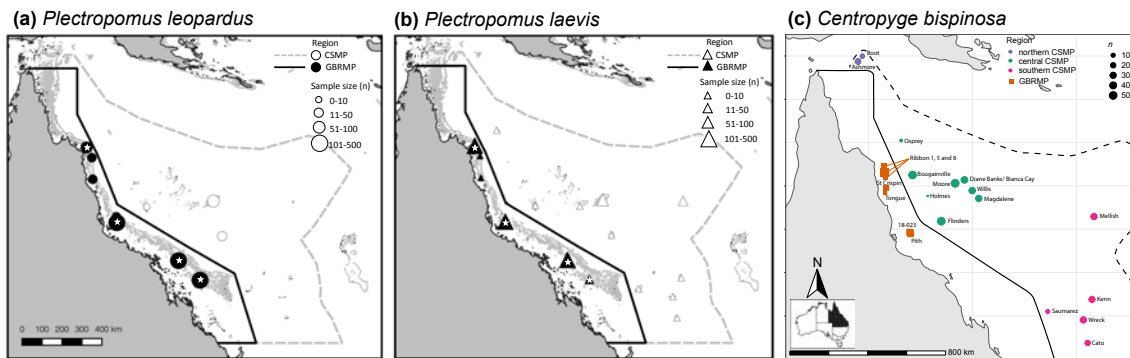
Figure 4.10 Map of sampling locations for (a) *Plectropomus leopardus*, (b) *Plectropomus laevis*, and (c) *Centropyge bispinosa* in the Coral Sea Marine Park and Great Barrier Reef Marine Park. Historical collections for *P. leopardus* and *P. laevis* from the Effects of Line Fishing Project (Mapstone *et al.,* 2004; Heupel *et al.,* 2010) are indicated by a star symbol. Sample size are relative to shape size in the legend key. Sample sizes for *Ctenochaetus striatus* were n = 53, 62, 154, and 217 for the northern, central, and southern Coral Sea Marine Park, and Great Barrier Reef Marine Park, respectively.

4.10   Data handling and analysis

Surveys from each voyage were compiled into a single database and analysed using R version 3.5 (R Core Team 2019). Data were wrangled using the *tidyverse* environment (Wickham 2017) and visualised using the *ggplot2* package (Wickham 2016). Colour palettes for figures were chosen in *RColorBrewer* (Neuwirth 2014) and *viridis* (Garnier 2018), with visualisations aided by *ggrepel* (Slowikowski 2018) and *ggpubr* (Kassambara 2018). All survey data were averaged across independent transects within zones (crest, slope) to obtain a site average prior to summarising data at the level of reefs or regions.

Non-metric multi-dimensional scaling (nMDS) were used to identify similarities in coral and fish assemblages among reefs in *a priori* defined regions (i.e., southern, central, and northern CSMP, southern, central, and northern GBRMP), among surveyed sites, and between reef habitats (reef slope and reef crest). The objective of nMDS is to summarise all available information on the presence and abundance of species, or taxa, into a simple similarity matrix. In the visual representations that follow, objects (i.e., sites or reefs) that are closer to one another are likely to be more similar than those further apart. Data were square-root transformed before Bray-Curtis dissimilarity measures were calculated. Analysis were done in R using the package *vegan* (Oksanen et al. 2018).

5   Findings

5.1 Impacts of 2016 and 2017 coral bleaching events

In the two years immediately preceding these surveys (i.e., 2016 and 2017), mass coral bleaching was recorded throughout the central and southern CSMP (Harrison et al. 2018, 2019). In 2016, extensive bleaching was recorded on five reefs surveyed in the central CSMP (Mellish, Lihou, Holmes Reefs, Herald Cays, and Chilcott Islet) with up to half of all coral colonies surveyed being moderately to severely bleached (>50% bleached), but no bleaching was reported for three reefs (Wreck, Saumarez, and Kenn Reefs) in the southern CSMP (Harrison et al. 2018). The 2017 bleaching event was more widespread, affecting reefs in both the central and southern CSMP, but generally less severe. Understanding the impacts of these events on the cover and composition of coral assemblages is critical in assessing the health of reefs in the CSMP.

Comparisons of coral assemblages among years (2016-20) revealed a distinct shift in species composition from 2016-17 to 2018-20 (Figure 5.1 a). Coral assemblages in 2016-17 were characterised by a higher relative cover of tabular *Acropora, Dipsastrea*, and, to a lesser extent, staghorn *Acropora*, whereas reefs in 2018-20 were characterised by a higher relative cover of *Montipora* and 'other' scleractinian corals (Figure 5.1 b). These changes in the composition of coral assemblages were most pronounced in the central CSMP, where coral assemblages shifted from being dominated by tabular and staghorn *Acropora* in 2016-17 to *Montipora, Isopora*, and 'other' sclertinian corals in 2018-20 (Figure 5.1 c,d). This shift in coral composition is consistent with, and likely reflects the impacts of, the 2016-17 bleaching events within the CSMP. The greater shift in species composition on central versus southern CSMP reefs is consistent with the spatial footprint and differences in the severity of the bleaching recorded in 2016 and 2017 (Harrison et al. 2018). Tabular and staghorn *Acropora* are among the most sensitive coral taxa to elevated water temperatures (e.g., Marshall and Baird 2000; Loya et al. 2001; McClanahan et al. 2004) and are often the first to be lost following large-scale bleaching (Bento et al. 2016; Hughes et al. 2018). Similar shifts in the composition of coral assemblages were evident following the 2016 bleaching event on the GBR (Hughes et al. 2018). The loss of these fast-growing tabular and staghorn *Acropora*

have been shown to reduce the three-dimensional structure and functionality of reef habitats (Hughes et al. 2018), and is likely to have flow-on effects to the fishes and invertebrates that rely on these corals for shelter and/or food (e.g., Pratchett et al. 2008, 2011b).
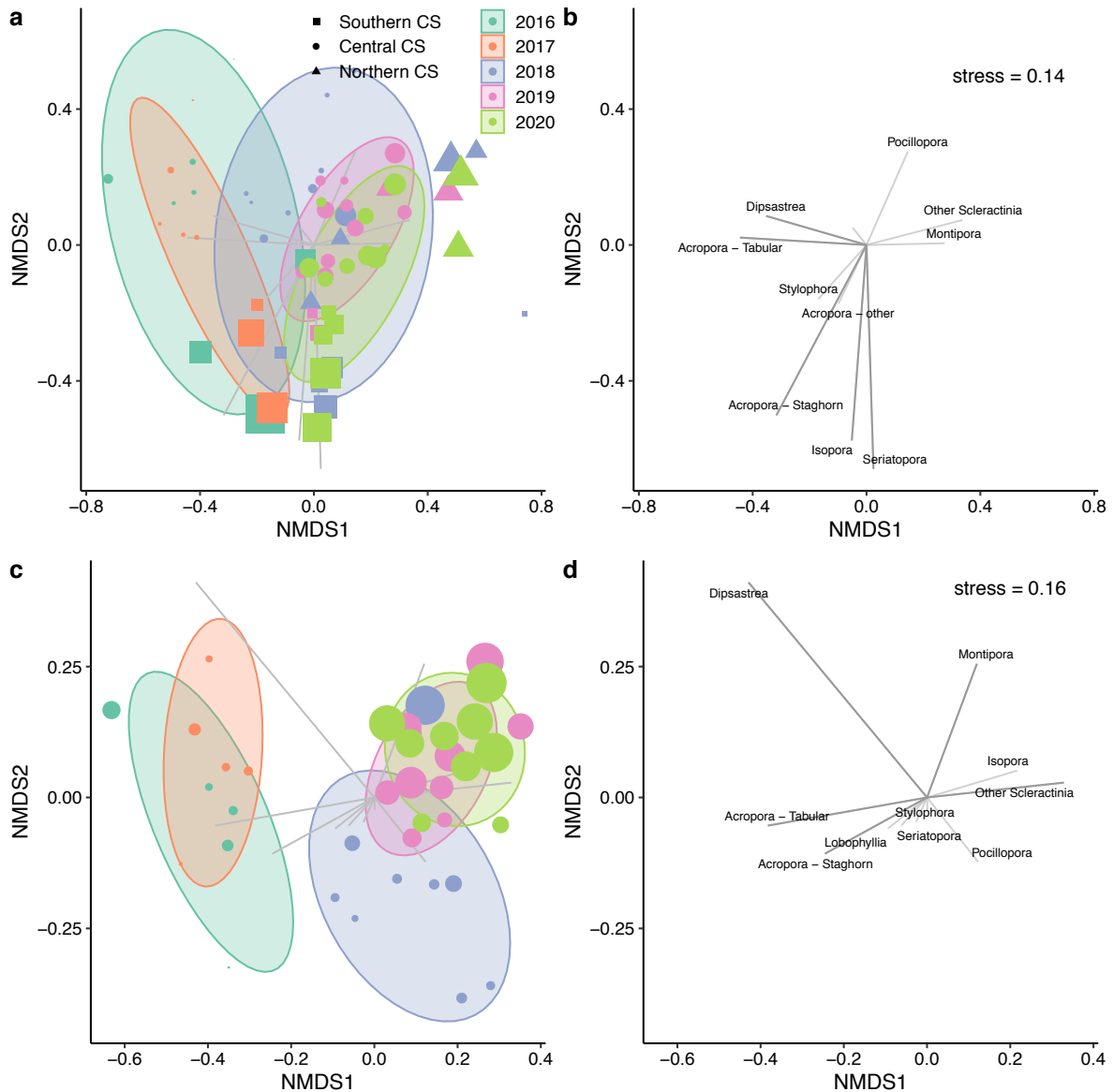
Figure 5.1 Temporal change in the composition of coral assemblages within the Coral Sea Marine Park. Non-metric multidimensional scaling (nMDS) plots showing the variation in coral composition (a,b) among years for all regions of the Coral Sea Marine Park (c,d) among years for reefs in the central Coral Sea Marine Park only. Analyses are based on data from Harrison et al. 2018 (2016-17) and the present surveys (2018-20) surveys. The size of individual points are proportion to the cover of live coral on each reef. Vectors in the right-hand side plot indicate key genera that account for variation in coral composition displayed in the corresponding left-hand side plot.

In contrast to the changes in the composition of coral assemblages following the 2016 and 2017 bleaching events, we found no evidence for reef-level declines in total coral cover for the nine CSMP reefs that were surveyed in 2016-17 and again in 2018-20 (Figure 5.2a). The only two reefs for which total coral decreased were Saumarez and Wreck Reefs in the southern CSMP. Total coral cover remained broadly comparable from 2016-17 to 2018-2020 at the other southern CSMP reef (Kenn Reef) and increased at the six reefs in the central CSMP over the same period (Figure 5.2a). While the apparent decline in total coral cover at Wreck Reef (2016: 56% to 2018: 32%) may be related to the 2017 bleaching event, the temporal changes in coral cover at other CSMP reefs are difficult to reconcile.

The decrease in coral cover at Saumarez Reef occurred from 2016 (32% cover) to 2017 (20% cover), prior to the extensive bleaching recorded at this reef in 2017 (Harrison et al. 2018). Corals typically take several weeks to months to either recover or die following heat stress and bleaching (Baird and Marshall 2002), and as such, the impacts of the 2017 bleaching would not have been realised at the time of the 2017 surveys. Further, total coral cover appeared to increase at all central CSMP reefs surveyed from 2016-17 to 2018-20, despite these reefs experiencing significant bleaching in both 2016 and 2017.

Our inability to detect consistent reef-level declines is likely related to differences in survey methods and/or the sites surveyed between 2016-17 and 2018-20. The 2016-17 surveys of coral cover were based on relatively short 10m line-intercept transects, whereby every coral colony directly beneath the transect tape was identified to genus and its intercept length measured to the nearest centimetre (Harrison et al. 2018), while the 2018-2020 surveys were based on longer 50-m point-intercept transects (described in Section 4.2 above). Line-intercept and point-intercept transects have been shown to be broadly comparable for estimating coral cover over the same transect length (Facon et al. 2016), however, differences in transect length may have contributed to some variation in estimates of coral cover given the heterogenous distribution of corals, and the general placement of the shorter 10m line-intercept transects on relatively small coral bommies versus larger areas of contiguous reef for the longer 50m point-intercept transects. There were also differences in depths surveyed among years with surveys conducted at 6m in

2016, and 2m and 6m in 2017 (Harrison et al. 2019), compared to 1-3m and 9-10m in 2018-20. The greater separation of the habitats in 2018-20 was related to the inclusion of surveys of other taxa, namely reef fishes, and the need to ensure independence of communities surveyed at each depth.

Comparisons of coral cover at 16 sites that had been surveyed at least twice during the period 2016-2020 revealed marked variation in the change in coral cover following the 2016 and 2017 bleaching events. Several sites displayed declines in coral cover of up to 50% 2016 to 2017 (i.e., Holmes 1, Holmes 2, Lihou 2), while coral cover at other sites others remained largely the same (i.e., Kenn 4, Lihou 3, Saumarez 1) or increased (i.e., Herald 1, Lihou 8) over the same time period (Figure 5.2 b). Similarly, changes in coral cover from 2017 to 2018 were highly variable, decreasing substantially at three sites (i.e., Lihou 1, Lihou 2, Saumarez 1), remaining largely unchanged at Herald 3, and increasing at seven other sites (Figure 5.2 b). While some of this temporal variation among sites is likely related to differences in coral composition at each site, it does highlight the potential variation among sites in response of total coral cover to the 2016 and 2017 bleaching events, and the need to resurvey the same sites using the same methods to fully understand the impacts of current and future bleaching events on the composition and cover of coral assemblages within the CSMP.

(a) Reef-level changes
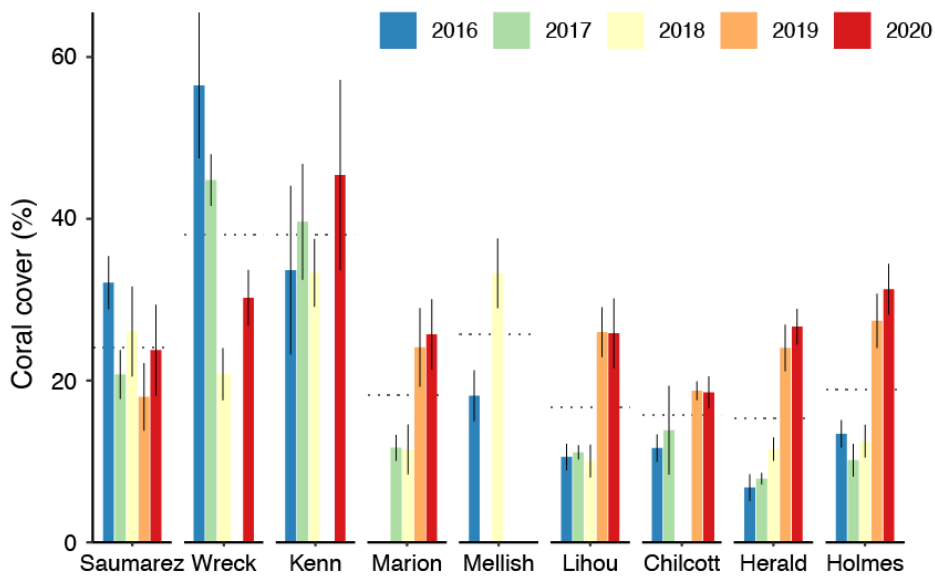
(b) Site-level changes
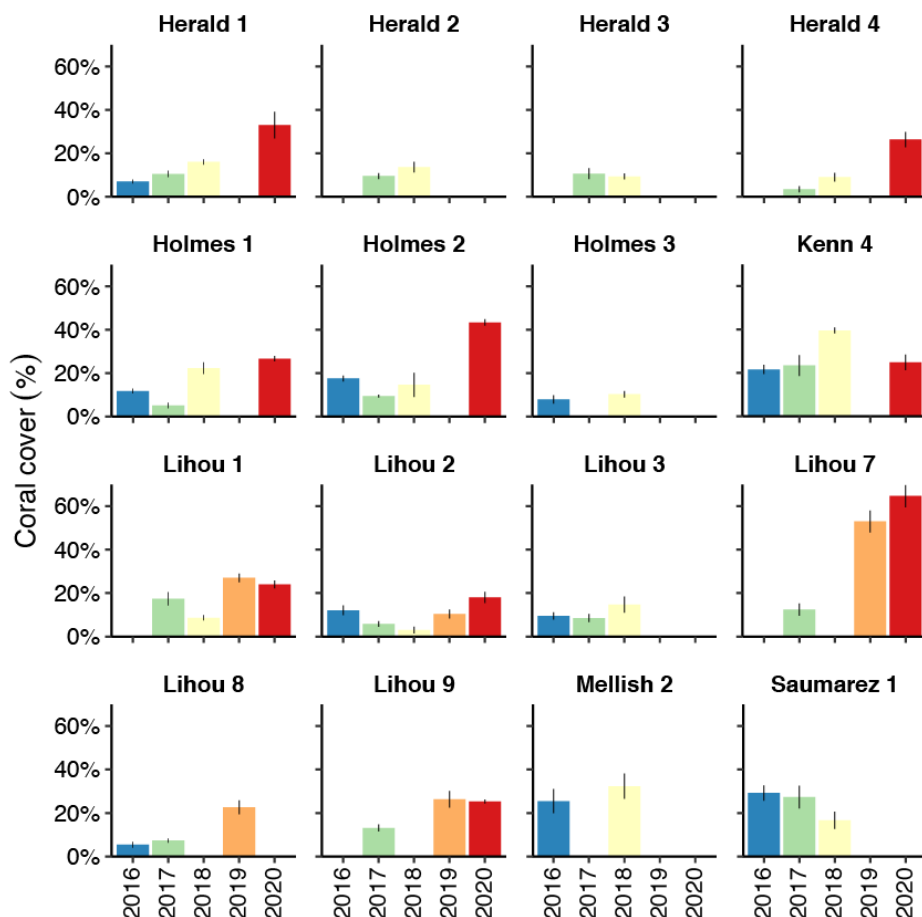
Figure 5.2 Temporal variation (2016-20) in average coral cover (+/- SE) on (a) 9 reefs and (b) 16 sites in the Coral Sea Marine Park. Reef-level coral cover estimates in (a) are averaged across sites and habitats. NB: the same sites within each reef were generally not surveyed in each year, and different survey methods were used in 2016-17 versus 2018-20.Site-level coral cover estimates in (b) are for reef slopes only.

5.2 Regional patterns of biodiversity

The non-metric multi-dimensional scaling (nMDS) plots indicate that coral assemblages, and especially fish assemblages in the CSMP are distinct from those on comparable reefs surveyed along the outer most edge of the GBRMP (Figures 5.3, 5.4). While there are similarities in the diversity of taxa between the two regions, they differ in the relative abundance of key species and/or genera. There are also marked differences in coral and fish communities among regions within the CSMP, but not among the three years of the surveys (Figures 5.3, 5.4). Our current understanding of the drivers of coral and fish communities for offshore coral reefs in the CSMP remains limited.

5.2.1 Coral community composition

The composition of coral assemblages varied regionally, both between the GBRMP and the CSMP, but also among regions within the CSMP (nMDS; Figure 5.3 a-d). Coral assemblages at reefs surveyed in the GBRMP were largely differentiated from the CSMP (most notably from the central and northern CSMP) by the higher relative cover of *Seriatopora,* as well as tabular and staghorn *Acropora*, which are largely absent from the northern and central CSMP reefs (Figure 5.3 a,b). However, reefs in the southern CSMP, where there was relatively high cover of *Acropora* and *Seriatopora*, had broadly similar coral composition to the central and northern GBRMP. The greatest differences in coral composition within the CSMP were found between the southern and northern CSMP reefs (e.g., Kenn, Cato and Wreck Reefs vs. Osprey and Bougainville Reefs; Figure 5.3 c,d). Relatively high cover of *Isopora* and *Seriatopora* characterised southern CSMP reefs (e.g. Cato Reef), whilst northern reefs such as Bougainville and Osprey Reefs had higher cover of branching *Porites* corals, and, to a lesser degree, *Pocillopora* (Figure 5.3 c,d). Sites surveyed at Mellish and Marion Reefs, some of the most isolated reefs in the CSMP, were most similar to the central CSMP reefs in terms of coral composition (Figure 5.3 a,b). Coral assemblages in the southern GBRMP were differentiated from all other reefs by their extremely low cover of live coral due to an ongoing population outbreak of crown-of-thorns starfish. Coral assemblages at

reefs in the central and northern GBRMP (e.g., Pith, 18-023 and Ribbon No.1 Reefs) were differentiated from reefs in the central and northern CSMP by the higher cover of *Acropora* at these reefs, which are typically rare or absent from shallow reef habitats in the central and northern CSMP (Figure 5.3 a,b).

Coral assemblages were differentiated among northern and southern CSMP reefs due to the relative high cover of branching *Porites*, *Diploastrea,* and, to a lesser extent, *Echinopora* on northern CSMP reefs (especially on the reef slope), and conversely the relatively high cover of *Seriatopora*, *Isopora*, and staghorn *Acropora* on southern CSMP reefs (Figure 5.3 c,d). Notably, it is these taxa, along with tabular *Acropora* and *Pocillopora* (Pratchett et al. 2020), that typically dominate shallow water coral assemblages, but all these taxa (with the exception of *Pocillopora*) were conspicuously rare (if not absent) across large areas of the central and northern CSMP (Figure 5.3 c,d). Coral assemblages at reefs in the central CSMP, with the exception of Mellish Reef, were characterised by relatively low cover of all coral taxa, though the coral species composition closely resembled that of reefs in the northern CSMP.

The composition of coral assemblages was relatively stable across the three-year period (2018-20), but distinct from those of 2016-17 (see Section 5.1 above), with almost complete overlap among years and no detectable shifts in coral composition for the southern, central, or northern CSMP (Figure 5.3 e,f).
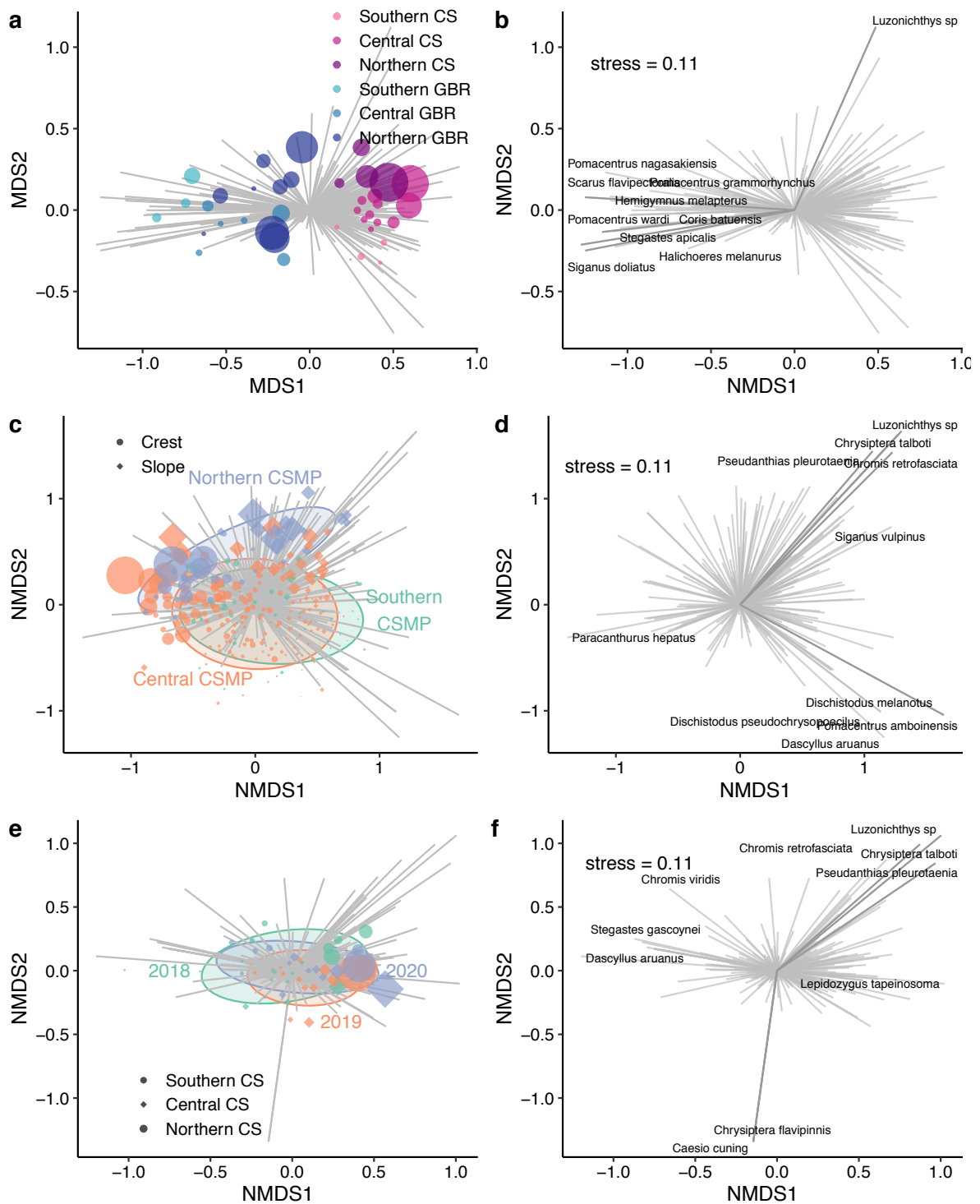
5.2.2 Fish community composition

As for corals, the composition of reef fish assemblages displayed marked variation both between the GBRMP and CSMP reefs and among regions within the CSMP, but not among years (nMDS; Figure 5.4). Species assemblages at surveyed reefs in the GBRMP were differentiated from the CSMP by higher densities of species of rabbitfish (f. Siganidae), wrasses and parrotfishes (f. Labridae) and coral bream (f. Nemipteridae) in the GBRMP and higher densities of species of anthias (f. Serranidae), angelfishes (f. Pomacanthidae) and damselfishes (f. Pomacentridae) in the CSMP. Within the CSMP, reefs within each region generally are clustered together. The main exceptions were Mellish and Moore Reefs that were most similar to the northern CSMP reefs in terms of fish composition (Figure 5.4 c,d).

Fish communities at reefs in the northern CSMP (Ashmore, Boot, Osprey and Bougainville Reefs) were differentiated from the southern and central CSMP by the relative high densities of pelagic species (barracudas: f. Sphyraenidae, trevally: f. Carangidae), herbivorous chubs (f. Kyphsoidae), and anthias (f. Serranidae) on northern CSMP reefs. Conversely, relatively high densities of herbivorous species (rabbitfishes: f. Siganidae, parrotfishes: f. Labridae, tribe Scarini) and some farming damselfishes (*Dischistodus* spp: f. Pomacentridae) characterised the southern and central CSMP reefs (Figure 5.4 c,d). Within the central and northern CSMP there was distinct separation of fish communities between the shallow reef crest and deeper reef slope. Notably, reefs within the southern CSMP were distinguished by the lower densities of all fish species and families (Figure 5.4 c,d). Importantly, these regional differences in fish assemblages were not driven by a limited number of species, with similar regional differences in species composition evident within individual fish families (Figure 5.5).

The composition of reef fish assemblages was relatively stable across the three-year period (2018-20), with almost complete overlap among years and no detectable shifts in fish composition for the southern, central, or northern CSMP (Figure 5.4 e,f).
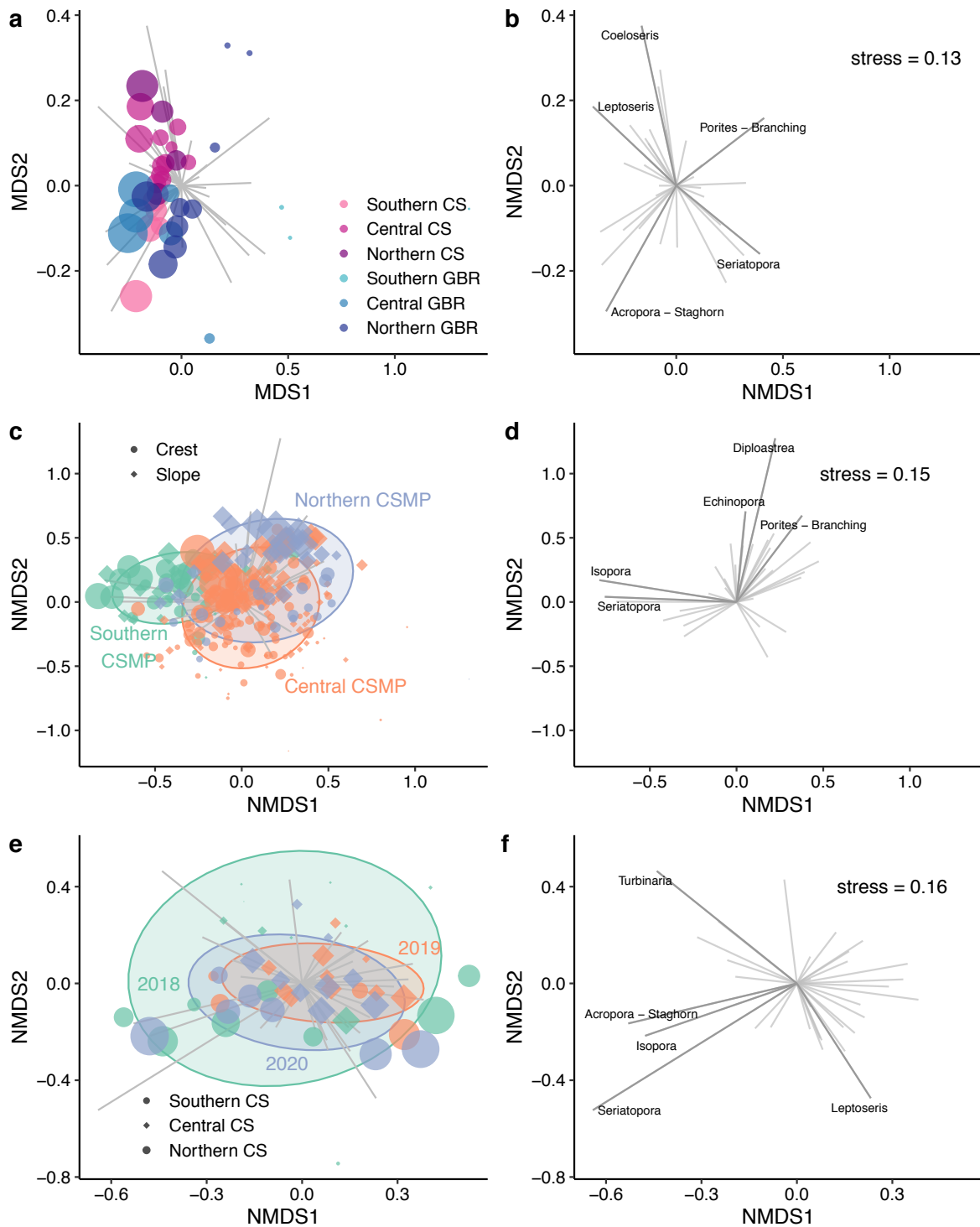
Figure 5.3 Coral community composition. Non-metric multidimensional scaling (nMDS) plots showing the variation in coral composition (a,b) among regions and reefs surveyed in the Coral Sea Marine Park and Great Barrier Reef Marine Park, (c,d) among regions, sites and habitats within the Coral Sea Marine Park, and (e,f) among regions and years in the Coral Sea Marine Park. Analyses are based on the 2018-20 surveys across 20 reefs in the Coral Sea Marine Park and 18 outer-shelf reefs in the Great Barrier Reef Marine Park. Vectors in the right-hand side plots indicate key genera that account for variation in coral composition displayed in the corresponding left-hand side plots.

Figure 5.4 Fish community composition. Non-metric multidimensional scaling (nMDS) plots to show variation in reef fish assemblages (a,b) among regions and reefs surveyed in the Coral Sea Marine Park and Great Barrier Reef Marine Park, (c,d) among regions, sites and habitats within the Coral Sea Marine Park, and (e,f) among regions and years in the Coral Sea Marine Park. Analyses are based on the 2018-20 surveys across 20 reefs in the Coral Sea Marine Park and 18 outer-shelf reefs in the Great Barrier Reef Marine Park. Vectors in the right-hand side plots indicate key species that account for variation in fish composition displayed in the corresponding left-hand side plots.
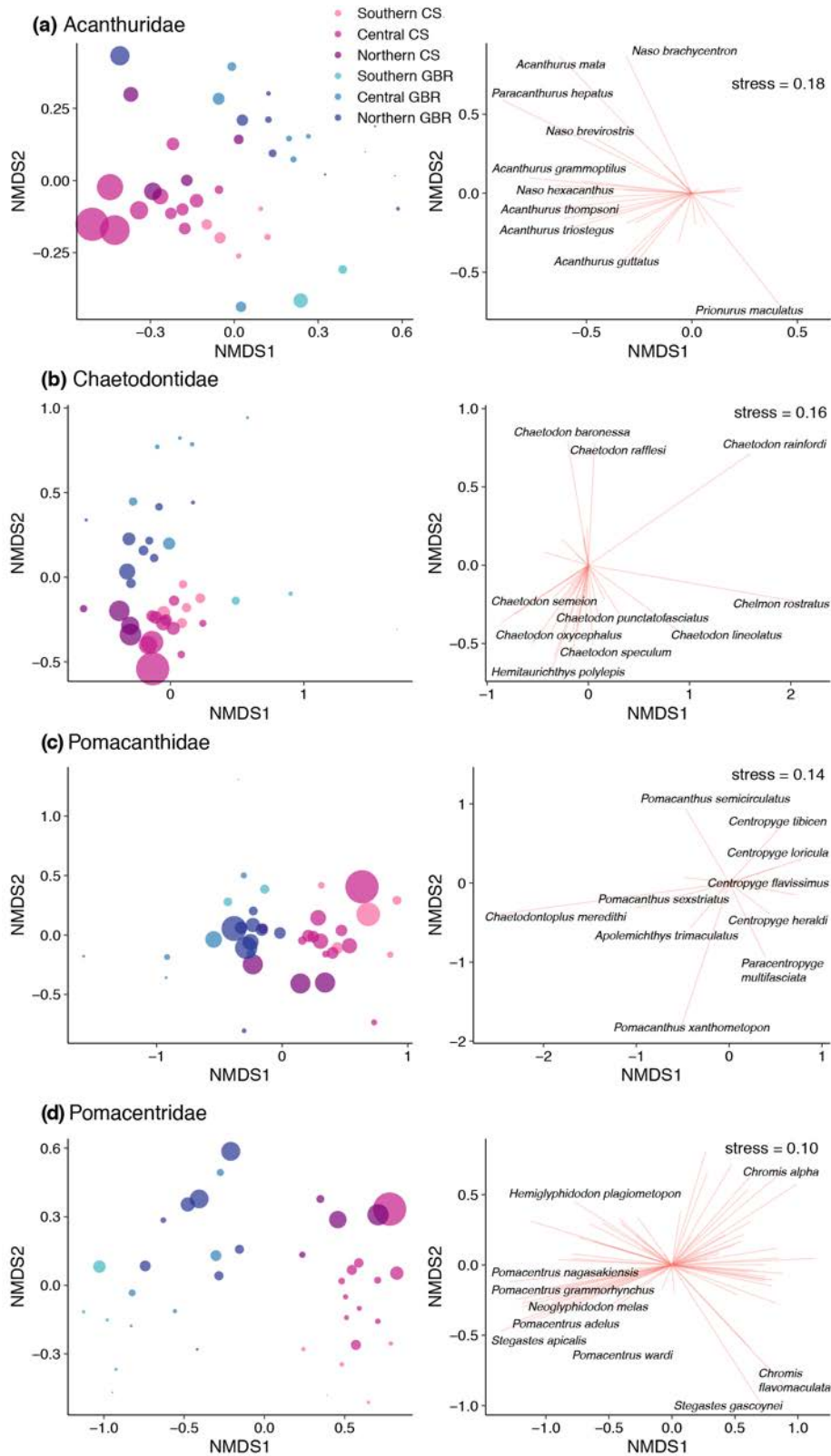
Figure 5.3 Variation the species composition of (a) surgeonfishes (f. Acanthuridae), (b) butterflyfishes (f. Chaetodontidae), (c) angelfishes (f. Pomacanthidae), (d) damselfishes (f. Pomacentridae). Analyses are based on the 2018-20 surveys across 20 reefs in the Coral Sea Marine Park and 18 outer-shelf reefs in the Great Barrier Reef Marine Park. Vectors in the right-hand side plots indicate key species that account for variation in fish composition displayed in the corresponding left-hand side plots.
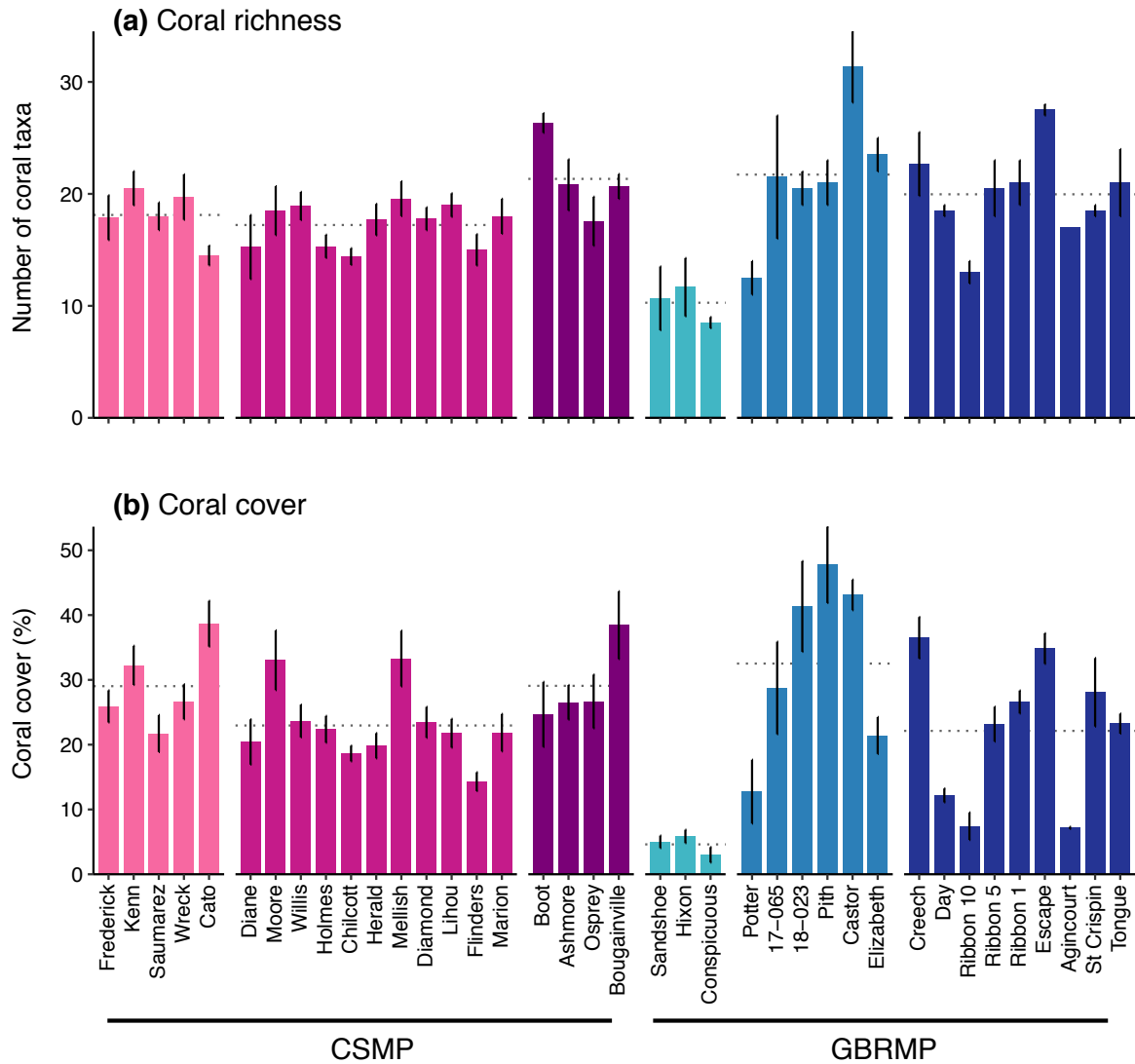
5.3 Coral diversity and abundance

Coral diversity – The average taxonomic richness of corals across the CSMP, based on the number of hard (Scleractinian) coral taxa (mostly genera) recorded at each survey site, was 18.3 taxa per site and ranged from 14.4 taxa per site (± 0.7 SE) at Chilcott Islet (Coringa-Herald Islets and Cays) to 26.3 taxa per site (± 0.9 SE) at Boot Reef (Figure 5.6 a). Coral diversity was generally greater in the northern CSMP (21.4 taxa per site), compared to the central (17.2 taxa per site) and southern CSMP (18.0 taxa per site; Figure 5.6 a).

There was much greater variation in coral species richness within the GBRMP, ranging from 12 species per site on Conspicuous Reef to 97 species per site on Castor Reef. While there was a tendency for coral richness to increase with decreasing latitude in the CSMP, such a trend was not evident within the GBRMP, with the highest richness recorded in the central GBRMP (21.6 species per site), lowest in the southern GBRMP (10.2 species per site), and intermediate in the northern GBRMP (19.9 species per site). The exceptionally low diversity on the southern GBRMP reefs was likely due to the effects of an outbreak of crown-of-thorns starfish in this region at the time of the surveys. Despite clear differences in coral community composition between the GBRMP and CSMP (Figure 5.3 a), there were no clear longitudinal patterns in coral richness or coral cover (Figures 5.7 and 5.8).

(a) Coral richness

(b) Coral cover

CSMP   GBRMP

Figure 5.6 Diversity and abundance of corals. Richness of coral taxa and coral cover were averaged (+/- SE) for each of 20 reefs in the Coral Sea Marine Park and 18 outer-shelf reefs in the Great Barrier Reef Marine Park. Reefs are arranged into three regions (southern, central, and northern) for each of the Coral Sea Marine Park and Great Barrier Reef Marine Park and coloured by *a priori* regional assignments (following Figure 4.1). Dotted lines represent regional averages.
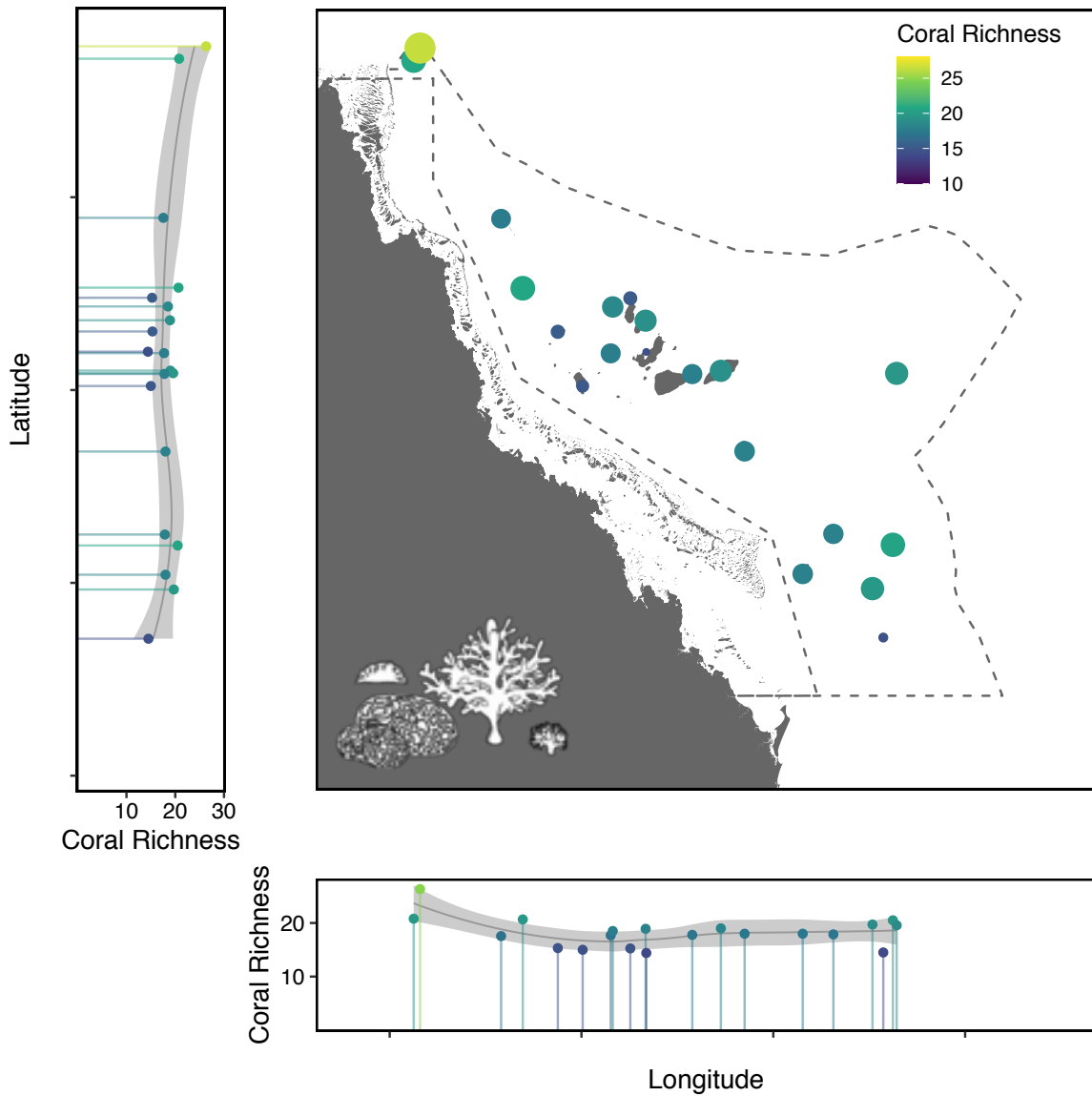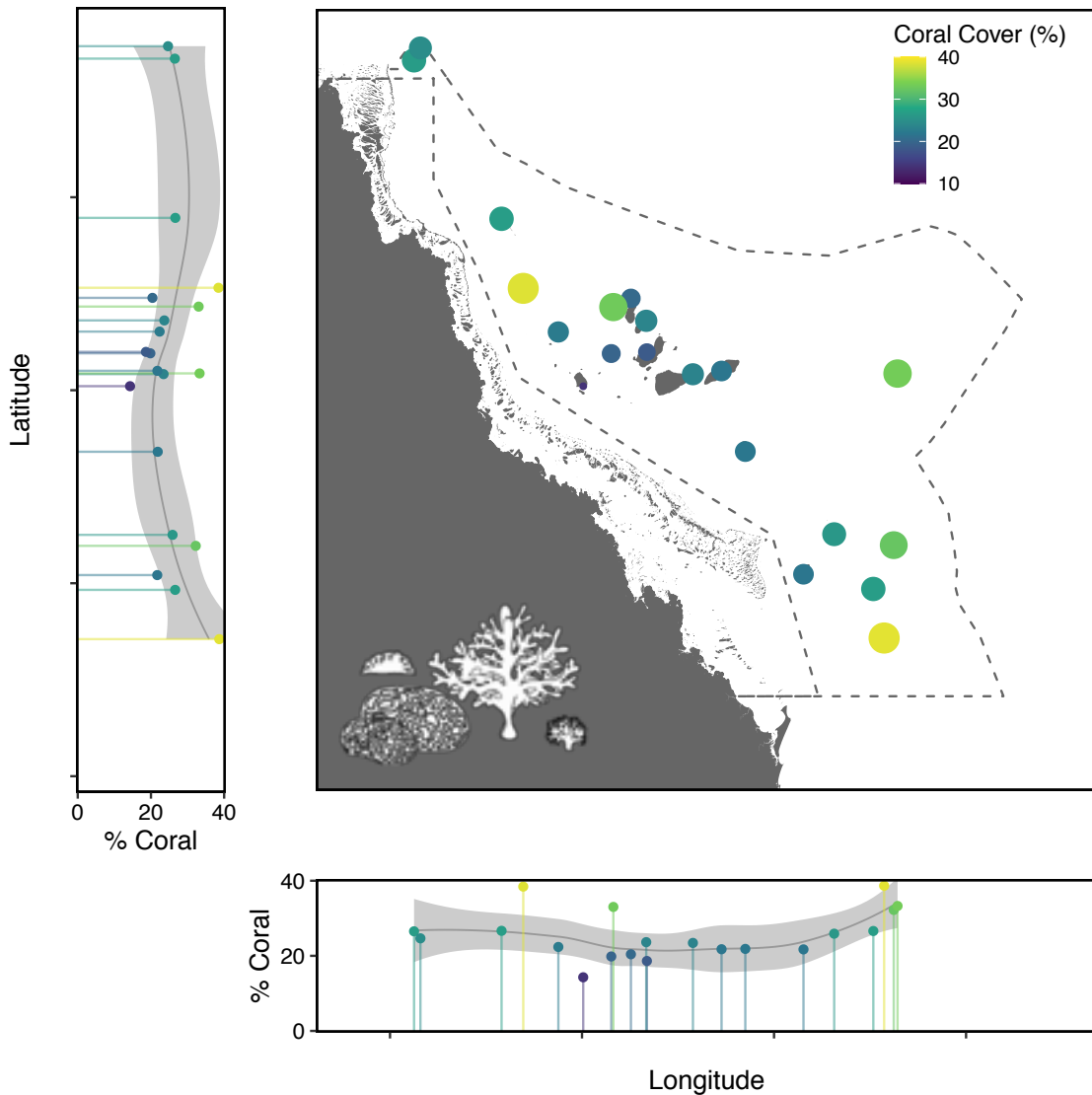
Figure 5.7 Spatial variation in coral richness (number of taxa.reef^{-1}) by latitude (left panel) and by longitude (bottom panel) for the 20 surveyed reefs in the Coral Sea Marine Park (2018-2020). Values for each reef are averaged across sites, habitats, and years.

Coral cover – Across the 3-year period (2018-2020), average cover of hard (Scleractinian) corals recorded across the CSMP sites was 24.80% (±0.45 SE), ranging from 12.19% (±1.33 SE) at Flinders Reefs in the central CSMP up to 38.67% (±2.32 SE) at Cato Reef in the southern CSMP (Figure 5.6 b). Coral cover was notably lower in the central CSMP (averaging 21.69%), compared to the northern (30.57%) and southern regions (26.97%).

There was much greater variation in coral cover on the outer-shelf reefs of the GBRMP, with average coral cover ranging from 4.81% (±0.53 SE) in the southern GBRMP to 23.82% (±1.36 SE) and 33.35% (±1.88 SE) in the northern and central GBRMP, respectively (Figure 5.6 b). Coral cover was also extremely variable within the central and northern GBRMP, ranging from 7.17% at Agincourt Reef to 36.5% at Creech Reef in the northern GBRMP, and from 12.75% at Potter Reef to 43.11% at Castor Reef in the central GBRMP.

Reefs with above average coral richness or cover in each region were Kenn, Mellish, Moore and Boot Reefs for coral richness, and Cato (38.67%), Kenn (32.24%), Mellish (33.29%), Moore (33.03%), and Bougainville (38.44%) Reefs for coral cover. Latitudinal and longitudinal patterns in coral cover also revealed a notable depression in coral cover in the central CSMP (Figure 5.8). Coral cover at individual sites in the central CSMP was often lower than 10%, except for Moore Reef and Mellish Reef where coral cover was high across all individual sites surveyed.

Figure 5.8 Spatial variation in mean coral cover (%) by latitude (left panel) and by longitude (bottom panel) for the 20 surveyed reefs in the Coral Sea Marine Park. Values for each reef are averaged across sites, habitats, and years.

Regional differences in coral cover were particularly apparent when comparing coral cover within specific habitats (Figure 5.9). We would generally expect to find high coral cover (upwards of 40%) on shallow reef crests of the mid- and outer-shelf reefs of the GBRMP due to typically extensive cover of tabular *Acropora* (Wismer et al. 2009; Linares et al. 2011). In the northern and central CSMP coral cover was greater or equal on the reef slope compared to the reef crest, whereas coral cover was generally higher on the reef crest in the southern CSMP (Figure 5.9). This lower coral cover in shallow reef crest habitats vs deeper reef slope

habitats indicates recent disturbance/s may have reduced coral cover in the shallow habitats in the northern and central CSMP. By way of comparison, coral cover on relatively undisturbed reefs (no active outbreaks of crown-of-thorns, bleaching or cyclonic impacts within previous 5-10 years) in the GBRMP is typically 20-40% (Hughes et al. 2018). As coral cover declines below 20%, and especially <10%, there are expected to be impacts on abundance and diversity of fishes and other reef associated organisms that directly or indirectly rely on corals (Bellwood et al. 2006, 2012; Wilson et al. 2006; Pratchett et al. 2008, 2015; Stella et al. 2011).
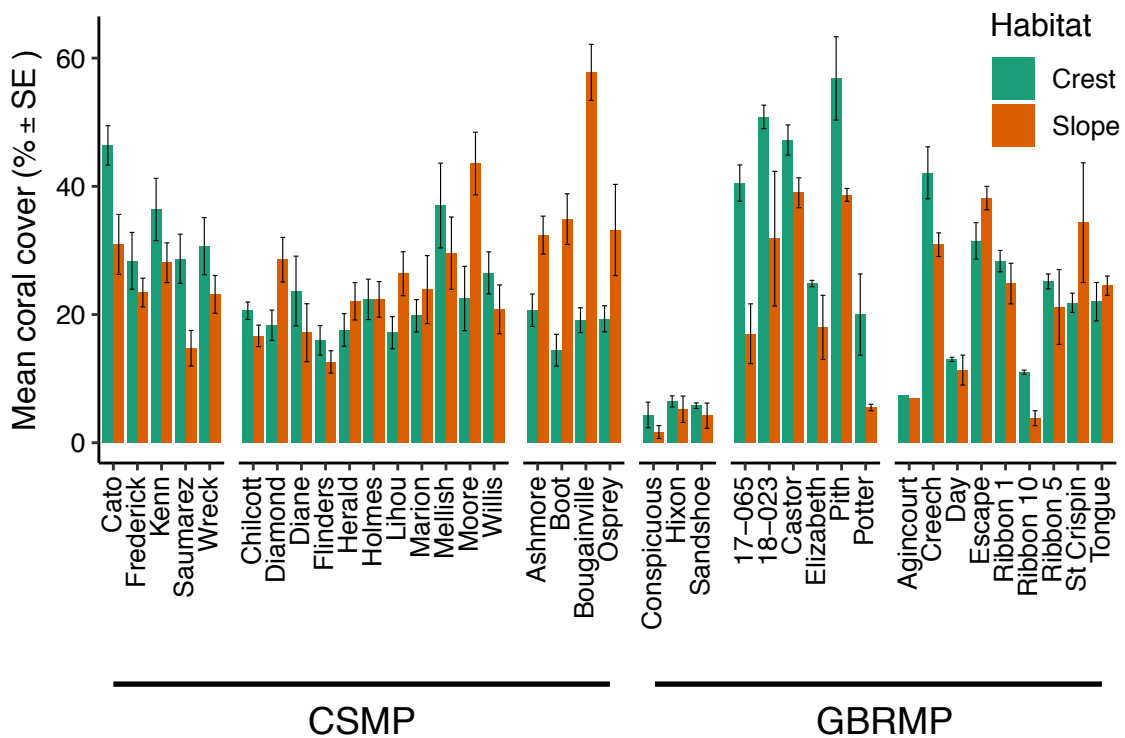
Figure 5.9 Differences in average coral cover (+/- SE) between habitats on 20 reefs in the Coral Sea Marine Park and 18 outer-shelf reefs in the Great Barrier Reef Marine Park. The shallow reef crest (1-3m) habitat is shown in green and the deeper reef slope (8-10m) is shown in orange. Means are based on sites surveyed throughout 2018-2020. Reefs are arranged into three regions (southern, central, and northern) for each of the Coral Sea Marine Park and Great Barrier Reef Marine Park.

Coral cover was relatively consistent among the 3 years of the surveys for each of the CSMP regions and reefs (Figure 5.10), however differences in the reefs and/or sites within each reef that were sampled in each year preclude any robust

comparisons. Although coral cover appeared to increase at some reefs from 2018 to 2020 (e.g., Coringa-Herald Islets and Cays and Holmes Reefs), and decline at others from 2018 to 2019 (e.g., Saumarez and Osprey Reefs; Figure 5.10), these changes are likely to reflect, at least to some extent, variation among sites rather than robust temporal change. Importantly, for those sites that were surveyed in multiple years there was no evidence for a decrease in coral cover, with the cover of live corals either remaining the same or increasing slightly. This is likely to change over the next 12 months as coral mortality due to the 2020 bleaching event (see Section 5.7 below) is realised.
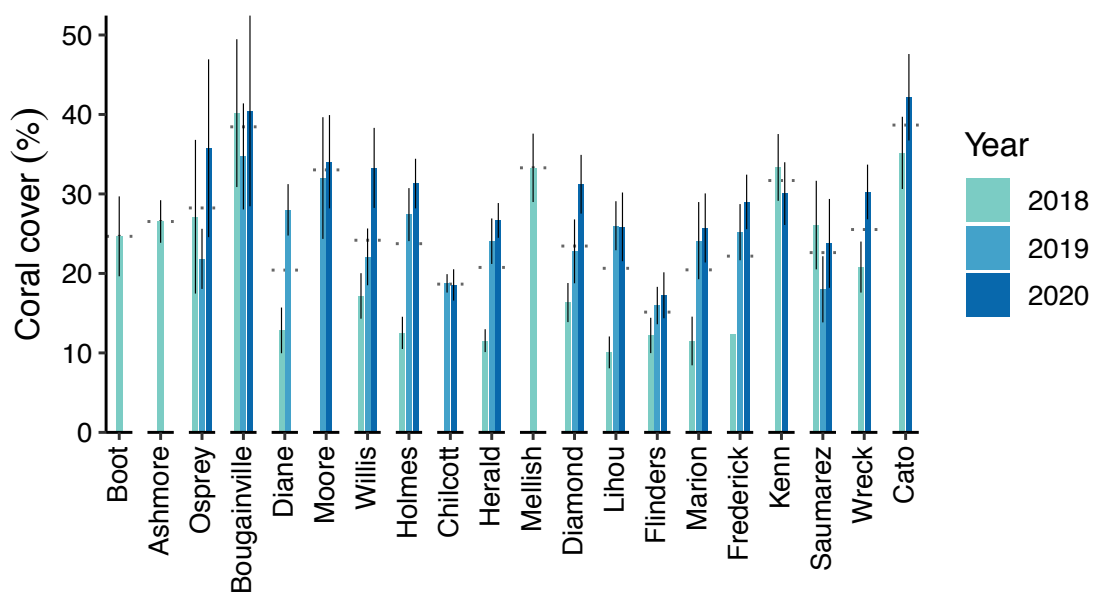
Figure 5.10 Temporal variation (2018-20) in average coral cover (+/- SE) on 20 reefs in the Coral Sea Marine Park. Values for each reef are averaged across sites and habitats. NB: the same sites within each reef were generally not surveyed in each year

5.4 Macroalgal abundance

Macroalgal cover - Across the 3-year period (2018-2020), average cover of macroalgae recorded across the CSMP was 8.74% (±0.36 SE) compared to 2.13% (±0.29 SE) on the outer-shelf GBRMP reefs (Figure 5.11). Although macroalgal cover was approximately 4-fold greater on reefs in the CSMP than GBRMP, this was mostly attributed to the higher cover of the green calcifying algae *Halimeda* spp. There were also marked differences in *Halimeda* cover among regions (Figure 5.11 a), with very low cover (ca. 1%) in the northern CSMP, low cover in the southern CSMP (2-6%) and moderate but variable cover (4-23%) in the central CSMP (Figure 5.11 a). *Halimeda* is a common feature of oceanic reefs where it often forms thick curtains on steep slopes and overhangs and is an important contributor to calcification and production of reef sediments (Drew 1983). Unlike many large canopy-forming algae, such as *Sargassum,* that predominate on coastal reefs of the GBRMP and elsewhere (e.g., Wismer et al. 2009; Hoey and Bellwood 2010; Rasher et al 2013), high abundances of *Halimeda* is not considered to be symptomatic of reef degradation. Cover of all other macroalgae was relatively low across all regions of the CSMP and the outer reefs of the GBRMP (Figure 5.11 b). The only exception was the sheltered back-reef habitats at Saumarez Reef, where we observed a moderate abundance of *Caleurpa*, a green algae with distinct creeping runners. The higher abundance of *Caulerpa* (ca. 7%) on Saumarez Reef may be related to numerous factors, such as local increases in nutrients (e.g., through upwelling) and/or reduced herbivory, or may reflect site specific differences. Identifying the drivers of the higher cover of *Caulerpa* at Saumarez Reef would require further investigation, however the current levels are relatively low compared to other oceanic reefs, such as Elizabeth and Middleton Reefs, and Lord Howe Island to the south (Hoey et al. 2011, 2018) and not cause for concern.

(a) *Halimeda*
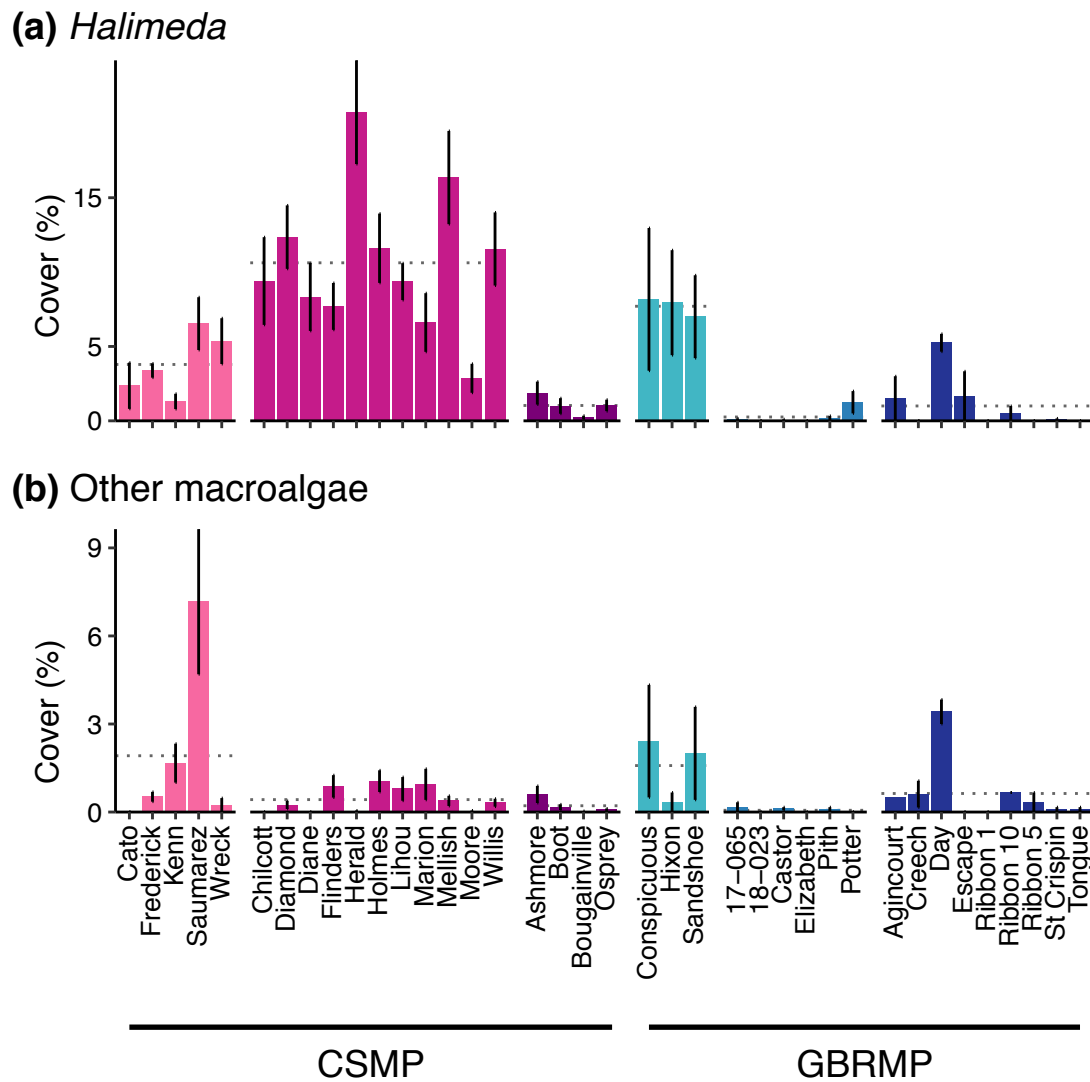
(b) Other macroalgae

Figure 5.11 Variation in the cover (±SE) of a) *Halimeda* and b) other macroalgae among the 20 reefs surveyed in the Coral Sea Marine Park and 18 outer-shelf reefs surveyed in Great Barrier Reef Marine Park during 2018-2020. Reefs are arranged into three regions (southern, central, and northern) for each of the Coral Sea Marine Park and Great Barrier Reef Marine Park and coloured by *a priori* regional assignments (following Figure 4.1).

5.5 Coral reef fishes

**Abundance and Diversity -** A total of 331,214 fishes were recorded across 163 sites throughout the 3-years of surveys. Within the CSMP both the richness and density of reef fishes were generally highest in reefs in the northern CSMP and decreased with increasing latitude (Figure 5.12 a). Regional species richness of reef fishes ranged from an average of 86 species per site in the northern CSMP to

69 and 59 species per site in the central and southern CSMP, respectively (Figure 5.12 a). The most diverse fish communities were recorded at Boot Reef in the northern CSMP with an average of 101 species per site, almost double that of Wreck Reef (52 species per site) in the southern CSMP. Similarly, regional averages in fish densities were up to 3-fold higher in the northern CSMP (244 individuals per 100 m^2) compared to the central (161 individuals per 100 m^2) and southern CSMP (81 individuals per 100 m^2; Figure 5.12 b). Despite these regional trends in fish species richness and density, there was considerable variation within each region of the CSMP, with Mellish and Moore Reefs supporting fish communities with markedly higher species richness and density than those of other central CSMP reefs. Similarly, in the northern CSMP, fish species richness was considerably lower at Osprey Reef, and fish density higher at Bougainville Reef than at other reefs within the northern CSMP (Figure 5.12 a,b). The higher diversity of reef fishes in the northern CSMP is consistent with well-known latitudinal gradients in the diversity of marine species (Hillebrand 2004) and reef fishes (Bellwood and Hughes 2001), though this does not explain why Mellish Reef had such high abundance and diversity of fishes compared to reefs in the central CSMP, and also does not explain the large difference in fish densities between the northern CSMP and regions further south (Figure 5.13, Figure 5.14).

Reef fish species richness on outer-shelf reefs in the GBRMP was slightly higher than those of the CSMP, but did not show any clear latitudinal patterns with the number of fish species recording ranging from 197 species per site in the central GBRMP to 206 species per site in the northern GBRMP (Figure 5.12 a). The mean density of reef fishes on outer-shelf reefs in the GBRMP was broadly comparable to those in the CSMP, although lowest densities were recorded in the central GBRMP (130 individuals per 100 m^2) and greatest densities in the northern GBRMP (210 individuals per 100 m^2).

(a) Reef fish richness

(b) Reef fish density

(c) Reef fish biomass

CSMP

GBRMP
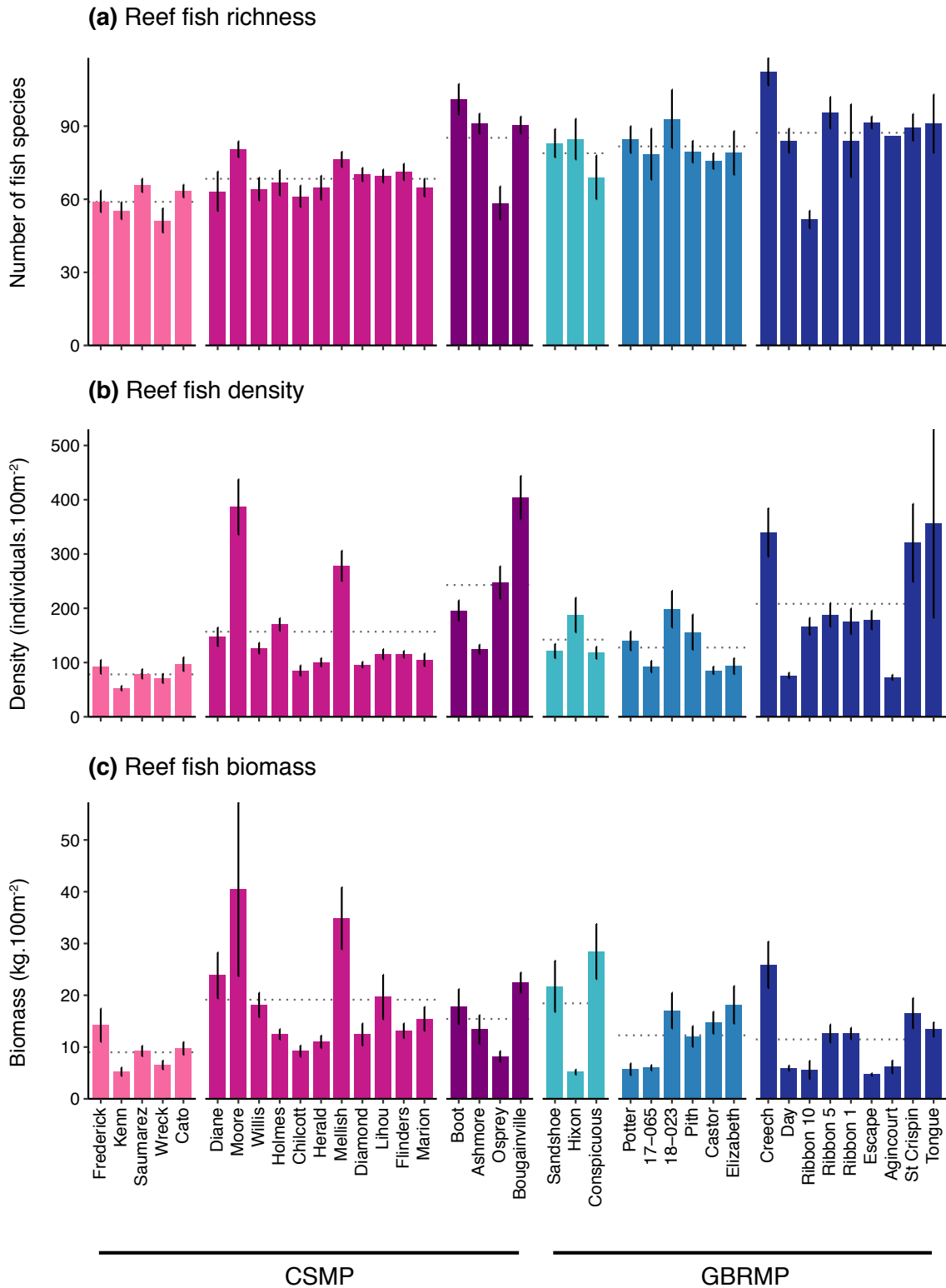
Figure 5.12 Diversity, abundance and biomass of coral reef fishes. Spatial variation in the (a) species richness, (b) abundance, and (c) biomass of coral reef fishes and sharks among the 20 reefs surveyed in the Coral Sea Marine Park and 18 outer-shelf reefs surveyed in Great Barrier Reef Marine Park during 2018-2020. Reefs are arranged into three regions (southern, central, and northern) for each of the Coral Sea Marine Park and Great Barrier Reef Marine Park and coloured by *a priori* regional assignments (following Figure 4.1).
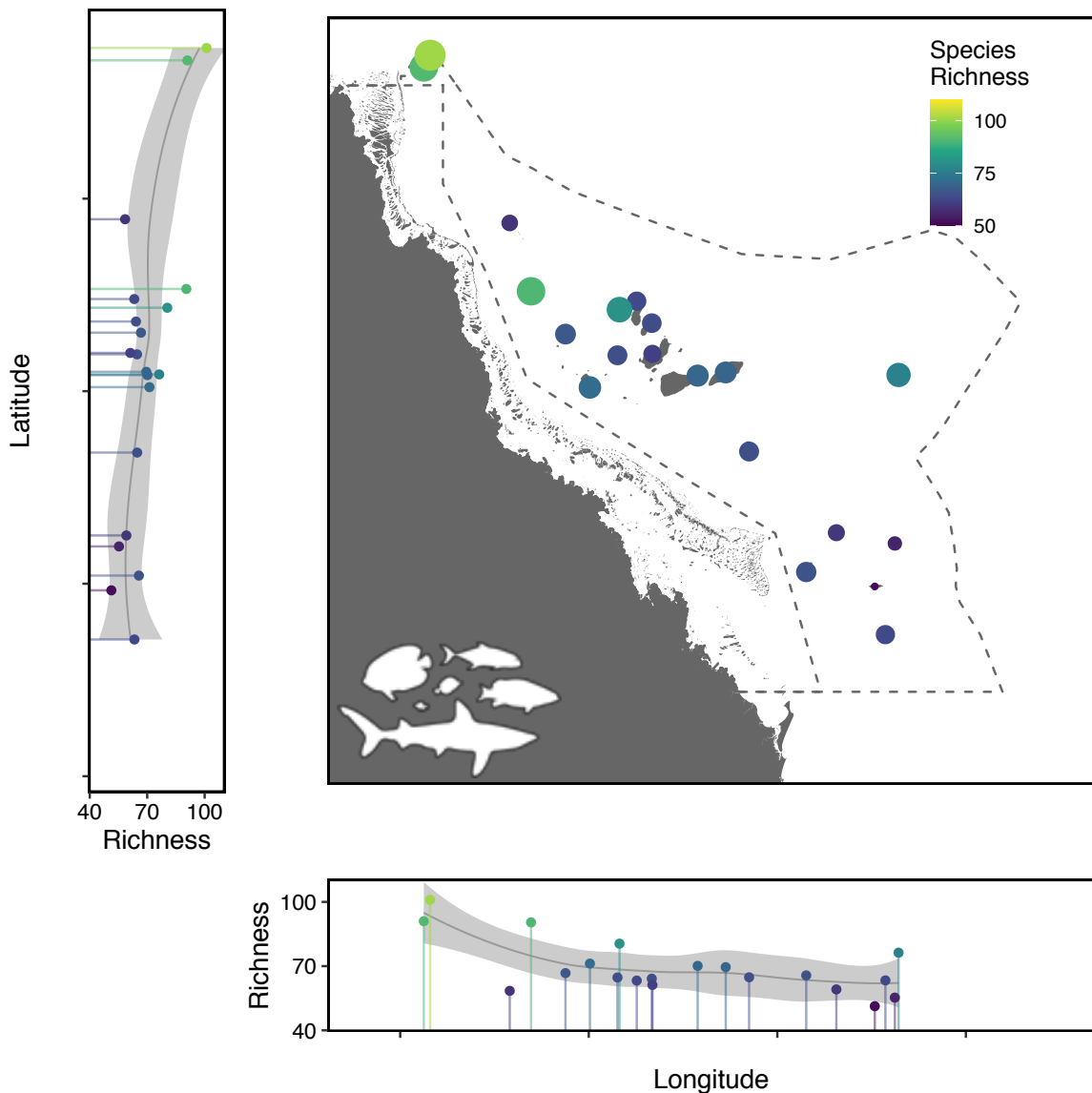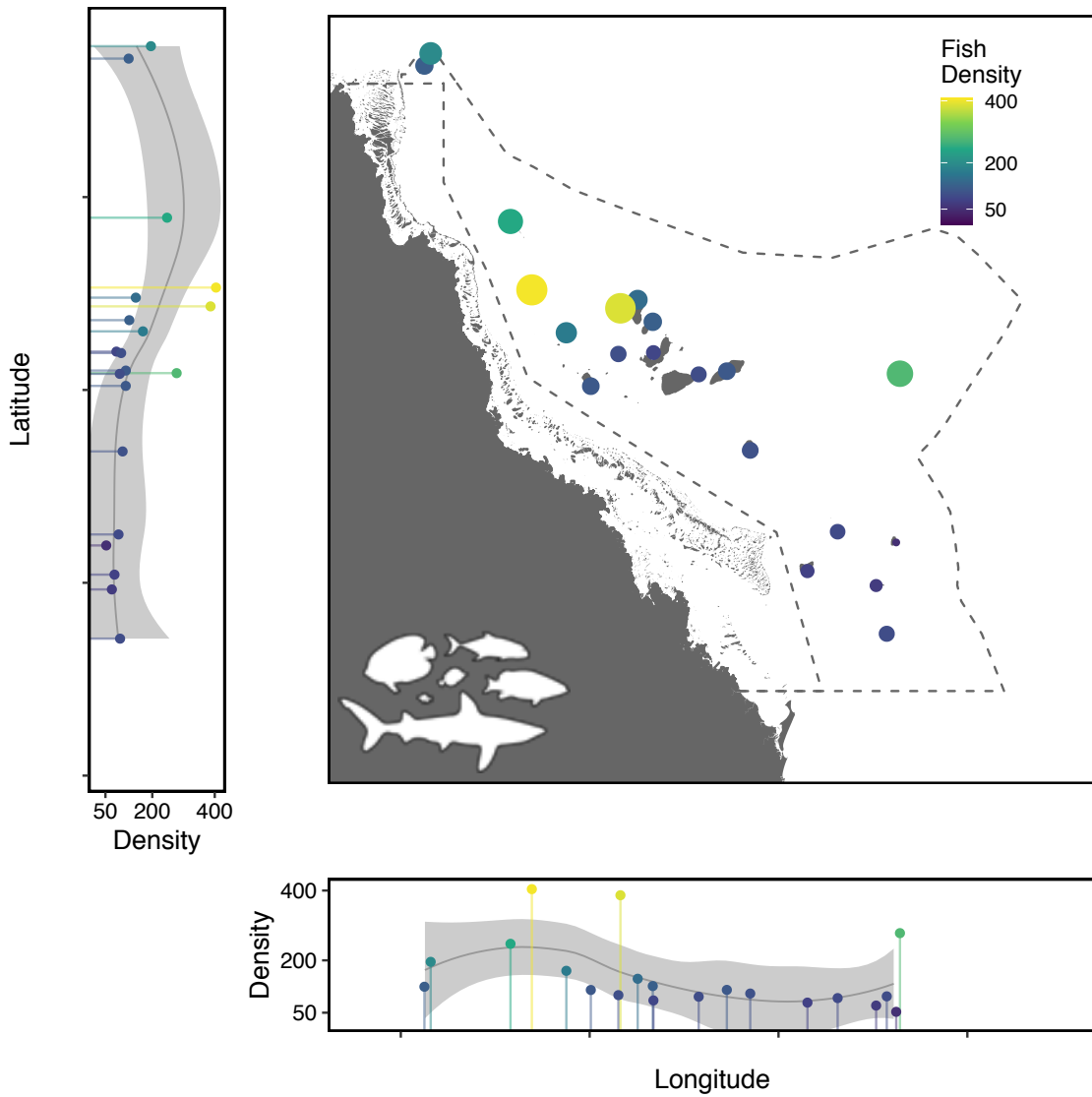
Figure 5.13 Distribution of the species richness of coral reef fishes and sharks (species.reef^{-1}) by latitude (left panel) and by longitude (bottom panel) for surveyed sites in the Coral Sea Marine Park. Values for each reef are averaged across sites, habitats, and years.

Biomass – We estimated the total biomass of sharks and reef fishes from known length-weight relationships for each fish species from Fishbase (Froese and Pauly 2019). In general, the biomass of reef fish and sharks was greater in the central and northern CSMP than the corresponding regions in the GBRMP, although this pattern was reversed in the southern CSMP and GBRMP (Figure 5.12 c). Regional patterns in fish biomass did not reflect latitudinal patterns in species richness or

density and varied widely between reefs within each region, ranging from approximately 5 to 40 kg per 100 m^2 (Figure 5.12 c, Figure 5.15). Reefs with particularly high biomass were Moore Reef (40 kg per 100 m^2), and Mellish Reef (35 kg per 100 m^2), both in the central CSMP. In the case of Mellish Reef, this high biomass is associated with the high densities of reef sharks observed in the surveys, and likely reflects the remoteness, and hence limited fishing, on this reef. Numerous studies have highlighted the significantly greater biomass of sharks and reef fishes on similar remote or highly protected reefs across the Indo-Pacific (e.g., Robbins et al. 2006, Graham and McClanahan 2013). The decoupling of density and biomass is likely related to differences in the size structure of fish assemblages with some reefs likely supporting high densities of relatively small fishes, while other may support a lower density of larger fishes.

Reef fish biomass and, to a lesser extent, reef fish density didn't display clear geographic patterns (i.e., differences among reefs within a region were generally greater than the differences between regions), suggesting that other factors may be influencing fish communities. Indeed, the mean density of reef fish was positively correlated to mean coral cover (r = 0.47) and negatively related to mean macroalgae cover (r = -0.25), but displayed no relationship with topographic complexity across the surveyed reefs. However, we found no relationship between reef fish biomass and coral cover, coral richness, macroalgae cover, or topographic complexity. This is in marked contrast with numerous studies that have found positive relationships between topographic complexity and the abundance and/or biomass of coral reef fish assemblages (reviewed by Graham and Nash 2013). More detailed analyses of potential drivers will be necessary to understand the reasons for the observed patterns in the fish community, and should consider factors reef size, isolation, habitat variables and disturbance history (e.g. Ceccarelli et al. 2016; Lam et al. 2018; Zinke et al. 2018).

Figure 5.14 Distribution of fish density (individuals.reef^{-1}) by latitude (left panel) and by longitude (bottom panel) for surveyed sites in the Coral Sea Marine Park. Values for each reef are averaged across sites, habitats, and years.
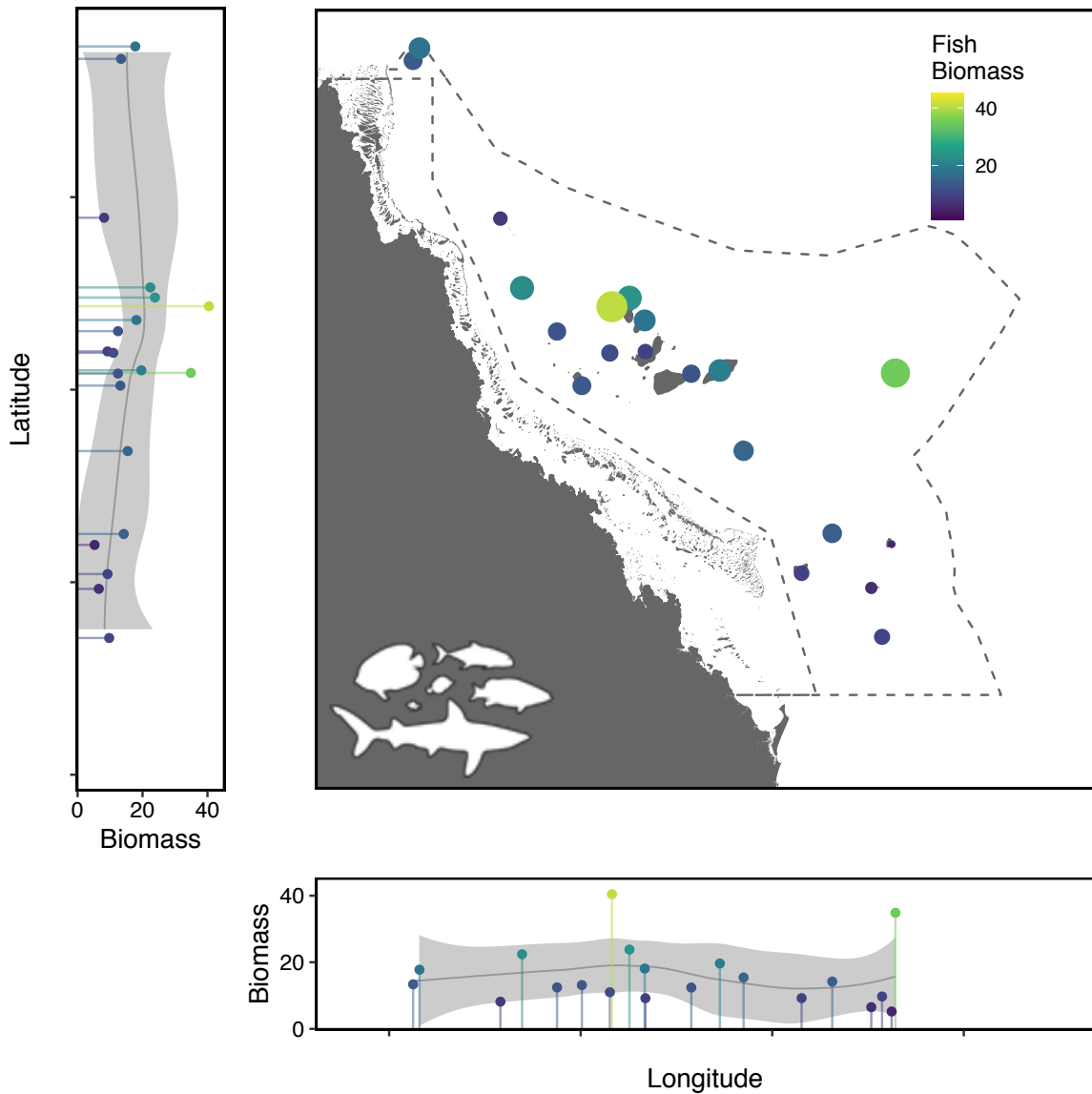
Figure 5.15 Distribution of fish biomass (kg.100^{-2}) by latitude (left panel) and by longitude (bottom panel) for surveyed sites in the Coral Sea Marine Park. Values for each reef are averaged across sites, habitats, and years.

Functional groups - Fishes were categorised into functional species groups based on their diet, morphology and feeding behaviour. Planktivorous fishes (e.g. fusiliers, anthias and some damselfishes) are typically the most abundant functional group on reefs in the CSMP and GBRMP, accounting for up to ~80% of the total fish abundance (Figure 5.16 a). The next most abundant groups were the grazing herbivores (e.g. primarily surgeonfishes and rabbitfishes) that typically feed

on short algal turfs and associated materials (i.e., the epilithic algal matrix; Wilson et al. 2003), and algal farming damselfishes (Figure 5.16 a). Except for some grazing fishes and planktivorous unicornfish (*Naso* spp), the majority of species within these functional groups are small bodied and hence only contribute a relatively small part of the total reef fish biomass (Figure 5.16 b). The only exception to this was Moore Reef in the central CSMP where schools of large-bodied *Naso caesius* and large schools (up to 200 individuals) of *Pterocasio tile* were commonly recorded. Reefs in the CSMP were typically characterised by a high biomass of piscivores (e.g., groupers, snappers and sharks), particularly in the central CSMP where the biomass of piscivores averaged almost 40 kg per 100 m^2, approximately 6-fold greater than our estimates from outer-shelf reefs in the central GBRMP (~7 kg per 100 m^2).
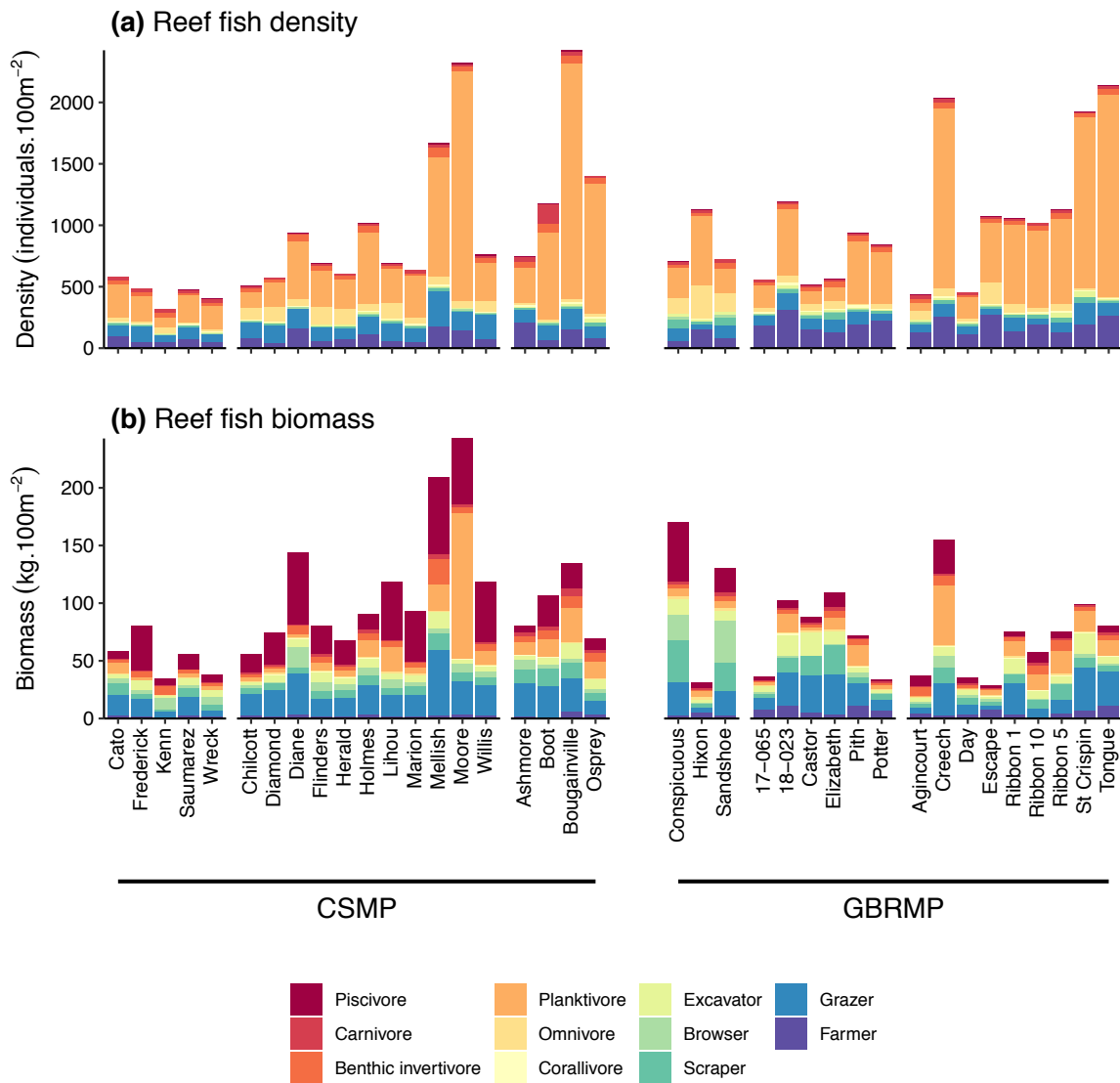
Figure 5.16 Variation in the functional composition of reef fish assemblages across the Coral Sea Marine Park and Great Barrier Reef Marine Park based on (a) density and (b) biomass. Values for each reef are averaged across habitats, sites and years.

5.6 Other reef taxa

Sea snakes – Sea snakes were regularly observed on all reefs in the southern CSMP and at Marion Reef, the southernmost reef of the central CSMP, but were not observed at any other reefs in the central or northern CSMP (Figure 5.17). This marked latitudinal gradient is similar to that reported on the GBRMP using baited remote underwater video station (BRUVS) sampling (Udyawer et al. 2014), with the highest prevalence and diversity of sea snakes occurring in central and southern GBRMP. These marked latitudinal gradients in the distribution and diversity of

Aipysurus sea snakes, which comprises the genera *Aipysurus* and *Emydocephalus* (e.g., Lukoschek et al. 2007) are generally attributed to the limited thermal tolerance of these species (Heatwole et al. 2012). If so, the northern limits of sea snakes recorded in the CSMP (at Marion Reef) could contract southwards with ongoing ocean warming. Importantly, there are global concerns about the conservation of sea snakes following the apparent disappearance of sea snakes at some locations where they were formerly abundant (Lukoschek et al. 2007, 2013), possibly due to changing environmental conditions or direct human pressures. This highlights the need to carefully monitor sea snake populations in the southern CSMP.

Aside from striking latitudinal limits, densities of sea snakes were highly variable within and among reefs in the southern CSMP. Similar small-scale variability was reported in the southern GBRMP (Lukoschek et al. 2007), where sea snakes (*A. laevis* and *E. annulatus*) were recorded more often on larger reefs, and mainly within sheltered reef habitats. The highest densities of sea snakes recorded in the CSMP were at Kenn Reef (1.9 snakes per hectare), moderate densities at Marion, Frederick and Saumarez Reefs (0.5-1.1 snakes per hectare), and lowest at Wreck and Cato Reefs (~0.2 snakes per hectare). There is very limited comparable data on the abundance of sea snakes, whereby most studies (e.g., Udyawer et al. 2014; Lukoschek et al. 2007) report only prevalence, rather than density estimates.

The olive sea snake (*Aipysurus laevis)* was the most abundant species recorded in the CSMP, accounting for >95% of all individuals observed during our surveys. Similarly, *A. laevis* accounted for 77% of sea snakes recorded during BRUVS sampling in the southern and central GBRMP (Udyawer et al. 2014). Other species recorded during surveys in the CSMP were *Acolyptophis peronii* and *Emydocephalus annulatus.* Both, *A. laevis* and *E. annulatus* are strongly associated with coral reef habitats (Lukoschek et al. 2007), whereas most other tropical sea snakes occur predominantly in deep water, inter-reef environments. Given our surveys were intentionally restricted to shallow reef habitats, it is likely that other species of sea snakes do occur in the CSMP, but in habitats that were not surveyed. No sea snakes were observed during surveys conducted in this study on 18 outer-shelf reefs of the GBRMP, though most of the sampling was

conducted in the northern GBRMP, which are known to be depauperate in terms of occurrence and diversity of sea snakes (Udyawer et al. 2014). Udyawer et al. (2014) also showed that sea snakes were more prevalent in inner and mid-shelf reefs, rather than outer-shelf reefs.
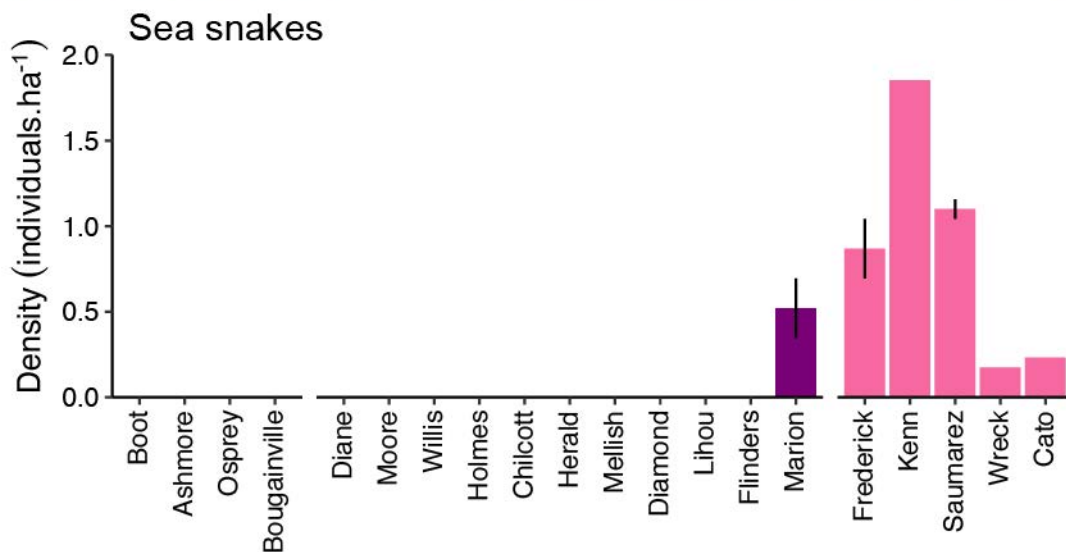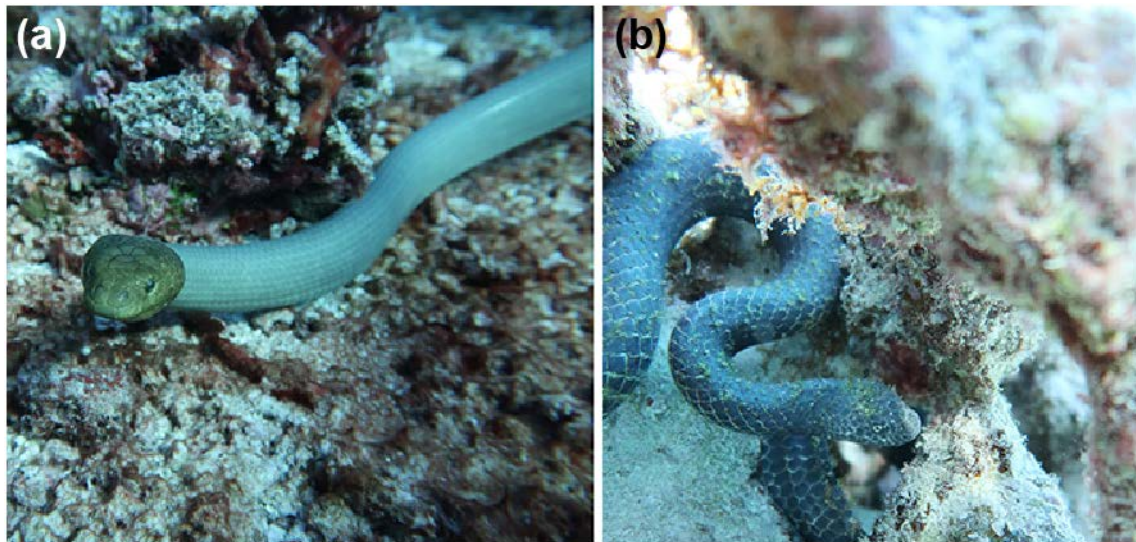
Figure 5.17 Spatial variation in the abundance of sea snakes, including a) *Aipysurus laevis,* b) *Acolyptophis peronii and Emydocephalus annulatus* (not shown) among the 20 reefs surveyed in the Coral Sea Marine Park during 2018-2020. Image credit: Deborah Burn

Giant Clams – A total of 2,028 giant clams were recorded in the 2018-2020 surveys. The density of giant clams (*Tridacna* spp.) was relatively consistent across the CSMP with < 1.7 clams per hectare (equivalent to 1.7×10^{-4} clams.m^{-2}) being recorded on most reefs (Figure 5.18 a). The only exception to this was Kenn

Reef in the southern CSMP where an average of 8.3 clams per hectare were recorded. The causes of the 6-fold greater densities of giant clams at Kenn Reef are difficult to reconcile but may be related to a chance recruitment event, and/or high levels of self-recruitment at this reef. Densities of giant clams were also notably low at Boot Reef and, to a lesser extent, Ashmore Reef, in the far northern CSMP. The overall mean density of giant clams was within the range of densities recorded throughout the Indo-Pacific (1×10^{-5} up to 1×10^{-3} clams.m^{-2}) by bin Othman et al. (2010). bin Othman et al. (2010) noted that densities of giant clams are highly supressed on reefs subjected to over-exploitation and some species (mainly *H. hippopus* and *T. gigas*) are functionally extinct in worst affected areas. Effects of fishing are also being compounded by environmental change, where mass-mortalities of giant clams have occurred following extreme temperature anomalies (e.g., Andréfouët et al. 2013).

The vast majority of giant clams recorded in the CSMP were *Tridacna maxima* and *Tridacna squamosa*, collectively accounting for 1,900 (94%) of the 2,028 clams recorded. The other species recorded (which were much less abundant) were *Tridacna derasa* (64 individuals, 2.3%), *Hippopus hippopus* (42 individuals, 1.5%), *Tridacna crocea* (19 individuals, 0.7%)*, and *Tridacna gigas* (3 individuals, 0.1%). The relative abundance of giant clam species surveyed, and the lack of the largest species *T. gigas* in particular, is attributable mainly to the specific habitat that we surveyed; *Tridacna gigas* is typically most abundant in lagoonal and shallow reef flat habitats (e.g., Brayley 1987) and would require dedicated surveys in these habitats to assess spatial and temporal changes in their abundances.

The average shell length of giant clams was relatively consistent among reefs and regions (approximately 200mm), the only exception being at Ashmore Reef where average shell length was 250mm (Figure 5.19). The size structure of giant clams reflects the predominance of smaller species (*T. maxima*: maximum size <40cm; *T. squamosa:* maximum size 45cm; Raymakers et al. 2003; van Wynsberge et al. 2017), and the lack of the largest species, *T. gigas* (maximum size 140cm) in the surveys. The apparent lack of giant clams on reefs in the northern CSMP may be attributable to harvesting, though clams are also vulnerable to elevated temperatures (e.g., Andréfouët et al. 2013), and it would be expected that the

northernmost reefs (Ashmore and Boot Reefs) are exposed to generally higher temperatures than reefs to the south.
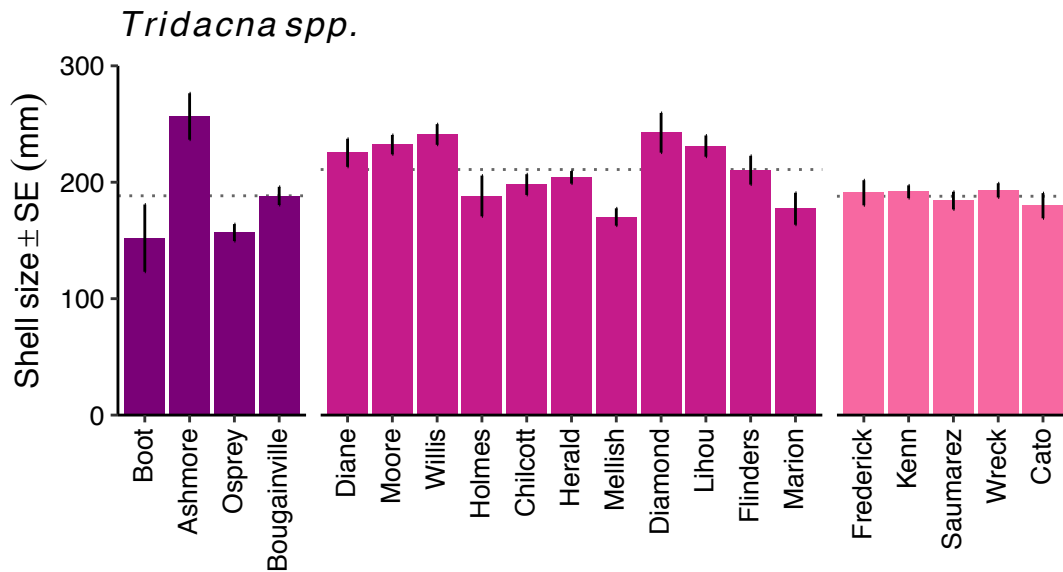
(a) Giant clams (*Tridacna* spp.)
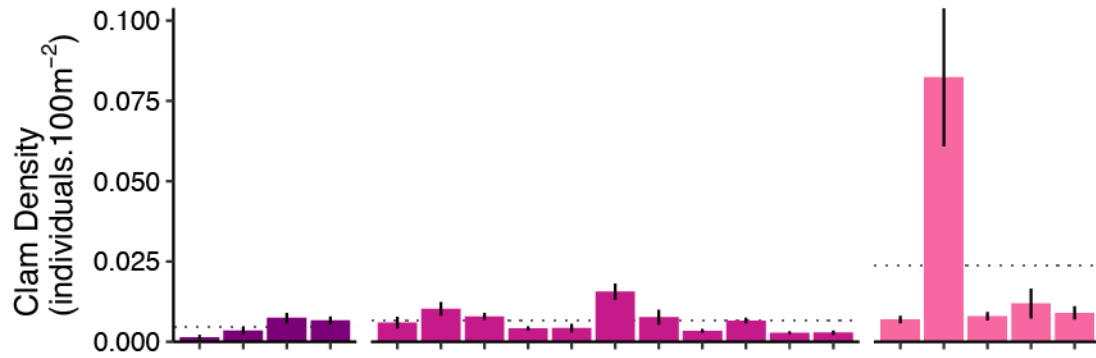
(b) Trochus (*Tectus* spp.)
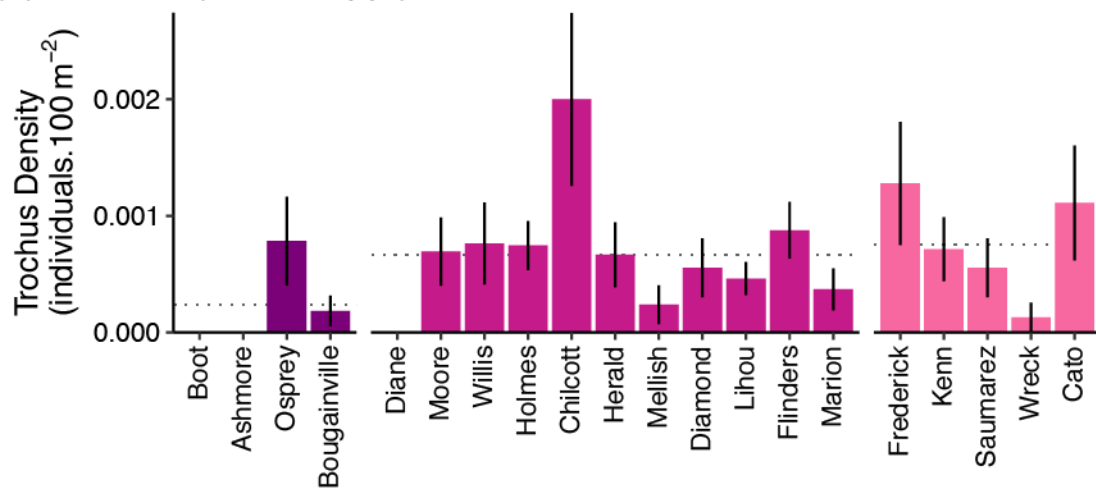
Figure 5.18 Spatial variation in the abundance of (a) giant clams, and (b) Trochus among the 20 reefs surveyed in the Coral Sea Marine Park during 2018-2020. Dotted lines represent the mean regional abundance.

Figure 5.19 Differences in the mean shell size of giant clams (*Tridacna* spp) among the 20 reefs surveyed in the Coral Sea Marine Park during 2018-2020. Dotted lines represent the regional mean sizes.

Trochus – The density of *Tectus* spp (formerly *Trochus*) was generally low (<0.2 individuals per hectare) and highly variable among reefs in the CSMP (Figure 5.18 b). The highest densities of *Tectus* were recorded at Chilcott Islet (Coringa-Herald Islets and Cays) (0.2 individuals per hectare) in the central CSMP, while *Tectus* were completely absent from Boot and Ashmore Reefs in the northern CSMP and Diane Bank in the central CSMP. The lack of historical data on the abundances of *Tectus* throughout the CSMP preclude any assessment of temporal change, however the complete absence of *Tectus* on Boot and Ashmore Reefs may be related to their proximity to the boundaries of the Torres Strait trochus fishery. The Torres Strait trochus fishery is a small commercial and subsistence fishery that operates throughout the islands of Torres Strait (D'Silva 2001). Annual landings approached 1,000 tonnes in the early 1900's, but declined to 2-20 tonnes in the late 1990's. Although these landings are relatively small, the potential effects of fishing on trochus populations on Ashmore and Boot Reefs warrants investigation and would facilitate greater collaboration and engagement with the Meriam people, the traditional owners of this sea country.

Sea urchins – The density of long-spined sea urchins (*Diadema* spp) was generally low (< 0.1 urchins per 100m^2) across the CSMP and GBRMP (Figure 5.20 a). The only exception to this were reefs in the southern CSMP where densities of *Diadema* spp. ranged from 0.35 urchins per 100m^2 at Frederick Reef

to 80.9 urchins per 100m^2 at Kenn Reef. The greater densities of *Diadema* in the southern CSMP may reflect latitudinal patterns in abundance with similar densities of *Diadema* (85.5 urchins per 100m^2) being recorded at Middleton Reef (Hoey et al. 2018), however the differences in *Diadema* densities among reefs in the southern CSMP warrants further investigation.

Many sea urchin species (including *Diadema* spp) are herbivorous, and together with herbivorous fishes can help to prevent the expansion and overgrowth of macroalgae on coral reefs (e.g., McClanahan et al 1994; Humphries et al. 2020). For example, the mass mortality of *Diadema antillarum* in 1983 triggered regime shifts to macroalgal dominance on many Caribbean coral reefs (Hughes et al. 1987, Hughes 1994). *Diadema* and some herbivorous fishes (i.e., parrotfishes) also bioerode carbonates from the reef when feeding (e.g., Hoey and Bellwood 2008). Unlike parrotfishes that bioerode the external surfaces of the reef, sea urchins such as *Diadema* are internal eroders and when present in high densities can destabilise the reef framework and result in net erosion of reef carbonates (Glynn et al. 1979; Eakin 1996).

Sea cucumbers – A total of 685 sea cucumbers (Holothuroidea) from 17 species were recorded in the 2018-2020 surveys. The most abundant species were *Holothuria atra* (20.1%), *Thelenota ananas* (19.6%), *Actinopyga mauritiana* (16.2%), and *Stichopus chloronotus* (14.2%). The density of sea cucumbers on reefs was generally low across the CSMP (average 0.38 individuals per 100m^2) ranging from 0.07 individuals per 100m^2 at Moore Reef to 0.86 individuals per 100m^2 at Cato Reef (Figure 5.20 b). These density estimates are substantially lower than those of previous dedicated sea cucumber surveys in the central CSMP that recorded a mean density of 1.33 individuals per 100m^2 for all species combined, and 1.06 individuals per 100m^2 for *H. atra* (Skewes and Persson 2017). The differences between these surveys may be related to the habitats surveyed, rather than a significant decline in sea cucumber populations. The surveys in the present study were conducted over hard reef substrate, due to the primary focus on corals and reef health, and thereby are not ideally suited to assess the population status of taxa that are predominantly found over sand, such sea

cucumbers. Robust assessments of sea cucumber populations would require dedicated surveys over these sandy habitats (*sensu* Kinch et al. 2008).

The density of sea cucumbers on the GBRMP reefs was generally greater than on CSMP reefs, but was highly variable among reefs within each region. These patterns are consistent with estimates of the density and richness of sea cucumbers from several previous studies (Oxley et al. 2003, 2004; Ceccarelli et al. 2008, 2009) and may reflect the greater exposure of CSMP reefs to cyclones and oceanic conditions or the generally steeper reef slopes in the CSMP.
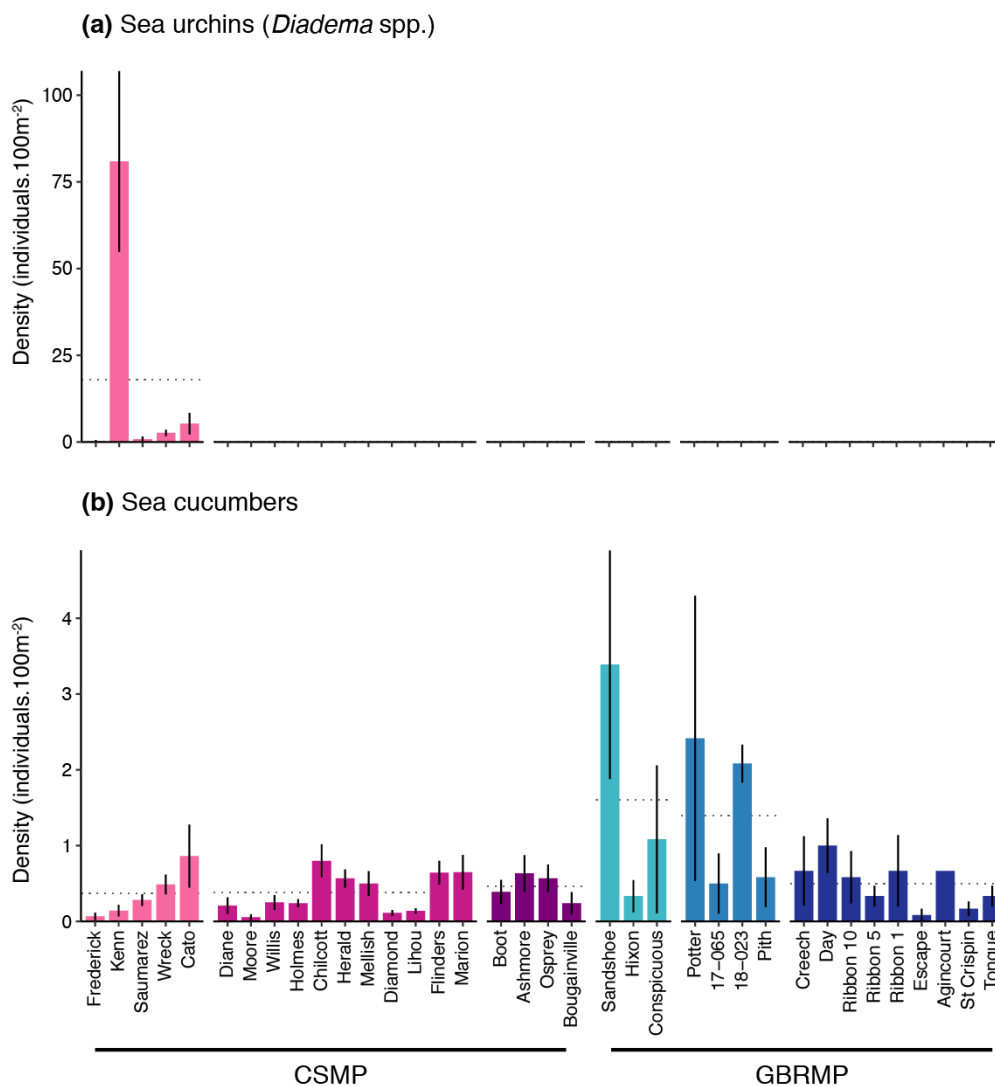
Figure 5.20 Spatial variation in the abundance of (a) sea urchins, and (b) sea cucumbers among 20 reefs in the Coral Sea Marine Park and 18 outer-shelf reefs in the Great Barrier Reef Marine Park during 2018-2020. Dotted lines represent the mean regional abundance. Note that sea urchins (*Diadema* spp) were extremely rare or absent from reefs in the central and northern Coral Sea Marine Park, and all regions of the Great Barrier Reef Marine Park

5.7 Coral health and injury

Coral colony size distribution – Coral assemblages within the CSMP and GBRMP were dominated by relatively small (< 20cm in diameter) coral colonies that accounted for >50% of all coral colonies observed, with few colonies larger than 40cm (Figure 5.21). The only exceptions to this were Bougainville and Mellish Reefs and Diane Bank that had higher proportions (i.e., >50%) of larger (>20cm diameter) corals. These reefs also had the highest coral cover in the CSMP, which may be indicative of older, more established coral communities and less frequent and/or less severe disturbances in recent years. Notably, larger (>20cm diameter) coral colonies were extremely rare (<10%) in the southern GBRMP and likely reflects the recent loss of large colonies due to an ongoing outbreak of crown-of-thorns starfish in that region.
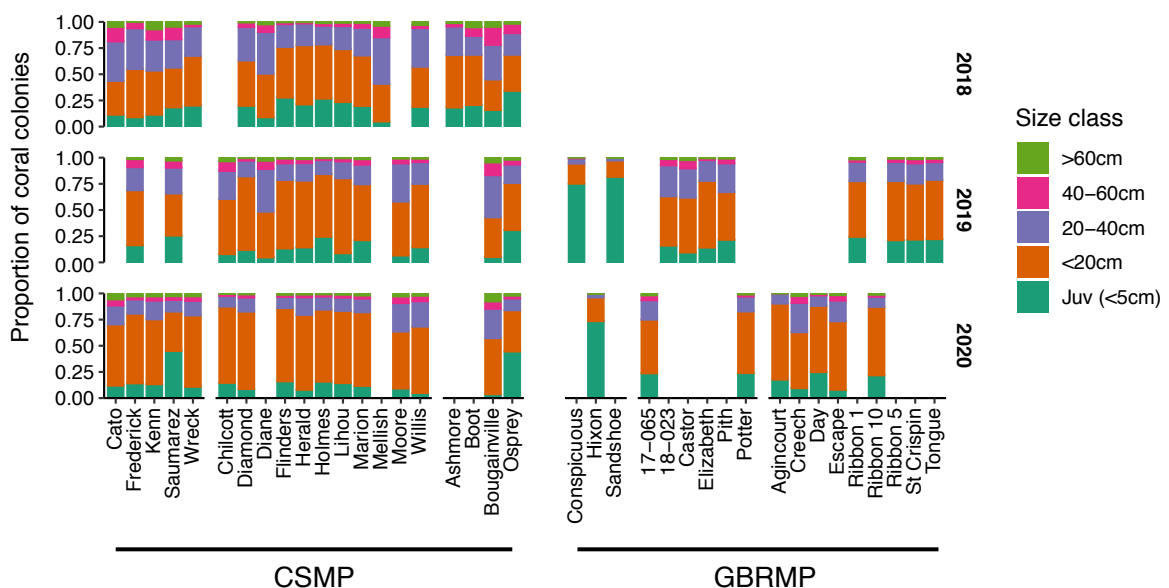
Figure 5.21 Size distribution of coral colonies at 20 reefs within the Coral Sea Marine Park and 18 outer-shelf reefs within the Great Barrier Reef Marine Park between 2018 and 2020.

Coral condition – While bleaching was recorded on reefs in the central CSMP in 2016 and the central and southern CSMP in 2017 (Harrison et al. 2018), the majority of coral colonies surveyed across the CSMP in 2018 and 2019 were healthy indicating no major disturbances had affected the CSMP reefs during that

period (Figure 5.22). Coral injuries associated with predation (e.g. *Drupella* and crown-of-thorns starfish) and localised bleaching at some reefs (i.e., Holmes and Lihou Reefs in 2018) ranged between 7% and 26% in 2018 and 2019, and is likely within the natural range of coral injury for coral reef systems. However, severe and widespread coral bleaching was detected throughout the CSMP in 2020 with the timing of our surveys coinciding with an extended period of elevated water temperatures throughout the region.
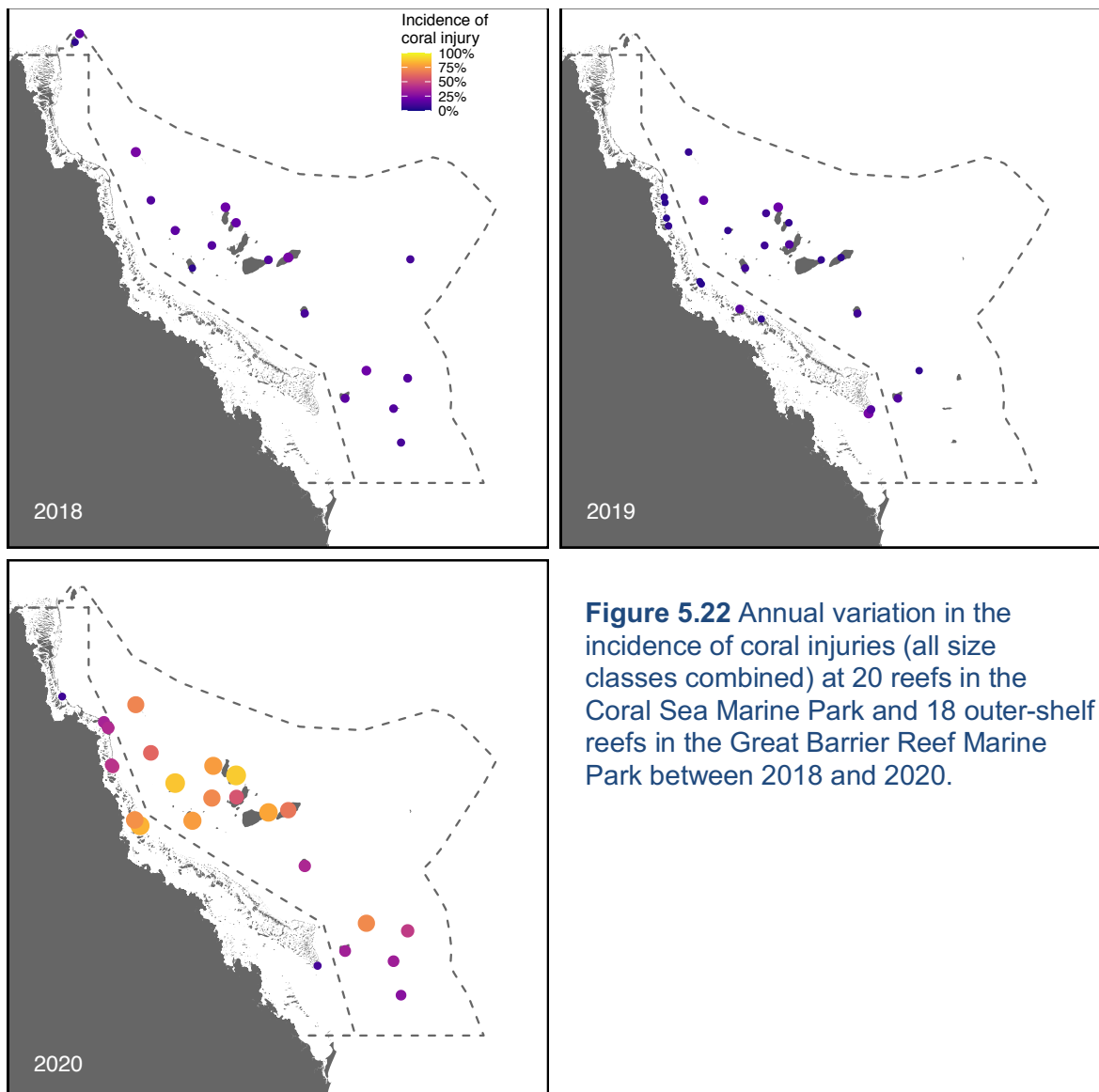

Figure 5.22 Annual variation in the incidence of coral injuries (all size classes combined) at 20 reefs in the Coral Sea Marine Park and 18 outer-shelf reefs in the Great Barrier Reef Marine Park between 2018 and 2020.

2020 Coral Bleaching Event - Coral bleaching was recorded throughout the CSMP and offshore reefs in the GBRMP during February to March 2020. Detailed surveys of coral communities at 16 reefs in the CSMP and 8 outer-shelf reefs in the GBRMP show the severity and geographic extent of the bleaching event. Overall, 63% of observed coral colonies surveyed across the CSMP showed signs of heat stress (Pale – Recently Dead; Figure 5.23) from elevated water temperatures, with the extent of bleaching ranging from 23% of colonies bleached at Cato Reef to 89% of colonies bleached at Willis Islets. Bleaching was also evident on all outer-shelf reefs surveyed in the central and northern regions of the GBRMP, where an average of 40% of coral colonies showed signs of heat stress (Pale – Recently Dead; Figure 5.23). The extent of the bleaching on GBRMP reefs was more variable than recorded for the CSMP ranging from <10% of colonies on Creech and Hixon Reefs to 81% of colonies on Reef 17-065. The 20% of recently dead coral colonies recorded on Hixon Reef were attributed to predation by crown-of-thorns starfish. When interpreting these estimates of bleaching it is important to consider that reefs in the central and southern CSMP and the northern GBRMP had recently experienced losses of bleaching susceptible corals, such as tabular and staghorn *Acropora*, as a result of the 2016 and 2017 bleaching events (Section 5.1; Hughes et al. 2018). As a consequence, these reefs were starting from a different baseline to those prior to the 2016 bleaching event, with a greater relative abundance of coral taxa considered to be less sensitive to elevated temperatures.

Within the CSMP, the central and northern reefs were the most severely affected by the 2020 bleaching event with 72% and 70% of coral colonies bleached or dead at the time of the surveys, respectively (Figure 5.23; 17 February – 11 March 2020). Bleaching was generally lower on the southern reefs within the CSMP, with an average of 40% of coral colonies affected, and ranged from 23% to 64% of coral colonies showing signs of heat stress at Cato and Frederick Reefs, respectively.
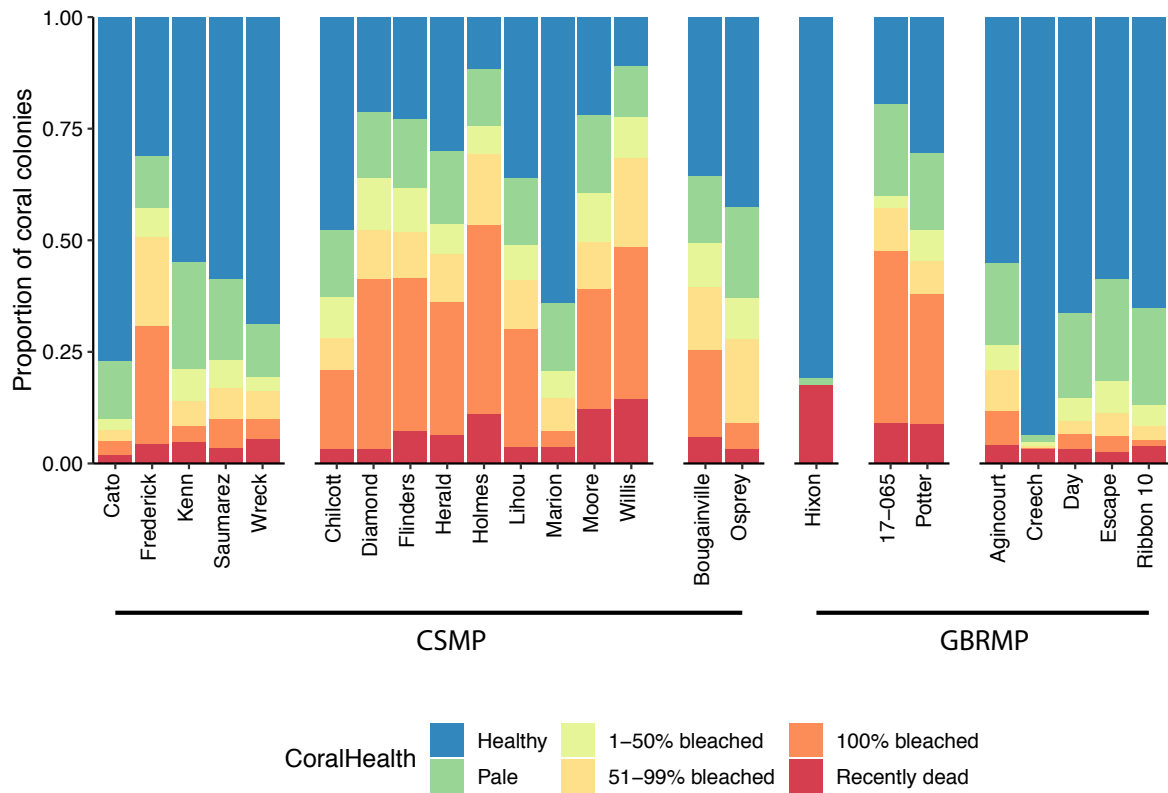
Figure 5.23 The proportion of coral colonies in each of six bleaching categories from 'healthy' to 'recently dead' observed at 16 reefs within the Coral Sea Marine Park and 8 reefs within Great Barrier Marine Park during Feb-Mar 2020.

Importantly, the extent of bleaching recorded was also influenced by the habitat or depth of the survey and the taxonomic composition of the coral assemblage at each site within the CSMP and GBRMP. The proportion of colonies affected by bleaching was lower on the deeper reef slope (8-10m depth) than the shallow reef crest (1-3m depth) across all reefs, and is consistent with numerous previous studies that have recorded declines in bleaching with depth (Marshall and Baird 2000; Muir et al. 2017). The percent of corals showing signs of heat stress (i.e., Pale – Recently Dead) on the reef crest in the central and northern CSMP mostly exceeded 80%, and across the entire CSMP bleaching was on average 28% higher on the shallow crest than on the reef slope (Figure 5.24).
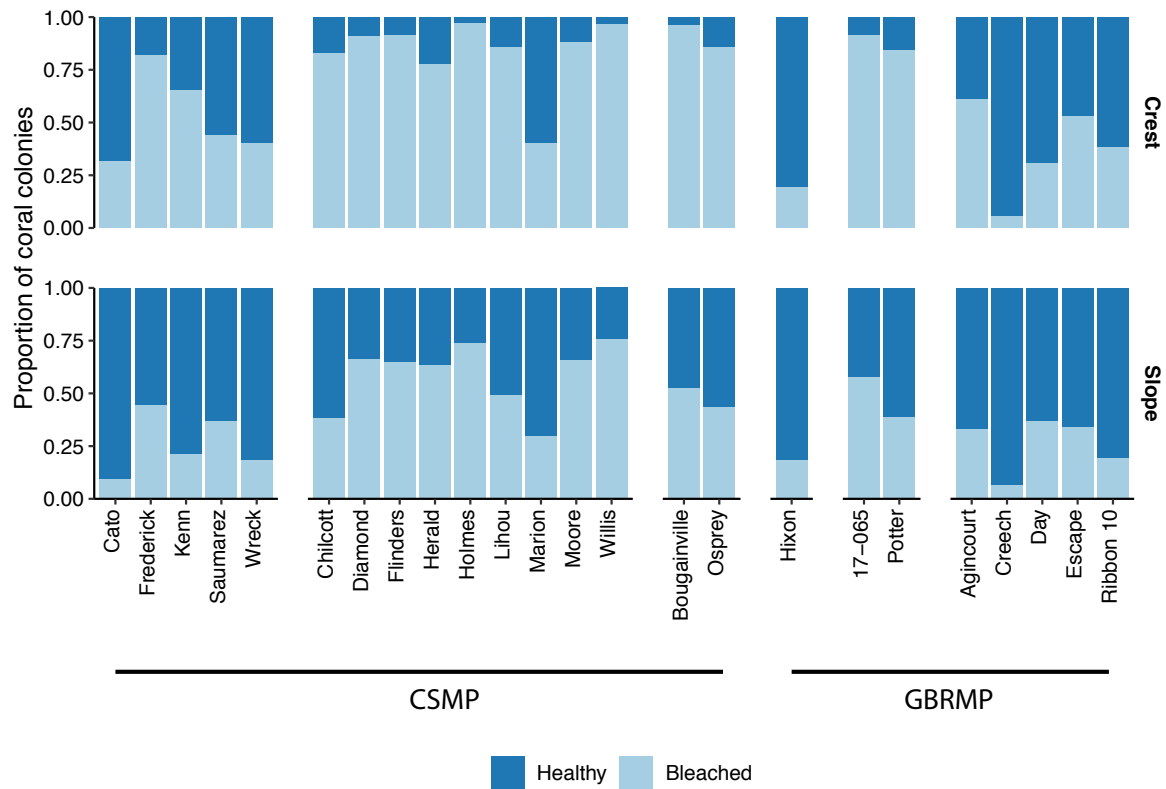
Figure 5.24 The proportion of coral colonies with signs of bleaching (Pale – Recently Dead) on shallow reef crest (1-3m) and reef slope (8-10m) observed at 16 reefs within the Coral Sea Marine Park and 8 reefs within Great Barrier Marine Park during Feb-Mar 2020.

The impact of elevated water temperatures has also been shown to vary considerably among coral taxa, with genera such as *Acropora*, *Pocillopora*, *Seriatopora* and *Stylophora* being most sensitive to increased temperature, while taxa such as *Porites* and *Montipora* are generally more robust (Loya et al. 2001). Our surveys of the 2020 bleaching event in the CSMP also show that *Acropora*, *Pocillopora*, *Seriatopora* and *Stylophora* were among the worst affected coral taxa, however we also recorded a high proportion of colonies of *Porites* (62%) and *Montipora* (61%) were bleached. This coupled with the high incidence of coral mortality across most coral genera (Figure 5.25), and the bleaching of other zooxanthellate organisms (e.g., giant clams, anemones; Figure 5.26) indicates 2020 was a very severe bleaching event. A comparison of the severity and spatial footprint of the 2016, 2017 and 2020 bleaching events in the CSMP show that the 2020 event was more widespread and severe than the two previous bleaching events (Figure 5.27). Moreover, the extent of 2020 bleaching event needs to be

interpreted against a shifted baseline; namely reductions in the cover of bleaching susceptible corals following the 2016 and 2017 bleaching events. It will be critical to re-survey at least some of these sites prior to April 2021 to gain an understanding of the impacts of this event on benthic communities, and the likely impacts on reef fish and invertebrate communities on these reefs.
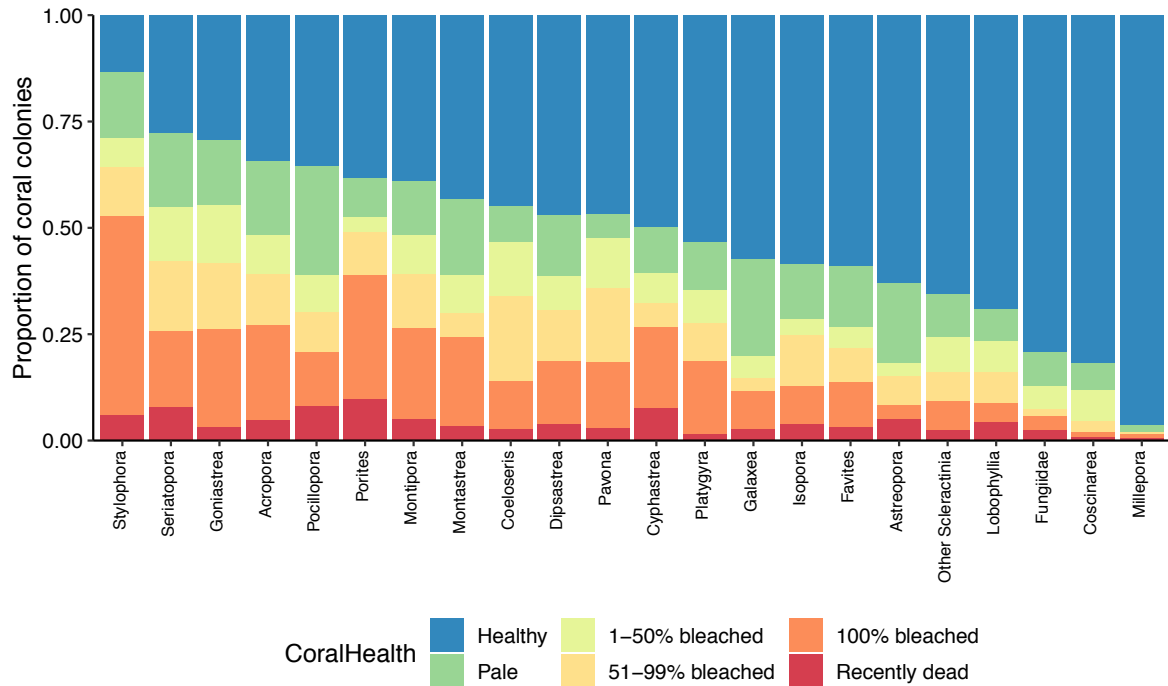
Figure 5.25 The proportion of coral colonies of the 25 most common coral genera in each of six bleaching categories from 'healthy' to 'recently dead' observed at 16 reefs within the Coral Sea Marine Park during Feb-Mar 2020.
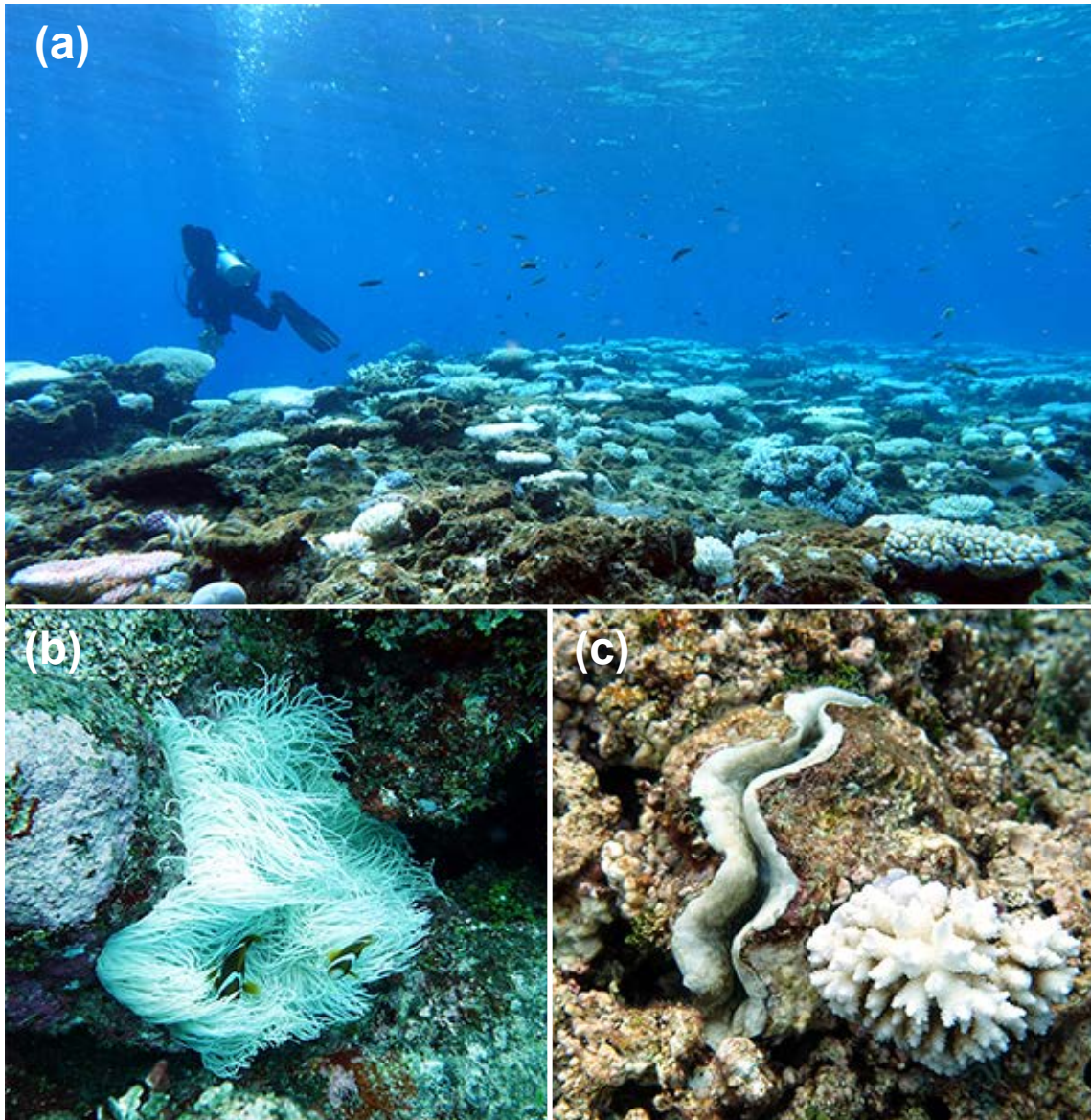
Figure 5.26 Photographs of the 2020 coral bleaching event in the Coral Sea Marine Park. (a) extensive bleaching of coral (mainly *Acropora* spp.) across shallow habitat on Holmes Reefs; (b) a bleached anemone at Flinders Reefs; (c) a bleached clam (*Tridacna* sp) next to a bleached *Acropora* coral colony at Flinders Reefs. Image credits: Dani Ceccarelli