CORAL SEA MARINE PARK CORAL REEF HEALTH SURVEY (2021)

Report on reef surveys February 2021

ARC Centre of Excellence Coral Reef Studies

JAMES COOK UNIVERSITY AUSTRALIA

PRODUCED FOR PARKS AUSTRALIA, JUNE 2021

BY JAMES COOK UNIVERSITY
Corresponding author: Professor Andrew Hoey,
ARC Centre of Excellence for Coral Reef Studies, James Cook University, Townsville QLD 4811.
ANDREW.HOEY1@JCU.EDU.AU | (07) 4781 5979 | (07) 4781 6722

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 thirteen reefs in the Coral Sea Marine Park.

On the cover – Extensive mortality of corals across shallow habitat on Holmes Reefs, central Coral Sea. Photograph taken by Andrew Hoey

Suggested citation: Hoey AS, Harrison HB, McClure EC, Burn D, Barnett A, Cresswell B, Doll PC, Galbraith G, Pratchett MS (2021) Coral Sea Marine Park Coral Reef Health Survey 2021. Report prepared for Parks Australia.

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

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

Acknowledgements

This *Coral Reef Health in the Coral Sea Marine Park* research project was jointly funded by the Director of National Parks, Australia and the ARC Centre of Excellence for Coral Reef Studies. The report was prepared by Andrew Hoey, Morgan Pratchett, Hugo Harrison, and Eva McClure, with important contributions by Deborah Burn, Adam Barnett, Ben Cresswell, Gemma Galbraith and Peter Doll.

The authors wish to thank the relevant staff at Parks Australia for managing this project, with particular mention given to Martin Russell (Manager, Coral Sea Marine Park) and Mitchell Baskys (Senior Marine Parks Officer, Coral Sea Marine Park).

The research presented herein was conducted with the full knowledge and support of Parks Australia, in the Australian Government Department of Agriculture, Water and the Environment. The findings and views expressed, however, are those of the authors and do not necessarily represent the views of Parks Australia, the Director of National Parks or the Australian Government.

We are indebted to Rob Benn (owner/skipper) and the entire crew and staff of MV Iron Joy for enabling this work, and successfully avoiding the worst of the conditions associated with Tropical Cyclone Niran.

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. In early 2020 shallow reef habitats across the Coral Sea Marine Park (CSMP) experienced severe and widespread bleaching, with 63% of all corals surveyed across the CSMP, and up to 89% of corals at individual reefs being bleached. James Cook University was commissioned by Parks Australia to assess (i) the latest condition of fish and benthic communities, (ii) the impacts of the 2020 bleaching event on benthic, fish and invertebrate communities, and (iii) gain some understanding of the potential resilience and recovery of corals throughout the CSMP. The project undertook detailed surveys of coral, fish and macro-invertebrate communities and associated reef health at thirteen CSMP reefs in February 2021. Surveys were conducted to provide rigorous quantitative information on temporal (i.e., 2020 vs 2021) and spatial (i.e., among reefs and regions) patterns in (i) cover and composition of corals and macroalgae; (ii) regional patterns of biodiversity; (iii) coral health, injury and recruitment; and (iv) abundance and composition of reef fishes, sea snakes, and ecologically or economically important invertebrates.

The project surveyed 43 sites and over 13 km of reef habitat across 13 reefs in the CSMP, spanning 9 degrees of latitude (~1,300 km) from Osprey Reef in the north (13.8ºS) to Wreck Reef in the south (22.5ºS). The surveys revealed that total shallow water coral cover decreased from 28% in 2020 to 17% in 2021 across the CSMP, a mean decline of 39%. It is also important to consider that the 2020 bleaching occurred against a shifted baseline of coral communities, with the abundance of bleaching sensitive coral taxa being reduced due to previous (i.e., 2016 and 2017) bleaching events. In the absence of other major disturbances these declines in coral cover are almost certainly attributable to the elevated ocean temperatures and subsequent coral bleaching recorded in February-March 2020. There was, however, considerable variation in the decline in coral cover among regions, ranging from a 17% decline on the two northern CSMP reefs, to 39% and 43% on southern and central CSMP reefs, respectively (Figure 1). Similarly, there were substantial differences in the decline in coral cover among reefs within regions (e.g., ranging from 24% at Wreck to 73% at Frederick Reef in the southern CSMP, and 13% at Chilcott Islet to 59% at Flinders Reefs in the central CSMP), and sites within reefs (e.g., 19% at Holmes 2 vs 59% at Holmes 6). This variation in coral loss across relatively small spatial scales could reflect differences in the composition of coral communities, local environmental conditions, resilience to heat stress, and/or other identified factors and warrants further investigation.

Together with the declines in coral cover, there were noticeable declines in coral richness in the southern and central CSMP, and shifts in the composition of coral assemblages across the CSMP. These changes reflect variation among coral taxa in their susceptibility, and consequent mortality, to elevated temperatures. Consistent with previous studies some of the greatest declines in coral cover were recorded for *Seriatopora*, *Stylophora,* tabular *Acropora* and 'other' *Acropora;* taxa generally considered to be most sensitive to elevated temperatures and coral bleaching. However, some coral taxa that are generally considered to be more tolerant to heat stress (e.g., massive *Porites* and *Favites*) experienced high levels of bleaching and subsequently bleaching-induced mortality in the CSMP, highlighting the severity of the 2020 bleaching event.

Despite the declines in coral cover attributable to the 2020 bleaching event, current coral cover in the CSMP (17%) is broadly comparable to recent estimates for the Great Barrier Reef (19%), and greater than the level seen as critical to avoid ecosystem collapse (>10%). Coral cover on central CSMP reefs (15.2%) is higher than estimates of 1-6% coral cover on some central CSMP reefs (i.e., Herald Cays, Chilcott Islet and Lihou Reef) from the early 2000's. Further, the majority of coral colonies surveyed across the CSMP in 2021 were healthy, with only low levels of bleaching recorded (1–17% of colonies surveyed) among reefs. The density of juvenile corals (an indicator of the recovery potential of coral populations), although low (~1.5 juveniles per m^2), was similar to levels recorded before the 2020 bleaching event.

Large-scale coral mortality commonly leads to declines in reef-associated taxa that rely on corals for food and/or shelter. While there were no substantive changes in the abundances of sea snakes or macro-invertebrates (i.e., sea urchins, sea cucumbers, *Trochus*, *Tridacna* clams) on CSMP reefs following the 2020 bleaching, there were declines in the species richness (7%), abundance (31%), and biomass (19%) of reef fishes within the central CSMP. These changes were largely driven by marked declines in corallivorous, planktivorous, and grazing fishes on Flinders and

Holmes Reefs, and Willis Islets, coinciding with some of the largest declines in coral cover. Fishes that have a direct reliance on live corals for food (i.e., corallivores) and/or habitat (i.e., small bodied planktivores) are typically the first and most adversely affected by coral loss, however, the observed declines in the biomass of grazing fishes (primarily surgeonfishes) is difficult to reconcile. Herbivorous fishes (including grazers) are widely viewed as being critical to the health, resilience, and recovery of reefs following large-scale disturbances, with reductions in their abundance being linked to shifts from coral- to macroalgal-dominated reefs in other areas of the Indo-Pacific. The causes and future implications of the declines in grazing fish populations on CSMP reefs is unclear and warrants future investigation.

While the immediate, or short-term, impacts of the 2020 bleaching event on CSMP reefs are already apparent, continued monitoring will be critical to assess any longer-term impacts on the structural complexity of habitats and reef associated taxa, and the potential recovery of coral assemblages and the shallow water reef habitats of the CSMP more broadly.

The latest (2021) surveys revealed:

- Total shallow water coral cover decreased from 28% in 2020 to 17% in 2021, a mean decline of 39%. The declines in coral cover varied among regions, and reefs ranging from 17% in the northern CSMP (13-29% among reefs), 39% in the southern CSMP (24-73% among reefs), to 43% in the central CSMP (13-59% among reefs). There was also considerable variation in the relative coral loss among sites within individual reefs (e.g., 19% vs 59% at two sites on Holmes Reef).
- The 2020 bleaching occurred against a shifted baseline of coral communities, with the cover of bleaching-susceptible coral taxa being reduced following the 2016 and 2017 bleaching events. This, coupled with the high levels of bleaching-induced mortality in coral taxa that are generally considered to be more tolerant to heat stress (e.g., massive *Porites*: 58%; *Favites*: 79% mortality) highlight the severity of the 2020 bleaching event.
- Bougainville Reef, previously identified as a ‘bright spot’ in terms of both coral cover and fish biomass, was again a standout. Despite experiencing significant bleaching in 2020 (65% of colonies bleached), the decline in coral cover was low (13%), and average coral cover remained high (35%).

- Declines in coral cover were relatively consistent between the reef crest (1-3m depth) and reef slope (7-10m depth) in the southern and central CSMP, but greater declines were recorded on the reef slope (22%) than reef crest (3%) on the two northern CSMP reefs.
- The differential susceptibility of coral taxa to bleaching and subsequent mortality led to shifts in the community composition of coral assemblages, with several fast growing and branching taxa (i.e., *Seriatopora*, *Stylophora,* tabular *Acropora* and 'other' *Acropora)* being lost.
- Despite the widespread loss of live corals, there were no increases in macroalgae across the CSMP, and the remaining coral cover on CSMP reefs (17%) is comparable to recent estimates for the Great Barrier Reef (19%), and considerably higher than that of some central CSMP reefs (i.e., Herald Cays, Chilcott Islet and Lihou Reef) from the early 2000's.
- There were no substantial or consistent changes in the abundance of sea snakes or macro-invertebrates (i.e., sea urchins, sea cucumbers, *Trochus*, *Tridacna* clams) on CSMP reefs following the 2020 bleaching.
- Ten fish species that had not been recorded during surveys or observations on the previous voyages (2018-2020) were recorded during the 2021 surveys, taking the total fish species recorded in the CSMP during the past four years of surveys to 631 species. No new species of coral were observed.
- There were declines in the species richness (7%), abundance (31%), and biomass (19%) of fish communities within the central CSMP from 2020 to 2021, while those on southern and northern CSMP remained relatively stable. These changes were most pronounced on Flinders and Holmes Reefs, and Willis Islets, coinciding with some of the largest declines in coral cover, and were largely driven by marked declines in corallivorous, planktivorous, and grazing fishes.
- Despite the changes in fish populations, the total biomass of reef fishes (a key indicator of reef health, together with coral cover) recorded across all reefs in the CSMP (500 - 3,000 kg per hectare) is high relative to coral reefs globally. These estimates of reef fish biomass are exceptional given the relatively low levels of coral cover, and altered composition of coral

assemblages on many reefs in the CSMP, and likely reflects their isolation and limited fishing pressure.

- The density of juvenile corals (an indicator of the recovery potential of coral populations) while generally low (~1.5 juveniles per m^2) across the CSMP, was similar in 2020 and 2021. 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, such as the 2020 bleaching event. However, current data suggests that the latest bleaching has not impacted the abundance of juvenile corals.
- The vast majority (96%) of coral colonies surveyed across the CSMP in 2021 were healthy, with only low levels of injury (incl bleaching) recorded across the 13 reefs (1–11% of colonies surveyed). This level of injury is likely within the natural range of coral injury for coral reef systems, although needs to be interpreted against a shifted baseline in coral composition due to reductions in bleaching-susceptible coral taxa following the 2020 bleaching event
- In addition to the monitoring undertaken, several additional projects were leveraged from this collaboration between James Cook University and Parks Australia and capitalised on available space during the voyage. These leveraged projects involved 12 researchers from 4 institutions and represent a significant in-kind contribution. Collectively, these projects will increase our understanding of the ecology of deep reef habitats, the movement and connectivity of sharks and large reef fishes, and fine scale hydrodynamics around reefs within the CSMP.

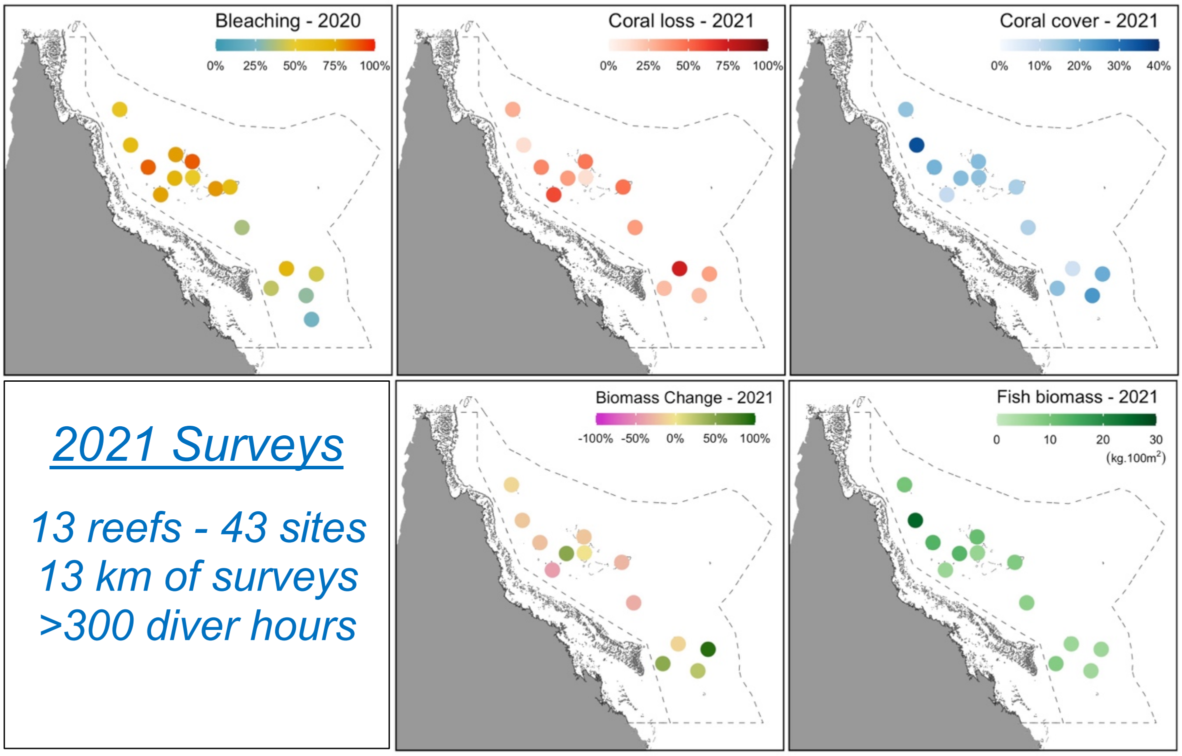


Figure 1. Summary of spatial variation in the intensity and effects of the 2020 coral bleaching event in the Coral Sea Marine Park: proportion of colonies bleached in February 2020, the relative decline in coral cover and fish biomass for 13 reefs between 2020-2021, and the cover of live corals and biomass of reef fishes ~1 year after the 2020 bleaching event. Values are averaged across habitats and sites on each reef, and based on all surveys conducted during 2020 and 2021.

In conclusion, the 2020 bleaching event has had a significant impact on coral and reef fish communities across all 13 CSMP reefs surveyed. This is the third major bleaching event in the CSMP in the last 6 years (2016, 2017 and 2020), and reflective of the increasing frequency and intensity of marine heatwaves that are affecting coral reefs globally. Continued surveys of CSMP reefs will be critical to assess the longer-term impacts of the 2020 bleaching event on reef fishes and other reef-associated species, and the potential recovery and resilience of these isolated reef systems in the absence of other stressors.

Recommendations for future research and monitoring:

- Continued annual monitoring of the sites surveyed in both 2020 and 2021 is critically important to determine any longer-term effects of the 2020 bleaching event on reef fish and other reef associated species, the potential recovery of coral assemblages, and any future disturbances that may push coral cover toward critical thresholds.

- Given the increasing incidence of major disturbances impacting CSMP reefs in recent years, coupled with predicted increases in the frequency and intensity of disturbances affecting reefs globally, and the logistical constraints of working in the CSMP (i.e., isolation and exposure) regular (annual or biennial) surveys are critical. In the absence of regular monitoring, the causes of any changes in reef communities (e.g., the declines in coral cover from 2020 to 2021) would be largely unknown, severely limiting the capacity of managers to make informed decisions.
- Annual monitoring of coral, fish, sea snake and other reef taxa communities on 12-14 reefs (minimum of 4 reefs in each CSMP region), with all 20 CSMP reefs to be surveyed once every 3-4 years.
- Re-survey of four ‘bright spot’ reefs that were not surveyed in 2021 (i.e., Ashmore, Boot, Moore and Mellish Reefs) prior to April 2022 to determine how coral and fish communities fared after the 2020 bleaching
- Dedicated sampling to directly quantify the settlement of coral larvae at a subset of accessible innermost reefs (e.g., Flinders, Holmes, Bougainville and Osprey Reefs). This would require an additional voyage in Oct/Nov to deploy settlement tiles.
- 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.
- Investigation of the potential causes (e.g., water temperature, upwelling, water flow) of the observed variation in coral mortality within individual reefs.
- Biennial surveys (Oct/Nov and Feb/Mar) to allow detailed investigation of seasonal processes (e.g., coral reproduction and settlement, fish spawning aggregations) and more effective deployment and maintenance of in-water sampling equipment. Some of this may be achieved through increased communication and collaboration among government and non-government organisations (e.g., dive tourism and fishing charter operators).
- 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.

Table of Contents

2 Background

The Coral Sea is situated off Australia's north-east coast, bounded by Papua New Guinea to the north, the Solomon Islands, Vanuatu and New Caledonia to the east, and the Tasman Sea to the south. 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 et al. 2020). Australia's marine estate within the Coral Sea is managed through the Coral Sea Marine Park (CSMP) that extends from the eastward margin of the Great Barrier Reef Marine Park (GBRMP) to the outer extent of Australia's Exclusive Economic Zone, some 1,200km offshore (Figure 2.1). The CSMP is among the world's largest and most isolated marine parks, encompassing an area of 989,836km^2, and is managed by the Australian Government, Director of National Parks. Within the CSMP there are approximately 56 islets and cays and 20 widely separated shallow reef systems, ranging from Ashmore and Boot reefs adjacent to the Torres Strait in the north, to Cato Reef in the south, and Mellish Reef (>1,000 km east of Cairns) in the far east. These shallow reefs systems, including Lihou Reef one of the world's largest atolls (~2,500km^2) have a combined reef area of 15,024 km^2; equating to 1.5% of the total CSMP (DNP 2018).

The reefs of the CSMP are fundamentally different to the more inter-connected reefs of the GBRMP, and are largely shaped by the geomorphic, oceanographic and environmental conditions of the region. Reefs within the CSMP rise from seamounts on four major deep-water plateaus; the Eastern Plateau in the north, the Queensland Plateau in the central region, and the Marion and Kenn Plateaus in the south, such that individual reefs are separated by oceanic waters up to 4,000 m deep (Davies et al.1989, Collot et al. 2011). Given the isolation of these reefs, potential connectivity among them is likely facilitated by major ocean currents. 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).

The CSMP is one of the most isolated coral reef environments in Australian waters, with limited exposure to direct human pressures (e.g., fishing, run-off) relative to more accessible coastal reefs. Despite this isolation, coral cover on many reefs within the Coral Sea, especially those in the central Coral Sea, has historically been relatively low (Ayling and Ayling 1985, Oxley et al. 2003, Ceccarelli et al. 2008, Hoey et al. 2020). Reefs in the central CSMP have been repeatedly exposed to severe tropical cyclones and also climate-induced coral bleaching (Ceccarelli et al. 2013, Harrison et al. 2019; Hoey et al. 2020). These disturbances, coupled with limited recovery potential due to poor connectivity and supply of coral larvae from other sources, most likely account for sustained low coral cover on these reefs (Oxley et al. 2003, 2004, Ceccarelli et al. 2008).

The shallow water reef habitats of the CSMP support unique coral and reef fish communities that are distinct from those of the adjacent GBRMP, and share many species with reefs in the Tasman Sea to the south (i.e., Elizabeth and Middleton Reefs and Lord Howe Island), and nations to the east (New Caledonia, Vanuatu and the Solomon Islands; Hoey et al. 2020). While there is some differentiation of fish and coral communities among the northern, central, and southern regions of the Coral Sea, a striking feature of these reefs is the diversity of reef fish (>600 species) and the high abundance and biomass of sharks (mainly the grey reef shark, *Carcharhinus amblyrhynchos,* and the silvertip shark, *C. albimarginatus*) and other large predatory fishes (Ceccarelli et al. 2013, Stuart-Smith et al. 2013, Hoey et al. 2020). The high biomass of large predatory fishes is comparable to the other isolated reef systems, such as the Chagos Archipelago in the central Indian Ocean (Graham and McClanahan 2013), and likely reflects the limited fishing that occurs on these reefs.

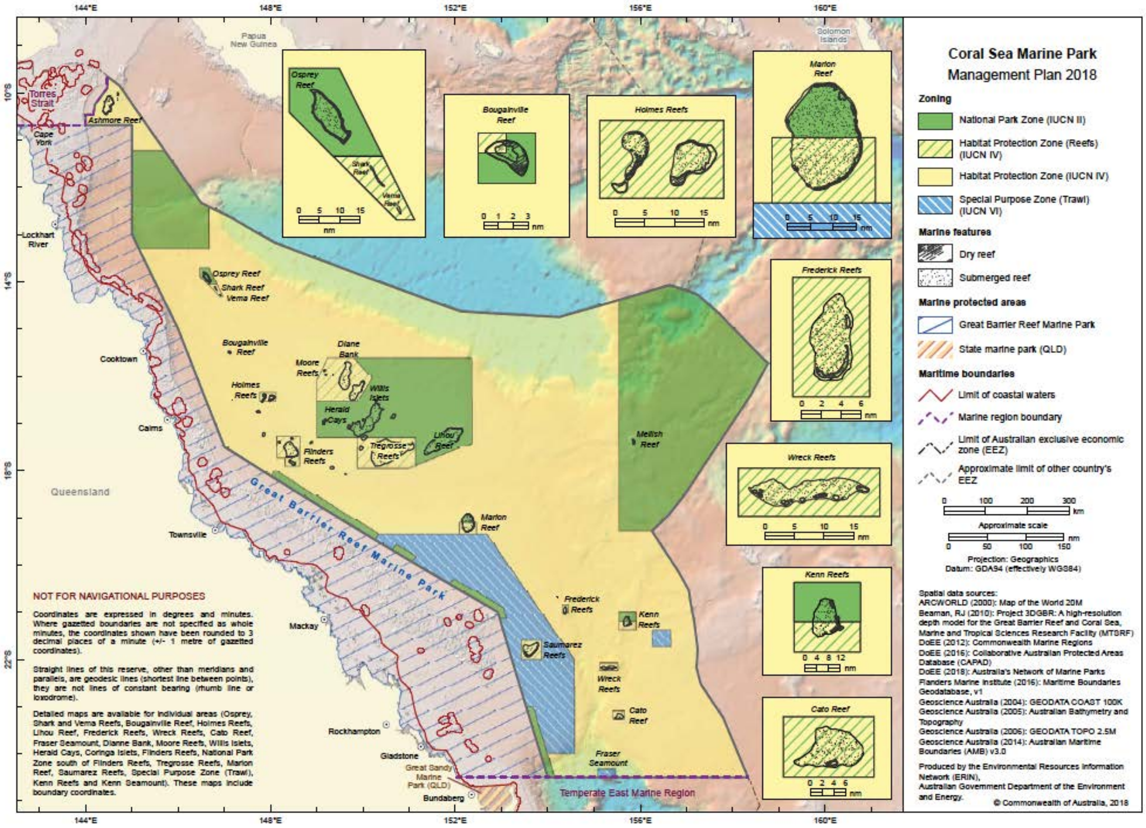


Figure 2.1. Map of the Coral Sea Marine Park, showing management zones implemented in July 2018. (Source: parksaustralia.gov.au)

2.1 Historical heat stress and the 2020 coral bleaching event

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). Despite the isolated nature and hence limited direct human pressures on CSMP reefs, they are increasingly being exposed to climate-induced marine heatwaves. Five major coral bleaching events have been reported in the CSMP in the past two decades (i.e., 2002, 2004, 2016, 2017 and 2020; Oxley et al. 2004, Harrison et al. 2018, 2019, Hoey et al. 2020), including the most recent bleaching event in 2020. Other bleaching events may have also affected CSMP reefs but went undetected due to its isolation and infrequent scientific surveys. Furthermore, comparison of the annual maximum Degree Heating Weeks (DHW, combines both the intensity and duration of heat stress into a single number) for the southern, central, and northern CSMP over the past 35 years shows steady increases across all sectors (Figure 2.2). Since 1985, 12-year means of the maximum DHW have more than doubled in the southern, central and northern CSMP, and are expected to increase further (van Hooindonk et al. 2013).

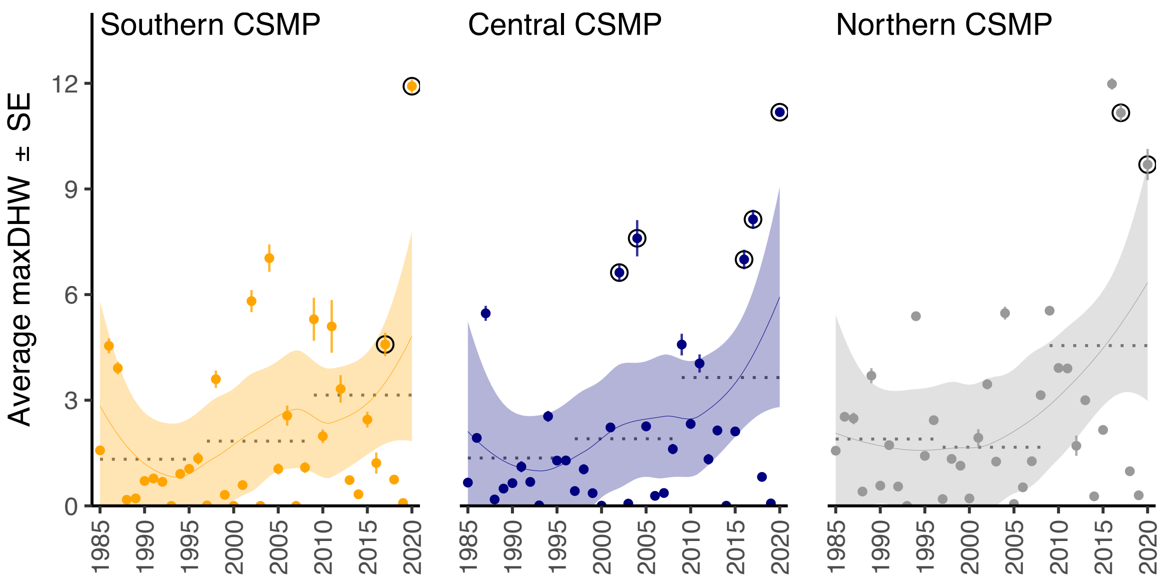


Figure 2.2 Maximum Degree Heating Weeks (DHW) averaged across reefs in the southern, central and northern Coral Sea Marine Park between 1985 and 2020. Cumulative heat stress above 3 DHW can lead to bleaching of shallow water corals, with widespread mortality occurring above 6 DHW (Hughes et al. 2018). Documented bleaching events are indicated by larger open circles and 12-year means are represented by dotted lines.

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 in the 35 years since 1985 (Figure 2.3a,b), with a concomitant decrease in the return time between events (where DHW> 3) to less than 2 years (Figure 2.3c). These events have undoubtedly shaped present, and will continue to shape future, coral reef communities in the CSMP.

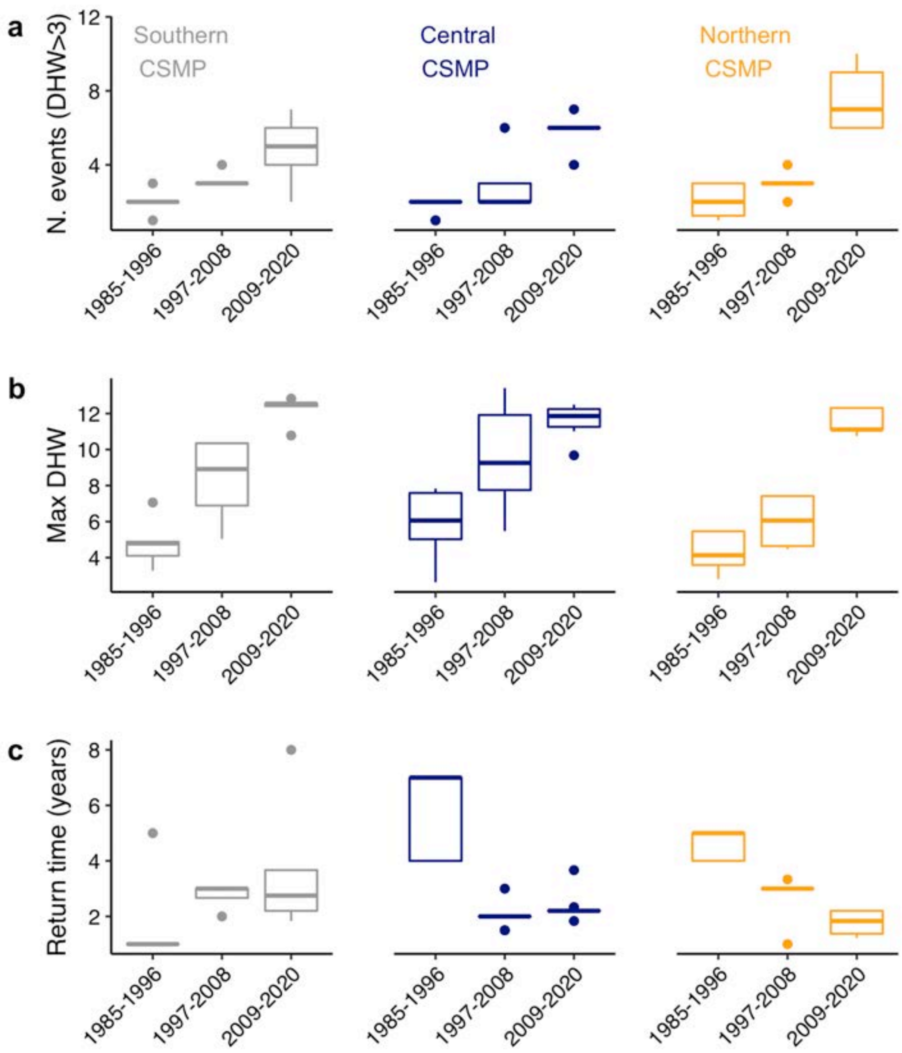


Figure 2.3 The (a) number, (b) intensity and (c) return time of marine heatwaves that are likely to induce coral bleaching (Degree Heating Weeks; DHW > 3) in the Coral Sea Marine Park between 1985 and 2020.

The 2020 bleaching event in the CSMP was severe and widespread, with 63% of all corals surveyed across 16 reefs exhibiting heat stress (from paling to recently dead) from high ocean temperatures (Hoey et al. 2020). The extent of bleaching varied regionally (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) within the CSMP. Comparisons of DHW (Figure 3.4) and incidence of bleaching among the three most recent bleaching events in the CSMP suggest the 2020 coral bleaching event in the CSMP was more severe and widespread than either the 2016 or 2017 events (Hoey et al. 2020). Large areas of the southern and central CSMP were exposed to >12 DHW during 2020 (Figure 2.4). It is also important to consider that the 2020 bleaching occurred against a shifted baseline of coral communities, with the abundance of bleaching sensitive coral taxa being reduced due to the 2016 and 2017 bleaching events (Harrison et al. 2019). However, bleaching does not directly equate to mortality, and some bleached corals can recover if water temperatures decrease sufficiently. Rates of mortality have been shown to vary considerably among coral taxa and with the extent of bleaching, with some corals taking up to 10 months to die (Baird and Marshall 2002). Therefore, future surveys are critical to assess the full extent of the 2020 bleaching event on reefs within the CSMP.

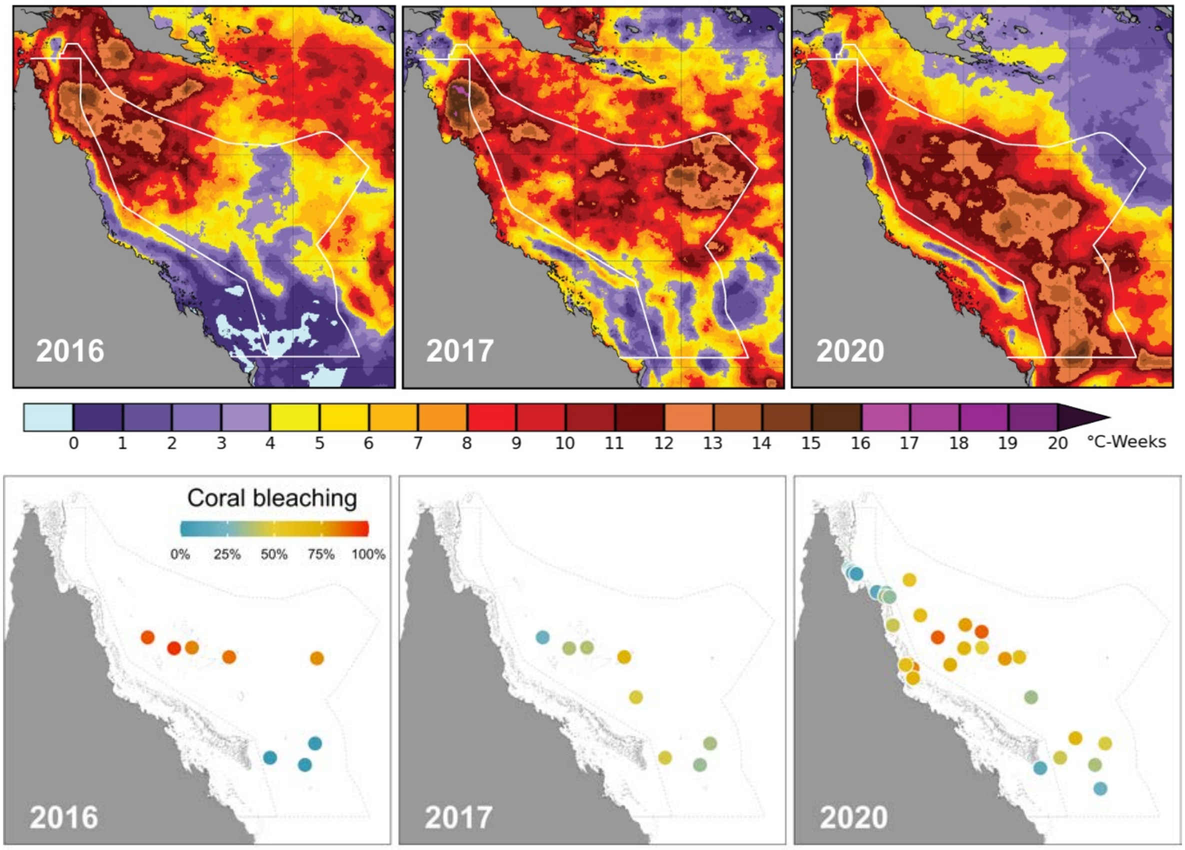


Figure 2.4. Comparison of the maximum Degree Heating Weeks (top row), and incidence of coral bleaching (bottom row) experienced throughout the Coral Sea Marine Park during 2016, 2017 and 2020.

2.2 Objectives and scope

The purpose of this study was to provide comprehensive assessments of the current condition of benthic and fish communities within the Coral Sea Marine Park (CSMP), assess the impacts of the 2020 bleaching event on benthic, fish and invertebrate communities throughout the CSMP, and gain some understanding of the potential resilience and recovery of corals.

Surveys were conducted at thirteen reefs throughout the CSMP following the methods of Hoey et al. (2020). At each site, surveys were conducted along three replicate transects within each of two habitats (reef crest: 1-3m depth; reef slope: 7-10m depth) to 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,
vii) abundance of holothurians, urchins and other ecologically or economically important reef-associated invertebrates, and
viii) the abundance and size of sea snakes

As well as the objectives listed above, several projects were leveraged from this collaboration between James Cook University and Parks Australia and capitalised on available vessel space during the voyage. These leveraged projects include:
i) Movement and population structure of sharks and large fishes within the CSMP;
ii) The ecology of deep reef habitats in the CSMP;
iii) Collection of video footage for the production of educational and promotional videos of the CSMP;
iv) Genetic diversity of giant clams (*Tridacna* spp.) within the CSMP;
v) Surveys for fish spawning aggregation sites within the CSMP; and
vi) *In situ* measurements of temperature and water flow.
Further details of these projects are provided in Appendix 1.

3 Methods

Surveys were undertaken at 43 sites across 13 reef systems within the CSMP during a 25-day voyage, 4th February – 1st March 2021 (Figure 3.1). The thirteen reefs surveyed were southern CSMP: Saumarez, Wreck, Kenn, Frederick Reefs; central CSMP: Marion, Flinders (north and south), Holmes (east and west), and Lihou (north and south) Reefs, Herald Cays, and Chilcott and Willis Islets; northern CSMP: Bougainville and Osprey Reefs (Appendix 2). At each reef, we re-visited sites that were surveyed in 2020 (Hoey et al. 2020) to facilitate direct comparisons in coral health and reef condition. Sites were relocated using GPS waypoints and a bearing of the direction of the transects from that waypoint. An additional six sites across three outer-shelf reefs of the central and northern GBRMP were surveyed using identical methodologies. We had planned on surveying several more GBRMP reefs that had been surveyed in 2020, however these surveys could not be completed due to adverse weather associated with Tropical Cyclone Niran at the time of the planned surveys. The surveys of GBRMP reefs were part of, and funded by, other projects but included here for comparative purposes.

3.1 Sampling design

25 days
13 reefs - 43 sites
13 km of UVC surveys
>300 diver hours

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).

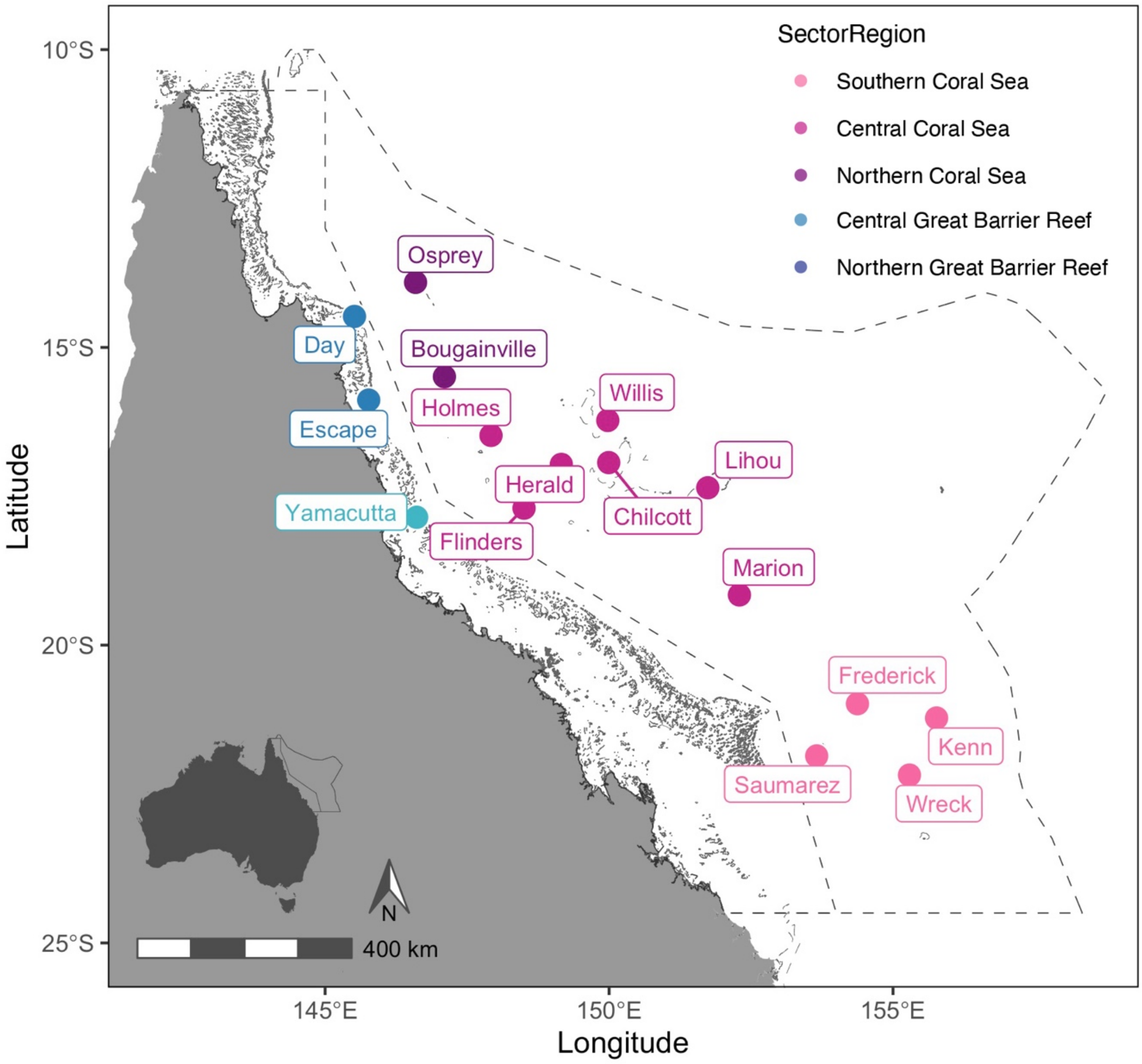


Figure 3.1 Map of the surveyed reefs in the Coral Sea Marine Park and Great Barrier Reef Marine Park in February and March 2021. Colours relate to the regional allocation of reefs in the southern, central, and northern Coral Sea Marine Park and central and northern Great Barrier Reef Marine Park, which 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. 2020). Deploying the transect while simultaneously recording fishes minimises disturbance prior to censusing, thereby minimising 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 substratum underlying evenly spaced (50cm apart) points along the entire length of the transect. The final (fourth) diver measured coral health, colony size, and abundance of juvenile corals (as a proxy of 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.

3.2 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 Hoey et al. 2020). 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, and encrusting with uprights). 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.

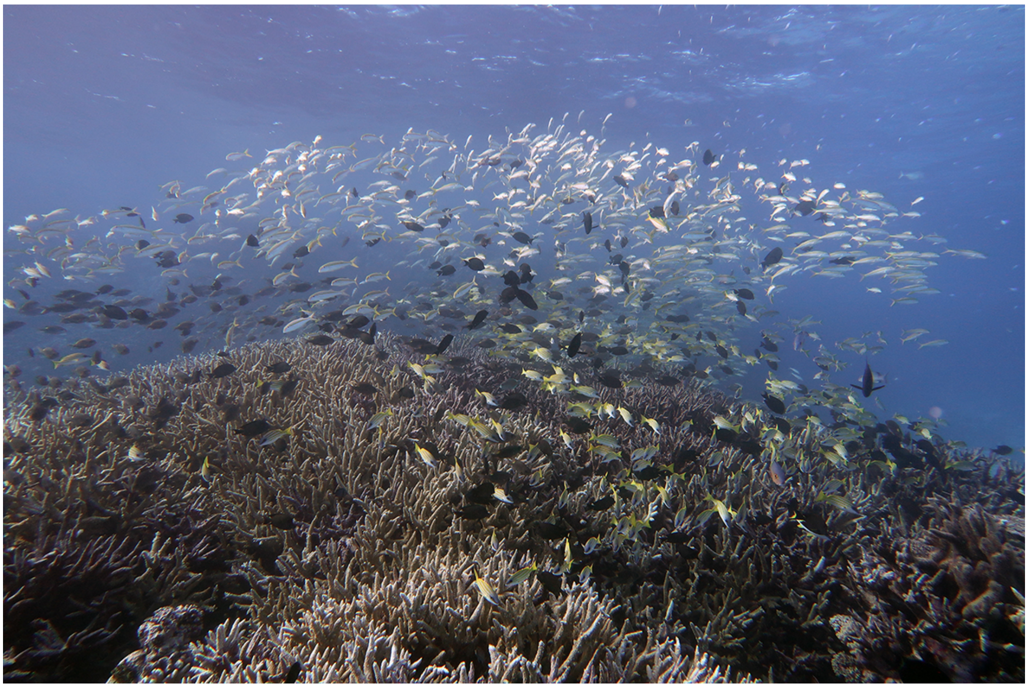


Figure 3.2 Herald Cays, February 2020. Areas of high coral cover and complexity in the Coral Sea Marine Park support a high density of reef associated fishes. Image credit: Andrew Hoey

Coral health – The health of all coral colonies was recorded 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 3.1), 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.

Table 3.1 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 

Coral recruitment - Densities of juvenile corals (≤5 cm maximum diameter, following Rylaarsdam 1983) are increasingly used to as a proxy for coral recruitment and hence the replenishment of coral populations as opposed to settlement studies that deploy experimental settlement substrata (e.g., tiles) and quantify the number of coral larvae that settle to these substrata. 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 3.3).

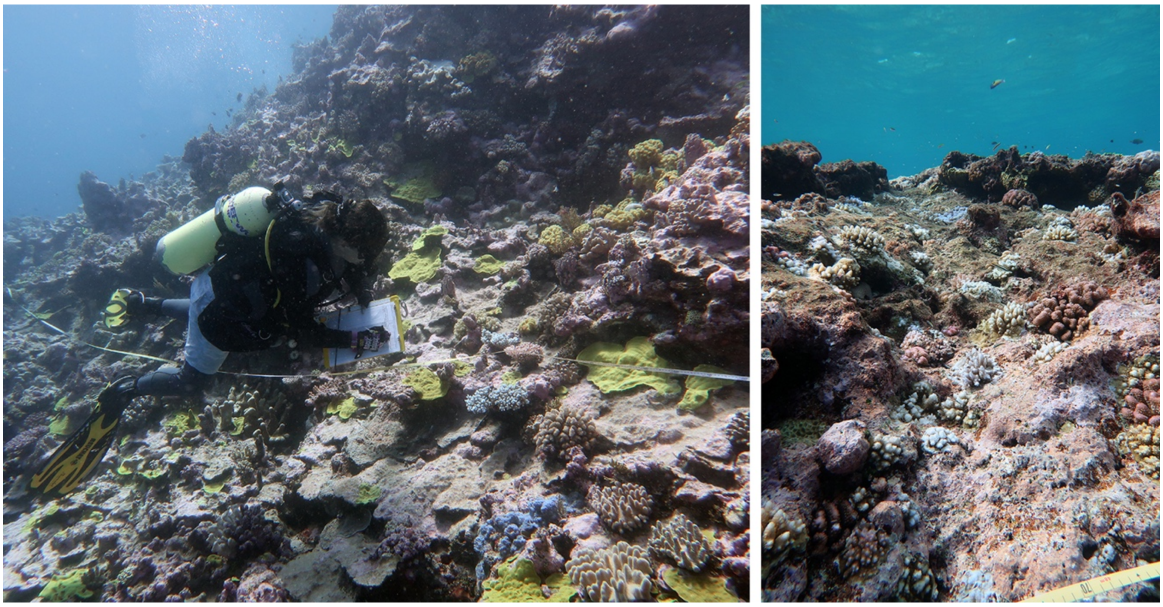


Figure 3.3 Deborah Burn conducting coral health and juvenile coral surveys at Lihou Reef (left), and high densities of juvenile coral on the reef crest at Osprey Reef (right), February 2021. Image credits: Andrew Hoey

3.3 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) were counted in a 2m wide belt ($100m^2$ per transect). Slightly larger bodied, site-attached species (e.g., Chaetodontidae, Labridae) 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.

Figure 3.4 Andrew Hoey surveying reef fishes while simultaneously deploying the transect tape in the shallow reef habitat on Willis Islet in February 2021. Image credit: Morgan Pratchett

3.4 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 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* spp., 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).

3.5 Data handling and analysis

Data from the 2021 surveys were combined with those of the previous voyages (2018-2020) into a single database and analysed using R version 4.0.2 (R Core Team 2021). 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). Maps of the GBRMP and marine park boundaries were reproduced from shape files contained in *gisaimsr* (Barneche and Logan 2021) and *dataaimsr* (AIMS Datacentre 2021), data courtesy of the Great Barrier Reef Marine Park. Maps of CSMP reefs and boundaries were reproduce from shapefiles generated by Project 3DGBR (Beaman 2012). All maps were produced in R using the package *sf* (Pebesma 2018) and *ggspatial* (Dunnington 2021) using the WGS84 coordinate system.

All survey data were averaged across independent transects to obtain a site average prior to summarising data at the level of reefs or regions. Data are generally presented using box and whisker plots. The box plots represent the distribution of the data based on the minimum, first quartile, median, third quartile and maximum values. The lower and upper hinges correspond to the first and third quartiles (the 25th and 75th percentiles). The upper whisker extends from the hinge to the largest value no further than 1.5 * IQR from the hinge (where IQR is the inter-quartile range, or distance between the first and third quartiles). The lower whisker extends from the hinge to the smallest value at most 1.5 * IQR of the hinge. Data beyond the end of the whiskers (i.e., outliers) are plotted individually.

Non-metric multi-dimensional scaling (nMDS) was used to identify similarities in coral and fish assemblages among reefs in *a priori* defined regions (i.e., southern, central, and northern CSMP) and between years. 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 to reduce the relative influence of the most frequent and variable taxa, which otherwise will tend to dominate the dissimilarity matrix. For the analysis of coral composition rare taxa were grouped as 'other Scleractinia' to reduce the influence of rare taxa in the dissimilarity matrix. The data were then standardised following a Wisconsin scaling, which removes the effect of absolute species abundance and also abundance between sites, so the comparison between sites becomes relative. Distances between points were determined with the *metaMDS* function using the Bray-Curtis dissimilarity metric. All data were analysed in the *vegan* package (Oksanen *et al.* 2020) using the statistical software package R version 4.0.2.

4 Findings

4.1 Impacts of the 2020 bleaching event on coral communities

4.1.1 Coral cover

Severe and widespread bleaching was recorded across CSMP reefs in February-March 2020, with 63% of all corals surveyed across 16 CSMP reefs showing signs of heat stress (from pale to recently dead) from elevated ocean temperatures (Hoey et al. 2020). The extent of bleaching varied regionally (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) within the CSMP. Understanding the impact of this event on the cover and composition of coral assemblages is critical in assessing the current health of reefs in the CSMP.

Comparisons of surveys of shallow reef habitats conducted in 2020 versus 2021 revealed a significant decline in coral cover on CSMP reefs (Figure 4.1), with these declines almost certainly being a result of the 2020 bleaching event. Overall, there was a 39.5% decline in mean coral cover across the 13 CSMP reefs surveyed (2020: 28.0%: 2021: 17.2%). There was, however, variation in the declines in coral cover among the three regions of the CSMP, with the greatest declines being recorded in the central and southern CSMP (mean declines of 43.1% and 38.8%, respectively) compared to the northern CSMP (mean decline of 17.1%). Interestingly, and in contrast to previous studies that have found the effects of bleaching to decline with depth (e.g., Bridge et al. 2013; Smith et al. 2014; Baird et al. 2018), the declines in coral cover were relatively consistent between the reef crest (1-3m depth) and reef slope (7-10m) in each of the three CSMP regions (Figure 4.2). Overall, coral cover declined by 35% on shallow reef crests (1-3m depth) and by 41% on deeper reef slope habitat (9-10m depth). The only exception to this trend was the northern CSMP where there was a negligible increase in coral cover on the reef crest and a decline of 20% within the deeper reef slope habitat (Figure 4.2). The lack of decline in coral cover on the reef crest is interesting and may reflect the coral composition of these sites, local acclimation of these corals to the temporal variability in water temperatures at these sites, and/or localised upwelling providing relief from heat stress (Choukroun et al. 2021). Future research is needed to identify the mechanism/s for the apparent resilience of these coral assemblages to temperature-induced bleaching.

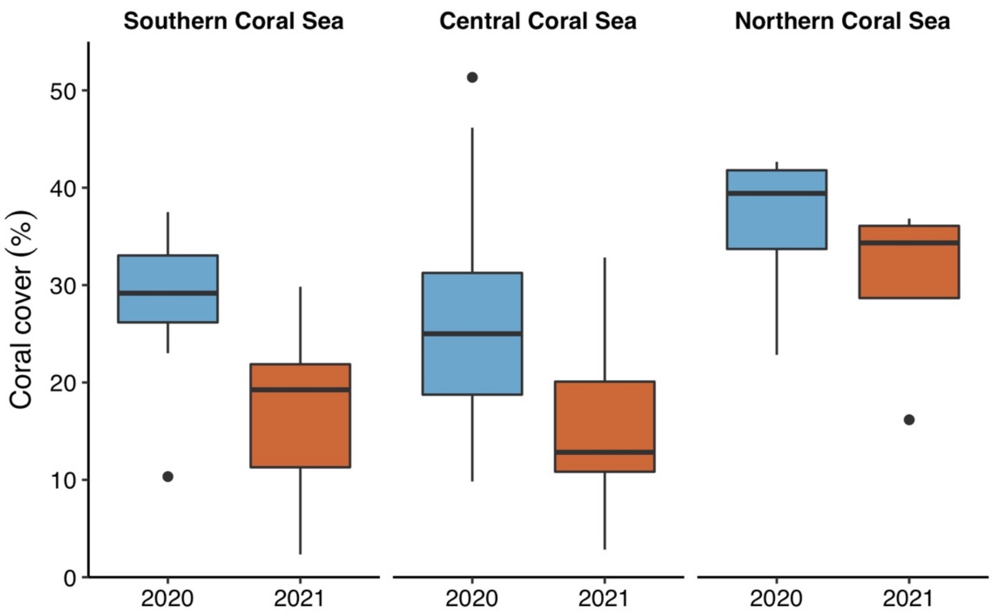


Figure 4.1 Temporal change in average coral cover (+/- SE) within the three regions of the Coral Sea Marine Park. Data are based on surveys of matching sites in 2020 and 2021 across 13 reefs (southern CSMP: Saumarez, Wreck, Kenn, Frederick Reefs; central CSMP: Marion, Flinders, Holmes, and Lihou Reefs, Herald Cays, and Chilcott and Willis Islets; northern CSMP: Bougainville and Osprey Reefs).

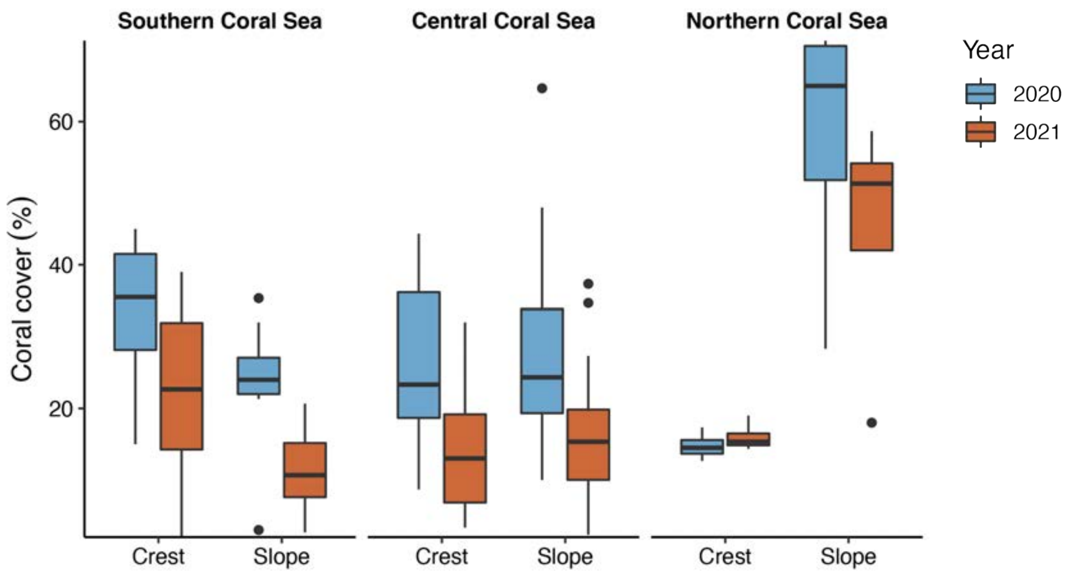


Figure 4.2 Temporal variation (2020-21) in average coral cover (+/- SE) between shallow reef habitats (reef crest and reef slope) within the three regions of the Coral Sea Marine Park. Data are based on surveys of matching sites in 2020 and 2021 across 13 reefs (southern CSMP: Saumarez, Wreck, Kenn, Frederick Reefs; central CSMP: Marion,

Flinders, Holmes, and Lihou Reefs, Herald Cays, and Chilcott and Willis Islets; northern CSMP: Bougainville and Osprey Reefs).

Declines in coral cover from 2020 to 2021 were evident at all thirteen reefs and 43 sites surveyed throughout the CSMP, however there was considerable variation in the declines recorded at individual reefs within each region, and also some variation among sites within each reef (Figure 4.3, 4.4). Within the southern CSMP recorded declines in coral cover ranged from 23.9% at Wreck Reef to 73.5% at Frederick Reef, within the central CSMP from 12.7% at Chilcott Islet to 59.2% at Flinders Reef, and within the northern CSMP from 13.1% at Bougainville Reef to 29.2% at Osprey Reef (Figure 4.3a, 4.4). The declines in total coral cover were relatively consistent among sites within several of the reefs surveyed (e.g., Bougainville, Frederick, and Kenn Reefs, Chilcott Islet, and Herald Cays), yet at other reefs, especially those in the central CSMP, there was considerable variation in the site-level declines in coral cover from 2020 to 2021 (Figure 4.3b, 4.5). For example, at Willis Islet declines in coral cover varied from 31.6% at Willis 2 to 56.7% at Willis 7; at Lihou Reef declines varied from 34.8% at Lihou 9 to 65.5% at Lihou 4; at Holmes varied from 19.3% at Holmes 2 to 58.5% at Holmes 6; and at Marion Reef varied from 7.1% at Marion 6 to 61.7% at Marion 7 (Figure 4.6). The causes of this considerable variation in coral mortality among sites, some of which are located within a couple of kilometres, is unknown and warrants further investigation.

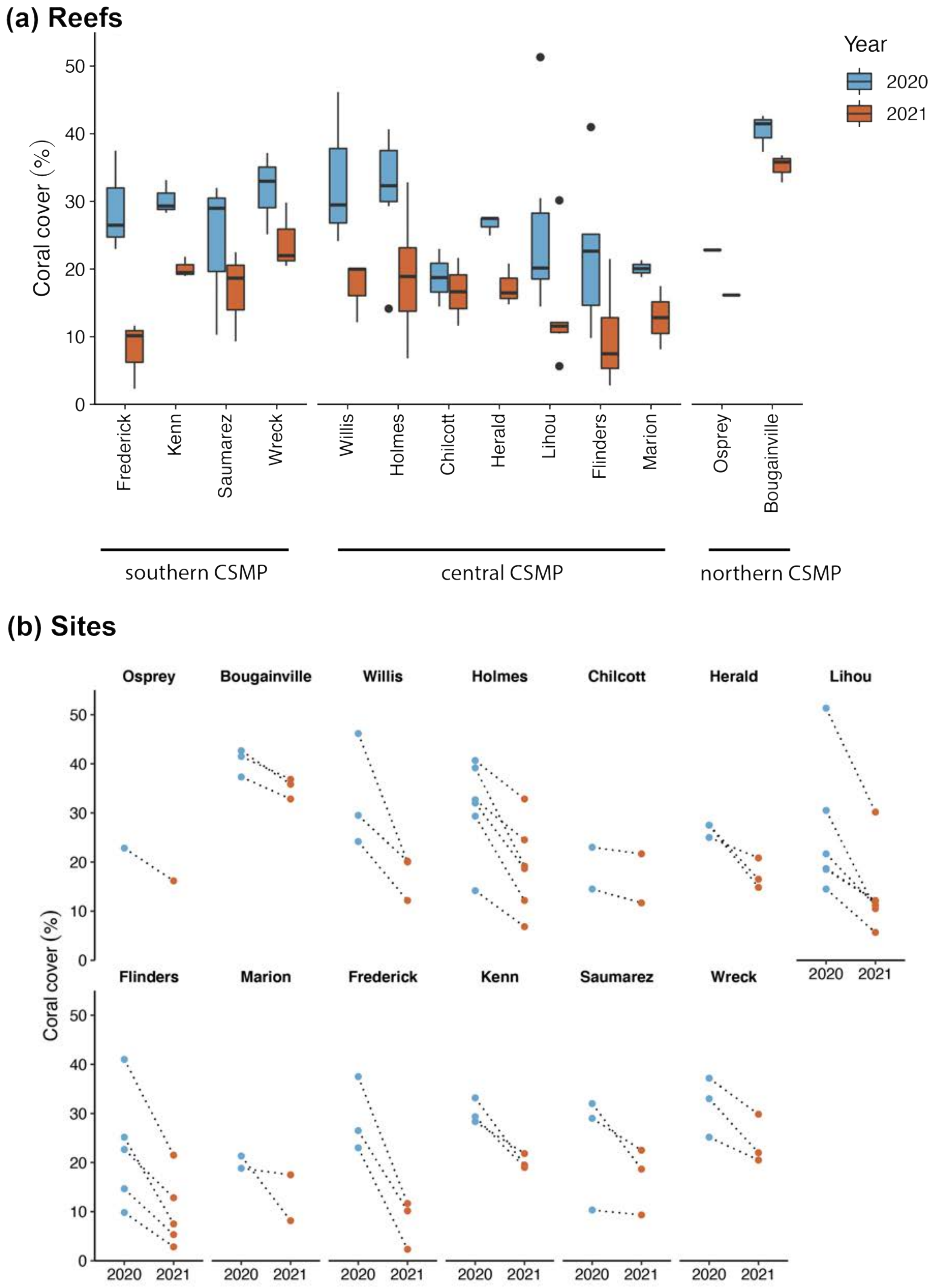


Figure 4.3 Temporal variation (2020-21) in average coral cover (+/- SE) among **(a)** thirteen reefs, and **(b)** 43 sites in the Coral Sea Marine Park. Data are based on surveys of matching sites in 2020 and 2021 and pooled between habitats (reef slope and reef crest) within each site.

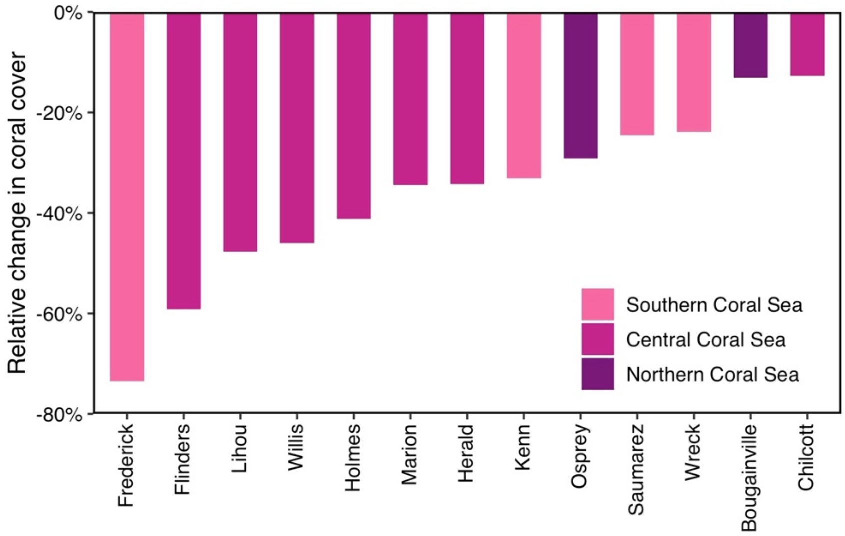


Figure 4.4 Relative change in coral cover following the 2020 coral bleaching event at thirteen reefs in the Coral Sea Marine Park. Data are based on the percentage change in coral cover at matching sites in 2020 and 2021 and pooled between habitats (reef slope and reef crest) within each site.

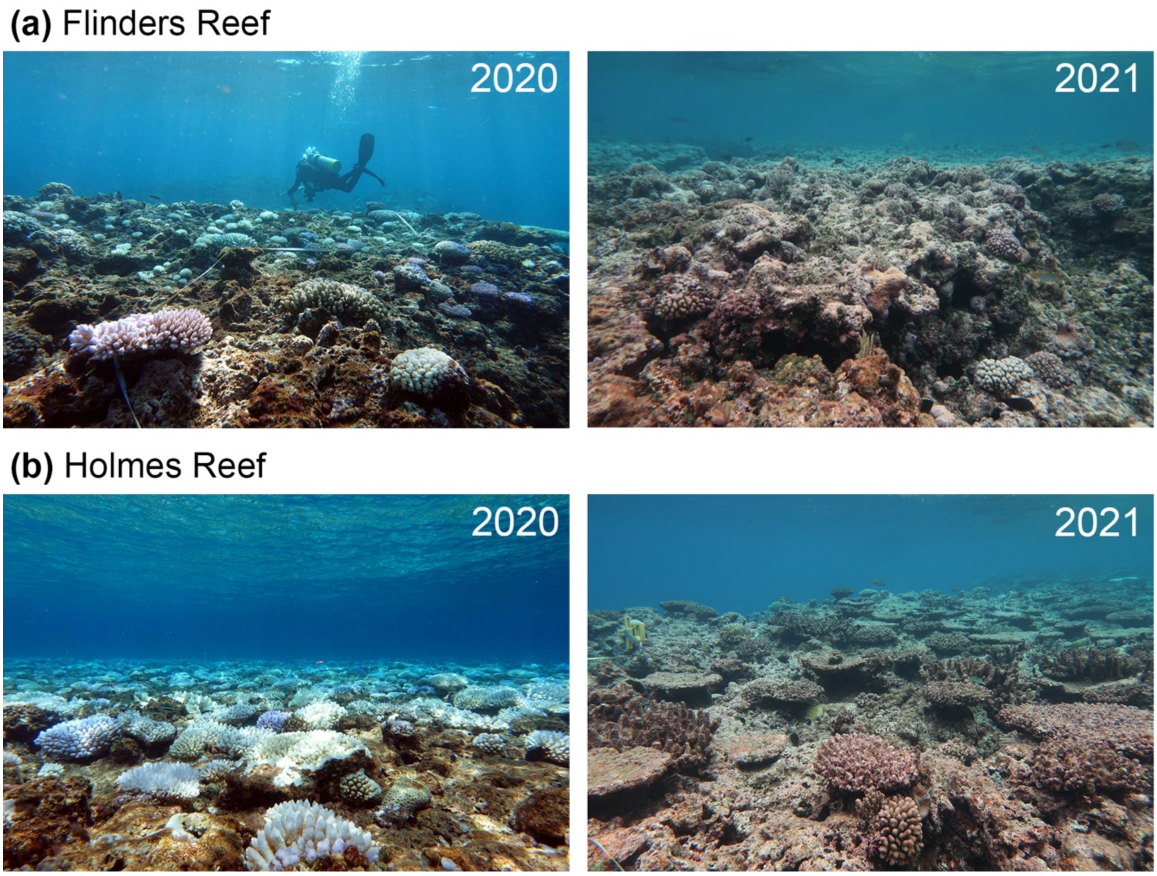


Figure 4.5 Images of the reef crest at **(a)** Flinders Reef and **(b)** Holmes Reef showing the loss of corals due to the 2020 bleaching event. Photographs were taken during the bleaching event (February 2020) and 12 months later (February 2021). Image credits: Dani Ceccarelli (top left, bottom left), Morgan Pratchett (top right), Andrew Hoey (bottom right).

Figure 4.6 Images showing the variation in coral cover at two reef crest sites at Holmes Reef in February 2021. The decline in coral cover between 2020 and 2021 at Holmes site 6 was 58.5% compared to 19.3% at Holmes site 2. Image credits: Andrew Hoey.

In contrast to the declines in coral cover recorded on CSMP reefs, only one of the three GBRMP reefs that were surveyed in both 2020 and 2021 showed a decline in coral cover. On Yamacutta Reef in the central GBRMP mean coral cover decreased from 40.5% in 2020 to 27.5% in 2021; a decline of 30.9%, whereas coral cover increased by 46.6% Day Reef (mean coral cover 2020: 12.2%; 2021: 17.6%) and by 3.7% at Escape Reef (2020: 34.8%; 2021: 36.0%) over the same time period.

4.1.2 Coral richness

Together with the declines in coral cover following the 2020 bleaching event, there were noticeable declines in the mean coral richness in the southern and central CSMP, but not the northern CSMP, from 2020 to 2021 (Figure 4.7). The mean number of coral taxa recorded per site decreased from *ca*. 20 to 16 taxa in the southern CSMP, and *ca*. 18 to 15 taxa in the central CSMP (Figure 4.7a). The greatest declines in coral richness were generally recorded at reefs that experienced the greatest declines in coral cover following the 2020 bleaching event (i.e., Frederick and Flinders Reef, and Willis Islet). The only exception to this was at Holmes Reef where a small increase in coral richness was recorded (Figure 4.7b).

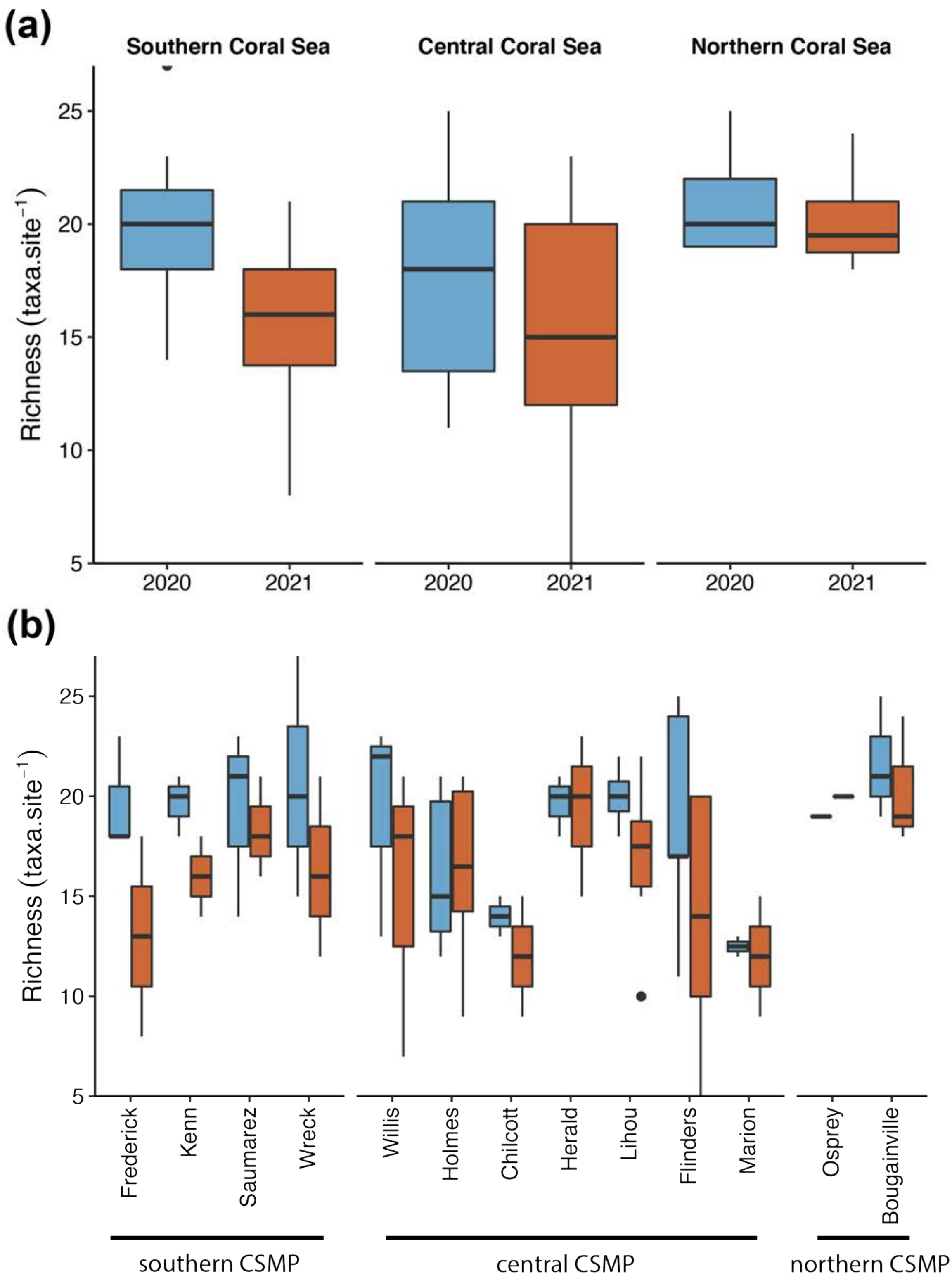


Figure 4.7 Temporal change in average coral richness (+/- SE) among **(a)** regions, and **(b)** reefs in the Coral Sea Marine Park from 2020 to 2021. Data are based on the number of coral taxa recorded at each of 43 sites (i.e., pooled across slope and crest habitats).

4.1.3 Coral composition

The greatest variation in the composition of coral assemblages was among the three CSMP regions (Figure 4.8). In 2021, reefs in the northern CSMP were characterised by a higher cover of *Coeloseris* and *Pocillopora,* while reefs in the southern CSMP were characterised by a higher cover of *Seriatopora, Isopora* and staghorn *Acropora* (Figure 4.8a,b). Together with this regional variation there was also some evidence of shifts in the composition of coral assemblages between 2020 and 2021, with these changes being most pronounced in the central CSMP (Figure 4.8, 4.9), reflecting variation in the susceptibility of coral taxa to elevated temperatures. In the northern CSMP coral assemblages in 2020 were characterised by a higher relative cover of *Leptastrea* and *Favites,* whereas in 2021 were characterised by a higher relative cover of staghorn *Acropora*, *Dipsastrea* and *Coeloseris* (Figure 4.9a,b), however these changes should be treated with some caution given the limited number of sites surveyed in both years (i.e., 4 sites). There was a distinct change in the composition of coral assemblages in the central CSMP, where coral assemblages shifted from being dominated by *Seriatopora, Isopora,* and staghorn and 'other' *Acropora* in 2020 to a mix of largely bleaching resistant taxa (including *Coeloceris, Galaxea, Lobophyllia)* in 2021 (Figure 4.9c,d). In the southern CSMP the composition of coral assemblages showed a greater degree of similarity among sites and reefs in 2020 (i.e., more clustered in the nMDS space) than 2021, likely reflecting the differential impacts of the 2020 bleaching event on individual reefs (Figure 4.4, 4.7b).

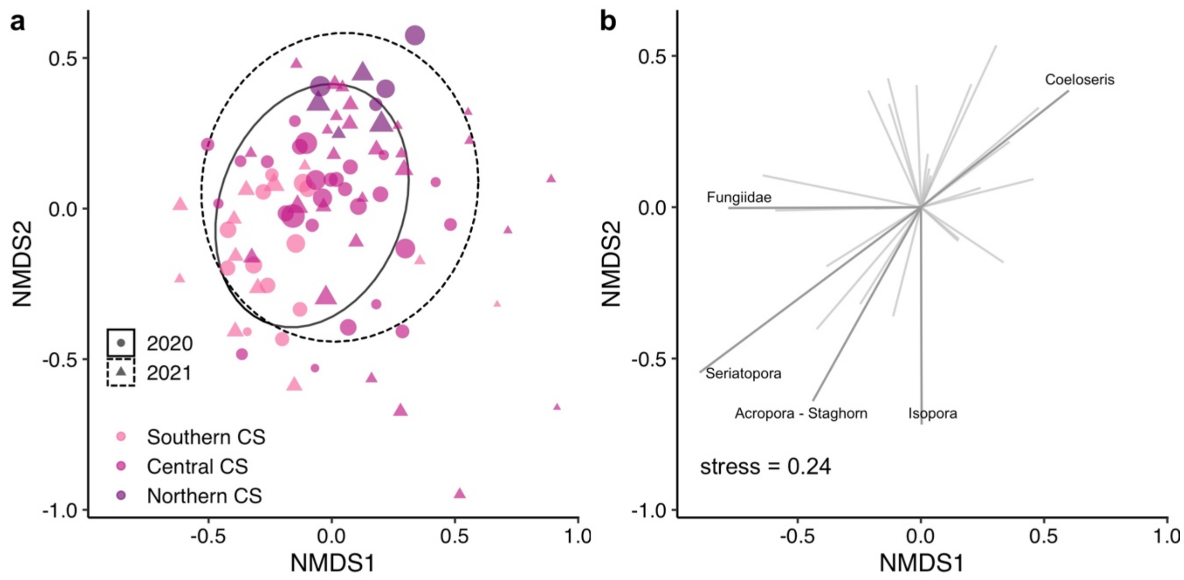


Figure 4.8 Regional and temporal (2020 vs 2021) variation in the composition of coral assemblages within the Coral Sea Marine Park. Non-metric multidimensional scaling (nMDS) plot showing the variation in coral composition among years for all regions of the Coral Sea Marine Park. Analyses are based on data from 43 sites that were surveyed in both years. The size of individual points is proportional to the cover of live coral on each reef. Vectors in the right-hand side plot indicate key taxa that account for the variation in coral composition displayed in the corresponding left-hand side plot.

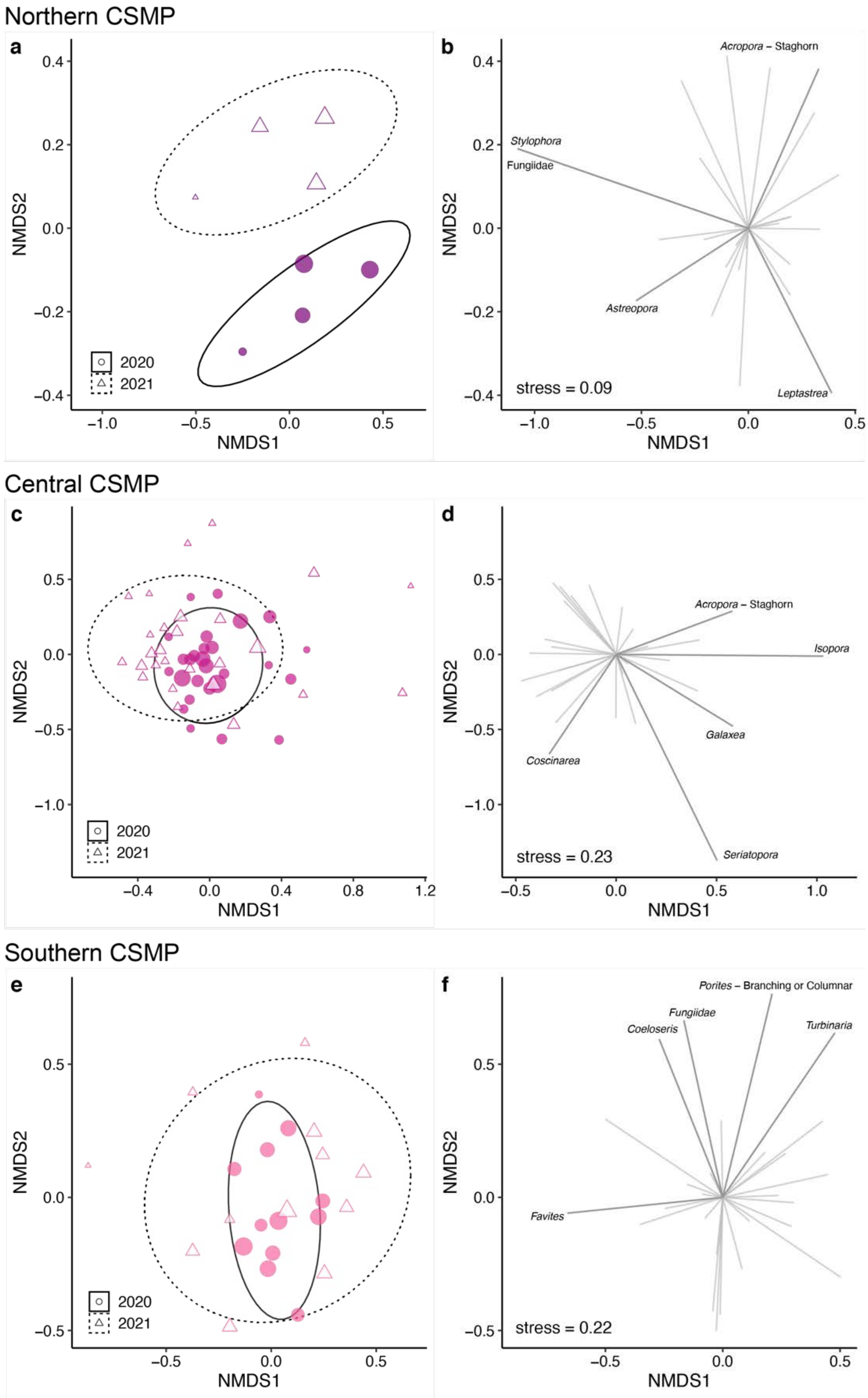
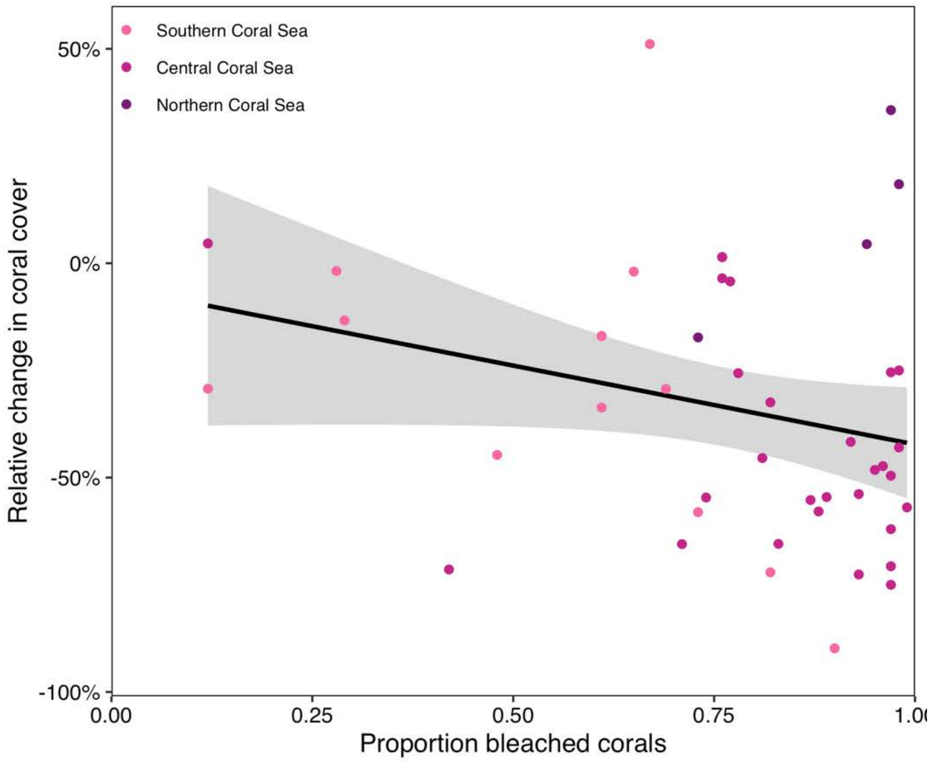


Figure 4.9 Non-metric multidimensional scaling (nMDS) plots showing the temporal variation (2020 vs 2021) in coral composition among reefs in the **(a)** northern, **(b)** central, and **(c)** southern Coral Sea Marine Park. Analyses are based on data from 43 sites that were surveyed in both years (northern: 4 sites; central: 27 sites; southern: 12 sites). The size of individual points is proportional to the cover of live coral on each reef. Vectors in the right-hand side plot indicate key taxa that account for variation in coral composition displayed in the corresponding left-hand side plot.

4.1.4 Relationship between bleaching and mortality

Comparisons of the change in coral cover recorded across the 43 CSMP sites in 2021 with the proportion of bleached corals recorded in 2020 reveal that the incidence of bleaching and subsequent mortality were only weakly related (Figure 4.10). On average, the recorded decline in coral cover was greater at sites that experienced higher levels of bleaching, however the change in coral cover was highly variable for any given level of bleaching. This relationship did not improve if corals that were recorded as ‘pale’ (as opposed to 1-50% bleached, 51-99% bleached, 100% bleached, or recently dead) in 2020 were excluded from the estimates of ‘bleached’ corals (Figure 4.10a). This relatively weak relationship may be related to the timing of the 2020 surveys (February 2020) that were undertaken prior to the maximum DHW experienced at most reefs, especially those in the southern CSMP. Differences in the extent of bleaching versus coral loss may also be attributable to variation in coral assemblages (and hence their susceptibility to heat stress) at each site, and/or differences in the disturbance history among sites and reefs.



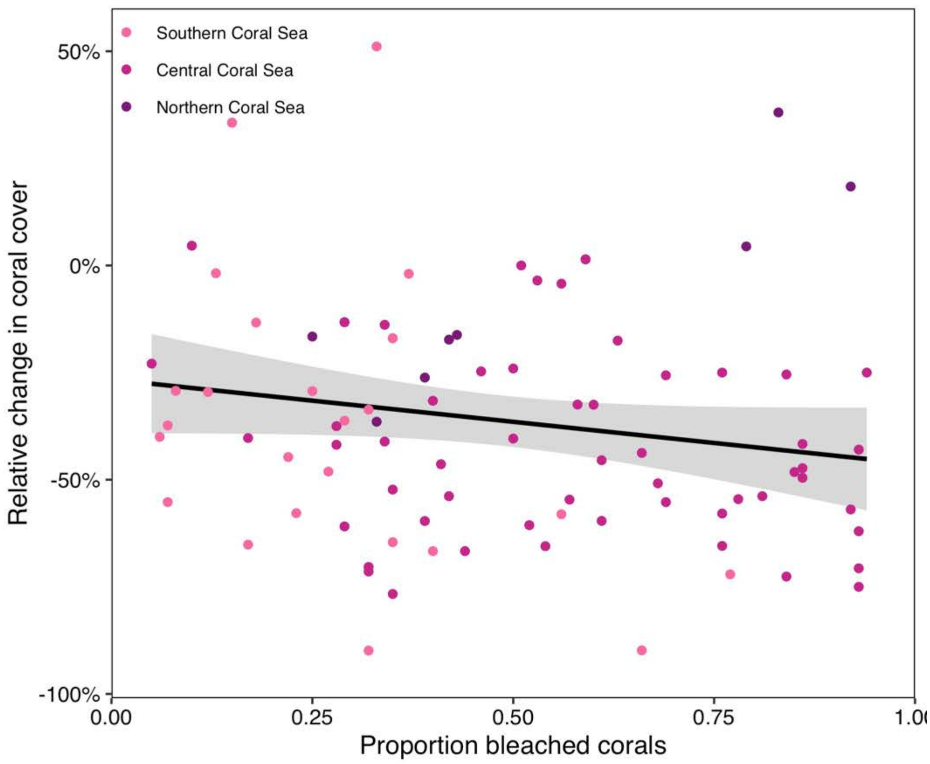


Figure 4.10 Relationship between the incidence of coral bleaching across habitats (crest and slopes) at 43 sites and thirteen CSMP reefs in 2020 and the subsequent change in coral cover twelve months later. (top panel) corals classified as bleached in 2020 includes those exhibiting any sign of heat stress (i.e., 'pale' to 'recently dead'), (bottom panel) corals classified as bleached in 2020 *excluding* those recorded as 'pale' in 2020.

The shifts in coral assemblages following the 2020 bleaching event largely reflect the variation in the susceptibility of coral taxa to elevated temperatures. Tabular and staghorn *Acropora* together with branching *Seriatopora, Stylophora,* and *Pocillopora* 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). Consistent with these previous studies, *Acropora*, *Pocillopora*, *Seriatopora* and *Stylophora* were among the worst affected coral taxa by the 2020 bleaching event in the CSMP, with >60% of colonies surveyed in February 2020 showing signs of heat stress (Hoey et al. 2020). While previous research in the GBRMP has shown that the proportion of corals bleached and the subsequent declines in coral cover of individual coral taxa are largely unrelated (Hughes et al. 2018), the findings from the 2020 bleaching event in the CSMP show that variation in bleaching susceptibility among coral taxa broadly reflected the declines in coral cover recorded in 2021 (Figure 4.11). Notably, the greatest declines in coral cover were recorded for the bleaching sensitive taxa *Seriatopora* (89%), *Stylophora* (79%), tabular *Acropora* (52%) and 'other' *Acropora* (49%). However, some coral taxa that are generally considered to be less sensitive to bleaching (e.g., massive *Porites* (58%) and *Favites* (78%)) also exhibited high levels of bleaching and declined in abundance thereafter (Figure 4.11). Conversely, there was a small increase in the relative cover of staghorn *Acropora,* generally considered to be a bleaching sensitive taxon. Staghorn *Acropora* was rare across the majority of CSMP sites surveyed, likely reflecting the effects of previous disturbances (Harrison et al. 2018, 2019; Hoey et al. 2020), with the increase in cover being driven by small increases at three of the least bleaching effected reefs, Wreck and Saumarez Reefs in the southern CSMP and Bougainville in the northern CSMP (Figure 4.4).

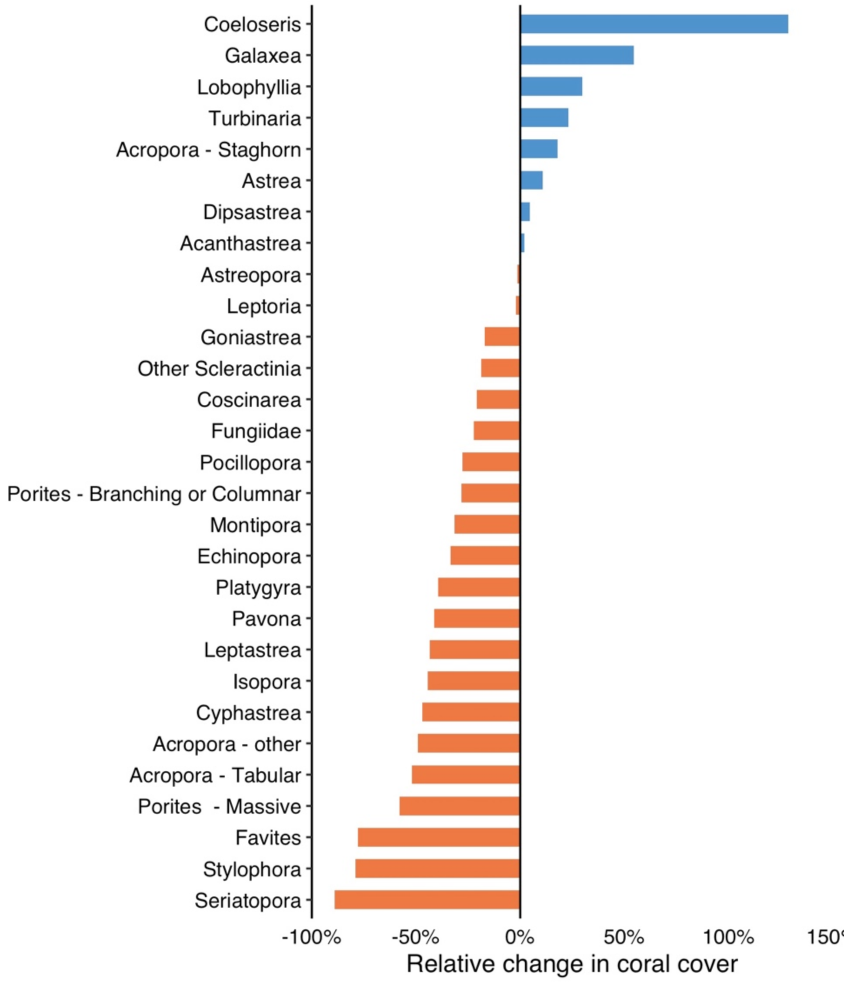


Figure 4.11 Variation in the relative change in coral cover among coral taxa from 2020 to 2021 in the Coral Sea Marine Park. Data are based on point-intercept transects across 13 reefs. Data are pooled across habitats, sites and reefs.

4.2 Macroalgal assemblages

Macroalgal cover – There is growing concern that the increasing frequency and intensity of temperature-induced bleaching events and subsequent declines in coral cover of reefs globally will lead to an increasing number of reefs becoming dominated by other benthic taxa, namely macroalgae, which rapidly colonise dead coral skeletons (Hughes et al. 2017, 2018). There was, however, little evidence of regional increases in macroalgal cover across shallow reef habitats in the CSMP following the 2020 bleaching event, with total macroalgal cover increasing from 6.60% in 2020 to 7.18% in 2021 (Figure 4.12). The dominant macroalga in shallow habitats of the CSMP was the green calcified alga *Halimeda* spp., accounting for

98% and 82% of all macroalgae recorded in 2020 and 2021, respectively. The cover of *Halimeda* was relatively consistent between years, with regional cover in 2021 being low in the northern and southern CSMP (<1% and 3.6%, respectively), and moderate (7.6%) in the central CSMP. There was, however, considerable spatial variation in the cover (Marion Reef: <1%; Chilcott Reef: 12.9%) and temporal change in the cover of *Halimeda* among reefs within the central CSMP (Figure 4.12a). *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 the three CSMP regions in both 2020 and 2021 (Figure 4.12b). The only exception was the sheltered back-reef sites at Saumarez Reef (sites 3 and 5), where the cover of the green alga *Caulerpa* was *ca.* 10% and had increased from 18% to 46% cover on the reef slope at Saumarez 5. *Caulerpa* has a creeping habit and can quickly grow to occupy areas free of other benthic taxa (i.e., hard corals, soft corals, sponges). The cause of the higher and increasing abundance of *Caulerpa* at these sheltered reef slope sites is unknown and may be related to numerous factors, such as local variation in nutrient availability (e.g., through upwelling) and/or reduced herbivory, or reflect differential recruitment and growth of *Caulerpa* among locations. With the exception of the high cover of *Caulerpa* on the reef slope at a single site, the overall cover of fleshy seaweeds was relatively low throughout the CSMP compared to other oceanic reefs, such as Elizabeth and Middleton Reefs, and Lord Howe Island to the south (Hoey et al. 2011, 2018).

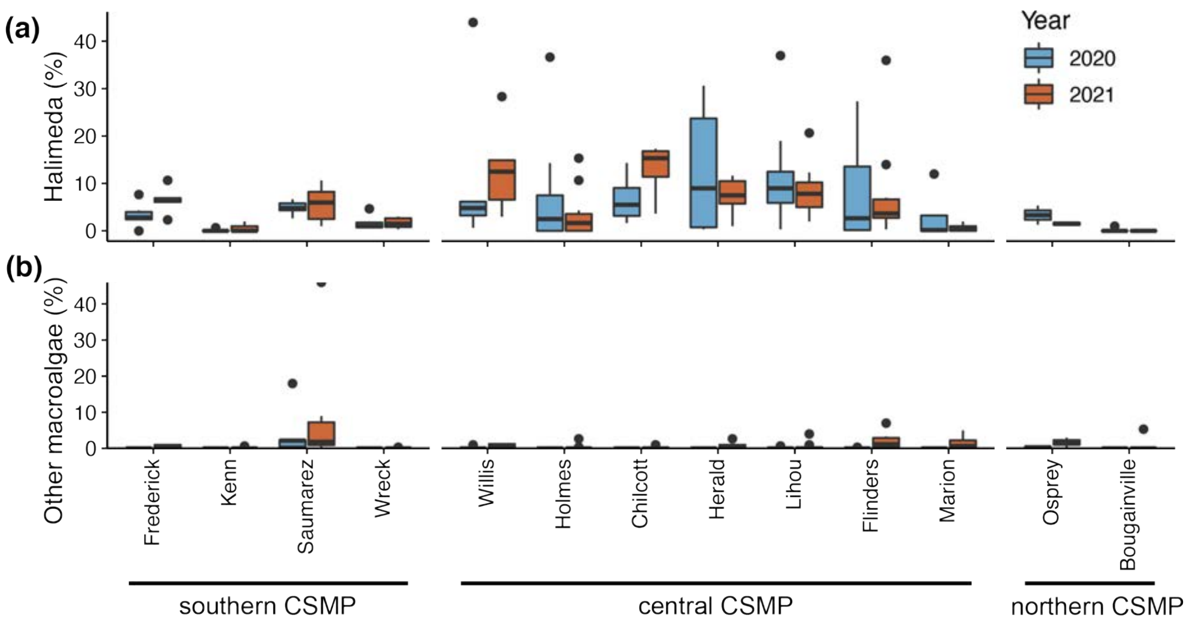


Figure 4.12 Variation in the cover of **(a)** *Halimeda* and **(b)** 'other' macroalgae between years (2020 vs 2021) and among the thirteen reefs surveyed in the Coral Sea Marine Park. Reefs are arranged into three regions (southern, central, and northern) within the Coral Sea Marine Park. Data are based on 1-6 sites per reef and pooled between habitats. The photograph shows a patch of *Halimeda* growing at the base of dead corals on Saumarez Reef in February 2021. Image credit: Andrew Hoey

4.3 Coral Reef Fishes

The loss of coral cover and shifts in the composition of coral assemblages following bleaching events has been shown to have the greatest and most immediate effects on fishes and invertebrates that rely on these corals for shelter and/or food (e.g., Pratchett et al. 2008, 2011b), leading to shifts in the community composition from coral specialists to habitat generalists (e.g., Bellwood et al. 2006a, 2012; Richardson et al. 2018). In particular, the loss of fast-growing tabular and staghorn *Acropora* have been shown to reduce the three-dimensional structure and functionality of reef habitats (Hughes et al. 2018). Reductions in live coral and the physical structure they provide is likely to have flow-on effects to populations of reef fishes and other reef-associated species, however such effects may take several years to be realised.

4.3.1 Fish diversity, abundance and biomass

A total of 36,707 and 25,020 fishes were recorded across the 43 sites within the CSMP in 2020 and 2021, respectively. Ten fish species that had not been recorded during surveys or observations on the previous voyages (2018-2020) were recorded during the 2021 surveys, taking the total fish species recorded in the CSMP during the past four years of surveys to 631 species (Appendix 4). In 2020, the species richness, density and biomass of reef fishes increased from the southern CSMP, to the central CSMP, and to the northern CSMP (Figure 4.13), and is consistent with well-known latitudinal gradients in the diversity of marine species (Hillebrand 2004) and reef fishes (Bellwood and Hughes 2001). However, in 2021 this latitudinal pattern was disrupted due to declines in the richness, abundance and biomass of reef fishes in the central CSMP, such that these three metrics of reef fish communities were similar between the southern and central CSMP reefs in 2021 (Figure 4.13).

The number of fish species recorded per site remained stable or increased from 2020 to 2021 in both the southern (2020: 60 species; 2021: 68 species per site) and northern CSMP (2020: 93 species; 2021: 94 species), however there was a 7% decline in species richness in the central CSMP (2020: 80 species; 2021: 73 species; Figure 4.13a). Fish species richness was relatively consistent among reefs and years in the southern and northern CSMP, but there was considerable variation among reefs in the central CSMP (Figure 4.14a). Declines in fish species richness were most pronounced at Willis Islets, Flinders, and Marion Reefs, while at the other central CSMP reefs fish species richness remained broadly comparable (Holmes Reef, Herald Cay), or experienced a relatively small decline (Lihou Reef; Figure 4.14a).

Similarly, the total biomass of reef fish remained relatively stable in the southern CSMP from 2020 to 2021, but declined by 19% in the central CSMP from 12.8 kg.100m^{-2} in 2020 to 9.6 kg.100m^{-2} in 2021 (Figure 4.13b), and by 16% in the northern CSMP from 26.6 kg.100m^{-2} in 2020 to 21.8 kg.100m^{-2} in 2021 (Figure 4.13b). The total fish biomass was relatively consistent among the four southern CSMP reefs in 2021, representing small increases on 2020 levels on Kenn and Saumarez Reefs, and small declines on Frederick and Wreck Reefs (Figure 4.14b). Total fish biomass was, however, highly variable among reefs and years in the central CSMP (Figure 4.14b). Total fish biomass decreased substantially on Flinders, Marion, and Holmes Reefs, and Willis Islets from 2020 to 2021 (43, 33, 22 and 19% declines, respectively), but was comparable or experienced small-moderate declines on the other three CSMP reefs (i.e., Lihou Reef, Herald Cays, and Chilcott Islets) over the same timeframe (Figure 4.14b).

Total fish abundance remained relatively stable across the southern and northern CSMP from 2020 to 2021, yet declined by 31% on central CSMP reefs (Figure 4.13c). Similar to fish biomass, the abundance of reef fishes was relatively consistent among the four southern CSMP reefs in 2021, representing small declines from 2020 levels on Frederick, Saumarez, and Wreck Reefs, that were offset by a doubling of fish abundance on Kenn Reef (Figure 4.14b). The marked increase on Kenn Reef is at least partly attributable to the very low fish abundance recorded during the 2020 surveys. The total reef fish abundance was highly variable among central CSMP reefs, with the greatest declines in abundance being recorded at Flinders, Holmes and Lihou Reefs, and Willis Islets.

These temporal changes in fish richness, biomass and abundance were relatively consistent among sites within each reef (Figure 4.15). The only exceptions being one site at Herald Cays (Herald 1) where the biomass of reef fishes increased markedly due to a school of 23 bumphead parrotfish, *Bolbometopon muricatum* (total length 80-110cm) being recorded along the reef slope (Figure 4.16), and variable responses of reef fish richness and biomass at Holmes Reef that may reflect differences in coral loss among sites (Figure 4.6).

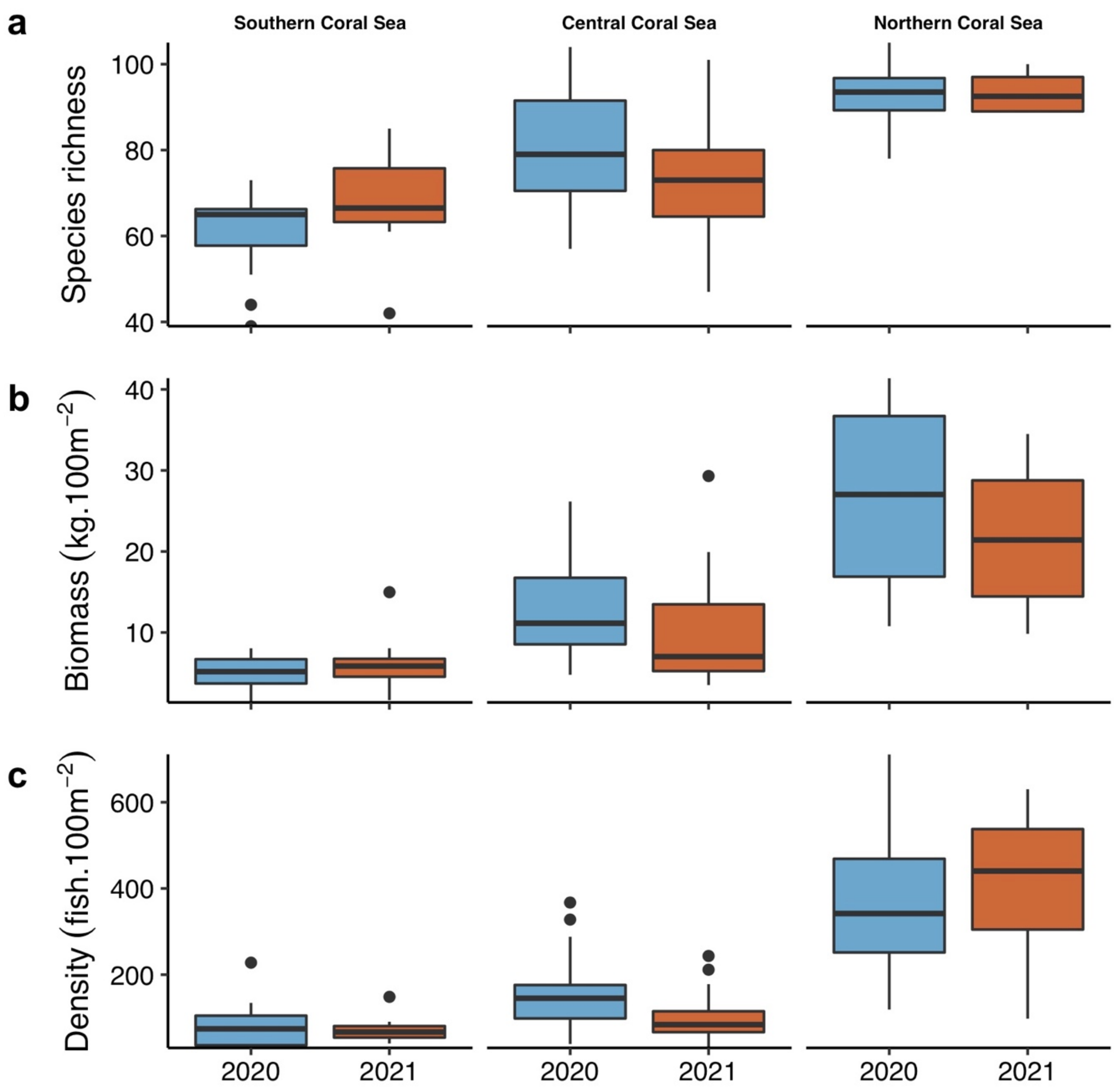


Figure 4.13 Spatial and temporal variation in **(a)** species richness, **(b)** biomass, and **(c)** abundance of coral reef fishes and sharks among the three regions of the Coral Sea Marine Park during 2020 and 2021.

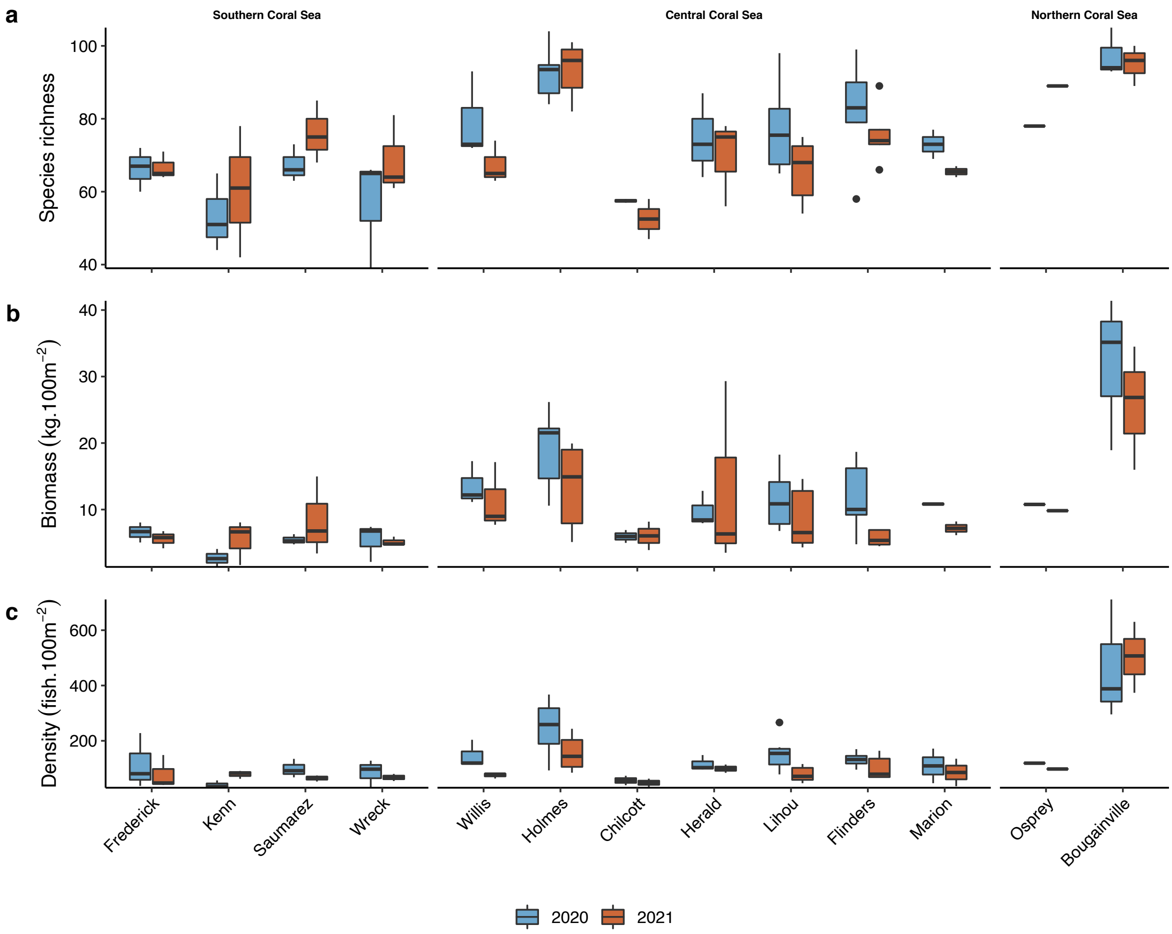


Figure 4.14 Temporal variation in **(a)** species richness, **(b)** biomass, and **(c)** abundance of coral reef fishes and sharks among the 13 reefs surveyed in the Coral Sea Marine Park during 2020 and 2021.

Interestingly, the greatest declines in richness, biomass and abundance of reef fishes were recorded at Flinders and Holmes Reefs, and Willis Islets (Figures 4.14, 4.15), three of the reefs that experienced the greatest declines in live coral cover from 2020 to 2021 (Figure 4.4). Despite the relatively high loss of corals recorded at these reefs (~42% - 59%), these declines in richness, abundance and biomass are greater than would be expected, especially only 12 months post-bleaching when most of the structure provided by the corals is still intact (Pratchett et al. 2011, 2014). Moreover, Frederick Reef in the southern CSMP experienced the greatest decline in coral cover (2020: 20.9% cover; 2021: 8.1% cover; a decline of 73.5%) yet fish communities remained relatively stable between 2020 and 2021. More detailed analyses of potential drivers will be necessary to understand the reasons for the observed changes in reef fish communities, and may need to consider factors such as reef size, isolation, hydrodynamics and habitat condition (e.g. Ceccarelli et al. 2016; Lam et al. 2018; Zinke et al. 2018).

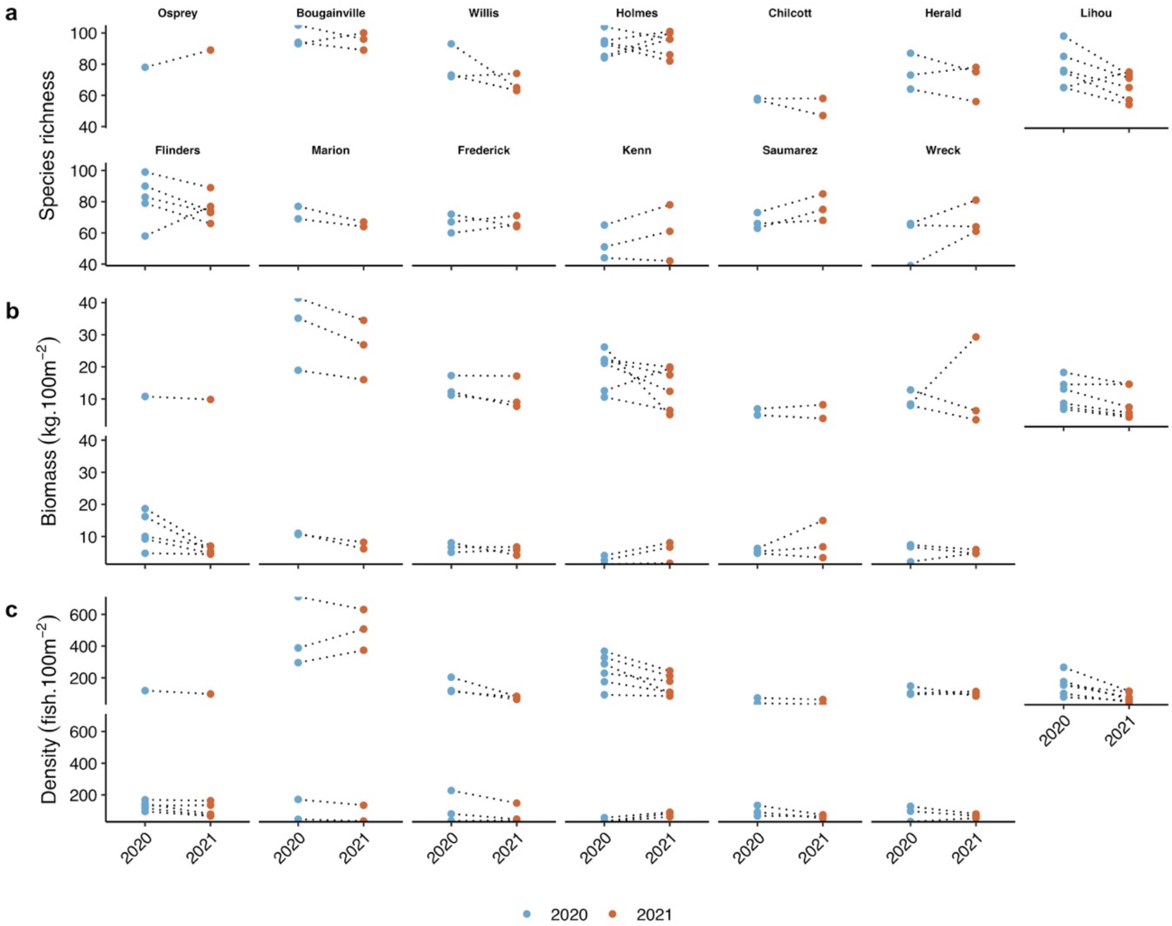


Figure 4.15 Temporal variation in **(a)** species richness, **(b)** biomass, and **(c)** abundance of coral reef fishes and sharks among the 43 sites across 13 reefs surveyed in the Coral Sea Marine Park during 2020 and 2021.

Figure 4.16
School of bumphead parrotfish, *Bolbometopon muricatum* (up to 110cm total length), surveyed on the reef slope at Herald 1 were largely responsible for the increase in biomass at that site in 2021. Image credit: Andrew Hoey

4.3.2 Fish functional groups

Fishes were categorised into eleven functional groups based on their diet, morphology and feeding behaviour. Planktivorous fishes (e.g., fusiliers, anthias and some damselfishes) were the most abundant functional group on reefs in the CSMP in both 2020 and 2021, accounting for up to ~80% of the total fish abundance (Figure 4.17a). The next most abundant groups were the grazing herbivores (primarily surgeonfishes and rabbitfishes) that typically feed on reef substrata covered by an epilithic algal matrix (i.e., short algal turfs and associated detritus, sediment and microbes, EAM; Wilson et al. 2003), and algal farming damselfishes (Figure 4.17a). All three of these groups declined in abundance on reefs in the central, and to a lesser extent southern, CSMP between 2020 and 2021 (Figure 4.17a). In contrast, the abundance of planktivorous fishes increased in the northern CSMP, while the abundance of grazers and algal farming damselfishes decreased (Figure 4.17a). The majority of planktivorous species and algal farming damselfishes are small-bodied and hence are not major contributors to reef fish biomass.

The biomass of reef fishes was more evenly distributed among functional groups on the 13 CSMP reefs, with grazers, scrapers, excavators and planktivores collectively contributing >60% of the total biomass on all reefs (Figure 4.17b). While there was an increase in the biomass of excavating fishes at Herald Cays attributable to a school of the giant bumphead parrotfish *Bolbometopon muricatum* between 2020 and 2021, the observed declines in biomass on central CSMP reefs (Figure 4.13, 4.14) was primarily driven by declines in the biomass of grazing herbivorous fishes (Figure 4.17b, 4.18), in particular *Acanthurus lineatus* and *Acanthurus nigrofuscus*. Similar declines in the biomass of grazing fishes were not evident on the four southern CSMP reefs or Bougainville Reef in the northern CSMP. Declines in biomass from 2020 to 2021 on central CSMP reefs were also evident for planktivores and corallivores (but not piscivores), however the magnitude of the decrease was greatest for grazing fishes (Figure 4.18). While the declines in those fishes that have a direct reliance on live corals for food (i.e., corallivores) and/or habitat (i.e., small bodied planktivores) may be expected following large-scale coral mortality (e.g., Pratchett et al. 2011, 2014), the observed declines in the biomass of grazing fishes is difficult to reconcile. Several previous studies have reported substantial increases in the abundance and/or biomass of grazing fishes following large-scale coral mortality (e.g., Adam et al 2011; Gilmour et al. 2013), with these increases being related to an increase in the availability of EAM communities (their preferred feeding substrata) that rapidly colonise the dead coral skeletons (Diaz-Pulido and McCook 2002). This positive response of grazing fishes to increases in dietary resources following coral loss is supported by increases in growth rates of individual fishes following coral bleaching events on the GBR and Chagos Archipelago (Taylor et al. 2020). The reason for the declines in grazing fish populations on CSMP reefs following the 2020 bleaching event is unclear and warrants further investigation.

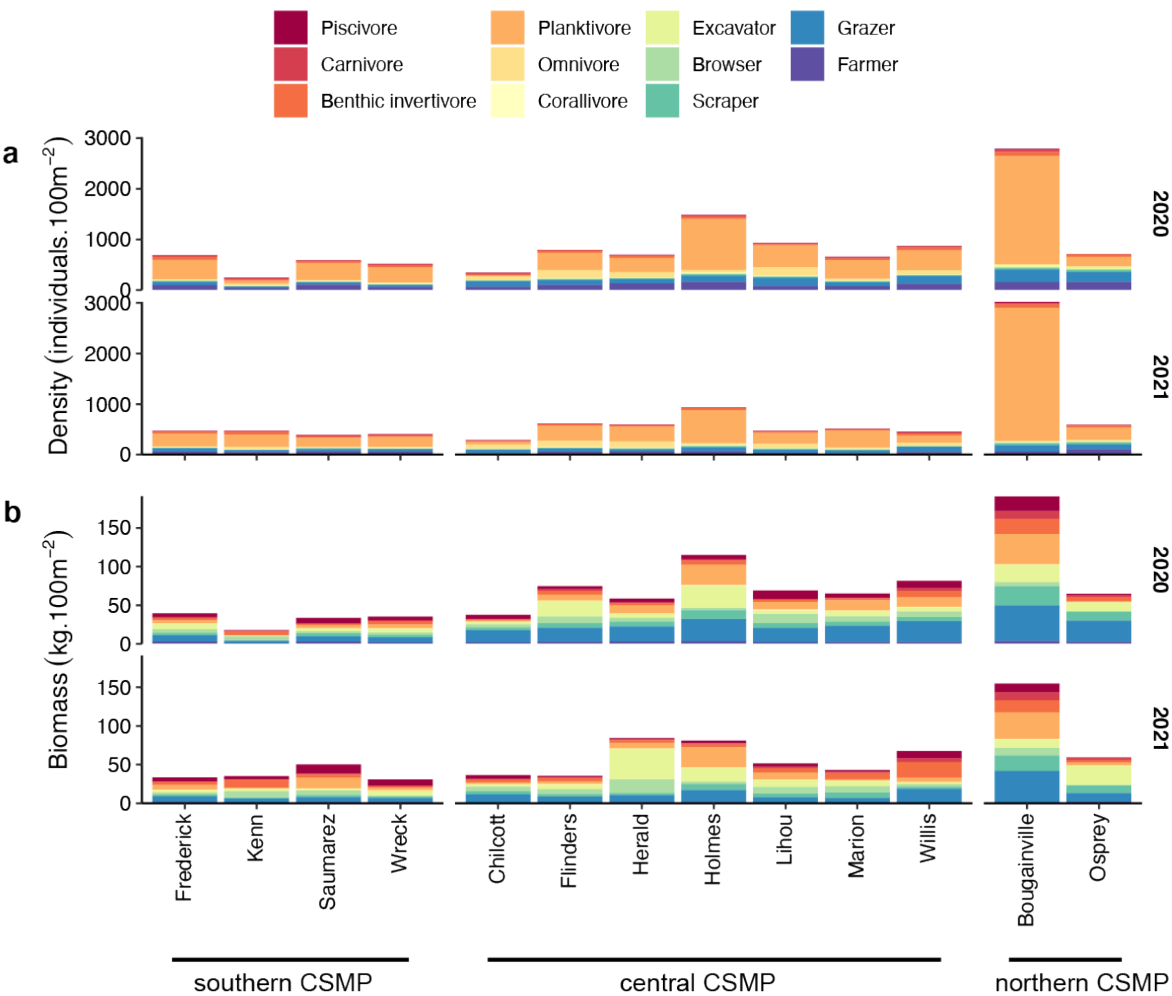


Figure 4.17 Temporal variation (2020 vs 2021) in the functional composition of reef fish assemblages across 13 reefs in the Coral Sea Marine Park based on **(a)** abundance, and **(b)** biomass. Values for each reef are averaged across habitats and sites.

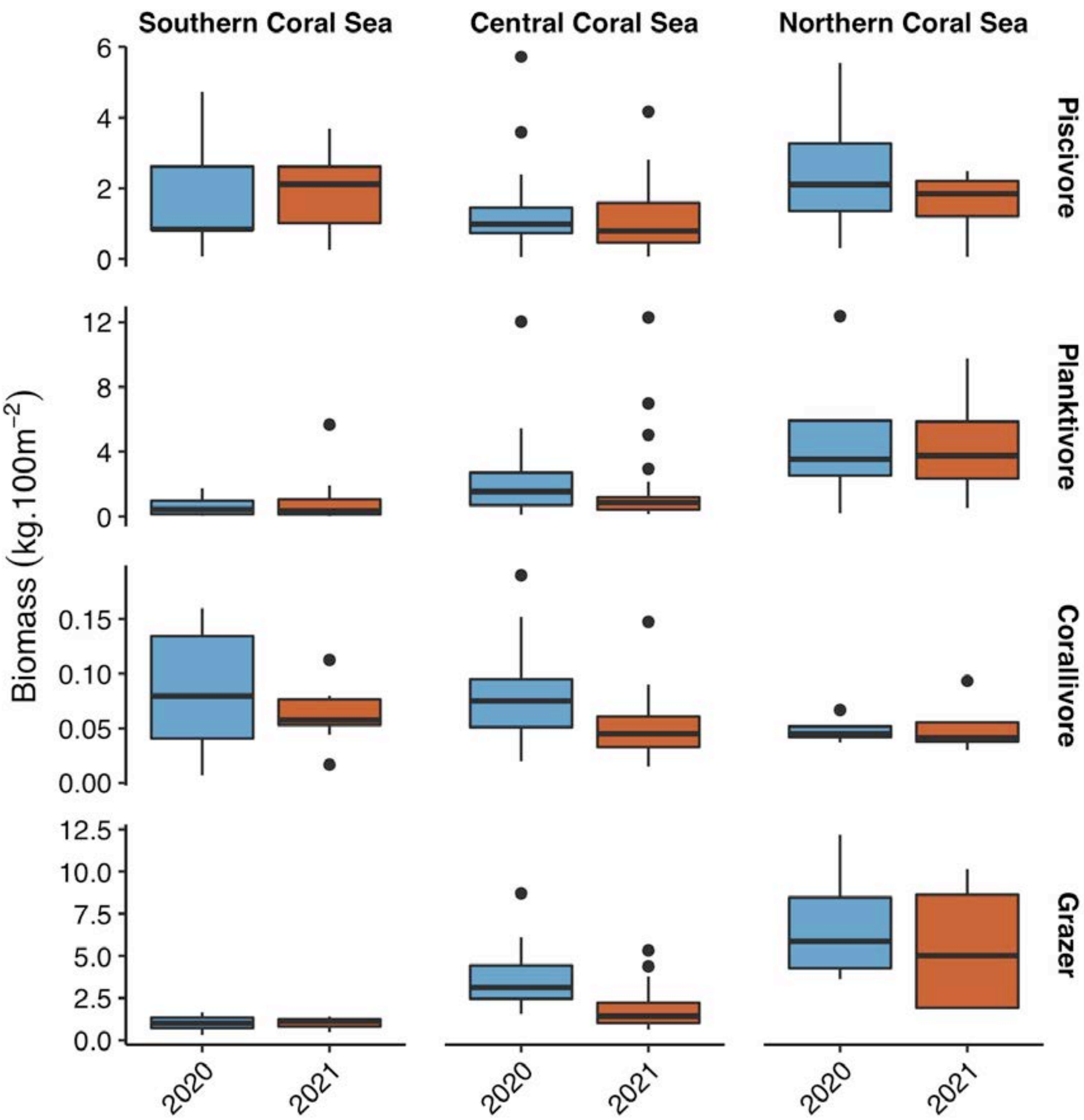


Figure 4.18 Spatial and temporal variation in the biomass of **(a)** piscivorous, **(b)** planktivorous, **(c)** corallivorous, and **(d)** grazing fishes among the three regions of the Coral Sea Marine Park during 2020 and 2021.

4.3.3 Fish community composition

Despite changes in the species richness and abundance of different functional groups of reef fishes on CSMP reefs from 2020 to 2021, the species composition of fish communities remained relatively stable over the same period, with almost complete overlap between years (Figure 4.19). Similarly, the species composition of individual fish families on CSMP reefs showed limited change between 2020 and 2021 (Figure 4.20).

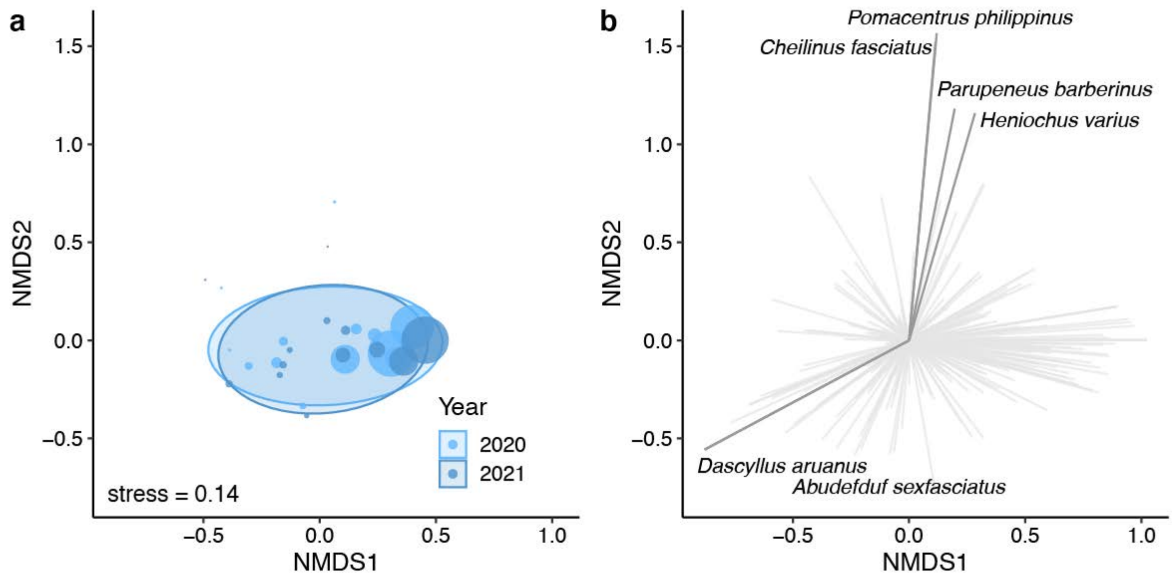


Figure 4.19 Non-metric multidimensional scaling (nMDS) plots showing **(a)** variation in reef fish assemblages among 13 reefs and between years (2020 vs 2021) in the Coral Sea Marine Park. The size of individual points are proportional to fish abundance on each reef; **(b)** vectors indicate key species that account for variation in fish composition displayed in the corresponding left-hand side plots.

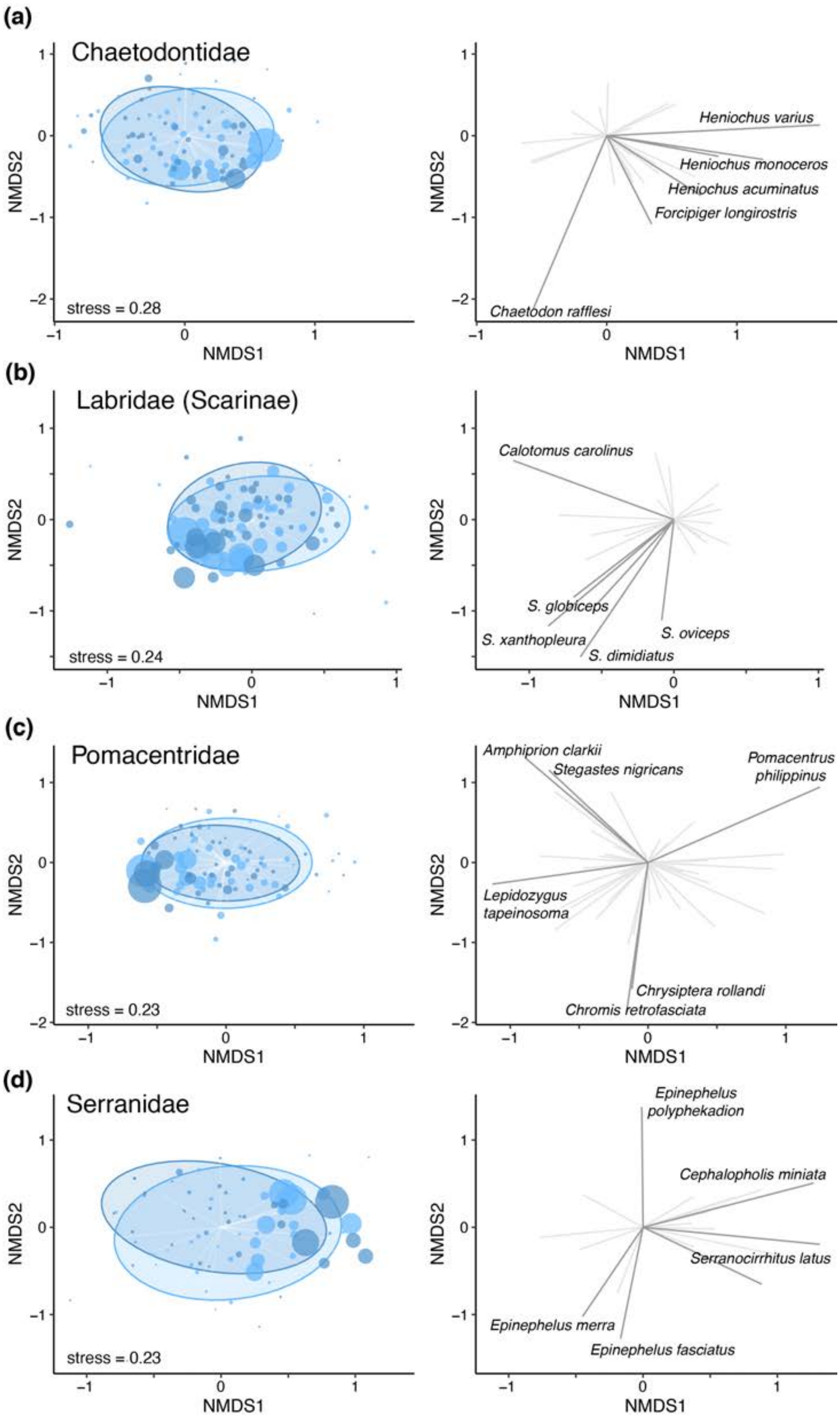


Figure 4.20 Variation the species composition of (a) butterflyfishes (f. Chaetodontidae), (b) parrotfishes (f. Labridae: Scarinae), (c) damselfishes (f. Pomacentridae), and (d) groupers (f. Serranidae) among 13 reefs and years (2020 vs 2021) in the Coral Sea Marine Park. The size of individual points in the left-hand plots are proportional to the abundance of each family at each site. 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 4.21 Photos of the coral dwelling planktivorous damselfish *Dascyllus reticulatus*. Left: Group of *D. reticulatus* associating with one of the remaining live *Pocillopora* colonies; Right: a solitary *D. reticulatus* associating with a dead *Acropora* colony. Species such as *D. reticulatus* are often among the first to be lost following large-scale coral mortality. Image credits: Deborah Burn (left), Morgan Pratchett (right)

4.4 Other reef taxa

4.4.1 Sea snakes

A total of 28 sea snakes were recorded across the 13 CSMP reefs in 2021 compared to 20 individuals in 2020, of which the vast majority were the olive sea snake *Aipysurus laevis* (2020: 95%; 2021: 90% of sea snakes observed). Consistent with previous surveys in 2019 and 2020 (Hoey et al. 2020) 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 (and presumably absent) at all other reefs in the central and northern CSMP (Figure 4.22). Importantly, the densities of sea snakes recorded on each of the four reefs in the southern CSMP (i.e., Frederick, Kenn, Saumarez, and Wreck Reefs) and Marion Reef in the central CSMP remained stable or increased slightly from 2020 to 2021 (Figure 4.22). The number of sea snakes increased on Saumarez (2020: 5 individuals; 2021: 10 individuals), Wreck (2020: 1 individual; 2021: 3 individuals), and Frederick Reefs (2020: 3 individuals; 2021: 5 individuals). Densities of snakes observed on Marion Reef were consistent between years (3 individuals in both

2020 and 2021), however their distribution among sites was more variable in 2021 leading to the lower median value shown in Figure 4.22.

The marked latitudinal gradient in the abundance of sea snakes within the CSMP 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 sea snakes of 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). However, the distribution of the olive sea snake *Aipysurus laevis,* which is by far the most abundant species observed in shallow reef habitats in the CSMP, extends into warmer, lower latitude waters of the north Western Australian coast, Timor Sea, Gulf of Carpentaria, and southern New Guinea (O'Shea 1996; Lukoschek et al. 2007) suggesting that environmental tolerances are unlikely to explain the marked patterns in the distribution and abundance of this species within the CSMP. While dedicated research would be required to identify the drivers of their latitudinal distribution in the CSMP, their potential susceptibility to increasing water temperatures highlights the need to carefully monitor sea snake populations in the southern CSMP.

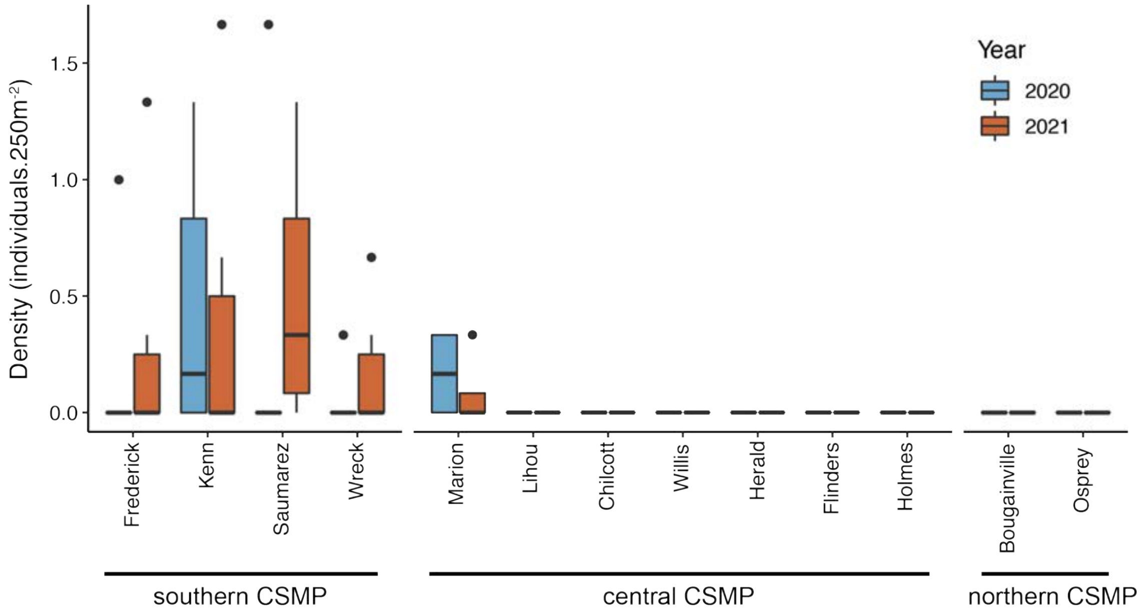


Figure 4.22 Spatial and temporal (2020 vs 2021) variation in the abundance of sea snakes among 13 reefs surveyed in the Coral Sea Marine Park. The majority of sea snakes surveyed were *Aipysurus laevis* (left), while *Acolyptophis peronii* (right) *and Emydocephalus annulatus* (not shown) were also recorded. Image credits: Deborah Burn

4.4.2 Macro-invertebrates

Giant Clams – A total of 688 giant clams (*Tridacna* spp.) were recorded across the 13 CSMP reefs in 2021, a decrease of approximately 20% from the 877 clams recorded across the same reefs in 2020. This decline was driven by a 40% decrease in the densities of clams recorded in the southern CSMP (2020: 8.5 clams.100m^{-2}; 2021: 5.4 clams.100m^{-2}), and in particular Kenn Reef (2020: 28.3 clams.100m^{-2}; 2021: 15.9 clams.100m^{-2}; Figure 4.23a). In contrast, the recorded densities of clams increased slightly in both the central (2020: 0.9 clams.100m^{-2}; 2021: 1.2 clams.100m^{-2}) and northern CSMP (2.0 clams.100m^{-2}; 2.4 clams.100m^{-2}). There was also an increase in the density of recently dead clams recorded in the southern (2020: 0.2 dead clams.100m^{-2}; 2021: 0.6 clams.100m^{-2}) and northern CSMP (2020: 0.3 dead clams.100m^{-2}; 2021: 0.5 clams.100m^{-2}), that may reflect increased mortality following the 2020 bleaching event in these regions (Figure 4.24). However, there was no change in the density of recently dead clams at reefs in the central CSMP, and the increase in the density of dead clams does not reconcile the observed declines in live giant clams in the southern CSMP. The apparent decrease in the density of giant clams is largely driven by variation in the density of clams at Kenn Reef, possibly due to inherent sampling variation at this site where densities of clams are very high, but also very patchy.

The density of giant clams (*Tridacna* spp.) in 2021 was relatively consistent across the CSMP with < 3.5 clams per transect (i.e., < 3.5 clams.100m^{-2}) being recorded on 12 of the 13 reefs (Figure 4.23a). The only exception to this was Kenn Reef in the southern CSMP where an average of 15.9 clams per transect were recorded. The causes of the 4.5-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.

The vast majority of giant clams recorded in the CSMP in 2021 were *Tridacna maxima* and *Tridacna squamosa*, collectively accounting for 665 (96.7%) of the 688 clams recorded. The other species recorded were *Tridacna derasa* (22 individuals, 3.3%), and *Tridacna gigas* (1 individual at Lihou Reef, 0.1%). No *Hippopus hippopus* or *Tridacna crocea* were recorded across the 13 CSMP reefs in either 2020 or 2021. When interpreting these density estimates and the species composition of giant clams recorded, consideration needs to be given to the habitats surveyed. Our surveys were designed primarily to provide robust estimates of coral and associated reef fish assemblages, and as such were conducted on areas of contiguous reef with a defined reef crest adjacent to a reef slope. However, many giant clam species, and *Tridacna gigas* in particular, are most abundant in lagoonal and shallow reef flat habitats (e.g., Braley 1987), and would require dedicated surveys in these habitats to assess spatial and temporal changes in their abundances.

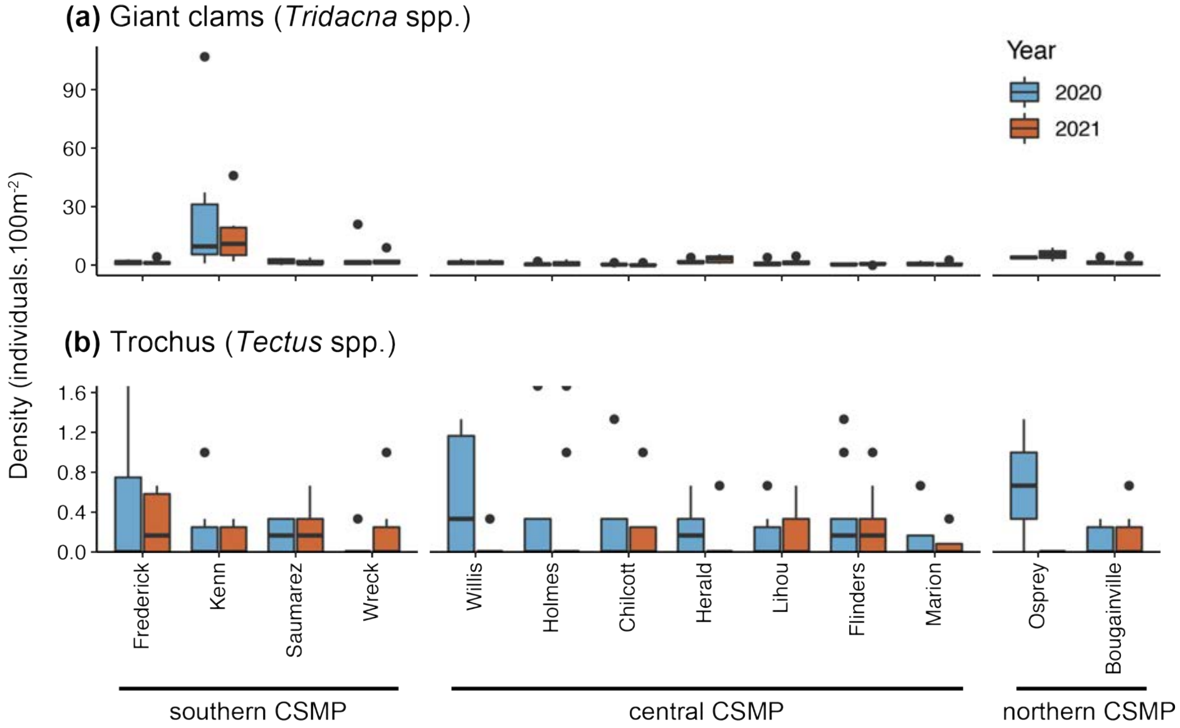


Figure 4.23 Spatial and temporal variation in the abundance of (a) giant clams, and (b) Trochus among the 13 reefs surveyed in the Coral Sea Marine Park during 2020 and 2021.

Figure 4.24 Photograph of a bleached clam (*Tridacna* sp.) next to a bleached *Acropora* coral colony on the reef crest at Flinders Reef in February 2020. Several clams likely died due to the 2020 bleaching, as reflected in the increased number of dead clams recorded. Image credit: Dani Ceccarelli

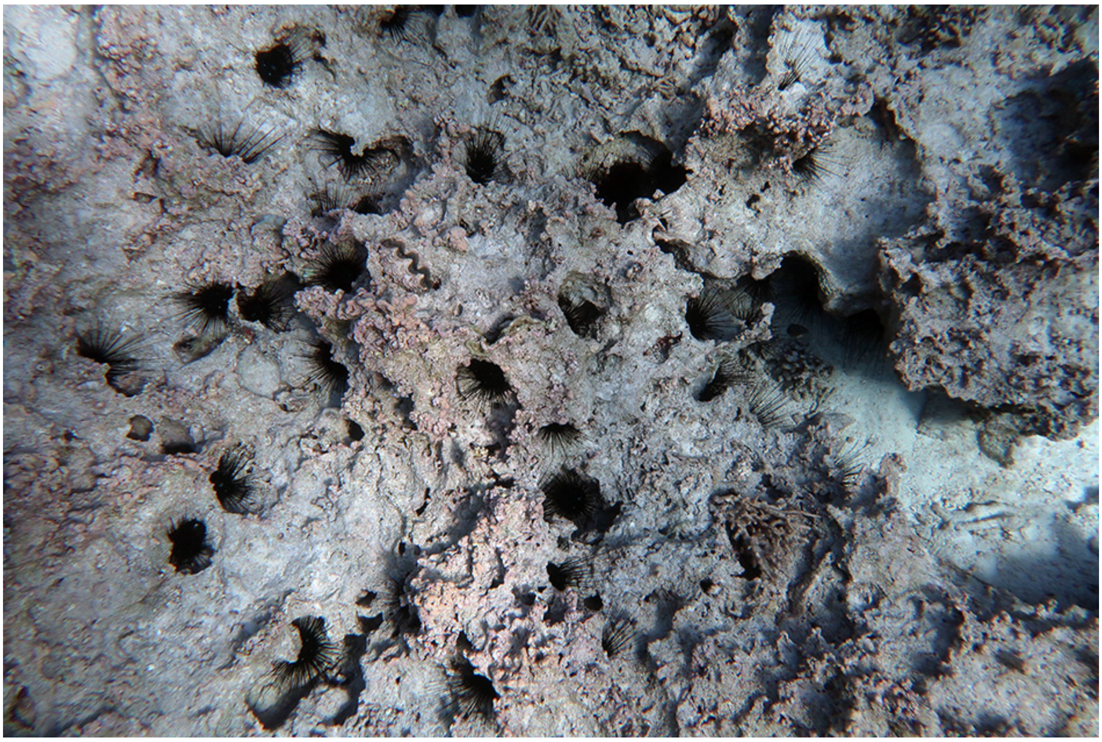
Trochus – A total of 49 *Tectus* spp. (formerly *Trochus*) were recorded across the 13 CSMP reefs in 2021, compared to 71 individuals recorded across the same reefs in 2020: a 30% decline. This decline was primarily driven by decreases in the numbers of *Tectus* surveyed at Willis (2020: 0.56 individuals.100m^{-2}; 2021: 0.06 individuals per100m^{2}) and Chilcott Islets (2020: 0.50 individuals.100m^{-2}; 2021: 0.17 individuals.100m^{2}) in the central CSMP and Osprey Reef in the northern CSMP (2020: 0.50 individuals.100m^{-2}; 2021: 0.06 individuals.100m^{-2}). In contrast, increases in the density of *Tectus* was recorded at Wreck Reef in southern CSMP, and Lihou Reef in the central CSMP, and densities remained the same at Kenn, Saumarez, Flinders, and Bougainville (Figure 4.23b). We do not believe the observed decline in *Tectus* spp. densities is a cause for concern and likely reflects the low and highly variable densities of *Tectus* among reefs and sites within the CSMP (Figure 4.23b).

Sea urchins – The density of long-spined sea urchins (*Diadema* spp.) was generally low (< 1 urchin.100m^{-2}) across the 13 CSMP reefs in both 2020 and 2021, with no *Diadema* recorded on central or northern CSMP reefs in 2021 (Figure 4.25a). The only exceptions to this were Kenn and Wreck Reefs in the southern CSMP, accounting for 1,943 of the 1,964 (i.e., 99%) urchins recorded in 2021. Densities of *Diadema* decreased from 154.3 to 97.3 urchins per 100m^{2} at Kenn Reef, and increased from 4.3 to 10.6 urchins per 100m^{2} at Wreck Reef between 2020 and 2021. 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), several hundred kilometres to the south. The differences in *Diadema* densities among reefs in the southern CSMP, and the temporal variation at Kenn and Wreck Reefs warrants further investigation.

Many sea urchin species (including *Diadema* spp.) are herbivorous, and as such are often viewed as having a positive effect on coral reefs through their ability to reduce the biomass of macroalgae and prevent shifts to macroalgae dominance (e.g., McClanahan et al 1994; Humphries et al. 2020; Williams et al. 2021). However, some urchins (including *Diadema*) also erode the internal structure of the reef (in contrast to parrotfishes that erode external surfaces; Hoey and Bellwood

2008) 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 159 sea cucumbers (Holothuroidea) from 9 species were recorded across the 13 CSMP reefs in 2021, an increase from the 125 sea cucmbers recorded across the same sites in 2021. The most abundant species were *Thelenota ananas* (36.5 %), *Actinopyga mauritiana* (24.5%), *Holothuria atra* (20.1%), *Holothuria whitmaei* (6.9%) and *Bohadschia argus* (6.9%). The other species recorded were *Stichopus chloronotus, Holothuria edulis, Pearsonothuria graeffei*, and *Thelenota anax*. The density of sea cucumbers within the shallow reef habitats surveyed was generally low across the CSMP (average 0.56 individuals.100m^{-2}) ranging from 0.05 individuals per 100m^2 at Frederick Reef to 1.44 individuals per 100m^2 at Marion and Osprey Reefs, and showed limited change between years (Figure 4.25b). As noted previously (Hoey et al. 2020), these density estimates are substantially lower than those of previous dedicated sea cucumber surveys in the central CSMP (average of 1.33 individuals.100m^2 for all species combined; 1.06 individuals.100m^{-2} for *H. atra*; Skewes and Persson 2017), and likely reflect differences in the habitats surveyed, rather than a significant decline in sea cucumber populations. Robust assessments of sea cucumber populations would require dedicated surveys over these sandy habitats (*sensu* Kinch et al. 2008).



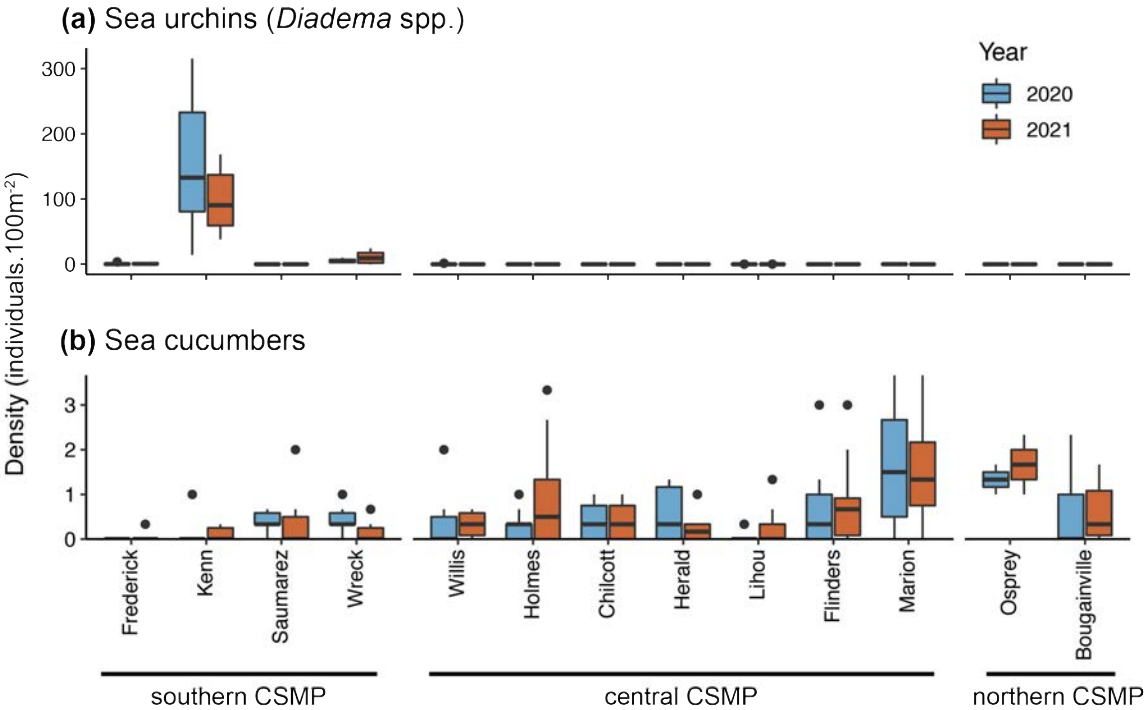


Figure 4.26 Spatial and temporal variation in the abundance of (a) sea urchins – *Diadema* spp., and (b) sea cucumbers among 13 reefs in the Coral Sea Marine Park during 2020 ands 2021. Dotted lines represent the mean regional abundance. Note that sea urchins (*Diadema* spp). were extremely rare or absent from reefs in the central and northern Coral Sea Marine Park. Photograph shows high densities of *Diadema* spp. on the reef slope at Kenn Reef in February 2021. Note the eroded reef framework in which the urchins are sheltering. Image credit: Andrew Hoey

4.5 Coral health and injury

4.5.1 Coral colony size distribution

Coral assemblages within the shallow reef habitats of the 13 CSMP reefs surveyed in 2020 and 2021 were dominated by relatively small (< 20cm in diameter) coral colonies, with few colonies larger than 40cm diameter recorded (Figure 4.26, 4.27). There were declines in the abundance of most size classes of corals in the southern and central CSMP between 2020 and 2021, with the most pronounced changes being the 5-20cm size class (Figure 4.26). The only exception to this was the smallest size class (i.e., juvenile corals <5cm diameter) that increased within the central CSMP, and may reflect the outcomes of partial mortality of larger corals following the 2020 bleaching event. In contrast, there was an increase in the abundance of 5-20cm diameter coral colonies (predominantly fast-growing *Acropora*) in the northern CSMP from 2020 to 2021, while the abundance of other size classes remained relatively stable (Figure 4.26). Collectively, these changes have resulted in a greater proportion of coral colonies in the smallest size class (<5cm diameter) in 2021 compared to the three previous years (Figure 4.27).

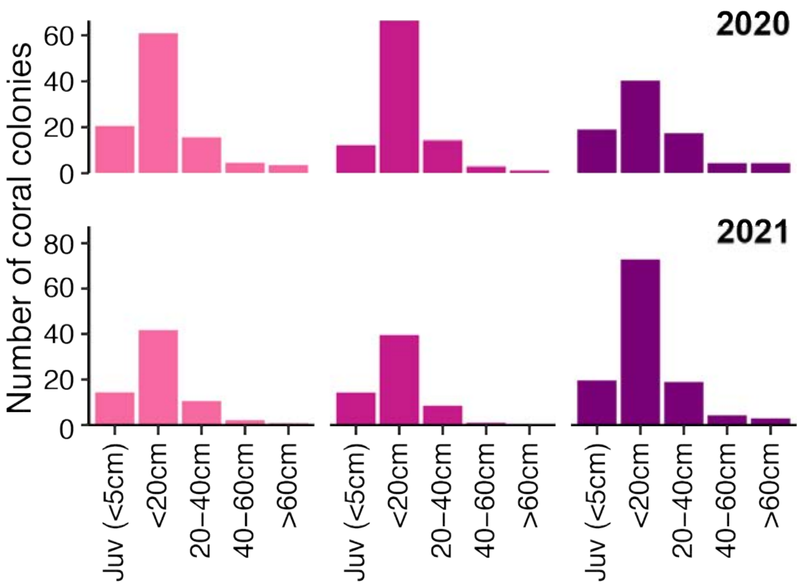


Figure 4.26 Comparison of the size distribution of coral colonies at 13 reefs within the Coral Sea Marine Park between 2020 and 2021. Data are based on the number of colonies recorded within replicate 10 x 1m belt transects within each of 43 sites (i.e., pooled across slope and crest habitats)

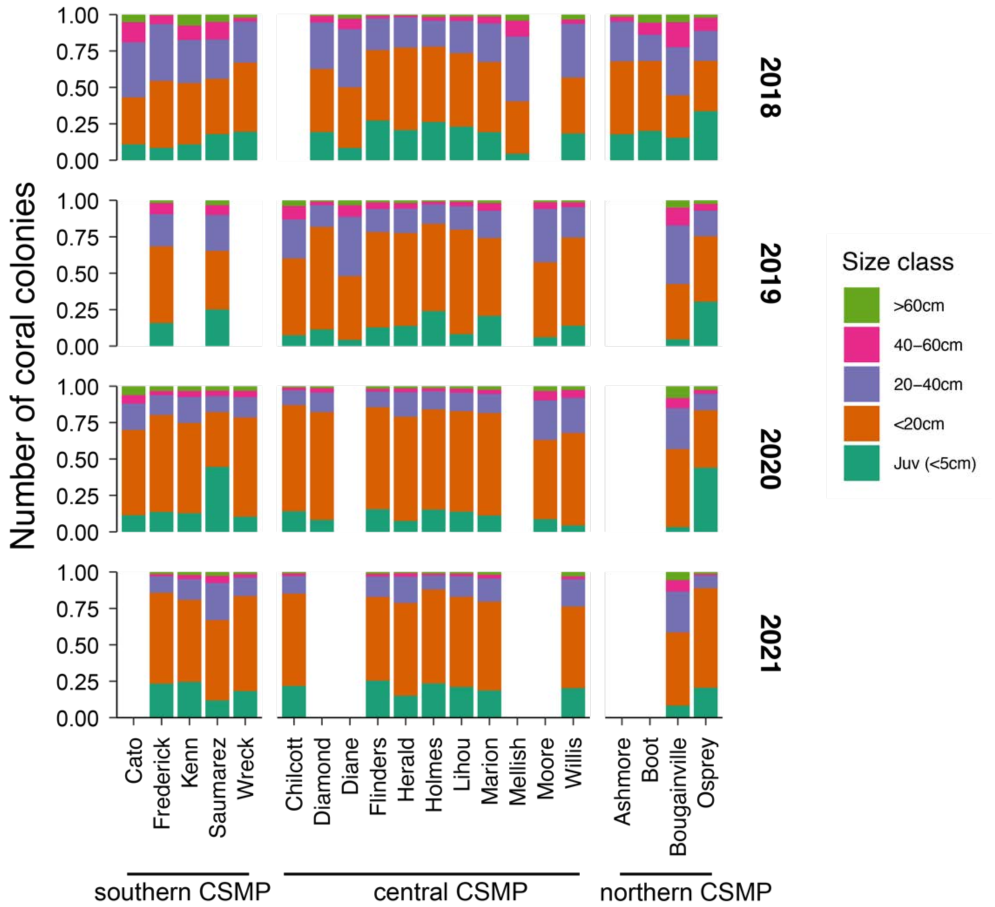


Figure 4.27 Size distribution of coral colonies at 13 reefs within the Coral Sea Marine Park between 2018 and 2021.

4.5.2 Coral condition

In contrast to 2020, the majority of coral colonies surveyed across the CSMP in 2021 were healthy, indicating no major disturbances had affected the CSMP reefs during that period. There were, however, low levels of bleaching (Pale - Recently Dead) recorded across all 13 CSMP reefs in February 2021, ranging from 1.6% of colonies surveyed at Marion Reef to 21.5% at Lihou Reef (Figure 4.28), with mean of 6.8% across the CSMP. The observed level of bleaching is within the range recorded on individual reefs in 2018 and 2019 (i.e., between 2% and 16% of colonies surveyed), and is likely within the natural range of coral injury for coral reef systems. Higher levels of bleaching at Lihou Reef in 2021 were driven by a high proportion of pale and partially bleached colonies of the thermally sensitive coral genera *Stylophora* (76% of colonies bleached) and *Acropora* (37%), which may indicate low levels of heat stress.

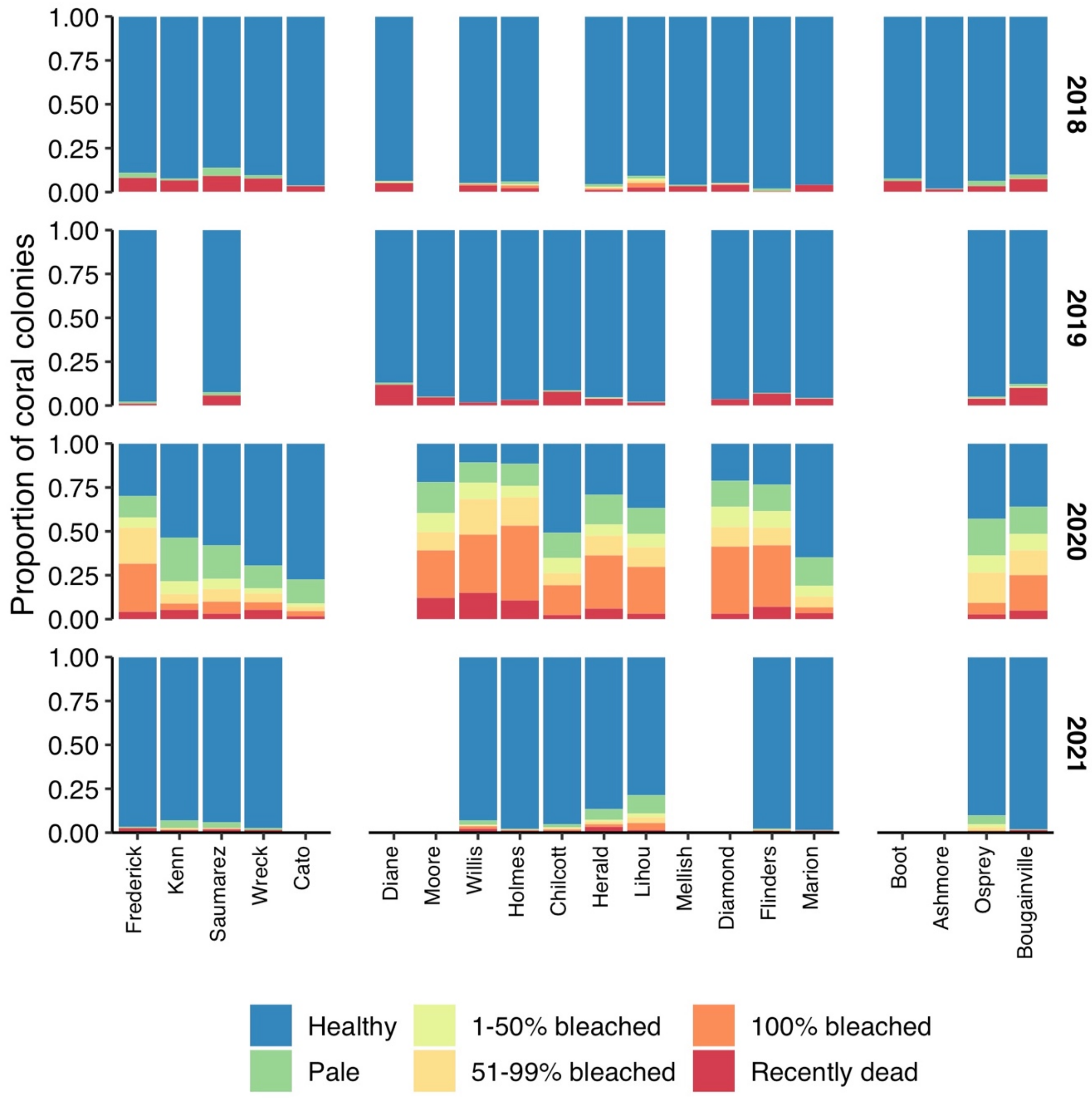


Figure 4.28 The proportion of coral colonies in each of six bleaching categories from 'healthy' to 'recently dead' observed at 20 reefs within the Coral Sea Marine Park from 2018 to 2021. Note: not all reefs were surveyed in each year, with 13 reefs surveyed in 2021.

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 (Loya et al. 2001). The incidence of bleaching recorded in the CSMP during February 2021, while relatively minor (i.e., 5.4% colonies bleached), also show that *Stylophora*

(25.3% of colonies bleached), *Acropora* (11.8%), and *Pocillopora* (9.0%) were among the worst affected coral taxa, however a number of taxa generally viewed to be less sensitive to heat stress displayed some incidence of bleaching, including branching *Porites* (6.2%), *Montipora* (11.1%), and *Galaxea* (6.5%; Figure 4.29). Although, the incidence of bleaching recorded across the 13 CSMP reefs in 2021 needs to be interpreted against a shifted baseline; namely due to the reductions in the cover of bleaching-susceptible corals following the 2020 bleaching event (Figure 4.11). The generally low incidence of bleaching among reefs and coral taxa in 2021 suggests the ongoing effects on coral assemblages will be negligible.

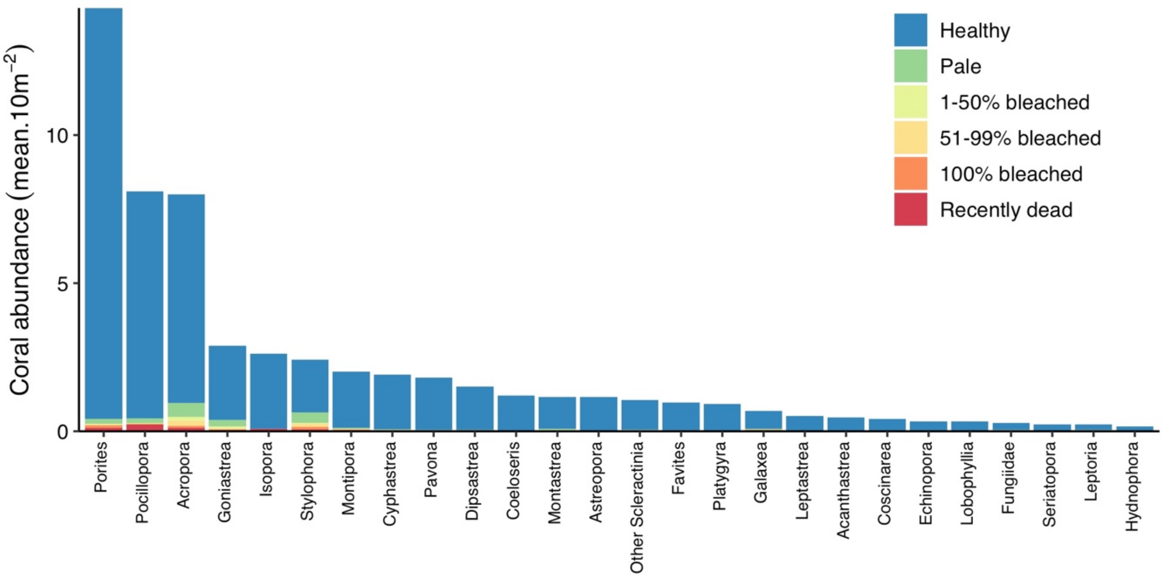


Figure 4.29 The mean abundance of coral colonies of the 26 most common coral genera in each of six bleaching categories from 'healthy' to 'recently dead' observed at sites across 13 reefs within the Coral Sea Marine Park during February 2021.

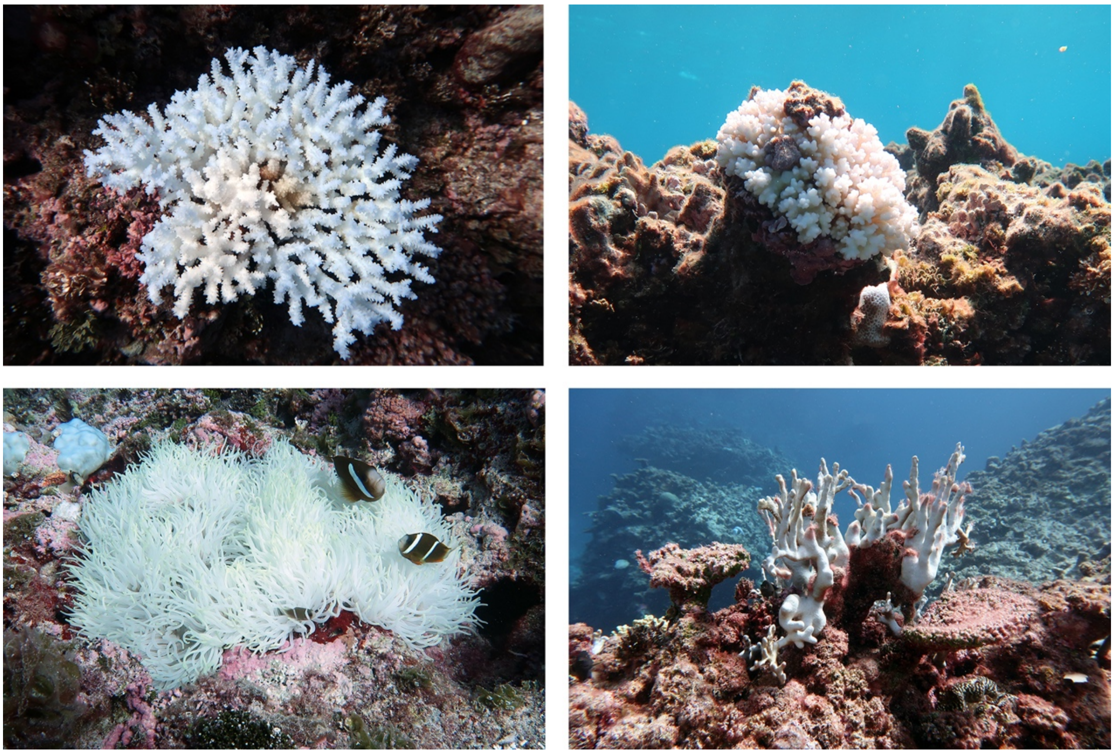


Figure 4.30 Photographs of bleached corals and anemone observed during February 2021 in the Coral Sea Marine Park. Clockwise from top left: *Acropora* at Lihou Reef; *Pocillopora* at Lihou Reef; *Millepora* at Lihou Reef- the algae growing on the branch tips indicate these areas of the colony have already died; anemone at Herald Cays. Image credits: Andrew Hoey

4.5.3 Coral recruitment

The mean density of juvenile corals (<5cm diameter) recorded across the 13 CSMP reefs in 2021 was 15.0 juveniles per $10m^2$, compared to 16.4 juveniles per $10m^2$ in 2020 (Figure 4.31). The densities of juvenile corals were broadly comparable among the three regions of the CSMP in 2021, ranging from 10.3 to 15.8 juveniles per $10m^2$ in the northern and central CSMP, respectively. There was, however, considerable variation in the density of juvenile corals both among and within sites (Figure 4.31, 4.32). With the exceptions of Saumarez Reef in the southern CSMP and Osprey Reef in the northern CSMP where substantial declines in the densities of juvenile corals were recorded, the densities of juvenile corals remained relatively consistent between 2020 and 2021 (Figure 4.32). However, the smallest juvenile corals recorded (~ 1cm diameter) would be > 1-year old (depending on taxa) meaning they would have been spawned and settled well before the 2020 bleaching event. As such, these data suggest that juvenile corals were generally resilient to the recent mass bleaching, while any potential effects of the 2020 bleaching on coral reproduction and settlement rates are yet to be realised.

Densities of juvenile corals recorded across the CSMP (15.0 juveniles per $10m^2$) are comparable to those recorded on the similarly isolated subtropical Elizabeth and Middleton Reefs (5 -15 juveniles per $10m^2$; Hoey et al. 2018), but are considerably lower than densities of juvenile corals previously recorded on mid-shelf reefs of the GBRMP (61 - 82 juveniles per $10m^2$; Trapon et al. 2013a). It should be noted, however, that surveys conducted on outer-shelf reefs of the GBRMP following the 2016 and 2017 bleaching events revealed lower densities of juvenile corals (10 - 60 juveniles per $10m^2$; Hoey et al. 2020), potentially reflecting widespread suppression of coral recruitment due to the loss of adult broodstock (Hughes et al. 2019). The lower densities of juvenile corals on CSMP reefs may be reflective of the isolated nature of these reefs and hence reliance on self-recruitment, and/or the frequency of disturbances that suppress the survivorship of small corals.

Local densities of juvenile corals are largely influenced by the abundance of adult broodstock, and hence rates of reproduction, and the supply and successful settlement of coral larvae, but may also be moderated by localised disturbances that cause elevated mortality of small corals (Trapon et al. 2013b; Harrison et al. 2018; Hughes et al. 2018). Rates of recruitment are expected to be lower on isolated, offshore reefs, reflecting lower levels of connectivity with other reefs and the limited extent of reef habitat (see Hoey et al. 2011), and have been shown to constrain the recovery of coral populations on isolated reefs, making them much more vulnerable to disturbances (Gilmour et al. 2013). Continued monitoring of juvenile coral assemblages within the CSMP will be critical to establish relationships with adult coral cover and detect the effect of disturbances on both densities and assemblage composition. However, additional monitoring of the settlement of coral larvae to artificial substrata (i.e., small terracotta tiles; Trapon et al. 2013b; Hughes et al. 2018) will help to resolve whether low densities of juvenile corals at most sites in the CSMP are attributable to limited supply of settling larvae, or low rates of post-settlment survival.

Comparing the composition of juvenile corals among regions of the CSMP reveals low densities of juvenile *Acropora* in the central and southern CSMP, with juvenile assemblages in these regions being dominated by *Porites* (Figure 4.33). While this lack of *Acropora* in the southern and central CSMP is consistent with surveys conducted in 2018-2020 (Hoey et al. 2020), it may reflect recent declines in abundance of corals that are highly susceptible to large-scale disturbances (Madin et al 2014; Hughes et al. 2018). Indeed, recent research in the GBRMP has shown that widespread mass-bleaching led to marked reductions in the settlement of corals (in particular *Acropora*), presumably due to decreases in populations of adult corals and decreased reproductive potential of surviving corals (Hughes et al 2019). *Acropora* are among the fatest growing coral taxa (Pratchett et al. 2015) and are often key to the recovery of live coral cover following disturbances. The low densities of juvenile *Acropora* in the CSMP suggests that the recovery and/or re-assembly of coral communities on these reefs is likely to be protracted.

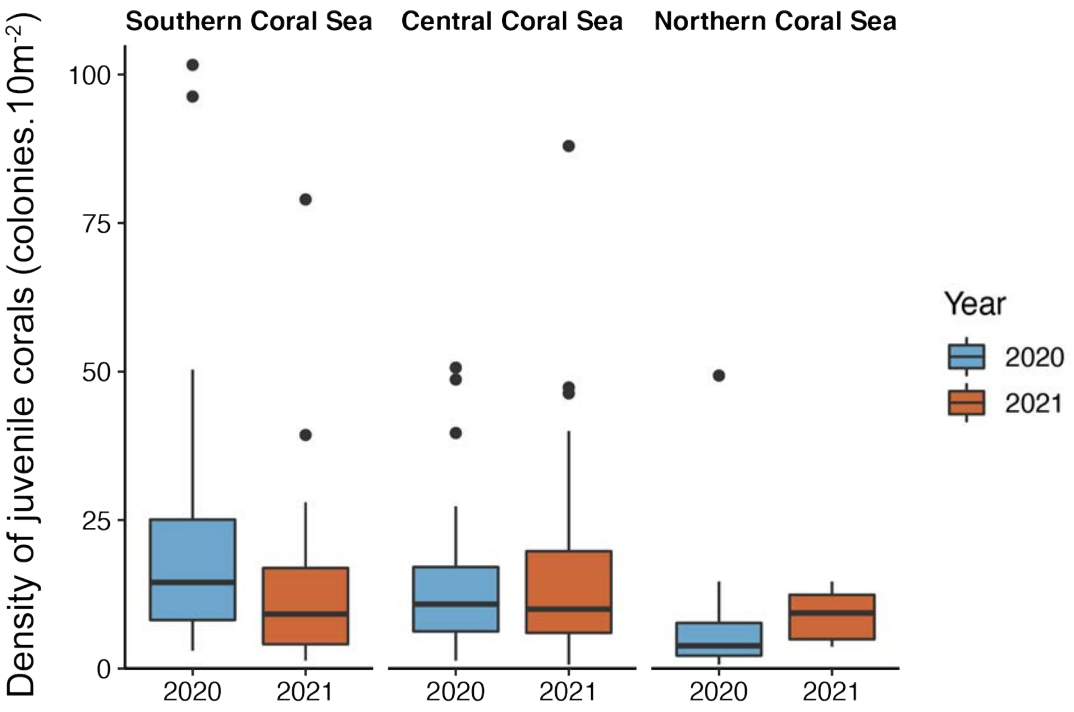


Figure 4.31 Spatial and temporal (2020 vs 2021) variation in the density of juvenile corals (<5cm diameter) among regions in the Coral Sea Marine Park. Data are based on surveys conducted at 43 sites across 13 reefs in February of each year.

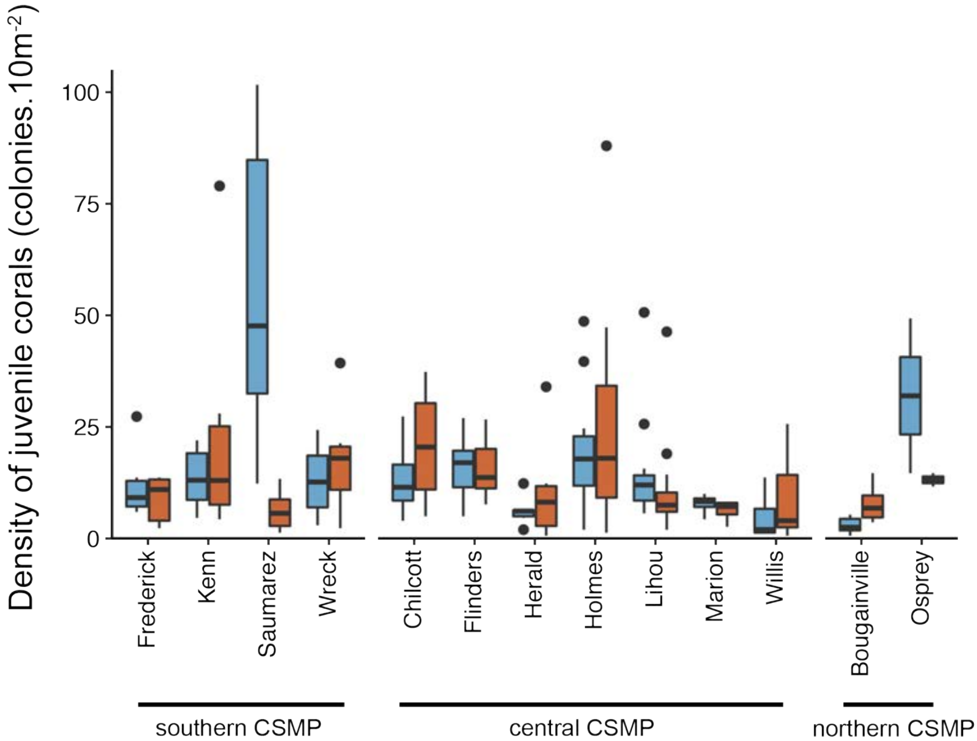


Figure 4.32 Spatial and temporal (2020 vs 2021) variation in the density of juvenile corals (<5cm diameter) among 13 reefs within the Coral Sea Marine Park. Data are based on surveys conducted at 43 sites across the 13 reefs in February of each year.

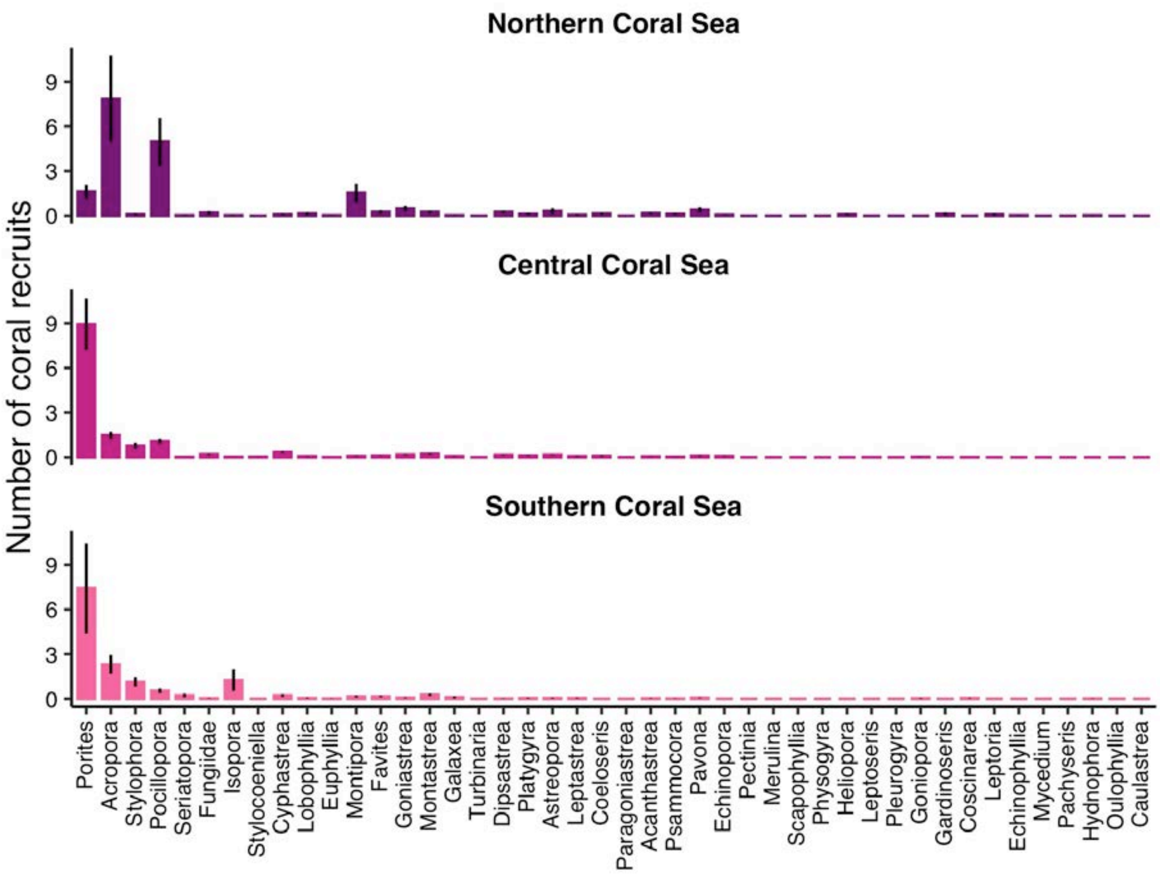


Figure 4.33 Regional patterns in the taxonomic composition of juvenile coral colonies in the Coral Sea Marine Park in 2021. Coral taxa are in order of overall abundance. Data are based on surveys of 43 sites across 13 reefs.

5 Conclusions

The coral reefs of the CSMP are some of the most isolated coral reef environments in Australian waters, supporting unique coral and reef fish communities that are distinct from those of the adjacent GBRMP (Hoey et al. 2020). While this isolation and inaccessibility reduces the exposure of the CSMP reefs to direct human pressures (e.g., fishing, run-off) relative to more accessible coastal or inshore reefs, reefs of the CSMP (like reefs globally) are being increasingly exposed to the effects of climate change. Climate-induced coral bleaching is now the foremost threat to coral reefs globally (Hughes et al. 2017), with the likelihood of global mass-coral bleaching occurring in any given year now being three-fold higher than pre-2000 (Hughes et al. 2018). Indeed, extensive coral bleaching has been recorded across shallow reef habitats in the CSMP in three of the past six years (i.e., 2016, 2017, 2020; Harrison et al. 2018, Hoey et al. 2020). While the 2016 and 2017 bleaching events led to shifts in the composition of coral assemblages, there were no regional declines in coral cover within the CSMP. The 2020 bleaching event in the CSMP, however, appeared to be more severe and widespread than the 2016 and 2017 events, with 63% of all corals surveyed across the CSMP and up to 89% of corals at individual reefs being bleached (Hoey et al. 2020). The surveys conducted in February 2021 under this project confirm that the 2020 bleaching event was the most severe and widespread bleaching event to affect the CSMP in recent years, and led to marked declines in the abundance and taxonomic richness, and shifts in the composition of coral and fish communities.

Total shallow water coral cover across the 13 CSMP reefs decreased from 28% in 2020 to 17% in 2021, a mean decline of 39%. There was, however, considerable variation in the decline in coral cover among regions (northern: 17%; central 43%; and southern 39%), reefs (ranging from 13% at Bougainville to 73% at Frederick Reef), and sites within reefs (e.g., 19% at Holmes 2 vs 59% at Holmes 6). In the absence of other major disturbances, these declines in coral cover are almost certainly attributable to the elevated ocean temperatures and subsequent coral bleaching recorded in February-March 2020 (Hoey et al. 2020). Indeed, 2020 saw an extended period of elevated temperatures throughout much of the CSMP, with most of the CSMP being exposed to >10 DHW (degree heating weeks) and some areas in the central and southern CSMP being exposed to 12-15 DHW (Figure 2.4). Such temperatures are well above those expected to cause bleaching-induced coral mortality (>6 DHW), with exposure to 10 DHW during the 2016 bleaching event on the GBR leading to a ~90% decline in coral cover (Hughes et al. 2018). Importantly, the observed declines in coral cover on CSMP reefs following the 2020 bleaching were not as great as may have been expected based on recorded levels of DHW, and may reflect differences in the composition of coral composition between the CSMP and GBRMP, a shifted baseline toward more bleaching resistant coral communities following previous heat stress (i.e., 2016 and 2017 bleaching events), and/or a greater resilience to heat stress in coral populations within the CSMP.

The geographic footprint of coral bleaching recorded in 2020 broadly matched the declines in the cover recorded in 2021, however the declines in coral cover were only weakly related to the observed bleaching levels at individual sites. The absence of a strong relationship between incidence of bleaching and subsequent mortality likely reflects differences in coral assemblages among reefs and sites, and/or the timing of the 2020 surveys at individual reefs. It is important to consider that the 2020 bleaching occurred against a shifted baseline of coral communities, with the abundance of bleaching sensitive coral taxa being reduced because of the 2016 and 2017 bleaching events (Harrison et al. 2018, 2019). Notably, prior to the 2020 bleaching, coral cover in the central CSMP was generally low (21.7%; Hoey et al. 2020), with coral taxa that are most sensitive to both severe storms and thermal stress (e.g., *Acropora*, *Seriatopora*; Madin and Connolly 2006; Hughes et al. 2018) conspicuously rare. Further, the 2020 estimates of coral bleaching were based on surveys conducted over a 3.5-week period, starting in the southern CSMP and working northwards to finish on Osprey Reef in the northern CSMP. The ocean temperatures were still well above average at the time of the surveys and consequently our surveys likely under-estimated the incidence and severity of bleaching, especially on the southern CSMP reefs.

Despite the declines in coral cover attributable to the 2020 bleaching event, coral cover in the CSMP (17.2%) is broadly comparable to recent estimates for the GBRMP (18.6%; Mellin et al. 2019). Moreover, current coral cover on central CSMP reefs (mean: 15.2%; range: 10.0-19.0%) is greater than that of early surveys at Herald Cays, Chilcott Islet and Lihou Reef where coral cover ranged from 1-5% in 2003 (following the 1998 coral bleaching event; Oxley et al. 2003) to ~6% in 2007 (Ceccarelli et al. 2008). Importantly, coral cover on CSMP reefs is greater than levels which may lead to fundamental changes in the structure and functioning of reef systems. Very low levels of coral cover (<10%) in some systems have been shown to disrupt positive, or reinforcing, feedbacks that limit recovery of coral and fish populations, and lead to shifts in habitat structure (Wilson et al. 2006; Graham et al. 2015). Once established, these shifts are difficult to reverse, and have lasting consequences for the diversity and functioning of such systems (Pratchett et al. 2021). Maintaining coral cover at levels >10% is therefore seen as critical to avoid ecosystem collapse.

The recovery of coral populations following widespread bleaching is dependent on the supply, settlement, and growth of new corals together with the growth of surviving corals. However, as the frequency of bleaching events increases there is concern that the return time between successive bleaching events is not sufficient for coral populations and communities to recover (Hughes et al. 2018). Even within well-connected reef systems such as the GBRMP, where the supply and settlement of coral larvae is not limiting, it has been estimated that reefs will require a minimum of 7-15 years free of any major disturbance to recover from a major bleaching event (Johns et al. 2014; Hughes et al. 2018). The recovery times for isolated reefs such as those in the CSMP are likely to be considerably longer (Gilmour et al. 2013), especially when the effects of these bleaching events are compounded by frequent damaging waves and cyclones (Hoey et al. 2020). Extensive coral bleaching has been recorded across shallow reef habitats in the CSMP in three of the past six years (i.e., 2016, 2017, 2020; Harrison et al. 2018, Hoey et al. 2020), with a maximum of 3 years between successive bleaching events. Given sustained and ongoing increases in global ocean temperatures it is very likely that there will be further severe and widespread mass-bleaching in CSMP in the coming years, which is likely to further suppress coral cover and delay recovery.

There was no detectable change in the density of juvenile corals within the CSMP following the 2020 bleaching event. Juvenile corals are generally less susceptible to bleaching than larger corals due to the favourable surface area to volume ratio of small corals, higher concentrations of fluorescent proteins, and/or their cryptic habitat shading them from direct sunlight (Alvarez-Noriega et al. 2018). However, the lack of decline in juvenile abundances in the present study is in contrast to other studies that have reported declines in the abundance of juvenile corals of up to 70% following large-scale bleaching events (e.g., Alvarez-Noriega et al. 2018; Dajka et al. 2019). This lack of decline may be related to the inclusion of remnant corals, resulting from partial mortality of larger coral colonies, in our density estimates for this size class. It should also be noted that the smallest juvenile corals recorded in the current study (~ 1cm diameter) would be at least 1-year old (depending on taxa), and therefore would have been spawned and settled on the reef well before the 2020 bleaching event. As such, any potential effects of the 2020 bleaching event on coral reproduction and the replenishment of coral populations are yet to be realised. The recent back-to-back 2016 and 2017 bleaching events on the GBR led to a collapse of coral recruitment in the following year (Hughes et al. 2018), presumably due to decline in the abundance of adult (breeding) corals, and the suppressed growth and reproductive output of surviving corals (Howells et al. 2016; Anderson et al. 2019). Understanding the effects of the 2020 bleaching event on the supply and recruitment of coral larvae, and the replenishment and recovery of coral populations will require continued monitoring of juvenile corals, together with the use of settlement tiles to quantify the composition and abundance of coral larvae settling to each reef.

Together with the declines in coral cover, there were marked declines in the richness, abundance and biomass of reef fishes on central, but not southern or northern, CSMP reefs. Despite these declines, the biomass of reef fishes (a key indicator of reef health, together with coral cover) recorded across all reefs in the CSMP was 5 – 30 kg per 100m^2 (or 500 - 3,000 kg per hectare), which is high relative to coral reef environments globally (Cinner et al. 2016) and likely reflects the isolation and limited fishing pressure on CSMP reefs. The reduction in the biomass of reef fishes was most pronounced on Flinders and Holmes Reefs, and Willis Islets, coinciding with substantial reductions in coral cover, and was largely driven by declines in planktivorous, corallivorous and grazing fishes. Numerous studies have reported that fishes that have a direct reliance on live corals for food (i.e., corallivores) and/or habitat (i.e., small bodied planktivores) are the first and most adversely affected by coral loss (e.g., Pratchett et al. 2011, 2014), however, the observed declines in the biomass of grazing fishes (primarily surgeonfishes) is difficult to reconcile.

Herbivorous fishes (i.e., grazers, scrapers, excavators, and browsers) are crtically important to the health and resilience of most coral reefs (e.g., Bellwood et al. 2006b; Hughes et al. 2007; Hoey and Bellwood 2009). Through their feeding activities these fishes reduce algal biomass, prevent macroalgal overgrowth, and clear benthic space for the settlement of corals and other organisms. Indeed, substantial declines in herbivorous fishes have been shown compromise the resilience of coral populations and assemblages following disturbance, and shifts in habitat structure (Rasher et al. 2013; Graham et al. 2015). In contrast to the declines in grazing fishes on the central CSMP reefs, several previous studies have reported substantial increases in the abundance and/or biomass of grazing fishes following large-scale coral mortality (e.g., Adam et al 2011; Gilmour et al. 2013). These increases have generally been related to the increase in the availability of their preferred feeding substrata as dead coral skeletons are rapidly colonised by algal communities (Diaz-Pulido and McCook 2002). This positive response of herbivorous fishes to increases in dietary resources following coral loss is supported by increases in growth rates of individual fishes following coral bleaching events on the GBR and Chagos Archipelago (Taylor et al. 2020). The reason for the declines in grazing fish populations on CSMP reefs is unclear, and could relate to short-term movement to alternate habitats (e.g., deeper habitats or spawning sites) or permanent declines in population size. Future monitoring is required to assess whether these population declines persist, and to assess the effects of these declines on reef health.

The abundances of macroalgae, sea snakes and macro-invertebrates displayed limited change between 2020 and 2021. The only exception was a ~20% decrease in the abundance of *Tridacna* clams, in particular *T. maxima* and *T. squamosa*, at Kenn Reef in southern CSMP. The decrease in the abundance of clams may be partly related to bleaching-induced mortality, or may reflect inherent patchiness and associated sampling error within sites at Kenn Reef. The lack of increases in macroalgae is encouraging, especially in light of the observed decreases in grazing fishes and coral cover. The green calcifying alga *Halimeda* spp., was again the predominant macroalgae within the CSMP following the 2020 bleaching. *Halimeda* is an important contributor to the production of reef sediments (Drew 1983), and is typically abundant on oceanic reefs, including those in the CSMP (Edgar et al. 2015). Importantly, *Halimeda* is not symptomatic of reef degradation (cf. canopy-forming brown algae that predominate on coastal and inshore reefs of the GBRMP and elsewhere: Wismer et al. 2009; Hoey and Bellwood 2010; Rasher et al. 2013).

The surveys conducted in this project together with the preceding 3-year project represent the most extensive assessment of coral reef health and marine biodiversity ever undertaken in the CSMP, and have greatly improved our understanding of the CSMP. However, it is important to recognise that these surveys were restricted to shallow reef habitats (<12m depth), and generally occurred on the sheltered aspect of each reef. There are some habitats (especially habitats on the weather exposed aspects of reefs, and in deeper areas below 12-15m) that are yet to be effectively surveyed. These habitats may provide refugia from elevated temperatures through decreasing temperature with depth (e.g., Bridge et al. 2013) and/or greater mixing of surface waters and upwelling on weather exposed aspects (Randall et al. 2020), and hence a source of larvae to re-populate shallow habitats. Surveys of deeper habitats of the CSMP using a remotely operated vehicle (ROV) and baited remote underwater videos (BRUVs) are currently underway, and will likely add considerably to the recorded biodiversity within the CSMP, and our understanding of the functioning of these unique reefs. Equally, surveys of habitats on the weather exposed aspects of reefs will add considerably to our understanding of these unique reefs, but will require modified survey methods due to difficulties in anchoring tenders in exposed areas.

Climate change and associated disturbances are increasingly shaping the composition and state of coral reefs globally (e.g., Hughes et al. 2017, 2018; Pratchett et al. 2020), and it is becoming increasingly important to understand the patterns of disturbance, resilience and recovery of individual reefs and reef systems. While the state and health of coral reef communities in the CSMP is likely to have been previously driven by a combination of reef geomorphology, reef size, habitat type, habitat complexity, and connectivity (Ceccarelli et al. 2013), it will be increasingly important to consider how these contemporary factors are influenced by ongoing and future effects of climate change.

5.1 Recommendations

Regular and ongoing comprehensive monitoring of coral reef environments in the CSMP is essential to understand its structure and function, ecological significance, and changing health and condition. Regular (annual) monitoring of CSMP reefs over the past six years has greatly improved our understanding of the unique nature of these reefs, and importantly attribute changes in reef communities to recent stressors, namely the 2020 marine heatwave. In the absence of regular monitoring, the causes of such changes would be largely unknown, severely limiting the capacity of managers to make informed decisions. As well as monitoring the current state or health of reefs (i.e., coral cover and population sizes of fishes and non-coral invertebrates), it is critical to quantify demographic processes of key reef taxa (e.g., recruitment, growth and mortality of corals and fishes) among reefs and regions within the CSMP to better understand the vulnerability and recovery potential of coral reef environments following recent (i.e., 2020 bleaching event) and likely future disturbances. Continued monitoring of juvenile corals coupled with targeted monitoring of coral settlement (described below) will be critical to understand the potential replenishment of coral populations following the 2020 bleaching event, and local stock-recruitment relationships for shallow water corals within the CSMP.

To effectively monitor the potential recovery of coral populations and communities, and any subsequent changes in fish communities following the effects of the 2020 bleaching event we recommend regular (i.e., annual or ideally biannual) monitoring of coral, fish, sea snakes and macro-invertebrate communities using the same methods and sites as the 2018-21 surveys. While the time between recurrent surveys of individual reefs could afford to be longer (2-5 years) in the absence of any major environmental disturbances, the increasing incidence of major disturbances impacting CSMP reefs in recent years (i.e., three mass bleaching events within the CSMP in the past 6 years), coupled with predicted increases in the frequency and intensity of disturbances affecting reefs globally (Hughes et al. 2018) and the logistical constraints of working in the CSMP (i.e. isolation and exposure), more regular surveys are critical. However, given the similarities of coral and fish assemblages on reefs within each region we recommend more detailed annual surveys of several 'representative' reefs in each region, with all 20 CSMP reefs to be surveyed every 3-4 years. These representative reefs should include 11-14 reefs (with a minimum of 4 reefs in each of the CSMP regions to allow rigorous statistical analyses). Consideration also needs to be given to the availability of suitable anchorages, and hence access to reefs and survey sites under all weather conditions. With representativeness and logistical considerations in mind we recommend as a minimum the following 13 reefs be surveyed annually Cato, Kenn, Saumarez, and Wreck Reefs in the southern CSMP, Flinders, Holmes, Lihou and Marion Reefs and Herald Cays in the central CSMP, and Ashmore, Boot, Bougainville and Osprey Reefs in the northern CSMP. Ideally, a minimum of 2 days should be spent at each reef to allow a greater number of sites and habitats (e.g., on the weather exposed aspect) to be surveyed.

Our current understanding of other habitat types (e.g., weather exposed aspects, soft-bottom/lagoon habitats, seagrass and algal habitats) is limited and should be incorporated into future monitoring. Spending a greater amount of time at the representative reefs (i.e., 2-3 days compared to only ~1 day in the surveys of the past 4 years) identified above would provide much greater opportunity to survey a diversity of different shallow, and potentially deeper, sites and habitats. This would also allow greater certainty around populations of sea cucumbers and giant clams that are not adequately captured through surveys of shallow reef habitats, and the identification of potentially important fish settlement and nursery habitats. Recent surveys of deeper water reef habitats in the CSMP has revealed substantial variation in benthic habitat among sites, and the identification of potentially important seagrass habitats. These additional habitats could be effectively surveyed using timed-swims with a towed GPS (e.g., Lynch et al. 2015), manta tow (e.g., Friedman et al. 2011), or underwater scooter surveys for shallow habitats, and remotely operated vehicles (ROV's) for deeper habitats. The use of ROVs, while effective, requires considerable time investment to process videos.

The 2021 surveys revealed considerable variation in coral mortality among sites (from 19% at Holmes 2 to 59% at Holmes 6; from 7% at Marion 6 to 62% at Marion 7). Investigation of the potential causes (e.g., water temperature, upwelling, water flow) of the observed variation in coral mortality within individual reefs should be a priority for future research. Surveying a greater number of sites at each reef would provide some insight into the potential causes of this variation, while also providing greater certainty around reef-level estimates of reef health. For example, surveying a greater number of sites would allow questions such as the following to be addressed: do reef areas that are closer to channels or passes through the reef have higher coral cover than those in sheltered back reef environments? If so, this could suggest that the high flushing of water through these channels are reducing bleaching stress on corals. Alternatively, it could highlight that these apparently resilient low-mortality sites are unique.

The results of the 2021 surveys show that the effects of the 2020 bleaching event were severe and widespread across the 13 CSMP reefs surveyed. The 2021 surveys, however, only included one of the five reefs previously identified as ‘bright spots’ (i.e., Bougainville, Ashmore, Boot, Moore and Mellish Reefs; Hoey et al. 2020). Importantly, Bougainville Reef (the only bright spot reef surveyed in 2021) had the highest coral cover and lowest recorded coral mortality (together with Chilcott Reef) of the 13 reefs surveyed. Given the geographic footprint of the heat stress across the CSMP in 2020 (Figure 2.4), it is likely that Mellish and Moore Reefs, and to a lesser extent Boot and Ashmore Reefs, experienced significant coral bleaching in 2020. If resources allow, we recommend re-surveying previous sites at these four bright spot reefs, together with the 13 representative reefs identified above, in early 2022 (i.e., February - March) to determine how these reefs fared following the 2020 bleaching event. Critically, if these reefs experienced little to no change in coral cover (similar to Bougainville Reef), then this apparent resilience to heat stress may explain the higher coral cover at these reefs. While further dedicated research would be required to ascertain the factors that contribute to these reefs being unique (i.e., recruitment, growth and mortality of corals and fishes, primary and secondary productivity, nutrient inputs, local hydrodynamics), their potential resilience to heat stress may warrant additional management consideration.

The potential recovery of coral populations and communities on CSMP reefs will be dependent on supply and settlement of coral larvae, together with the growth of both these newly-settled corals and surviving corals. The current monitoring of juvenile corals (< 5cm diameter) provides an indication of the abundance and composition of corals that will likely enter the adult population, however it doesn't differentiate the differences in the supply and settlement of coral larvae from post-settlement processes (i.e., mortality, competition, growth). Further, differences in growth rates among coral taxa, and hence the time taken to reach the 5cm threshold may bias assessments of the composition of juvenile corals toward slower growing coral taxa. We recommend continued monitoring of juvenile corals coupled with targeted monitoring of coral settlement to gain a greater understanding of the potential replenishment of coral populations following the 2020 bleaching event, and local stock-recruitment relationships for shallow water corals within the CSMP. Directly quantifying the settlement of corals requires the deployment of artificial substrata (i.e., small terracotta 'settlement' tiles) several weeks prior to the predicted coral spawning (in October-November), and their collection and processing 2-4 months later (February-March). Quantifying the settlement of coral larvae at a subset of reefs should be a future priority.

Continued monitoring of reef fish communities is essential to fully understand the impacts of coral loss following the 2020 bleaching event. Numerous studies and meta-analyses have shown that fish species that are reliant on live corals for food and/or shelter are the first and most adversely affected by coral loss (e.g., Bellwood et al. 2006a; Pratchett et al. 2008, 2011). However, many other fish species display protracted declines over several years as the physical structure of the habitat erodes, and the loss of juvenile habitat limits the replenishment of populations (e.g., Graham et al. 2007; Halford and Caley 2009; Pratchett et al. 2014). A key to understanding these relationships in the CSMP will be to repeat the 3-dimensional habitat mapping of sites mapped during 2019-2020 in the next 3-4 years. Matching the sites previously mapped will allow the relative contribution of live corals versus the underlying reef matrix and coralline algae in providing habitat structure to be assessed.

Scheduling of surveys for late summer-early autumn (i.e., Februay-April) is ideal to capture the incidence and extent of bleaching (as evidenced by the extensive bleaching recorded in 2020), however it limits the capacity to explore other important biological and ecological processes, especially those related to coral reproduction, coral settlement (detailed above) and fish spawning aggregations which typically peak in mid- to late-spring (i.e., October-November). Biannual surveys would allow for much more detailed understanding of reproduction and other seasonal processes, as well as allowing for the more effective deployment and maintenance of in-water sampling devices (e.g., batteries in acoustic receivers typically last ~9 months, tilt current meters only record for ~3 months). Given the logistics and costs of undertaking voyages to the CSMP we recommend any voyage in mid- to late-spring be limited to a small number of accessible innermost reefs (e.g., Flinders, Holmes, Bougainville and Osprey Reefs).

Finally, surveys conducted over the past 4 years have pointed toward the importance and unique nature of shallow water reef communities of the CSMP. Comparable monitoring and research in all regions within and bordering the CSMP, including the GBRMP, Australia’s Temperate East Marine Parks Network, New Caledonia, Solomon Islands and Papua New Guinea, is required to establish the biogeographical significance of the CSMP. Cross-jurisdictional meetings, workshops, and ultimately scientific expeditions will be invaluable to better understand biological and ecological connections among these regions. Given the increasing use of online meeting platforms (e.g., Zoom, Microsoft Teams) during the COVID-19 pandemic, now may the ideal time to initiate such cross-jurisdictional meetings without the expense of travel.

References

AIMS Datacentre (2021) dataaimsr: AIMS Data Platform API Client. R package version 1.0.2. https://open-aims.github.io/dataaimsr

Álvarez-Noriega, M., Baird, A.H., Bridge, T.C., Dornelas, M., Fontoura, L., Pizarro, O., Precoda, K., Torres-Pulliza, D., Woods, R.M., Zawada, K. and Madin, J.S. (2018) Contrasting patterns of changes in abundance following a bleaching event between juvenile and adult scleractinian corals. *Coral Reefs* 37: 527-532.

Anderson, K.D., Cantin, N.E., Casey, J.M. and Pratchett, M.S., 2019. Independent effects of ocean warming versus acidification on the growth, survivorship and physiology of two Acropora corals. *Coral Reefs* 38: 1225-1240.

Ayling, A.M. and Ayling, A.L. (1985) Report on a preliminary survey of the Lihou and Coringa/Herald Nature Reserves. Australian National Parks and Wildlife Service.

Baird, A.H. and Marshall, P.A. (2002) Mortality, growth and reproduction in scleractinian corals following bleaching on the Great Barrier Reef. *Marine Ecology Progress Series* 237: 133-141.

Baird, A.H., Madin, J.S., Álvarez-Noriega, M., Fontoura, L., Kerry, J.T., Kuo, C.Y., Precoda, K., Torres-Pulliza, D., Woods, R.M., Zawada, K.J. and Hughes, T.P. (2018) A decline in bleaching suggests that depth can provide a refuge from global warming in most coral taxa. *Marine Ecology Progress Series* 603: 257-264.

Barneche, D. and Logan, M. (2021) gisaimsr: Assortment of GBR GIS Files. R package version 0.0.1. https://open-aims.github.io/gisaimsr

Beaman, R. (2012) Project 3DGBR: Great Barrier Reef and Coral Sea Geomorphic Features. School of Earth and Environmental Sciences, James Cook University. eAtlas. https://eatlas.org.au/data/uuid/25685ba5-6583-494f-974d-cce2f3429b78

Bellwood, D.R. and Hughes, T.P. (2001) Regional-scale assembly rules and biodiversity of coral reefs. *Science* 292: 1532-1535.

Bellwood, D.R., Hoey, A.S., Ackerman, J.L. and Depczynski, M. (2006a) Coral bleaching, reef fish community phase shifts and the resilience of coral reefs. *Global Change Biology* 12: 1587-1594.

Bellwood, D.R., Hughes, T.P. and Hoey, A.S. (2006b) Sleeping functional group drives coral-reef recovery. *Current Biology* 16: 2434-2439.

Bellwood, D.R., Baird, A.H., Depczynski, M., et al. (2012) Coral recovery may not herald the return of fishes on damaged coral reefs. *Oecologia* 170: 567-573.

Bento, R., Hoey, A.S., Bauman, A.G., Feary, D.A. and Burt, J.A. (2016) The implications of recurrent disturbances within the world's hottest coral reef. *Marine Pollution Bulletin* 105: 466-472.

Braley, R.D. (1987) Distribution and abundance of the giant clams *Tridacna gigas* and *T. derasa* on the Great Barrier Reef. *Micronesica* 20: 215-223.

Bridge, T.C., Hoey, A.S., Campbell, S.J., Muttaqin, E., Rudi, E., Fadli, N. and Baird, A.H. (2013) Depth-dependent mortality of reef corals following a severe bleaching event: implications for thermal refuges and population recovery. *F1000 Research* 2.

Cinner, J.E., Huchery, C., MacNeil, M.A., et al. (2016). Bright spots among the world's coral reefs. *Nature* 535: 416-419.

Ceccarelli, D., Choat, J.H., Ayling, A.M., et al. (2008) Coringa-Herald National Nature Reserve Marine Survey – 2007. Report to the Department of the Environment, Water, Heritage and the Arts by C&R Consulting and James Cook University.

Ceccarelli, D.M., Emslie, M.J., Richards, Z.T. (2016) Post-disturbance stability of fish assemblages measured at coarse taxonomic resolution masks change at finer scales. *PLoS ONE* 11: e0156232.

Ceccarelli, D.M., McKinnon, A.D., Andrefouet, S., et al. (2013) The coral sea: physical environment, ecosystem status and biodiversity assets. *Advances in Marine Biology* 66: 213-290.

Choukroun, S., Harrison, H.B., Hoey, A.S., Pratchett, M.S. (2021) Coral Sea – Coral reef health multi-year project. *In-situ* measurements of temperature and water flow. Report prepared for Parks Austrlia.

Collot J., Lafoy Y., Geli L. (2011) Explanatory notes of the structural provinces of the Southwest Pacific map. Geological Survey of New Caledonia, DIMENC, IFREMER, New Caledonia.

Dajka, J.C., Wilson, S.K., Robinson, J.P., Chong-Seng, K.M., Harris, A. and Graham, N.A. (2019) Uncovering drivers of juvenile coral density following mass bleaching. *Coral Reefs* 38: 637-649.

Davies, P.J., Symonds, P.A., Feary, D.A., Pigram, C.J. (1989) The evolution of carbonate platforms of northeast Australia. In: Crevello, P.D., Wilson, J.L., Sarg, J.F., Read, J.F. (Eds.), Controls on Carbonate Platform and Basin Development. SEPM Special Publications, Tulsa, pp. 233–258.

De'ath, G., Fabricius, K. E., Sweatman, H. and Puotinen, M. L. (2012) The 27-year decline of coral cover on the Great Barrier Reef and its causes. *PNAS*. 109**:** 17995–17999.

Director of National Parks (DNP) (2018) *Coral Sea Marine Park Management Plan 2018*. Australian Government Director of National Parks, Canberra.

Doherty, P. J., Mather, P. and Planes, S. (1994). Acanthochromis polyacanthus, a fish lacking larval dispersal, has genetically differentiated populations at local and regional scales on the Great Barrier Reef. *Mar Biol* 121**:** 11–21.

Drew, E.A. (1983) Halimeda biomass, growth rates and sediment generation on reefs in the central Great Barrier Reef province. Coral Reefs 2: 101-110.

Dunnington, D. (2021) ggspatial: Spatial Data Framework for ggplot2. R package version 1.1.5. https://CRAN.R-project.org/package=ggspatial

Eakin, C.M. (1996) Where have all the carbonates gone? A model comparison of calcium carbonate budgets before and after the 1982–1983 El Nino at Uva Island in the eastern Pacific. *Coral Reefs* 15: 109-119.

Edgar GJ, Ceccarelli DM, Stuart-Smith RD (2015) Assessment of coral reef biodiversity in the Coral Sea. Report for the Department of the Environment. The Reef Life Survey Foundation Inc. and Institute of Marine and Antarctic Studies.

Edmunds, P.J. and Carpenter, R.C. (2001) Recovery of *Diadema antillarum* reduces macroalgal cover and increases abundance of juvenile corals on a Caribbean reef. *Proceedings of the National Academy of Sciences* 98: 5067-5071.

Emslie, M.J., Pratchett, M.S., Cheal, A.J. and Osborne, K. (2010) Great Barrier Reef butterflyfish community structure: the role of shelf position and benthic community type. *Coral Reefs* 29: 705-715.

Friedman, K., Eriksson, H., Tardy, E., and Pakoa, K. (2011). Management of sea cucumber stocks: patterns of vulnerability and recovery of sea cucumber stocks impacted by fishing. *Fish and Fisheries* 12: 75-93.

Froese, R. and D. Pauly. Editors. 2019.FishBase. World Wide Web electronic publication. www.fishbase.org

Garnier, S. (2018) viridis: Default Color Maps from 'matplotlib'. R package version 0.5.1.

Gilmour, J.P., Smith, L.D., Heyward, A.J., Baird, A.H., and Pratchett, M.S. (2013) Recovery of an isolated coral reef system following severe disturbance. *Science* 340: 69-71.

Glynn, P.W., Wellington, G.M. and Birkeland, C. (1979) Coral reef growth in the Galapagos: limitation by sea urchins. *Science* 203: 47-49.

Graham, N.A. and McClanahan, T.R. (2013) The last call for marine wilderness? *Bioscience* 63: 397-402.

Graham, N.A., Wilson, S.K., Jennings, S., et al. (2007) Lag effects in the impacts of mass coral bleaching on coral reef fish, fisheries, and ecosystems. *Conservation Biology* 21:1291-1300.

Graham, N.A., Jennings, S., MacNeil, M.A., Mouillot, D. and Wilson, S.K. (2015) Predicting climate-driven regime shifts versus rebound potential in coral reefs. *Nature* 518: 94-97.

Halford, A.R. and Caley, M.J., 2009. Towards an understanding of resilience in isolated coral reefs. *Global Change Biology* 15: 3031-3045.

Harrison, H.B., Àlvarez-Noriega, M., Baird, A.H., Heron, S.F., MacDonald, C., and Hughes, T.P. (2019) Back to back coral bleaching events on isolated atolls in the Coral Sea. *Coral Reefs* 38:713-719.

Harrison, H.B., Àlvarez-Noriega, M., Baird, A.H. and MacDonald, C. (2018) Recurrent Coral Bleaching in the Coral Sea Commonwealth Marine Reserve between 2016 and 2017. Report to the Director of National Park and Department of Environment and Energy by James Cook University. 41 pp.

Heatwole, H., Grech, A., Monahan, J.F., King, S. and Marsh, H. (2012) Thermal biology of sea snakes and sea kraits. Integrative and Comparative Biology 52: 257-273.

Heron, S.F., Maynard, J.A., Van Hooidonk, R., and Eakin, C.M. (2016) Warming trends and bleaching stress of the world's coral reefs 1985–2012. *Scientific Reports* 6: 38402.

Hillebrand, H. (2004). On the generality of the latitudinal diversity gradient. *The American Naturalist* 163: 192-211.

Hoey, A.S. and Bellwood, D.R. (2008) Cross-shelf variation in the role of parrotfishes on the Great Barrier Reef. *Coral Reefs* 27: 37-47.

Hoey, A.S. and Bellwood, D.R. (2009) Limited functional redundancy in a high diversity system: single species dominates key ecological process on coral reefs. *Ecosystems* 12: 1316-1328.

Hoey, A.S. and Bellwood, D.R. (2010) Cross-shelf variation in browsing intensity on the Great Barrier Reef. *Coral Reefs* 29: 499-508.

Hoey, A.S., Pratchett, M.S., Cvitanovic, C. (2011) High macroalgal cover and low coral recruitment undermines the potential resilience of coral reef assemblages at Lord Howe Island. *PLoS ONE* 6: e25824

Hoey A.S., Pratchett M.S., Sambrook K., Gudge S., and Pratchett D.J. (2018) Status and trends for shallow reef habitats and assemblages at Elizabeth and Middleton reefs, Lord Howe Marine Park. Report for Department of the Environment. 65 pp.

Hoey A.S. *et al.* (2020) Coral reef health in the Coral Sea Marine Park. Report on reef surveys April 2018 – March 2020. Report for Parks Australia. Pp. 171.

Howells, E.J., Ketchum, R.N., Bauman, A.G., Mustafa, Y., Watkins, K.D. and Burt, J.A. (2016) Species-specific trends in the reproductive output of corals across environmental gradients and bleaching histories. *Marine Pollution Bulletin 105*: 532-539.

Hughes, T.P., Rodrigues, M.J., Bellwood, D.R., Ceccarelli, D., Hoegh-Guldberg, O., McCook, L., Moltschaniwskyj, N., Pratchett, M.S., Steneck, R.S. and Willis, B. (2007) Phase shifts, herbivory, and the resilience of coral reefs to climate change. *Current Biology* 17: 360-365.

Hughes, T.P., Kerry J.T., Álvarez-Noriega M. et al. (2017) Global warming and recurrent mass bleaching of corals. *Nature* 543: 373–377.

Hughes T.P., Anderson K.D., Connolly S.R., et al. (2018) Spatial and temporal patterns of mass bleaching of corals in the Anthropocene. Science 359: 80-83.

Hughes, T.P., Kerry, J.T., Baird, A.H., et al. (2019) Global warming impairs stock-recruitment dynamics of corals. *Nature* 568: 387-390.

Humphries, A.T., McClanahan, T.R. and McQuaid, C.D. (2020) Algal turf consumption by sea urchins and fishes is mediated by fisheries management on coral reefs in Kenya. *Coral Reefs* doi: 10.1007/s00338-020-01943-5

Kassambara, A. (2018) ggpubr: 'ggplot2' Based Publication Ready Plots. R package version 0.1.8.

Kinch, J., Purcell, S., Uthicke, S. and Friedman, K. (2008) Population status, fisheries and trade of sea cucumbers in the Western Central Pacific. *Sea cucumbers. A global review of fisheries and trade. FAO Fisheries and Aquaculture Technical Paper*, *516*, pp.7-55.

Kitahara, M.V., Fukami, H., Benzoni, F., and Huang, D. (2016) The New Systematics of Scleractinia: Integrating Molecular and Morphological Evidence. In: Goffredo S, Dubinsky Z (eds) *The Cnidaria, Past, Present and Future: The world of Medusa and her sisters*. Springer International Publishing, Cham, pp. 41-59

Lam, V.Y.Y., Chaloupka, M., Thompson, A., Doropoulos, C., and Mumby, P.J. (2018) Acute drivers influence recent inshore Great Barrier Reef dynamics. *Proceedings of the Royal Society B: Biological Sciences* 285: 2063.

Loya, Y., Sakai, K., Yamazato, K., Nakano, Y., Sambali, H., and Van Woesik, R. (2001) Coral bleaching: the winners and the losers. *Ecology letters* 4: 122-131.

Lukoschek V, Heatwole H, Grech A, Burns G, and Marsh H (2007) Distribution of two species of sea snakes, *Aipysurus laevis* and *Emydocephalus annulatus*, in the southern Great Barrier Reef: metapopulation dynamics, marine protected areas and conservation. *Coral Reefs* 26: 291-307

Lynch, T., Green, M., and Davies, C. (2015) Diver towed GPS to estimate densities of a critically endangered fish. *Biological Conservation* 191: 700-706.

Madin, J. S., Dornelas, M., Baird, A. H. and Connolly, S. R. (2014). Mechanical vulnerability explains size-dependent mortality of reef corals. *Ecol. Lett.* 17: 1008–1015.

Marshall, P.A. and Baird, A.H. (2000) Bleaching of corals on the Great Barrier Reef: differential susceptibilities among taxa. *Coral Reefs* 19:155-163

McClanahan, T.R. and Shafir, S.H. (1990) Causes and consequences of sea urchin abundance and diversity in Kenyan coral reef lagoons. *Oecologia* 83: 362-370.

McClanahan, T.R., Nugues, M. and Mwachireya, S. (1994) Fish and sea urchin herbivory and competition in Kenyan coral reef lagoons: the role of reef management. *Journal of Experimental Marine Biology and Ecology* 184: 237-254.

McClanahan, T.R., Baird, A.H., Marshall, P.A. and Toscano, M.A. (2004) Comparing bleaching and mortality responses of hard corals between southern Kenya and the Great Barrier Reef, Australia. *Marine Pollution Bulletin* 48:327-335

Neuwirth, E. (2014) RColorBrewer: ColorBrewer Palettes. R package version 1.1-2.

Oksanen, J., Guillaume Blanchet, F., Friendly, M., Kindt, R., Legendre, P., McGlinn, Peter, D., Minchin, R., O'Hara, R. B., Simpson, G.L., Solymos, P., Henry, M., Stevens, H., Szoecs, E., Wagner, H. (2018). vegan: Community Ecology Package. R package version 2.5-2.

Oliver, E.C., Donat, M.G., Burrows, M.T., et al. (2018) Longer and more frequent marine heatwaves over the past century. *Nature Communications* 9: 1324.

Oxley, W.G., Ayling, A.M., Cheal, A.J., Thompson, A.A. (2003) Marine surveys undertaken in the Coringa-Herald National Nature Reserve, March-April 2003. Report produced for CRC Reef for Environment Australia by the Australian Institute of Marine Science, Townsville.

Oxley W.G., Emslie M., Muir P., Thompson A.A. (2004) Marine surveys undertaken in the Lihou Reef Nature Reserve, March 2004. Department of the Environment and Heritage.

Pebesma, E. (2018) Simple Features for R: Standardized Support for Spatial Vector Data. The R Journal 10 (1), 439-446, https://doi.org/10.32614/RJ-2018-009

Pratchett, M.S., Munday, P.L., Wilson, S.K., et al. (2008) Effects of climate-induced coral bleaching on coral-reef fishes: ecological and economic consequences. *Oceanography and Marine Biology: An Annual Review* 46: 251-296.

Pratchett M.S., Hobbs J-P.A., Hoey A.S., Baird A.H., Ayling A.A., Gudge S., Choat J.H. (2011a) Elizabeth and Middleton Reef Reserves Marine Survey 2011. Report to the Department of Sustainability, Environment, Water, Population and Communities. 71 pp.

Pratchett, M. S., Hoey, A. S., Wilson, S. K., Messmer, V., and Graham, N. A. (2011b) Changes in biodiversity and functioning of reef fish assemblages following coral bleaching and coral loss. *Diversity* 3: 424-452

Pratchett, M.S., Caballes, C.F., Rivera-Posada, J.A., and Sweatman, H. (2014a) Limits to understanding and managing outbreaks of crown-of-thorns starfish (*Acanthaster* spp.). *Oceanography and Marine Biology* 52: 133-200.

Pratchett, M.S., Hoey, A.S. and Wilson, S.K. (2014b) Reef degradation and the loss of critical ecosystem goods and services provided by coral reef fishes. *Current Opinion in Environmental Sustainability* 7: 37-43.

Pratchett, M.S., Blowes, S.A., Coker, D., et al. (2015) Indirect benefits of high coral cover for non-corallivorous butterflyfishes. *Coral Reefs* 34*:* 665-672.

Pratchett, M.S., Anderson, K.D., Hoogenboom, M.O., Widman, E., Baird, A.H., Pandolfi, J.M., Edmunds, P.J. and Lough, J.M. (2015) Spatial, temporal and taxonomic variation in coral growth—implications for the structure and function of coral reef ecosystems. *Oceanography and Marine Biology: An Annual Review* 53: 215-295.

Pratchett, M.S., McWilliam, M.J. and Riegl, B. (2020) Contrasting shifts in coral assemblages with increasing disturbances. *Coral Reefs* 39: 783-793.

Pratchett, M.S., Heron, S.F., Mellin, C. and Cumming, G.S. (2021) Recurrent mass-bleaching and the potential for ecosystem collapse on Australia's Great Barrier Reef. In: *Ecosystem Collapse and Climate Change* (eds. J.G. Canadell, R.B. Jackson) Springer, Cham. pp. 265-289.

R Core Team (2018) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Randall, C.J., Toth, L.T., Leichter, J.J., Maté, J.L. and Aronson, R.B. (2020) Upwelling buffers climate change impacts on coral reefs of the eastern tropical Pacific. *Ecology* 101: e02918.

Rasher D.B., Hoey A.S., and Hay M.E. 2013. Consumer diversity interacts with prey defenses to drive ecosystem function. *Ecology* 94: 1347-1358.

Ridgway, K.R., Benthuysen, J.A., Steinberg, C. (2018) Closing the gap between the Coral Sea and the equator: Direct observations of the north Australian western boundary currents. *Journal of Geophysical Research: Oceans* 123: 9212–9231.

Rousselet, L., Doglioli, A.M., Maes, C., Blanke, B., and Petrenko, A.A. (2016) Impacts of mesoscale activity on the water masses and circulation in the Coral Sea. *Journal of Geophysical Research: Oceans* 121: 7277–7289.

Rylaarsdam, K.W. (1983) Life histories and abundance patterns of colonial corals on Jamaican reefs. *Marine Ecology Progress Series* 13*:* 249-260.

Skewes, T.D. and Persson, S.I. (2017) Coral Sea sea cucumber survey, 2017. A report for Parks Australia. Tim Skewes Consulting. Brisbane

Slowikowski, K. (2018) ggrepel: Automatically Position Non-Overlapping Text Labels with 'ggplot2'. R package version 0.8.0.

Smale, D.A., Wernberg, T., Oliver, E.C., et al. (2019) Marine heatwaves threaten global biodiversity and the provision of ecosystem services. *Nature Climate Change* 9: 206-312.

Smith, T.B., Glynn, P.W., Maté, J.L., Toth, L.T. and Gyory, J. (2014) A depth refugium from catastrophic coral bleaching prevents regional extinction. *Ecology* 95: 1663-1673.

Stuart-Smith, R.D., Crawford, T., Cooper, A., Kininmonth, S., Stuart-Smith, J., Berkhout, J. and Edgar, G. (2013) Coral Sea marine biodiversity. IMAS and Reef Life Survey, Australia

Taylor, B.M., Benkwitt, C.E., Choat, H., Clements, K.D., Graham, N.A. and Meekan, M.G. (2020) Synchronous biological feedbacks in parrotfishes associated with pantropical coral bleaching. *Global Change Biology* 26: 1285-1294.

Trapon, M.L., Pratchett, M.S., Hoey, A.S. (2013a) Spatial variation in abundance, size and orientation of juvenile corals related to the biomass of parrotfishes on the Great Barrier Reef, Australia. *PLoS ONE* 8: e57788.

Trapon, M.L., Pratchett, M.S., Hoey, A.S. and Baird, A.H. (2013b) Influence of fish grazing and sedimentation on the early post-settlement survival of the tabular coral *Acropora cytherea*. *Coral Reefs* 32:1051-1059.

Udyawer, V., Cappo, M., Simpfendorfer, C.A., Heupel, M.R. and Lukoschek, V. (2014) Distribution of sea snakes in the Great Barrier Reef Marine Park: observations from 10 yrs of baited remote underwater video station (BRUVS) sampling. Coral Reefs 33(3): 777-791.

Wickham, H. (2016) ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag New York.

Wickham, H. (2017) tidyverse: Easily Install and Load the 'Tidyverse'. R package version 1.2.1.

Williams, S.M. (2021) The reduction of harmful algae on Caribbean coral reefs through the reintroduction of a keystone herbivore, the long spined sea urchin, *Diadema antillarum*. *Restoration Ecology*, p.e13475.

Wilson, S.K., Bellwood, D.R., Choat, J.H. and Furnas, M.J. (2003) Detritus in the epilithic algal matrix and its use by coral reef fishes. *Oceanography and Marine Biology: an Annual Review* 41: 279-310.

Wilson, S.K., Graham, N.A.J., Polunin, N.V.C. (2007) Appraisal of visual assessments of habitat complexity and benthic composition on coral reefs. *Marine Biology* 151: 1069-1076.

Wilson, S.K., Graham, N.A.J., Pratchett, M.S., Jones, G.P., Polunin, N.V.C. (2006) Multiple disturbances and the global degradation of coral reefs: are reef fishes at risk or resilient? *Global Change Biology* 12: 2220-2234.

Wismer, S., Hoey, A.S., Bellwood, D. R. (2009) Cross-shelf benthic community structure on the Great Barrier Reef: relationships between macroalgal cover and herbivore biomass. *Marine Ecology Progress Series* 376: 45-54.

Zinke, J., Gilmour, J.P., Fisher, R., Puotinen, M., Maina, J., Darling, E., Stat, M., Richarts, Z.T., Mcclanahan, T.R., Beger, M., Moore, C., Graham, N.J., Feng, M., Hobbs, J.-P.A., Evans, S., Field, S., Shedrawi, G., Babcock, R.C., Wilson, S.K. (2018) Gradients of disturbance and environmental conditions shape coral community structure for south-eastern Indian Ocean reefs. *Diversity and Distributions* 24:605-620.

6 APPENDIX 1 – Leveraged projects.

Five additional projects were leveraged from this collaboration between James Cook University and Parks Australia and capitalised on available space during the voyage. * indicates projects funded under an *Our Marine Parks – Round 2 Grant*, however the scope of these projects was increased substantively through activities completed during the February 2021 voyage

Project description	Key Personnel	Institution
Movement and population structure of sharks and large fishes within the CSMP*	Dr Adam Barnett Prof Andrew Hoey Prof Morgan Pratchett	James Cook University James Cook University James Cook University
The ecology of deep reef habitats in the CSMP*	Ms Gemma Galbriath Mr Ben Cresswell Prof Andrew Hoey Prof Morgan Pratchett Dr Eva McClure	James Cook University James Cook University James Cook University James Cook University James Cook University
Production of educational and promotional videos of the CSMP*	Ms Rebecca Piper Mr Richard Fitzpatrick	Biopixel Biopixel
Genetic diversity of giant clams (*Tridacna* spp) within the CSMP	Mr Peter Doll Prof Morgan Pratchett Prof Andrew Hoey Assoc. Prof. Jane Williamson	James Cook University James Cook University James Cook University Macquarie University
Opportunistic surveys for fish spawning aggregations	Prof Andrew Hoey Prof Morgan Pratchett Mr Martin Russell	James Cook University James Cook University Science and Conservation of Fish Aggregations
In situ measurements of temperature and water flow	Dr Severine Choukroun Prof Morgan Pratchett Dr Hugo Harrison Prof Andrew Hoey	James Cook University James Cook University James Cook University James Cook University

7 APPENDIX 2 – Sites surveyed.

List of sites surveyed across 13 reefs in the Coral Sea Marine Park (CSMP) and three reefs in Great Barrier Reef Marine Park (GBRMP) during 2020 and 2021. Voyage dates were 16 February – 12 March 2020, 4 February – 10 March 2021.

Region	Sector	Site	Exposure	Aspect	Lat	Long
CSMP	Southern	Frederick 1	Sheltered	NE	-21.01111	154.351
CSMP	Southern	Frederick 2	Semi-sheltered	W	-21.01043	154.34743
CSMP	Southern	Frederick 4	Sheltered	W	-20.93838	154.39737
CSMP	Southern	Kenn 1	Sheltered	NE	-21.2476	155.76616
CSMP	Southern	Kenn 2	Sheltered	NE	-21.25323	155.76216
CSMP	Southern	Kenn 4	Sheltered	W	-21.20459	155.77238
CSMP	Southern	Saumarez 3	Sheltered	NW	-21.9178	153.58452
CSMP	Southern	Saumarez 5	Sheltered	W	-21.75002	153.76973
CSMP	Southern	Saumarez 7	Sheltered	NW	-21.91194	153.59119
CSMP	Southern	Wreck 1	Sheltered	NW	-22.19267	155.33405
CSMP	Southern	Wreck 2	Sheltered	W	-22.17814	155.17674
CSMP	Southern	Wreck 3	Sheltered	NW	-22.18667	155.17049
CSMP	Central	Chilcott 2	Sheltered	NW	-16.93962	149.99644
CSMP	Central	Chilcott 4	Sheltered	NE	-16.93601	149.99835
CSMP	Central	Flinders 1	Sheltered	NW	-17.71357	148.43713
CSMP	Central	Flinders 2	Sheltered	N	-17.70218	148.46655
CSMP	Central	Flinders 5	Sheltered	W	-17.86163	148.46652
CSMP	Central	Flinders 6	Sheltered	S	-17.83089	148.51353
CSMP	Central	Flinders 7	Exposed	NE	-17.53675	148.55112
CSMP	Central	Herald 1	Semi-exposed	N	-16.94348	149.18565
CSMP	Central	Herald 4	Sheltered	SW	-16.97254	149.12865
CSMP	Central	Herald 6	Sheltered	W	-16.99189	149.13075
CSMP	Central	Holmes 1	Sheltered	NW	-16.52613	147.80701
CSMP	Central	Holmes 2	Semi-sheltered	W	-16.51181	147.84
CSMP	Central	Holmes 5	Semi-sheltered	NW	-16.50534	147.96745
CSMP	Central	Holmes 6	Semi-sheltered	NW	-16.41898	147.98981
CSMP	Central	Holmes 7	Semi-sheltered	NW	-16.42693	147.98442
CSMP	Central	Holmes 10	Semi-exposed	NW	-16.52143	147.83772
CSMP	Central	Lihou 1	Sheltered	NW	-17.59707	151.48956
CSMP	Central	Lihou 2	Sheltered	N	-17.59065	151.50027
CSMP	Central	Lihou 4	Semi-sheltered	N	-17.12527	151.82535
CSMP	Central	Lihou 5	Semi-sheltered	N	-17.12113	151.82939
CSMP	Central	Lihou 7	Exposed	SE	-17.41725	151.86607
CSMP	Central	Lihou 9	Lagoon	SE	-17.13022	151.83931
CSMP	Central	Marion 6	Exposed	SE	-19.12125	152.39993

CSMP	Central	Marion 7	Sheltered	N	-19.29511	152.23782
CSMP	Central	Willis 2	Sheltered	W	-16.28728	149.9593
CSMP	Central	Willis 4	Semi-exposed	NE	-16.28256	149.9657
CSMP	Central	Willis 7	Semi-sheltered	NW	-16.11697	149.97095
CSMP	Northern	Bougainville 1	Sheltered	N	-15.49273	147.08638
CSMP	Northern	Bougainville 4	Semi-exposed	SW	-15.50667	147.11234
CSMP	Northern	Bougainville 5	Semi-exposed	SW	-15.50083	147.09891
CSMP	Northern	Osprey 7	Lagoon	E	-13.88845	146.5594
GBRMP	Central	Yamacutta 1			-17.85383	146.6088
GBRMP	Central	Yamacutta 2			-17.85199	146.61508
GBRMP	Northern	Day 1			-14.48258	145.51596
GBRMP	Northern	Day 2			-14.47672	145.51762
GBRMP	Northern	Escape 1			-15.88507	145.7648
GBRMP	Northern	Escape 2			-15.88303	145.76949

8 APPENDIX 3 – Fish species surveyed.

List of fish species recorded from the southern, central and northern reefs in the CSMP and GBRMP and the area in which fish are counted in each transect.

Species	Transect area	Species2	Transect area2
Abudefduf sexfasciatus	50 x 2	*Acanthurus olivaceus*	50 x 5
Abudefduf vaigiensis	50 x 2	*Acanthurus pyroferus*	50 x 5
Abudefduf whitleyi	50 x 2	*Acanthurus thompsoni*	50 x 5
Acanthochromis polyacanthus	50 x 2	*Acanthurus triostegus*	50 x 5
Amblyglyphidodon aureus	50 x 2	*Acanthurus xanthopterus*	50 x 5
Amblyglyphidodon curacao	50 x 2	*Anyperodon leucogrammicus*	50 x 5
Amblyglyphidodon leucogaster	50 x 2	*Aphareus furca*	50 x 5
Amphiprion akindynos	50 x 2	*Aprion virescens*	50 x 5
Amphiprion chrysopterus	50 x 2	*Balistapus undulatus*	50 x 5
Amphiprion clarkii	50 x 2	*Balistoides conspicillum*	50 x 5
Amphiprion melanopus	50 x 2	*Balistoides viridescens*	50 x 5
Amphiprion perideraion	50 x 2	*Bolbometopon muricatum*	50 x 5
Chromis agilis	50 x 2	*Caesio cuning*	50 x 5
Chromis alpha	50 x 2	*Caesio lunaris*	50 x 5
Chromis amboinensis	50 x 2	*Calotomus carolinus*	50 x 5
Chromis atripectoralis	50 x 2	*Carangoides bajad*	50 x 5
Chromis atripes	50 x 2	*Carangoides ferdau*	50 x 5
Chromis chrysura	50 x 2	*Carangoides fulvoguttatus*	50 x 5
Chromis flavomaculata	50 x 2	*Carangoides orthogrammus*	50 x 5
Chromis iomelas	50 x 2	*Caranx ignobilis*	50 x 5
Chromis lepidolepis	50 x 2	*Caranx lugubris*	50 x 5
Chromis margaritifer	50 x 2	*Caranx melampygus*	50 x 5
Chromis retrofasciata	50 x 2	*Caranx sexfasciatus*	50 x 5
Chromis ternatensis	50 x 2	*Caranx sp.*	50 x 5
Chromis vanderbilti	50 x 2	*Carcharhinus albimarginatus*	50 x 5
Chromis viridis	50 x 2	*Carcharhinus amblyrhynchos*	50 x 5
Chromis weberi	50 x 2	*Cephalopholis argus*	50 x 5
Chromis xanthochira	50 x 2	*Cephalopholis cyanostigma*	50 x 5
Chromis xanthura	50 x 2	*Cephalopholis leopardus*	50 x 5
Chrysiptera biocellata	50 x 2	*Cephalopholis miniata*	50 x 5
Chrysiptera brownriggii	50 x 2	*Cephalopholis spiloparea*	50 x 5
Chrysiptera flavipinnis	50 x 2	*Cephalopholis urodeta*	50 x 5
Chrysiptera glauca	50 x 2	*Cetoscarus ocellatus*	50 x 5
Chrysiptera rex	50 x 2	*Cheilinus chlorourus*	50 x 5
Chrysiptera rollandi	50 x 2	*Cheilinus fasciatus*	50 x 5
Chrysiptera talboti	50 x 2	*Cheilinus oxycephalus*	50 x 5
Chrysiptera taupou	50 x 2	*Cheilinus trilobatus*	50 x 5
Dascyllus aruanus	50 x 2	*Cheilinus undulatus*	50 x 5
Dascyllus reticulatus	50 x 2	*Chlorurus bleekeri*	50 x 5
Dascyllus trimaculatus	50 x 2	*Chlorurus frontalis*	50 x 5
Dischistodus melanotus	50 x 2	*Chlorurus japanensis*	50 x 5
Dischistodus pseudochrysopoecilus	50 x 2	*Chlorurus microrhinos*	50 x 5
Hemiglyphidodon plagiometopon	50 x 2	*Chlorurus spilurus*	50 x 5
Lepidozygus tapeinosoma	50 x 2	*Choerodon cyanodus*	50 x 5
Neoglyphidodon melas	50 x 2	*Choerodon fasciatus*	50 x 5
Neoglyphidodon nigroris	50 x 2	*Choerodon graphicus*	50 x 5
Neopomacentrus asyzron	50 x 2	*Cromileptes altivelis*	50 x 5
Neopomacentrus cf cyanomos	50 x 2	*Ctenochaetus binotatus*	50 x 5
Plectroglyphidodon dickii	50 x 2	*Ctenochaetus cyanocheilus*	50 x 5
Plectroglyphidodon imparipennis	50 x 2	*Ctenochaetus striatus*	50 x 5
Plectroglyphidodon johnstonianus	50 x 2	*Diploprion bifasciatum*	50 x 5

Plectroglyphidodon lacrymatus	50 x 2	*Elagatis bipinnulatus*	50 x 5
Plectroglyphidodon leucozonus	50 x 2	*Epibulus insidiator*	50 x 5
Plectroglyphidodon phoenixensis	50 x 2	*Epinephelus coeruleopunctatus*	50 x 5
Pomacentrus adelus	50 x 2	*Epinephelus coioides*	50 x 5
Pomacentrus amboinensis	50 x 2	*Epinephelus fasciatus*	50 x 5
Pomacentrus bankanensis	50 x 2	*Epinephelus fuscoguttatus*	50 x 5
Pomacentrus brachialis	50 x 2	*Epinephelus hexagonatus*	50 x 5
Pomacentrus chrysurus	50 x 2	*Epinephelus howlandensis*	50 x 5
Pomacentrus coelestis	50 x 2	*Epinephelus lanceolatus*	50 x 5
Pomacentrus grammorhynchus	50 x 2	*Epinephelus merra*	50 x 5
Pomacentrus imitator	50 x 2	*Epinephelus polyphekadion*	50 x 5
Pomacentrus lepidogenys	50 x 2	*Epinephelus quoyanus*	50 x 5
Pomacentrus moluccensis	50 x 2	*Epinephelus tukula*	50 x 5
Pomacentrus nagasakiensis	50 x 2	*Gnathodentex aureolineatus*	50 x 5
Pomacentrus pavo	50 x 2	*Gracilla albomarginata*	50 x 5
Pomacentrus philippinus	50 x 2	*Gymnocranius euanus*	50 x 5
Pomacentrus vaiuli	50 x 2	*Gymnocranius microdon*	50 x 5
Pomacentrus wardi	50 x 2	*Hemigymnus fasciatus*	50 x 5
Pomachromis richardsoni	50 x 2	*Hemigymnus melapterus*	50 x 5
Stegastes apicalis	50 x 2	*Hipposcarus longiceps*	50 x 5
Stegastes fasciolatus	50 x 2	*Hologymnosus annulatus*	50 x 5
Stegastes gascoynei	50 x 2	*Hologymnosus doliatus*	50 x 5
Stegastes nigricans	50 x 2	*Kyphosus cinerascens*	50 x 5
Anampses caeruleopunctatus	50 x 4	*Kyphosus vaigiensis*	50 x 5
Anampses femininus	50 x 4	*Lethrinus atkinsoni*	50 x 5
Anampses meleagrides	50 x 4	*Lethrinus erythracanthus*	50 x 5
Anampses neoguinaicus	50 x 4	*Lethrinus miniatus*	50 x 5
Anampses twistii	50 x 4	*Lethrinus nebulosus*	50 x 5
Apolemichthys trimaculatus	50 x 4	*Lethrinus obsoletus*	50 x 5
Bodianus axillaris	50 x 4	*Lethrinus olivaceus*	50 x 5
Bodianus dictynna	50 x 4	*Lethrinus sp. 1*	50 x 5
Bodianus loxozonus	50 x 4	*Lethrinus xanthocheilus*	50 x 5
Bodianus mesothorax	50 x 4	*Lutjanus argentimaculatus*	50 x 5
Bodianus perditio	50 x 4	*Lutjanus bohar*	50 x 5
Centropyge bicolor	50 x 4	*Lutjanus carponotatus*	50 x 5
Centropyge bispinosus	50 x 4	*Lutjanus fulviflamma*	50 x 5
Centropyge fisheri	50 x 4	*Lutjanus fulvus*	50 x 5
Centropyge flavissimus	50 x 4	*Lutjanus gibbus*	50 x 5
Centropyge heraldi	50 x 4	*Lutjanus kasmira*	50 x 5
Centropyge loricula	50 x 4	*Lutjanus monostigma*	50 x 5
Centropyge smokey	50 x 4	*Lutjanus rivulatus*	50 x 5
Centropyge tibicen	50 x 4	*Lutjanus semicinctus*	50 x 5
Centropyge vrolikii	50 x 4	*Luzonichthys sp*	50 x 5
Chaetodon auriga	50 x 4	*Macolor macularis*	50 x 5
Chaetodon baronessa	50 x 4	*Macolor niger*	50 x 5
Chaetodon bennetti	50 x 4	*Melichthys vidua*	50 x 5
Chaetodon citrinellus	50 x 4	*Monotaxis grandoculis*	50 x 5
Chaetodon ephippium	50 x 4	*Monotaxis heterodon*	50 x 5
Chaetodon flavirostris	50 x 4	*Mulloidichthys flavolineatus*	50 x 5
Chaetodon kleinii	50 x 4	*Mulloidichthys vanicolensis*	50 x 5
Chaetodon lineolatus	50 x 4	*Naso annulatus*	50 x 5
Chaetodon lunula	50 x 4	*Naso brachycentron*	50 x 5
Chaetodon lunulatus	50 x 4	*Naso brevirostris*	50 x 5
Chaetodon melannotus	50 x 4	*Naso caesius*	50 x 5
Chaetodon mertensii	50 x 4	*Naso hexacanthus*	50 x 5
Chaetodon meyeri	50 x 4	*Naso lituratus*	50 x 5
Chaetodon ocellicaudus	50 x 4	*Naso tonganus*	50 x 5
Chaetodon ornatissimus	50 x 4	*Naso unicornis*	50 x 5
Chaetodon oxycephalus	50 x 4	*Naso vlamingii*	50 x 5
Chaetodon pelewensis	50 x 4	*Odonus niger*	50 x 5

Chaetodon plebeius	50 x 4	*Oxycheilinus digramma*	50 x 5
Chaetodon punctatofasciatus	50 x 4	*Oxycheilinus orientalis*	50 x 5
Chaetodon rafflesi	50 x 4	*Oxycheilinus oxycephalus*	50 x 5
Chaetodon rainfordi	50 x 4	*Oxycheilinus unifasciatus*	50 x 5
Chaetodon reticulatus	50 x 4	*Paracanthurus hepatus*	50 x 5
Chaetodon semeion	50 x 4	*Parupeneus barberinoides*	50 x 5
Chaetodon speculum	50 x 4	*Parupeneus barberinus*	50 x 5
Chaetodon trifascialis	50 x 4	*Parupeneus ciliatus*	50 x 5
Chaetodon ulietensis	50 x 4	*Parupeneus crassilabris*	50 x 5
Chaetodon unimaculatus	50 x 4	*Parupeneus cyclostomus*	50 x 5
Chaetodon vagabundus	50 x 4	*Parupeneus multifasciatus*	50 x 5
Chaetodontoplus meredithi	50 x 4	*Parupeneus pleurostigma*	50 x 5
Chelmon rostratus	50 x 4	*Platax pinnatus*	50 x 5
Cirrhilabrus exquisitus	50 x 4	*Plectorhinchus albovittatus*	50 x 5
Cirrhilabrus laboutei	50 x 4	*Plectorhinchus chaetodontoides*	50 x 5
Cirrhilabrus lineatus	50 x 4	*Plectorhinchus lessoni*	50 x 5
Cirrhilabrus punctatus	50 x 4	*Plectorhinchus lineatus*	50 x 5
Cirrhilabrus scottorum	50 x 4	*Plectorhinchus picus*	50 x 5
Coris aygula	50 x 4	*Plectropomus areolatus*	50 x 5
Coris batuensis	50 x 4	*Plectropomus laevis*	50 x 5
Coris dorsomacula	50 x 4	*Plectropomus leopardus*	50 x 5
Coris gaimard	50 x 4	*Pomacanthus imperator*	50 x 5
Diproctacanthus xanthurus	50 x 4	*Pomacanthus semicirculatus*	50 x 5
Forcipiger flavissimus	50 x 4	*Pomacanthus sexstriatus*	50 x 5
Forcipiger longirostris	50 x 4	*Pomacanthus xanthometopon*	50 x 5
Gomphosus varius	50 x 4	*Prionurus maculatus*	50 x 5
Halichoeres biocellatus	50 x 4	*Pseudanthias cooperi*	50 x 5
Halichoeres hortulanus	50 x 4	*Pseudanthias pascalus*	50 x 5
Halichoeres margaritaceus	50 x 4	*Pseudanthias pleurotaenia*	50 x 5
Halichoeres marginatus	50 x 4	*Pseudanthias squamipinnis*	50 x 5
Halichoeres melanurus	50 x 4	*Pseudanthias tuka*	50 x 5
Halichoeres ornatissimus	50 x 4	*Pseudobalistes flavimarginatus*	50 x 5
Halichoeres prosopeion	50 x 4	*Pseudobalistes fuscus*	50 x 5
Halichoeres trimaculatus	50 x 4	*Pterocaesio digramma*	50 x 5
Hemitaurichthys polylepis	50 x 4	*Pterocaesio tile*	50 x 5
Heniochus acuminatus	50 x 4	*Pterocaesio trilineata*	50 x 5
Heniochus chrysostomus	50 x 4	*Rhinecanthus rectangulus*	50 x 5
Heniochus monoceros	50 x 4	*Scarus altipinnis*	50 x 5
Heniochus varius	50 x 4	*Scarus chameleon*	50 x 5
Labrichthys unilineatus	50 x 4	*Scarus dimidiatus*	50 x 5
Labroides bicolor	50 x 4	*Scarus flavipectoralis*	50 x 5
Labroides dimidiatus	50 x 4	*Scarus forsteni*	50 x 5
Labroides pectoralis	50 x 4	*Scarus frenatus*	50 x 5
Labropsis australis	50 x 4	*Scarus ghobban*	50 x 5
Labropsis xanthonota	50 x 4	*Scarus globiceps*	50 x 5
Macropharyngodon choati	50 x 4	*Scarus longipinnis*	50 x 5
Macropharyngodon kuiteri	50 x 4	*Scarus niger*	50 x 5
Macropharyngodon meleagris	50 x 4	*Scarus oviceps*	50 x 5
Macropharyngodon negrosensis	50 x 4	*Scarus psittacus*	50 x 5
Paracentropyge multifasciata	50 x 4	*Scarus rivulatus*	50 x 5
Pseudocheilinus evanidus	50 x 4	*Scarus rubroviolaceus*	50 x 5
Pseudocheilinus hexataenia	50 x 4	*Scarus schlegeli*	50 x 5
Pseudocoris yamashiroi	50 x 4	*Scarus spinus*	50 x 5
Pseudodax moluccanus	50 x 4	*Scarus viridifucatus*	50 x 5
Pteragogus sp.	50 x 4	*Scarus xanthopleura*	50 x 5
Pygoplites diacanthus	50 x 4	*Scolopsis bilineatus*	50 x 5
Stethojulis bandanensis	50 x 4	*Scomberoides lysan*	50 x 5
Stethojulis interrupta	50 x 4	*Scomberoides sp*	50 x 5
Stethojulis strigiventer	50 x 4	*Serranocirrhites latus*	50 x 5

Thalassoma amblycephalum	50 x 4	*Siganus argenteus*	50 x 5
Thalassoma hardwicke	50 x 4	*Siganus corallinus*	50 x 5
Thalassoma lunare	50 x 4	*Siganus doliatus*	50 x 5
Thalassoma lutescens	50 x 4	*Siganus puellus*	50 x 5
Thalassoma nigrofasciatum	50 x 4	*Siganus punctatissimus*	50 x 5
Thalassoma purpureum	50 x 4	*Siganus punctatus*	50 x 5
Thalassoma quinquevittatum	50 x 4	*Siganus vulpinus*	50 x 5
Acanthurus albipectoralis	50 x 5	*Siganus woodlandi*	50 x 5
Acanthurus blochii	50 x 5	*Stegostoma fasciatum*	50 x 5
Acanthurus dussumieri	50 x 5	*Sufflamen bursa*	50 x 5
Acanthurus grammoptilus	50 x 5	*Sufflamen chrysopterus*	50 x 5
Acanthurus guttatus	50 x 5	*Trachinotus blochii*	50 x 5
Acanthurus lineatus	50 x 5	*Triaenodon obesus*	50 x 5
Acanthurus mata	50 x 5	*Variola louti*	50 x 5
Acanthurus nigricans	50 x 5	*Zanclus cornutus*	50 x 5
Acanthurus nigricauda	50 x 5	*Zebrasoma scopas*	50 x 5
Acanthurus nigrofuscus	50 x 5	*Zebrasoma veliferum*	50 x 5
Acanthurus nigroris	50 x 5		

9 APPENDIX 4 – Fish species records.

List of conspicuous (i.e., non-cryptic) fish species recorded and/or observed within each region of the CSMP during 2018-2021. A separate column is provided for cryptobenthic fish species that were identified during targeted collections using clove oil. * indicates species that were recorded for the first time in 2021

Count	Species	Southern	Central	Northern	Cryptobenthic
1	*Abudefduf sexfasciatus*	1		1	
2	*Abudefduf vaigiensis*	1	1	1	
3	*Acanthochromis polyacanthus*		1	1	1
4	*Acanthurus albipectoralis*	1	1	1	
5	*Acanthurus blochii*	1	1	1	
6	*Acanthurus dussumieri*	1	1	1	
7	*Acanthurus grammoptilus*		1		
8	*Acanthurus guttatus*	1	1	1	
9	*Acanthurus lineatus*	1	1	1	
10	*Acanthurus maculiceps**		1		
11	*Acanthurus mata*		1	1	
12	*Acanthurus nigricans*	1	1	1	
13	*Acanthurus nigricauda*	1	1	1	
14	*Acanthurus nigrofuscus*	1	1	1	1
15	*Acanthurus nigroris*	1	1	1	
16	*Acanthurus nubilis*		1		
17	*Acanthurus olivaceus*	1	1	1	
18	*Acanthurus pyroferus*	1	1	1	
19	*Acanthurus thompsoni*	1	1	1	
20	*Acanthurus triostegus*	1	1	1	
21	*Acanthurus xanthopterus*	1	1	1	
22	*Aethaloperca rogaa*			1	
23	*Aetobatus narinari*		1		
24	*Aetobatus ocellatus*	1			
25	*Aluteres scriptus*	1	1	1	
26	*Amanses scopas*	1		1	
27	*Amblycirrhitus bimacula*				1
28	*Amblyeleotris steinitzi*		1	1	
29	*Amblyglyphidodon aureus*	1	1	1	
30	*Amblyglyphidodon curacao*	1	1		
31	*Amblyglyphidodon leucogaster*	1	1	1	
32	*Amphiprion akindynos*	1	1		
33	*Amphiprion chrysopterus*		1	1	
34	*Amphiprion clarkii*	1		1	
35	*Amphiprion melanopus*	1	1	1	
36	*Amphiprion perideraion*		1	1	
37	*Anampses caeruleopunctatus*	1	1	1	
38	*Anampses femininus*	1	1		
39	*Anampses geographicus**		1	1	
40	*Anampses meleagrides*	1			
41	*Anampses neoguinaicus*	1	1	1	
42	*Anampses twistii*	1	1	1	
43	*Antennarius nummifer*				1
44	*Antennarius pictus*				1
45	*Anyperodon leucogrammicus*			1	
46	*Aphareus furca*	1	1	1	
47	*Apogon crassiceps*				1

48	*Apogon doederleini*			1	
49	*Apogon doryssa*				1
50	*Apogon seminigricaudus*				1
51	*apogonid sp.*				1
52	*Apolemichthys trimaculatus*			1	
53	*Aprion virescens*	1	1	1	
54	*Arothron hispidus*	1			
55	*Arothron nigropunctatus*	1	1	1	
56	*Arothron stellatus*	1	1		
57	*Aseraggodes sp.*				1
58	*Assessor flavissimus*			1	
59	*Asterropteryx semipunctata*				1
60	*Aulostomus chinensis*	1	1	1	
61	*Balenoperca chabanaudi*		1	1	
62	*Balistapus undulatus*	1	1	1	
63	*Balistoides conspicillum*	1	1	1	
64	*Balistoides viridescens*	1	1	1	
65	*Belonoperca chabanaudi**			1	
66	*Bodianus anthioides*		1		
67	*Bodianus axillaris*	1	1	1	
68	*Bodianus dictynna*		1	1	
69	*Bodianus loxozonus*		1	1	
70	*Bodianus mesothorax*	1	1	1	
71	*Bodianus perditio*	1			
72	*Bolbometopon muricatum*		1	1	
73	*Brachaluteres prionurus*		1		
74	*Brosmophyciops pautzkei*				1
75	*Bryaninops sp.*				1
76	*bythitid sp.*				1
77	*Cabillus tongarevae*				1
78	*Caesio caerulaurea**			1	
79	*Caesio cuning*		1		
80	*Caesio lunaris*		1	1	
81	*Caesio teres*		1	1	
82	*Callogobius sclateri*				1
83	*Calotomus carolinus*	1	1	1	
84	*Cantherhines dumerilii*	1	1		
85	*Cantherhines pardalis**		1		
86	*Canthigaster amboinensis*	1	1		
87	*Canthigaster axiologus*	1			
88	*Canthigaster bennetti*	1	1		
89	*Canthigaster janthinoptera**		1		
90	*Canthigaster papua*		1		1
91	*Canthigaster valentini*	1	1	1	1
92	*Caracanthus maculatus*	1	1	1	1
93	*Caracanthus unipinna*				1
94	*Carangoides ferdau*		1	1	
95	*Carangoides fulvoguttatus**			1	
96	*Carangoides orthogrammus*	1	1	1	
97	*Carangoides plagiotaenia*			1	
98	*Caranx ignobilis*	1	1	1	
99	*Caranx lugubris*		1	1	
100	*Caranx melampygus*	1	1	1	
101	*Caranx papuensis**		1		
102	*Caranx sexfasciatus*	1	1	1	
103	*Caranx sp.*			1	
104	*Carcharhinus albimarginatus*	1	1	1	
105	*Carcharhinus amblyrhynchos*	1	1	1	
106	*Celotomus carolinus*	1			
107	*Centropyge bicolor*	1	1	1	

108	*Centropyge bispinosa*	1	1	1	1
109	*Centropyge fisheri*		1		
110	*Centropyge flavissima*	1	1	1	
111	*Centropyge heraldi*	1	1	1	1
112	*Centropyge hybrid 'smokey'*	1	1		1
113	*Centropyge loricula*	1	1	1	
114	*Centropyge tibicen*	1			1
115	*Centropyge vrolikii*	1	1	1	
116	*Centropyge woodheadi*	1			
117	*Cephalopholis argus*	1	1	1	
118	*Cephalopholis leopardus*		1	1	1
119	*Cephalopholis miniata*		1	1	
120	*Cephalopholis spiloparaea*		3		
121	*Cephalopholis urodeta*	1	1	1	1
122	*Cercamia eremia*				1
123	*Cetoscarus ocellatus*	1	1	1	1
124	*Chaetodon auriga*	1	1	1	
125	*Chaetodon baronessa*			1	
126	*Chaetodon bennetti*	1		1	
127	*Chaetodon citrinellus*	1	1	1	
128	*Chaetodon ephippium*	1	1	1	
129	*Chaetodon flavirostris*	1	1	1	
130	*Chaetodon kleinii*	1	1	1	
131	*Chaetodon lineolatus*	1	1	1	
132	*Chaetodon lunula*	1	1	1	
133	*Chaetodon lunulatus*	1	1	1	
134	*Chaetodon melannotus*	1	1	1	
135	*Chaetodon mertensii*	1	1	1	
136	*Chaetodon meyeri*		3	1	
137	*Chaetodon ocellicaudus*	1			
138	*Chaetodon ornatissimus*	1	1	1	
139	*Chaetodon oxycephalus*			1	
140	*Chaetodon pelewensis*	1	1	1	
141	*Chaetodon plebeius*	1	1	1	
142	*Chaetodon punctatofasciatus*			1	
143	*Chaetodon rafflesi*		1		
144	*Chaetodon reticulatus*	1	1	1	
145	*Chaetodon semeion*		1	1	
146	*Chaetodon speculum*	1	1	1	
147	*Chaetodon trifascialis*	1	1	1	
148	*Chaetodon ulietensis*	1	1	1	
149	*Chaetodon unimaculatus*	1	1	1	
150	*Chaetodon vagabundus*	1	1	1	
151	*Chanos chanos*			1	
152	*Cheilinus chlorourus*	1	1	1	
153	*Cheilinus fasciatus*		1	1	
154	*Cheilinus oxycephalus*	1	1	1	
155	*Cheilinus trilobatus*	1	1	1	
156	*Cheilinus undulatus*	1	1	1	
157	*Cheilodipterus macrodon*		1		
158	*Chlorurus bleekeri*			1	
159	*Chlorurus frontalis*	1	1		
160	*Chlorurus japanensis*	1		1	
161	*Chlorurus microrhinos*	1	1	1	
162	*Chlorurus spilurus*	1	1	1	
163	*Choerodon fasciatus*		1		
164	*Chromis agilis*	1	1	1	
165	*Chromis alpha*		1		
166	*Chromis amboinensis*	1	1	1	
167	*Chromis atripectoralis*	1	1	1	

168	*Chromis atripes*	1	1	1	
169	*Chromis chrysura*	1	1	1	
170	*Chromis flavomaculata*	1			
171	*Chromis iomelas*	1	1	1	1
172	*Chromis lepidolepis*	1	1	1	
173	*Chromis margaritifer*	1	1	1	1
174	*Chromis retrofasciata*	1	1	1	
175	*Chromis ternatensis*	1	1	1	
176	*Chromis vanderbilti*	1	1	1	1
177	*Chromis viridis*	1	1		
178	*Chromis weberi*		1	1	
179	*Chromis xanthochira*	1	1		
180	*Chromis xanthura*	1	1	1	
181	*Chrysiptera biocellata*	1	1	1	
182	*Chrysiptera brownriggii*		1	1	
183	*Chrysiptera flavipinnis*		1		
184	*Chrysiptera glauca*	1			
185	*Chrysiptera rollandi*		1		1
186	*Chrysiptera talboti*			1	
187	*Chrysiptera taupou*	1	1	1	1
188	*Cirrhilabrus exquisitus*	1	1	1	
189	*Cirrhilabrus laboutei*	1	1		1
190	*Cirrhilabrus lineatus*		1		
191	*Cirrhilabrus punctatus*	1	1	1	1
192	*Cirrhilabrus scottorum*	1	1	1	
193	*Cirrhitichthys falco*	1	1		1
194	*Cirrhitichthys oxycephalus**			1	
195	*Cirrhitus pinnulatus*	1			
196	*Cirripectes castaneus*		1	1	1
197	*Cirripectes filamentosus*				1
198	*Cirripectes stigmaticus*	1	1		1
199	*Coris aygula*	1	1	1	
200	*Coris batuensis*			1	1
201	*Coris dorsomacula*	1	1		
202	*Coris gaimard*	1	1	1	
203	*Cosmocampus banneri*				1
204	*Crossosalarias macrospilus*				1
205	*Ctenochaetus binotatus*	1	1	1	
206	*Ctenochaetus cyanocheilus*	1	1	1	
207	*Ctenochaetus striatus*	1	1	1	
208	*Ctenogobiops pomastictus*				1
209	*Cypho purpurascens*	1	1	1	1
210	*Dascyllus aruanus*	1			
211	*Dascyllus reticulatus*	1	1	1	1
212	*Dascyllus trimaculatus*	1	1	1	
213	*Dasyatis kuhlii*		1		
214	*Decapterus macarellus*		1		
215	*Dinematichthys ilucoetiodes*				1
216	*Dinematichthys sp.?*				1
217	*Diodon hystrix*		1		
218	*Diplogrammus goramensis*				1
219	*Dischistodus melanotus*	1			
220	*Dischistodus pseudochrysopoecilus*	1			
221	*Doryrhamphus melanopleura*				1
222	*Doryrhamphus sp.**				1
223	*Echeneis naucrates*	1	1	1	
224	*Echidna polyzona*				1
225	*Ecsenius bicolor*			1	
226	*Ecsenius fourmanoiri*	1			
227	*Ecsenius stictus*				1

228	*Ecsenius tigris*				1
229	*Elegatis bipinnulata*		1	1	
230	*Encheliophis homei?*				1
231	*Enneapterygius atrogulare?*				1
232	*Enneapterygius flavoccipitis*				1
233	*Enneapterygius sp.*				1
234	*Enneapterygius sp. 1*				1
235	*Enneapterygius sp. 1*				1
236	*Enneapterygius tutuilae*				1
237	*Epibulus insidiator*	1	1	1	
238	*Epinephelus coioides*		1		
239	*Epinephelus cyanopodus*	1			
240	*Epinephelus fasciatus*	1		1	
241	*Epinephelus fuscoguttatus*			1	
242	*Epinephelus hexagonatus*	1	1	1	
243	*Epinephelus howlandensis*	1			
244	*Epinephelus lanceolatus*		1		
245	*Epinephelus merra*	1	1	1	
246	*Epinephelus polyphekadion*	1	1	1	
247	*Epinephelus quoyanus*		1		
248	*Epinephelus tauvina*		1		
249	*Epinephelus tukula*			1	
250	*Euthynnus affinis*	1			
251	*Eviota afelei*				1
252	*Eviota ancora**				1
253	*Eviota atriventris*				1
254	*Eviota cf. teresae**				1
255	*Eviota cometa*				1
256	*Eviota distigma*				1
257	*Eviota fallax**				1
258	*Eviota fasciola*				1
259	*Eviota flebilis**				1
260	*Eviota guttata*		1		
261	*Eviota herrei*				1
262	*Eviota infulata*				1
263	*Eviota latifasciata*				1
264	*Eviota melanosphena*				1
265	*Eviota melasma*				1
266	*Eviota monostigma*				1
267	*Eviota nebulosa*				1
268	*Eviota occasa**				1
269	*Eviota prasites*	1			1
270	*Eviota punctulata*				1
271	*Eviota queenslandica*				1
272	*Eviota readeri**				1
273	*Eviota sigillata*				1
274	*Eviota singula**				1
275	*Eviota sp.*				1
276	*Eviota sp. 1**				1
277	*Eviota sp. 1**				1
278	*Eviota sp. 1a**				1
279	*Eviota sp. 3**				1
280	*Eviota sp. 4**				1
281	*Eviota sp. 5**				1
282	*Eviota sparsa*				1
283	*Eviota specca**				1
284	*Eviota variola*				1
285	*Eviota zebrina*				1
286	*Exallias brevis*	1	1		
287	*Fistularia commersonii*	1	1	1	

288	*Forcipiger flavissimus*	1	1	1	
289	*Forcipiger longirostris*	1	1	1	
290	*Fowleria aurita*				1
291	*Fowleria vaiulae*				1
292	*Fusigobius gracilis*				1
293	*Fusigobius humeralis*				1
294	*Fusigobius neophytus*				1
295	*Fusigobius sp.*				1
296	*Galeocerdo cuvier*	1			
297	*Genicanthus melanospilos*		1	1	
298	*Genicanthus watanabei*		1		
299	*Glyptoparus delicatulus*				1
300	*Gnathanodon speciosus*	1			
301	*Gnathodentex aureolineatus*	1	1	1	
302	*Gnatholepis cauerensis*		1		1
303	*Gnatholepis sp.*				1
304	*gobiid sp.*				1
305	*Gobiodon prolixus*				1
306	*Gobiodon quinquestrigatus*				1
307	*Gobiodon rivulatus*				1
308	*Gomphosus varius*	1	1	1	
309	*Gracila albomarginata*			1	
310	*Grammistes sexlineatus*		1	1	
311	*Gymnapogon philippinus*				1
312	*Gymnapogon sp.*				1
313	*Gymnocranius euanus*	1	1		
314	*Gymnocranius grandoculis*			1	
315	*Gymnocranius microdon*	1	1		
316	*Gymnosarda unicolor*	1	1	1	
317	*Gymnothorax favagineus*		1		
318	*Gymnothorax flavimarginatus*				1
319	*Gymnothorax fuscomaculatus*				1
320	*Gymnothorax gracilicauda*				1
321	*Gymnothorax javanicus*	1	1	1	
322	*Gymnothorax meleagris*	1			
323	*Gymnothorax sp.*				1
324	*Gymnothorax zonipectis*				1
325	*Halicampus dunckeri*				1
326	*Halichoeres biocellatus*	1	1	1	1
327	*Halichoeres chrysus*			1	
328	*Halichoeres hortulanus*	1	1	1	
329	*Halichoeres margaritaceus*	1	1	1	
330	*Halichoeres marginatus*	1	1	1	
331	*Halichoeres melanurus*				1
332	*Halichoeres melanurus*			1	
333	*Halichoeres nebulosus*	1			
334	*Halichoeres ornatissimus*	1	1	1	
335	*Halichoeres prosopeion*		1	1	
336	*Halichoeres trimaculatus*	1	1	1	1
337	*Helcogramma sp.*				1
338	*Helcogramma striatum*				1
339	*Hemigymnus fasciatus*	1	1	1	
340	*Hemitaurichthys polylepis*	1	1	1	
341	*Heniochus acuminatus*		1	1	
342	*Heniochus chrysostomus*	1	1	1	
343	*Heniochus monoceros*	1	1	1	
344	*Heniochus varius*	1	1	1	
345	*Heteropriacanthus carolinus*				1
346	*Heteropriacanthus cruentatus*			1	
347	*Himantura fai*		1		

348	*Hipposcarus longiceps*	1	1	1	
349	*Hologymnosus annulatus*	1	1	1	
350	*Hologymnosus doliatus*	1	1		
351	*Hoplolatilus starcki*			1	
352	*Iniistius pavo**	1			
353	*Kaupichthys brachychirus*				1
354	*Kyphosus bigibbus*	1			
355	*Kyphosus cinerascens*	1	1	1	
356	*Kyphosus vaigiensis*	1	1	1	
357	*Labrichthys unilineatus*			1	1
358	*labrid sp.*				1
359	*Labroides bicolor*	1	1	1	
360	*Labroides dimidiatus*	1	1	1	1
361	*Labroides pectoralis*	1		1	
362	*Labropsis australis*	1	1	1	
363	*Labropsis xanthonota*		1	1	
364	*Lepadichthys frenatus*				1
365	*Lepadichthys sp.*				1
366	*Lepidozygus tapeinosoma*		1	1	
367	*Lethrinus atkinsoni**		1		
368	*Lethrinus erythracanthus*		1	1	
369	*Lethrinus nebulosus*	1	1	1	
370	*Lethrinus olivaceus*	1	1	1	
371	*Lethrinus sp. 1*		1		
372	*Lethrinus xanthocheilus*	1	1	1	
373	*Limnichthys fasciatus*				1
374	*Liopropoma susumi*	1			1
375	*Luposicya lupus*				1
376	*Lutjanus argentimaculatus*			1	
377	*Lutjanus bohar*	1	1	1	
378	*Lutjanus fulvus*		1	1	
379	*Lutjanus gibbus*	1	1	1	
380	*Lutjanus kasmira*	1	1	1	
381	*Lutjanus monostigma*		1	1	
382	*Lutjanus rivulatus*	1	1	1	
383	*Lutjanus semicinctus*			1	
384	*Luzonichthys sp*			1	
385	*Luzonichthys waitei*			1	
386	*Macolor macularis*	1	1	1	
387	*Macolor niger*	1	1	1	
388	*Macropharyngodon choati*		1		
389	*Macropharyngodon kuiteri*		1		
390	*Macropharyngodon meleagris*	1	1	1	
391	*Macropharyngodon negrosensis*	1	1		
392	*Malacanthus latovittatus*	1	1	1	
393	*Meiacanthus atrodorsalis*		1	1	1
394	*Melichthys vidua*	1	1	1	
395	*Monotaxis grandoculis*	1	1	1	
396	*Monotaxis heterodon*	1	1	1	
397	*Mulloidichthys flavolineatus*	1	1		
398	*Mulloidichthys vanicolensis*	1	1	1	
399	*Myripristis adusta*			1	
400	*Myripristis kuntee*	1	1	1	
401	*Myripristis murdjan*		1		
402	*Myripristis vittata*		1		
403	*Naso annulatus*	1	1	1	
404	*Naso brachycentron*		1	1	
405	*Naso brevirostris*	1	1	1	
406	*Naso caesius*	1	1	1	
407	*Naso hexacanthus*	1	1	1	

408	*Naso lituratus*	1	1	1	
409	*Naso tonganus*	1	1	1	
410	*Naso unicornis*	1	1	1	
411	*Naso vlamingii*	1	1	1	
412	*Neamia octospina*				1
413	*Nebrius ferrugineus*	1	1	1	
414	*Nemateleotris magnifica*	1		1	1
415	*Neocirrhites armatus*	1	1	1	1
416	*Neoniphon sammara*	1	1	1	
417	*Neopomacentrus cf cyanomos*		1		
418	*Neosynchiropus morrisoni*				1
419	*Neotrygon kuhlii*	1	1		
420	*Norfolkia thomasi*				1
421	*Novaculichthys taeniourus*	1	1		1
422	*Odonus niger*		1		
423	*Ogilbyina queenslandiae*				1
424	*Opistognathus seminudus*				1
425	*Opistognathus stigmosus*				1
426	*Ostorhinchus cyanosoma*				1
427	*Ostracion cubicus*	1	1		
428	*Ostracion meleagris*		1	1	
429	*Oxycheilinus digramma*	1	1	1	
430	*Oxycheilinus orientalis*	1	1	1	1
431	*Oxycheilinus unifasciatus*	1	1	1	
432	*Oxymonacanthus longirostris*	1	1	1	
433	*Paracaesio sordida*			1	
434	*Paracanthurus hepatus*	1	1	1	
435	*Paracentropyge multifasciatus*		1	1	
436	*Paracirrhites arcatus*	1	1	1	1
437	*Paracirrhites forsteri*	1	1	1	
438	*Paracirrhites hemistictus*	1	1		
439	*Paragobiodon echinocephalus*				1
440	*Paragobiodon lacunicolus*				1
441	*Paragobiodon xanthosoma*				1
442	*Parapercis clathrata*				1
443	*Parupeneus barberinoides*		1		
444	*Parupeneus barberinus*	1	1	1	
445	*Parupeneus ciliatus*	1	1	1	
446	*Parupeneus crassilabris*	1	1	1	
447	*Parupeneus cyclostomus*	1	1	1	
448	*Parupeneus multifasciatus*	1	1	1	
449	*Parupeneus pleurostigma*	1	1	1	
450	*Pempheris oualensis*	1			
451	*Pervagor alternans*	1	1		
452	*Pervagor janthinosoma*	1	1		1
453	*Plagiotremus rhinorhynchos*		1	1	
454	*Plagiotremus tapeinosoma*		1	1	
455	*Platax pinnatus*		1		
456	*Platax teira*		1		
457	*platycephalid sp.*				1
458	*Plectorhinchus albovittatus*		1	1	
459	*Plectorhinchus chaetodonoides*	1	1	1	
460	*Plectorhinchus lessonii*		1	1	
461	*Plectorhinchus lineatus*		1	1	
462	*Plectorhinchus picus*	1	1		
463	*Plectranthias nanus*				1
464	*Plectroglyphidodon dickii*	1	1	1	
465	*Plectroglyphidodon imparipennis*	1	1	1	
466	*Plectroglyphidodon johnstonianus*	1	1	1	
467	*Plectroglyphidodon lacrymatus*	1	1	1	1

468	*Plectroglyphidodon leucozonus*			1	
469	*Plectroglyphidodon phoenixensis*	1	1		
470	*Plectropomus areolatus*		1	1	
471	*Plectropomus laevis*	1	1	1	
472	*Plectropomus leopardus*	1	1	1	
473	*Plectropomus oligacanthus*			1	
474	*Plectrypops lima*				1
475	*Plesiops caeruleolineatus*				1
476	*Pleurosicya mossambica*				1
477	*Plotosus lineatus*	1	1	1	1
478	*Pomacanthus imperator*	1	1	1	
479	*Pomacanthus sexstriatus*			1	
480	*Pomacentrus amboinensis*			1	1
481	*Pomacentrus auriventris*			1	
482	*Pomacentrus bankanensis*	1	1	1	
483	*Pomacentrus brachialis*	1		1	1
484	*Pomacentrus chrysurus*		1	1	
485	*Pomacentrus coelestis*	1	1	1	
486	*Pomacentrus imitator*	1	1	1	
487	*Pomacentrus lepidogenys*	1	1	1	
488	*Pomacentrus moluccensis*	1	1	1	
489	*Pomacentrus nagasakiensis*				1
490	*Pomacentrus pavo*			1	
491	*Pomacentrus philippinus*	1		1	1
492	*Pomacentrus vaiuli*	1	1	1	1
493	*Pomacentrus wardi*	1			
494	*Pomachromis richardsoni*	1	1	1	
495	*Priacanthus blochii*		1		
496	*Priacanthus hamrur*		1		
497	*Priolepis cincta*				1
498	*Priolepis compita*				1
499	*Priolepis inhaca*				1
500	*Priolepis kappa*				1
501	*Priolepis pallidicincta*				1
502	*Priolepis psygmophila*				1
503	*Priolepis sp.*				1
504	*Prionurus maculatus*	1			
505	*Pristiapogon exostigma*				1
506	*Prteragogus sp.*	1			
507	*Pseudanthias cooperi*		1		
508	*Pseudanthias pascalus*	1	1	1	
509	*Pseudanthias pleurotaenia*		1	1	
510	*Pseudanthias squamipinnis*	1	1	1	
511	*Pseudanthias tuka*	1	1	1	
512	*Pseudobalistes flavimarginatus*		1	1	
513	*Pseudobalistes fuscus*	1	1	1	
514	*Pseudocheilinus evanidus*	1	1	1	1
515	*Pseudocheilinus hexataenia*	1	1	1	1
516	*Pseudochromis sp.*				1
517	*Pseudochromis tapeinosoma*				1
518	*Pseudocoris yamashiroi*			1	
519	*Pseudodax moluccanus*	1	1	1	
520	*Pseudogramma polyacanthus*				1
521	*Pseudojuloides cerasinus*		1		
522	*Pseudoplesiops annae*				1
523	*Pseudoplesiops sp.*				1
524	*Pseudoplesiops wassi*				1
525	*Pteragogus cryptus*	1	1		1
526	*Pteragogus sp.*	1	1		
527	*Ptereleotris evides*	1	1	1	

528	*Ptereleotris zebra*		1	1	
529	*Pterocaesio digramma*	1	1		
530	*Pterocaesio marri*		1	1	
531	*Pterocaesio tile*	1	1	1	
532	*Pterocaesio trilineata*	1	1	1	
533	*Pterois volitans*	1		1	1
534	*Pygoplites diacanthus*	1	1	1	1
535	*Rhinecanthus aculeatus*			1	
536	*Rhinecanthus rectangulus*	1	1	1	
537	*Sargocentron caudimaculatum*		1		
538	*Sargocentron ittodai*				1
539	*Sargocentron spiniferum*	1	1	1	
540	*Saurida gracilis*	1			
541	*Scarini sp.*				1
542	*Scarus altipinnis*	1	1	1	
543	*Scarus chameleon*	1	1	1	
544	*Scarus dimidiatus*		1	1	
545	*Scarus forsteni*	1	1	1	
546	*Scarus frenatus*	1	3	1	
547	*Scarus globiceps*	1	1	1	
548	*Scarus longipinnis*	1	1	1	
549	*Scarus niger*	1	1	1	
550	*Scarus oviceps*	1	1	1	
551	*Scarus psittacus*	1	1	1	
552	*Scarus rubroviolaceus*	1	1	1	
553	*Scarus schlegeli*	1	1	1	
554	*Scarus spinus*	1	1	1	
555	*Scarus viridifucatus*			1	
556	*Scarus xanthopleura*	1	1	1	
557	*Scolopsis bilineata*	1		1	
558	*Scomberoides commersonianus*		1		
559	*Scomberoides lysan*		1	1	
560	*Scomberoides sp*			1	
561	*Scomberomorus commerson*			1	
562	*scorpaenid sp.*				1
563	*Scorpaenodes corallinus*				1
564	*Scorpaenodes guamensis*				1
565	*Scorpaenopsis macrochir*				1
566	*Scorpaenopsis sp.*				1
567	*Sebastapistes corallinus*				1
568	*Sebastapistes cyanostigma*				1
569	*Sebastapistes cyanostigma*			1	
570	*Serranocirrhites latus*	1	1	1	
571	*Siganus argenteus*	1	1	1	
572	*Siganus corallinus*	1	1		
573	*Siganus puellus*	1			
574	*Siganus punctatissimus*		1		
575	*Siganus punctatus*	1	1	1	
576576	*Siganus vulpinus*	1	1	1	
577	*Siganus woodlandi*	1	1		
578	*Siphamia tubifer*				1
579	*Sphyraena barracuda*	1	1	1	
580	*Sphyraena forsteri*		1		
581	*Stegastes fasciolatus*	1	1	1	
582	*Stegastes gascoynei*	1			
583	*Stegastes nigricans*	1	1	1	1
584	*Stegostoma fasciatum*	1	1		
585	*Stethojulis bandanensis*	1	1	1	1
586	*Stethojulis interrupta*	1			
587	*Stethojulis strigiventer*	1	1	1	

588	*Sufflamen bursa*	1	1	1	
589	*Sufflamen chrysopterum*	1	1	1	
590	*Suttonia lineata*				1
591	*Synodus binotatus*				1
592	*Synodus dermatogenys*				1
593	*Synodus variegatus*	1	1	1	
594	*Synodus varigatus*				1
595	*Taeniura lymma*		1		
596	*Taeniura meyeni*	1	1		
597	*Thalassoma amblycephalum*	1	1	1	1
598	*Thalassoma hardwicke*	1	1	1	
599	*Thalassoma lunare*	1	1	1	
600	*Thalassoma lutescens*	1	1	1	1
601	*Thalassoma nigrofasciatum*	1	1	1	
602	*Thalassoma purpureum*	1	1	1	
603	*Thalassoma quinquevittatum*	1	1	1	
604	*Thalassoma trilobatum*		1	1	
605	*Thysanophrys celebicus*				1
606	*Trachinotus baillonii*			1	
607	*Trachinotus blochii*			1	
608	*Triaenodon obesus*	1	1	1	
609	*Trimma caesiura*				1
610	*Trimma emeryi*				1
611	*Trimma lantana*				1
612	*Trimma macrophthalma*				1
613	*Trimma maiandros*				1
614	*Trimma milta*				1
615	*Trimma necopinna*				1
616	*Trimma okinawae*				1
617	*Trimma sp.*				1
618	*Trimmatom eviotops*				1
619	*Trimmatom macropodus*				1
620	*Trimmatom nanus*				1
621	*Trimmatom sp.*				1
622	*Ucla xenogrammus*				1
623	*Valenciennea strigata*		1	1	
624	*Variola albimarginata*		1	1	
625	*Variola louti*	1	1	1	
626	*Xenisthmus eirospilus*				1
627	*Zanclus cornutus*	1	1	1	
628	*Zebrasoma scopas*	1	1	1	
629	*Zebrasoma velifer*	1	1	1	
		318	**383**	**347**	**213**