CORAL SEA MARINE PARK CORAL REEF HEALTH SURVEY 2023

Report on reef surveys February - March 2023

PRODUCED FOR PARKS AUSTRALIA, MAY 2023
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In responding to a tender from Parks Australia, a team of researchers representing the College of Science and Engineering at James Cook University (JCU) completed surveys of eleven reefs in the Coral Sea Marine Park.

On the cover – A mixed school of planktivorous fishes and high coral cover on the reef crest of Ashmore Reef, northern Coral Sea Marine Park, 7th March 2023. Photograph taken by Victor Huertas

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

Eight members of the Meriam people joined our team during surveys of Ashmore and Boot Reefs during Feb-Mar 2023. Taiku Wailu can be seen here observing Josie Chandler (JCU) surveying coral assemblages on Ashmore Reef.
Image credit: Victor Huertas

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We are indebted to Rob Benn (owner/skipper) and the entire crew and staff of MV Iron Joy for enabling this work, despite sometimes trying conditions.

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. Indeed, previous surveys (2020-2022) of shallow reef habitats across the Coral Sea Marine Park (CSMP) documented widespread and severe bleaching of corals in 2020 and 2021. These back-to-back bleaching events led to a substantial (52%) decline in coral cover in shallow (<15m depth) reef habitats throughout the CSMP.

James Cook University was commissioned by Parks Australia to assess:

(i) the latest condition of benthic, fish and invertebrate communities within the CSMP;

(ii) any ongoing impacts of the back-to-back (2020 and 2021) bleaching events on benthic, fish and invertebrate communities; and

(iii) gain some understanding of the resilience and biodiversity of the CSMP 'bright spot' reefs.

The project undertook detailed surveys of coral, fish and macro-invertebrate communities and associated reef health at ten CSMP reefs over a 4.5-week voyage February-March 2023. Surveys were conducted to provide rigorous quantitative information on temporal (i.e., 2020, 2021, 2022 and 2023) 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 48 sites across 11 reefs in the CSMP, spanning 7.6 degrees of latitude (~1,600 km) from Boot Reef in the northern CSMP (10.0ºS) to Diamond Islets in the central CSMP (17.6ºS).

The surveys revealed that average cover of hard (scleractinian) corals in shallow habitats across the 11 reefs in 2023 was 20.0% (±1.9 SE), ranging from 6.0% at Lihou Reef in the central CSMP up to 35.2% at Ashmore Reef in the northern CSMP. There was, however, a marked difference in the temporal change (2022 to 2023) in coral cover between the central and northern CSMP. Shallow water coral cover decreased by 6.7% from 2022 to 2023 in the central CSMP, and increased by 8.9% in the northern CSMP over the same period. The decline in coral cover on the central CSMP reefs from 2022 to 2023, albeit relatively small, occurred against a shifted baseline of coral communities, with the abundance of bleaching sensitive coral taxa being reduced due to previous (i.e., 2016, 2017, 2020, and 2021) bleaching events. This reduction in coral cover compounded on previous declines due to the 2020 and 2021 bleaching events, resulting in a 66.9% decline in coral cover from 2020 to 2023 (2020: 30.1%; 2023: 10.0%). Coral cover on northern CSMP reefs has declined by 29.6% over the same period.

Although only low-moderate levels of bleaching (Pale - Recently Dead) were recorded across central CSMP reefs in February 2022 (11.9% of colonies surveyed), a large area of the central CSMP was exposed to significant heat stress in March-April 2022. The geographic footprint of this heat stress event coincides with the recorded declines in coral cover from our surveys (Figure 1). In the absence of other major disturbances, the recorded declines in coral cover are most likely attributable to the elevated ocean temperatures experienced in March-April 2022. To our knowledge this is the first record of three consecutive bleaching events on coral reefs globally. It is important to note that there was significant heat stress around Marion and Mellish Reefs in March-April 2022, however these reefs were not surveyed in 2023. Surveys in February 2024 will be critical to determine the effects of this heat stress on these reefs.

Importantly, coral cover on previously identified 'bright spot' reefs remained relatively high. For example, despite a 51.9% decline in coral on Moore Reefs since 2020 (2020: 39.5%; 2023: 19.0%), the present coral cover is the highest of the seven central CSMP reefs surveyed, and almost double that of the average for central CSMP reefs (10.0%). Similarly coral cover at the three other 'bright spot' reefs surveyed remained considerably higher than the regional average (Ashmore: 35.2%; Boot: 22.8%; Bougainville: 31.3%). In contrast, four reefs in the central CSMP had average coral cover lower than 10% (Lihou: 6.0%; Diamond Islets: 7.0%; Herald Cays: 7.5%; Willis Islets: 8.9%). While these levels of coral cover are equal to or higher than historical estimates of coral cover on some central CSMP reefs (i.e., 1-6%: Herald Cays, Chilcott Islet and Lihou Reef), such low levels of coral cover have been shown to disrupt key processes and have lasting consequences for the diversity and functioning in other reef systems. It is currently unknown if low coral cover will have the same consequences in reef systems such as the CSMP where coral cover in shallow reef habitats has been historically low. Continued monitoring over the next 5-10 years will be critical to assess whether coral populations and coral cover on these reefs recovers, or collapses, and any associated changes in reef fish and invertebrate communities.

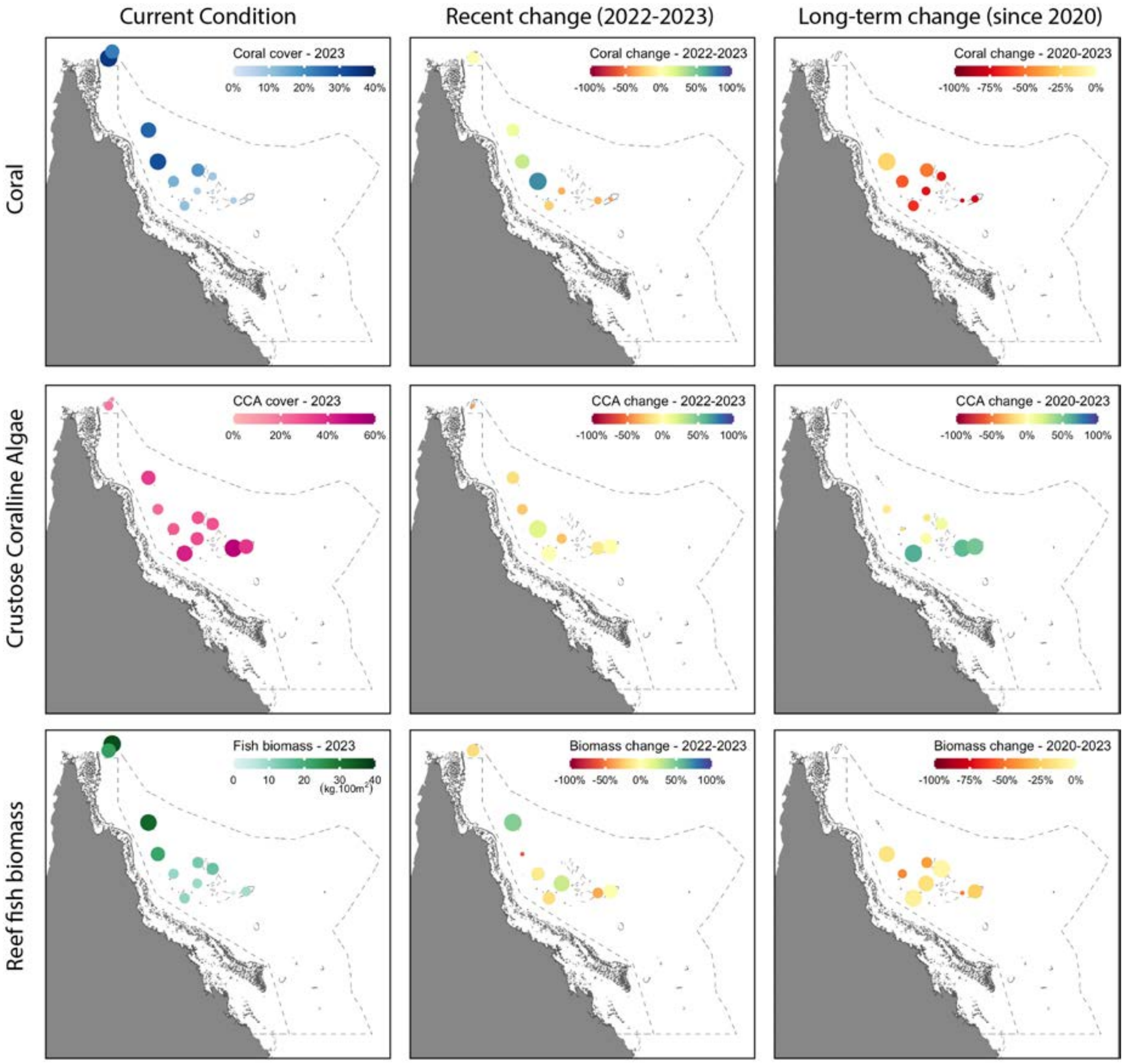

Figure 1. Summary of the current (2023) condition and recent and longer-term trends in coral cover, crustose coralline algae cover, and reef fish biomass across the central and northern Coral Sea Marine Park. Values are averaged across habitats and sites on each reef, and based on surveys conducted during 2020, 2022 and 2023.

Turf- and/or macro-algae often increase in abundance following widespread coral loss, however we found no evidence of such increases following widespread coral loss in the CSMP from 2020 to 2023. Rather, the cover of crustose coralline algae (CCA) increased in both the central and northern CSMP from 2020 to 2023, and largely offset the declines in coral cover over the same period. It appears therefore that CCA's, rather than turf- or macro-algae, are able to rapidly colonise and/or grow to cover recently dead coral skeletons. This is important as CCA's are a critical component of healthy coral reef ecosystems, contributing to reef calcification, the induction of coral larvae to settle, and the provision of 3-dimensional structure for reef associated species.

The density of juvenile corals (an indicator of the recovery potential of coral populations) recorded across the 11 CSMP reefs in 2023 was high (44.2 juveniles per 10m^2), and almost double that recorded in 2022 (23.1 juveniles per 10m^2). This increase was largely driven by the higher densities recorded at Ashmore and Boot Reefs (76.0 and 63.5 juveniles per 10m^2, respectively) in the far north of the CSMP, and likely reflects their proximity and hence connectivity with reefs of the Torres Straits and Eastern Fields (PNG). Comparisons of sites that have been repeatedly surveyed since 2020, show the density of juvenile corals have increased by 10-20% from 2022 to 2023 (central CSMP: 24.4 to 29.6 juvenile per 10m^2; northern CSMP 27.5 to 30.0 juvenile per 10m^2), and are 2- to 6-times greater than the densities recorded prior to and during the 2020 bleaching event. While the increase in the densities of juvenile corals will aid in the recovery of the coral populations across CSMP reefs, the majority of these juvenile corals likely settled onto these reefs prior to the 2020 bleaching event, and as such the full impacts of the 2020, 2021, and 2022 bleaching events on the supply and settlement of coral larvae are yet to be realised. The deployment of coral settlement tiles on some CSMP reefs (namely Osprey and Bougainville Reefs) is planned for October 2023, and will provide greater insight into the supply and settlement of coral larvae to CSMP reefs.

There was very little evidence of thermal stress (i.e., bleaching) across the eleven CSMP reefs surveyed in 2023, with less than 1% of colonies surveyed showing signs of bleaching (pale – recent mortality). This extremely low level of bleaching, coupled with the negligible heat stress experienced over much of the CSMP during early (Jan-Apr) 2023, suggests these reefs are unlikely to experience any significant bleaching in 2023. Assuming this holds, it will be the first year in the past four years, and only the third year in the last eight years (i.e., since 2016) that reefs in the CSMP have not experienced severe bleaching.

Widespread 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 macro-invertebrates (i.e., sea urchins, sea cucumbers, *Trochus*, *Tridacna* clams) on CSMP reefs in 2023, the biomass of reef fishes declined on central and northern CSMP reefs. The initial decrease in reef fish biomass (2020-2022) was largely driven by declines in small-bodied planktivorous fishes (e.g., damselfishes), and corallivorous butterflyfishes that are reliant on live coral for shelter and/or food, as well as grazing herbivorous fishes. While the biomass of corallivorous and planktivorous fishes remained relatively unchanged between 2022 and 2023, the biomass of grazing fishes (primarily surgeonfishes) continued to decline and is now >60% lower than 2020 levels. The continued declines in the biomass of grazing fishes are difficult to reconcile as several studies have reported substantial increases in the abundance and/or biomass of herbivorous fishes following large-scale bleaching-induced coral mortality. Such increases have generally been related to an increase in the availability of their preferred feeding substrata (i.e., algal turfs). The immediate and sustained decline of grazing fishes following the 2020 bleaching event suggest that these changes may be related to the physiological response of these fishes to heat stress, and/or the rapid colonisation of dead coral skeletons by CCA (as opposed to algal turfs which are the favoured feeding substrata of these fishes). Further dedicated investigation into the diet and fitness of these fishes on CSMP reefs is required to identify the likely mechanism/s for these declines.

Despite the declines in reef fish biomass on CSMP reefs from 2020 to 2023, the biomass of reef fishes (a key indicator of reef health, together with coral cover) recorded across all CSMP reefs in 2023 remained high (mean = 1,806 kg per hectare) and likely reflects the isolation and relatively low fishing pressure on CSMP reefs.

While the immediate impacts of the 2020, 2021, and 2022 bleaching events on CSMP reefs are apparent, continued monitoring will be critical to assess any longer-term impacts on the structural complexity of habitats and reef associated fishes and invertebrates, and the potential recovery of shallow water coral assemblages across the CSMP.

The latest (2023) surveys revealed:

- Total shallow water coral cover decreased from 14.7% in 2022 to 14.0% in 2023 across the ten reefs that were surveyed in each year, a mean decline of 5%. The change in coral cover varied among regions ranging from a 6.7% decline in the central CSMP, to a 8.9% increase in the northern CSMP. There was also considerable variation in the change in coral cover among reefs (e.g., 40% decline at Lihou Reef vs a 56% increase at Holmes Reef).
- Although only low-moderate levels of bleaching were recorded across central CSMP reefs in February 2022 (11.9% of colonies), the central and eastern region of the Queensland Plateau in the central CSMP (including Herald Cays, Diamond and Willis Islets, and Lihou and Moore Reefs) was exposed to seawater temperatures above those expected to cause bleaching-induced mortality March-April 2022. In the absence of any other major disturbance, the observed declines in coral cover on central CSMP reefs in 2023 are most likely attributable to elevated temperatures experienced in March-April 2022, and represents the fifth major bleaching event in the CSMP in the past seven years. It is also the first record of three consecutive bleaching events on coral reefs globally.
- Low levels of bleaching (<1% of colonies surveyed) were recorded across CSMP reefs in February-March 2023, and coupled with the negligible heat stress experienced over most of the CSMP during early (Jan-Apr) 2023, suggests these reefs are unlikely to experience any significant bleaching in 2023. Assuming this holds, it will be the first year in the past four years, and only the third year in the last eight years (i.e., since 2016) that reefs in the CSMP have not experienced severe bleaching.
- The reduction in coral cover in 2023 (5% decline) while considerably lower than the declines recorded following the two previous bleaching events (2020: 39% decline; 2021: 18% decline), occurred against an increasingly shifted baseline of coral communities, with the cover of bleaching-susceptible coral taxa being severely reduced following the 2016, 2017, 2020, and 2021 bleaching events. Collectively, the three most recent bleaching events (2020, 2021, 2022) have led to a 67% decline in shallow water coral cover on central CSMP reefs, and a 30% decline in coral cover on northern CSMP reefs.

- Four reefs in the central CSMP reefs had very low (< 10%) coral cover (Lihou: 6.0%; Diamond Islet: 7.0%; Heralds Cays: 7.5%; Willis Islets: 8.9%). Such low coral cover has been shown to disrupt key processes and have lasting consequences for the diversity and functioning of other reef systems. Continued monitoring over the next 5-10 years will be critical to assess whether coral populations on these reefs recover, or collapse, and any associated changes in reef fish and invertebrate communities.
- Despite the significant loss of live corals over the past three years, there were no substantial increases in macroalgae across the CSMP. Rather the cover of crustose coralline algae, an important component of healthy reef ecosystems, has increased in both the central and northern CSMP.
- Eleven fish species that had not been recorded during surveys or observations on the previous voyages (2018-2022) were recorded during the 2023 surveys, taking the total fish species recorded in the CSMP during the past six years of surveys to 650 species. All of these eleven species were recorded at Ashmore Reef in the far north of the CSMP. No new species of coral were observed.
- The biomass of reef fishes declined by 14% from 2022 to 2023 across the ten reefs that were surveyed in each year, and was primarily related to declines in grazing fishes. Grazing fishes are widely viewed as key functional group on coral reefs because of their capacity to remove algal biomass and prevent algal overgrowth following disturbance. The sustained declines in the abundance and biomass of grazing fishes following the 2020 bleaching event are counter to those reported for coastal and continental shelf reef systems, and suggest that these changes may be related to the rapid colonisation of dead coral skeletons by CCA (as opposed to algal turfs which are the favoured feeding substrata of these fishes).
- Ashmore, Boot, Bougainville, and Moore Reefs, previously identified as ‘bright spots’ in terms of coral cover, richness and/or fish biomass, were again standouts. All four reefs appear to have been less adversely affected by the recent bleaching events than other CSMP reefs, with the highest coral cover being recorded on Ashmore Reef (35.2%), while coral cover at Moore Reefs (19% cover) was almost double that of other reefs in the central CSMP

(average = 10% cover). This is despite coral cover at Moore Reefs declining by 52% from the level recorded in 2020 (39.5%) when it was last surveyed. The fifth bright spot reef, Mellish Reef, was not surveyed in 2023 and will be surveyed in February 2024.

- 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 voyages. These leveraged projects represent a significant in-kind contribution and collectively, will increase our understanding of the movement and connectivity of sharks and large reef fishes, identify fish spawning aggregation sites, and promote the unique nature of the CSMP.

In conclusion, the three consecutive bleaching events (2020, 2021, and 2022) have had a significant impact on coral and reef fish communities across most CSMP reefs surveyed. The 2022 bleaching event while spatially restricted to the central CSMP and causing a relatively small (6.7%) decline in coral cover, occurred against an increasing shifted baseline of coral communities, with the cover of bleaching-susceptible coral taxa being reduced following the 2016, 2017, 2020, and 2021 bleaching events. Importantly, the 2022 bleaching event was the fifth major bleaching event in the CSMP in the last 7 years (2016, 2017, 2020, 2021, and 2022), and the first record of three consecutive bleaching events on coral reefs globally. These recent bleaching events in the CSMP are 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 potential recovery and resilience of these isolated reef systems in the absence of local anthropogenic stressors, and the longer-term and ongoing impacts of these bleaching events on reef associated species.

Recommendations for future monitoring and research:

- Given the increasing incidence of major disturbances impacting CSMP reefs in recent years (namely five bleaching events in the past seven years, and three consecutive bleaching events in the past three years), coupled with 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 would be largely unknown, severely limiting the capacity of managers to understand the health status of these reefs and make informed decisions. For example, Marion and Mellish Reefs experienced significant heat stress in Mar-Apr 2022, however the impacts on reef communities are currently unknown and may be confounded by any future events that occur prior to these reefs being re-surveyed.

- Continued monitoring (annual or biennial) should prioritise reefs and sites that have been repeatedly surveyed since 2020. Continued monitoring of these existing sites is critically important to determine any longer-term effects of the three recent bleaching events (2020, 2021, and 2022) 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 of collapse.
- Continued regular (annual or biennial) monitoring of coral, fish, sea snake and invertebrate communities should be conducted on a subset of 10-12 representative reefs, with all 22 CSMP reefs to be re-surveyed every 3-5 years. These representative reefs should include the five ‘bright spot’ reefs (i.e., Ashmore, Boot, Bougainville, Moore and Mellish Reefs), as well as reefs that are adjacent to the ‘bright spot’ reefs and/or on-route between reefs to facilitate comparisons and maximise the available vessel time.
- Dedicated monitoring of deep reef and non-reef habitats using remotely operated underwater vehicles (ROVs) should be continued and expanded. These deep habitats are more extensive than shallow reef habitats, largely unexplored and likely less impacted by disturbance. Monitoring will provide a more comprehensive understanding of the composition and health of these unique deep water habitats, increase our understanding of potential links between shallow reef and deep reef and non-reef habitats, while also maximising the use of available berths on the vessel.
- At least 2 days should be spent at each of the representative reefs (weather and conditions permitting) to allow for surveys of additional sites and habitats and targeted research and monitoring. Ideally 5-7 days should be spent at

one select reef during each (annual or biennial) voyage year to allow a greater number of sites to be surveyed (i.e., 3-4 sites per day) and thereby provide a more comprehensive understanding of the composition and health of both shallow and deep reef and non-reef habitats of that reef.

- Additional means for accessing CSMP should be considered, including the provision of berths on the CSMP Island Health voyages (and vice-versa) and the use of berths on dive tourism vessels.
- Repeat the 3-dimensional habitat mapping of sites mapped during the 2019-2020 voyages in the next 2-3 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.
- Increased focus on quantifying demographic rates of benthic (namely corals and crustose coralline algae; CCA) and fish taxa to better understand the replenishment and potential resilience of populations to environmental change. Devices to quantify the settlement and calcification of CCA's were deployed across eight CSMP reefs during the 2023 voyage and coral settlement tiles will be deployed on two CSMP reefs in Oct 2023. Quantifying demographic rates for fish and identifying potential settlement and nursery habitats will require dedicated research.
- Dedicated investigation into the diet, fitness, and demographics of grazing fishes on CSMP reefs is required to identify the likely mechanism/s for the observed declines in this group following the recent bleaching events.
- The maintenance and replenishment of populations, and the resilience of reef systems within the CSMP is largely dependent on the connectivity among and within reefs in the CSMP and adjacent regions (i.e., GBRMP, Temperate East Marine Parks Network, New Caledonia, Vanuatu, Solomon Islands and Papua New Guinea). Dedicated collections of animal tissue across these regions and subsequent genetic analyses of these samples are required to understand patterns of connectivity, and how they differ among taxa.
- 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.

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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 together with the adjacent Parc naturel de la mer de Corail (Natural Park of the Coral Sea – New Caledonia) form the largest protected area in the world (*ca.* 2.3 million km^2; Figure 2.1). 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).

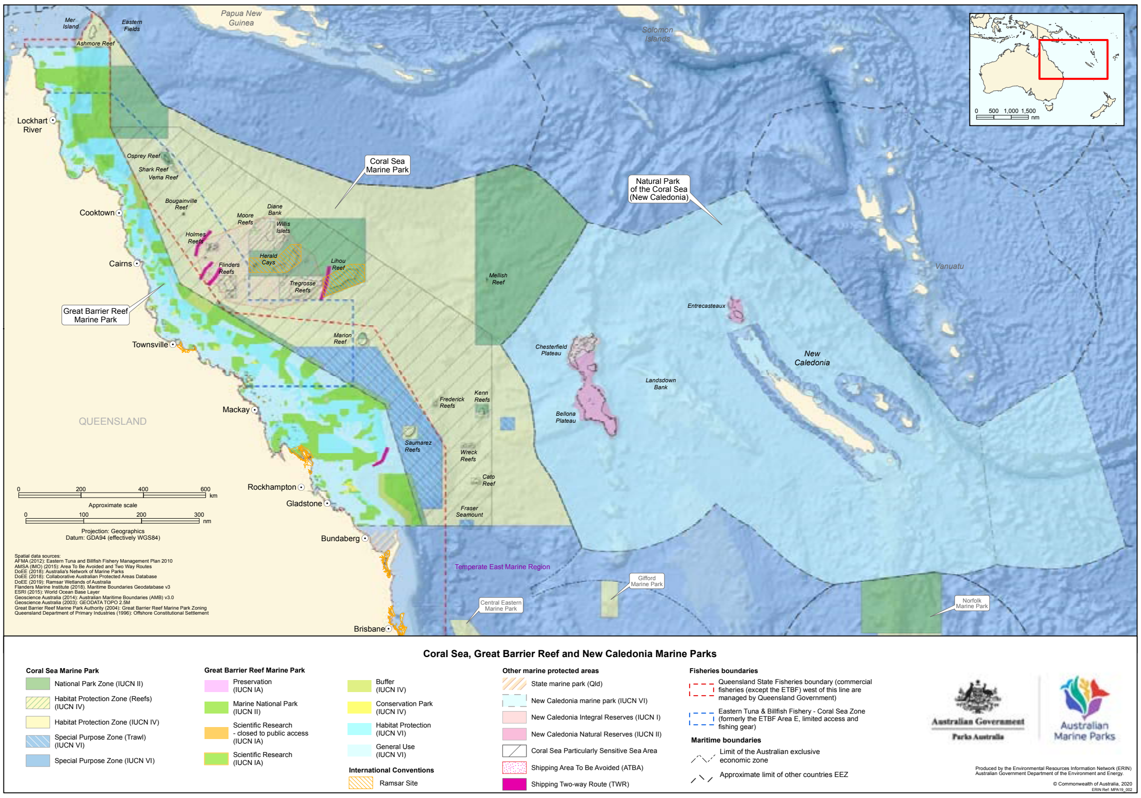


Figure 2.1. Map of the Coral Sea showing the location of the Coral Sea Marine Park, Natural Park of the Coral Sea (New Caledonia). Together these two Marine Parks represent the world's largest protected area showing management zones implemented in July 2018. (Source: parksaustralia.gov.au)

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).

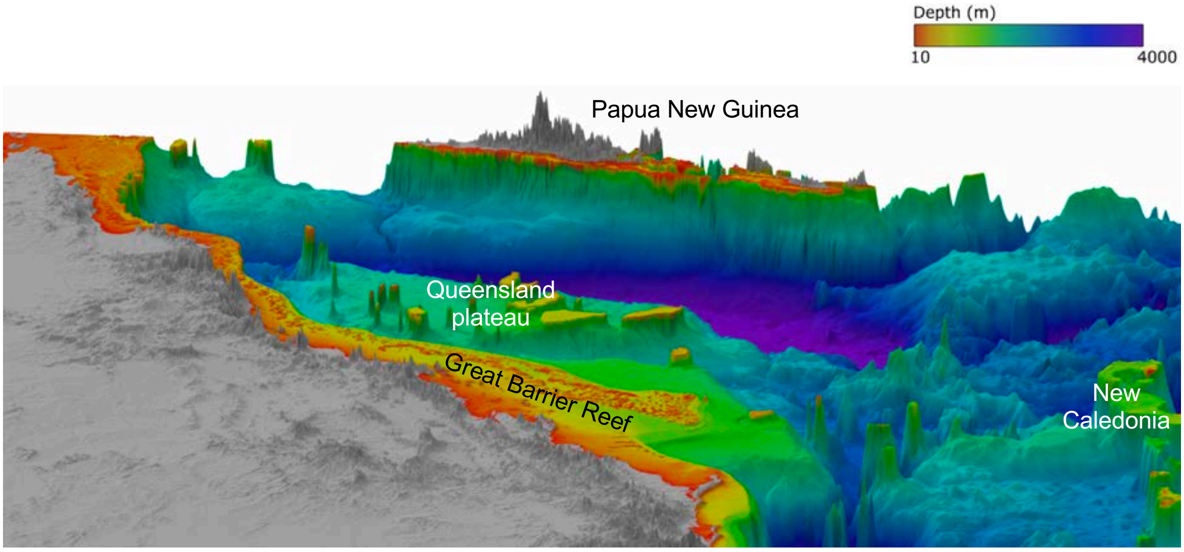


Figure 2.2. Bathymetric map of the Coral Sea showing the location of the Queensland plateau that gives rise to many of the reefs in the central and northern regions of the Coral Sea Marine Park.

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 CSMP has historically been relatively low (ca. 1-6% cover; Ayling and Ayling 1985; Oxley et al. 2003; Ceccarelli et al. 2008). In particular, estimates of coral cover on reefs in the central CSMP indicate coral cover has been relatively low for at least the past 30+ years, with this low coral cover linked to repeated exposure to severe tropical cyclones and more recently climate-induced coral bleaching (Ceccarelli et al. 2013; Harrison et al. 2019; Hoey et al. 2020, 2021, 2022). These frequent disturbances, coupled with the likely reliance on self-recruitment for the recovery of coral populations on isolated reefs (Gilmour et al. 2013), most likely account for sustained low coral cover on these reefs (Oxley et al. 2003, 2004; Ceccarelli et al. 2008; Hoey et al. 2020, 2021, 2022).

The reef habitats (down to 100m depth) 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 Marine Park, 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, 2021, 2022; Galbraith et al. 2022). 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.

Despite the isolated nature and hence limited direct human pressures on CSMP reefs, they are increasingly being exposed to the effects of climate change. Indeed, six major coral bleaching events have been recorded in the CSMP in the past two decades (2002, 2004, 2016, 2017, 2020, and 2021), with four of these bleaching events occurring in the past seven years (Oxley et al. 2004, Harrison et al. 2018, 2019, Hoey et al. 2020, 2021, 2022). The two most recent bleaching events (i.e., 2020 and 2021) were the most severe and widespread, and led to a 52% decline in coral cover in shallow (<15m depth) reef habitats throughout the CSMP. Other bleaching events may have also affected CSMP reefs but went undetected due to its isolation and infrequent scientific surveys. These bleaching events reflect the increasing frequency and intensity of marine heatwaves that are affecting coral reefs globally (van Hooidonk et al. 2016; Hughes et al. 2018; Figure 2.3), and are becoming a major driver of the cover and composition of coral communities on contemporary reefs, and the assemblages of reef fish and other reef-associated taxa they support (e.g., Bellwood et al. 2006a, 2012; Richardson et al. 2018). The effects of these bleaching events, and other major disturbances, may be particularly pronounced on isolated reefs such as those in the CSMP due to the reliance on self-recruitment of coral larvae (i.e., larvae spawned from adult corals on the same reef rather than those nearby) to replenish coral populations (Gilmour et al. 2013).

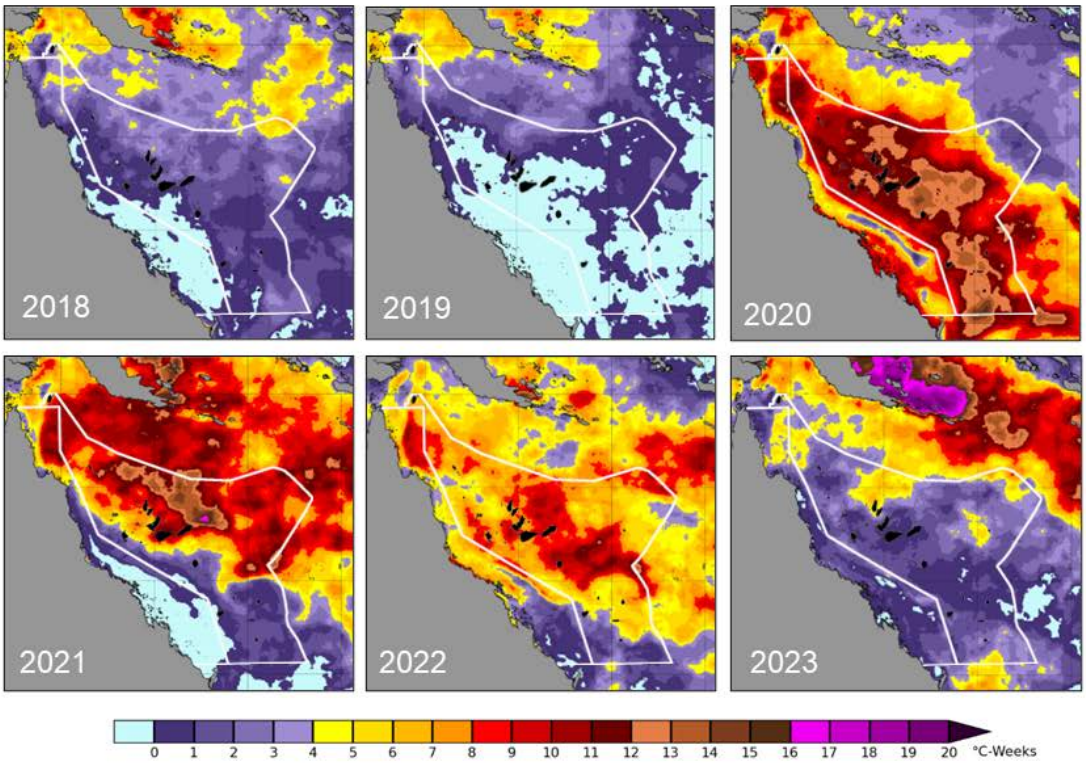


Figure 2.3 Comparison of the maximum Degree Heating Weeks (DHW) experienced throughout the Coral Sea Marine Park during the past six years (2018-2023). Note the maximum DHW in 2021 and 2022 occurred in March of each year. Images produced using the NOAA CRW 5km product v3.1

The combined effects of the back-to-back 2020 and 2021 bleaching events in the CSMP resulted in a mean coral cover decreasing from 27% in 2020 to 13% in 2022 across the eleven reefs surveyed in each year, a mean decline of 52% (Hoey et al. 2022). There was, however, considerable variation in the decline in coral cover among regions (2022: 28% and 26% declines in the northern and central CSMP, respectively, and a 4% increase in the southern CSMP), among reefs (22% increase to a 59% decline), and sites within reefs (Hoey et al. 2022). Importantly, previously identified ‘bright spot’ reefs due to their higher coral cover, richness and/or fish biomass (Hoey et al. 2020) appeared to be less adversely affected by recent bleaching events than other CSMP reefs. For example, coral cover on Ashmore Reef in the far north of the CSMP increased by 35% from 2018 (26.5% cover) to 2022 (35.9% cover), and remained largely unchanged (*ca*. 30% coral cover) on Mellish Reef in the far east of the CSMP while coral cover declined substantially on non ‘bright spot’ reefs over the same period (Hoey et al. 2022). This variation in the response of coral assemblages to heat stress across relatively small spatial scales could reflect differences in the composition of coral communities, local environmental conditions, resilience to heat stress, and/or other unidentified factors (Marzonie et al. 2023), and will likely have flow-on effects to the recovery of coral populations, changes in associated assemblages of reef fish and invertebrates, and the potential resilience of the system as a whole. Future surveys are critical to assess the potential recovery of shallow water coral assemblages following the 2020 and 2021 bleaching events, any ongoing effects of coral loss on associated fish and invertebrate communities, and to better understand the dynamics, and factors that contribute to the performance, of the five 'bright spot' reefs.

2.1 Objectives and scope

The purpose of this study was to provide comprehensive assessments of the current condition of benthic and fish communities within the CSMP, assess the impacts of the back-to-back (i.e., 2020 and 2021) bleaching events on benthic, fish and invertebrate communities throughout the CSMP, and gain some understanding of the resilience and biodiversity of the CSMP 'bright spot' reefs.

Surveys were conducted at eleven reefs throughout the CSMP following the methods of Hoey et al. (2020, 2021, 2022). At each site, diver-based 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) coral health and injuries caused by coral bleaching, disease, or coral predators (e.g., *Acanthaster* spp. and *Drupella* spp.);
iv) 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.

Additional surveys of deeper reef habitats (up to 100m depth) were conducted at each reef using Remotely Operated Vehicles (ROV) fitted with forward-facing stereo-video systems, and side- and down-facing time lapse cameras.

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) Surveys for fish spawning aggregation sites within the CSMP;
iii) Cultural and ecological significance of Ashmore and Boot Reefs;
iv) Filming of a documentary on Sea Country featuring the Meriam people;
v) Vessel grounding assessment at Moore Reefs;
Further details of these projects are provided in Appendix 1.

Figure 2.4 The *MV Iron Joy* anchored off Mer Island, in the eastern Torres Straits, with the JCU research team, CSMP Manager, vessel crew, representatives of the Meriam people and cinematographers from Millstream Productions on the foredeck as part of the leveraged project: *Jewel in the Coral Sea: the cultural and ecological significance of Ashmore and Boot Reefs*. Additional funding for this project was provided through an Our Marine Parks Round Three Grant. Image credit: Stuart Ireland, Millstream Productions

3 Methods

Surveys were undertaken at 48 sites across 11 reef systems within the CSMP during a 31-day voyage, 10th February – 12th March 2023 (Figure 3.1). The 11 reefs surveyed were central CSMP: Flinders (north and south), Holmes (east and west), Lihou, and Moore Reefs, Herald Cays, and Willis, and Diamond Islets; northern CSMP: Ashmore, Boot, Bougainville and Osprey Reefs (Appendix 2). To facilitate direct comparisons in coral health and reef condition among years we re-visited the sites that were surveyed during 2020-2022 (i.e., including the two most recent bleaching events; Hoey et al. 2020, 2021, 2022), or if the reef wasn't surveyed in 2020 or 2021 (i.e., Boot Reef) we attempted to revisit sites that had been surveyed prior to 2020. Sites were relocated using GPS waypoints and a bearing of the direction of the transects from that waypoint.

3.1 Sampling design – diver-based surveys

31 days
11 reefs - 48 sites
15 km of UVC surveys
>390 diver hours

At each site, diver-based surveys were generally 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). The only exceptions to this were one site at Willis Islets (Willis site 4) where the reef crest could not be safely accessed due to excessive surge and wave action, and one site inside the lagoon at Boot Reef (Boot site 8) where there wasn't sufficient depth to differentiate the reef slope and crest habitats. 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).

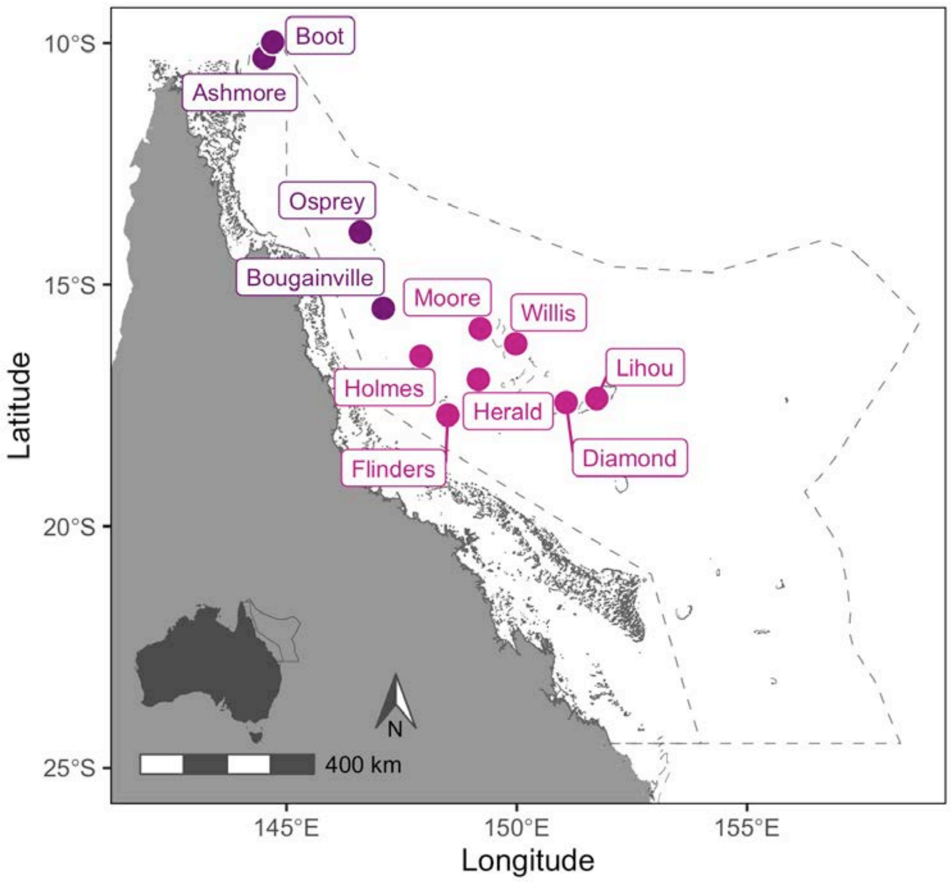


Figure 3.1 Map of the surveyed reefs in the Coral Sea Marine Park in February and March 2023. Colours relate to the regional allocation of reefs in the central (magenta), and northern (purple) Coral Sea Marine Park which are used throughout the report. Regional allocation is based on our current understanding of coral and fish communities. Note: no reefs in the southern CSMP were surveyed in 2023. The five southern CSMP reefs, together with Mellish and Marion Reefs are scheduled to be surveyed in Feb 2024.

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 the size and identity of all larger (>10 cm total length, TL) or motile fish species, within a 5m wide belt (following Hoey et al. 2020, 2021, 2022). 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 fish 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

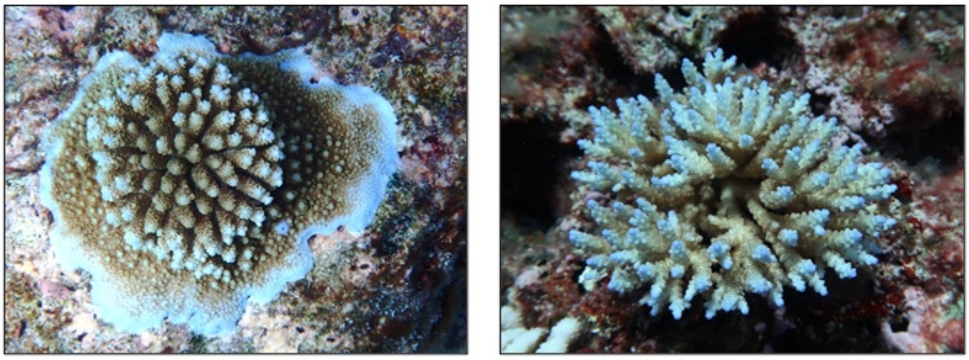
3). The third diver conducted a point intercept survey, providing important information on coral cover and benthic composition, by recording the sessile organisms or substratum underlying evenly spaced (50cm apart) points along the entire length of the transect. The final (fourth) diver assessed coral health, estimated colony size, and counted 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, sea urchins, *Tectus* (formerly *Trochus*), and crown-of-thorns starfish) within a 2m wide belt along the full length of each transect.

3.1.1 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, 2021, 2022). Corals were mostly identified to genus (using contemporary, molecular-based classifications for scleractinian corals), though some of the less abundant genera were pooled to ‘other’ for analyses. We also distinguished major growth forms for *Acropora* (tabular, staghorn, and other) and *Porites* (massive versus columnar or branching). Macroalgae were identified to genus. For survey points that did not intersect corals or macroalgae, the underlying substratum was categorised as either crustose coralline algae (CCA), sponge, sand/ rubble, carbonate pavement, or other (including gorgonians, hydroids, anemones).

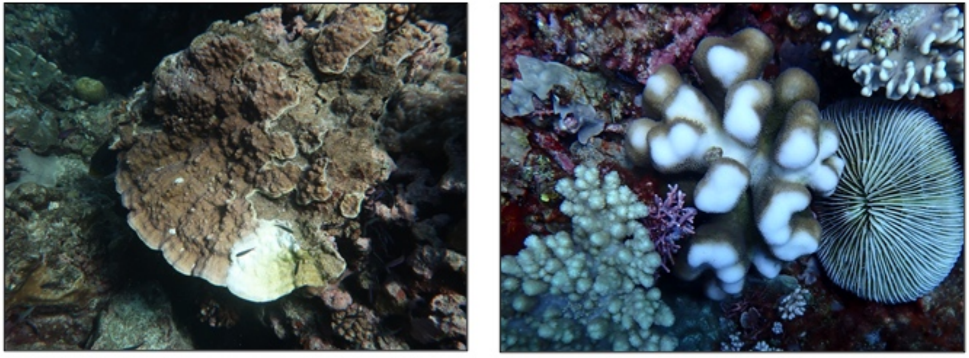
Topographic complexity – Topographic complexity was estimated visually at the start of each transect, using the six-point scale formalised by Wilson et al. (2007), where 0 = no vertical relief (essentially flat homogenous habitat), 1 = low and sparse relief, 2 = low but widespread relief, 3 = moderately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

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 genus and growth form (as described for PIT above), and then assigned to one of five size classes based on their maximum diameter (≤ 5cm, 6-20cm, 21-40cm, 41-60cm and >60cm). The health of each coral colony was then assigned to one of 8 categories (Figure 3.2), 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 injury was also recorded, be it due to coral predators (e.g., *Drupella* spp., crown-of-thorns starfish or parrotfish) observed within or nearby the injured colony, or coral disease.



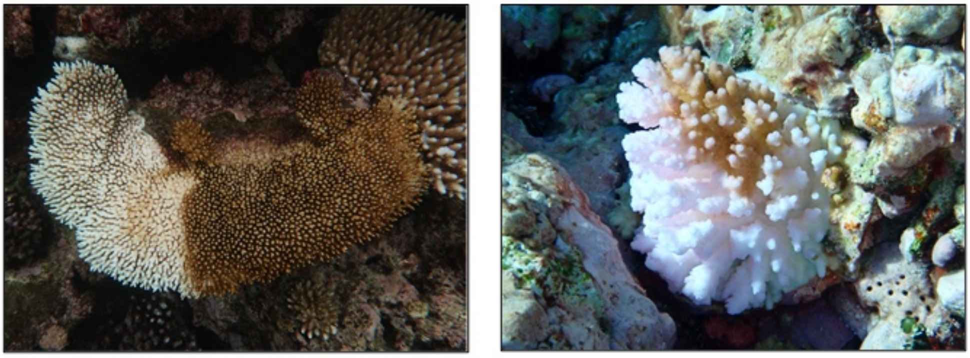
Healthy

Pale



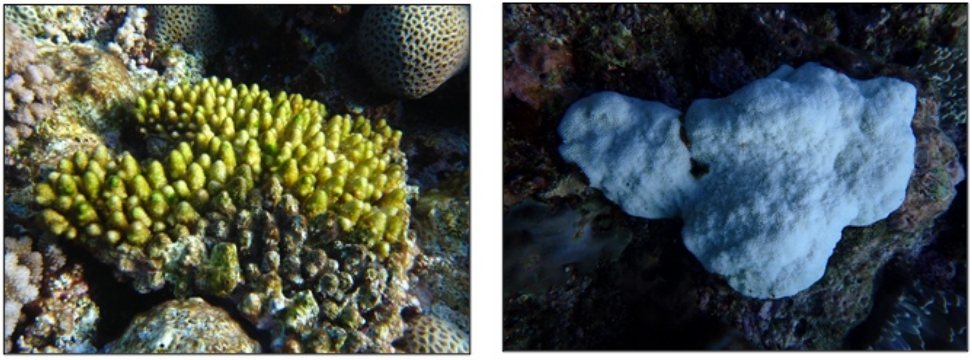
5-50% recent mortality

5-50% bleached



51-99% recent mortality

51-99% bleached



100% recent mortality

100% bleached

Figure 3.2 Coral health categories used for the in-water coral health assessments. Images on the left provide examples of the four injury categories, whilst images on the right are examples of the coral bleaching categories. Image credits: Deborah Burn; Morgan Pratchett

Juvenile corals - Densities of juvenile corals (≤5 cm maximum diameter, following Rylaarsdam 1983) are increasingly used as a proxy for recovery potential of coral assemblages as opposed to quantifying the number of coral larvae that settle on experimental settlement substrata (e.g., tiles). Counting juvenile corals accounts somewhat for the high mortality rates of newly settled corals, and logistically only requires a single visit to the study site. Therefore, 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 potential coral recovery among habitats, sites and reefs across the CSMP. All juvenile corals within the 10 x 1m coral health transect were recorded to genus (Figure 3.3).

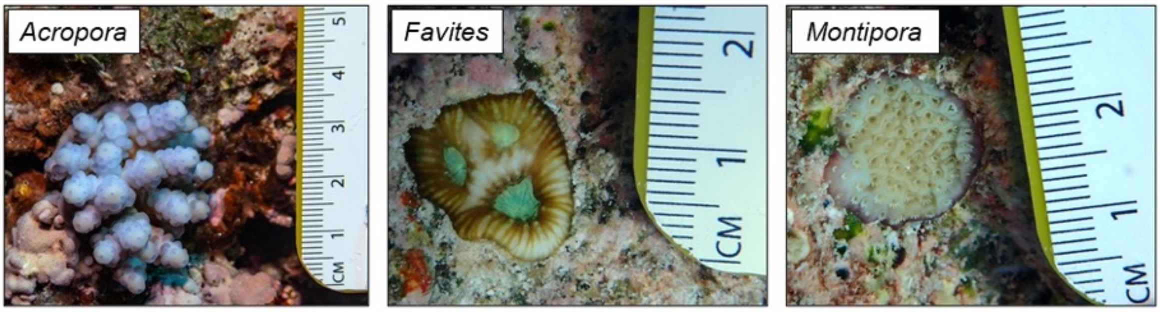


Figure 3.3 Photographs of juvenile (≤5cm diameter) corals recorded within 10m^2 belt transects within the Coral Sea Marine Park. Each juvenile coral within the 10m^2 belt transects were identified to genus and recorded. Image credits: Deborah Burn

Coral settlement – To directly quantify the supply and settlement of coral larvae among reefs and habitats in the CSMP, coral settlement tiles (terracotta tiles: 11 x 11 x 1cm) will be deployed in each of two habitats (reef crest and reef slope) at 2-3 sites around Bougainville and Osprey Reefs in late October 2023, approximately 6-weeks prior to the predicted coral spawning.

Crustose Coralline Algae (CCA) settlement and growth - To directly quantify the settlement and growth (calcification) of crustose coralline algae (CCA) among reefs and habitats in the CSMP, a series of CCA devices were deployed across eight reefs in Feb-Mar 2023. The CCA devices consisted of a length of PVC pipe (15mm diameter x 250mm length) that were attached to a steel bar (12mm diameter x 450 mm length) using cable ties (following Kennedy et al. 2017). The reinforcing bar was driven vertically into the reef framework using a hammer, avoiding areas of live coral (Figure 3.4). Three replicate CCA devices were deployed in each of two habitats (reef crest and reef slope) at 19 sites across eight reefs (2 sites at each of Flinders, Lihou, Moore and Holmes Reefs, and East Diamond Islet; 3 sites at each of Bougainville, Osprey, and Ashmore Reefs; Appendix 3). Each CCA device was individually numbered. These deployments were more widespread and comprehensive than the 2-3 sites originally planned at Bougainville and Osprey Reefs. These additional devices are not scheduled for collection in 2024, rather will be collected on future voyages (e.g., 2025) together with the temperature loggers.

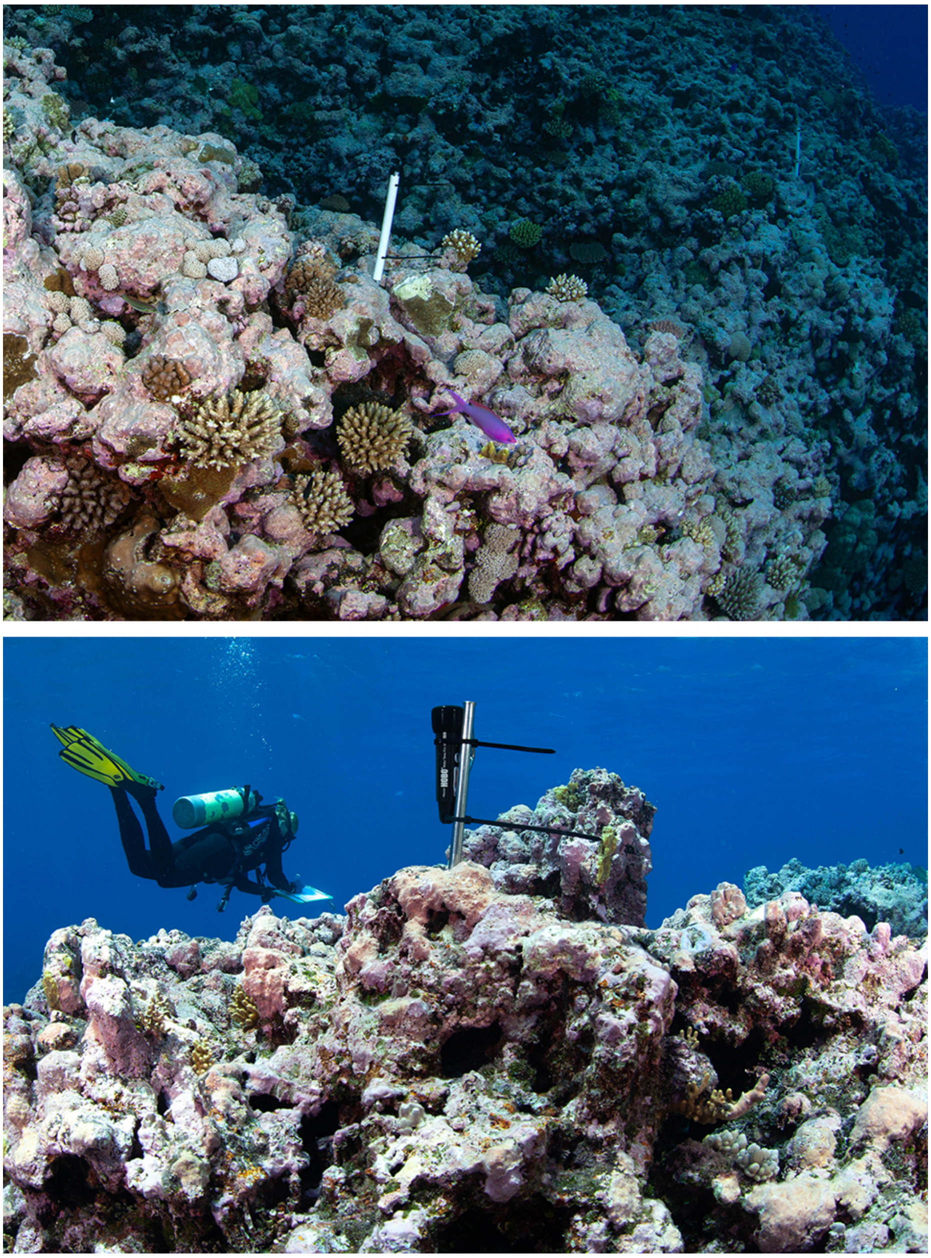


Figure 3.4 Photographs of crustose coralline algae (CCA) devices and temperature loggers deployed in Feb-Mar 2023. Top: CCA devices deployed on the reef slope of Osprey Reef. Bottom: Hobo temperature logger deployed on the reef crest at Willis Islets. Image credits: Victor Huertas.

3.1.2 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 4 for a comprehensive list of species surveyed.

3.1.3 Other reef taxa

Sea snakes – The abundance and size of sea snakes (including the Olive sea snake, *Aipysurus laevis*; Dubois' sea snake, *Aipysurus duboisii*; Spiny headed or Horned sea snake, *Hydrophis peronii*; Turtle-headed sea snake, *Emydocephalus annulatus*; Figure 3.5) 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. Only a single sea snake was recorded during surveys of shallow reef habitats during the 2023 surveys, and as such figures of their abundance are not presented.

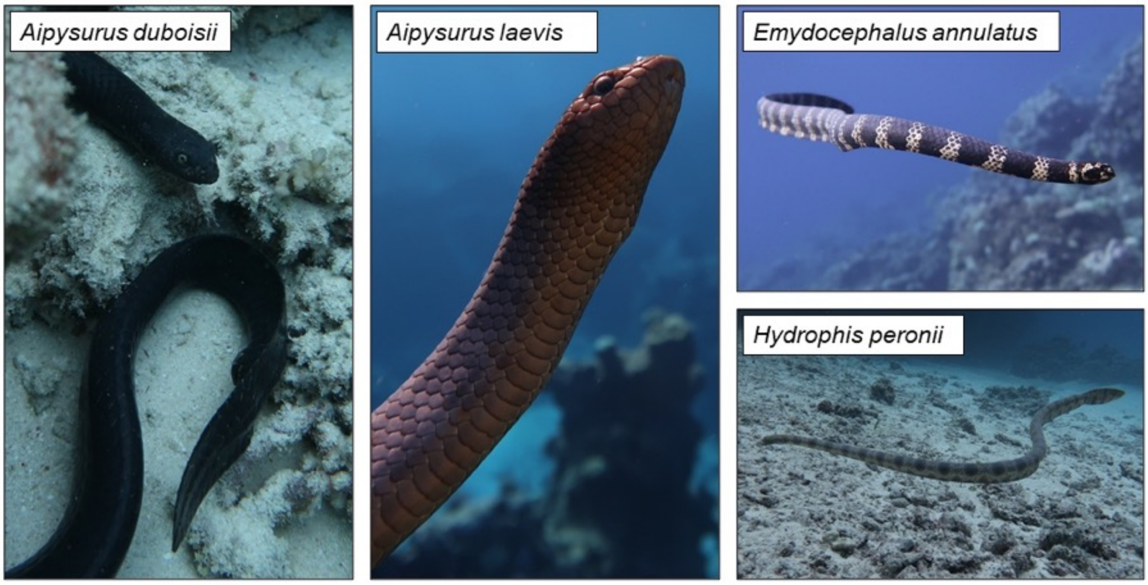


Figure 3.5 Photographs of the four species of sea snake that are commonly observed within the Coral Sea Marine Park; Dubois' sea snake, *Aipysurus duboisii*; Olive sea snake, *Aipysurus laevis*; Turtle-headed sea snake, *Emydocephalus annulatus;* Spiny headed or Horned sea snake, *Hydrophis peronii*. Image credits: Deborah Burn

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; Figure 3.6), 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).

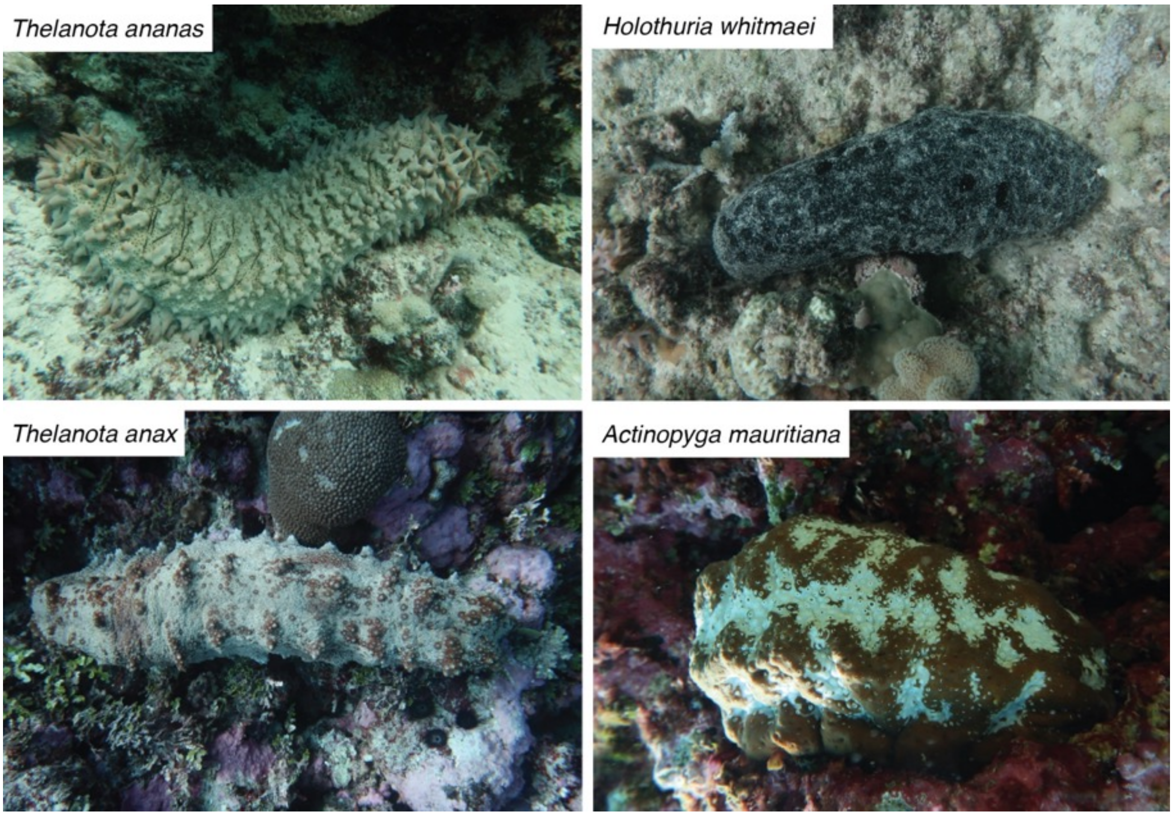


Figure 3.6 Photographs of four species of sea cucumber that are commonly observed within the Coral Sea Marine Park; Prickly redfish, *Thelanota ananas*; Black teatfish, *Holothuria whitmaei*; Amber fish, *Thelanota anax*; and Surf redfish, *Actinopyga mauritiana*. Image credits: Deborah Burn

Coral predators are potentially important contributors to coral reef health and habitat structure, especially during periods of elevated densities (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 population irruptions 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 and 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).

3.2 Sampling design – ROV surveys

Remotely Operated Vehicles (ROVs) were used to survey deeper (15 - 100m depth) reef and non-reef habitats throughout the CSMP. ROVs (BlueRobotics BlueRov2) were fitted with a forward-facing stereo-video system (SVS) to record fish assemblages and enable length estimates of individual fish to be made. SVS cameras (Paralenz + or GoPro Hero 8 systems) were calibrated prior to surveys using the software CAL and associated calibration method (SeaGis Australia). The ROVs were also fitted with downward and sideward facing time lapse cameras (GoPro Hero 8 cameras inside aluminium T-housings) to record benthic assemblages. The time lapse cameras were set to take a photo every 10 seconds, capturing an image of the benthos every ~2m. Combining the SVS to record fish and the time lapse cameras to record the benthos, the ROVs were essentially able to replicate diver-based surveys (see Section 3.1 above) at depths of up to 100m.

The ROVs were deployed, piloted and retrieved from a tender to the main vessel (Figure 3.7). At each site, the ROV was deployed and descended to the maximum depth possible depending on the habitat type, sea conditions, and maximum depth rating of the ROV (i.e., 100m). Once at the target depth, the ROV was positioned ~0.5m above the substratum (or alongside for vertical reef walls), and two timed transects were conducted at a constant depth, with 5-10m between replicate transects. Each transect was 2.5 minutes long and by travelling at a known speed of 0.2 m/s, equated to a distance of approximately 30m. After the second transect, the ROV ascended by ~10m and two transects were conducted at this shallower depth in the opposite direction to the previous two transects (i.e., at the depth band immediately below). This survey pattern was repeated at ~10m depth bands until the two final transects were conducted in the upper 10m (Figure 3.8).

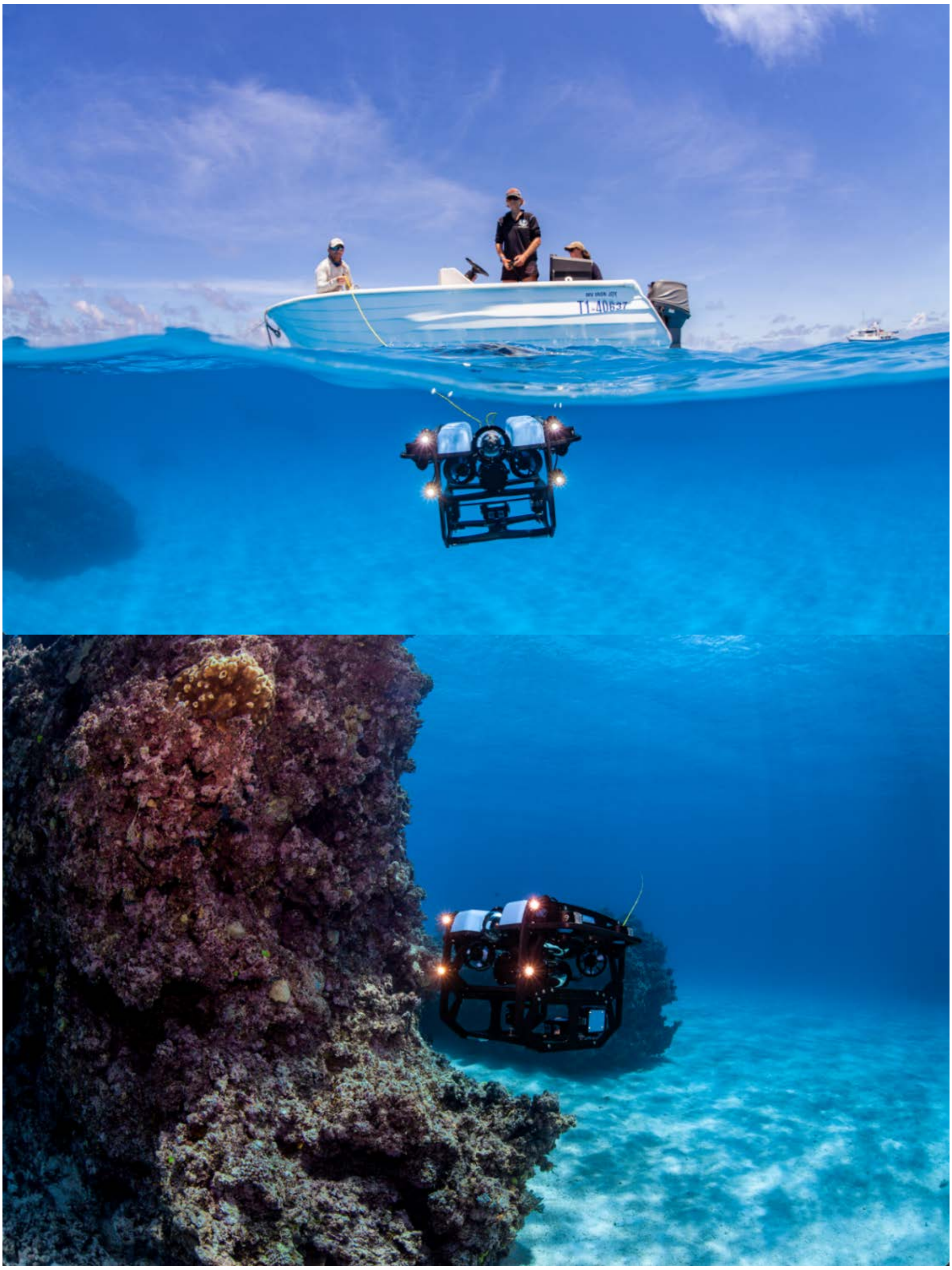


Figure 3.7 Photographs showing the operation of the Remotely Operated Vehicle (ROV) from a tender to the *MV Iron Joy* in the Coral Sea Marine Park. Top: The ROV (Blue Robotics BlueROV2) being deployed from the tender with the operator (Gemma Galbraith, standing) piloting the ROV, while an assistant manages the tether (Ben Cresswell). Bottom: The ROV navigating around a shallow bommie. Image credits: Victor Huertas

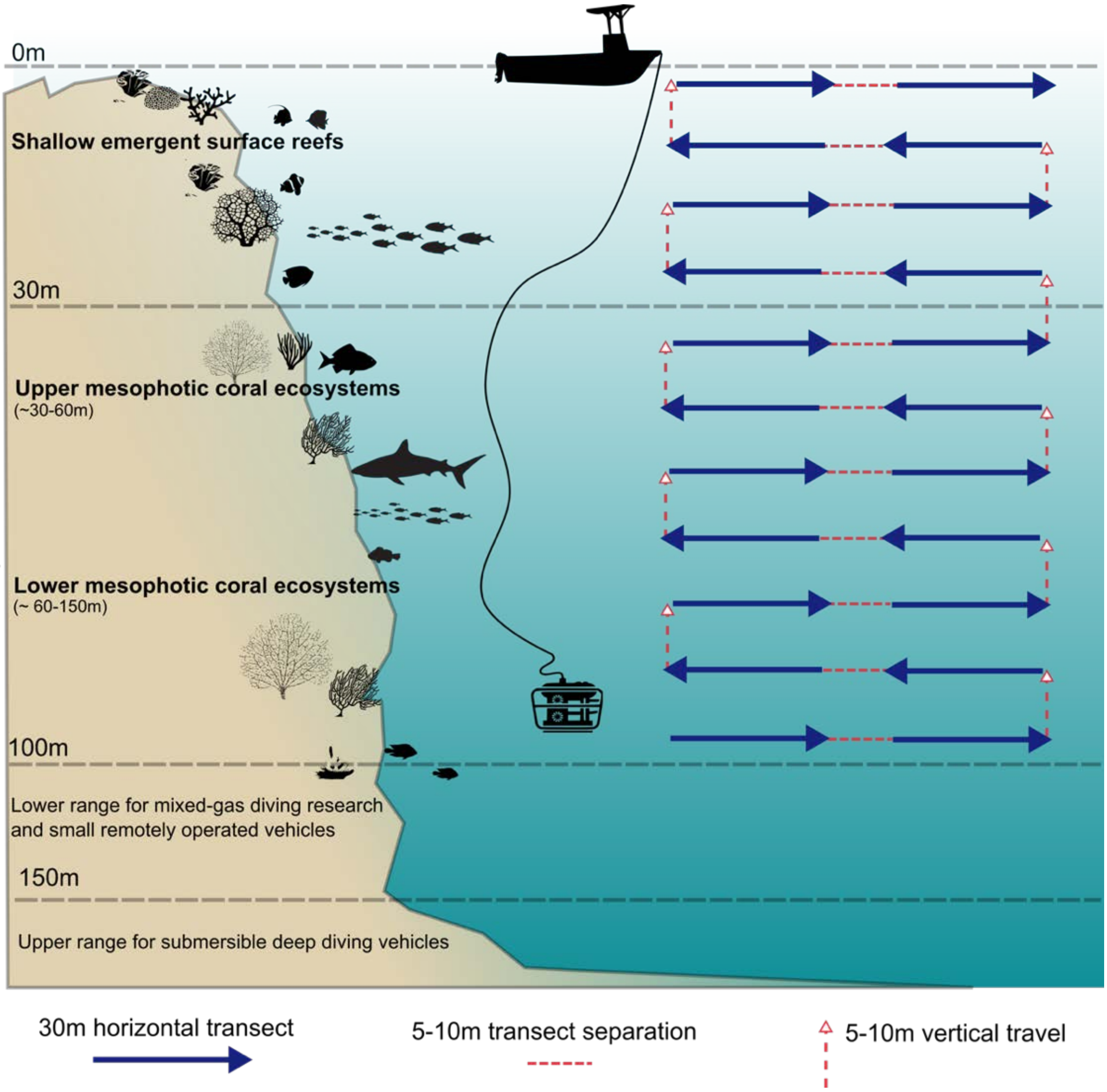


Figure 3.8 Remotely Operated Vehicle (ROV) transect survey methodology used to survey fish and benthic assemblages in the Coral Sea Marine Park. All ROV surveys were conducted at depths between 5 – 100m using the BlueRobotics BlueRov2.

During the 31-day voyage, 10th February – 12th March 2023, 150 ROV transects were undertaken at 18 sites across nine reef systems within the CSMP. The nine reefs surveyed were central CSMP: Flinders (north and south), Holmes (east and west) and Lihou Reefs, Herald Cays, and East Diamond Islet; northern CSMP: Ashmore, Boot, Bougainville and Osprey Reefs. Both ROV units experienced significant technical issues with overheating batteries during the course of the voyage. This led to unpredictable off gassing of electrical components in the electronics enclosure, and caused two significant flooding events. Replacement components were sourced in order to make repairs at sea, however, these equipment issues did reduce the capacity to survey sites at Moore Reefs, Herald Cays and Willis Islets as originally planned.

3.3 Temperature loggers

To directly quantify water temperatures at ‘bright spot’ and adjacent reefs a series of long-term temperature loggers were deployed during Feb-Mar 2023. The temperature loggers (Hobo Water Temp Pro v2 Data Logger - U22-001) were programmed to record water temperature every 30 minutes, and attached to a stainless steel stake driven vertically into the reef substrata, avoiding areas of live coral (Figure 3.4). A temperature logger was deployed in each of two habitats (reef crest and reef slope) at 22 sites across nine reefs (2 sites at each of Flinders, Lihou, and Moore Reefs, and East Diamond and Willis Islets; 3 sites at each of Bougainville, Holmes, Osprey, and Ashmore Reefs; Appendix 3).

3.4 Data handling and analysis

Data from the 2023 surveys were combined with those of the previous voyages (2018-2022) into a single database and analysed using R version 4.1.1 with RStudio interface version 2022.02.3 (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 Authority. Two-dimensional maps of CSMP reefs and boundaries were reproduced from shapefiles generated by Project 3DGBR (Beaman 2012). These maps were produced in R using the package *sf* (Pebesma 2018) and *ggspatial* (Dunnington 2021) using the WGS84 coordinate system. Data for the three-dimensional digital elevation model (i.e., Figure 2.2) came from Project 3DGBR Version 6 (Beaman 2020), rendered in R using the *rayshader* package (Morgan-Wall 2023).

All survey data were averaged across independent transects to obtain a site, or where appropriate a zone (i.e., crest, slope), average prior to summarising data at the level of reefs or regions. For calculations of taxonomic richness, the number of species/taxa were calculated at the level of site (i.e., pooled among transects and reef zone) to give the total number of species/taxa observed at a site, prior to being summarised to the level of reefs or regions. Data are generally presented using box and whisker plots (i.e., box 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 dissimilarity 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 these 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 matrix. All data were analysed in the *vegan* package (Oksanen *et al.* 2020) using the statistical software package R version 4.1.1.

4 *Findings*

4.1 Benthic communities

The back-to-back (2020 and 2021) bleaching events in the CSMP were severe and widespread, and resulted in a decrease in shallow water coral cover from 27% in 2020 to 13% in 2022, a mean decline of 52% (Hoey et al. 2021, 2022). There was, however, considerable variation in the change in coral cover among regions (e.g., change from 2021 to 2022: 28% and 26% declines in the northern and central CSMP, respectively, and a 4% increase in the southern CSMP), among reefs (59% decline at Holmes Reefs to a 22% increase at Saumarez Reef), and sites within reefs (e.g., 18% vs 45% declines at Herald 1 and Herald 4, respectively; Hoey et al. 2022). Understanding the ongoing impacts of, and the potential recovery from, these bleaching events on the cover and composition of coral assemblages, and the associated fish and invertebrate communities, is critical in assessing the current health of reefs in the CSMP.

4.1.1 Coral cover and richness

The average cover of hard (Scleractinian) corals recorded across the 48 CSMP sites surveyed in 2023 was 20.01% (±1.87 SE), ranging from 6.00% (±1.08 SE) at Lihou Reef in the central CSMP up to 35.19% (±2.84 SE) at Ashmore Reef in the northern CSMP (Figure 4.1a). Average coral cover was approximately three-fold greater on reefs in the northern CSMP (averaging 30.95 ± 1.89 %; excl Ashmore Reef: 26.32 ± 1.62 %), compared to the central CSMP reefs (9.95 ± 1.13 %).

The average taxonomic richness of corals across the CSMP, based on the number of hard (Scleractinian) coral taxa (mostly genera) recorded using the 50m point-intercept transects at each survey site, was 15.7 taxa per site and ranged from 10.7 taxa per site (± 3.2 SE) at Willis Islets in the central CSMP to 18.7 taxa per site (± 0.3 SE) at Osprey Reef in the northern CSMP (Figure 4.1b). Coral richness displayed a similar pattern to coral cover among CSMP regions, with coral richness being generally greater in the northern CSMP (17.5 taxa per site), compared to the central CSMP (14.1 taxa per site; Figure 4.1b).

There was considerable variation in both coral cover and richness among reefs within each of the CSMP regions. In the central CSMP, Moore Reefs had both the highest coral cover (19.00 ± 0.03 %) and highest richness (19.0 taxa per site) of the seven central CSMP reefs, while both coral cover and coral richness at Willis Islets (cover: 8.89%; richness: 10.7 taxa per site), Lihou Reef (cover: 6.00%; richness: 12.8 taxa per site) and Herald Cays (cover: 7.50%; richness: 13.0 taxa per site) were lower than the regional average (Figure 4.1). In the northern CSMP, average coral cover ranged from 22.83% at Boot Reef to 35.19% at Ashmore Reef, while coral richness was relatively consistent across reefs (17.0 – 18.7 0 taxa per site; Figure 4.1).

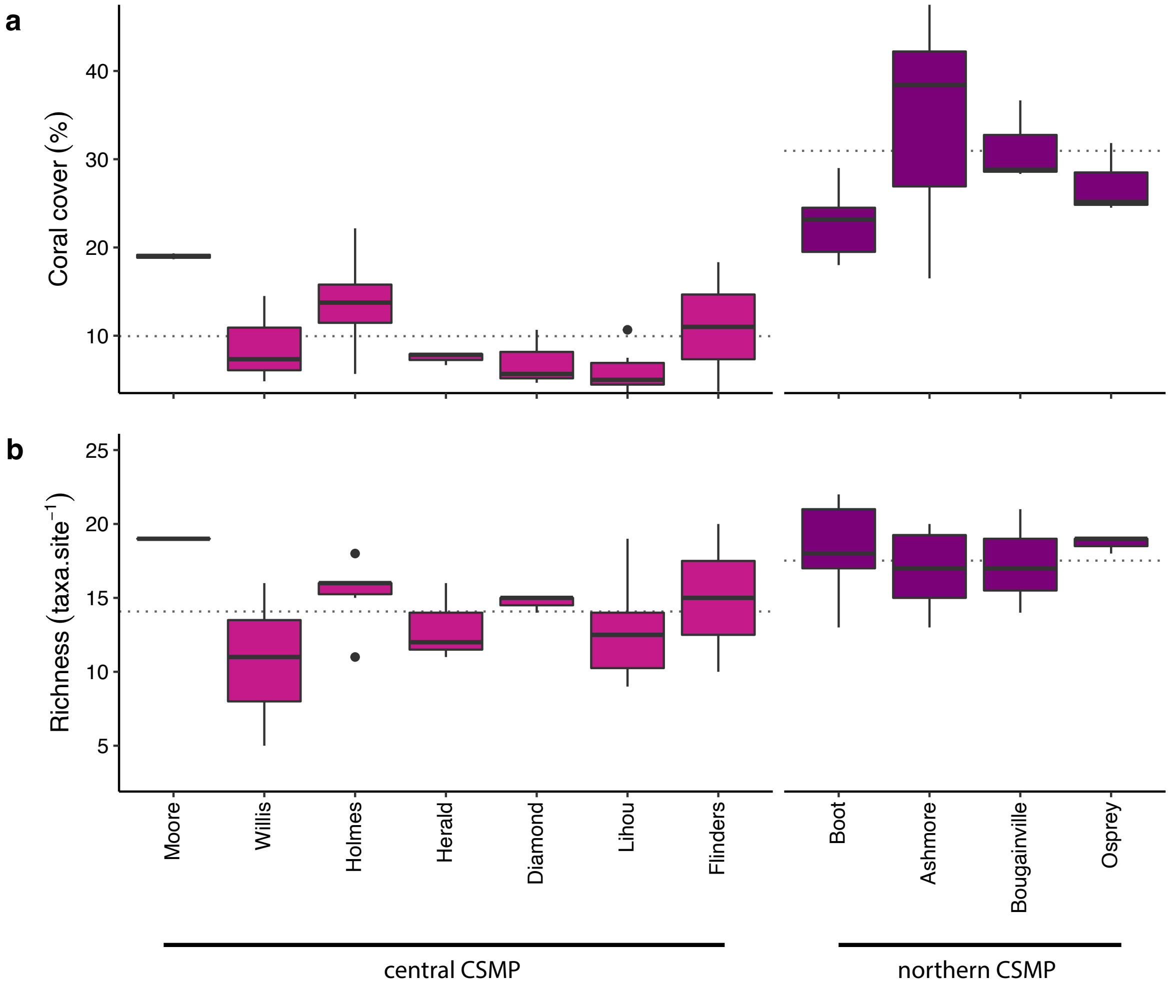


Figure 4.1 Variation in coral cover and coral richness among 11 reefs in the Coral Sea Marine Park (CSMP) in 2023. Data are based on the 50m point-intercept transects, with data for richness based on the number of coral taxa recorded at each of the 48 sites (i.e., pooled across transects and slope and crest habitats). Reefs are arranged into the central and northern CSMP and coloured by *a priori* regional assignments (following Figure 3.1). Dotted lines represent regional averages.

4.1.2 Temporal changes in coral cover and richness

Coral cover - Comparisons of coral cover in shallow reef habitats across the ten reefs that were surveyed in 2020, 2021, 2022, and/or 2023 (i.e., excluding Boot Reef) revealed a marked difference in the temporal patterns in coral cover among the two CSMP regions, most notably from 2022 to 2023 (Figure 4.2). While coral cover declined in both the central and northern CSMP from 2020 to 2022 (i.e., following the 2020 and 2021 coral bleaching events; Hoey et al. 2021, 2022), coral cover continued to decline between 2022 and 2023 in the central CSMP (2022: 10.67%; 2023: 9.95%; a decline of 6.7%), and increased by 8.9% in the northern CSMP over the same period (2022: 26.17%; 2023: 28.50%; Figure 4.2). The decrease in coral cover from 2022 to 2023 was relatively consistent between habitats (i.e., the reef crest: 1-3m depth; reef slope: 7-10m) in the central CSMP, while changes in coral cover on northern CSMP reefs differed between habitats (Figure 4.3). Coral cover was consistently lower on the reef crest (range: 13.8 – 17.0 %) than the reef slope (35.5 – 67.1%) across all years (i.e., 2020-2023), with the increase in coral cover from 2022 to 2023 being largely driven by an 17.2% increase in coral cover on the reef slope (2022: 35.5%; 2023: 41.6%). The decline in coral cover on the central CSMP reefs from 2022 to 2023, albeit relatively small, compounded on previous declines due to the 2020 and 2021 bleaching events (Hoey et al. 2021, 2022). Overall, coral cover has declined by 66.89% from 2020 to 2023 across the central CSMP (2020: 30.05%; 2023: 9.95%), while over the same period the net change in coral cover on the northern CSMP reefs has been a 29.63% decline (2020: 40.50%; 2023: 28.50%).

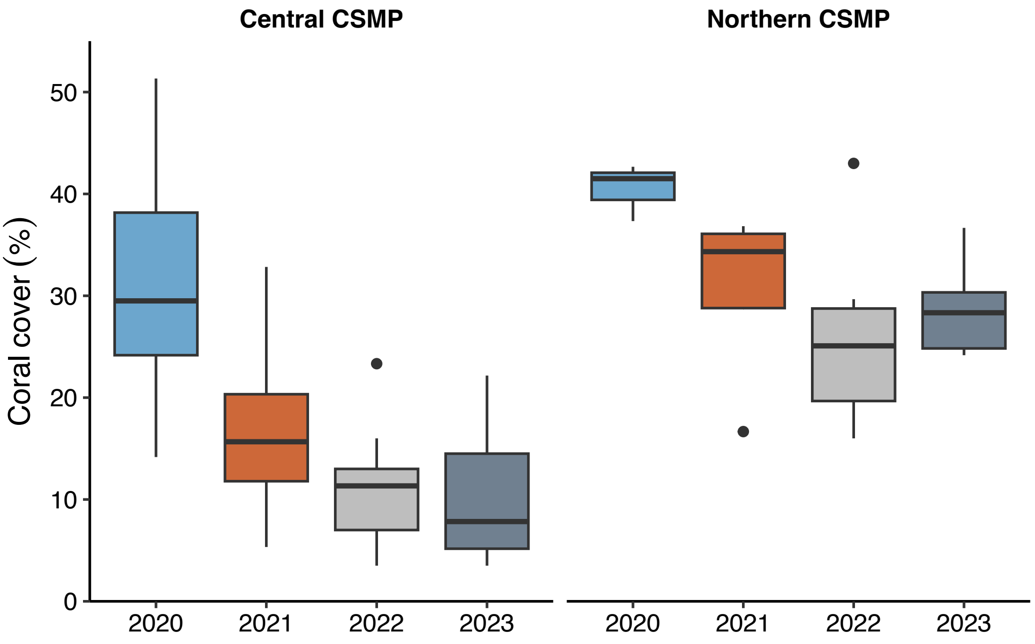


Figure 4.2 Temporal change in coral cover within the two regions of the Coral Sea Marine Park. Data are based on surveys of matching sites in 2020, 2021, 2022, and/or 2023 across 10 reefs (central CSMP: Flinders, Holmes, Lihou, and Moore Reefs, Willis and Diamond Islets, and Herald Cays; northern CSMP: Ashmore, Bougainville and Osprey Reefs).

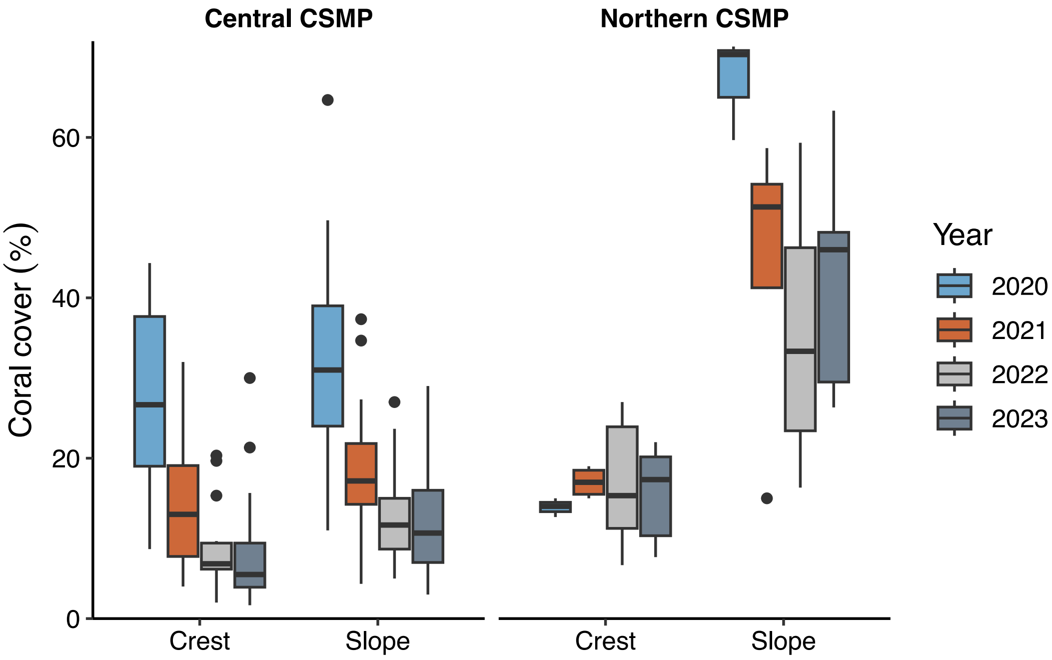


Figure 4.3 Temporal variation in coral cover between shallow reef habitats (reef crest and reef slope) within the two regions of the Coral Sea Marine Park. Data are based on surveys of matching sites in 2020, 2021, 2022, and/or 2023 across 10 reefs (central CSMP: Flinders, Holmes, Lihou, and Moore Reefs, Willis and Diamond Islets, and Herald Cays; northern CSMP: Ashmore, Bougainville and Osprey Reefs) and pooled between habitats (reef slope and reef crest) within each site.

Changes in coral cover varied among reefs and sites within the northern CSMP with coral cover being relatively consistent on Ashmore and Osprey Reefs between 2022 and 2023, and increasing from 23.92% to 31.28% on Bougainville Reef over the same timeframe (Figure 4.4, 4.5). Declines in coral cover in the central CSMP from 2022 to 2023 were relatively consistent (22.3 – 40.1% decline) among most reefs and sites in the central CSMP, the only exception being Holmes Reefs where coral cover increased from 8.83% in 2022 to 13.75% in 2023 (Figure 4.4, 4.5). Some of this increase at Holmes Reefs is likely due to unfavourable weather conditions that were experienced during the 2022 surveys. As such some sites and the reef crest habitat at other sites were not able to be surveyed in 2022, and likely resulted in coral cover being under-estimated for Holmes Reefs in 2022. Notably, coral cover at Moore Reefs, a previously identified 'bright spot' reef that hadn't been resurveyed since 2020, decreased from 39.50% in 2020 to 19.00% in 2023, a decline of 51.9%. Despite this decline in coral cover, the present coral cover at Moore Reefs (19.00%) is the highest of the seven central CSMP surveyed, and almost double that of the average for central CSMP reefs (9.95%).

The cause/s of the declines in coral cover on the central CSMP reefs are difficult to determine, however the relatively widespread nature of the declines suggest that they are unlikely to be related to localised disturbances. Moderate levels of bleaching (Pale - Recently Dead) were recorded across some central CSMP reefs in February 2022, ranging from 4.1% of colonies surveyed at Holmes Reefs to 17.6% at Diamond Islets, with mean of 11.9% across the central CSMP (Hoey et al. 2021). Water temperatures experienced with the central CSMP were again higher than average in 2022, with the greatest heat stress occurring in March 2022 and continuing into April 2022 in some areas (i.e., after the 2022 surveys had been completed; Figure 4.6). Importantly, the central and eastern region of the Queensland Plateau (including Herald Cays, Diamond and Willis Islets, and Lihou and Moore Reefs) were exposed to 8-11 Degree Heating Weeks (DHW) in March 2022 (Figure 4.6), coinciding with the greatest declines in coral cover from our 2023 surveys. DHW combines the intensity and duration of heat stress experienced during the previous 3 months, and is a strong predictor of bleaching with DHW >4 likely to lead to significant bleaching, and DHW>8 likely to lead to significant mortality (Hughes et al. 2018), especially in more thermally sensitive species. While we cannot rule out other potential causes for the recorded declines in coral cover, there was only one tropical cyclone recorded in the CSMP in the period between our 2022 and 2023 surveys (TC Gabrielle) that passed through the central CSMP (approximately 100km to the west of Lihou Reef) as a Category 1-2 system on the 8th-9th February 2023, a few days prior to our surveys. We didn't observe any areas of recently broken and/or dislodged corals that would be consistent with physical damage from severe storms (e.g., Fabricius et al. 2008). Further, the spatial footprint of this storm does not coincide with the recorded declines in coral cover from our surveys. It appears most likely that the observed declines in coral cover on central CSMP reefs in 2023 were related to the heat stress experienced in March 2022.

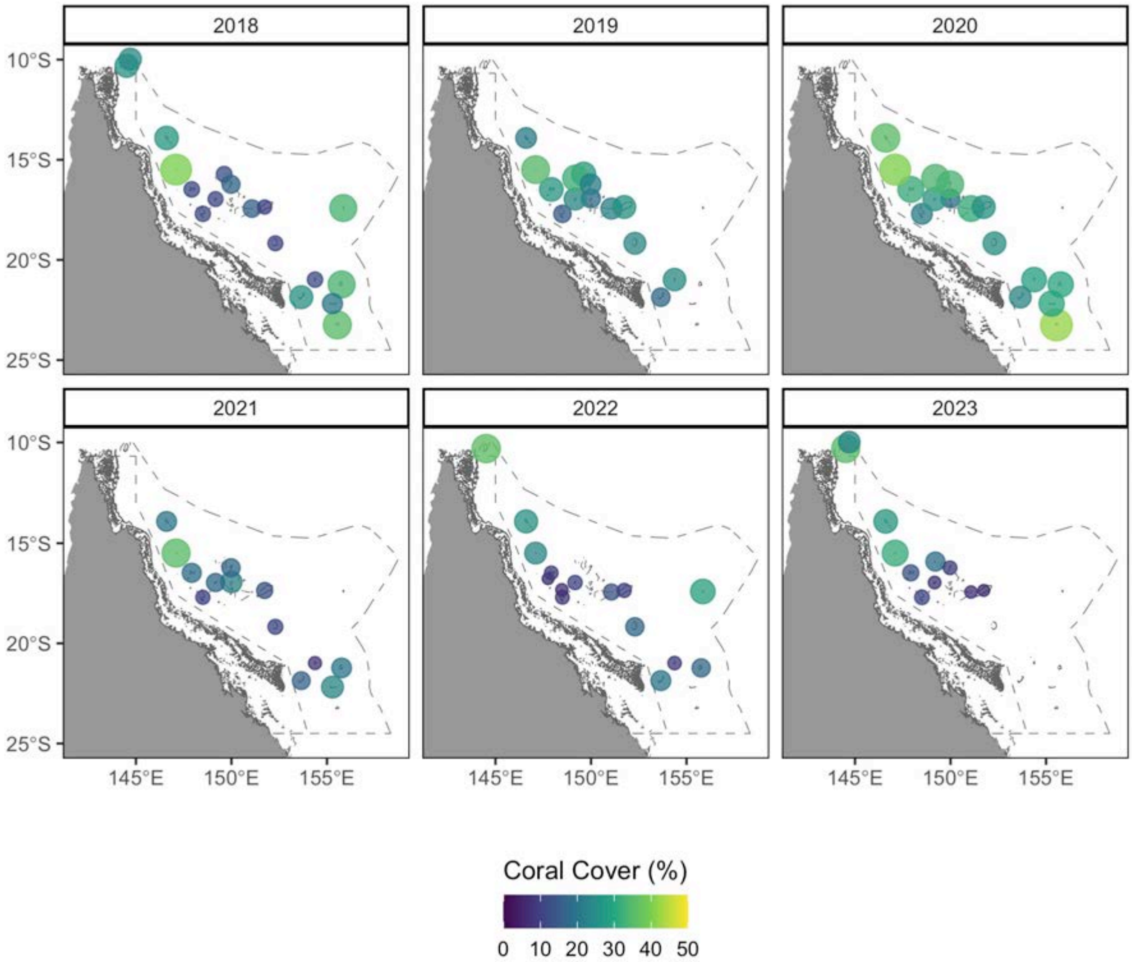
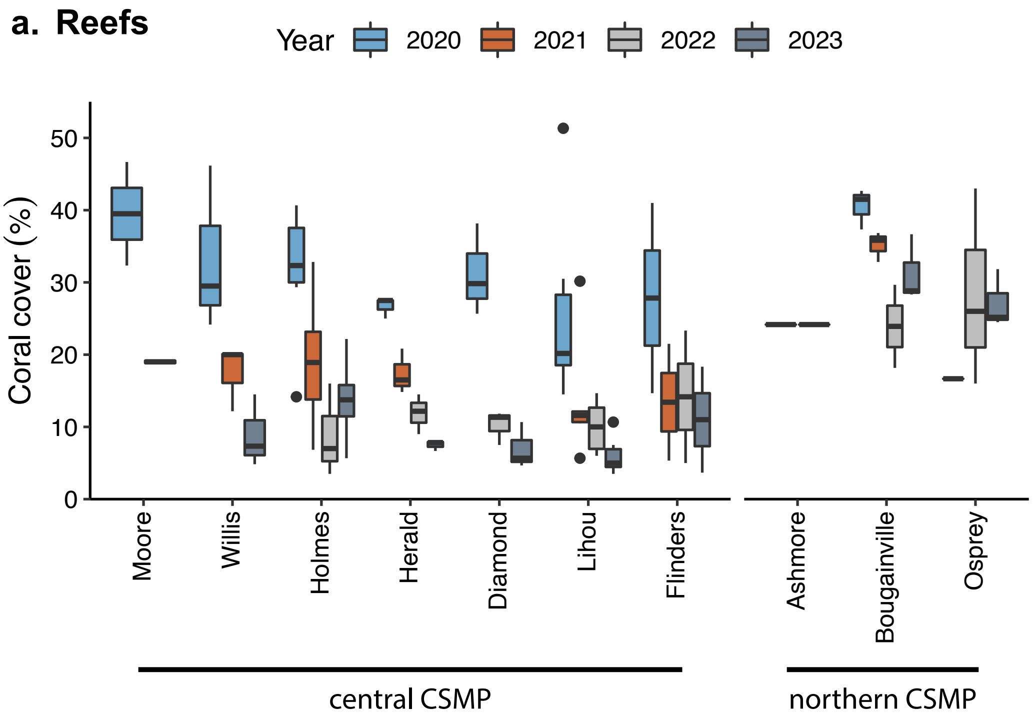
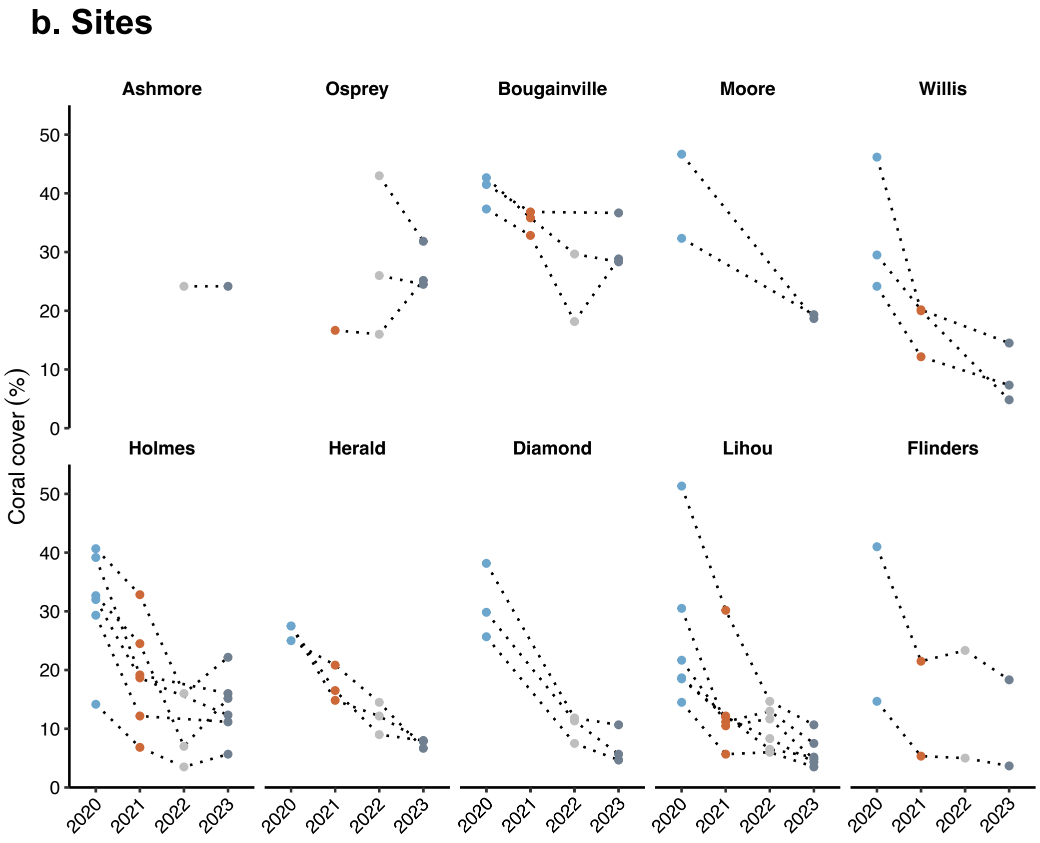


Figure 4.4 Spatial and temporal (2018-2023) variation in coral cover on shallow reef habitats (reef crest and reef slope) across 22 reef systems in the Coral Sea Marine Park. The size of individual points is proportional to the cover of live coral at each reef.



a. Reefs



b. Sites

Figure 4.5 Temporal variation in coral cover among **(a)** ten reefs, and **(b)** 32 sites in the Coral Sea Marine Park that were surveyed in 2020, 2021, 2022, and/or 2023. Data are based on surveys of matching sites in each year and pooled between habitats (reef slope and reef crest) within each site.

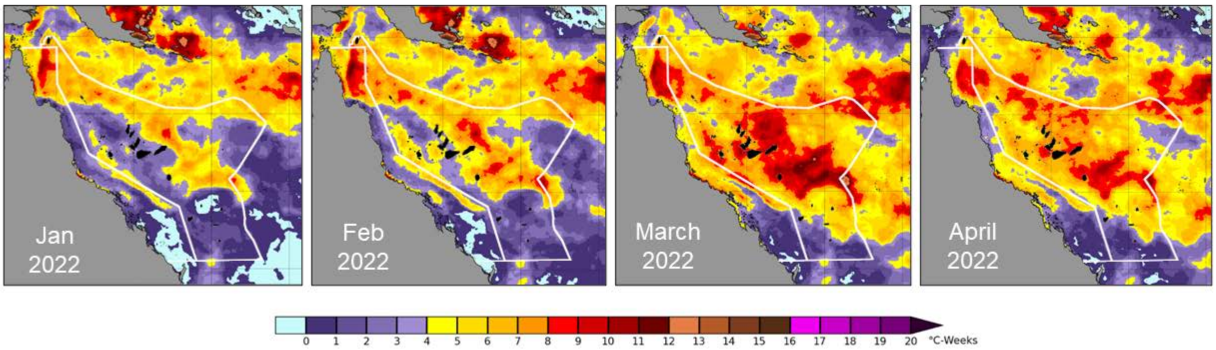


Figure 4.6 Progression of heat stress experienced throughout the Coral Sea Marine Park from January to April 2022. Colours represent the maximum Degree Heating Weeks (DHW). DHW combines the intensity and duration of heat stress experienced during the previous 3 months into one single number. It is a strong predictor of bleaching with DHW >4 likely to lead to significant bleaching, and DHW>8 likely to lead to significant mortality (Hughes et al. 2018), especially in more thermally sensitive species.

Coral richness – In contrast to coral cover, coral richness declined in both the central and northern CSMP from 2022 to 2023 (Figures 4.7, 4.8). Average coral richness declined from 16.2 to 14.1 taxa per site on central CSMP reefs, and from 20.3 to 17.6 taxa per site on northern CSMP reefs from 2022 to 2023 (Figure 4.7). These declines were generally consistent among reefs in each region, the only exception being Holmes Reefs where average coral richness increased from 12.3 to 15.3 taxa per site from 2022 to 2023 (Figure 4.8). This apparent increase in richness is likely related to some sites and zones on Holmes Reef not being surveyed in 2022 (as discussed previously).

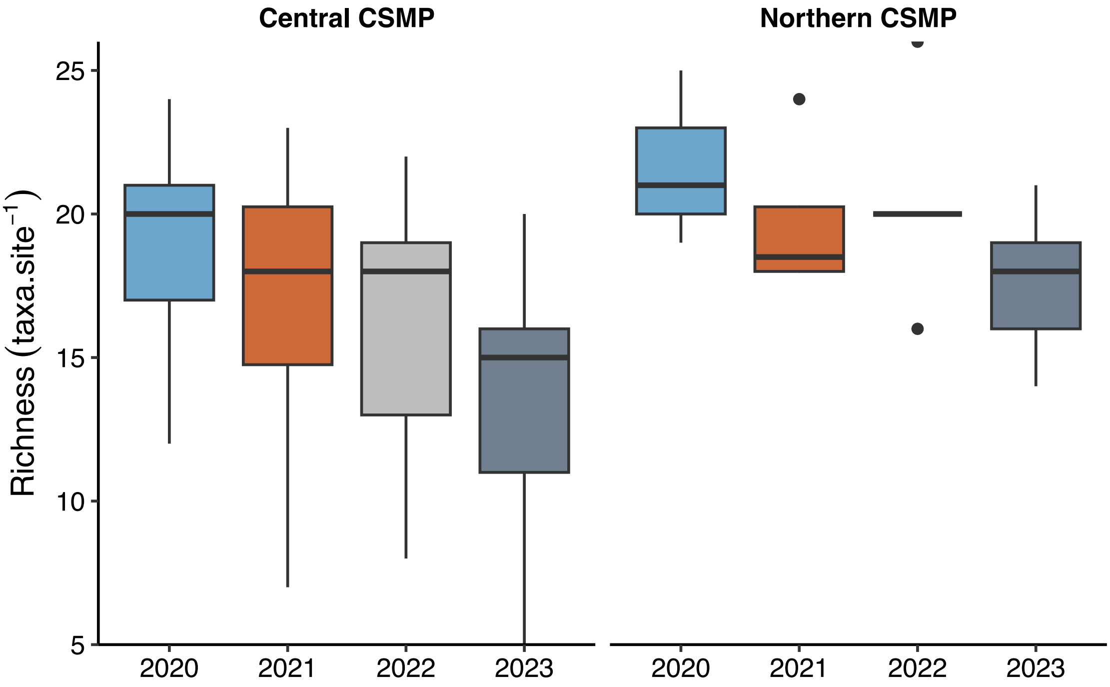


Figure 4.7 Temporal change in coral richness among the two regions in the Coral Sea Marine Park from 2020 to 2023. Data are based on the number of coral taxa recorded at each of 32 sites (i.e., pooled across slope and crest habitats).

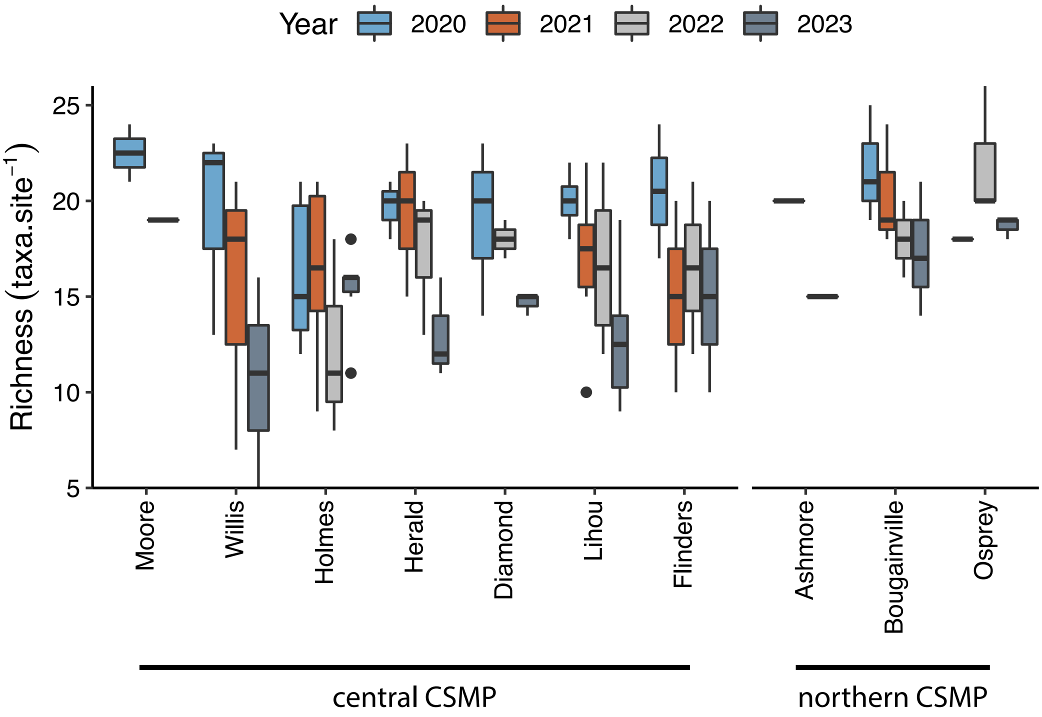


Figure 4.8 Temporal change in coral richness among ten reefs in the Coral Sea Marine Park from 2020 to 2023. Data are based on the number of coral taxa recorded at each of 32 sites (i.e., pooled across slope and crest habitats). One to six sites were surveyed at each reef.

4.1.3 Coral composition

The greatest variation in the composition of shallow water coral assemblages was between the two CSMP regions, that were largely separated along the first dimension of the nMDS plot (Figure 4.9a). In general, reefs in the northern CSMP were characterised by a relatively higher cover of *Echinopora, Platygyra* and branching *Porites,* while reefs in the central CSMP were characterised by a higher cover of *Coscinarea, Stylophora and Astrea*, and, to a lesser degree, tabular *Acropora* (Figure 4.9a,b).

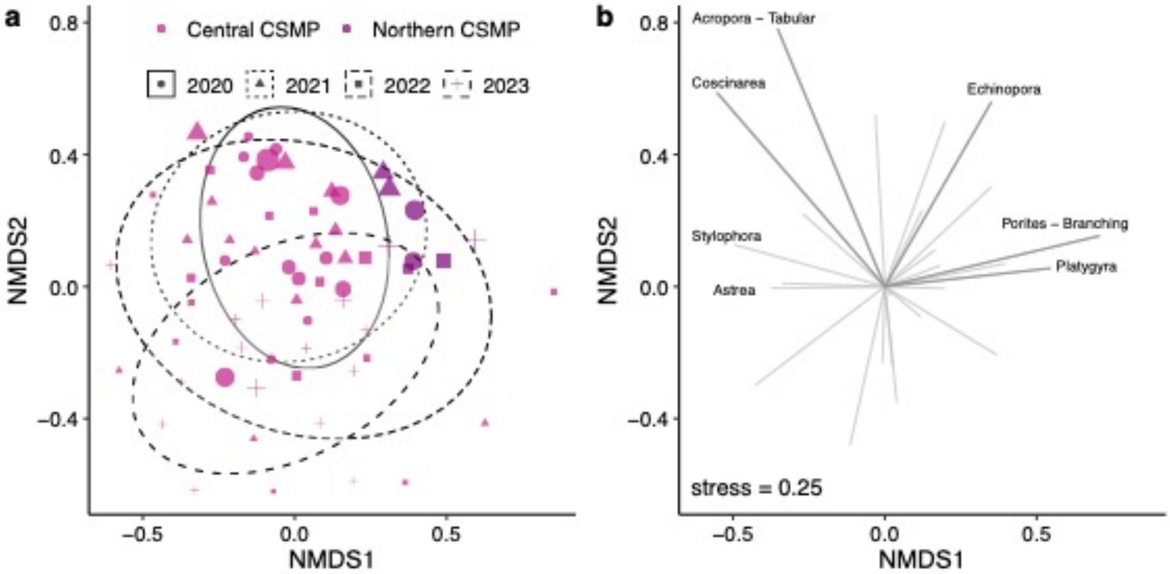


Figure 4.9 Regional and temporal (2020-2023) variation in the composition of shallow water coral assemblages within the Coral Sea Marine Park. Non-metric multidimensional scaling (nMDS) plot showing the variation in coral composition among years for the two regions of the Coral Sea Marine Park. Analyses are based on data from 16 sites that were surveyed every year of the past four years. The size of individual points is proportional to the cover of live coral at each site. 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.

There was also evidence of shifts in the composition of shallow water coral assemblages among years along the second dimension on the nMDS (Figure 4.9). The composition of coral assemblages has been particularly pronounced in the central CSMP, where coral assemblages shifted from being dominated by bleaching sensitive coral taxa, such as *Seriatopora,* tabular *Acropora*, and

Coscinarea in 2020, to a mix of largely bleaching resistant taxa (including branching *Porites* and *Coeloseris*) in 2022 and 2023 (Figure 4.10c,d). There was also evidence of a temporal change in the variability of coral assemblages among reefs within the central CSMP, with the composition of coral assemblages showing greater dissimilarity among sites in 2021-2023 (i.e., less clustered in the nMDS space) compared to 2020. This increasing variability in the composition of coral assemblages, likely reflects the differential exposure to, and impacts of, the 2020, 2021 and likely 2022 bleaching events on individual reefs (Figures 4.9, 4.10c,d; Hoey et al. 2021, 2022, this report). There was also some evidence of temporal changes in the composition of coral assemblages in the northern CSMP, with coral assemblages shifting after the 2020 bleaching, before returning to a pre-bleaching configuration (i.e., 2020) in 2023 (Figure 4.10a,b).

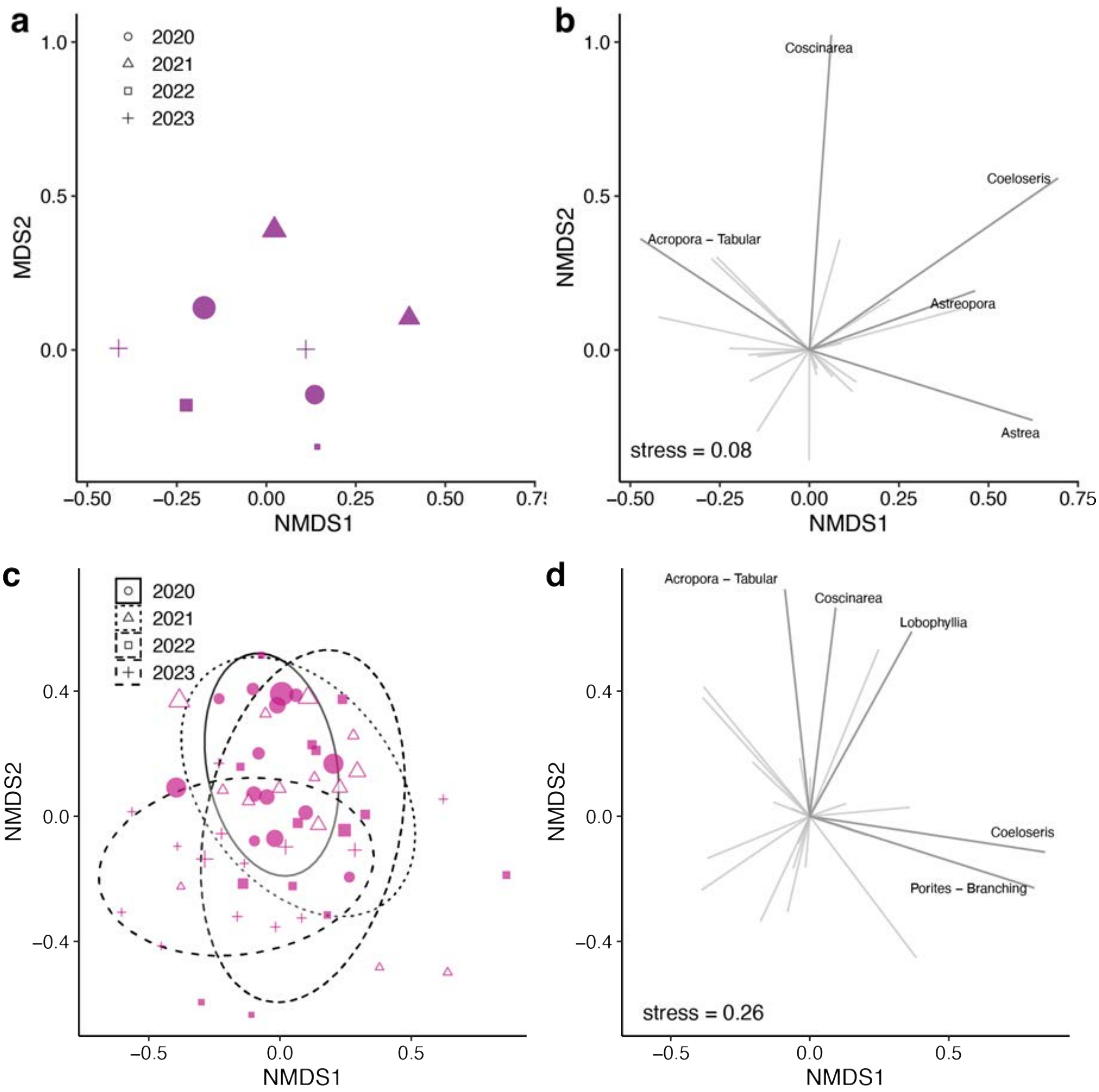
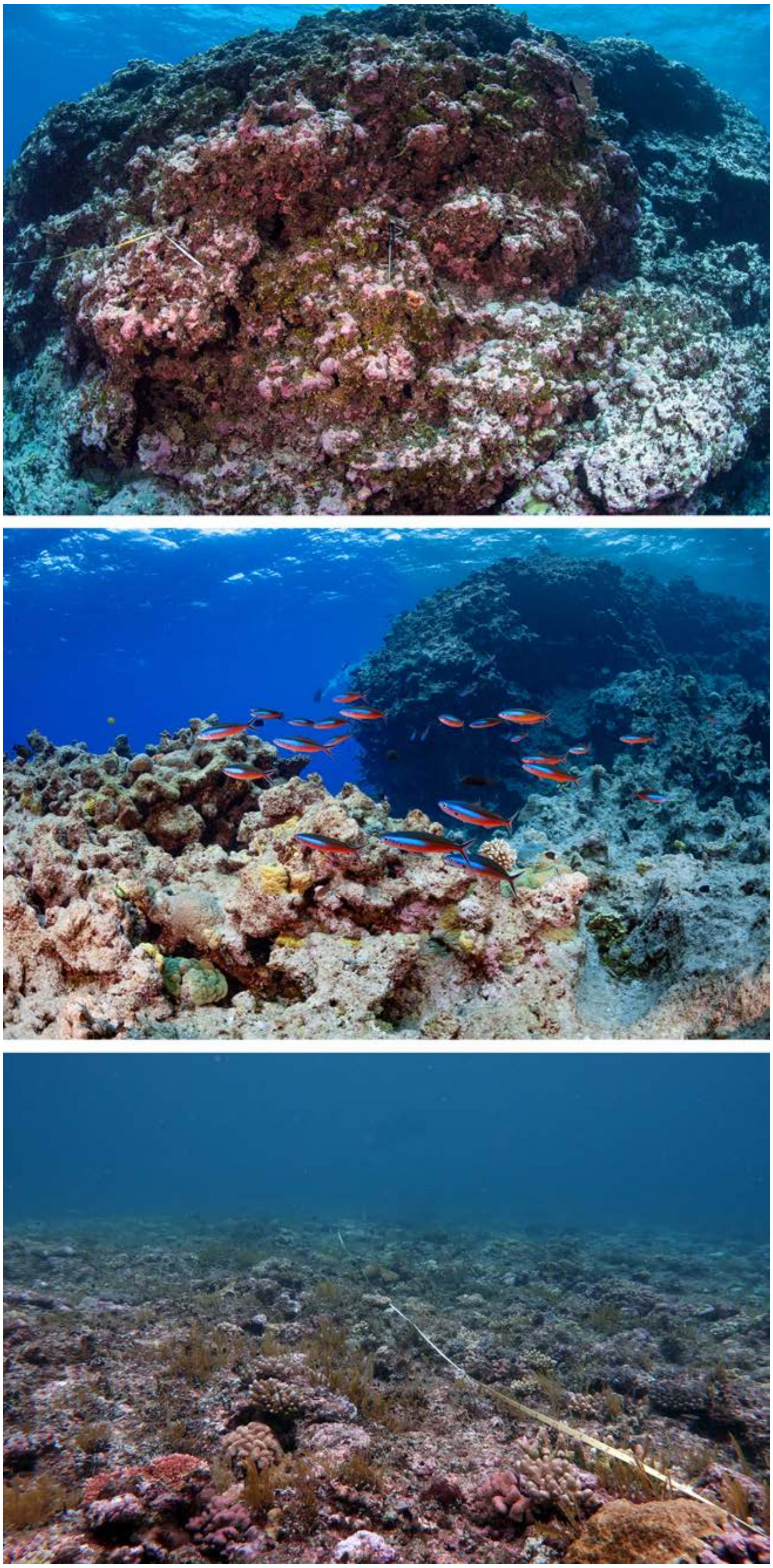


Figure 4.10 Non-metric multidimensional scaling (nMDS) plots showing the temporal variation (2020-2023) in shallow water coral composition among reefs in the **(a,b)** northern, and **(c,d)** central Coral Sea Marine Park. Analyses are based on data from 16 sites that were surveyed consistently in the past four years (northern: 2 sites; central: 14 sites). The size of individual points is proportional to the cover of live coral at each site. 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.

Figure 4.11 Photographs of diverse coral assemblages at Moore Reefs (top; February 2023), Beva Reef (middle; March 2023) and Ashmore Reef (bottom; March 2023) within the Coral Sea Marine Park. Note the prostrate morphologies of coral on the exposed face of Ashmore Reef in the bottom image. Image credits: Victor Huertas



Photographs of low coral cover habitats at Willis Islets (top; February 2023), Holmes Reefs (middle; February 2023) and Boot Reef (bottom; March 2023) within the Coral Sea Marine Park. Note the coral cover on the exposed sites at Boot Reef was markedly higher on the reef slope, than the reef crest shown here. Image credits: Victor Huertas

4.2 Algal assemblages

Macroalgae - Following the mortality of corals, other benthic taxa (in particular algae) increase in abundance as they rapidly colonise the dead coral skeletons (Diaz-Pulido and McCook 2002). If these changes in benthic composition persist, they can inhibit the recovery of coral populations, and the associated assemblages and ecosystem goods and services they provide (Moberg and Folke 1999; Pratchet et al. 2014; Woodhead et al. 2019). Notably, shifts from coral- to macroalgal-dominance have been documented following large-scale coral mortality (e.g., Jamaica: Hughes 1994; Great Barrier Reef: Cheal et al. 2010; Seychelles: Graham et al. 2015). Once established, these macroalgal-dominated areas tend to persist due to a series of positive, or reinforcing, feedbacks that promote macroalgal assemblages and limit the recovery of coral populations (e.g., Hoey and Bellwood 2011; Van de Leemput et al. 2016; Johns et al. 2018). Predicted increases in the frequency and intensity of temperature-induced bleaching events and severe tropical storms under ongoing and future climate change has led to concerns that an increasing number of reefs may be overgrown by macroalgae (Hughes et al. 2017, 2018; Souter et al. 2021).

The cover of macroalgae across the eleven CSMP reefs surveyed in 2023 was generally low, with total macroalgal cover averaging 8.14%. Macroalgal cover was 5.5-fold greater on reefs in the central CSMP (13.22%) than the northern CSMP (2.37%). Macroalgal cover also varied among reefs within each region, ranging from 0.0% to 3.03% on Bougainville and Ashmore Reefs, respectively, in the northern CSMP, and from 6.28% to 22.28% on Moore Reefs and Herald Cays, respectively, in the central CSMP (Figure 4.12).

Comparisons of macroalgal cover in shallow reef habitats across the ten reefs that were surveyed in 2020, 2021, 2022, and/or 2023 revealed a marked difference in the temporal patterns in macroalgal cover among the two CSMP regions (Figure 4.12). While macroalgal cover has remained low and relatively stable on reefs in the northern CSMP from 2020-2023 (0.22 – 2.58%), total macroalgal cover declined in the central CSMP from 8.44% (2020) to 6.54% (2022), before increasing to 13.73% in 2023 (an increase of 110%; Figure 4.12).

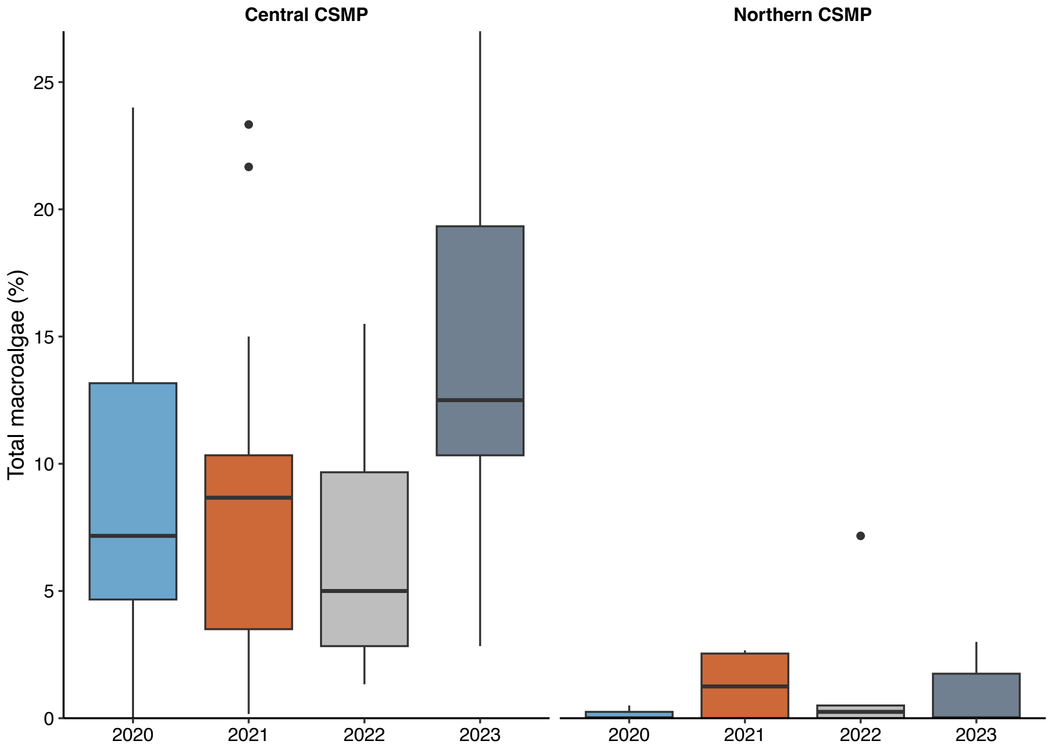


Figure 4.12 Temporal change in total macroalgae cover within the two regions of the Coral Sea Marine Park. Data are based on surveys of matching sites in 2020, 2021, 2022, and/or 2023 across 10 reefs (central CSMP: Flinders, Holmes, Lihou, and Moore Reefs, Willis and Diamond Islets, and Herald Cays; northern CSMP: Ashmore, Bougainville and Osprey Reefs) and pooled between habitats (reef slope and reef crest) within each site.

The increase in total macroalgal cover in shallow reef habitats of the central CSMP was largely driven by an increase in cover of the green calcified alga *Halimeda* spp. (Figure 4.13). *Halimeda* spp. accounted for >90% of all macroalgae recorded across all sites in 2023, and the cover of *Halimeda* spp. almost doubled on reefs in the central CSMP from 2022 (6.26%) to 2023 (12.38%; Figure 4.13). The cover of *Halimeda* increased on most reefs in the central CSMP from 2022 to 2023, the only exception being Holmes Reefs where *Halimeda* cover remained stable (Figure 4.14). *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. The cover of other macroalgae was extremely low across the northern and central CSMP regions in 2023 (0.0% and 1.35%, respectively; Figure 4.13).

There was, however, some variation in the cover of ‘other’ macroalgae among reefs in the central CSMP, ranging from 0.06% at Herald Cays and Holmes Reefs to 3.50% at Flinders Reef (primarily *Caulerpa*) and 4.06% (primarily *Rhipiliopsis*) at Lihou Reef (Figure 4.14).

Caulerpa and *Rhipiliopsis* are both green algae (Chlorophyta). *Caulerpa* has a creeping habit and can quickly grow to occupy areas free of other benthic taxa (i.e., hard corals, soft corals, sponges), while *Rhipiliopsis* has flattened fan-like blades that can form clumps but rarely exceed 1cm in height (Littler and Littler 2003). The cause of the increases in *Caulerpa* at Flinders Reef and *Rhipiliopsis* at Lihou Reef are unknown, and although not currently a concern, should be monitored to detect any further increases. Despite the declines in coral cover on shallow reef habitats throughout the central and northern CSMP from 2020 to 2023 (Figure 4.2), the cover of fleshy macroalgae (excl *Halimeda*) has remained low throughout the CSMP (1.05%), and considerably lower than other oceanic reefs, such as Elizabeth and Middleton Reefs, and Lord Howe Island to the south (Hoey et al. 2011, 2018).

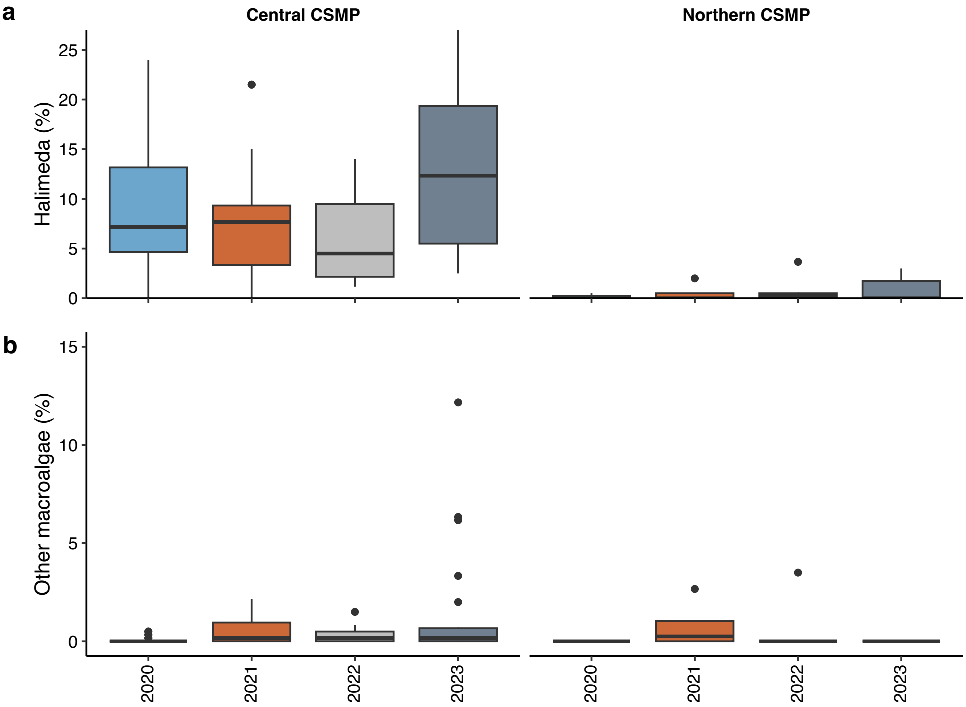


Figure 4.13 Temporal variation in the cover of **(a)** *Halimeda* spp. and **(b)** ‘other’ macroalgae within the two regions of the Coral Sea Marine Park. Data are based on surveys of matching sites in 2020, 2021, 2022, and/or 2023 across 10 reefs (central CSMP: Flinders, Holmes, Lihou, and Moore Reefs, Willis and Diamond Islets, and Herald Cays; northern CSMP: Ashmore, Bougainville and Osprey Reefs) and pooled between habitats (reef slope and reef crest) within each site.

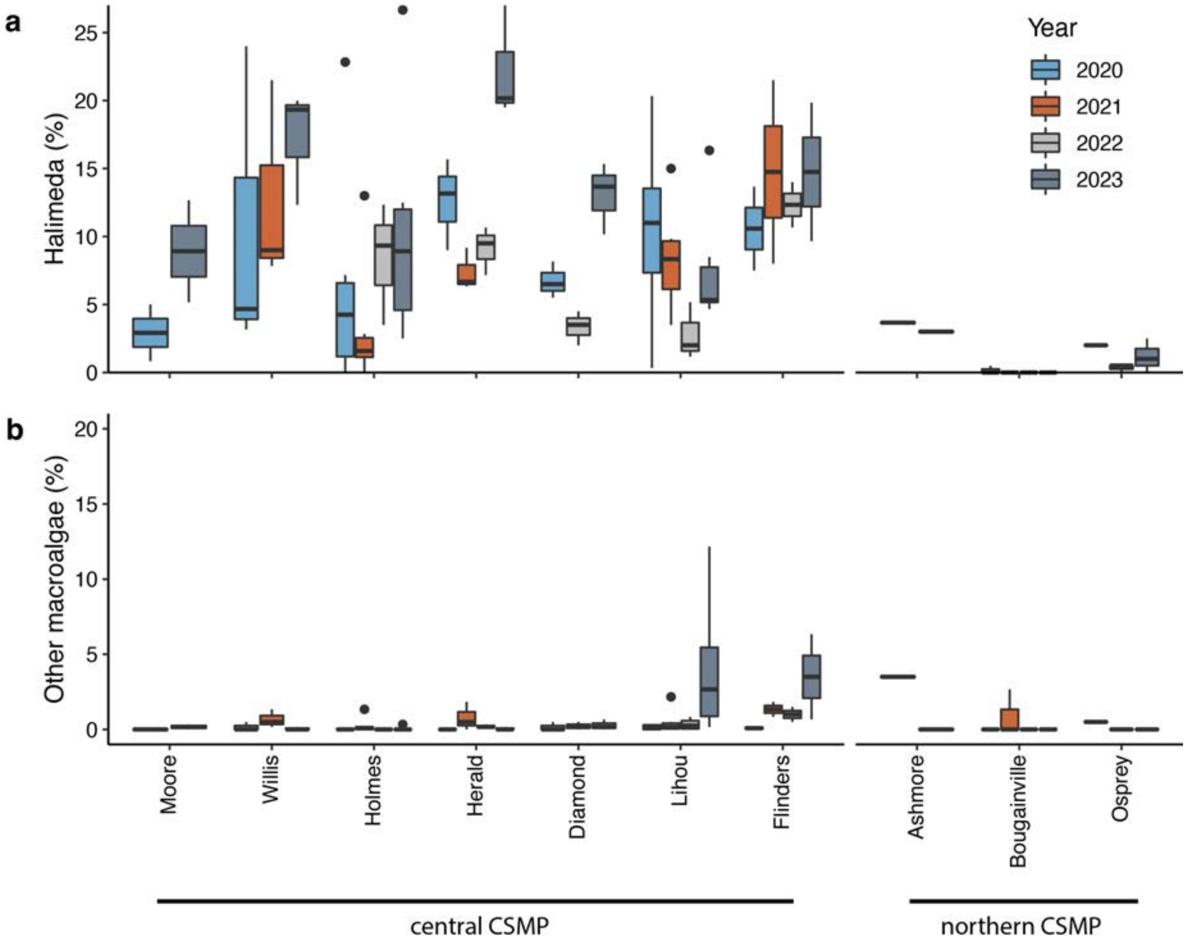


Figure 4.14 Temporal variation in the cover of **(a)** *Halimeda* spp. and **(b)** 'other' macroalgae among ten reefs in the Coral Sea Marine Park that were surveyed at least twice in the past 4 years (i.e., 2020-2023). Data are based on surveys of matching sites in 2020, 2021, 2022, and/or 2023 across 10 reefs (central CSMP: Flinders, Holmes, Lihou, and Moore Reefs, Willis and Diamond Islets, and Herald Cays; northern CSMP: Ashmore, Bougainville and Osprey Reefs) and pooled between habitats (reef slope and reef crest) within each site.

Crustose coralline algae (CCA) – The average cover of crustose coralline algae recorded across the 48 CSMP sites surveyed in 2023 was 27.58% (±1.23 SE), ranging from 13.85% (±1.08 SE) at Boot Reef in the northern CSMP up to 51.72% (±2.84 SE) at Diamond Islets in the central CSMP (Figure 4.15). Average CCA cover was generally greater on reefs in the central CSMP reefs (33.29 ± 1.81%) compared to the northern CSMP (averaging 21.12 ± 1.82%), although there was considerable variation among reefs within each region. Average CCA cover varied from 19.67% (Moore Reefs) to 51.72% (Diamond Islets) in the central CSMP, and from 13.85% (Boot Reef) to 35.56% (Osprey Reef) in the northern CSMP (Figure 4.15).

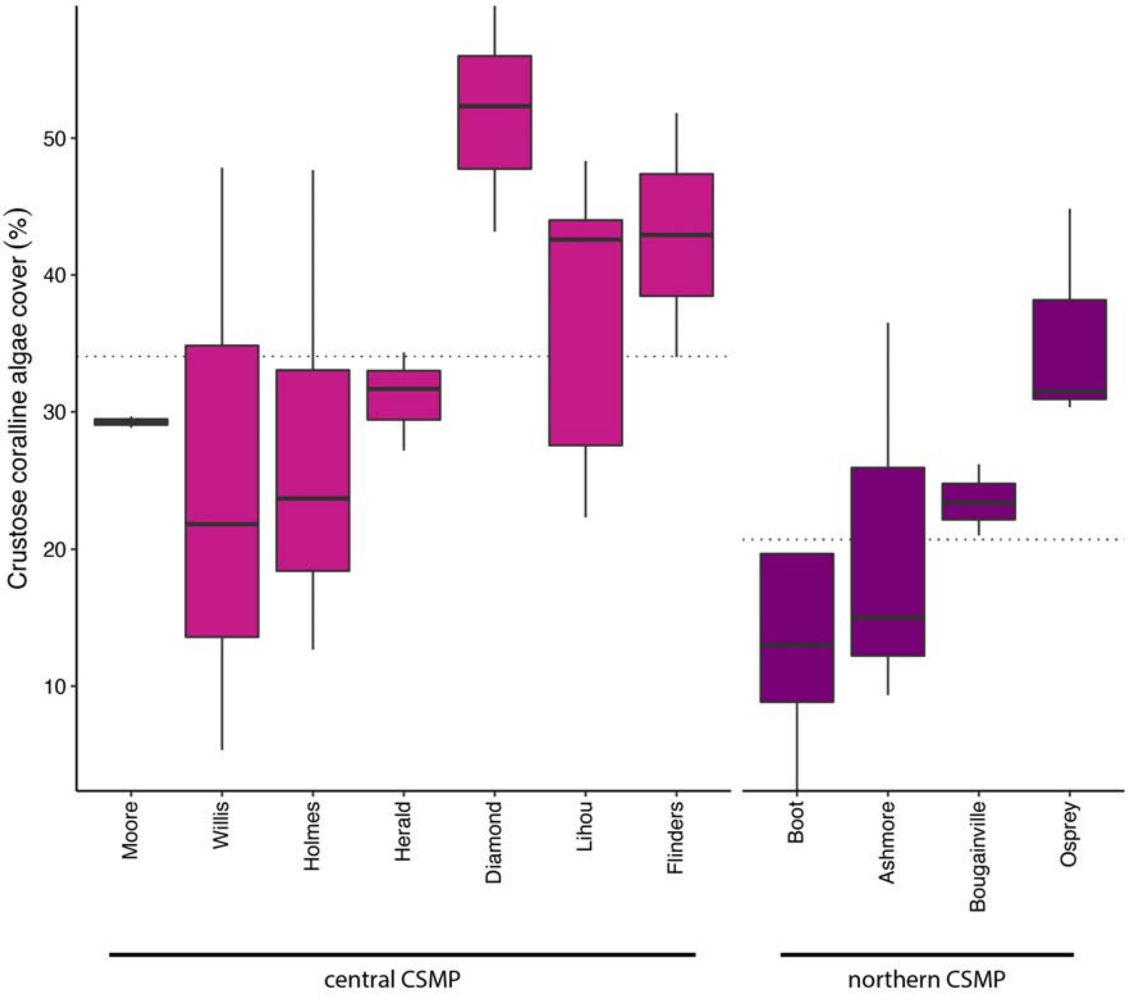


Figure 4.15 Variation in coral cover and coral richness among 11 reefs in the Coral Sea Marine Park (CSMP) in 2023. Data are based on the 50m point-intercept transects, with data for richness based on the number of coral taxa recorded at each of the 48 sites (i.e., pooled across transects and slope and crest habitats). Reefs are arranged into the central and northern CSMP and coloured by *a priori* regional assignments (following Figure 3.1). Dotted lines represent regional averages.

Comparisons of CCA cover in shallow reef habitats across the ten reefs that were surveyed in 2020, 2021, 2022, and/or 2023 revealed similar temporal patterns in the CCA cover in both the central and northern CSMP. CCA cover increased in both the central and northern CSMP from 2020 to 2022 (Figure 4.16), coinciding with declines in coral cover over the same period in both these regions (Figure 4.2). For example, CCA cover in the central CSMP increased from 28.15% to 41.77% from 2020 to 2022 (an absolute increase of 13.62%), which largely offset the decline in coral cover in the central CSMP over the same period (2020: 28.89%, 2022: 10.05%; an absolute decline of 18.84%). It appears therefore that

CCA's, rather than turf algae or macroalgae, are able rapidly colonise and/or grow to cover recently dead coral skeletons. This is important as CCA's are generally viewed as a critical component of healthy coral reef ecosystems, contributing to reef calcification, cementing and infilling (e.g., Teichert et al. 2020; Cornwall et al. 2023), inducing the settlement of coral larvae (e.g., Harrington et al. 2004; Abdul Wahab et al. 2023), potentially the provision of 3-dimensional structure for reef associated species (Hoey et al. 2022), and the resilience of the system as a whole.

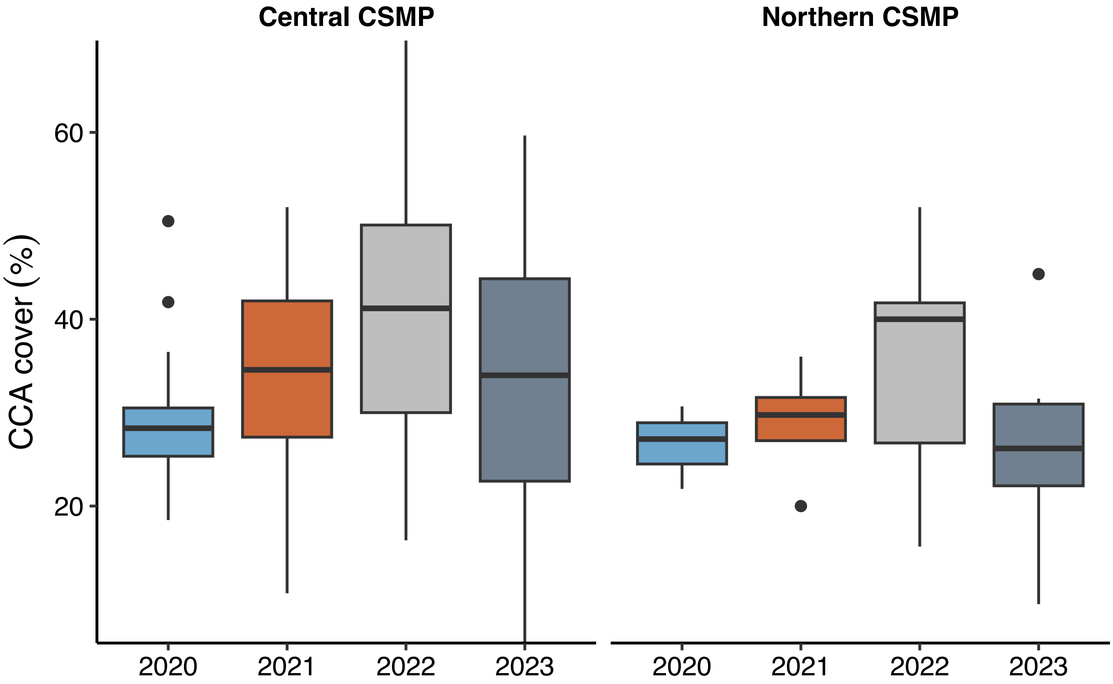
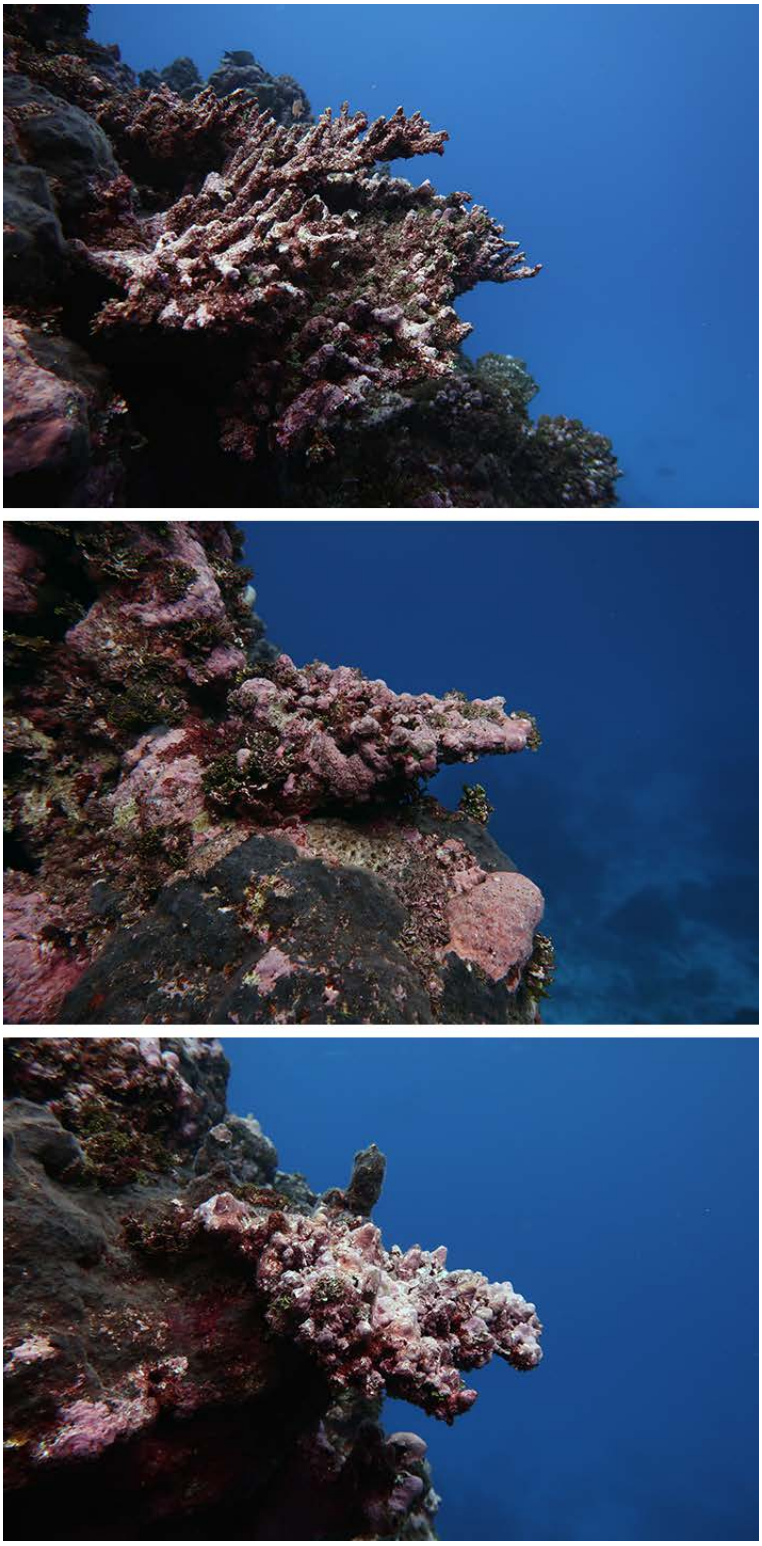


Figure 4.16 Temporal change in the cover of Crustose Coralline Algae (CCA) within the two regions of the Coral Sea Marine Park. Data are based on surveys of matching sites in 2020, 2021, 2022, and/or 2023 across 10 reefs (central CSMP: Flinders, Holmes, Lihou, and Moore Reefs, Willis and Diamond Islets, and Herald Cays; northern CSMP: Ashmore, Bougainville and Osprey Reefs) and pooled between habitats (reef slope and reef crest) within each site.

The cover of CCA declined in both regions of the CSMP from 2022 to 2023; from 41.77% to 34.63% in the central CSMP, and 35.39% to 26.67% in the northern CSMP (Figure 4.16). The decline in CCA cover within the central CSMP was largely driven by declines at two reefs (Diamond Islets and Herald Cays), while

CCA cover was relatively stable at the other five central CSMP reefs (Figures 4.17, 4.18). The decline in CCA cover from 2022 to 2023 was relatively consistent across the three northern CSMP reefs (Figures 4.17, 4.18).



Images showing dead branching and tabular coral skeletons at Willis Islets colonised by crustose coralline algae. Image credits: Andrew Hoey

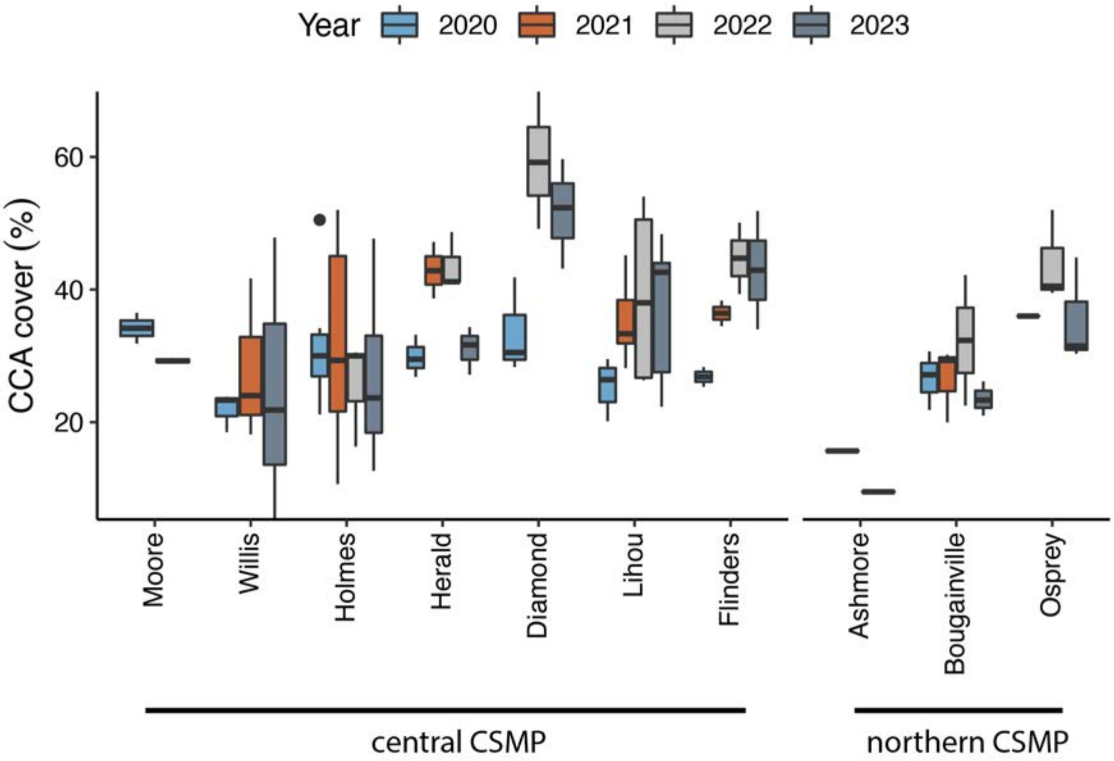


Figure 4.17 Temporal variation in the cover of Crustose Coralline Algae (CCA) among ten reefs in the Coral Sea Marine Park that were surveyed at least twice in the past 4 years (i.e., 2020-2023). Data are based on surveys of 32 matching sites in 2020, 2021, 2022, and/or 2023 across 10 reefs (central CSMP: Flinders, Holmes, Lihou, and Moore Reefs, Willis and Diamond Islets, and Herald Cays; northern CSMP: Ashmore, Bougainville and Osprey Reefs) and pooled between habitats (reef slope and reef crest) within each site.

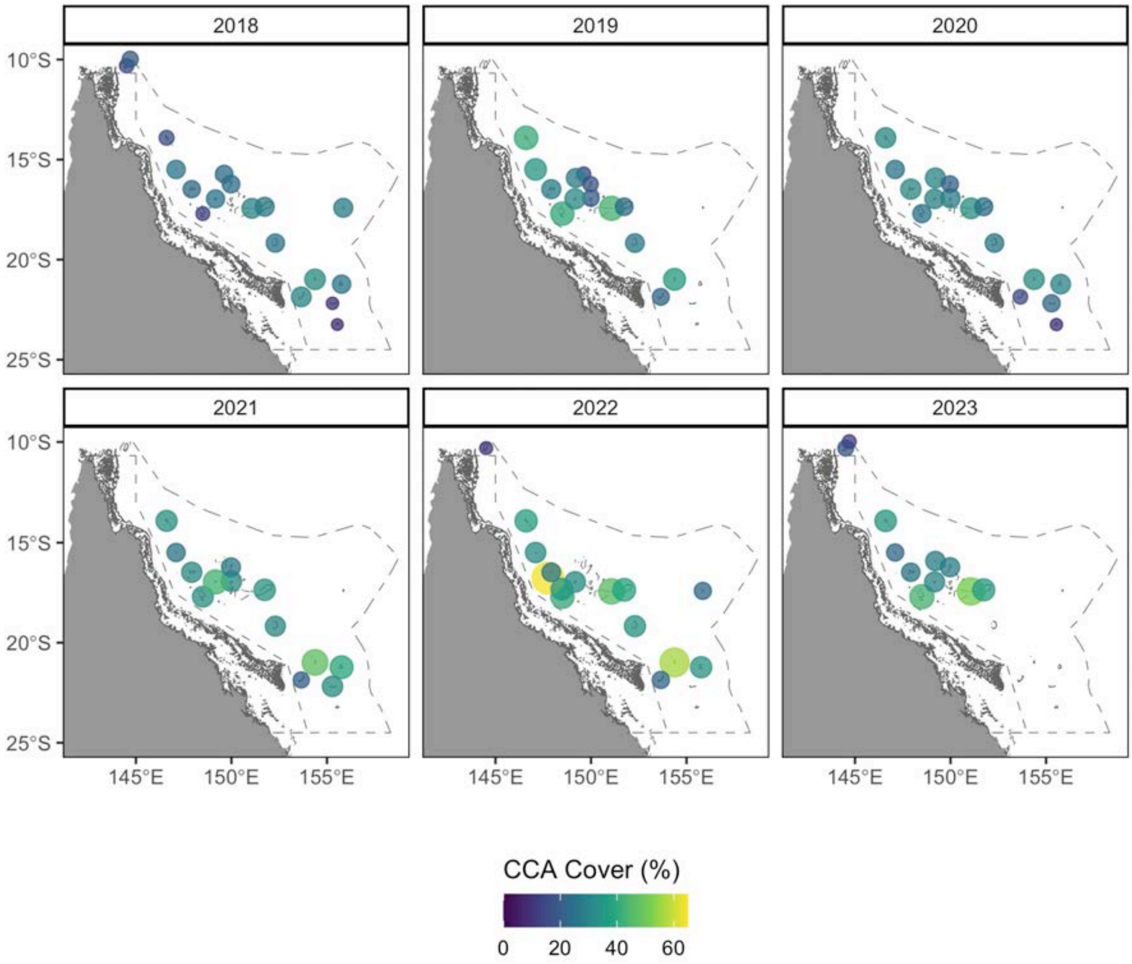


Figure 4.18 Spatial and temporal (2018-2023) variation in crustose coralline algae (CCA) cover on shallow reef habitats (reef crest and reef slope) across 22 reef systems in the Coral Sea Marine Park. The size of individual points is proportional to CCA cover at each reef.

4.3 Coral Reef Fishes

Reductions in coral cover and shifts in the composition of coral assemblages as has been experienced across the CSMP following the 2020, 2021, and now 2022 bleaching events (Figures 4.2, 4.9), often lead to reductions in the structural complexity of reef habitats and the associated reef fish and invertebrate assemblages (e.g., Wilson et al. 2006; Stella et al. 2011; Hoey et al. 2016; Robinson et al. 2019). The greatest and most immediate effects on bleaching-induced coral mortality are on fishes that rely on these corals for food (i.e., corallivores) or shelter (e.g., Pratchett et al. 2008; Hoey et al. 2016), and have been shown to lead to shifts in the composition of fish assemblages from coral specialists to habitat and/or diet generalists (e.g., Bellwood et al. 2006a, 2012; Richardson et al. 2018). In particular, the loss of fast-growing, and thermally sensitive tabular and staghorn *Acropora* (Burn et al. 2023) have been shown to reduce the three-dimensional structure and functionality of reef habitats (Hughes et al. 2018; McWilliam et al. 2020). While reductions in live coral cover may have immediate effects on species that rely on live corals as adults, it has been estimated that approximately 75% of reef fish species use live coral at some stage during their life cycle (e.g., as a settlement or juvenile habitat; Coker et al. 2014). Any effects of coral loss on these species may take several years to be realised (e.g., Graham, et al. 2007).

4.3.1 Richness, density and biomass of reef fishes

A total of 72,371 fishes were recorded across the 48 sites surveyed across 11 reefs in 2022. Eleven fish species that had not been recorded during surveys or observations of shallow reef habitats of the CSMP on the previous voyages (2018-2022) were recorded during the 2023 surveys. All of these eleven species (*Chromis richardsoni, Cirrhilabrus* sp., *Epinephelus spilotoceps, Lutjanus biguttatus, Naso lopezi, N. minor, N. thynnoides, Pentapodus aureofasciatus, Pycnochromis lineatus, Scarus festivus,* and *Sphyraena qenie*) were recorded from Ashmore and Boot Reefs in the far north of the CSMP. Three of these species (*N. lopezi, N. minor,* and *P. aureofasciatus*) have been previously recorded in deeper (>40m) reef habitats in the CSMP using baited remote underwater video systems (BRUVs; Galbraith et al. 2022), and the remaining eight species were new records for the CSMP, taking the total fish species recorded in the CSMP during the past six years of surveys to 650 species (Appendix 5).

The richness, density, and biomass of reef fishes and sharks were generally lower on reefs in the central CSMP than those in the northern CSMP, although there was considerable variation among reefs in each region (Figure 4.19). Regional species richness of reef fishes ranged from an average of 75 species per site in the central CSMP to 90 species per site in the northern CSMP, and from 59 species (Diamond Islets) to 100 species per site (Osprey Reef) among individual reefs. In the central CSMP average species richness varied from 59 species per site at Diamond Islets to 87-89 species per site at Holmes and Moore Reefs (Figure 4.19a). There was less variation in fish species richness among reefs in the northern CSMP, varying from 87 to 100 species per site at Ashmore and Osprey Reefs, respectively. As noted previously (Hoey et al. 2020, 2021), the higher species richness of corals and reef fishes in the northern CSMP (Figures 4.1b, 4.19a) is consistent with well-known latitudinal gradients in the diversity of marine species (Hillebrand 2004; Bellwood and Hughes 2001).

In 2023 regional averages in fish densities were approximately 2-fold higher in the northern CSMP (219.0 individuals per 100 m^2) compared to the central (114.9 individuals per 100 m^2) (Figure 4.19b). Like fish species richness there was considerable variation in the density of reef fish recorded among reefs, especially in the central CSMP where the mean density of reef fish (including sharks) varied 4-fold among reefs, from 56.3 individuals per 100 m^2 at Diamond Islets to 223.0 individuals per 100 m^2 at Moore Reefs (Figure 4.14b). Mean density of reef fish was less variable among reefs in the northern CSMP, ranging from 192.3 to 275.3 individuals per 100 m^2 at Boot and Osprey Reefs, respectively (Figure 4.14b).

Regional patterns in reef fish biomass were similar in direction but greater in magnitude to those of fish species richness and density. The mean reef fish biomass recorded on reefs in the northern CSMP (27.1 kg per 100m^2) was 3-fold greater than the biomass recorded on reefs in central CSMP (9.7 kg per 100m^2, respectively; Figure 4.19c). Reef fish biomass varied widely (~12-fold) among reefs, ranging from 3.1 kg per 100 m^2 at Diamond Islets in the central CSMP to 36.5 kg per 100 m^2 at Boot Reef in the northern CSMP (Figure 4.19c). In the central CSMP reef fish biomass was greatest at Willis Islets and Moore Reefs (13.2-15.1 kg per 100 m^2), while in the northern CSMP was greatest at Boot and Osprey Reefs (32.2-36.5 kg per 100 m^2) and lowest at Ashmore Reef (23.2 kg per 100 m^2). It is important to note that Moore Reefs, that hadn't been surveyed since 2020, was again a standout among reefs in the central CSMP, supporting the highest richness, density and biomass of reef fishes in that region (Figure 4.19).

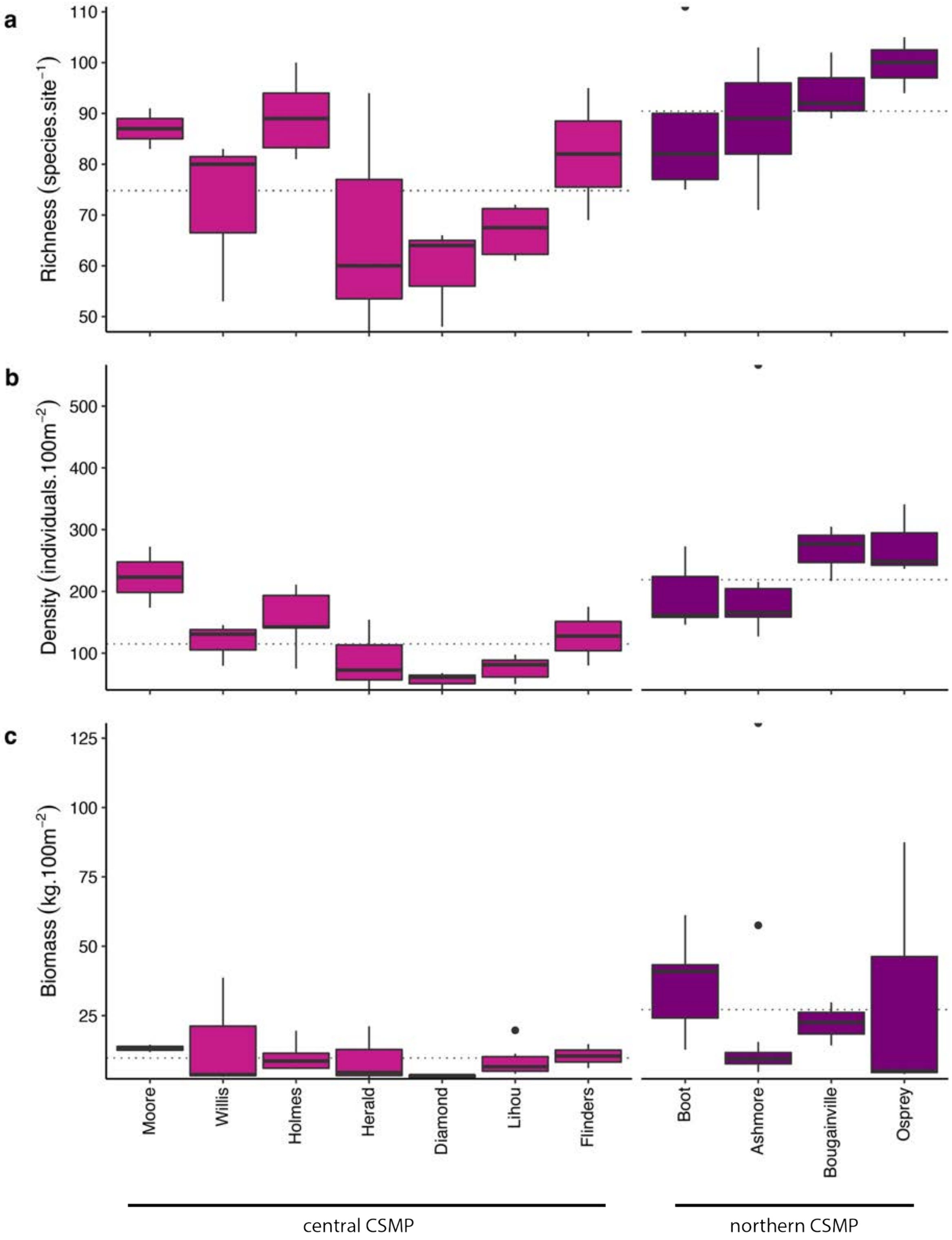


Figure 4.19 Spatial variation in the **(a)** species richness, **(b)** abundance, and **(c)** biomass of coral reef fishes and sharks among the 11 reefs surveyed in the Coral Sea Marine Park during 2023. Data are based on the 50m belt transects, with data for richness based on the number of fish species recorded at each of the 48 sites (i.e., pooled across transects and slope and crest habitats). Reefs are arranged into the central and northern CSMP and coloured by *a priori* regional assignments (following Fig 3.1). Dotted lines represent regional averages.

4.3.2 Temporal changes in reef fish richness, density and biomass

Richness – On the reefs that were surveyed in at least two of the past four years (2020-2023), the number of fish species recorded per site declined gradually from 2020 to 2022 in the central CSMP (2020: 82 species; 2021: 76 species; 2022: 72 species), before stabilising in 2023 (75 species per site; Figure 4.20a). This reduction in species richness likely reflects the loss of fish species that are dependent of corals for food and/or shelter following the 66.9% reduction in coral cover on central CSMP reefs over this period (Figure 4.2). There was a small decline in fish species richness on reefs in the northern CSMP from 2020 to 2022 (2020: 97 species; 2022: 95 species), before it increased again in 2023 (97 species (Figure 4.20a). Despite some interannual differences, the declines in species richness were relatively consistent among reefs in each region. The only exception being Moore Reefs, where species richness increased from 82 to 87 species per site from 2020 to 2023 (Figures 4.21a, 4.22).

Density – The density of reef fishes showed a similar pattern to that of species richness in the central CSMP, declining by ~50% from 2020 to 2022 (2020: 188.7 individuals per 100m^2; 2022: 93.7 individuals per 100m^2), before recovering slightly in 2023 (130.4 individuals per 100m^2; Figure 4.20b). In contrast, the density of reef fish increased in the northern CSMP from 2020 to 2022, before declining (Figure 4.20b). These changes in density were primarily driven by changes in abundance of corallivores, planktivores and grazing herbivores (see Section 4.3.3 below). Declines in mean density were variable among the seven central CSMP reefs, with some reefs displaying gradual declines among years (e.g., Herald Cays and Diamond Islets), while others underwent an initial decline and then stabilised or increased (e.g., Lihou and Holmes Reefs; Figure 4.21b). These differences among reefs are likely related to differences in the magnitude of coral loss and changes in coral composition among reefs.

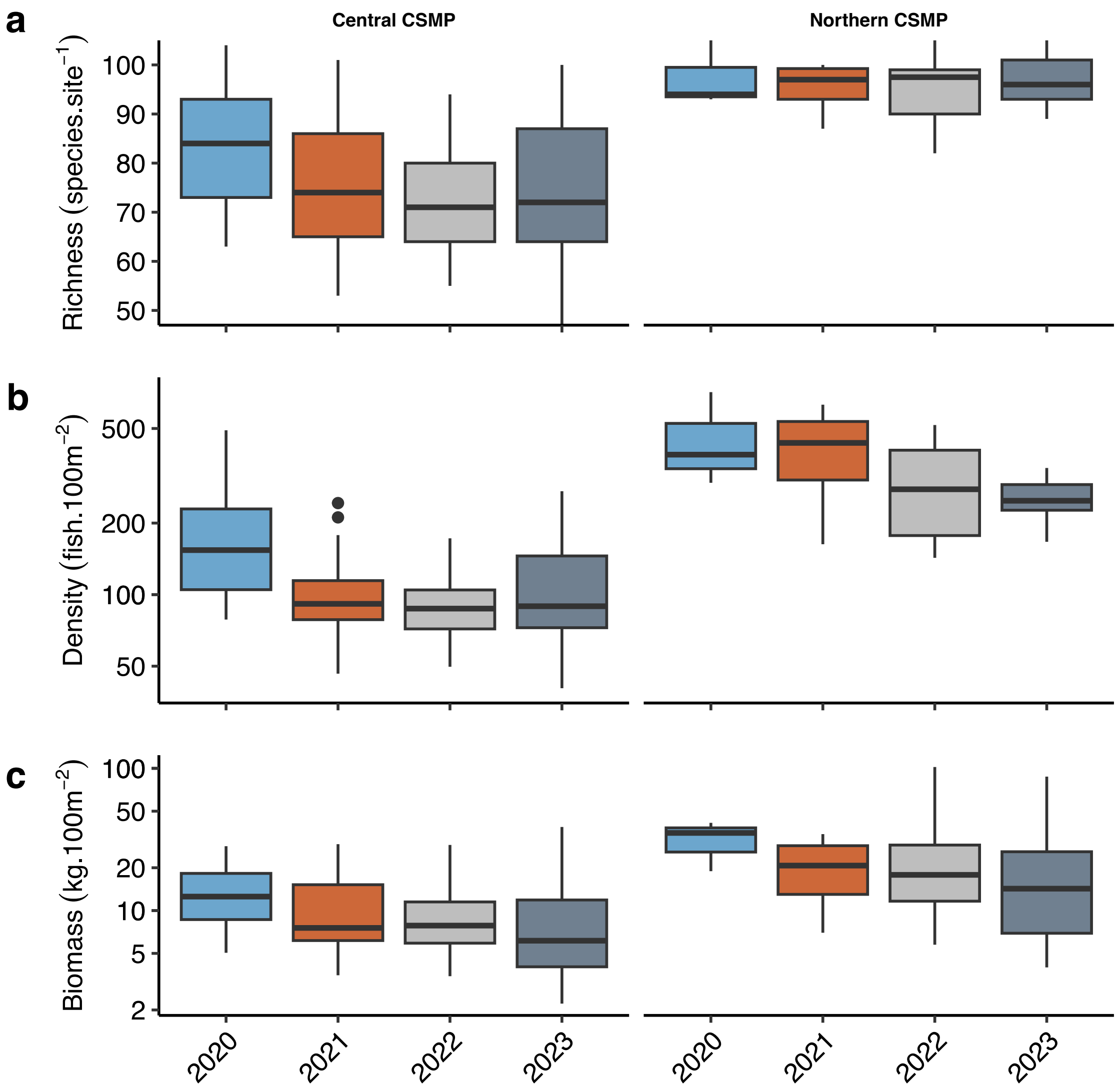


Figure 4.20 Temporal variation in the **(a)** species richness, **(b)** density, and **(c)** biomass of reef fish and shark assemblages among the two regions of the Coral Sea Marine Park. Data are based on surveys of 32 matching sites in 2020, 2021, 2022, and/or 2023 across 10 reefs (central CSMP: Flinders, Holmes, Lihou, and Moore Reefs, Willis and Diamond Islets, and Herald Cays; northern CSMP: Ashmore, Bougainville and Osprey Reefs) and pooled between habitats (reef slope and reef crest) within each site. Note: the data for **(b)** density, and **(c)** biomass are presented on a $\log_{10}$-scale.

Biomass – Declines in reef fish biomass from 2020 to 2023 were evident for both the central (2020: 13.9 kg per 100m^2; 2023: 9.7 kg per 100m^2) and northern CSMP reefs (2020: 31.8 kg per 100m^2; 2023: 24.7 kg per 100m^2) (Figure 4.20c). Note, the large increase in biomass in 2022 on northern CSMP reefs was driven by two schools (100 individuals in total) of the bumphead parrotfish, *Bolbometopon muricatum* (each 80-100cm in total length) being recorded along the reef crest at Bougainville 5 (Figures 4.21c, 4.23). These schools were not observed during the

2023 surveys. With the exception of the increase in reef fish biomass at Bougainville Reef in 2022, the declines in reef fish biomass have been largely consistent across reefs (Figures 4.21c, 4.24), and sites within each reef (Figure 4.23).

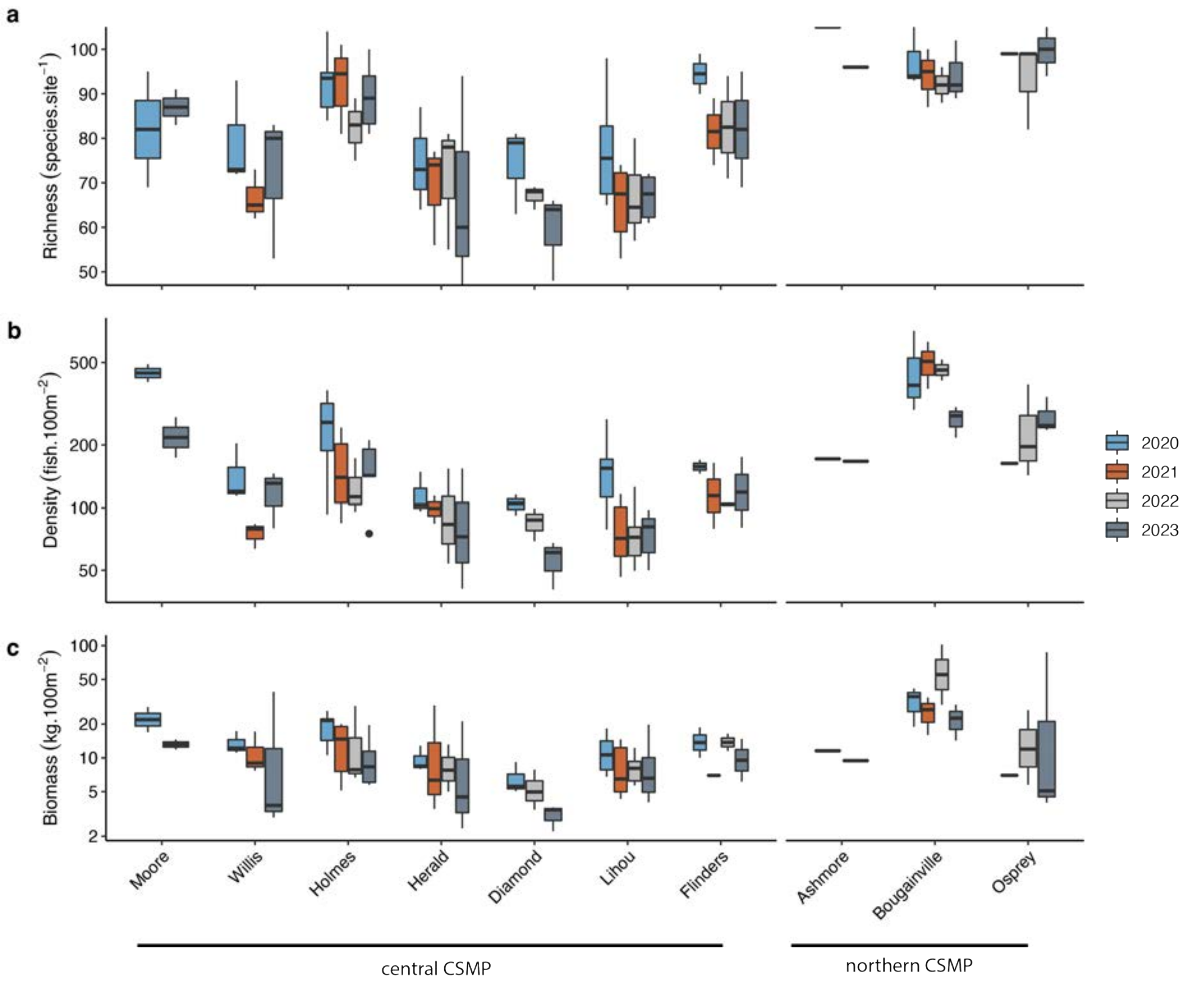


Figure 4.21 Temporal variation in the **(a)** species richness, **(b)** density, and **(c)** biomass of reef fish and shark assemblages among reefs in the Coral Sea Marine Park. Data are based on surveys of 32 matching sites in 2020, 2021, 2022, and/or 2023 across 10 reefs (central CSMP: Flinders, Holmes, Lihou, and Moore Reefs, Willis and Diamond Islets, and Herald Cays; northern CSMP: Ashmore, Bougainville and Osprey Reefs) and pooled between habitats (reef slope and reef crest) within each site. Note: the data for **(b)** density, and **(c)** biomass are presented on a $\log_{10}$-scale.

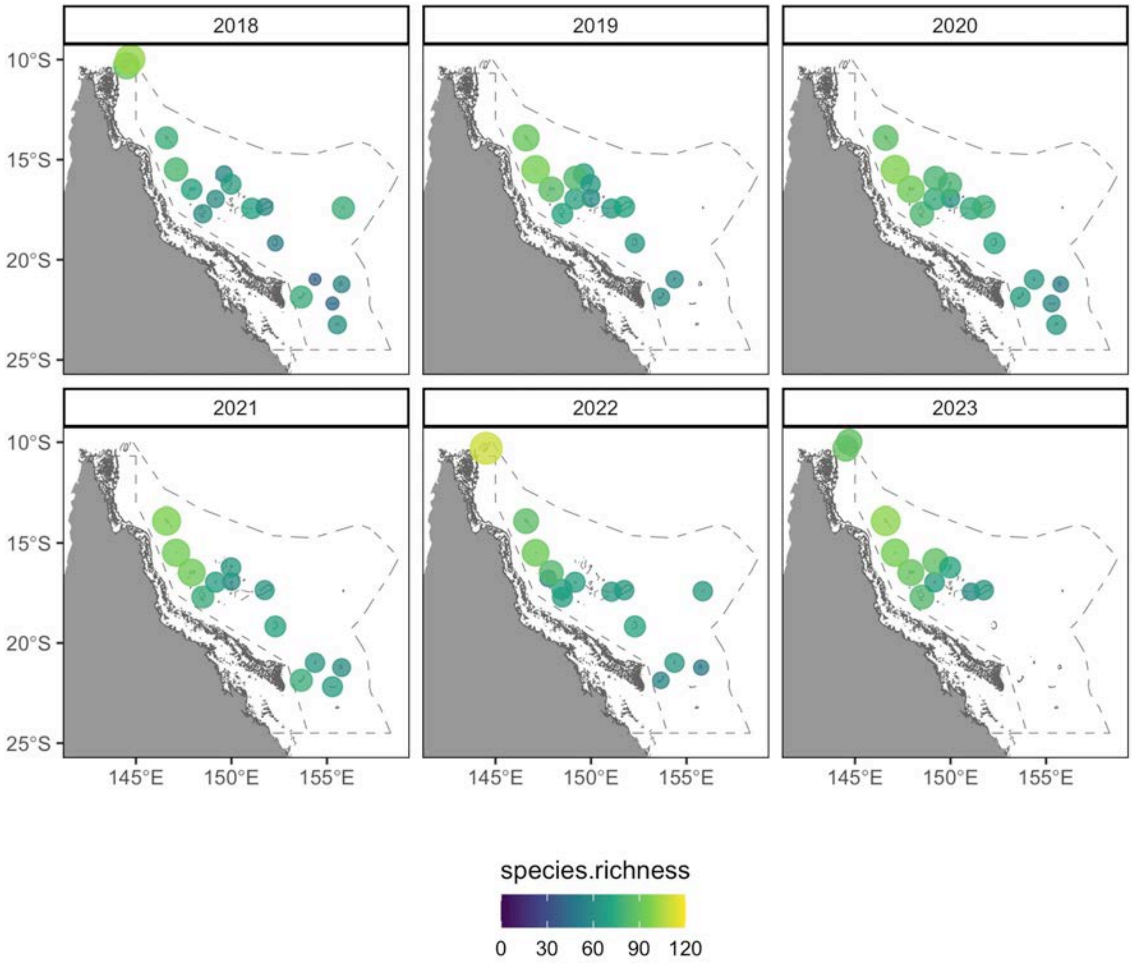


Figure 4.22 Spatial and temporal variation in the species richness of reef fish and sharks on shallow reef habitats (reef crest and reef slope) across 22 reef systems in the Coral Sea Marine Park (2018-2023). The size of individual points is proportional to the number of fish species recorded at each reef.

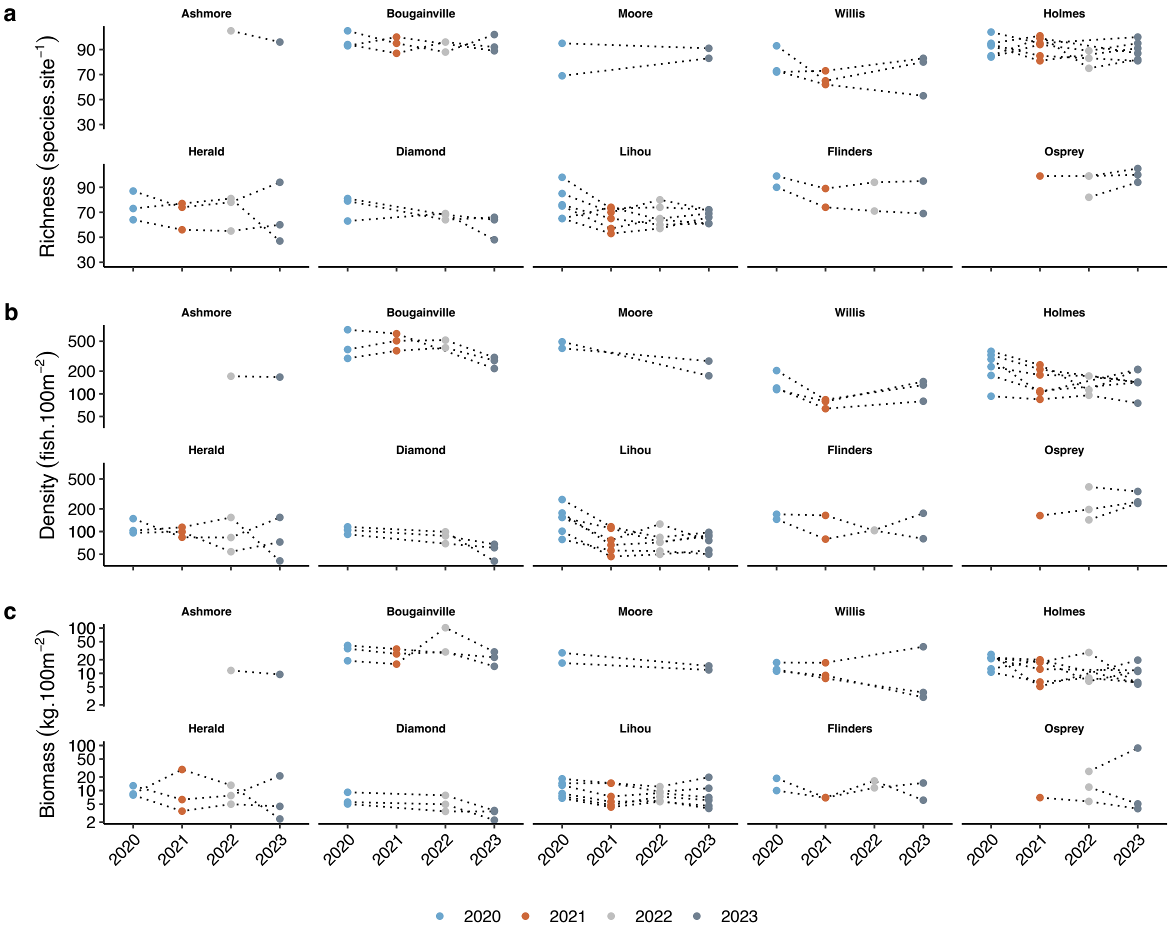


Figure 4.23 Temporal variation in the **(a)** species richness, **(b)** density, and **(c)** biomass of reef fish and shark assemblages among individual sites in the Coral Sea Marine Park. Data are based on surveys of 32 matching sites in 2020, 2021, 2022, and/or 2023 across 10 reefs (central CSMP: Flinders, Holmes, Lihou, and Moore Reefs, Willis and Diamond Islets, and Herald Cays; northern CSMP: Ashmore, Bougainville and Osprey Reefs) and pooled between habitats (reef slope and reef crest) within each site. Note: the data for **(b)** density, and **(c)** biomass are presented on a $\log_{10}$-scale.

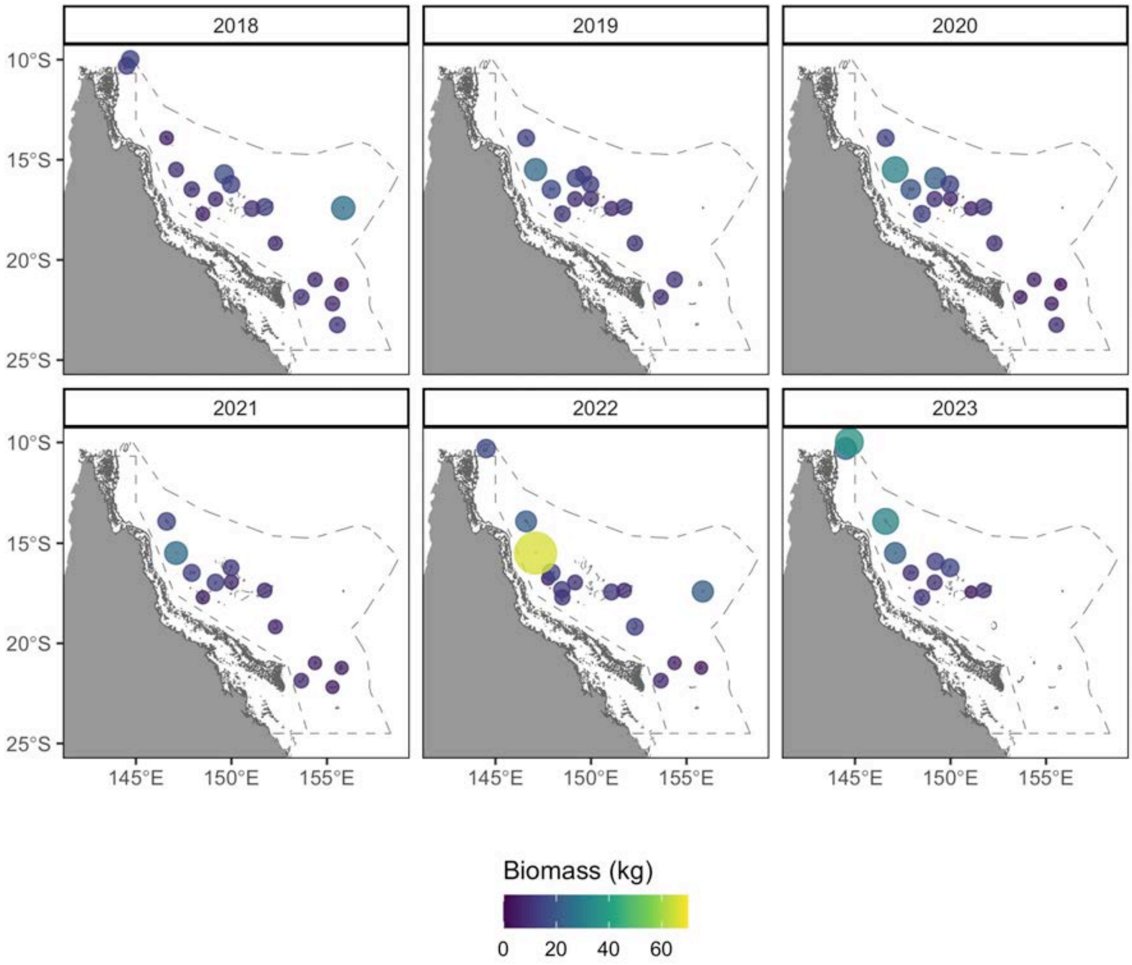


Figure 4.24 Spatial and temporal variation in the biomass of reef fish and sharks on shallow reef habitats (reef crest and reef slope) across 22 reef systems in the Coral Sea Marine Park. The size of individual points is proportional to the average fish biomass at each reef.

4.3.3 Functional composition of fish assemblages

Fishes were categorised into eleven functional groups (piscivore, mixed carnivore, benthic invertivore, planktivore, omnivore, corallivore, excavator, scraper, browser, grazer, and farmer) 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 accounting for approximately 60.3% of all fish recorded, but only 16.3% of total fish biomass, from 2020-2023 (Figure 4.25). Fish biomass was more evenly spread among functional groups with grazing herbivores (14.9%), piscivores (15.3%), planktivores (16.3%), and excavating parrotfishes (18.3%) together accounting for 64.8% of total fish biomass from 2020-2023 (Figure 4.25).

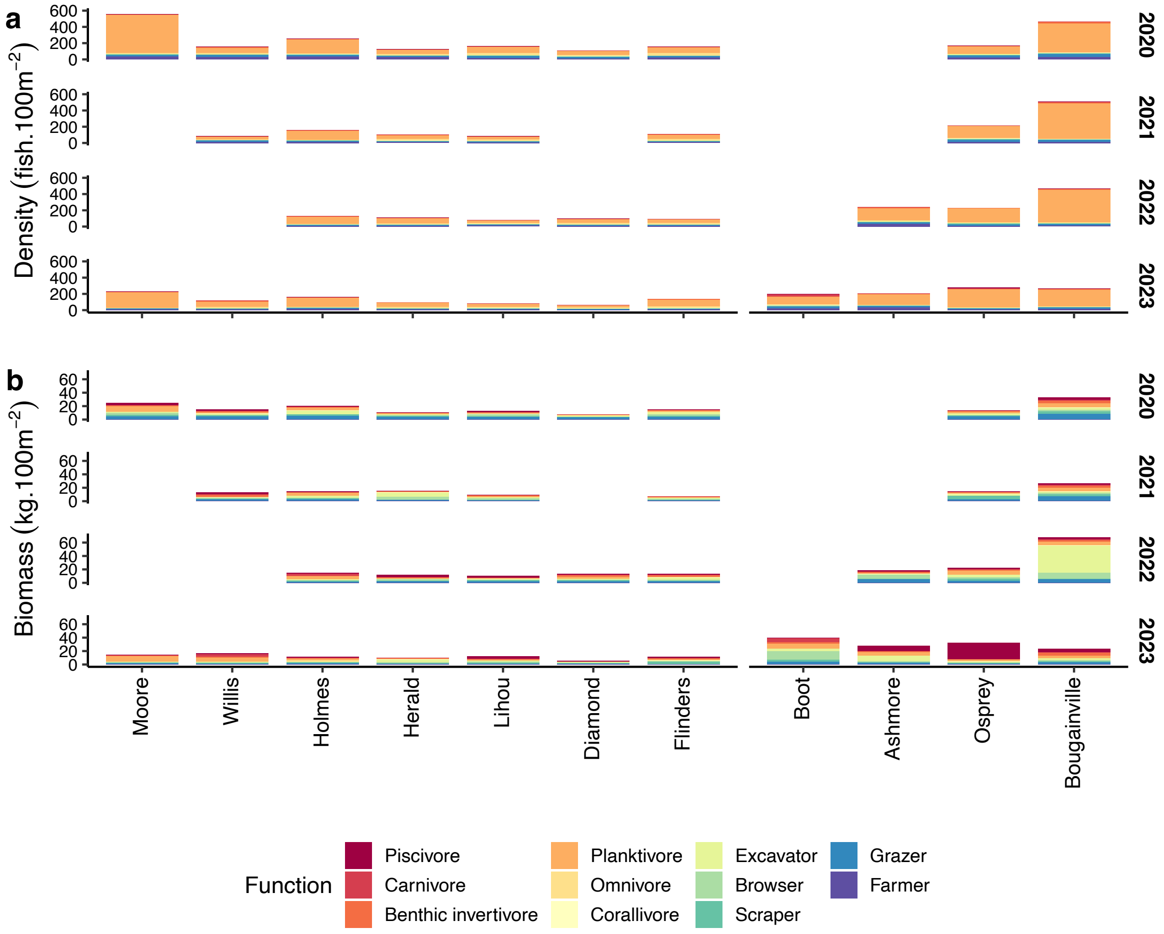


Figure 4.25 Temporal variation (2020-23) in the functional composition of reef fish assemblages across 11 reefs in the Coral Sea Marine Park based on **(a)** abundance, and **(b)** biomass. Data are based 50m belt transects and values for each reef are averaged across habitats and sites.

Following the 2020 and 2021 coral bleaching events, the abundance of corallivorous fishes in both the central and northern CSMP and planktivorous fishes in the central CSMP had declined, likely reflecting their reliance of live corals for food and shelter, respectively (Hoey et al. 2022). The abundance of corallivores remained relatively unchanged between 2022 and 2023 in both the central and northern CSMP (Figure 4.26b). There was, however, a small increase in the abundance of planktivorous fish (primarily fusiliers, anthias and damselfish) on central CSMP reefs from 2022 to 2023 (2022: 44.7 individuals per 100m^2; 2023: 73.1 individuals per 100m^2; Figure 4.26b), that partly offset previous declines (2020-2022) in the abundance of this group. In contrast, the abundance of planktivorous fish declined in the northern CSMP from 2022 to 2023 (2022: 244.0

individuals per 100m^2; 2023: 193.7 individuals per 100m^2), after increasing from 2020 to 2023 (Figure 4.26b). The majority of planktivorous and corallivorous fishes are small-bodied and hence are not major contributors to reef fish biomass. The abundance of piscivorous fishes remained relatively unchanged in the central CSMP from 2022 to 2023, and there was a large increase in the northern CSMP due to a school of several hundred big eye trevally, *Caranx sexfaciatus*, at Osprey Reef (North Horn, Figures 4.26a, 4.28).

Previous declines in the density and biomass of grazing herbivorous fishes on central and northern CSMP reefs between 2020 and 2022 (Hoey et al. 2022) were maintained in 2022 with further declines in abundance and biomass of this group evident in both regions in 2023 (Figures 4.26d, 4.27d). The abundance (and biomass) of grazing fishes has now declined by 59.3% (biomass: 66.7%) and 65.4% (biomass: 71.4%) in the central and northern CSMP, respectively, since 2020 (Figures 4.26d, 4.27d). These declines were primarily driven by reductions in the density and biomass of grazing surgeonfishes (in particular *Acanthurus lineatus* and *Acanthurus nigrofuscus*). The continued declines in the density and biomass of grazing surgeonfishes are difficult to reconcile as several studies have reported substantial increases in the abundance and/or biomass of herbivorous fishes following large-scale bleaching-induced coral mortality (e.g., Adam et al 2011; Gilmour et al. 2013). Such increases have generally been related to an increase in the availability of EAM-covered substrata and subsequent increases in the growth rates of individual fishes (e.g., parrotfishes: Taylor et al. 2020). The immediate and sustained decline of grazing fishes following the 2020 bleaching event suggest that these changes may be related to the physiological response of these fishes to heat stress (Stuart-Smith et al. 2018), and/or the rapid colonisation of dead coral skeletons by CCA (as opposed to algal turfs which are the favoured feeding substrata of these fishes). Further dedicated investigation into the diet and fitness of these fishes on CSMP reefs is required to identify the likely mechanism/s for these declines.

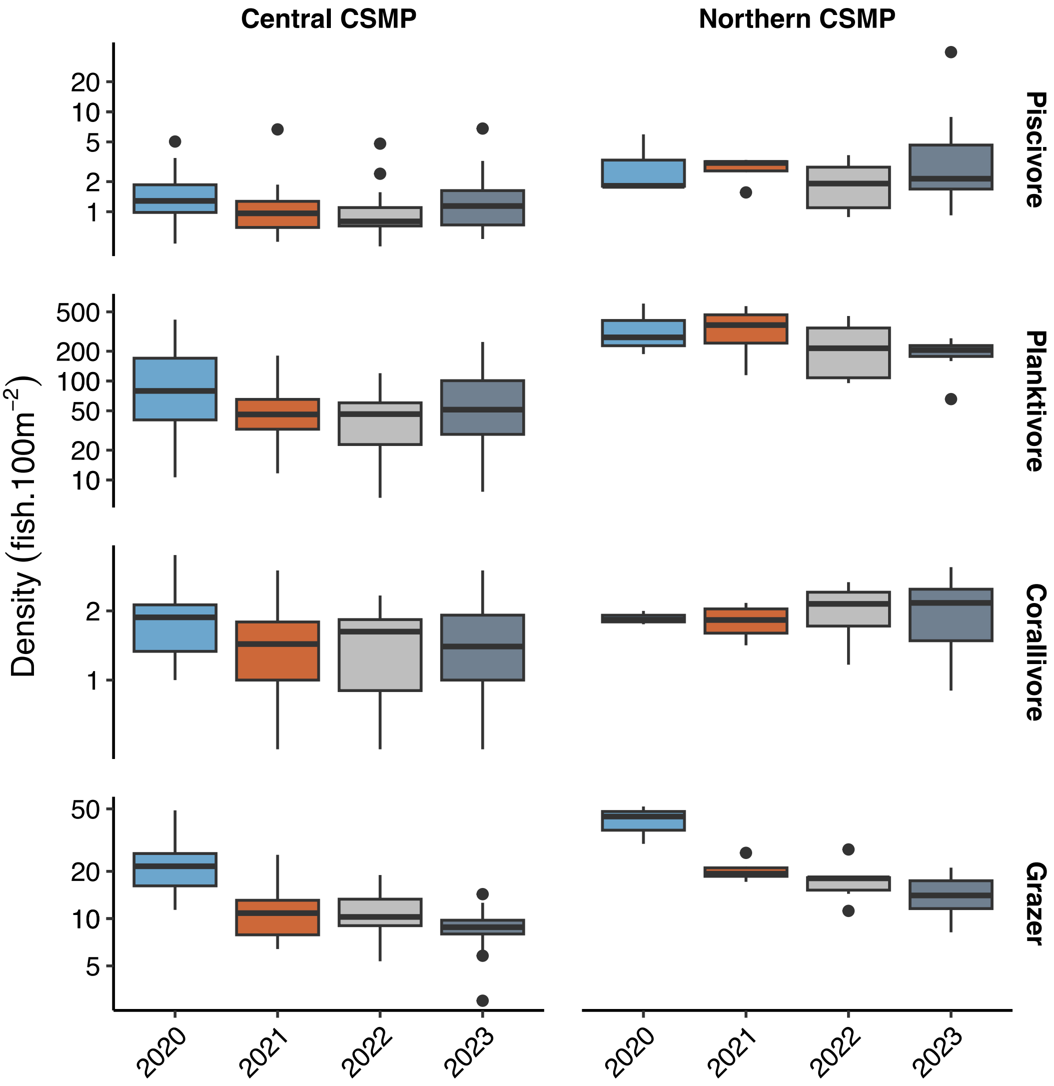


Figure 4.26 Spatial and temporal variation in the density of **(a)** piscivorous, **(b)** planktivorous, **(c)** corallivorous, and **(d)** grazing fishes among the three regions of the Coral Sea Marine Park during 2020, 2021 and 2022. Data are based on replicate 50m transects at each of 32 matching sites across 10 reefs that were surveyed in 2020, 2021, 2022, and/or 2023 (central CSMP: Flinders, Holmes, Lihou, and Moore Reefs, Willis and Diamond Islets, and Herald Cays; northern CSMP: Ashmore, Bougainville and Osprey Reefs). Note: data are presented on a $\log_{10}$-scale.

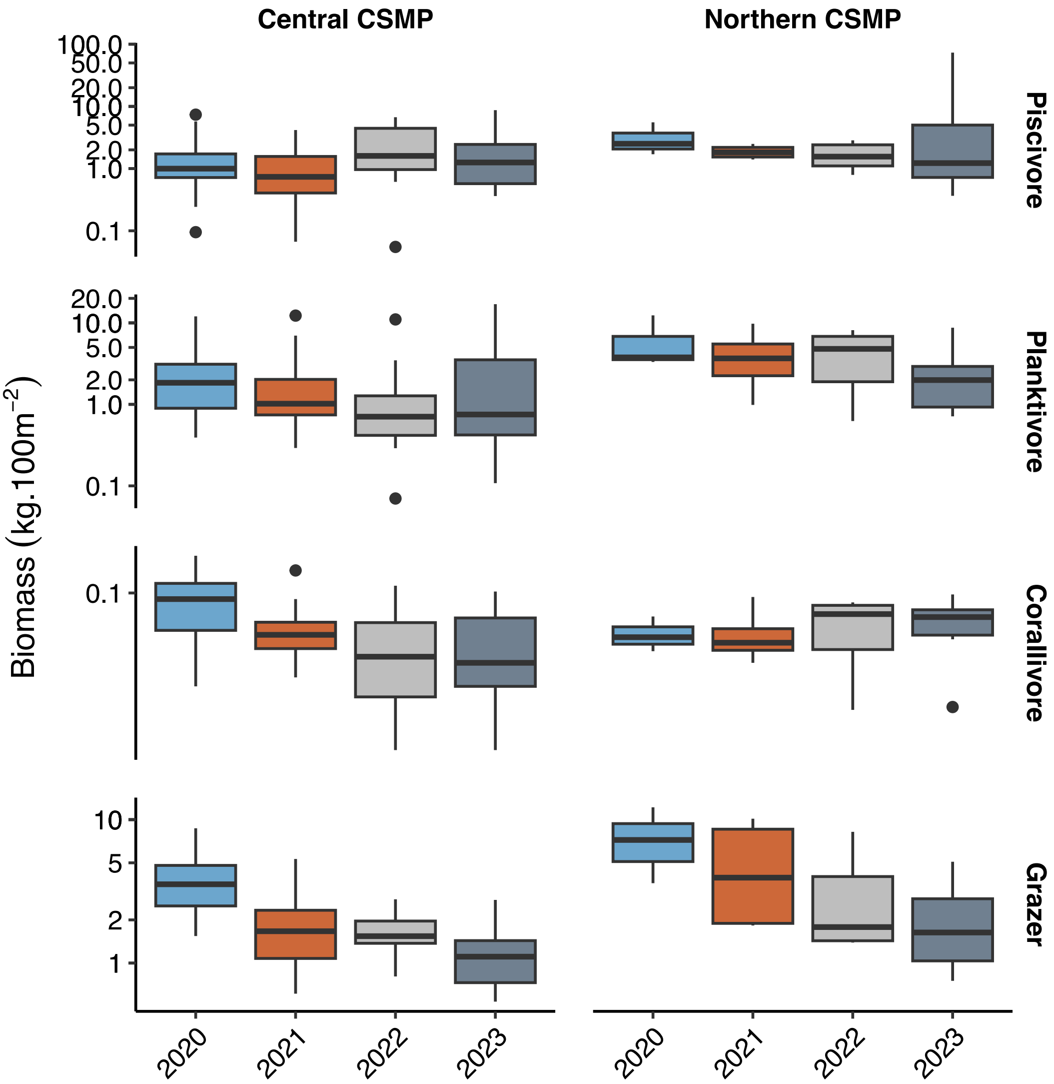


Figure 4.27 Spatial and temporal variation in the biomass of **(a)** piscivorous, **(b)** planktivorous, **(c)** corallivorous, and **(d)** grazing fishes among the two regions of the Coral Sea Marine Park (2020-23). Data are based on replicate 50m transects at each of 32 matching sites across 10 reefs that were surveyed in 2020, 2021, 2022, and/or 2023 (central CSMP: Flinders, Holmes, Lihou, and Moore Reefs, Willis and Diamond Islets, and Herald Cays; northern CSMP: Ashmore, Bougainville and Osprey Reefs). Note: data are presented on a $\log_{10}$-scale.

Figure 4.28 Photographs of fish assemblages at Osprey Reef, northern Coral Sea Marine Park, February 2023. Top: School of mixed parrotfishes (*Scarus altipinnis* and *Chlorurus microrhinos*) feeding on the shallow reef crest. Bottom: Large school of big eye trevally (*Caranx sexfasciatus*) over a coral bommie at False Entrance, Osprey Reef. Image credits: Victor Huertas.

4.3.4 Fish community composition

Taxonomic composition - The greatest variation in the taxonomic composition of reef fish assemblages, like coral assemblages (see Section 4.1.3 above), was between the two CSMP regions (Figure 4.29). The northern CSMP reefs were tightly clustered in the upper left-hand space of the nMDS, whereas the central CSMP reefs were more evenly spread throughout the nMDS space (Figure 4.29). Despite changes in the species richness, abundance and biomass of different functional groups of reef fishes on CSMP reefs from 2020 to 2023, the species composition of fish communities remained relatively stable over the same period, with almost complete overlap between years (Figure 4.29). Similarly, there was no evidence of a shift in the taxonomic composition of reef fish and shark assemblages within the central or northern CSMP from 2020 to 2023 (Figure 4.30). The nMDS failed to converge for the northern CSMP sites, likely due to the limited number of sites surveyed in each year.

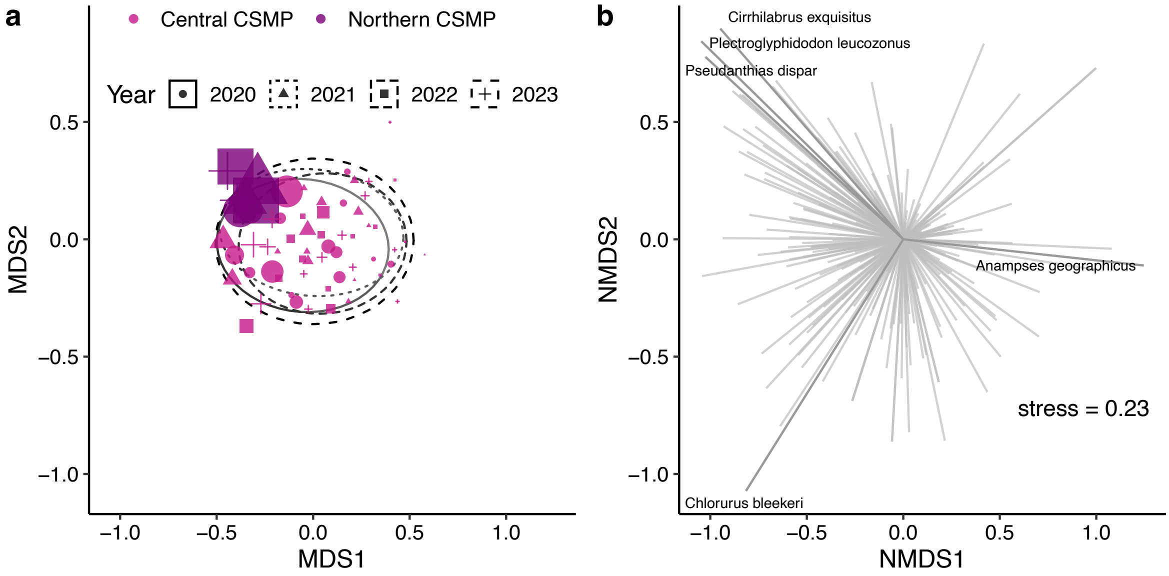


Figure 4.29 Regional and temporal (2020-23) variation in the taxonomic composition of reef fish and shark assemblages within the Coral Sea Marine Park. Non-metric multidimensional scaling (nMDS) plot showing the variation in reef fish composition among years for the two regions of the Coral Sea Marine Park. Analyses are based on abundance data from 16 sites that were surveyed in each of the past four years. The size of individual points is proportional to the total fish abundance on each reef. Vectors in the right-hand side plot indicate key taxa that account for the variation in fish composition displayed in the corresponding left-hand side plot.

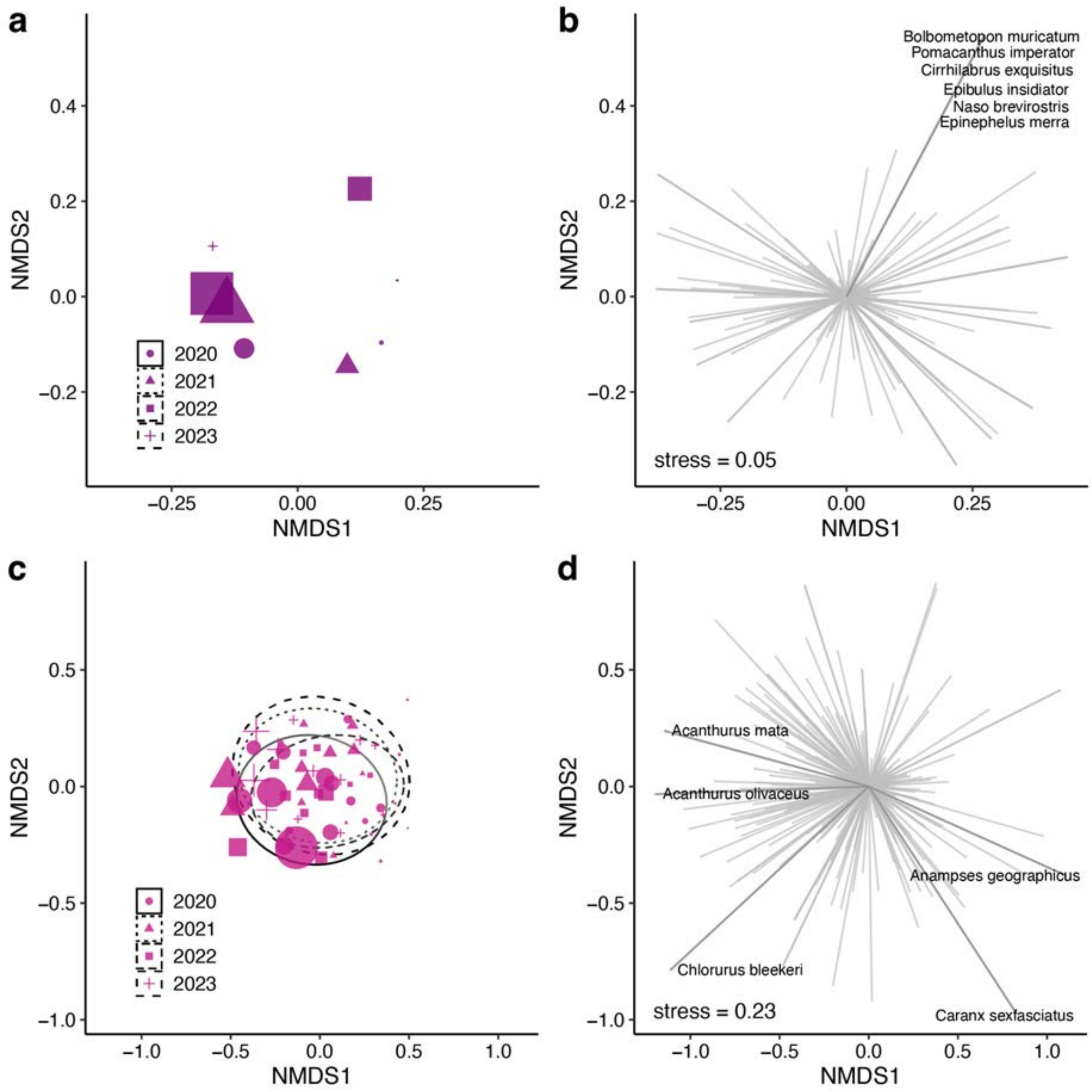


Figure 4.30 Non-metric multidimensional scaling (nMDS) plots showing the temporal variation (2020-23) in the taxonomic composition of reef fish and shark assemblages among reefs in the **(a)** northern, and **(c)** central Coral Sea Marine Park. Analyses are based on abundance data from 16 sites that were surveyed in each of the past four years. The size of individual points is proportional to the total fish abundance at each site. Vectors in the right-hand side plot indicate key taxa that account for variation in fish composition displayed in the corresponding left-hand side plot. Note: the model for the northern CSMP sites failed to converge.

Functional composition – Similar to taxonomic composition, the greatest variation in the functional composition of reef fish assemblages, was between the two CSMP regions (Figure 4.29). The northern CSMP and central CSMP were differentiated along the first dimension of the nMDS, with northern CSMP reefs being characterised by a higher abundance of planktivorous fishes, and to a lesser extent benthic invertivores (Figure 4.31). In contrast, central CSMP reefs were characterised by a higher abundance of omnivorous, and to a lesser extent browsing herbivores and farming damselfish (Figure 4.31). There was also some evidence of a shift in the functional composition of fish assemblages among years. Fish assemblages in 2020 (i.e., prior to the bleaching events) were more variable being spread throughout the nMDS space, with assemblages in subsequent years being more tightly clustered around the centre of the nMDS space indicating reductions in several groups (Figure 4.31). Similar shifts were evident when assessing changes in the northern and central CSMP independently (Figure 4.32). The nMDS failed to converge for the northern CSMP sites, likely due to the limited number of sites surveyed in each year.

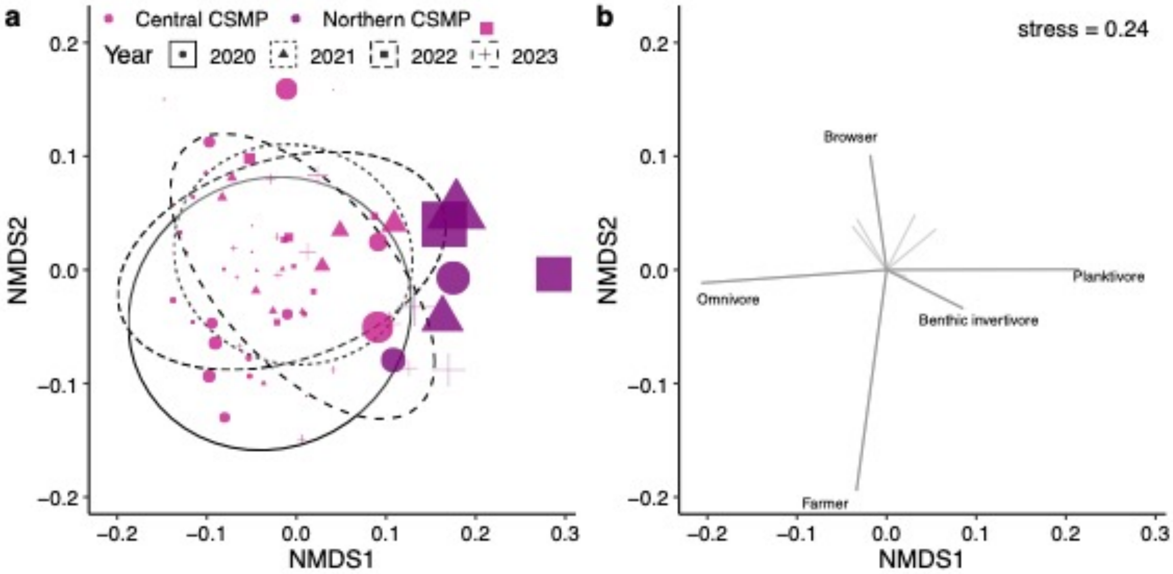


Figure 4.31 Regional and temporal (2020-23) variation in the functional composition of reef fish and shark assemblages within the Coral Sea Marine Park. Non-metric multidimensional scaling (nMDS) plot showing the variation in reef fish functional composition among years for the two regions of the Coral Sea Marine Park. Analyses are based on abundance data from 16 sites that were surveyed in each of the past four years. The size of individual points is proportional to the total fish abundance on each reef. Vectors in the right-hand side plot indicate key groups that account for the variation in fish composition displayed in the corresponding left-hand side plot.

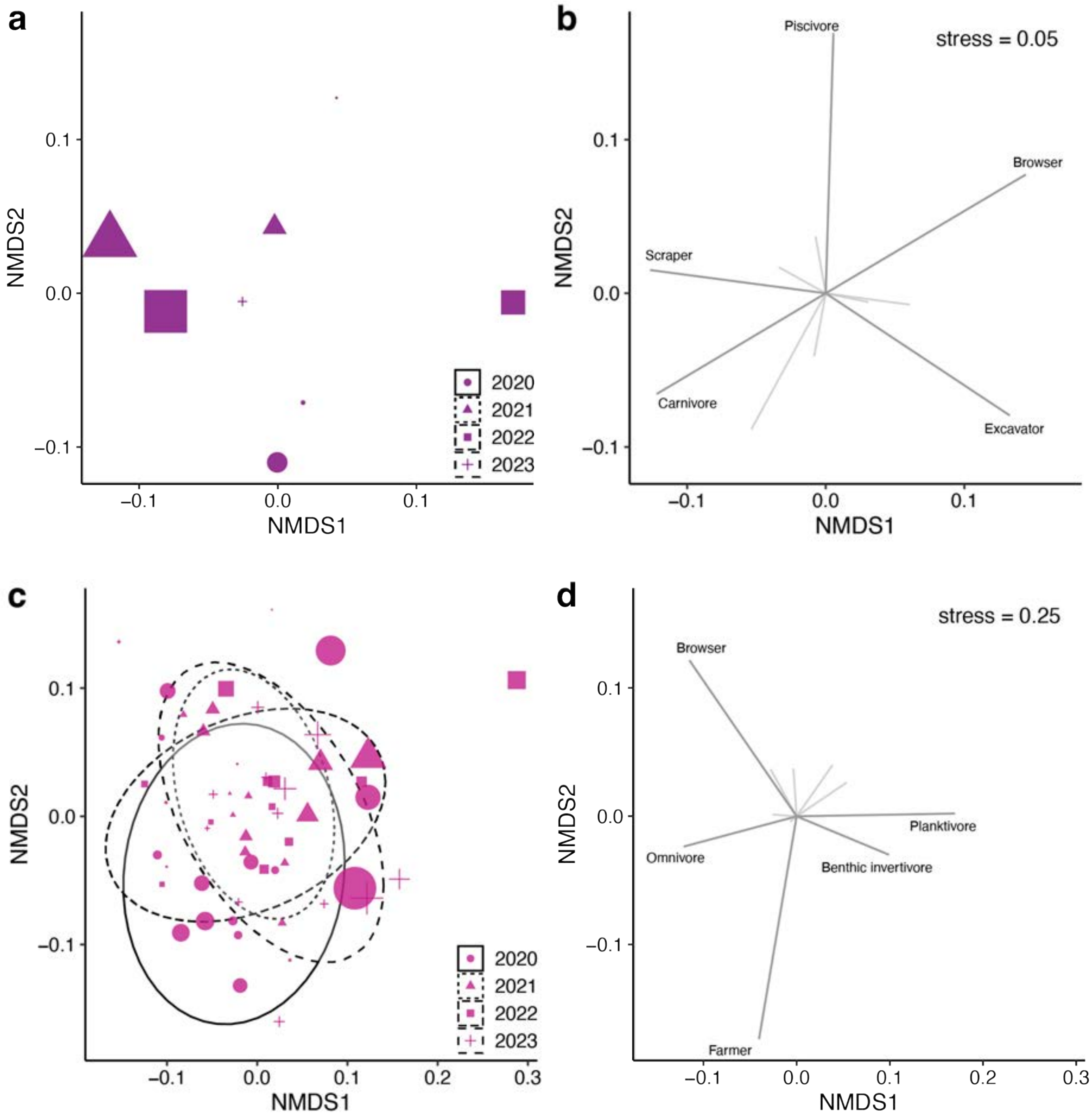


Figure 4.32 Non-metric multidimensional scaling (nMDS) plots showing the temporal variation (2020-23) in the functional composition of reef fish and shark assemblages among reefs in the **(a,b)** northern, and **(c,d)** central Coral Sea Marine Park. Analyses are based on abundance data from 16 sites that were surveyed in each of the past four years. The size of individual points is proportional to the total fish abundance at each site. Vectors in the right-hand side plot indicate key groups that account for variation in fish composition displayed in the corresponding left-hand side plot. Note: the model for the northern CSMP sites failed to converge.

Photographs of abundant and high biomass fish communities on the exposed aspect of Boot Reef, northern Coral Sea Marine Park. Top: Large school of bumphead parrotfish (*Bolbometopon muricatum*) on the shallow reef crest. Each individual is 80-100cm long. Bottom: School of paddletail snapper (*Lutjanus gibbus*) closely associated with the benthos at 12m on the reef slope. Note the difference in the benthic communities between habitats. Image credits: Andrew Hoey

4.4 Other reef taxa

4.4.1 Sea snakes

Previous surveys in 2019-2022 have shown that sea snakes are abundant on all reefs in the southern CSMP from Cato Reef to Marion Reef, and at Ashmore Reef in the far north of the CSMP, but were not observed (and presumably absent) at all other reefs in the central CSMP, and Bougainville and Osprey Reefs in the northern CSMP (Hoey at al. 2020, 2021, 2022). Only a single sea snake (*A. laevis*) was observed during surveys of 12 sites at Ashmore Reef in 2023, equating to a density of 0.01 individuals per 250m^2. This is markedly lower than the mean density recorded at Ashmore Reef in 2022 (0.2 individuals per 250m^2). This reduction is likely to be related to the sites surveyed, rather than a reduction in the sea snake population at Ashmore Reef. The vast majority of sites surveyed in 2022 were inside the lagoon, however these sites were largely inaccessible in 2023 due to strong westerly and north-westerly winds. In contrast, the majority of sites surveyed in 2023 were on the exposed eastern and south-eastern aspect of Ashmore Reefs, and the only sea snake observed was recorded at the only sheltered lagoon site surveyed. Analysis of the ROV footage (to be completed later this year) will provide a greater understanding of the overall population size of sea snakes on Ashmore Reef.

4.4.2 Macro-invertebrates

Giant Clams – Overall, 374 giant clams (*Tridacna* spp. and *Hippopus hippopus*) were recorded across the 11 CSMP reefs in 2023, with the vast majority (357 individuals, 95.4%) being *Tridacna maxima* and *Tridacna squamosa.* The other species recorded were *Tridacna derasa* (5 individuals, 1.3%), *Tridacna gigas* (5 individuals, 4 of which were recorded on Ashmore Reef, 1.3%), *Tridacna crocea* (7 individuals, 1.9%). No *Hippopus hippopus* were recorded in 2022.

The density of giant clams (*Tridacna* spp.) across all reefs in 2023 was low (1.3 clams per 100m^2), however there was considerable variation between regions (central CSMP: 1.6 clams per 100m^2; northern CSMP: 0.9 clams per 100m^2) and among reefs (Flinders Reef: 0.3 clams per 100m^2; Herald Cays: 3.2 clams per 100m^2; Figure 4.33a).

Comparisons of the ten CSMP reefs that were surveyed at least twice between 2020-2023 reveal the abundance of giant clams has seen an almost 50% increase (2020: 1.1 clams per 100m^2; 2022: 1.7 clams per 100m^2), with a relatively small (12%) increase from 2022 to 2023 (2022: 1.5 clams per 100m^2; 2023: 1.7 clams per 100m^2). These increases were driven by increases in the abundance of clams on the central CSMP reefs (2022: 1.3 clams per 100m^2; 2023: 1.7 clams per 100m^2), while there were small decreases in the abundance of clams on northern CSMP reefs (2022: 2.3 clams per 100m^2; 2023: 1.8 clams per 100m^2; Figure 4.34a).

Trochus –*Tectus* spp. (formerly *Trochus*) were relatively rare across the CSMP, with 38 individuals recorded across the 11 CSMP reefs in 2023 (mean density: 0.13 individuals per 100m^2). The density of *Tectus* was similar between the central (0.14 individuals per 100m^2) and northern CSMP (0.13 individuals per 100m^2), however varied considerable among individual reefs (0 to 0.46 individuals per 100m^2 at Flinders Reef and Willis Islets, respectively; Figure 4.33b). Comparisons of the ten CSMP reefs that were surveyed at least twice in the past 4 years show the densities of *Tectus* have gradually declined from 2020 to 2023 (2020: 0.22 individuals per 100m^2; 2023: 0.14 individuals per 100m^2), although there is considerable variation in the direction and magnitude of the change among individual reefs (Figure 4.34b).

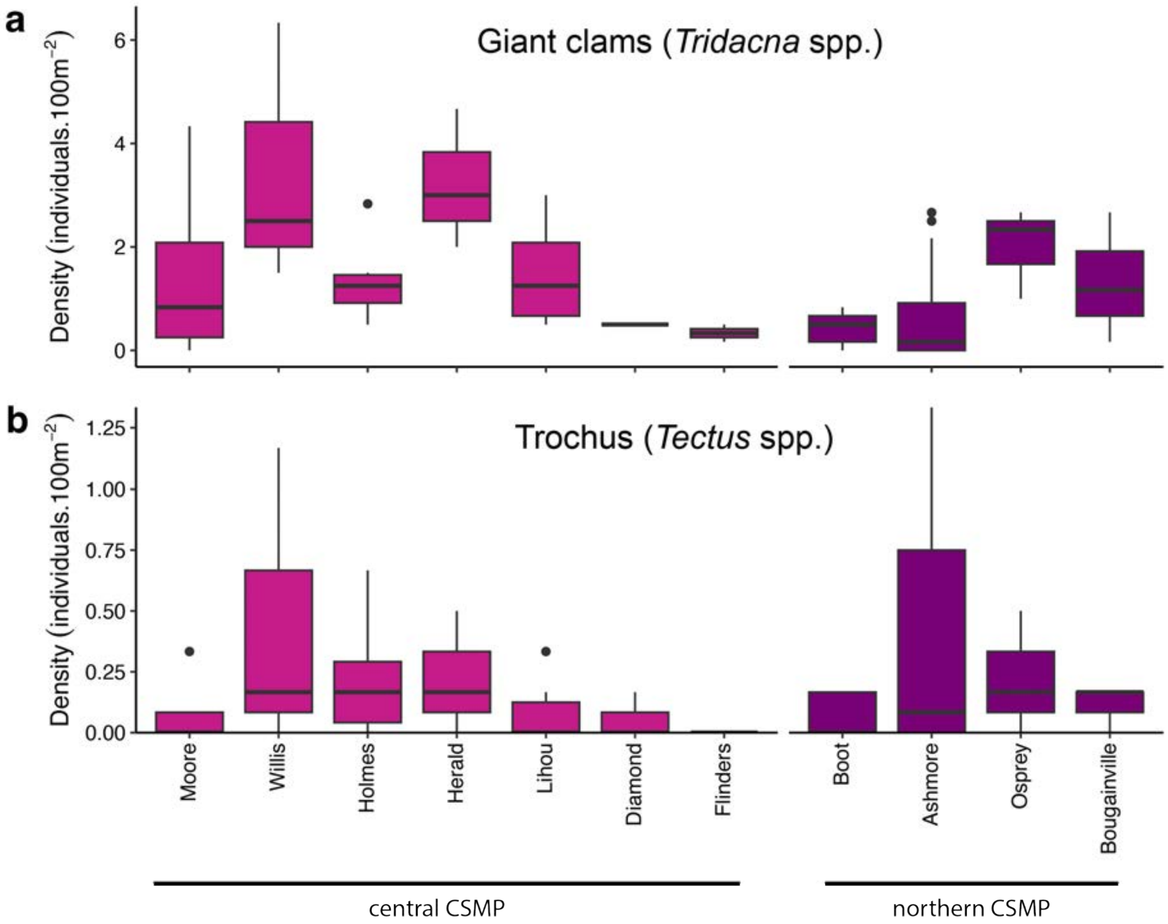


Figure 4.33 Spatial and temporal variation in the abundance of **(a)** giant clams, and **(b)** Trochus among the 11 reefs surveyed in the Coral Sea Marine Park during 2023. Reefs are arranged into the central, and northern CSMP and coloured by *a priori* regional assignments (following Figure 3.1). Bottom: giant clam (*Tridacna*) on the reef slope at Willis Reef, central CSMP. Note the green calcifying alga *Halimeda* on the surrounding substrata Image credit: Victor Huertas

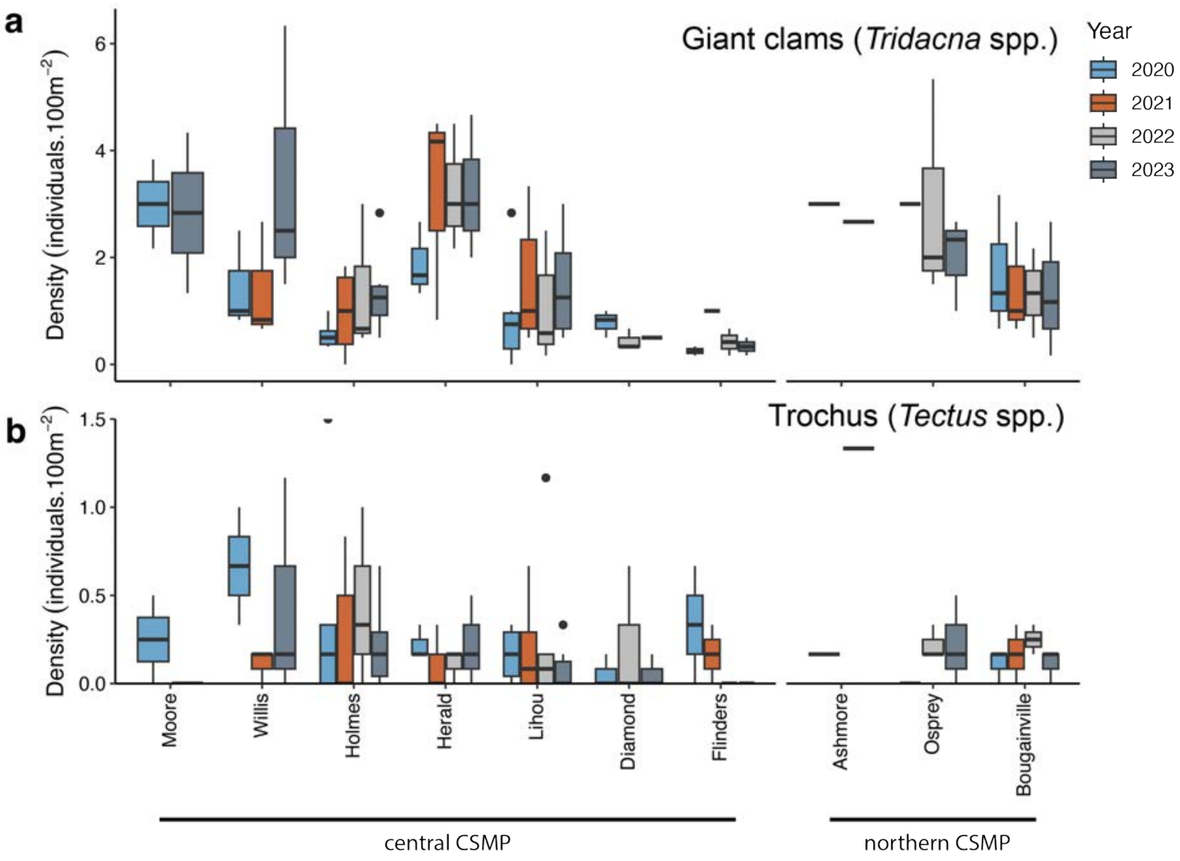


Figure 4.34 Spatial and temporal variation in the abundance of **(a)** giant clams, and **(b)** Trochus among reefs surveyed in the Coral Sea Marine Park (2020-23). Data are based on replicate 50m transects at each of 32 matching sites across 10 reefs that were surveyed in 2020, 2021, 2022, and/or 2023.

Sea urchins – Long-spined sea urchins (*Diadema* spp.) were extremely rare across the 11 CSMP reefs in 2023, with only 2 individuals being recorded across all sites (average density: 0.007 urchins per 100m^2; Figure 4.35a). This low density of *Diadema* is consistent with previous surveys (2018-2022) across the central and northern CSMP (0 – 0.04 urchins per 100m^2; Hoey et al. 2022; Figure 4.36a). *Diadema* are generally more abundant on subtropical reefs, such as Lord Howe Island, and Elizabeth and Middleton Reefs, as well as some of the reefs (e.g., Kenn Reef) in the southern CSMP (Hoey et al. 2011, 2018, 2022).

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., Humphries et al. 2020; Williams 2022). However, on Indo-Pacific reefs high densities of sea urchins, and *Diadema* in particular, are seen as a sign of overfishing and/or reef degradation (McClannahan et al. 1994; Glynn and Manzello 2015) and can result in net erosion of reef carbonates and destabilisation of the reef framework through their feeding (Glynn et al. 1979; Eakin 1996).

Sea cucumbers – A total of 70 sea cucumbers (Holothuroidea) from 10 species were recorded across the 11 CSMP reefs in 2023, equating to an average of 0.24 individuals per 100m^2. The densities of sea cucumbers were generally greater (0.33 individuals per 100m^2) and less variable among reefs in the northern CSMP (0.26-0.35 individuals per 100m^2) than in the central CSMP (average: 0.17 individuals per 100m^2; range 0.0-0.58 individuals per 100m^2; Figure 4.35b). The most abundant species were *Actinopyga mauritiana* (34.3%), *Pearsonothuria graeffei* (14.3%), *Thelenota ananas* (11.4 %), *Stichopus chloronotus* (10.0%) and *Bohadschia argus* (10.0%). The other species recorded were *Holothuria atra Holothuria edulis, Holothuria fuscopunctata, Actinopyga palauensis,* and *Holothuria whitmaei*. The density of sea cucumbers within the shallow reef habitats surveyed showed limited change between years (Figure 4.36b).

When interpreting the density estimates of these macroinvertebrates (i.e., giant clams, trochus and sea cucumbers), and the species composition of giant clams and sea cucumbers across the CSMP, consideration needs to be given to the sampling design, and in particular 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. These are not the preferred habitats for many of these macroinvertebrates. For example, most giant clam (*Tridacna*) species, and *T. 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 populations. Similarly, and as noted previously (Hoey et al. 2020, 2021), the density estimates of sea cucumbers provided herein are substantially lower than those of previous dedicated sea cucumber surveys in the central CSMP (average of 1.33 individuals per 100m^2 for all species combined; 1.06 individuals per 100m^2 for *H. atra*; Skewes and Persson 2017). These differences likely reflect differences in the habitats surveyed, rather than significant changes in sea cucumber populations. Robust assessments of giant clam, trochus, and sea cucumber populations would require dedicated surveys over the preferred habitat of each species. Specifically, these would include deeper lagoonal habitats dominated by sand for sea cucumbers (*sensu* Kinch et al. 2008), shallow exposed reef flat habitats for trochus (Ahmed and Hill 1994), and lagoonal shallow reef flat habitats for giant clams (Braley 1987).

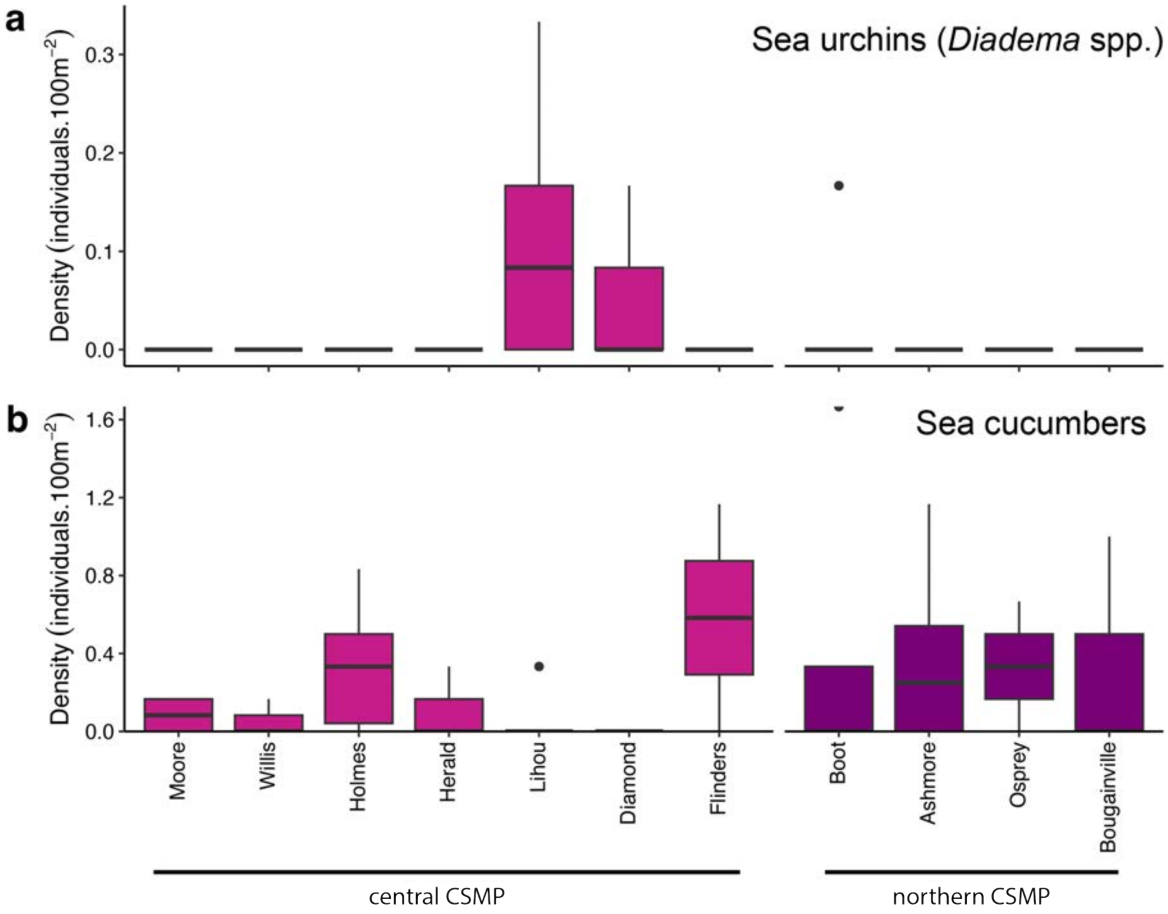


Figure 4.35 Spatial and temporal variation in the abundance of **(a)** the sea urchin, *Diadema* spp., and **(b)** sea cucumbers among the 12 reefs surveyed in the Coral Sea Marine Park during 2023. Data are based on 50 x 2 m belt transects. Reefs are arranged into the central, and northern CSMP and coloured by *a priori* regional assignments (following Figure 3.1).

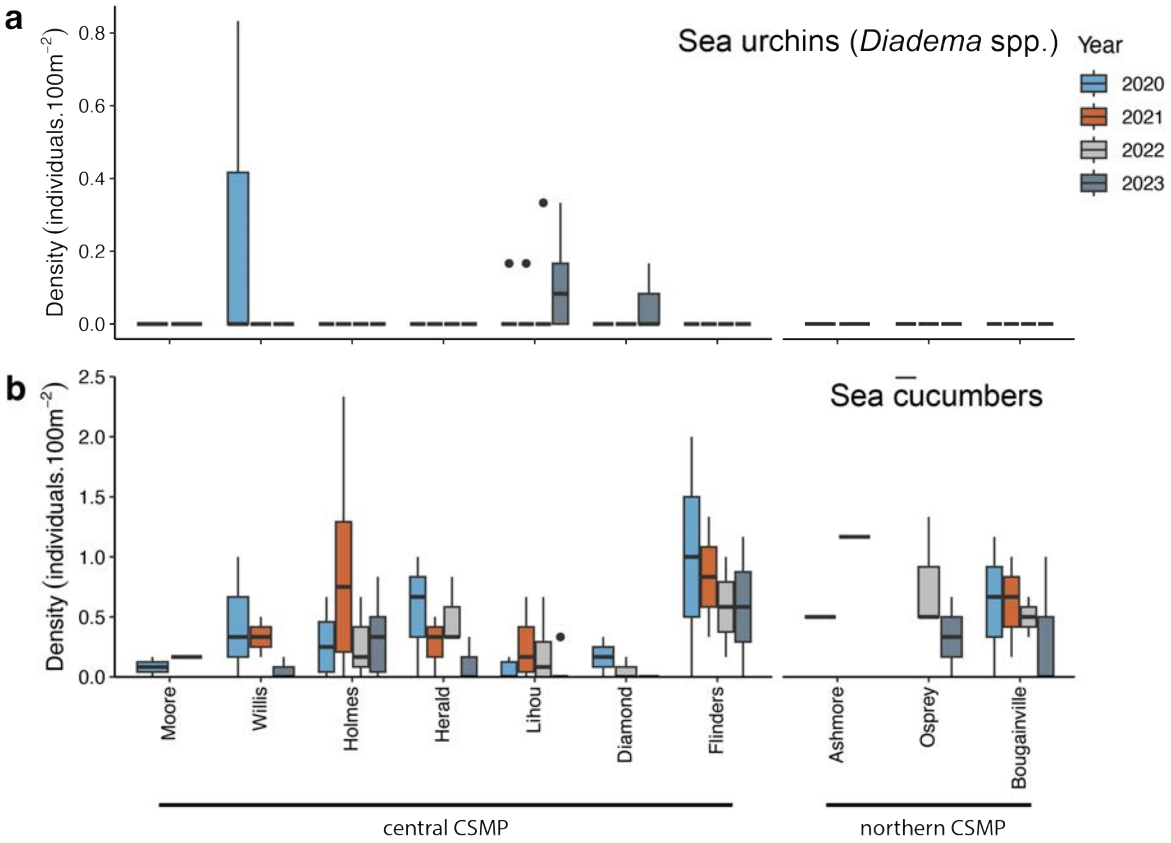


Figure 4.36 Spatial and temporal variation in the abundance of **(a)** sea urchins – *Diadema* spp., and **(b)** sea cucumbers among 11 reefs in the Coral Sea Marine Park (2020-23). Data are based on replicate 50 x 2m transects at each of 32 matching sites across 10 reefs that were surveyed in 2020, 2021, 2022, and/or 2023.

4.5 Coral health and injury

4.5.1 Coral colony size distribution

Shallow coral assemblages of the 11 CSMP reefs surveyed in 2023 were dominated by relatively small coral colonies (<20cm diameter), with few colonies larger than 40cm diameter recorded (Figures 4.37, 4.38). This predominance of small colonies has been evident in the CSMP since this series of surveys was initiated in 2018 (Figure 4.38; Hoey et al. 2020, 2021, 2022), and is indicative of a system that is exposed to frequent disturbance and/or low rates of recovery (Dietzel et al. 2020). Comparisons among regions reveal that the central CSMP has a similar abundance of juvenile coral colonies (<5cm diameter) but a lower abundance of small (6-20cm) and large coral colonies (21-40cm, 41-60cm and >60cm) than the northern CSMP (Figure 4.37).

Declines in the abundance of most coral colony size classes in the central CSMP were evident from 2020 to 2023, except for the abundance of the smallest size class (<5cm diameter) that has gradually increased each year since 2020 (Figure 4.37). In contrast to the central CSMP, the size structure of coral colonies in the northern CSMP remained relatively unchanged between 2020 and 2023, except for an increase in the 6-20cm size class from 2020 to 2021 (Figure 4.37). While the increases in the smallest size classes of corals in the central CSMP could reflect the growth and survival of previously settled corals, the concurrent reduction in abundance of larger corals suggest that at least some of this increase may be attributable to partial mortality of larger colonies, resulting in a greater number of smaller colonies through fission. In the absence of major disturbances, it would be expected that abundance of each size class would increase. This provides further evidence for the likely bleaching related mortality in the central CSMP due to heat stress after our surveys in February 2022 (see also Section 4.1.2).

There has been a marked reduction (40.9% decline) in the abundance of larger coral colonies (>20cm diameter) from 2020-2023 across all CSMP reefs re-surveyed from 2020-2023, which is consistent the effects of major bleaching events (Dietzel et al. 2020).

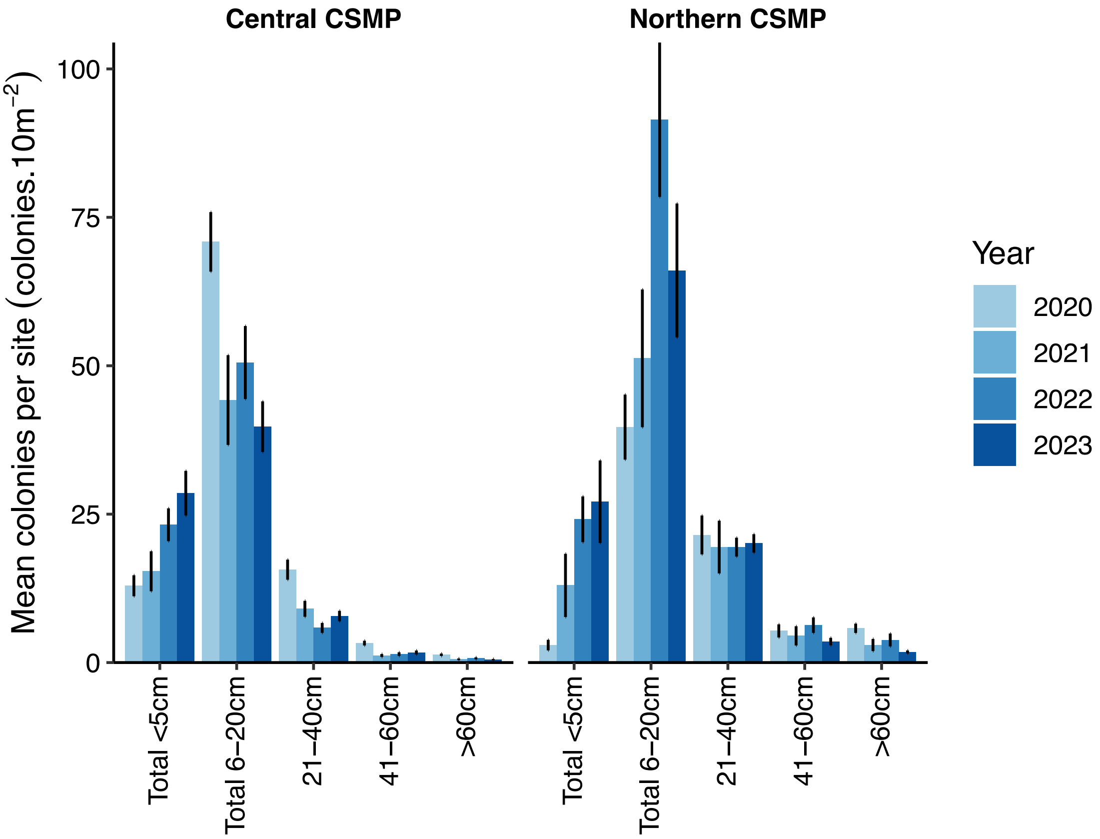


Figure 4.37 Temporal variation in the size frequency distribution of coral colonies surveyed across three regions of the Coral Sea Marine Park. Data are based on 32 sites across 10 reefs that were surveyed in 2020, 2021, 2022 and/or 2023.

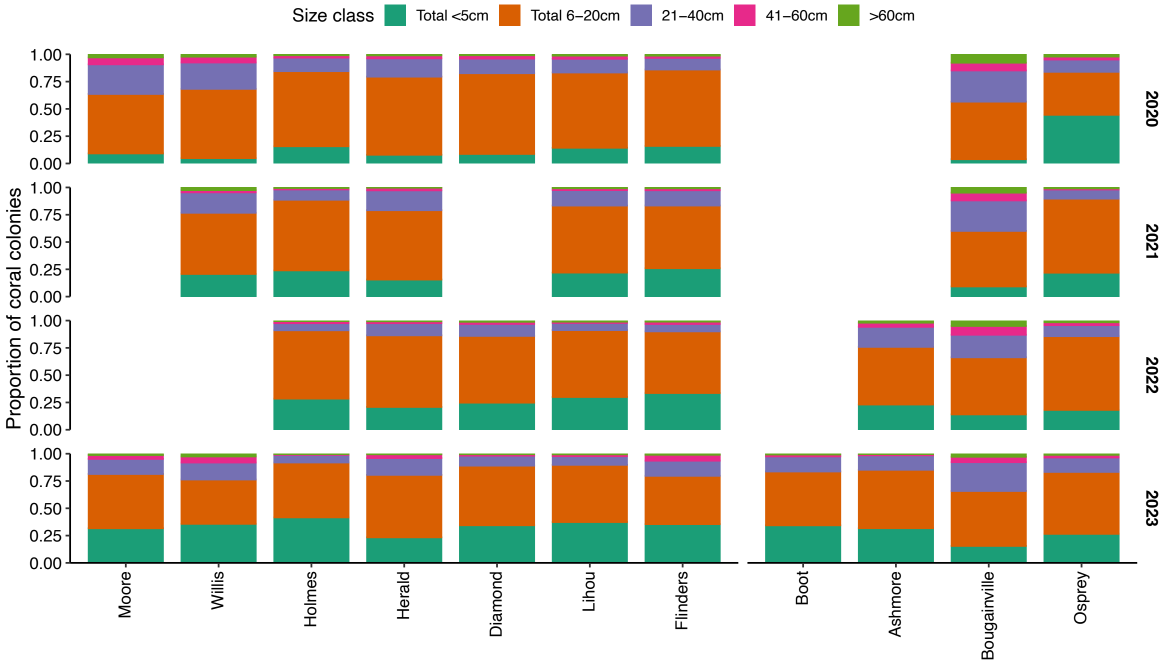


Figure 4.38 Proportion of coral colonies within each size class at 11 reefs within the Coral Sea Marine Park from 2020 to 2023.

4.5.2 Coral condition

In 2023, the vast majority (98.0%) of corals surveyed at each reef were healthy (Figure 4.39). Consistent with previous surveys (i.e., 2018-2022), the proportion of colonies exhibiting signs of injury (5-100% recent mortality) as a result of various stressors was low (1.2%) in 2023 (Burn et al. 2022; Figure 4.39). This low prevalence of injury in the CSMP, is in marked contrast to estimates from the GBRMP where more than half of all colonies were found to exhibit signs of injury (*Acropora:* 71%, branching *Pocillopora*: 59%; encrusting *Montipora*: 85%; *Porite*s: 92%: Pisapia et al. 2016). While some of this difference is likely attributable to the differences in the definition of ‘injury’ with our surveys only considering recent injuries (within the previous 4-6 weeks), it suggests levels of background injury and partial mortality are low across the CSMP.

There was also very little evidence of thermal stress (i.e., bleaching) across the eleven reefs surveyed in the central and northern CSMP, with less than 1% of colonies surveyed showing signs of bleaching (pale – recent mortality) (Figure 4.39). Across the eleven reefs surveyed the percent of coral colonies showing signs of bleaching (pale – recent mortality) ranged from 0.2% at Herald Cays to

1.2% at Ashmore Reef (Figure 4.39). Signs of bleaching was very low among coral taxa, with the majority of bleaching stress manifesting as the paling of colonies, rather than the complete loss of zooxanthellae from the tissue (Figures 4.40). The extremely low level of bleaching observed during our 2023 surveys, coupled with the negligible heat stress experienced over much of the CSMP during early (Jan-Apr) 2023, suggests these reefs are unlikely to experience any significant bleaching in 2023. Assuming this holds, it will be the first year in the past four years, and only the third year in the last eight years (i.e., since 2016) that reefs in the CSMP have not experienced severe bleaching.

Given the ongoing and predicted future effects of climate change, further heat stress events within the CSMP are inevitable, and as such continued monitoring will be critical to not only quantify the impacts and potential recovery of coral and fish populations, but also to understand the capacity of coral and fish populations to adapt to changing environmental conditions.

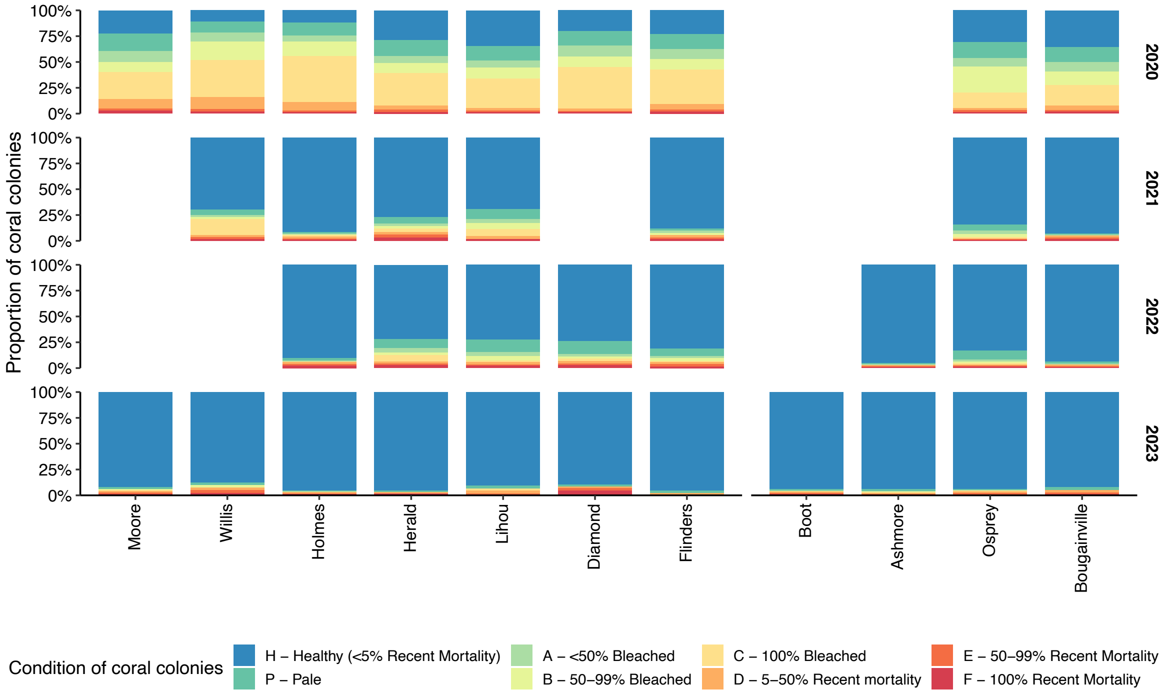


Figure 4.39 The proportion of coral colonies in each of eight health categories from 'healthy' to 'recently dead' recorded at 11 reefs within the Coral Sea Marine Park from 2020 to 2023. Note: not all reefs were surveyed in each year, with 11 reefs surveyed in 2023.

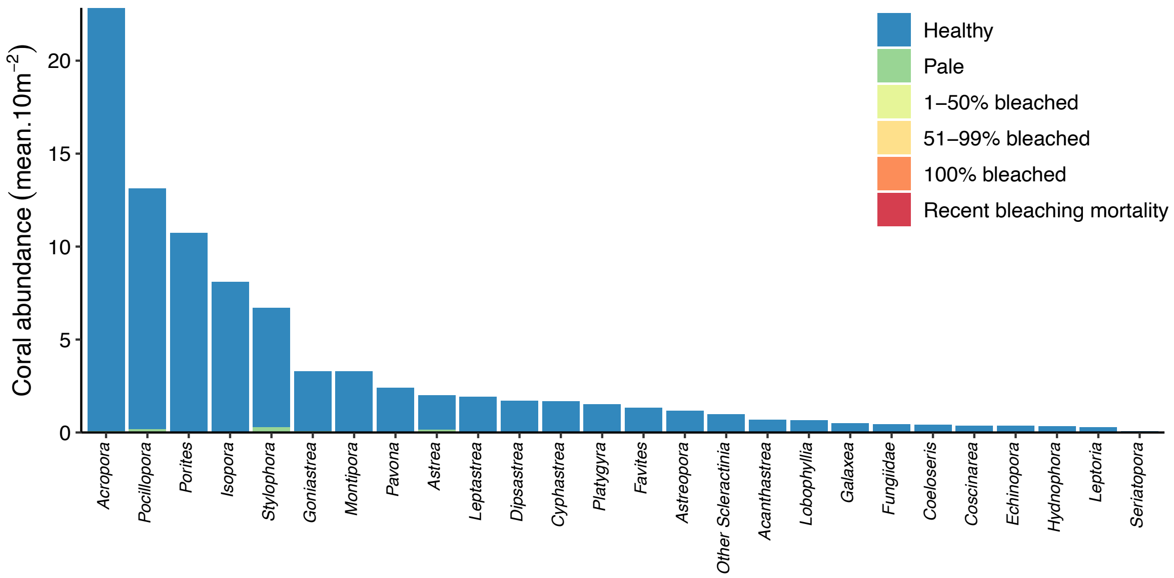


Figure 4.40 Mean density of coral colonies (per 10m^2) in the 26 most common scleractinian genera (including a pooled ‘other Scleractinia’ category) in each of six bleaching health categories from ‘healthy’ (blue) to ‘recent bleaching mortality’ (red) observed at sites across 11 reefs in the CSMP during February – March 2023.

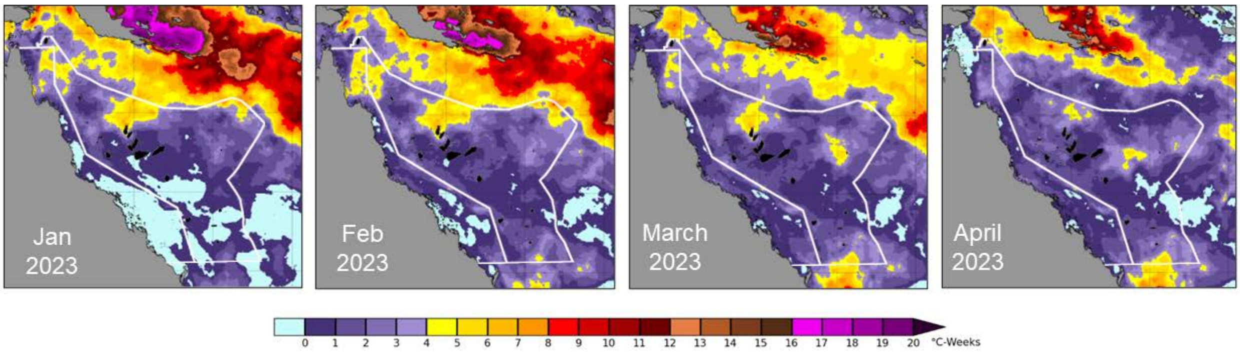


Figure 4.41 Mean monthly maximum degree heating weeks (DHW) in the Coral Sea Marine Park for January - April 2023. Images produced using the NOAA CRW 5km product v3.1

4.5.3 Juvenile corals

A total of 12,733 juvenile corals (≤5cm diameter; Rylaarsdam 1983) were recorded across the 11 reefs surveyed in the CSMP in 2023, equating to a mean density of 44.2 juvenile corals per 10m^2, and representing an increase in the densities of juvenile corals reported in previous years (2020: 15 juvenile corals per 10m^2; 2021: 16.4 juvenile corals per 10m^2; 2022: 23.1 juvenile corals per 10m^2). Some caution needs to be applied when comparing across all reefs surveyed, rather than those that have been resurveyed in multiple years. The higher overall density of juvenile corals in 2023 was largely driven by the higher densities recorded at Ashmore and

Boot Reefs in the far north of the CSMP (Figure 4.42), and likely reflects their proximity and hence connectivity with reefs of the Torres Straits and Eastern Fields (PNG). Overall, the mean densities of juvenile corals were 2-fold higher in the northern CSMP (60.1 juvenile corals per $10m^2$) than the central CSMP (29.6 juvenile corals per $10m^2$). There was however considerable variation among individual reefs in each region, ranging from 13.5 to 76.0 juvenile corals per $10m^2$ at Bougainville Reef and Ashmore Reef, respectively, in the northern CSMP, and from 15.7 to 52.6 juvenile corals per $10m^2$ at Willis Islets and Holmes Reefs, respectively, in the central CSMP (Figure 4.42). Notably, the lowest density of juvenile corals recorded (13.5 juvenile corals per $10m^2$) was at Bougainville Reef, one of the ‘bright spot’ reefs within the CSMP.

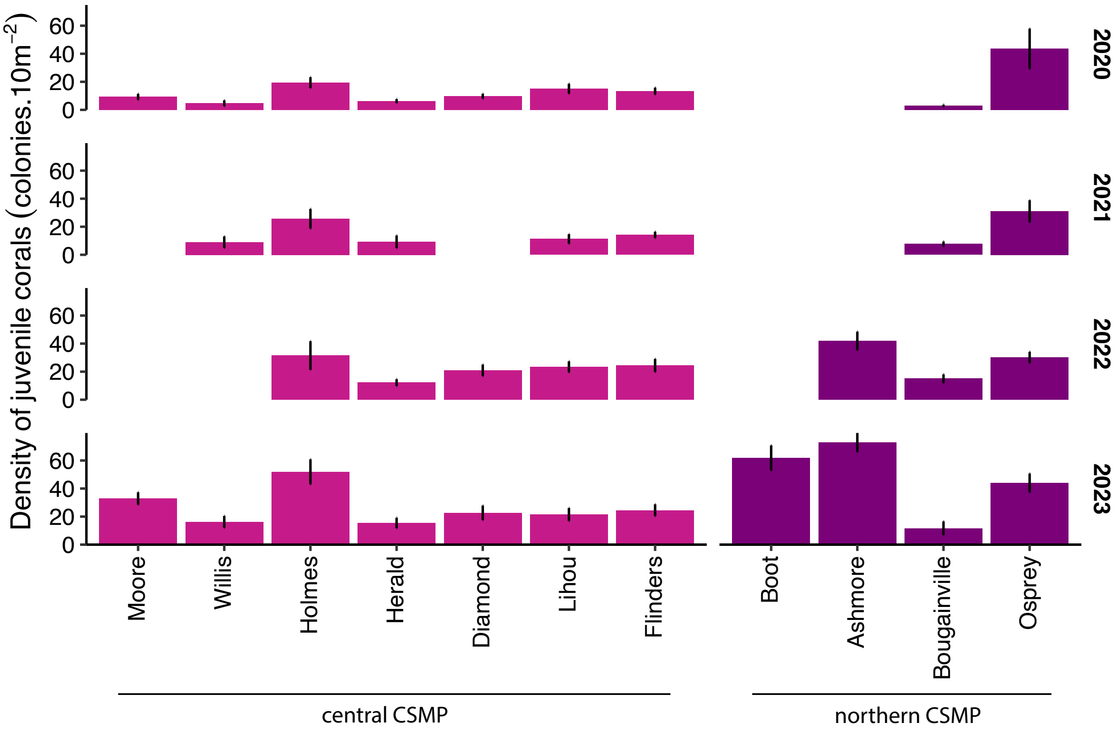


Figure 4.42 Mean (±SE) density of juvenile corals at each reef surveyed in the Coral Sea Marine Park from 2020 to 2022. Data are based on the number of juvenile corals (<5cm diameter) surveyed within 10 x 1 m belt transects at each site. Reefs are arranged into the southern, central, and northern CSMP and coloured by *a priori* regional assignments (following Figure 3.1). Densities of juvenile corals surveyed were always >1 colony.$10m^{-2}$, reefs with zero juveniles indicate they were not surveyed in that year.

Comparisons of the ten reefs that were surveyed at least twice in the past four years (2020-23), show the density of juvenile corals have increased in both regions between 2022 and 2023 (central CSMP from 24.4 to 29.6 juvenile corals per $10m^2$; northern CSMP from 27.5 to 30.0 juvenile corals per $10m^2$) and are now considerably greater than the densities recorded prior to and during the 2020 bleaching event (Hoey et al. 2020; Figure 4.44). These increases in the densities of juvenile corals have been largely consistent among reefs (Figure 4.45), and will aid in the recovery of the coral populations following the 2020, 2021, and 2022 bleaching events. It should be noted, however, that the majority of these juvenile corals likely settled onto these reefs during or prior to 2020 (e.g., Doropoulos et al. 2021). Consequently, the effects of the three back-to-back bleaching events (i.e., 2020, 2021, 2022) on adult coral brood stock, the production and settlement of coral larvae, and hence the replenishment of coral populations in the CSMP may yet to be fully realised. The next few years are likely to be critical in fully understanding the effects of these bleaching events on the recovery potential of these unique reefs.

The densities of juvenile corals in the central and northern regions of the CSMP in 2023 (average 4.4 juvenile corals per m^2), while greater than those reported in previous years are still at the lower end of density estimates for other regions (e.g., mid-shelf GBR: 6.1-8.2 juvenile corals per m^2, Trapon et al. 2013; Palmyra Atoll: 17.1 juvenile corals per m^2, Roth and Knowlton 2009; New Caledonia: 2 - 11.6 juvenile corals per m^2, Adjeroud et al. 2010). They are, however, greater than the densities of juvenile corals recorded following major bleaching events on oceanic reefs in the Maldives (2.9 juvenile corals per m^2, Pisapia et al. 2019).

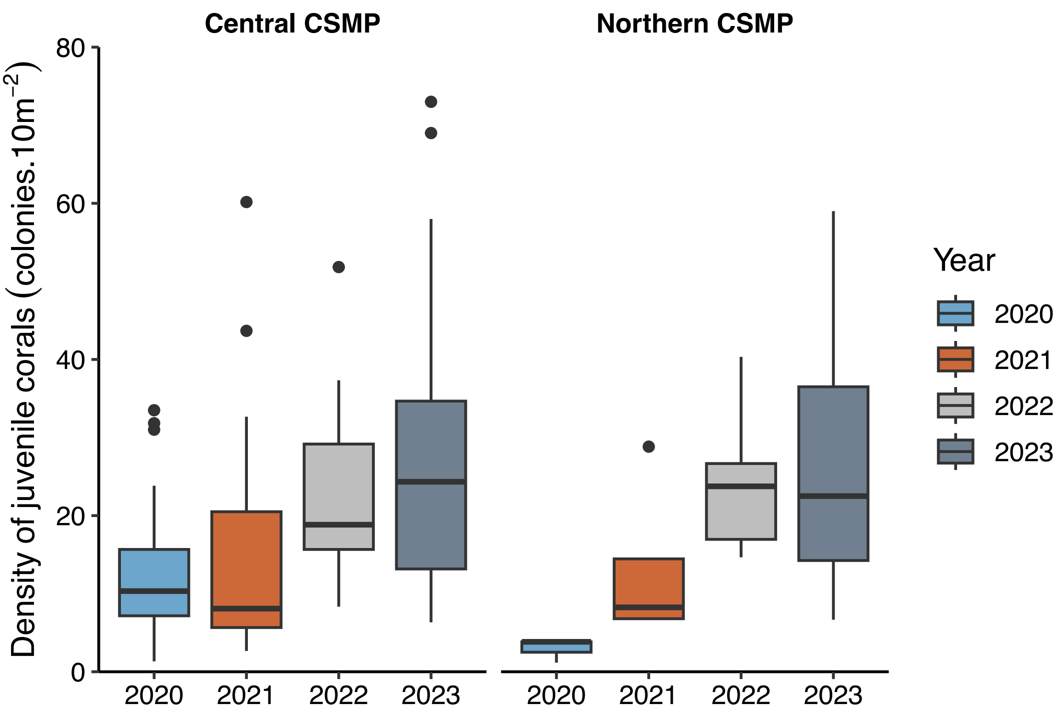
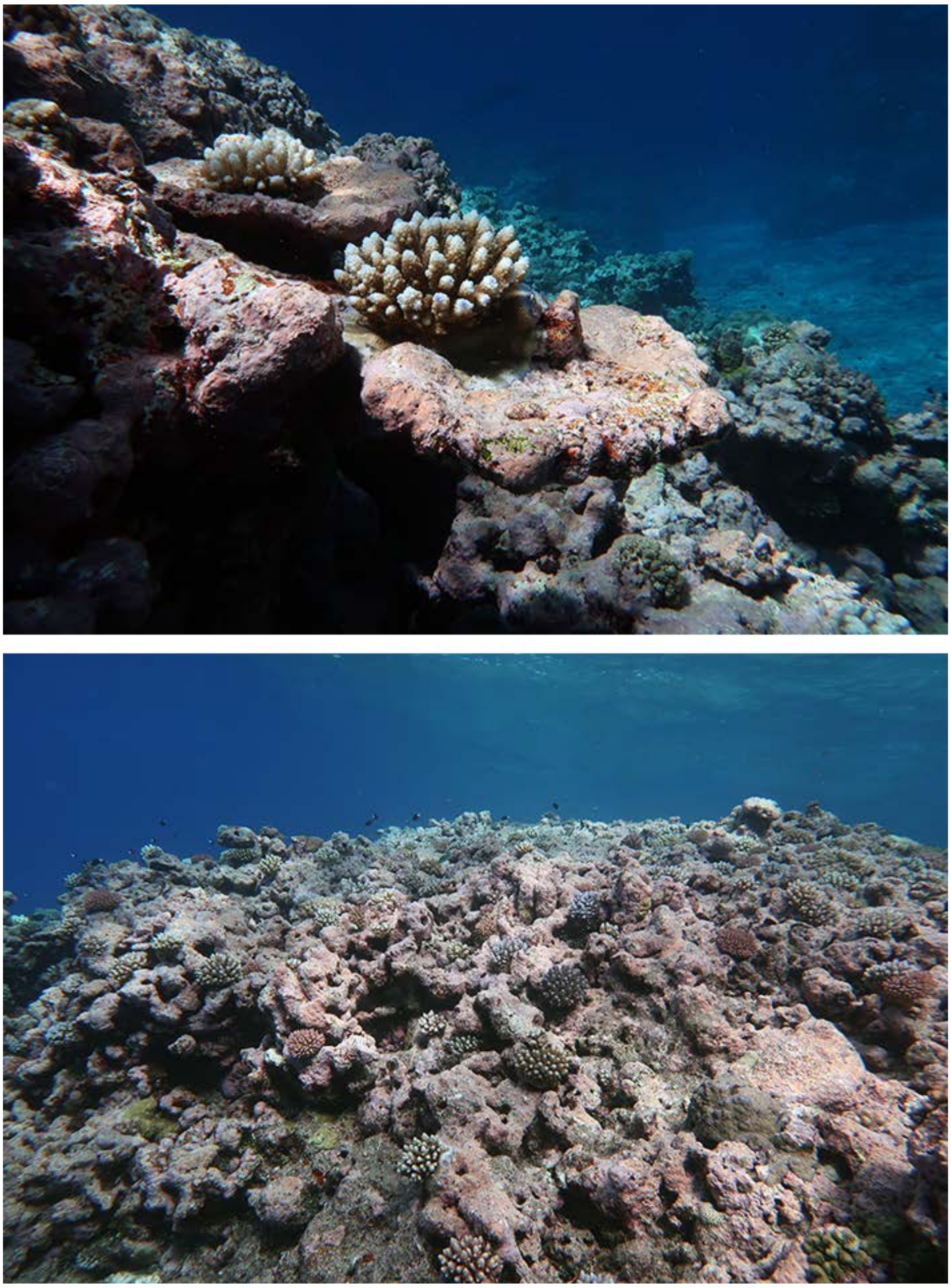


Figure 4.43 Spatial and temporal (2020-2023) variation in the density of juvenile corals (<5cm diameter) among two regions of the Coral Sea Marine Park. Data are based on surveys conducted at 32 sites across the 10 reefs in February-March of 2020, 2021, 2022 and/or 2023.



Images showing colonies of *Acropora* corals that have settled and established on areas of crustose coralline algae (CCA). Top: Two corymbose *Acropora* corals that have settled on dead tabular coral skeletons covered by CCA on Osprey Reef. Bottom: shallow reef crest habitat on Osprey Reef showing high cover of CCA's and a high density of juvenile and small *Acropora* colonies. Image credits: Andrew Hoey

The abundance of juvenile corals on a reef is a product of the supply and successful settlement of larvae, together with the survival and growth of newly settled corals. In the CSMP, larval supply from external sources (i.e., other reefs) is likely to be limited by the isolation and limited connectivity among reefs, with reefs relying largely on locally produced larvae for the replenishment of coral populations (i.e., self-recruitment; Gilmour et al. 2013). Following major disturbance events (e.g., mass bleaching) that cause extensive mortality of corals, local production of coral larvae is impeded due to the mortality of brood stock, and reduced fecundity as energy is partitioned away from reproduction and toward growth and colony repair (Hughes et al. 2019; Frisch et al. 2019). Continued monitoring of the juvenile assemblages in the CSMP will be critical to understand the full effects of the 2020, 2021, and 2022 bleaching events on the replenishment of coral populations and the future recovery of these isolated and unique reefs.

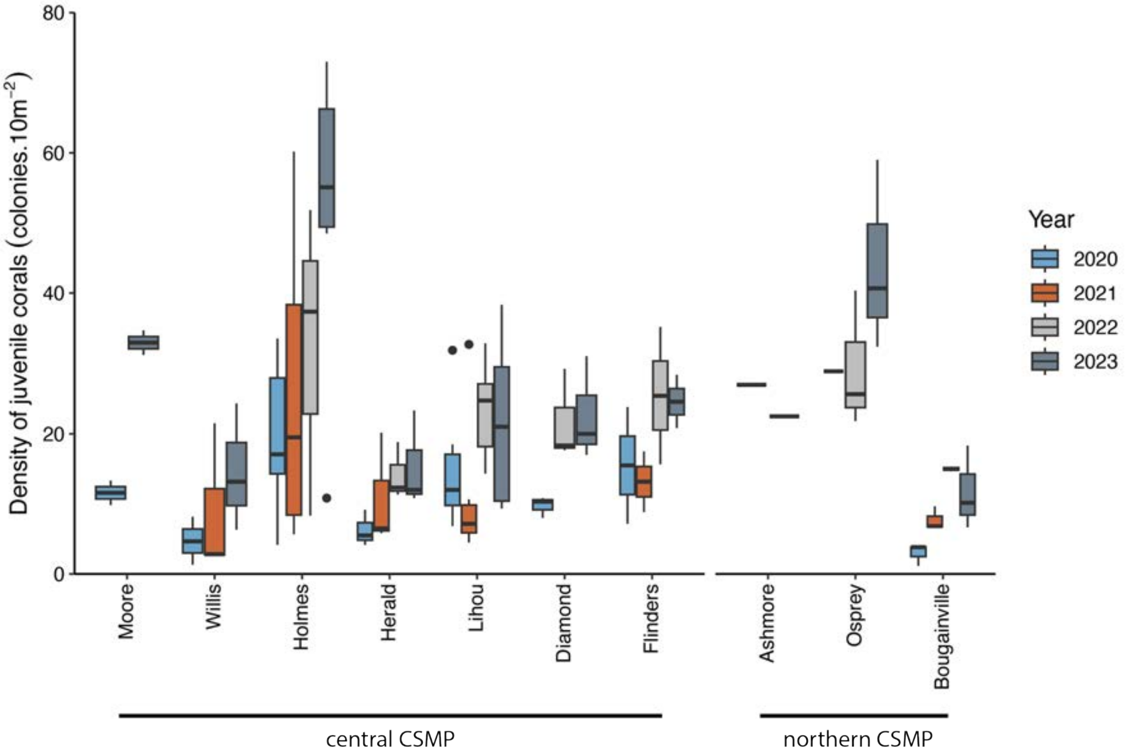


Figure 4.44 Spatial and temporal (2020-2023) variation in the density of juvenile corals (<5cm diameter) among 10 reefs within the Coral Sea Marine Park. Data are based on surveys conducted at 32 sites across the 11 reefs in February-March of each year.

4.6 ROV surveys – preliminary observations

Preliminary analysis of the ROV video surveys conducted on this voyage revealed the presence of three 'new' fish species within the CSMP, and represent significant range extensions for these species. ROV surveys conducted on this voyage confirmed the presence of Randall's Tilefish (*Hoplolatius randalli*) at East Diamond Islet, and Lihou, Ashmore and Boot Reefs. Individuals thought to be *H.randalli* were initially recorded by ROV at Lihou reef during ROV surveys in 2021, however, exact identification could not be made from the footage at this time. A total of eight individuals have now been recorded at reefs spanning the northern and central CSMP (Ashmore, Boot and Lihou Reefs and East Diamond Islet), all at depths below 50m. These observations by ROV in the CSMP represent the southernmost occurrence records for the species and expand the known extent of occurrence for *H. randalli* by almost 10 degrees of latitude (Figure 4.45a).

The ROV surveys on the 2023 voyage also recorded two new species that have not previously been recorded in shallow or deep habitats of the CSMP: the Harlequin hind (*Cephalopholis polleni*) and the spotted soapfish (*Pogonoperca punctata*). *Cephalopholis polleni,* previously only known in Australian waters from the Cocos (Keeling) and Christmas Islands in the Indian Ocean, was observed at Osprey reef at 97m (Figure 4.45b). *Pogonoperca punctata,* previously recorded from the Northern Territory and also the Cocos (Keeling) and Christmas Islands in Australian waters, was recorded at East Diamond Islet, 53m (Figure 4.45c). Both *C. polleni* and *P. punctata* are known from the wider Indo-Pacific region but these observations in the CSMP represent the southernmost records of these species.

ROV surveys in 2021 discovered several areas of very high coral cover at depth, including a large area of high coral cover (70-80%) at depths between 70 -80m adjacent to Edna Cay, Lihou Reef. Surveys conducted during this voyage located at the same site confirmed the persistence of this extensive Mesophotic Coral Ecosystem (Figure 4.46). Quantitative analyses of the video and still imagery captured by the ROV is ongoing and will be included in subsequent reports.

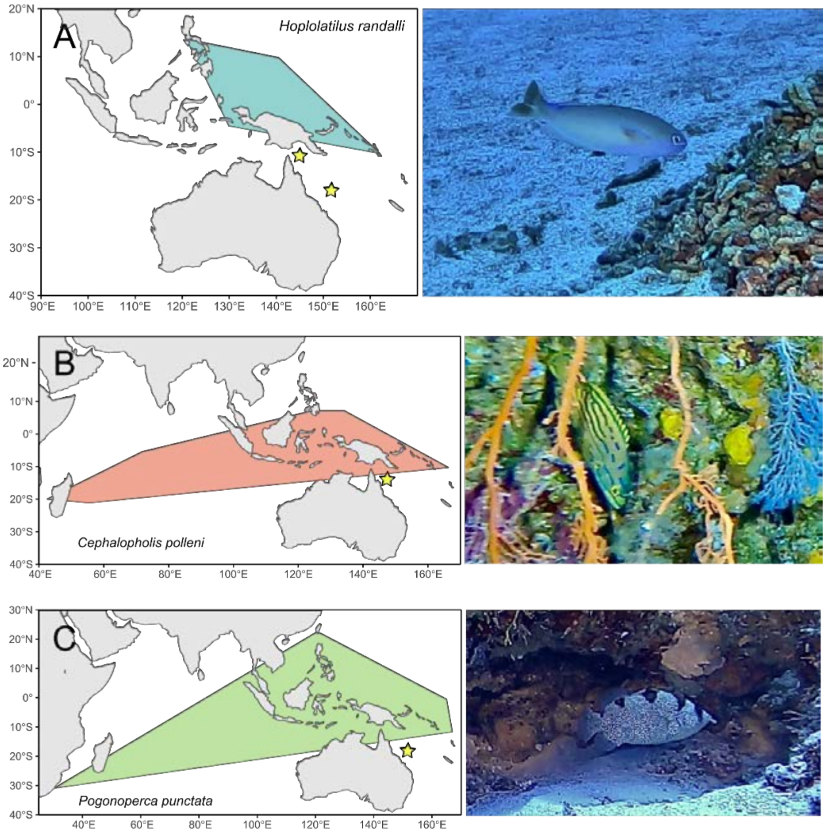


Figure 4.45 Current extent of occurrence plotted as colored hulls for a) *Hoplolatilus randalli,* b) *Cephalopholis polleni,* and c) *Pogonoperca punctata.* Occurrence data were obtained from Ocean Biodiversity Information System (www.obis.org) and the Global Biodiversity Information Facility (www.gbif.org). New observations of each species from the CSMP during 2023 ROV surveys are represented by yellow stars.

Figure 4.46 High coral cover mesophotic coral ecosystem surveyed at Edna Cay, 77m. This site was first discovered by ROV surveys in July 2021 and was resurveyed during the current voyage in February 2023.

4.7 Additional observations

4.7.1 Fish spawning aggregations

An aggregation of >20 large humphead maori wrasse (*Cheilinus undulatus*) was recorded at Bougainville site 1 on the morning of 21st February 2023. The majority of individuals were large (>90cm total length) and were positioned in open water 10-40m from the reef edge (Figure 4.47). This is a significant observation as this species usually occurs individually or in small groups (i.e., 2-3 individuals). It is also one of the largest species of teleosts (bony fish) that associates with coral reefs and is listed as *Vulnerable* on the International Union for Conservation of Nature (IUCN) Red List.

Figure 4.47 Several large Humphead Maori Wrasse (*Cheilinus undulatus*) aggregating in open water 10-40m off the reef edge at Bougainville Site 1 on the 21st February 2023. Image credit: Andrew Hoey

4.7.2 Vessel Moorings

Visual inspections were made of any existing vessel moorings sighted on Holmes and Bougainville Reefs. Where possible photographs were taken of the mooring lines, and the GPS coordinates of the moorings recorded. Two moorings were recorded and inspected at Holmes Reef and one mooring at Bougainville Reef. The moorings generally consisted of a length of chain or rope that passed through holes in the reef and was secured back onto itself. The mooring lines themselves were of variable condition (Figure 4.48).

4.7.3 Debris

Several large square 'quadrats' were observed on the reef slope at Holmes site 1 on the 20th February 2023. The quadrats were constructed of 2-3m lengths of PVC tube (orange electrical conduit) and secured with cable ties, with one of the quadrats having a temperature logger attached (Figure 4.49). These quadrats have been observed at this site previously, and given the level of fouling appear to have been in place for several years. The quadrats are in generally poor condition and should be considered for removal on future voyages.

Figure 4.48 Images of mooring lines attached to large bommies at Holmes Reef. Image credits: Andrew Hoey

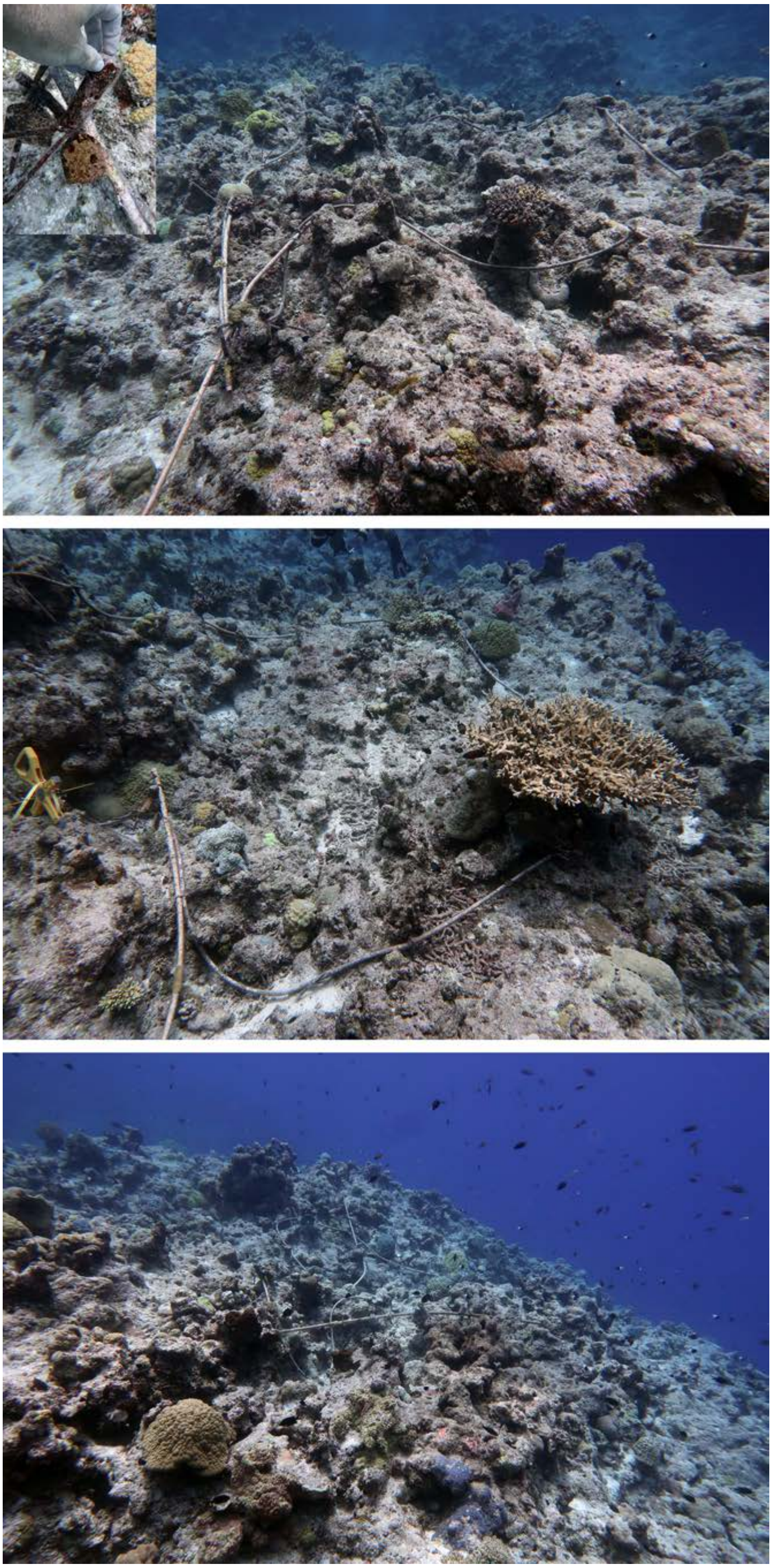


Figure 4.49 Large abandoned quadrats observed on the reef slope at site 1 on Holmes Reef, 20th February 2023. Image credits: Andrew Hoey

5 Conclusions

Globally, coral reefs are being increasingly exposed to the effects of climate change, with climate-induced coral bleaching now recognised as the foremost threat to coral reefs globally (Hughes et al. 2017). The severity and frequency of marine heatwaves, and associated bleaching of corals, have increased over recent decades, with the likelihood of mass-coral bleaching events occurring in any given year now being three-fold higher than prior to 2000 (Hughes et al. 2018). The ongoing and predicted future effects of climate change on reefs have led to concerns that some reefs will become overgrown by macroalgae and/or approach critical thresholds at which key processes are disrupted (Wilson et al. 2006; Graham et al. 2015). While isolated reef systems, such as those in the CSMP, are often described as being 'pristine' or 'near pristine' due to their limited exposure to local anthropogenic pressures (e.g., fishing, terrestrial run-off) relative to more accessible coastal or inshore reefs (e.g., McCauley et al. 2010; Graham and McClanahan 2013), the effects of climate change are pervasive. Indeed, four major coral bleaching events have been recorded in the CSMP in the past seven years (2016, 2017, 2020, and 2021; Harrison et al. 2018, 2019, Hoey et al. 2020, 2021, 2022). The two most recent bleaching events (i.e., 2020 and 2021) were the most severe and widespread, and led to a 52% decline in coral cover in shallow (<15m depth) reef habitats throughout the CSMP. There was, however, considerable variation in the decline in coral cover among regions, reefs, and sites within reefs (Hoey et al. 2022). Importantly, the five 'bright spot' reefs (Hoey et al. 2020) appeared to be less adversely affected by recent bleaching events than other CSMP reefs. Assessing the potential recovery of shallow water coral assemblages following recent bleaching events, any ongoing effects of coral loss on associated fish and invertebrate communities, are critical to better understand the dynamics, and factors that contribute to the performance, of the five 'bright spot' reefs, and the longer-term health of this unique reef system as a whole.

The surveys conducted in February-March 2023 under this project revealed further declines in coral cover in the central, but not northern, CSMP. While only low-moderate levels of bleaching were recorded across central CSMP reefs in February and March 2022 (11.9% of colonies), the central and eastern region of the Queensland Plateau in the central CSMP (including Herald Cays, Diamond and Willis Islets, and Lihou and Moore Reefs) was exposed to seawater temperatures above those expected to cause bleaching-induced mortality (>6 DHW) in March-April 2022, with some areas exposed to 8-11 DHW (Figure 4.6). In the absence of any other major disturbance, the observed declines in coral cover on central CSMP reefs in 2023 are most likely attributable to elevated temperatures experienced in March-April 2022, and represents the fifth major bleaching event in the CSMP in the past seven years. To our knowledge this is the first record of three consecutive bleaching events on coral reefs globally.

The observed declines in coral cover on the central CSMP reefs in 2023 (6% decline) were not as great as may have been expected based on recorded levels of DHW (i.e., 8-11 DHW) in March-April 2022. DHW combines the intensity and duration of heat stress experienced during the previous 3 months into one single number. It is a strong predictor of bleaching with DHW >4 likely to lead to significant bleaching, and DHW>8 likely to lead to significant mortality, especially in more sensitive species (Hughes et al. 2017). The relatively low incidence of mortality likely reflects a shifted baseline toward more bleaching resistant coral communities due to the loss of thermally sensitive species following the four previous bleaching events (i.e., 2016, 2017, 2020, and 2021), and as such the severity of this bleaching event should not be underestimated. Similar changes in the incidence of bleaching in response to heat stress were observed following the 2016 bleaching event on the GBR, with reefs exposed to 8-9 DHW having >90% probability of severe bleaching in 2016, compared to only a 50% probability for reefs exposed to the same heat stress in 2017 (Hughes et al. 2019). Variation in heat tolerance among corals has also been linked to a range of factors, including annual temperature ranges, the rate of warming, the frequency of, and prior exposure to, heat stress events (e.g., Ainsworth et al. 2016; Jurriaans and Hoogenboom 2020; Marzonie et al. 2023).

The most recent decline in coral cover on central CSMP reefs, albeit relatively small, compounded on previous declines due to the 2020 and 2021 bleaching events. The net effect of these three consecutive bleaching events is a 66.9% decline in shallow water (<15m) coral cover across the central CSMP, with several reefs in the central

CSMP having levels of coral cover that are approaching or below critical thresholds (<10% cover) that have been shown to disrupt key processes and the maintenance of biodiversity and ecosystem functions in other reef systems (Wilson et al. 2006; Graham et al. 2015; Pratchett et al. 2021). Despite the extensive coral loss there have been no concomitant increases in macroalgae, rather the dead coral skeletons appear to have been rapidly colonised by crustose coralline algae (CCA). This differentiation is critical, as increases in macroalgal biomass can break key ecological feedbacks, further suppressing the recovery of coral populations (e.g., Hoey and Bellwood 2011; Van de Leemput et al. 2016; Johns et al. 2018). In contrast, CCA's are a critical component of healthy reef ecosystems, contributing to reef calcification and stabilisation of the reef framework (e.g., Teichert et al. 2020; Cornwall et al. 2023), promoting the settlement and survival of coral larvae (e.g., Harrington et al. 2004; Abdul Wahab et al. 2023), and inhibiting the settlement and colonisation of fleshy fouling organisms (Littler and Littler 2013).

Current levels of coral cover on central CSMP reefs (mean: 10.0%; range: 6.0-19.0%) are generally greater than that of historical surveys of isolated bommies within the lagoons at Herald Cays, Chilcott Islet and Lihou Reef (1-5% in 2003 to ~6% in 2007; Oxley et al. 2003; Ceccarelli et al. 2008), indicating that such low coral cover may not be uncommon on these reefs. While it is currently unknown if coral cover <10% will disrupt key processes and limit the recovery of these central CSMP reefs, any further coral loss is likely to have lasting consequences on the health and resilience of these unique ecosystems. The next few years may be critical in determining whether coral populations and coral cover on these reefs recovers, or collapses, as well as the implications for reef fish and invertebrate communities.

The biomass of reef fishes has steadily declined on central and northern CSMP reefs since 2020, and is likely related (at least to some degree) to the reductions in coral cover during this period (2020-2023). The initial decrease in reef fish biomass (2020-2022) was largely driven by declines in small-bodied planktivorous fishes (e.g., damselfishes), and corallivorous butterflyfishes that are reliant on live coral for shelter and food, respectively. These fishes are often the first and most adversely affected groups following coral loss (e.g., Pratchett et al. 2011, 2014). Declines in the biomass of grazing fishes (primarily surgeonfishes) were also evident from 2020-

2022. While the biomass of corallivorous and planktivorous fishes remained relatively unchanged between 2022 and 2023, the biomass of grazing fishes (primarily surgeonfishes) continued to decline and is now >60% lower than 2020 levels. Grazing fishes are widely viewed as a critical functional group on coral reefs preventing algal overgrowth and maintaining a healthy balance between corals and algae (e.g., Bellwood et al. 2006b; Hoey and Bellwood 2009, 2011; Rasher et al. 2013), and herbivorous fishes of the Queensland Plateau are recognised as a Key Ecological Feature in the CSMP. The continued and sustained declines in the biomass of grazing fishes are difficult to reconcile as several previous studies have reported substantial increases in the abundance and/or biomass of herbivorous fishes following large-scale bleaching-induced coral mortality (e.g., Adam et al 2011; Gilmour et al. 2013). Such increases have generally been related to an increase in the availability of their preferred feeding substrata (i.e., algal turf assemblages that rapidly colonise dead coral skeletons; Diaz-Pulido and McCook 2002), and subsequent increases in the growth rates of individual fishes (Taylor et al. 2020). The recorded declines in the biomass of grazing fishes may be related to the physiological response of these fishes to heat stress (Stuart-Smith et al. 2018), although this seems unlikely as other groups (e.g., piscivores) were largely unaffected. It seems more likely that the rapid colonisation of dead coral skeletons by CCA, as opposed to algal turf assemblages, may be limiting the food available for these fishes. Further dedicated investigation into the diet and fitness of these fishes on CSMP reefs is required to identify the mechanism/s for these declines.

Despite these declines, the biomass of reef fishes (a key indicator of reef health) recorded across all reefs in the CSMP remained high (range: 309 – 3,684 kg per hectare; mean = 1,807 kg per hectare) relative to coral reef environments globally (Cinner et al. 2016) and higher than estimates of unfished biomass for coral reefs globally (1,000-1,250 kg per hectare; MacNeil et al. 2015; McClanahan 2018). This high biomass, especially of sharks and large-bodied piscivores, likely reflects the isolation and limited fishing pressure on CSMP reefs.

Importantly, coral cover and fish biomass on previously identified ‘bright spot’ reefs (Hoey et al. 2020) remained relatively high. For example, coral cover on Moore Reefs (19%) is the highest of the seven central CSMP surveyed, and almost double that of the average for central CSMP reefs (10.0%); this is despite *ca.* 50% decline in coral since 2020. Similarly coral cover at the three other ‘bright spot’ reefs surveyed remained considerably higher than the regional average (Ashmore: 35.2%; Boot: 22.8%; Bougainville: 31.3%). Unfortunately, differences in the survey frequency at these ‘bright spot’ reefs make temporal comparisons difficult. For example, Moore Reefs was surveyed in 2020 and 2023, Boot Reef in 2018 and 2023, Ashmore Reef in 2018, 2022, and 2023, and Bougainville Reef annually since 2018. As such understanding their response to, and recovery from, disturbance (namely bleaching), and how this compares to other reefs is difficult to isolate.

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, as well as the responses, recovery and resilience of individual reefs and reef systems. Reefs in the CSMP have experienced five major coral bleaching events over the past 7 years (i.e., 2016, 2017, 2020, 2021, 2022), and three consecutive bleaching events in the past three years. To our knowledge this is the first time major bleaching events have been recorded over three successive years on reefs anywhere. While previous research has highlighted the importance of reef geomorphology, reef size, habitat type, habitat complexity, and connectivity in shaping the status and health of reef communities in the CSMP (Ceccarelli et al. 2013), it will be increasingly important to understand how interactions between these contemporary factors and ongoing and future effects of climate change shape these unique reefs into the future.

5.1 Recommendations

Regular comprehensive monitoring of coral reef environments in the CSMP is essential to understand its structure and function, ecological significance, and changing health and condition, especially in light of the increasing incidence of heat stress events. Annual monitoring of CSMP reefs since 2018 has greatly improved our understanding of the unique nature of these reefs, and importantly identified drivers of change (i.e., major bleaching events). 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 status of reefs (i.e., coral cover and population sizes of fishes and non-coral invertebrates), quantifying demographic processes of key reef taxa (e.g., recruitment, growth and mortality of corals, coralline algae and fishes) among reefs and regions within the CSMP will greatly improve our understanding of the vulnerability, recovery potential, and resilience of shallow coral reef environments in the CSMP to ongoing and future disturbances, as well as potential interactions among increasingly frequent and more intense heat stress events. Continued monitoring of both coral settlement and the density of juvenile corals will be critical to understand the potential replenishment of coral populations following repeated bleaching events, as well as local stock-recruitment relationships for shallow water corals within the CSMP.

To effectively monitor the potential recovery of coral populations and communities, as well as any changes in the associated fish and invertebrate communities following major disturbances, we recommend annual monitoring of benthic (coral, macroalgae, CCA), fish, sea snakes and macro-invertebrate communities using the same methods and sites as previous (2018-23) surveys. The consistency of survey method is critical to ensure any changes are due to changes in the ecological communities, rather than an artefact of any difference/s in the survey methods. In the absence of any major environmental disturbances the time between recurrent surveys of individual reefs could be extended to 2-5 years, however this appears unlikely given predicted increases intensity of disturbances affecting reefs globally (Hughes et al. 2018), and as highlighted by the three successive bleaching events experienced in the CSMP in the last 3 years. Given this increased incidence of disturbance, coupled with the logistical constraints of working in the CSMP (i.e., isolation and exposure), regular (i.e., annual or biennial) surveys of at least a subset of representative reefs are critical. We recommend a subset of 10-12 representative reefs should be surveyed each year, with all 22 CSMP reefs to be re-surveyed every 3-5 years. These representative reefs should prioritise the five ‘bright spot’ reefs (i.e., Ashmore, Boot, Bougainville, Moore and Mellish Reefs), as well as reefs that are adjacent to the ‘bright spot’ reefs and/or on-route between reefs to facilitate comparisons and maximise the available vessel time. With these considerations in mind, we recommend as a minimum the following 11 reefs be surveyed annually Saumarez, and Kenn Reefs in the southern CSMP; Flinders, Holmes, Lihou, Marion, Moore and Mellish Reefs in the central CSMP, and Bougainville and Osprey Reefs in the northern CSMP. We do not include Ashmore and Boot Reefs here given their location in the far north of the CSMP, and hence the addition travel time and cost of accessing these reefs.

On reefs where suitable overnight anchorages are available, a minimum of 2 days should be allocated to each of the representative reefs (weather and conditions permitting) to allow for surveys of additional sites and habitats (e.g., weather exposed aspects) and targeted research and collections. In addition to the regular surveys of representative reefs, we recommend intensive sampling (5-7 days) of a single reef during each voyage, with a different reef to be surveyed in consecutive voyages. Spending 5-7 days at a reef would allow *ca.* 15-28 sites to be surveyed compared to the current 3-5 sites per reef, thereby providing a much more comprehensive understanding of the status and health of each reef, as well as allowing questions such as the following to be addressed: do reef areas that are adjacent to channels have higher coral cover and/or densities of juvenile corals than those in sheltered back reef or lagoonal environments? During the 2023 voyage, ~2 weeks were spent at Ashmore and Boot Reefs which allowed surveys to be conducted over a much greater range of habitats and sites, including the exposed south-east aspect of these reefs, and within the enclosed lagoon at Boot Reef. The sites on the exposed south-east aspect of Ashmore and Boot Reefs have not been observed or surveyed by western scientists previously, and were found to support rich coral communities and possibly the greatest reef fish biomass recorded across all of our surveys since 2018. Greater detail of the surveys on these reefs will be provided in the report for the *Our Marine Parks Round 3 Grant – The Jewel in the Coral Sea: The cultural and ecological significance of Ashmore and Boot Reefs* that funded this aspect of the voyage.

Dedicated monitoring of deep reef and non-reef (i.e., soft-bottom, macroalgae beds, seagrass) habitats using remotely operated underwater vehicles (ROVs) should be continued and expanded. These deep habitats occupy a greater area than shallow reef habitats throughout the CSMP, are largely unexplored and likely less impacted by disturbance. Continued and expanded monitoring of these habitats using ROVs, as well as other technologies (e.g., towed videos) will provide a more comprehensive understanding of the composition and health of these unique deep water habitats, increase our understanding of potential links between shallow reef and deep reef and non-reef habitats, while also maximising the use of available berths on the vessel. Repeat surveys of deep habitats should be prioritised and are critical to understand the dynamics of these habitats and their response to disturbance, while spending more time at each reef would allow the opportunity to conduct dedicated surveys of species of commercial and/or conservation interest (e.g., sea cucumbers and giant clams) that are not adequately captured through current surveys of shallow reef habitats, as well as identify novel habitats (e.g., *Halimeda* bioherms), and important fish settlement and nursery and habitats.

Several projects aimed at understanding potential variation in water temperatures, and the settlement and calcification rates of crustose coralline algae (CAA) between 'bright spot' and other reefs were initiated during the 2023 voyage, and coral settlement tiles will be deployed on some CSMP reefs in October 2023. These projects are aimed at better understanding key processes on CSMP and should be continued and expanded upon to include projects to quantify key demographic rates of corals and reef fish. Establishing fixed plots at a select number of sites and using high resolution photogrammetry to create 3-dimensional maps would allow the fate of individual coral colonies, and the topographic complexity of the habitat to be tracked through time. Repeating the 3-dimensional habitat mapping of sites mapped during the 2019-2020 voyages in the next 1-2 years would provide some insight into relative contribution of live corals versus the underlying reef matrix and coralline algae in providing habitat structure. These existing 3-dimensional maps were not created for fixed plots and were not of sufficient resolution to quantify the growth of individual corals. We also recommend dedicated research and collections to quantifying demographic rates (growth, mortality) for fish and identifying key settlement and nursery habitats. Ideally this would include grazing fish species so that the likely mechanism/s for the observed declines in this group following the recent bleaching events could be identified.

The maintenance and replenishment of populations, and the resilience of reef systems within the CSMP is largely dependent on the supply of larvae, and hence the connectivity among and within reefs in the CSMP and adjacent regions (i.e., GBRMP, Temperate East Marine Parks Network, New Caledonia, Vanuatu, Solomon Islands and Papua New Guinea). Dedicated collections of animal tissues across these regions, and subsequent genetic analyses of these samples are required to understand patterns of connectivity, and how they differ among taxa. We recommend focusing on several fish taxa that vary in their dispersal potential (i.e., reproductive mode, pelagic larval duration, body size), as well as macro-invertebrates of potential commercial value (i.e., sea cucumber, *Tridacna* clams). While corals are the foundation species of coral reefs, we would be hesitant to use corals as a focal taxon for investigating connectivity due to the taxonomic uncertainty and ongoing taxonomic revisions of this group (e.g., Huang et al. 2016).

The current scheduling of surveys for late summer-early autumn (i.e., February-March) is designed to capture the incidence and extent of bleaching. Indeed the 2020 surveys coincided with the peak in the heat stress, although surveys were conducted prior to the peak in heat stress in both 2021 and 2022. Targeting this period for the surveys limits the capacity to explore other important biological and ecological processes, especially those related to the spawning and settlement of corals, fishes and invertebrates. While 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., tilt current meters only record for ~3 months), the costs of running multiple dedicated voyages per year are likely prohibitive. We recommend that additional research and monitoring could be achieved by either making use of existing dive tourism expeditions to the CSMP, or combining with other planned voyages (e.g., CSMP Island Health).

Finally, surveys conducted over the past 5 years have highlighted 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 would be invaluable to better understand biological and ecological connections among these regions.

References

Abdul Wahab MA, Ferguson S, Snekkevik VK, McCutchan G, Jeong S, Severati A, Randall CJ, Negri AP, Diaz-Pulido G (2023) Hierarchical settlement behaviours of coral larvae to common coralline algae. *Scientific Reports* 13: 5795.

Adam TC, Schmitt RJ, Holbrook SJ, Brooks AJ, Edmunds PJ, Carpenter RC, Bernardi G (2011) Herbivory, connectivity, and ecosystem resilience: response of a coral reef to a large-scale perturbation. *PloS one* 6:e23717.

Adjeroud M, Fernandez JM, Carroll AG, Harrison PL, Penin L (2010) Spatial patterns and recruitment processes of coral assemblages among contrasting environmental conditions in the southwestern lagoon of New Caldedonia. *Mar Poll Bull* 61: 375-386

Ahmad W, Hill GJ. (1994) A classification strategy for mapping trochus shell habitat in Torres Strait, Australia. *Geocarto International* 9:39-47.

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

Ainsworth TD, Heron SF, Ortiz JC, Mumby PJ, Grech A, Ogawa D, Eakin CM, Leggat W (2016) Climate change disables coral bleaching protection on the Great Barrier Reef. *Science* 352:338-42.

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

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

Beaman RJ (2020) High-resolution depth model for the Great Barrier Reef and Coral Sea – 100 m. Geoscience Australia, Canberra. http://dx.doi.org/10.26186/5e2f8bb629d07

Bellwood DR, Hughes TP (2001) Regional-scale assembly rules and biodiversity of coral reefs. *Science* 292: 1532-1535.

Bellwood DR, Hoey AS, Ackerman JL, Depczynski M (2006) Coral bleaching, reef fish community phase shifts and the resilience of coral reefs. *Global Change Biology* 12: 1587-1594.

Bellwood DR, Hughes TP, Hoey AS (2006b) Sleeping functional group drives coral-reef recovery. *Current Biology* 16: 2434-2439.

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

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

Burn D, Matthews S, Pisapia C, Hoey AS, Pratchett MS (2022) Changes in the incidence of coral injuries during mass bleaching across Australia’s Coral Sea Marine Park. *Mar Ecol Prog Ser* 682:97-109.

Burn D, Hoey AS, Matthews S, Harrison HB, Pratchett MS (2023). Differential bleaching susceptibility among coral taxa and colony sizes, relative to bleaching severity across Australia's Great Barrier Reef and Coral Sea Marine Parks. *Marine Pollution Bulletin* 191:114907

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

Cheal AJ, MacNeil MA, Cripps E, Emslie MJ, Jonker M, Schaffelke B, Sweatman H (2010) Coral–macroalgal phase shifts or reef resilience: links with diversity and functional roles of herbivorous fishes on the Great Barrier Reef. Coral reefs. 29:1005-15.

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

Coker DJ, Wilson SK, Pratchett MS (2014) Importance of live coral habitat for reef fishes. *Reviews in Fish Biology and Fisheries* 24:89-126.

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

Cornwall CE, Carlot J, Branson O, Courtney TA, Harvey BP, Perry CT, Andersson AJ, Diaz-Pulido G, Johnson MD, Kennedy E, Krieger EC (2023) Crustose coralline algae can contribute more than corals to coral reef carbonate production. *Communications Earth & Environment* 4: 105.

Davies PJ, Symonds PA, Feary DA, Pigram CJ (1989) The evolution of carbonate platforms of northeast Australia. In: Crevello PD, Wilson JL, Sarg JF, Read JF (Eds.) Controls on Carbonate Platform and Basin Development. SEPM Special Publications, Tulsa, pp. 233–258.

De'ath G, Fabricius KE, Sweatman H, Puotinen ML (2012) The 27-year decline of coral cover on the Great Barrier Reef and its causes. *Proc. Natl. Acad. Sci. USA* 109**:** 17995–17999.

Diaz-Pulido G, McCook LJ (2002) The fate of bleached corals: patterns and dynamics of algal recruitment. *Marine Ecology Progress Series* 232: 115-28.

Dietzel A, Bode M, Connolly SR, Hughes TP (2020) Long-term shifts in the colony size structure of coral populations along the Great Barrier Reef. *Proceedings of the Royal Society B* 287: 20201432.

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

Doropoulos C, Bozec YM, Gouezo M, Priest MA, Thomson DP, Mumby PJ, Roff G (2021) Cryptic coral recruits as dormant'seed banks': an unrecognised mechanism of rapid reef recovery. *Ecology* 103:e3621

Drew EA (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 CM (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.

Edmunds PJ, Carpenter RC (2001) Recovery of *Diadema antillarum* reduces macroalgal cover and increases abundance of juvenile corals on a Caribbean reef. *Proc. Natl Acad. Sci. USA* 98: 5067-5071.

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

Fabricius KE, De'Ath G, Puotinen ML, Done T, Cooper TF, Burgess SC (2008) Disturbance gradients on inshore and offshore coral reefs caused by a severe tropical cyclone. *Limnology and Oceanography* 53:690-704.

Frisch J, Drury C, Towle EK, Winter RN, Miller MW (2019) Physiological and reproductive repercussions of consecutive summer bleaching events of the threatened Caribbean coral Orbicella 114aveolate. *Coral Reefs* 38: 863-876

Galbraith G, McClure E, Barnett A, Cresswell B, Burn D, Huertas V, Pratchett MS, Hoey AS (2022) Diving into the Deep: the Unique Deep Habitats of the Coral Sea Marine Park. Report prepared for Parks Australia. pp. 162.

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

Gilmour JP, Smith LD, Heyward AJ, Baird AH, Pratchett MS (2013) Recovery of an isolated coral reef system following severe disturbance. Science 340(6128): 69-71

Glynn PW, Wellington GM, Birkeland C (1979) Coral reef growth in the Galapagos: limitation by sea urchins. *Science* 203: 47-49.

Glynn PW, Manzello DP (2015) Bioerosion and coral reef growth: a dynamic balance. Coral reefs in the Anthropocene. Pp. 67-97.

Graham NA, McClanahan TR (2013) The last call for marine wilderness? *Bioscience* 63: 397-402.

Graham NA, Wilson SK, 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 NA, Jennings S, MacNeil MA, Mouillot D, Wilson SK (2015) Predicting climate-driven regime shifts versus rebound potential in coral reefs. *Nature* 518: 94-97.

Harrington L, Fabricius K, De'Ath G, Negri A (2004) Recognition and selection of settlement substrata determine post-settlement survival in corals. *Ecology* 85: 3428-3437.

Harrison HB, Àlvarez-Noriega M, Baird AH, Heron SF, MacDonald C, Hughes TP (2019) Back to back coral bleaching events on isolated atolls in the Coral Sea. *Coral Reefs* 38:713-719.

Harrison HB, Àlvarez-Noriega M, Baird AH, 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.

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

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

Hoey AS, Bellwood DR (2010) Cross-shelf variation in browsing intensity on the Great Barrier Reef. *Coral Reefs* 29: 499-508.

Hoey AS, Bellwood DR (2011) Suppression of herbivory by macroalgal density: a critical feedback on coral reefs? *Ecology Letters* 14:267-73.

Hoey AS, Pratchett MS, 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 AS, Howells E, Johansen JL, Hobbs JPA, Messmer V, McCowan DM, Wilson SK, Pratchett MS (2016) Recent advances in understanding the effects of climate change on coral reefs. *Diversity* 8:1-12.

Hoey AS, Pratchett MS, Sambrook K, Gudge S, Pratchett DJ (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 AS, Harrison HB, Pratchett MS (2020) Coral Reef Health in the Coral Sea Marine Park – Surveys 2018-2020. Report prepared for Parks Australia

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

Hoey AS, McClure EC, Burn D, Chandler J, Huertas V, Creswell B, Galbraith G, Pratchett MS (2022) Coral Sea Marine Park Coral Reef Health Survey 2022. Report prepared for Parks Australia.

Huang D, Arrigoni R, Benzoni F, Fukami H, Knowlton N, Smith ND, Stolarski J, Chou LM, Budd AF (2016) Taxonomic classification of the reef coral family Lobophylliidae (Cnidaria: Anthozoa: Scleractinia). *Zoological Journal of the Linnean Society* 178:436-81.

Hughes TP (1994) Catastrophes, phase shifts, and large-scale degradation of a Caribbean coral reef. *Science* 265:1547-51.

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

Hughes TP, Anderson KD, Connolly SR, Heron SF, Kerry JT, Lough JM, Baird AH, Baum JK, Berumen ML, Bridge TC, Claar DC, et al. (2018) Spatial and temporal patterns of mass bleaching of corals in the Anthropocene. *Science* 359:80-3.

Hughes TP, Kerry JT, Baird AH, Connolly SR, Chase TJ, Dietzel A, Hill T, Hoey AS, Hoogenboom MO, Jacobson M, Kerswell A, Madin JS, Mieog A, Paley AS, Pratchett MS, Torda G, Woods RM (2019) Global warming impairs stock-recruitment dynamics of corals. Nature 568: 387-390

Hughes TP, Kerry JT, Baird AH, Connolly SR, Dietzel A, Eakin CM, Heron SF, Hoey AS, Hoogenboom MO, Liu G, McWilliam MJ, Pears RJ, Pratchett MS, Skirving WJ, Stella JS, Torda G (2018) Global warming transforms coral reef assemblages. Nature 556:492-496

Hughes TP, Kerry JT, Connolly SR, Baird AH, Eakin CM, Heron SF, Hoey AS, Hoogenboom MO, Jacobson M, Liu G, Pratchett MS. Ecological memory modifies the cumulative impact of recurrent climate extremes (2019). *Nature Climate Change* 9:40-3.

Humphries AT, McClanahan TR, McQuaid CD (2020) Algal turf consumption by sea urchins and fishes is mediated by fisheries management on coral reefs in Kenya. *Coral Reefs* 39:1137-1146

Johns KA, Emslie MJ, Hoey AS, Osborne K, Jonker MJ, Cheal AJ (2018) Macroalgal feedbacks and substrate properties maintain a coral reef regime shift. *Ecosphere* 9:e02349.

Jurriaans S, Hoogenboom MO (2020). Seasonal acclimation of thermal performance in two species of reef-building corals. *Marine Ecology Progress Series* 635:55-70

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

Kinch J, Purcell S, Uthicke S, 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.

Littler DS, Littler MM (2003) South Pacific Reef Plants. Offshore Graphics, Washington. pp. 331.

Littler MM, Littler DS (2013) The nature of crustose coralline algae and their interactions on reefs. Research and discoveries: the revolution of science through SCUBA

Marzonie MR, Bay LK, Bourne DG, Hoey AS, Matthews S, Nielsen JJV, Harrison HB (2023) Mild heatwaves increase acute tolerance in corals. *Global Change Biology* 29: 404-16.

MacNeil MA, Graham NA, Cinner JE, Wilson SK, Williams ID, Maina J, Newman S, Friedlander AM, Jupiter S, Polunin NV, McClanahan TR (2015) Recovery potential of the world’s coral reef fishes. *Nature* 520: 341-4.

McCauley DJ, Micheli F, Young HS, Tittensor DP, Brumbaugh DR, Madin EM, Holmes KE, Smith JE, Lotze HK, DeSalles PA, Arnold SN (2010) Acute effects of removing large fish from a near-pristine coral reef. *Marine Biology* 157:2739-50.

McClanahan TR (2018) Community biomass and life history benchmarks for coral reef fisheries. *Fish and Fisheries* 19:471-88.

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

McClanahan TR, Nugues M, 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: 237-54.

McCook LJ (1999) Macroalgae, nutrients and phase shifts on coral reefs: scientific issues and management consequences for the Great Barrier Reef. *Coral Reefs* 18:357-67

McWilliam M, Pratchett MS, Hoogenboom MO, Hughes TP (2020) Deficits in functional trait diversity following recovery on coral reefs. *Proc. R. Soc. B* 287: 20192628.

Morgan-Wall T (2023). *Rayshader: Create Maps and Visualize Data in 2D and 3D*. https://www.rayshader.com, https://github.com/tylermorganwall/rayshader

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

Oksanen J, Guillaume Blanchet F, Friendly M, Kindt R, Legendre P, McGlinn PD, Minchin R, O’Hara RB, Simpson GL, Solymos P, Henry M, Stevens H, Szoecs E, Wagner H (2018) vegan: Community Ecology Package. R package version 2.5-2.

Oxley WG, Ayling AM, Cheal AJ, Thompson AA (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 WG, Emslie M, Muir P, Thompson AA (2004) Marine surveys undertaken in the Lihou Reef Nature Reserve, March 2004. Department of the Environment and Heritage.

Pebesma EJ (2018) Simple features for R: standardized support for spatial vector data. *The R Journal* 10:439.

Pisapia C, Burn D, Pratchett MS (2019) Changes in the population and community structure of coras during recent disturbances (February 2016-October 2017) on Maldivian coral reefs. Sci Rep 9:8402

Pratchett MS, Hoey AS, Wilson SK, Messmer V, Graham NA (2011) Changes in biodiversity and functioning of reef fish assemblages following coral bleaching and coral loss. *Diversity* 3: 424-452

Pratchett MS, Hoey AS, Wilson SK (2014) 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 MS, McWilliam MJ, Riegl B (2020) Contrasting shifts in coral assemblages with increasing disturbances. *Coral Reefs* 39: 783-793

Pratchett MS, Heron SF, Mellin C, Cumming GS (2021) Recurrent mass-bleaching and the potential for ecosystem collapse on Australia's Great Barrier Reef. In: *Ecosystem Collapse and Climate Change* (eds. JG Canadell, RB Jackson) Springer, Cham. pp. 265-289.

Rasher DB, Hoey AS, Hay ME (2013) Consumer diversity interacts with prey defenses to drive ecosystem function. *Ecology* 94:1347-1358.

Richardson LE, Graham NA, Pratchett MS, Eurich JG, Hoey AS (2018) Mass coral bleaching causes biotic homogenization of reef fish assemblages. *Global Change Biology* 24:3117-29.

Ridgway KR, Benthuysen JA, 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.

Robinson JP, Wilson SK, Jennings S, Graham NA (2019) Thermal stress induces persistently altered coral reef fish assemblages. *Global Change Biology* 25:2739-50.

Roth MS, Knowlton N (2009) Distribution, abundance and microhabitat characterization of small juvenile corals at Palmyra Atoll. Mar Ecol Prog Ser 376: 133-142

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

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

Skewes TD, Persson SI (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.

Souter D, Planes S, Wicquart J, Logan L, Obura D, Staub F (eds) (2021) Status of coral reefs of the world: 2020. GCRMN.

Stella JS, Pratchett MS, Hutchings P, Jones GP (2011) Coral-associated invertebrates: diversity, ecological importance and vulnerability to disturbance. *Oceanography and Marine Biology: An Annual Review* 49: 43-104.

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

Stuart-Smith RD, Brown CJ, Ceccarelli DM, Edgar G (2018) Ecosystem restructuring along the Great Barrier Reef following mass coral bleaching. *Nature* 560:92-6.

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.

Teichert S, Steinbauer M, Kiessling W (2020) A possible link between coral reef success, crustose coralline algae and the evolution of herbivory. *Scientific Reports* 10: 1-12.

Trapon ML, Pratchett MS, Hoey AH (2013) 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(2): e57788

van Hooidonk R, Maynard J, Tamelander J, Gove J, Ahmadia G, Raymundo L, Williams G, Heron SF, Planes S (2016) Local-scale projections of coral reef futures and implications of the Paris Agreement. *Scientific Reports* 6:39666

Van de Leemput IA, Hughes TP, van Nes EH, Scheffer M (2016) Multiple feedbacks and the prevalence of alternate stable states on coral reefs. *Coral Reefs* 35:857-65.

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 SM (2022) 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* 30:e13475

Wilson SK, Graham NAJ, Polunin NVC (2007) Appraisal of visual assessments of habitat complexity and benthic composition on coral reefs. *Marine Biology* 151:1069-1076.

Wilson SK, Graham NAJ, Pratchett MS, Jones GP, Polunin NVC (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 AS, Bellwood DR (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.

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 3 Grant*, however the scope of these projects was increased substantively through activities completed during the 2023 voyage

Project description	Key Personnel	Institution
Movement and population structure of sharks and large fishes within the CSMP	Dr Adam Barnett Prof Andrew Hoey Mr Ben Cresswell	James Cook University James Cook University James Cook University
Opportunistic surveys for fish spawning aggregations	Prof Andrew Hoey Mr Martin Russell	James Cook University Science and Conservation of Fish Aggregations
The cultural and ecological significance of Ashmore and Boot Reefs*	Prof Andrew Hoey Dr Eva McClure Dr Gemma Galbriath Mr Ben Cresswell Dr Victor Huertas Ms Deborah Burn Ms Josie Chandler Mr Martin Russell 8 representatives of the Meriam People	James Cook University James Cook University James Cook University James Cook University James Cook University James Cook University James Cook University Parks Australia Mer Island PBC
Sea Country Documentary*	Mr Alaneo Gloor Mr Stuart Ireland Prof Andrew Hoey Mr Martin Russell Meriam People	Millstream Productions Millstream Productions James Cook University Parks Australia Mer Island PBC
Vessel Grounding at Moore Reefs	Prof Andrew Hoey Dr Eva McClure Dr Gemma Galbriath Mr Ben Cresswell Dr Victor Huertas Ms Deborah Burn Ms Josie Chandler Dr Natalie Bool	James Cook University James Cook University James Cook University James Cook University James Cook University James Cook University James Cook University Parks Australia

7 APPENDIX 2 – Sites surveyed

List of sites surveyed across 11 reefs in the Coral Sea Marine Park (CSMP) during February - March 2023. * indicates sites that were surveyed in 2020, 2021, 2022, and/or 2023, and form the basis of the temporal comparisons. # indicates new sites that were surveyed for the first time in 2023

Sector	Reef	Site	Exposure	Aspect	Lat	Long
Central	Diamond	Diamond 1*	Semi-sheltered	NE	-17.442	151.0626
Central	Diamond	Diamond 2*	Sheltered	N	-17.4368	151.0697
Central	Diamond	Diamond 6*	Sheltered	W	-17.4187	151.0712
Central	Flinders	Flinders 5*	Sheltered	W	-17.8616	148.4665
Central	Flinders	Flinders 7*	Exposed	NE	-17.5368	148.5511
Central	Herald	Herald 1*	Semi-exposed	N	-16.9435	149.1857
Central	Herald	Herald 4*	Sheltered	SW	-16.9725	149.1287
Central	Herald	Herald 6*	Sheltered	W	-16.9919	149.1308
Central	Holmes	Holmes 1*	Sheltered	NW	-16.5261	147.807
Central	Holmes	Holmes 10*	Semi-exposed	NW	-16.5214	147.8377
Central	Holmes	Holmes 2*	Semi-sheltered	W	-16.5118	147.84
Central	Holmes	Holmes 5*	Semi-sheltered	NW	-16.5053	147.9675
Central	Holmes	Holmes 6*	Semi-sheltered	NW	-16.419	147.9898
Central	Holmes	Holmes 7*	Semi-sheltered	NW	-16.4269	147.9844
Central	Lihou	Lihou 1*	Sheltered	NW	-17.5971	151.4896
Central	Lihou	Lihou 2*	Sheltered	N	-17.5907	151.5003
Central	Lihou	Lihou 4*	Semi-sheltered	N	-17.1253	151.8254
Central	Lihou	Lihou 5*	Semi-sheltered	N	-17.1211	151.8294
Central	Lihou	Lihou 7*	Exposed	SE	-17.4173	151.8661
Central	Lihou	Lihou 9*	Lagoon	SE	-17.1302	151.8393
Central	Moore	Moore 3*	Semi-exposed	N	-15.8774	149.1596
Central	Moore	Moore 4*	Sheltered	W	-15.9648	149.1943
Central	Willis	Willis 2*	Sheltered	W	-16.2873	149.9593
Central	Willis	Willis 4*	Semi-exposed	NE	-16.2826	149.9657
Central	Willis	Willis 7*	Semi-sheltered	NW	-16.117	149.971
Northern	Ashmore	Ashmore 7*	Sheltered	North	-10.4391	144.429
Northern	Ashmore	Ashmore 8#	Exposed	E	-10.2535	144.5743
Northern	Ashmore	Ashmore 9#	Exposed	E	-10.2607	144.5568
Northern	Ashmore	Ashmore 10#	Sheltered	E	-10.3831	144.3813
Northern	Ashmore	Ashmore 11#	Sheltered	E	-10.3895	144.3839
Northern	Ashmore	Ashmore 12#	Exposed	SE	-10.3983	144.4905
Northern	Ashmore	Ashmore 13#	Exposed	SE	-10.4031	144.4869
Northern	Ashmore	Ashmore 14#	Exposed	SE	-10.4149	144.4773
Northern	Ashmore	Ashmore 15#	Exposed	SE	-10.4085	144.4811
Northern	Ashmore	Ashmore 16#	Exposed	SE	-10.3983	144.5394
Northern	Ashmore	Ashmore 17#	Lagoon	S	-10.1588	144.5812
Northern	Ashmore	Ashmore 18#	Lagoon	S	-10.0691	144.5298

Northern	Boot	Boot 4#	Exposed	E	-9.98998	144.6943
Northern	Boot	Boot 5#	Exposed	E	-10.0021	144.6958
Northern	Boot	Boot 6#	Exposed	E	-9.97152	144.7215
Northern	Boot	Boot 7#	Exposed	E	-9.97534	144.7131
Northern	Boot	Boot 8#	Lagoon	W	-9.98221	144.6956
Northern	Bougainville	Bougainville 1*	Sheltered	N	-15.4927	147.0864
Northern	Bougainville	Bougainville 4*	Semi-exposed	SW	-15.5067	147.1123
Northern	Bougainville	Bougainville 5*	Semi-exposed	SW	-15.5008	147.0989
Northern	Osprey	Osprey 1*	Exposed	N	-13.8013	146.5461
Northern	Osprey	Osprey 2*	Sheltered	W	-13.901	146.5619
Northern	Osprey	Osprey 6*	Sheltered	S	-13.8808	146.5588

8 APPENDIX 3 – CCA devices and temperature loggers

List of Coral Sea Marine Park (CSMP) reefs and sites in which Crustose Coralline Algae (CCA) devices and/or temperature loggers were deployed during February - March 2023

Reef	Site	Temperature loggers	CCA devices	Date
Flinders	Flinders 5	Y	Y	11/2/2023
	Flinders 7	Y	Y	11/2/2023
Diamond	Diamond 1	Y	Y	13/2/2023
	Diamond 2	Y	Y	13/2/2023
Lihou	Lihou 1	Y	Y	14/2/2023
	Lihou 4	Y	Y	15/2/2023
Willis	Willis 2	Y		16/2/2023
	Willis 7	Y		17/2/2023
Moore	Moore 4	Y	Y	18/2/2023
	Moore 3	Y	Y	18/2/2023
Holmes	Holmes 6	Y	Y	19/2/2023
	Holmes 5	Y	Y	19/2/2023
	Holmes 2	Y		20/2/2023
Bougainville	Bougainville 5	Y	Y	21/2/2023
	Bougainville 4	Y	Y	21/2/2023
	Bougainville 1	Y	Y	21/2/2023
Osprey	Osprey 6	Y	Y	22/2/2023
	Osprey 2	Y	Y	22/2/2023
	Osprey 1	Y	Y	22/2/2023
Ashmore	Ashmore 7	Y	Y	28/2/2023
	Ashmore 17	Y	Y	7/3/2023
	Ashmore 18	Y	Y	7/3/2023

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

10 APPENDIX 5 – Fish species records

List of conspicuous (i.e., non-cryptic) fish species recorded and/or observed within each region of the CSMP during 2018-2023. 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 2023

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	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 fumea*		1		
172	*Chromis iomelas*	1	1	1	1
173	*Chromis lepidolepis*	1	1	1	
174	*Chromis margaritifer*	1	1	1	1

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

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

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

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

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

479	*Plectroglyphidodon dickii*	1	1	1	
480	*Plectroglyphidodon imparipennis*	1	1	1	
481	*Plectroglyphidodon johnstonianus*	1	1	1	
482	*Plectroglyphidodon lacrymatus*	1	1	1	1
483	*Plectroglyphidodon leucozonus*			1	
484	*Plectroglyphidodon phoenixensis*	1	1		
485	*Plectropomus areolatus*		1	1	
486	*Plectropomus laevis*	1	1	1	
487	*Plectropomus leopardus*	1	1	1	
488	*Plectropomus oligacanthus*			1	
489	*Plectrypops lima*				1
490	*Plesiops caeruleolineatus*				1
491	*Pleurosicya mossambica*				1
492	*Plotosus lineatus*	1	1	1	1
493	*Pomacanthus imperator*	1	1	1	
494	*Pomacanthus sexstriatus*			1	
495	*Pomacentrus amboinensis*			1	1
496	*Pomacentrus auriventris*			1	
497	*Pomacentrus bankanensis*	1	1	1	
498	*Pomacentrus brachialis*	1		1	1
499	*Pomacentrus chrysurus*		1	1	
500	*Pomacentrus coelestis*	1	1	1	
501	*Pomacentrus imitator*	1	1	1	
502	*Pomacentrus lepidogenys*	1	1	1	
503	*Pomacentrus moluccensis*	1	1	1	
504	*Pomacentrus nagasakiensis*		1	1	1
505	*Pomacentrus pavo*			1	
506	*Pomacentrus philippinus*	1		1	1
507	*Pomacentrus vaiuli*	1	1	1	1
508	*Pomacentrus wardi*	1			
509	*Pomachromis richardsoni*	1	1	1	
510	*Priacanthus blochii*		1		
511	*Priacanthus hamrur*		1		
512	*Priolepis cincta*				1
513	*Priolepis compita*				1
514	*Priolepis inhaca*				1
515	*Priolepis kappa*				1
516	*Priolepis pallidicincta*				1
517	*Priolepis psygmophila*				1
518	*Priolepis sp.*				1
519	*Prionurus maculatus*	1			
520	*Pristiapogon exostigma*				1
521	*Prteragogus sp.*	1			
522	*Pseudanthias cooperi*		1		
523	*Pseudanthias pascalus*	1	1	1	
524	*Pseudanthias pleurotaenia*		1	1	
525	*Pseudanthias squamipinnis*	1	1	1	
526	*Pseudanthias tuka*	1	1	1	
527	*Pseudobalistes flavimarginatus*		1	1	
528	*Pseudobalistes fuscus*	1	1	1	
529	*Pseudocheilinus evanidus*	1	1	1	1
530	*Pseudocheilinus hexataenia*	1	1	1	1
531	*Pseudocheilinus octotaenia*		1		
532	*Pseudochromis sp.*				1
533	*Pseudochromis tapeinosoma*				1
534	*Pseudocoris yamashiroi*			1	
535	*Pseudodax moluccanus*	1	1	1	
536	*Pseudogramma polyacanthus*				1
537	*Pseudojuloides cerasinus*		1		
538	*Pseudoplesiops annae*				1
539	*Pseudoplesiops sp.*				1

540	*Pseudoplesiops wassi*				1
541	*Pteragogus cryptus*	1	1		1
542	*Pteragogus sp.*	1	1		
543	*Ptereleotris evides*	1	1	1	
544	*Ptereleotris zebra*		1	1	
545	*Pterocaesio digramma*	1	1		
546	*Pterocaesio marri*		1	1	
547	*Pterocaesio tile*	1	1	1	
548	*Pterocaesio trilineata*	1	1	1	
549	*Pterois volitans*	1		1	1
550	*Pygoplites diacanthus*	1	1	1	1
551	*Pycnochromis lineatus**	1			
552	*Rhinecanthus aculeatus*			1	
553	*Rhinecanthus rectangulus*	1	1	1	
554	*Sargocentron caudimaculatum*		1		
555	*Sargocentron ittodai*				1
556	*Sargocentron spiniferum*	1	1	1	
557	*Saurida gracilis*	1			
558	*Scarini sp.*				1
559	*Scarus altipinnis*	1	1	1	
560	*Scarus chameleon*	1	1	1	
561	*Scarus dimidiatus*		1	1	
562	*Scarus festivus**	1			
563	*Scarus forsteni*	1	1	1	
564	*Scarus frenatus*	1	1	1	
565	*Scarus ghobban*			1	
566	*Scarus globiceps*	1	1	1	
567	*Scarus longipinnis*	1	1	1	
568	*Scarus niger*	1	1	1	
569	*Scarus oviceps*	1	1	1	
570	*Scarus psittacus*	1	1	1	
571	*Scarus rubroviolaceus*	1	1	1	
572	*Scarus schlegeli*	1	1	1	
573	*Scarus spinus*	1	1	1	
574	*Scarus viridifucatus*			1	
575	*Scarus xanthopleura*	1	1	1	
576	*Scolopsis bilineata*	1		1	
577	*Scomberoides commersonianus*		1		
578	*Scomberoides lysan*		1	1	
579	*Scomberoides sp*			1	
580	*Scomberomorus commerson*			1	
581	*scorpaenid sp.*				1
582	*Scorpaenodes corallinus*				1
583	*Scorpaenodes guamensis*				1
584	*Scorpaenopsis macrochir*				1
585	*Scorpaenopsis sp.*				1
586	*Sebastapistes corallinus*				1
587	*Sebastapistes cyanostigma*				1
588	*Sebastapistes cyanostigma*			1	
589	*Serranocirrhites latus*	1	1	1	
590	*Siganus argenteus*	1	1	1	
591	*Siganus corallinus*	1	1		
592	*Siganus doliatus*				
593	*Siganus puellus*	1			
594	*Siganus punctatissimus*		1		
595	*Siganus punctatus*	1	1	1	
596	*Siganus vulpinus*	1	1	1	
597	*Siganus woodlandi*	1	1		
598	*Siphamia tubifer*				1
599	*Sphyraena barracuda*	1	1	1	
600	*Sphyraena forsteri*		1		

601	*Sphyraena qenie**				
602	*Stegastes fasciolatus*	1	1	1	
603	*Stegastes gascoynei*	1			
604	*Stegastes nigricans*	1	1	1	1
605	*Stegostoma fasciatum*	1	1		
606	*Stethojulis bandanensis*	1	1	1	1
607	*Stethojulis interrupta*	1			
608	*Stethojulis strigiventer*	1	1	1	
609	*Sufflamen bursa*	1	1	1	
610	*Sufflamen chrysopterum*	1	1	1	
611	*Suttonia lineata*				1
612	*Synodus binotatus*				1
613	*Synodus dermatogenys*				1
614	*Synodus variegatus*	1	1	1	
615	*Synodus varigatus*				1
616	*Taeniura lymma*		1		
617	*Taeniura meyeni*	1	1		
618	*Thalassoma amblycephalum*	1	1	1	1
619	*Thalassoma hardwicke*	1	1	1	
620	*Thalassoma lunare*	1	1	1	
621	*Thalassoma lutescens*	1	1	1	1
622	*Thalassoma nigrofasciatum*	1	1	1	
623	*Thalassoma purpureum*	1	1	1	
624	*Thalassoma quinquevittatum*	1	1	1	
625	*Thalassoma trilobatum*		1	1	
626	*Thysanophrys celebicus*				1
627	*Trachinotus baillonii*			1	
628	*Trachinotus blochii*			1	
629	*Triaenodon obesus*	1	1	1	
630	*Trimma caesiura*				1
631	*Trimma emeryi*				1
632	*Trimma lantana*				1
633	*Trimma macrophthalma*				1
634	*Trimma maiandros*				1
635	*Trimma milta*				1
636	*Trimma necopinna*				1
637	*Trimma okinawae*				1
638	*Trimma sp.*				1
639	*Trimmatom eviotops*				1
640	*Trimmatom macropodus*				1
641	*Trimmatom nanus*				1
642	*Trimmatom sp.*				1
643	*Ucla xenogrammus*				1
644	*Valenciennea strigata*		1	1	
645	*Variola albimarginata*		1	1	
646	*Variola louti*	1	1	1	
647	*Xenisthmus eirospilus*				1
648	*Zanclus cornutus*	1	1	1	
649	*Zebrasoma scopas*	1	1	1	
650	*Zebrasoma velifer*	1	1	1	
Total		**325**	**385**	**355**	**213**