ELIZABETH AND MIDDLETON REEFS (LORD HOWE MARINE PARK) CORAL REEF HEALTH SURVEYS

Report on reef surveys
February 2024

PRODUCED FOR PARKS AUSTRALIA, JUNE 2024
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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 Elizabeth and Middleton Reefs in the Lord Howe Marine Park.

On the cover – A Galapagos shark (*Carcharhinus galapagensis*) cruises along the reef crest at Middleton Reef. Image credit: Victor Huertas

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1 Executive Summary

Elizabeth and Middleton Reefs are the world's southernmost open ocean platform reefs, supporting a unique mix of tropical and subtropical species. While isolated reefs, such as Elizabeth and Middleton Reefs, are often described as being 'near pristine' due to limited local anthropogenic impacts, these isolated reefs (like coral reefs globally) are being increasingly exposed to changing environmental conditions, particularly ocean warming. Quantifying the status and trends on these reefs is critical to understand the responses of these reef ecosystems to contemporary and future disturbances.

James Cook University was commissioned by Parks Australia to:

(i) assess the current condition of benthic, fish and invertebrate communities within Elizabeth and Middleton Reefs using methods that were consistent with previous surveys (2011, 2014, 2018) of these reefs, and directly comparable with those used in the Coral Sea Marine Park (2018-2024).

(ii) assess the status of endemic and/or threatened species, including Black cod (*Epinephelus daemelii*), McCulloch's anemonefish (*Amphiprion mccullochi*), and doubleheader wrasse (*Coris bulbifrons*).

(iii) explore benthic and fish communities within deeper (>15m) habitats around Elizabeth and Middleton Reefs using Remotely Operated Vehicles (ROVs) and/or Baited Remote Underwater Videos (BRUVs).

The project undertook detailed surveys of coral, fish and macro-invertebrate communities and associated reef health at 14 sites on Elizabeth and Middleton Reefs over 6-days from 9th – 14th February, 2024. Surveys were conducted to provide rigorous quantitative information on temporal (i.e., 2011, 2014, 2018, 2024) and spatial (i.e., among sites, 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 (including endemic and/or threatened species), and ecologically or economically important invertebrates.

The 2024 surveys revealed:

- Average coral cover across Elizabeth and Middleton Reefs (31.3%) was the highest recorded between 2011 and 2024, and represents a 4% increase on 2018 coral cover (30.1%). The limited increase in coral cover from 2018 to 2024 compared to the 34% increase in coral cover between 2014 and 2018 indicates there was likely some mortality of corals between 2018 and 2024. Given the time between successive surveys the cause of this mortality is difficult, if not impossible, to identify.
- Current coral cover on Elizabeth and Middleton Reefs is 2- to 3-fold greater, and reef fish density and biomass 2- to 4-fold greater than that of reefs in the southern Coral Sea Marine Park, >780km to the north.
- Low to moderate levels of bleaching were recorded on Elizabeth and Middleton Reefs during early February 2024 (13.6% of coral colonies were pale or bleached). However, the 2024 marine heatwave in the Tasman Sea was still building at the time of our surveys and did not reach its peak until late March where large areas of the Tasman Sea were exposed to >16 Degree Heating Weeks (DHW) and up to 20 DHW in some areas. This level of heat stress is unprecedented, and over double that expected to lead to substantial bleaching and mortality of corals (>8 DHW).
- Densities of juvenile corals at Elizabeth and Middleton Reefs (18.0 juveniles per $10m^2$) are low, and likely reflect their isolation and reliance on the recruitment of locally spawned larvae. This coupled with the lower growth and calcification rates of corals on subtropical reefs limit the recovery potential of coral populations following disturbance.
- The biomass of reef fish on Elizabeth and Middleton Reefs (3,880 kg per hectare) is high relative to coral reefs globally, and together with the high biomass of Galapagos sharks and large-bodied piscivores (e.g., Black cod), likely reflects the limited fishing pressure on these reefs.
- The density of endemic and vulnerable/threatened fish species was generally stable or increased from 2018 to 2024. Importantly, there were small increases in the populations of McCulloch's anemonefish, *Amphiprion mccullochi*, on Elizabeth and Middleton Reefs over the past 6 years, despite

populations on reefs surrounding Lord Howe Island experiencing a >50% decline over the same period

- The density of the ‘vulnerable’ Black cod, *Epinephelus daemelii*, recorded across Elizabeth and Middleton Reefs was consistent between the 2018 and 2024 surveys (2.7 – 2.8 individuals per hectare), and appears to be stable.

Recommendations:

- Monitoring of coral and fish communities (in particular McCulloch’s anemonefish, *A. mccullochi*) on Elizabeth and Middleton Reefs in late 2024 or early 2025 (i.e., before any future heat stress) is critical to quantify the effects of the 2024 marine heatwave on these unique ecosystems. In the absence of future monitoring, any changes in the population status of endemic and threatened species would be largely unknown, severely limiting the capacity of managers to make informed decisions.
- To effectively monitor the potential changes in the unique coral reef ecosystems of Elizabeth and Middleton Reefs following major disturbances, and their potential recovery, we recommend regular monitoring of benthic, fish, and macro-invertebrate communities using the same methods and sites as previous (2011-24) surveys. This series of surveys represents one of the longest running monitoring programs of any Australian Marine Park, and is invaluable in providing contemporary baselines and detecting change. In the absence of any major environmental disturbances the time between recurrent surveys of individual reefs could be 3-5 years, however more frequent and responsive monitoring is recommended to assess any effects of future disturbances.
- In the absence of frequent and responsive monitoring, the utility of occasional visits to Elizabeth and Middleton Reefs should be maximised. This could provide managers with timely information regarding any emerging concerns and/or threats to the ecosystem health of these two unique reefs.
- Maintain collaboration and regular communication between the managers of the Lord Howe Marine Park (Commonwealth waters) and the Lord Howe Island Marine Park (NSW waters) to ensure insight is gained from

monitoring and visitors observations from accessible reefs close to Lord Howe Island.

- As well as monitoring the current status of benthic, fish and macro-invertebrate populations, dedicated research to quantify the ecology and demographic processes of key taxa (e.g., recruitment, growth and mortality of corals, nursery habitats of Black cod, and diet, behaviour and demographics of browsing herbivores) will greatly improve our understanding of the vulnerability, recovery potential, and resilience of Elizabeth and Middleton Reefs.

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2 Background

Coral reefs are one of the world's most biodiverse ecosystems, yet are also one of the most threatened. The effects of local anthropogenic degradation and exploitation (Jackson et al. 2001; Pandolfi et al. 2003), are being greatly compounded by the effects climate change (Hughes et al. 2017a, 2018). Indeed regional declines in coral cover have been reported for the many of the world's coral reefs over the past few decades (e.g., Caribbean: Gardner et al. 2003; Great Barrier Reef: De'ath et al. 2010), with a disproportionate number of coral reefs being lost due high levels of sustained human activities (e.g., chronic pollution, eutrophication, sedimentation, overfishing and/or destructive fishing practices) in some regions (Pandolfi et al. 2003; Wilkinson 2008). Predicted increases in the frequency, intensity and diversity of disturbances to which coral reefs will be exposed (van Hooidonk et al. 2016; Hughes et al. 2017b; Hoegh-Guldberg et al. 2018) paints a bleak picture of further declines in coral cover and increases in other taxa (e.g., macroalgae) over coming decades (Hoey et al. 2016; Bellwood et al. 2019). These declines in coral cover and shifts in benthic composition have flow-on effects to fish and invertebrate communities (Stella et al. 2011; Pratchett et al. 2014; Richardson et al. 2018), the ecological process they support (e.g., Richardson et al. 2020), and the ecosystem goods and services they provide (Lam et al. 2020).

Isolated and high latitude coral reef ecosystems are often considered to be less susceptible to direct anthropogenic pressures and global warming than those close to human populations and situated in low latitudes respectively (Graham and McClanahan 2013; Beger et al. 2014), and has led to suggestions they may act as future refugia for coral reef species. However, isolated reefs are being increasing exposed to the effects of climate-induced disturbances (namely marine heatwaves that cause widespread coral bleaching and severe storms) that are leading to widespread coral mortality (e.g., Gilmour et al. 2013; Hoey et al. 2020, 2024). The effects of these disturbances are particularly pronounced and long-lasting on isolated reefs due to their reliance on self-recruitment of larvae (i.e., larvae spawned from adults on the same reef rather than other reefs nearby) to replenish coral, invertebrate and fish populations (Graham et al. 2007; Gilmour et al. 2013). Similarly, high-latitude reefs systems may also be more (not less) susceptible to climate change because they are disproportionately affected by changing seawater chemistry (specifically, ocean acidification and declining aragonite saturation), which may impose increasing constraints on calcification and reef accretion (Anderson et al. 2015), and are not as immune to thermally-induced coral bleaching as once thought. Coral bleaching has been documented on high-latitude reefs in Australia (Lord Howe Island: Harrison et al. 2011; Moriarty et al. 2023; Solitary Islands: Kim et al. 2019; Houtman Abrolhos: Abdo et al. 2012), South Africa (Celliers and Schleyer 2022), Northwest Hawaiian Islands (Kenyon et al. 2006), Arabian Gulf (Burt et al. 2019) and Japan (Nishiguchi et al. 2018) though the relevant susceptibility and severity of bleaching (compared to low-latitude reefs) is unclear.

2.1 Elizabeth and Middleton Reefs

Elizabeth and Middleton Reefs are the worlds southernmost open ocean platform reefs, located within the Tasman Sea, and supporting a unique mix of tropical and subtropical species (Choat et al. 2006; Hoey et al. 2014, 2018). These emergent reef systems are contained within the Lord Howe Marine Park in the Temperate East Network of Australian Marine Parks (Figure 2.1). The northernmost zone encompassing Middleton Reef is a National Marine Park (IUCN II) which allows “passive use by the public”, but no fishing. The southern zone encompasses Elizabeth Reef and is a Recreational Use Zone (IUCN IV) that allows recreational fishing. Due to its isolation from the Australian mainland and the small human population on Lord Howe Island, however, fishing pressure at Elizabeth Reef is low.

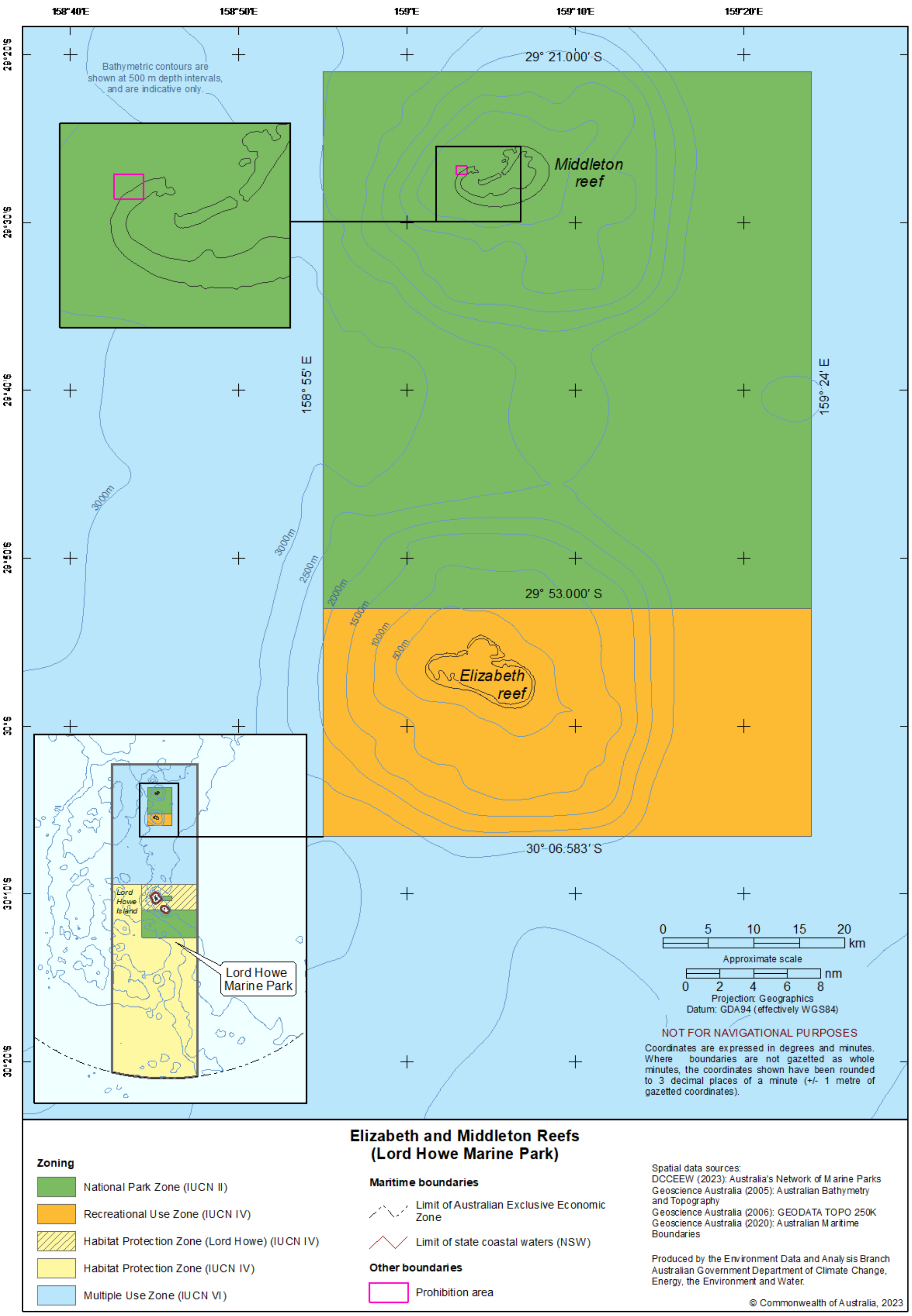

Figure 2.1: Map showing the spatial extent and location of specific management zones within the Lord Howe Marine Park, and the location of Middleton and Elizabeth Reefs. Note: the pink box on the northwestern aspect of Middleton Reef is an 'exclusion zone' due to the presence of a suspected unexploded ordnance. Source: www.parksaustralia.gov.au

Elizabeth and Middleton Reefs are critically important coral reef environments, owing to their isolation and location. These reefs support a mix of tropical and subtropical species, including a number of endemic species or species that are generally rare over most of their range. Most notably, Elizabeth and Middleton Reefs are renowned for being the last remaining stronghold for the black cod (*Epinephelus daemelii*), which occurs throughout the southwestern Pacific, but has been overfished throughout much of its range (Choat et al. 2006; Harasti and Malcolm 2013; Francis et al. 2015). *Epinephelus daemelii* is listed as vulnerable in Commonwealth environment legislation, and as 'near threatened' on the *International Union for the Conservation of Nature (IUCN) Red List of Threatened Species: 2023*.

Several other fish species are endemic to Elizabeth and Middleton Reefs, Lord Howe Island, and/or Norfolk Island, and include McCulloch's anemonefish (*Amphiprion mccullochi*), the doubleheader wrasse (*Coris bulbifrons*), and the three-striped butterflyfish (*Chaetodon tricinctus*). Notably, *A. mccullochi* was recently assessed by the NSW Fisheries Scientific Committee for a proposed determination of 'critically endangered' based on population declines on the reefs surrounding Lord Howe Island (NSW Fisheries Scientific Committee 2023).

2.1.1 History of Coral Reef Surveys at Elizabeth and Middleton Reefs

There have been at least 14 scientific surveys undertaken at Elizabeth and/ or Middleton Reefs since 1979, with the most recent surveys being conducted by the National Environmental Science Program (NESP) Marine and Coastal Hub, led by Geoscience Australia in 2020 (Table 2.1). Collectively, these surveys represent the longest-running surveys of coral reef habitats across the entire network of Australian Marine Parks (Hoey and Pratchett 2017). Recurrent surveys of the shallow reef environments have revealed a gradual, but sustained, recovery of coral cover at Elizabeth and Middleton Reefs over the past two and a half decades, from *ca.* 10% cover in 1994 up to 20-30% in 2018 (Elizabeth Reef: 29.2%; Middleton Reef: 19.3%; Hoey et al. 2018). Low levels of coral cover reported in the early 1990's are widely attributed to a localised population irruption of the corallivorous crown-of-thorns starfish (*Acanthaster* cf. *solaris*) in the mid- to late-1980s (Harriot 1998). While the recovery of coral assemblages over the past 25 years is encouraging, it is apparent that rates of coral recovery at these locations are slow, especially compared with isolated reefs at lower latitudes (e.g., Scott Reef, Gilmour et al. 2013; Seychelles: Graham et al. 2015). The protracted recovery of coral populations likely reflects the reliance on self-recruitment and hence limited supply of coral larvae (Gilmour et al. 2013; Pratchett et al. 2015), and the lower calcification and growth of corals at higher latitudes (Pratchett et al. 2015). It is possible, however, that rates of coral recruitment will increase as the abundance, size and/or fecundity of local adult coral populations increase (Gilmour et al. 2013).

Table 2.1. List of marine surveys undertaken at Elizabeth and/ or Middleton Reef since 1979. For a comprehensive overview of surveys conducted prior to 1979 see Australian Museum (1992).

Year	Elizabeth	Middleton	Purpose	Source
1979	X	X	Coral diversity	Australian Museum 1992
1984	X	X	Coral diversity	Australian Museum 1992
1981		X	Assess impacts of COTS	Harriot 1998
1987	X	X	Biodiversity assessment	Australian Museum 1992
1994	X	X	Marine ecological survey	Choat et al. unpublished data
2003	X		Marine ecological survey	Oxley et al. 2003
2006	X	X	Marine ecological survey (JCU)	Choat et al. 2006
2007	X	X	Rapid assessment of reef health (JCU)	Hobbs and Feary 2007
2011	X	X	Marine ecological survey (JCU)	Pratchett et al. 2011
2012	X	X	Reef Life Survey (RLS)	Edgar et al. 2017
2014	X	X	Marine ecological survey (JCU)	Hoey et al. 2014
2018	X	X	Reef Life Survey (RLS)	Edgar et al. 2018
2018	X	X	Marine ecological survey (JCU)	Hoey et al. 2018
2020	X	X	Seabed mapping and stereo-BRUV sampling	Carroll et al. 2021

2.2 Objectives and scope

The objective of this project was to provide comprehensive assessments of the current condition of benthic, fish and macro-invertebrate communities, populations of endemic and/or threatened fish species within Elizabeth and Middleton Reefs, Lord Howe Marine Park, using methods that were consistent with previous surveys of these reefs (Pratchett et al. 2011; Hoey et al. 2014, 2018), and directly comparable with those used in the Coral Sea Marine Park (Hoey et al. 2020, 2024). Additionally, we aimed to explore benthic and fish communities within deeper (>15m) habitats around Elizabeth and Middleton Reefs using Remotely Operated Vehicles (ROVs) and/or Baited Remote Underwater Videos (BRUVs) (Figure 2.2).

Surveys were conducted at 14 sites across Elizabeth and Middleton Reefs following the methods of Hoey et al. (2020, 2024). At each site, diver-based surveys were conducted along three replicate transects within each of two reef zones (reef crest: 1-3m depth; reef slope: 7-10m depth) to provide rigorous quantitative information on spatial (i.e., among sites, 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 (including endemic and threatened species);
vii) abundance of holothurians, urchins and other ecologically or economically important reef-associated invertebrates.

Unfavourable weather conditions at the time of the surveys made it difficult and unsafe to deploy the ROVs and BRUVs from the tenders, and severely limited our ability to survey deeper habitats.

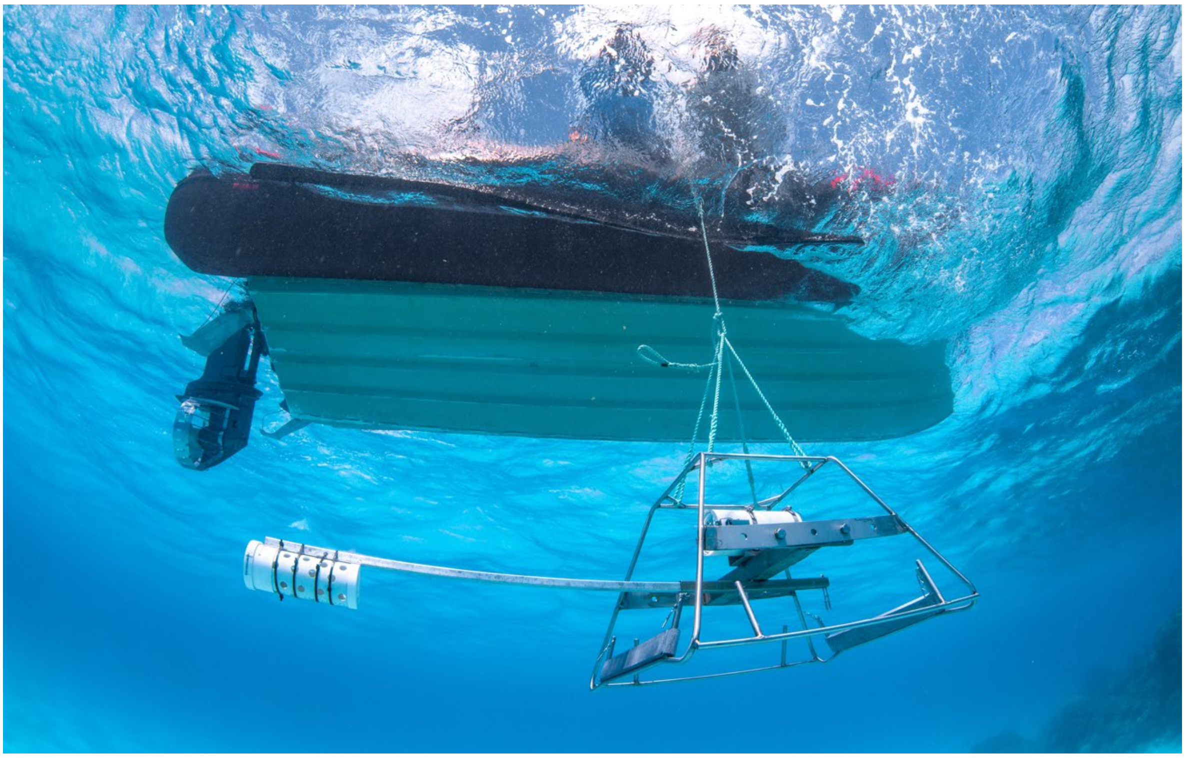

Figure 2.2 Deployment of a Baited Remote Underwater Video (BRUV) instrument to sample fish communities. Image credit: Victor Huertas.

3 Methods

Surveys were undertaken at 14 sites across Elizabeth and Middleton Reefs during a 6-day voyage, 9th – 14th February 2024 (Figure 3.1; Appendix 1). To facilitate comparisons among years we re-visited the sites that were surveyed during previous voyages to these reefs (Pratchett et al. 2011; Hoey et al. 2014, 2018). Six sites were surveyed on Elizabeth Reef (2 back reef, 2 lagoon, and 2 reef front sites) and eight sites were surveyed on Middleton Reef (3 back reef, 2 lagoon, and 3 reef front sites; Figure 3.1). Several reef front sites on the exposed eastern and southern aspects could not be accessed at the time of the voyage due to strong winds and swell (Elizabeth sites 1, 8, 9, and 10; Middleton sites 7 and 9). Sites were relocated using GPS waypoints and a bearing of the direction of the transects from that waypoint. The surveys at Middleton site 5 had to be moved several hundred metres to the south as the original site 5 was located within the exclusion zone, adjacent to the wreck of the *SS Runic.*

3.1 Sampling design – diver-based surveys

At each site, diver-based surveys were generally conducted within each of two different reef zones, i) the reef crest (approximately 1-3m depth) and ii) the reef slope (9-10m depth, where possible). The only exceptions to this were two sites where the reef crest could not be safely accessed due to excessive surge and wave action (Elizabeth site 2 and Middleton sites 6 and 10). In shallow reef environments (inside lagoons or in some 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.

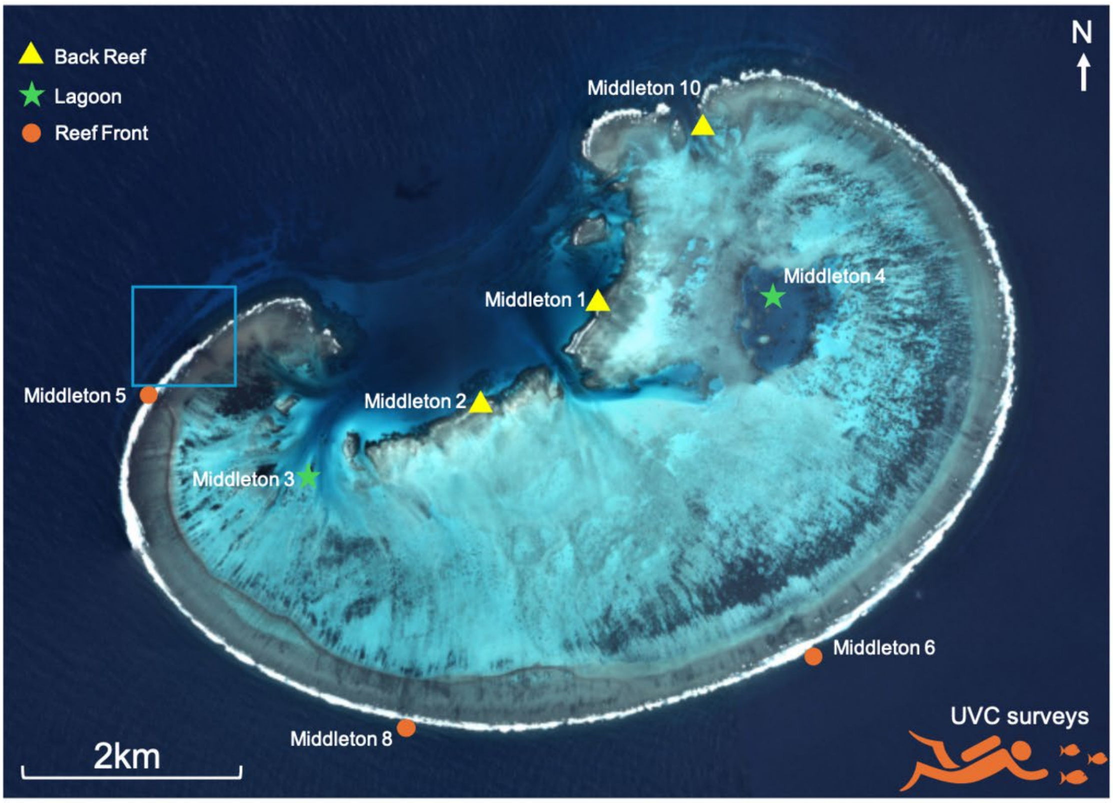

Figure 3.1 Map showing the location of the survey sites on Middleton Reef (top) and Elizabeth Reef (bottom). Sites on Elizabeth Reef were surveyed on the 9th-11th February, and sites on Middleton Reef were surveyed 12th-14th February 2024. Yellow triangles = back reef locations, green stars = lagoon locations, and orange circle = reef front locations. The blue square on the northwestern aspect of Middleton Reef is the exclusion area. Note Middleton 10 is a new site that had not been surveyed previously. Satellite images sourced from Allen Coral Atlas (www.allencoralatlas.org)

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 (or 5-person) dive team, whereby the lead diver deployed the transect tape while simultaneously recording the size and identity of larger (>10 cm total length, TL) and generally more mobile 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 (Emslie et al. 2018). 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 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 Pratchett et al. 2011; Hoey et al. 2014, 2018, 2020). 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 where possible. 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. 2017a). 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 or disease 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.

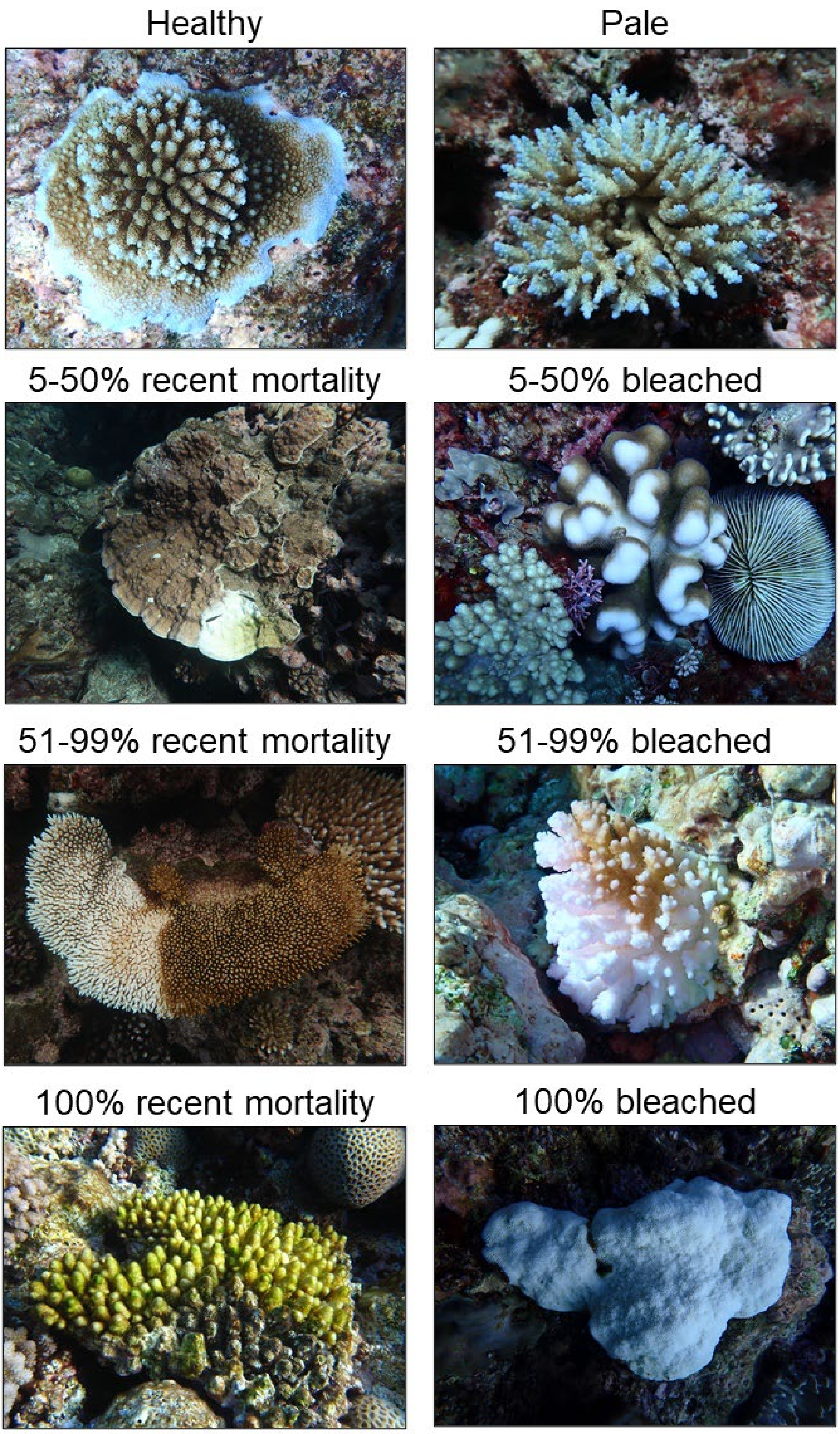

Figure 3.2 Representative images of the coral health categories used for the in-water coral health assessments. Images on the left provide examples of the four injury categories, whilst images on the right are examples of the coral bleaching categories. Note these images were not from Elizabeth and Middleton Reefs. 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 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 Elizabeth and Middleton Reefs. All juvenile corals within the 10 x 1m coral health transect were recorded to genus (Figure 3.3).

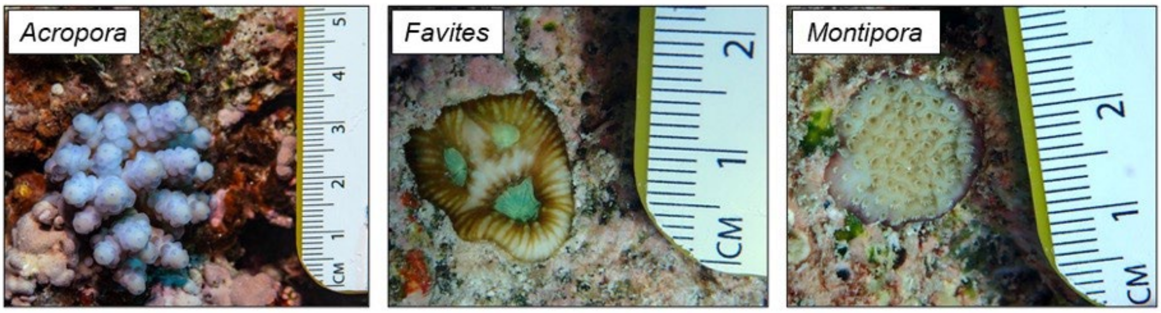

Figure 3.3 Photographs of juvenile (≤5cm diameter) corals recorded within $10m^2$ belt transects within the Coral Sea Marine Park. Each juvenile coral within the $10m^2$ belt transects were identified to genus and recorded. Image credits: Deborah Burn

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. Unlike the previous surveys at Elizabeth and Middleton Reefs in which only herbivorous fishes and endemic and site-attached fishes were surveyed (Pratchett et al 2011; Hoey et al 2014, 2018), all non-cryptic, diurnally active fishes observed within the transects were recorded. 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., Coral Sea Marine Park: Hoey et al. 2020, 2021, 2022, 2024). Smaller site-attached species (e.g., 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 2 for a comprehensive list of species surveyed.

Figure 3.4 Photographs of fish fauna at Elizabeth and Middleton Reefs. Top: Mixed assemblage of herbivorous fishes including the Pacific Drummer *Kyphosus sectatrix*, and the surgeonfishes *Prionurus maculatus* and *Acanthurus dussumieri*, Bottom: School of trevally *Pseudocaranx* sp. Image credits: Victor Huertas.

3.1.3 Non-coral invertebrates

Non-coral invertebrates, including potential coral predators (e.g., crown-of-thorns starfish *Acanthaster* cf. *solaris*, pin-cushion starfish *Culcita novaeguineae*, and coral snails *Drupella* spp.) as well as ecologically and economically important species, namely long-spined sea urchins (*Diadema* spp.), sea cucumbers (holothurians), giant clams (*Tridacna* spp.) and trochus (*Tectus* spp., formerly *Trochus* spp.) were surveyed in a 2m wide belt along each transect, giving a sample area of 100m^2. For all crown-of-thorns starfish (*Acanthaster* cf. *solaris*) and giant clams (*Tridacna* spp.) observed, the size (diameter and length, respectively) was also recorded (to the nearest 10cm). No trochus were recorded during the 2024 surveys at Elizabeth and Middleton Reefs.

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* spp.) 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). The eastern Pacific crown-of-thorns starfish is now recognised as being morphologically and genetically distinct from the predominant Indian Ocean species (*Acanthaster planci*) and is nominally referred to as *Acanthaster* cf. *solaris* (Pratchett et al. 2017). *Acanthaster* cf. *solaris* has been frequently sighted at Elizabeth and Middleton Reefs, as well as at Lord Howe Island and Ball's Pyramid, though densities are consistently very low.

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).

3.1.4 Deep reef habitats – Video-based surveys

Deep water surveys (> 30m) using Baited Remote Underwater Video systems (BRUVs) or Remotely Operated Vehicles (ROV) were not possible during voyage time at Elizabeth and Middleton due to strong winds (> 25kts) and large swells that made deploying and piloting the ROV from a tender unsafe. This restricted the area of operation for both BRUV and ROV surveys conducted from small tenders. Where possible, sampling was undertaken at depths between 15-25m but this was limited to sheltered lagoon and channel sites in close proximity to the *MV Iron Joy*.

ROV sampling was conducted at 2 sites in the northeast channel (22 transects) and 2 sites at the northern outer shelf (18 transects) of Elizabeth Reef at depths between 20 – 25m (Figure 3.5). Ten BRUV drops were completed in the lagoon of Middleton Reef at depths between 2 – 10m. Recommended minimum separation of 500m (Currey-Randall et al. 2020) was not attainable between individual drops due to unfavourable sea state restricting the area of operation however, all drops were at least 300m apart (Figure 3.5).

Baited Remote Underwater Video systems (BRUVs) - BRUVs were deployed broadly following methods and established standard operating procedures outlined in Langlois et al. (2020). Briefly, ~1 kg of crushed pilchards (*Sardinops sp)* was used as bait in a burley pot attached 1.5m in front of a GoPro Hero7 action camera. Cameras were mounted in an underwater housing attached to a weighted frame (SeaGis.com). All BRUVs were deployed between daylight hours of 0800 – 1600 and allowed to “soak” for the standard recommended time of 60 mins each (Currey-Randall et al. 2020).

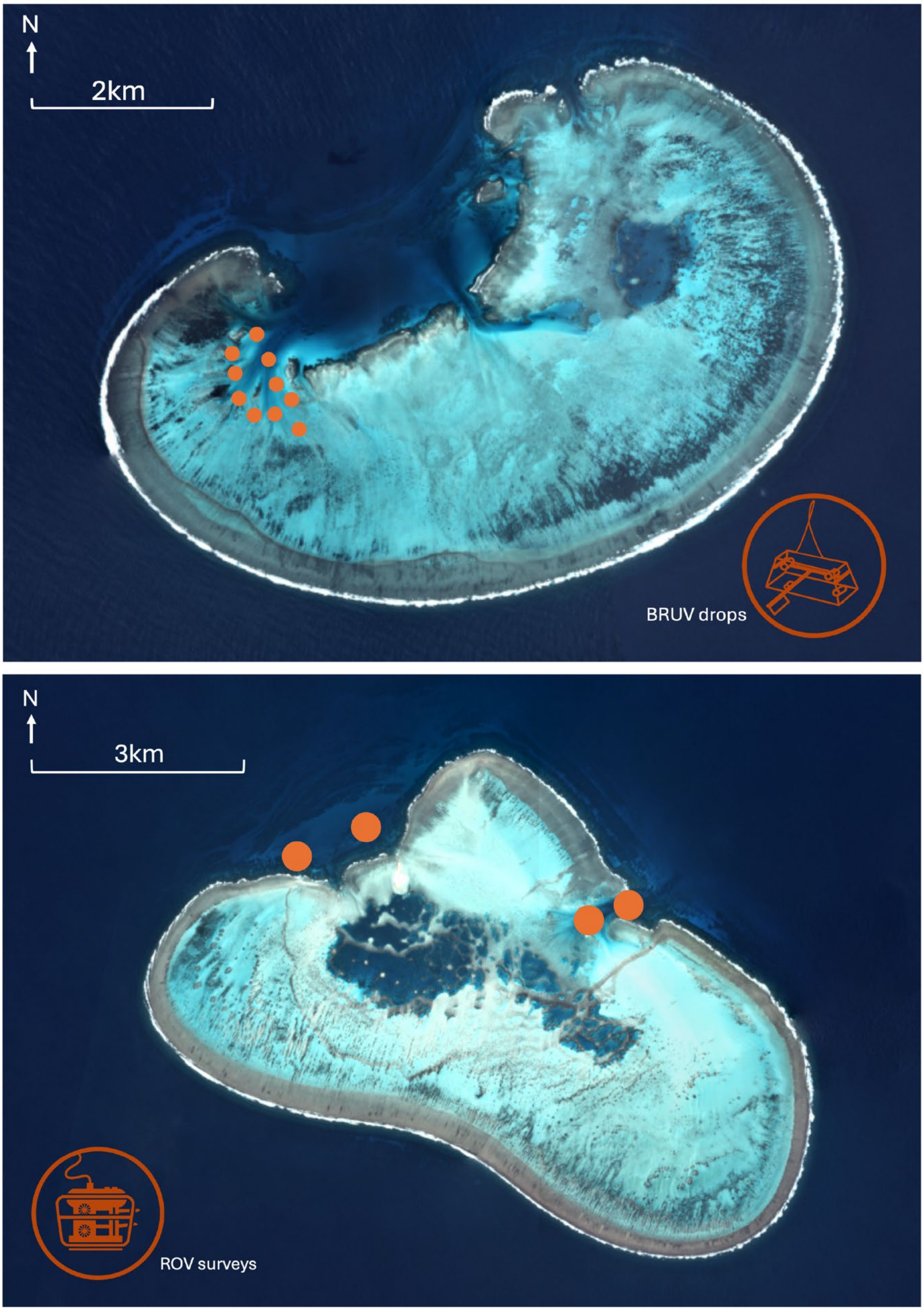

Figure 3.5 Map showing the location of the Baited Remote Underwater Video (BRUV) deployments on Middleton Reef (top) and Remotely Operated Vehicle (ROV) surveys on Elizabeth Reef (bottom). Satellite images sourced from Allen Coral Atlas (www.allencoralatlas.org)

Remotely Operated Vehicle - A BlueROV2 (BlueRobotics) micro-Remotely Operated Vehicle (ROV) was used to conduct area-based sampling of fish and benthic communities at depths below the diver-based visual census surveys (>10m). The ROV was constructed with an 8-thruster vectored configuration and 2 high-powered lumen Subsea lights. In addition to the onboard high-definition (1080p, 30fps), wide-angle, low-light optimized camera that was used for piloting the ROV, the ROV was fitted with a forward-facing GoPro Hero 8 housed inside a deep-rated aluminum T-housing to survey fish communities.

Additionally, a time-lapse benthic camera system consisting of three GoPro Hero 7 action cameras inside deep-rated aluminum T-housings was used. These GoPros were mounted on the left and right side of the ROV to allow the benthic communities on steep habitats (i.e., walls) to be photographed, and one GoPro mounted facing downwards on the ROV payload skid to allow the benthic assemblages on relatively flat, or horizontal, habitats to be photographed. The cameras were set to take timelapse photos resulting in an average of 35 benthic photos (range 16-60) per transect. 30m long transects were conducted using a timed swim method (2.5 minutes at 0.2m/s; Galbraith et al. 2022) and all cameras set to record continuously during the survey. Depth and temperature were recorded by the ROV onboard logger at 10 second intervals during each dive.

Video surveys of fish communities (ROV and BRUV) were interrogated using the software EventMeasure, where every species entering the camera field of view was counted and identified to the lowest possible taxonomic resolution (usually species). For ROV surveys, mean species richness, density and diversity were calculated for each site. For BRUV drops, MaxN was used as a measure of relative abundance; the maximum number of individuals of a species in one frame at one time. Species richness was calculated as total number of species observed per drop.

The ROV benthic cameras were set to timelapse and took photographs of the benthos every 10 seconds. 250 benthic photos were analysed using the software TransectMeasure. Briefly, 25 points are projected onto a grid overlaid on the photo and the primary benthic habitat in each grid square and the specific benthic group directly under each point identified. Mean percentage cover was calculated for each site. Forward facing imagery from the video transect was scored at 5 independent points (separated by 20 seconds footage) in each video to give an estimate of habitat complexity (0 – 5, following Wilson et al. 2007) and slope (0 – 4, where 0 = slope angle of 0 degrees and 4 = slope angle of 90 degrees).

3.2 Data handling and analysis

All data were handled in R Version 4.3.2. (R Core Team 2023). Data were wrangled using the *tidyverse* environment (Wickham 2017) and visualised using the *ggplot2* package (Wickham 2016). Colour palettes for figures were chosen in *RColorBrewer* (Neuwirth 2014) and *viridis* (Garnier 2018), with visualisations aided by *ggrepel* (Slowikowski 2018) and *ggpubr* (Kassambara 2018).

All survey data were averaged across independent transects to obtain a site average prior to summarising data at the level of reef. 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 (not average) number of species/taxa observed at a site, prior to being summarised to the level of reefs. While the focus of this report is on Elizabeth and Middleton Reefs, survey data collected at reefs within the southern Coral Sea Marine Park (CSMP) during Feb-Mar 2024 are used for comparison.

Data are generally presented using box and whisker plots (i.e., box plots). The boxplots 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.

4 Findings

4.1 Shallow water benthic communities

4.1.1 Coral cover and richness

The average taxonomic richness of corals across Elizabeth and Middleton Reefs, based on the number of hard (Scleractinian) coral taxa (mostly genera) recorded using the 50m point-intercept transects at each survey site, was 15.5 taxa per site and ranged from 13.5 taxa per site at Middleton Reef to 18.2 taxa per site at Elizabeth Reef (Figure 4.1a). The taxonomic richness of corals was broadly consistent within the reef front (18.5 and 18.7 taxa per site) and lagoon (13.0 and 10.5 taxa per site on Elizabeth and Middleton Reefs, respectively) at both reefs, but differed markedly in back reef (Elizabeth: 23 taxa per site; Middleton: 10.3 taxa per site).

The average cover of hard (Scleractinian) corals recorded at the 14 sites surveyed across Elizabeth and Middleton Reefs in 2024 was 31.3% (±2.2SE), and varied between reefs and among sites within each reef. Average coral cover was generally higher on Elizabeth Reef (mean ± SE: 35.9 ±3.7 %) than Middleton Reef (27.4 ±1.42%; Figure 4.1b), and ranged from 6.8% to 68.6% on Elizabeth Reef (Elizabeth sites 6 and 2, respectively) and from 7.3% to 40.7% on Middleton Reef (Middleton site 1 and 6, respectively). Although coral cover was greatest on the exposed reef crest on both reefs (Elizabeth: 60.5%; Middleton: 34.7%), differences in coral cover among habitats were not consistent between reefs. On Elizabeth Reef, the lowest coral cover was recorded at the lagoon sites (17.8%) and intermediate coral cover was recorded at the back reef sites (41.8%). In contrast, on Middleton Reef the lowest coral cover was recorded at the back reef sites (14.9%) and intermediate at lagoon sites (31.9%; Figure 4.1b). The higher cover in lagoon habitats at Middleton Reef was driven by the high coral cover at Middleton 4.

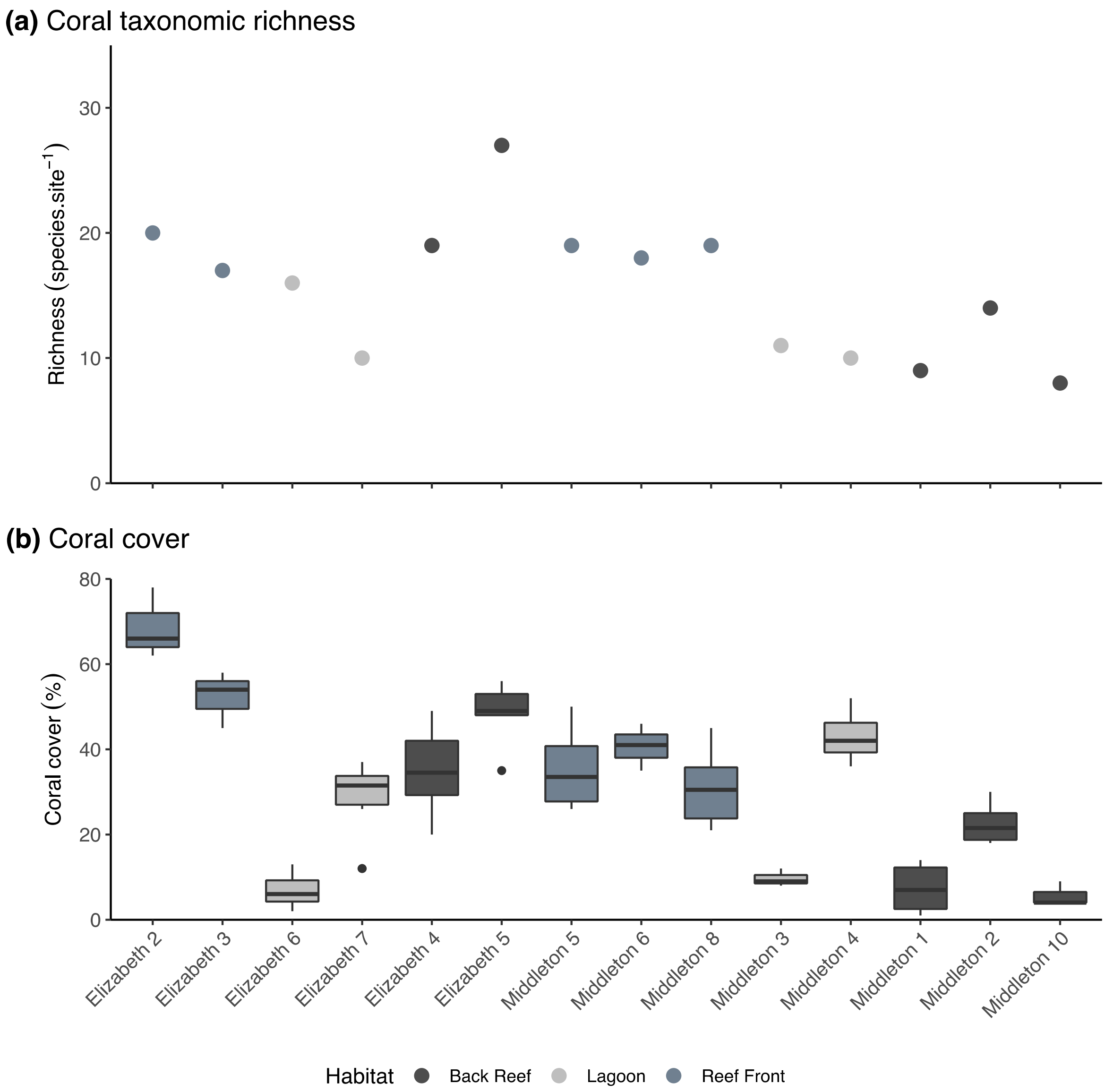

Figure 4.1 Variation in **(a)** coral richness and **(b)** coral cover among 14 sites on Elizabeth and Middleton Reefs in February 2024. Data are based on the 50m point-intercept transects, with data for richness based on the number of coral taxa recorded at each of the 14 sites (i.e., pooled across transects and reef zones).

Temporal changes in coral and macroalgal cover

Coral Cover - Comparisons of coral cover in shallow reef habitats across the 13 reefs that were surveyed on Elizabeth and Middleton Reefs in 2011, 2014, 2018 and 2024 (i.e., excluding Middleton 10) revealed differences in the temporal patterns in coral cover between reefs and among habitats (Figure 4.2a). On Elizabeth Reef average coral cover has increased from 28.4% in 2014 to 33.2% in 2018 and 35.9% in 2024 (an 8.1% increase since 2018, and a 26.4% increase since 2014). These increases have been driven by increases in average coral cover within back reef and reef front habitats, while average coral cover has declined within the lagoon habitat (Table 4.1).

Coral cover has remained relatively stable on Middleton Reef from 2018 to 2024 (27.4%), maintaining the 60.2% increase from 2014 levels (17.1%; Figure 4.2a). In contrast to Elizabeth Reef, coral cover within each habitat was also relatively stable from 2018 to 2024 (Table 4.1).

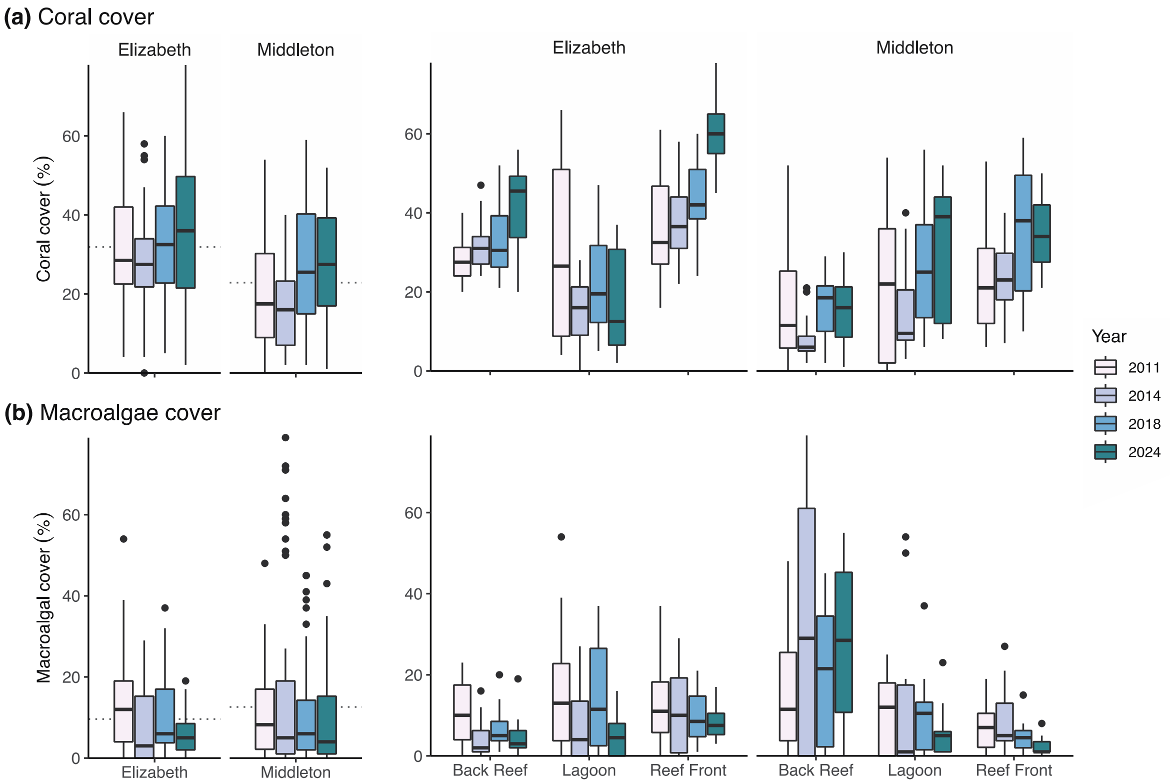

Figure 4.2 Temporal change (2011-2024) in **(a)** coral cover and **(b)** macroalgal cover on Elizabeth and Middleton Reefs. Data are based on replicate 50m point-intercept transects within each of two reef zones (reef crest and slope) at 13 sites. Left-hand panels show reef averages and right-hand panel shows variation among habitats within each reef. Dashed lines on left-hand panels represent reef-wide averages.

Macroalgae cover - Average cover of macroalgae at Elizabeth and Middleton reefs was 9.1% (±1.6 SE) and generally similar (<10% cover) across most habitats, with the exception of the back reef habitat on Middleton Reef where cover of macroalgae (mostly *Codium* spp) was high (28.0%; Figure 4.2b). Importantly, macroalgal cover has been consistently high on the back reef habitat at Middleton Reef (2014: 32.7%; 2018: 20.4%; 2024: 28.0%) for at least a decade. Overall, macroalgal cover varied between reefs and habitats in 2024, with average cover of macroalgae being over 2-fold greater on Middleton (11.8 ± 4.1%) than on Elizabeth Reef (5.8 ± 1.0%). These observed differences were primarily driven by high macroalgal cover in back reef sites at Middleton (Figure 4.2b).

Table 4.1 Temporal variation (2011-2024) in average coral cover between reefs and habitats on Elizabeth and Middleton Reefs, Lord Howe Marine Park. Data are based on replicate 50m point-intercept transects within each of two reef zones (reef crest and slope) at 13 sites.

	2011	2014	2018	2024
Elizabeth	**31.6%**	**28.4%**	**33.2%**	**35.9%**
Back Reef	28.4%	31.6%	33.3%	41.8%
Lagoon	30.4%	15.3%	22.8%	17.8%
Reef Front	36.1%	38.3%	43.5%	60.5%
Middleton	**20.7%**	**17.1%**	**27.4%**	**27.4%**
Back Reef	15.7%	8.3%	16.3%	14.9%
Lagoon	22.7%	15.1%	26.1%	31.9%
Reef Front	23.0%	24.0%	35.6%	34.7%

Coral composition - Coral assemblages on Elizabeth and Middleton Reefs were dominated by *Acropora* (16.4%), *Montipora* (14.9%), *Isopora* (14.8%), and *Platygyra* (11.3%; Figure 4.3) corals. Although there was some variation between reefs, *Acropora, Isopora* and *Platygyra* were generally most abundant on the reef front and back reef habitats, while *Montipora* was the most abundant coral taxon in the lagoon habitat (Figure 4.3).

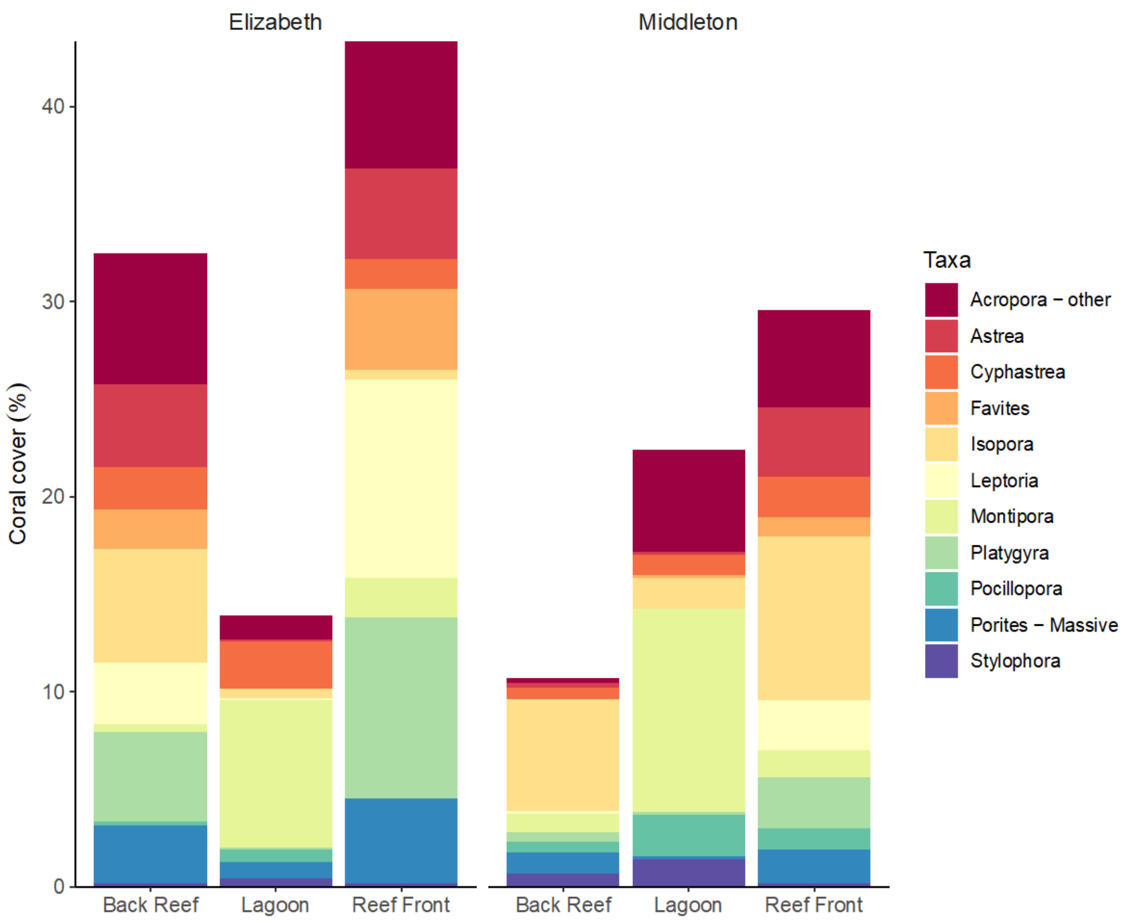

Figure 4.3 Variation in the taxonomic composition of coral assemblages among habitats on Elizabeth and Middleton Reefs in February 2024. Data are based on the 50m point-intercept transects at each of the 14 sites (i.e., pooled across transects and reef zones).

Coral recruitment - A total of 1,165 juvenile corals (≤5cm diameter; Rylaarsdam 1983) were recorded across the 14 sites on Elizabeth and Middleton Reefs in 2024, equating to an average density of 18.0 juveniles per 10m^2; a 58% increase on densities recorded in 2018. The density of juvenile corals was 28% greater on Elizabeth Reef (20.3 juveniles per 10m^2) compared to Middleton Reef (15.9 juveniles per 10m^2), and were generally greater within reef front and lagoon, than back reef habitats (Figure 4.4). Temporal trends in the density of juvenile corals were variable among reefs and habitats, with an 88.0% increase on Elizabeth (2018: 10.8 juveniles per 10m^2; 2024: 20.3 juveniles per 10m^2) compared to a 33.6% increase on Middleton Reef (2018: 11.9 juveniles per 10m^2; 2024: 15.9 juveniles per 10m^2) from 2018 to 2024 (Figure 4.4). The recorded increases in the density of juvenile corals over the past 6 years (i.e., 2018-2024) were driven by increases in reef front and lagoon habitats at each reef, while the density of juvenile corals either remained relatively stable (Elizabeth Reef) or declined (Middleton Reef) in the back reef habitat over the same period (Figure 4.4).

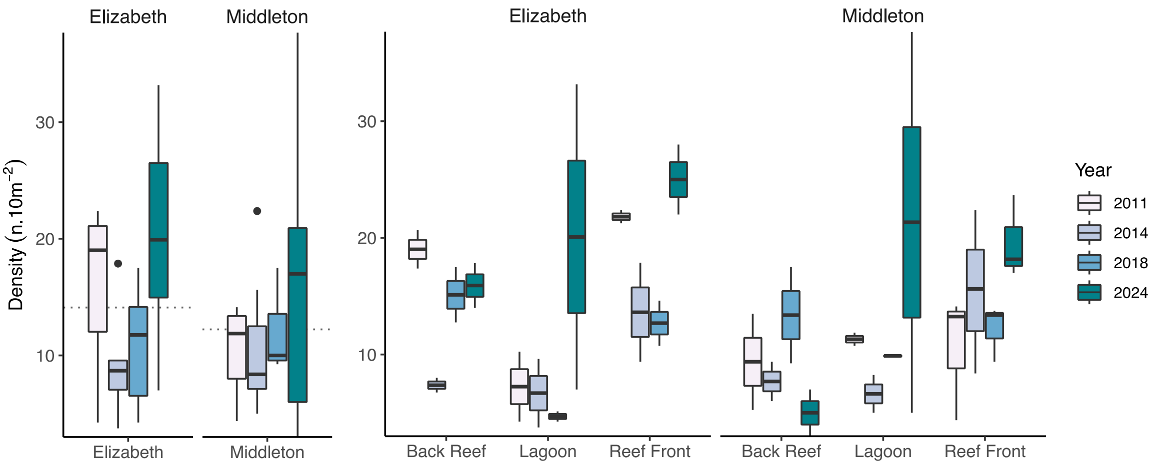

Figure 4.4 Temporal variation (2011-2024) in the density of juvenile corals (<5cm diameter) on Elizabeth and Middleton Reefs. Data are based on replicate 50m point-intercept transects within each of two reef zones (reef crest and slope) at 13 sites. Dashed lines on left-hand panels show reef averages and right-hand panel shows variation among habitats within each reef.

In contrast to adult coral assemblages that were dominated by *Acropora* and *Isopora* (Figure 4.3), juvenile coral assemblages at Elizabeth and Middleton Reefs were dominated by encrusting *Montipora* (27.4%), *Astrea* (17.9%), *Platygyra* (13.5%), *Cyphastrea* (7.2%), and *Pocillopora* (5.9%), with the remaining 20 taxa collectively accounting for 27.1% of juvenile corals recorded (Figure 4.5). There were distinct differences in the composition of juvenile coral assemblages among habitat, with the lagoon habitats on both reefs being dominated by encrusting *Montipora* (59.7-74.8% of juveniles), and the back reef and reef front habitats being dominated by submassive *Astrea* (34.8-47.8%) and *Platygyra* (22.3-35.9%; Figure 4.5). Recent and ongoing changes in the taxonomic classifications of scleractinian corals (e.g., Huang et al. 2009, 2014; Kitihara et al. 2016) preclude any temporal comparison of the composition of juvenile coral assemblages.

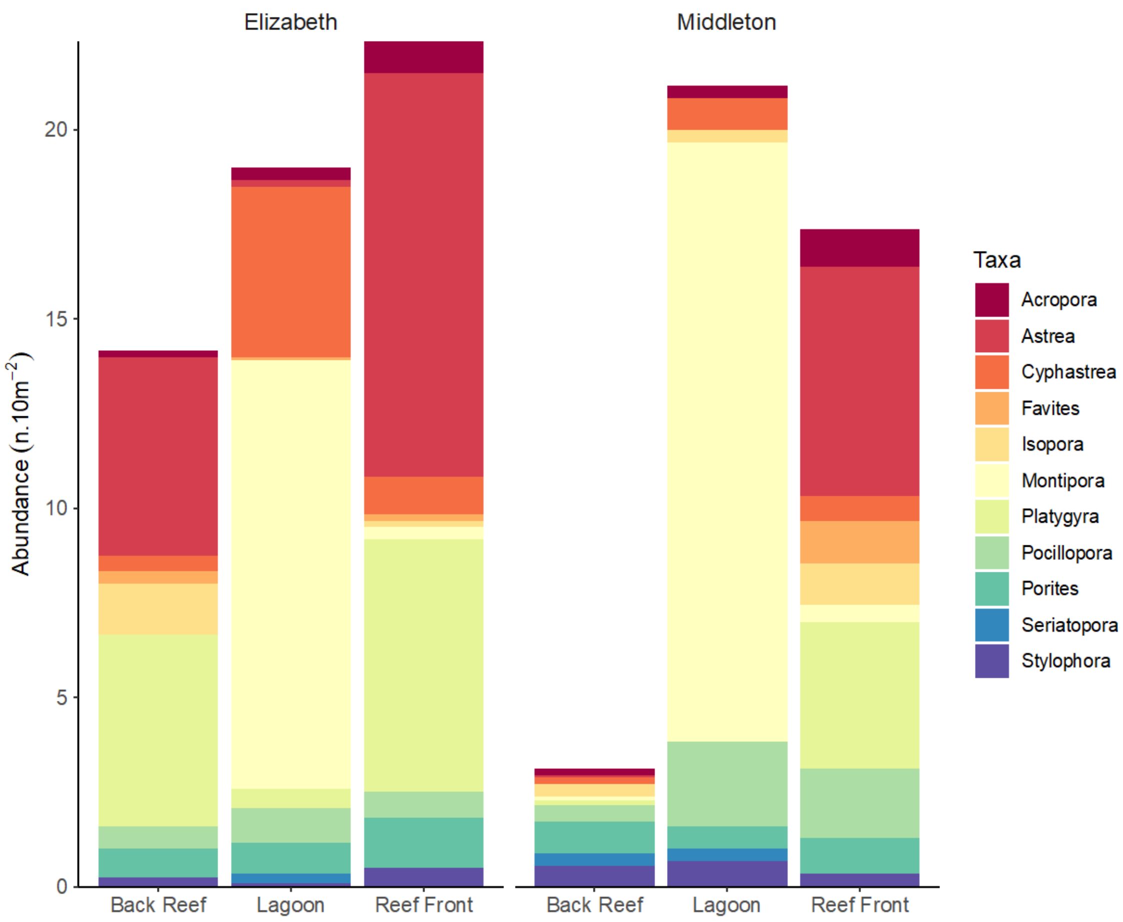

Figure 4.5 Variation in the taxonomic composition of juvenile coral assemblages among habitats on Elizabeth and Middleton Reefs in February 2024. Data are based on 10m^2 belt transects at each of the 14 sites (i.e., pooled across transects and reef zones).

4.1.2 Regional variation in coral assemblages

Comparisons with surveys conducted on five reefs in the southern Coral Sea Marine Park (CSMP) during February – March 2024 show that Elizabeth and Middleton Reefs had higher average taxonomic richness and cover of scleractinian (hard) corals than reefs in the southern CSMP, 780 – 1,100 km to the north (Figure 4.6). Importantly, average coral cover at Elizabeth and Middleton Reefs (31.1%) was almost double that of the southern CSMP (15.8%), although there was considerable variation among reefs in the southern CSMP, ranging from 4.0% at Frederick Reef to 33.7% at Cato Reef.

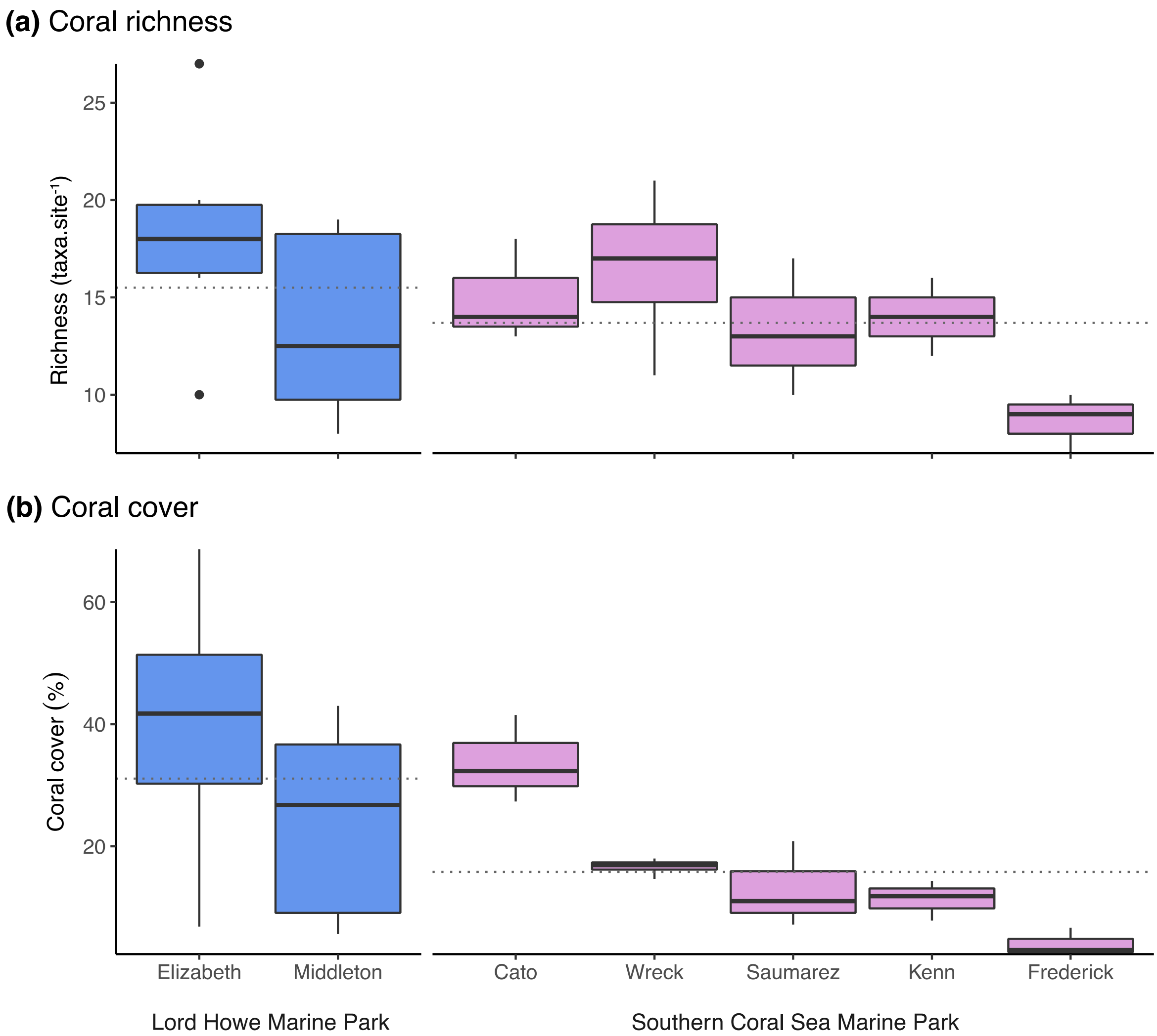

Figure 4.6 Variation in the **(a)** taxonomic richness and **(b)** cover of coral assemblages among reefs and regions (Elizabeth and Middleton Reefs and southern Coral Sea Marine Park) in February 2024. Data are based on 50m point-intercept transects at three to eight sites per reef (i.e., pooled across transects and reef zones). Dashed lines represent region averages.

The composition of coral assemblages also differed between the two regions, with *Isopora, Astrea, Cyphastrea, Platygyra,* and *Montipora* being more abundant, and *Acropora* being less abundant on Elizabeth and Middleton Reefs, compared to reefs of the southern CSMP (Figure 4.7).

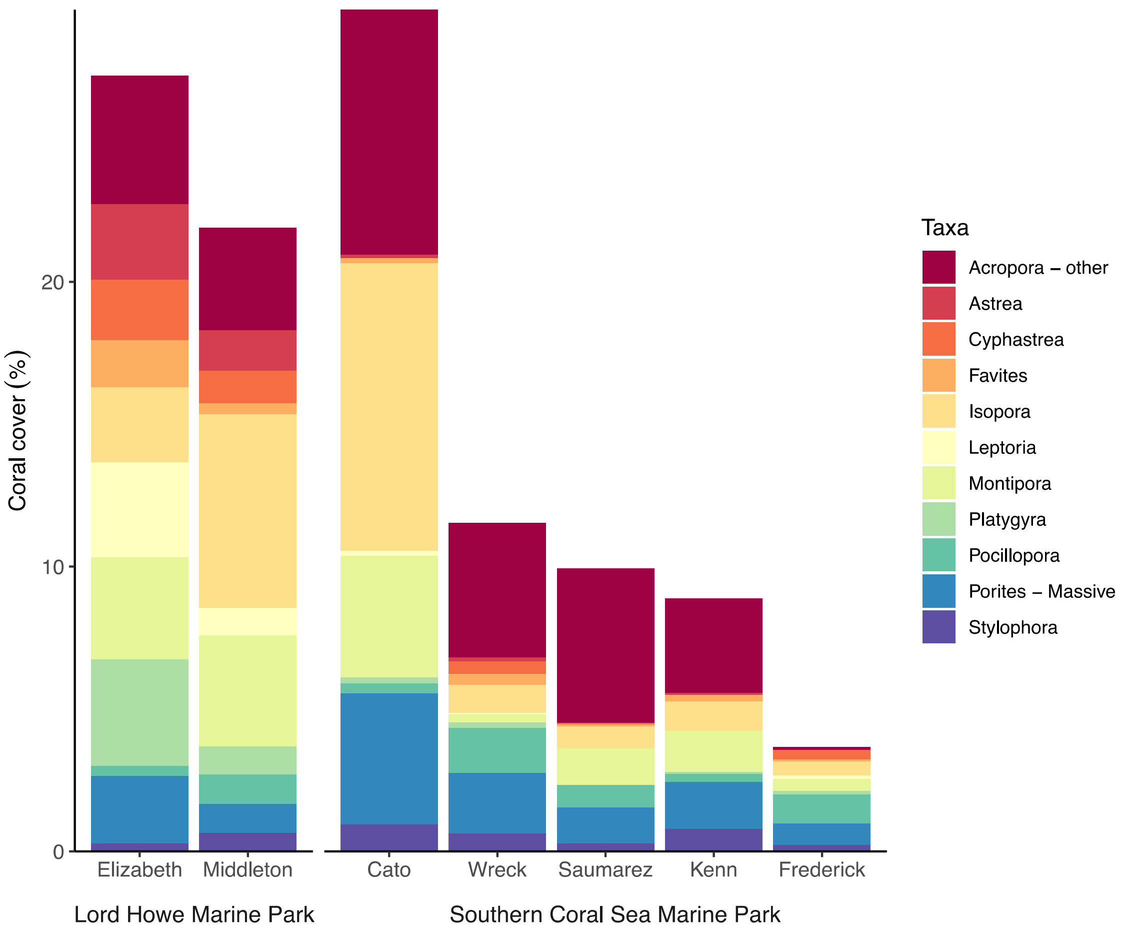

Figure 4.7 Variation in the taxonomic composition of coral assemblages among reefs and regions (Elizabeth and Middleton Reefs and southern Coral Sea Marine Park) in February 2024. Data are based on 50m point-intercept transects at three to eight sites per reef (i.e., pooled across transects and reef zones). Colours show the relative contribution of the eleven most common coral morpho-taxa.

4.1.3 Coral health

The majority corals surveyed across Elizabeth and Middleton Reefs between 9th – 14th February were healthy (86.4%), and varied from 82.6% on Middleton Reef to 89.0% on Elizabeth Reef (Figure 4.8). The percent of healthy colonies also varied among habitats and was generally greatest on the reef front (92.4 - 100%) compared to the lagoon (80.5 - 86.0%) and back reef (74.8 – 93.2%).

At the time of our surveys the majority of heat stress manifested primarily as the paling of colonies (41.2% of colonies that showed signs of heat stress), with only 13.4% of heat-stressed colonies being completely bleached (Figures 4.8). As expected, the incidence of paling and bleaching varied among coral taxa with heat sensitive taxa such as *Stylophora* (43%), *Seriatopora* (20%), *Montipora* (16%), and

Pocillopora (12%) being more affected than other taxa (Loya et al. 2001; Figure 4.9). Interestingly, the incidence of paling and bleaching among *Acropora* colonies was much lower than expected (0%) and may reflect their predominant distribution on the wave exposed reef front habitat where wave action and mixing of cooler deeper waters may have reduced the heat stress. While this level of heat stress and bleaching is relatively low, the heat stress experienced at Elizabeth and Middleton Reefs in 2024 was still building at the time of the surveys and did not reach its peak until late March (Figure 4.10).

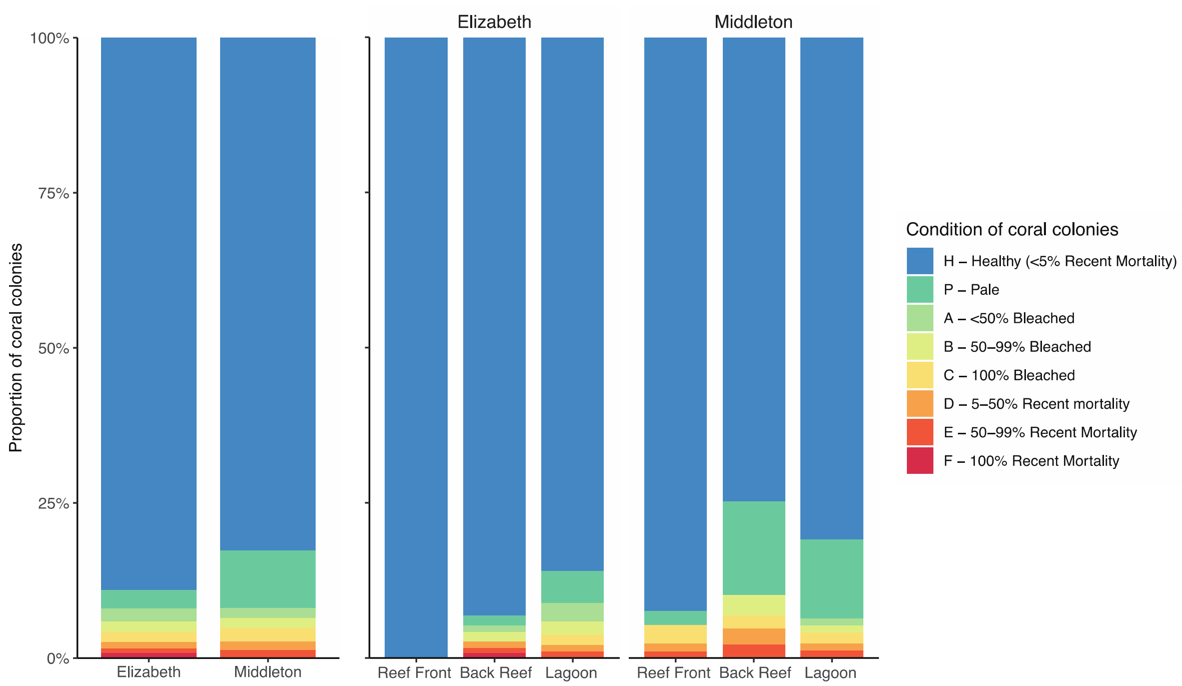

Figure 4.8 The proportion of coral colonies in each of eight health categories from 'healthy' to 'recently dead' recorded at 14 sites across Elizabeth and Middleton Reefs, Lord Howe Marine Park during 9-14 February 2024.

Importantly, the lowest incidence of bleaching in 2024 was recorded on Elizabeth Reef, with higher levels being recorded on Middleton Reef just a few days later. While this variation could be related to several factors, it is consistent with increasing heat stress over the duration of our surveys. At the time of our last surveys on Middleton Reef (14th February 2024) large areas of the southern CSMP and Lord Howe Marine Park were exposed to > 8 Degree Heating Weeks (DHW), and up to 13 DHW in some areas (Figure 4.9). Importantly, the marine heat wave continued to build through March with large areas of the southern Coral Sea and

Tasman Sea exposed to >16 DHW and up to 20 DHW in some areas. DHW is a metric that combines the intensity and duration of heat stress experienced during the previous 3 months into a single index. It is a strong predictor of bleaching in corals with DHW >4 likely to lead to significant bleaching, and DHW >8 likely to lead to significant mortality, especially in more thermally sensitive species (Hughes et al. 2017). Given the heat stress experienced on or near Elizabeth and Middleton Reefs in late March 2024, extensive and severe bleaching, and bleaching-induced mortality may be expected. Future monitoring (ideally in late 2024 or early 2025) will be critical to assess the impacts of this heat stress on shallow water coral communities of Elizabeth and Middleton Reefs.

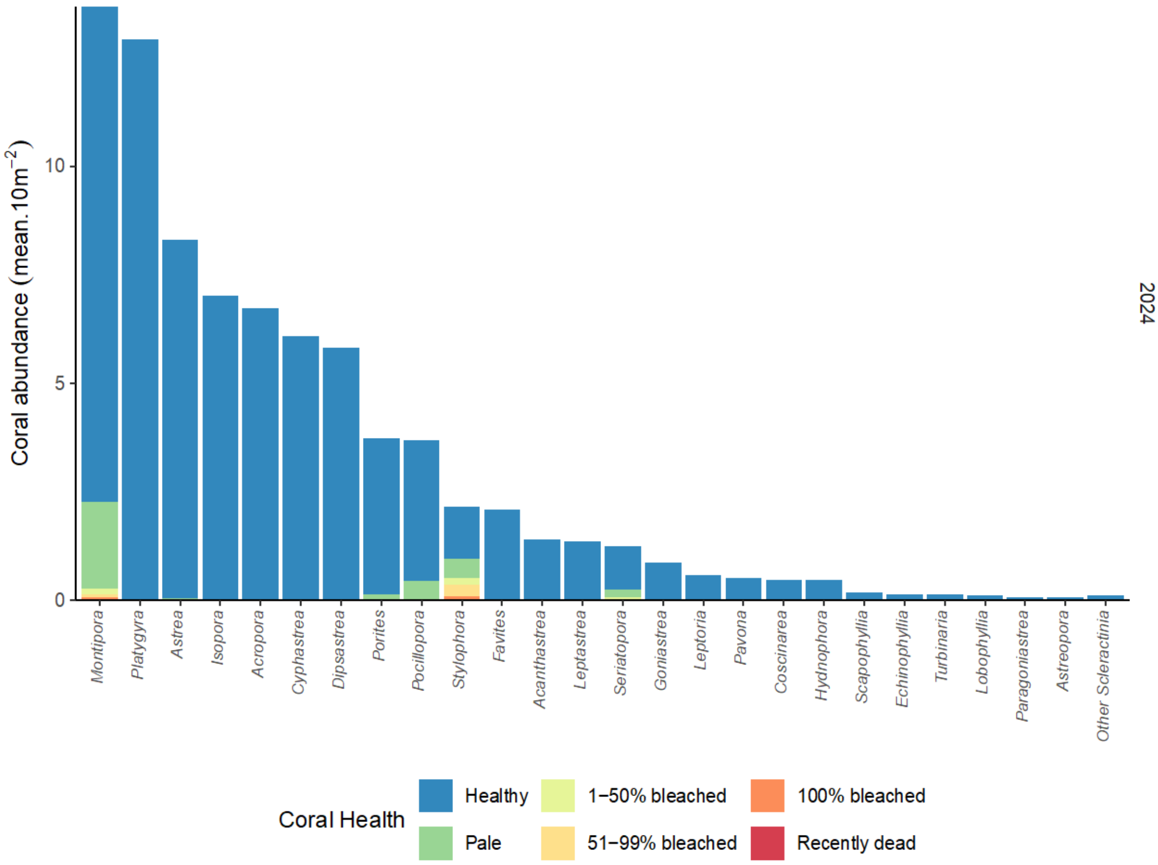

Figure 4.9 Mean density of coral colonies (per 10m^2) in the 26 most common scleractinian genera (including a pooled 'other Scleractinia' category) in each of six bleaching health categories from 'healthy' (blue) to 'recent bleaching mortality' (red) observed at sites across Elizabeth and Middleton Reefs during 9-14th February 2024.

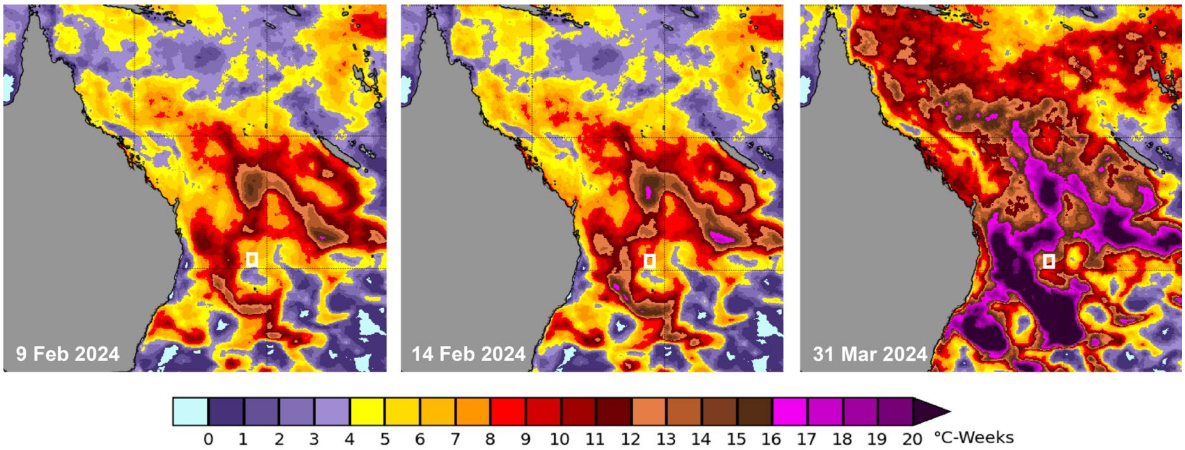

Figure 4.10 Thermal stress, measured as Degree heating weeks (DHW), in the Coral Sea and Tasman Sea for February – March 2024. The three panels show the progression of thermal stress from the start of the surveys on Elizabeth Reef (9th February) to the end of the surveys on Middleton Reef (14th February), and the maximum heat stress recorded in 2024 (31st March 2024). The white rectangle shows the approximate location of Elizabeth and Middleton Reefs. Images produced using the NOAA CRW 5km product v3.1

4.2 Non-coral invertebrates

Coral predators - Densities of coral-feeding invertebrates, specifically crown-of-thorns starfish (*Acanthaster* cf. *solaris*), the pin cushion starfish (*Culcita novaeguineae*) and coral snails (*Drupella* spp.) were negligible across Elizabeth and Middleton Reefs in 2024. In 2024, only a single pincushion starfish, and no crown-of-thorns starfish or *Drupella* were recorded across all 69 transects. These numbers are lower than were recorded in 2011, 2014 and 2018, though abundance of coral predators have been consistently low across all surveys (Table 4.2). This low abundance of coral predators likely reflects the isolation of these reefs and hence the limited supply of larvae from other reefs.

Table 4.2 Temporal variation (2011-2024) in average coral cover between reefs and habitats on Elizabeth and Middleton Reefs, Lord Howe Marine Park. Data are based on replicate 50m point-intercept transects within each of two reef zones (reef crest and slope) at 13 sites.

Species	2011	2014	2018	2024
Sites surveyed	16	16	19	14
Crown-of-thorns starfish (*Acanthaster cf. solaris)*	1	4	1	0
Pin cushion starfish (*Culcita novaeguineae*)	3	6	1	1
Coral Snails (*Drupella* spp.)	5	3	2	0

Sea cucumbers (Holothurians) - A total of 240 sea cucumbers from 11 species were recorded across all transects in 2024, corresponding to a mean density of 3.04 (± 1.16 SE) individuals per 200m^2, which is lower than those recorded the same sites in 2018 and 2014 (3.95 ± 1.61 and 3.25 ± 1.14 individuals per 200m^2, respectively), but greater than estimates from 2011 (2.11 ± 1.07 individuals per 200m^2; Figure 4.10a). The decline in density of sea cucumbers from 2018 to 2024 were largely driven by the lagoon sites at both reefs, with densities of sea cucumbers declining by 45.2% in the lagoon at Elizabeth Reef (2018: 8.3 individuals per 200m^2; 2024: 4.6 individuals per 200m^2) and by 40.3% in the lagoon at Middleton Reef (2018: 13.7 individuals per 200m^2; 2024: 8.2 individuals per 200m^2). In contrast, the density of sea cucumbers almost doubled at the back reef sites on Middleton Reef (2018: 3.5 individuals per 200m^2; 2024: 6.8 individuals per 200m^2; Figure 4.11a). These temporal changes in the density of sea cucumbers at Elizabeth and Middleton Reefs likely reflect natural variability in populations (Uthicke et al. 2009), rather than the effects of external pressures, namely harvesting. Similar to previous surveys, the dominant species of sea cucumber were *Holothuria edulis* (125 out of 240 individuals; 52%) and *Holothuria atra* (61 out of 240 individuals; 25%). The remaining nine species only accounted for 23% of individuals recorded (54 individuals).

Sea urchins (Echinoids) – Overall, 1,164 long-spined sea urchins (mostly *Diadema savigni*) were recorded across the 14 sites at Elizabeth and Middleton Reefs in 2024, equating to an average density of 16.9 (± 5.4 SE) urchins per

200m^2. The density of urchins was 5-fold greater on Middleton Reef (26.9 urchins per 200m^2) than Elizabeth Reef (5.2 urchins per 200m^2; Figure 4.11b) and was largely driven by high densities of *D. savigni* on the back reef of Middleton Reef (in particular Middleton site 2: 159.1 urchins per 200m^2). The densities of *D. savigni* within the back reef habitat at Middleton Reef, and hence the densities of sea urchins on Middleton Reef, have displayed considerable variability among surveys, ranging from 47.9 to 261.8 urchins per 200m^2 in 2011 and 2014, respectively.

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 cover and biomass of macroalgae, and thereby prevent overgrowth of reefs by macroalgae (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).

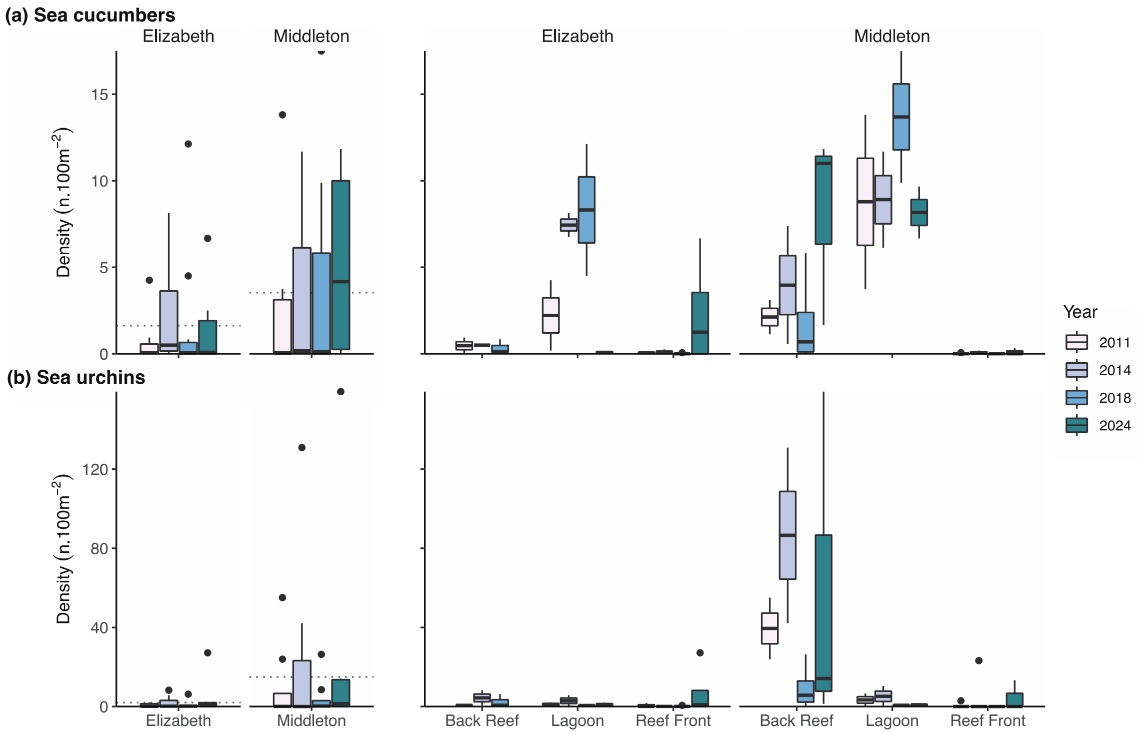

Figure 4.11 Spatial and temporal variation (2011-2024) in the density of **(a)** sea cucumbers (Holothuroidea), and **(b)** sea urchins (Echinoidea) on Elizabeth and Middleton Reefs. Data are based on replicate 50 x 2m belt transects within each of two reef zones

(reef crest and slope) at 13 sites. Dashed lines on left-hand panels show reef averages and right-hand panel shows variation among habitats within each reef.

Giant Clams – Twenty giant clams (*Tridacna* spp.) were recorded across the 14 sites on Elizabeth and Middleton Reefs in 2024, equating to an average density of 0.29 clams per 100m^2. The vast majority of giant clams recorded were *Tridacna maxima* and *Tridacna squamosa* (16 individuals, 80%), with the *Tridacna crocea* (3 individuals, 15%) and *Tridacna derasa* (1 individual, 5%) being the only other species recorded. Although giant clams were not recorded during previous surveys of Elizabeth and Middleton Reefs, the average density recorded on these reefs in 2024 was an order of magnitude lower than the densities recorded across 18 reefs in the Coral Sea Marine Park in 2023/24 (2.7 clams per 100m^2). The lower densities of giant clam on Elizabeth and Middleton Reefs appear comparable to those of other subtropical reefs (e.g., Solitary Islands: Smith 2011).

4.3 Coral reef fishes

The richness, density, and biomass of reef fishes were generally lower on Middleton than Elizabeth Reef, although there was greater variation among habitats and sites within each reef than between the two reefs (Figure 4.12). The species richness of reef fishes ranged from 33 species per site at Middleton site 6 (reef front) to 71 species per site at Elizabeth site 7 (lagoon). Within both reefs, average species richness was generally lowest within the reef front habitat (Elizabeth Reef: 40 species per site; Middleton Reef: 46 species per site), and higher within the lagoon (Elizabeth Reef: 57 species per site; Middleton Reef: 56 species per site) and back reef habitats (Elizabeth Reef: 64 species per site; Middleton Reef: 53 species per site; Figure 4.12a). These differences in the species richness of reef fishes among habitats contrast with those of corals in which the highest taxonomic richness of corals was recorded within the reef front habitat (Figure 4.1). Notably, the differences in species richness among habitats was more pronounced on Elizabeth Reef (range: 40 to 64 species per site) than Middleton Reef (46 to 56 species per site).

Average density of reef fish densities was approximately 25% higher on Elizabeth Reef (169.8 individuals per 100 m^2) compared to Middleton Reef (142.2 individuals per 100 m^2; Figure 4.12b). Like fish species richness there was considerable variation in the density of reef fish recorded among habitats and sites within each reef, ranging from 57.1 individuals per 100 m^2 at Middleton site 3 (lagoon) to 244.1 individuals per 100 m^2 at Elizabeth site 3 (reef front). In contrast to species richness, average density of reef fish was highest within the reef front habitat (Elizabeth Reef: 231.8 individuals per 100 m^2; Middleton Reef: 229.0 individuals per 100 m^2), intermediate within the back reef (Elizabeth Reef: 178.1 individuals per 100 m^2; Middleton Reef: 92.2 individuals per 100 m^2) and lowest in the lagoon habitat (Elizabeth Reef: 99.3 individuals per 100 m^2; Middleton Reef: 86.7 individuals per 100 m^2; Figure 4.12b).

The average biomass of reef fish was remarkably consistent between the two reefs (Elizabeth: 38.7 kg per 100m^2; Middleton 38.9 kg per 100m^2), but varied almost 20-fold among sites (Middleton site 1: 5.3 kg per 100m^2; Middleton site 8: 99.1 kg per 100m^2; Figure 4.12c). Average reef fish biomass was greatest on the reef front of both reefs (Elizabeth: 53.9 kg per 100m^2 Middleton: 54.3 kg per 100m^2), and was lowest in the lagoon at Elizabeth Reef (23.1 kg per 100m^2) and in the back reef at Middleton Reef (24.1 kg per 100m^2; Figure 4.12c).

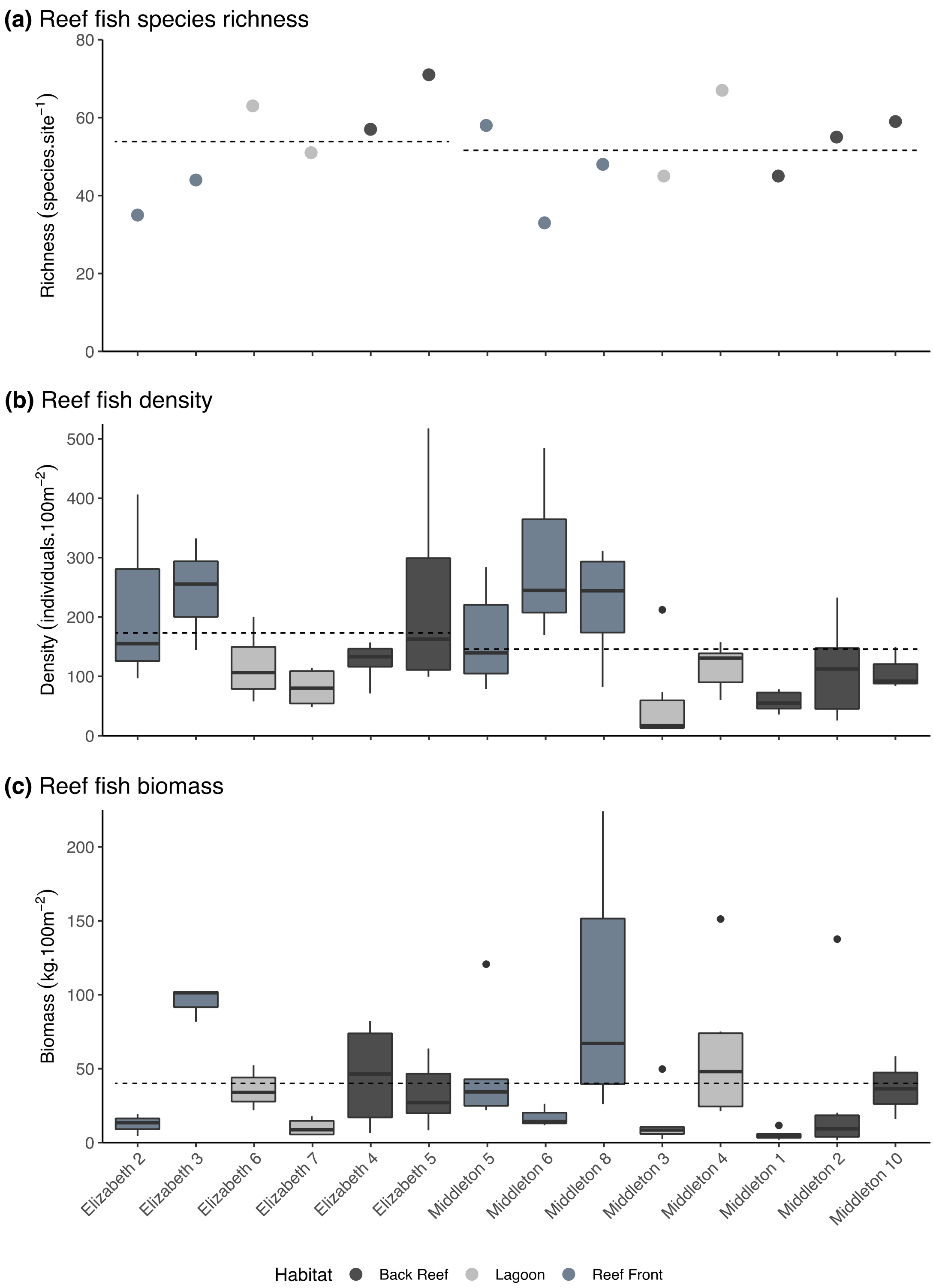

Figure 4.12 Variation in **(a)** richness, **(b)** density, and **(c)** biomass of reef fish assemblages among 14 sites on Elizabeth and Middleton Reefs in February 2024. Data are based on the 50m belt transects, with data for richness based on the number of fish species recorded at each of the 14 sites (i.e., pooled across transects and reef zones). Dashed lines represent reef averages.

Functional composition of reef 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 (e.g., fusiliers and some damselfishes) and omnivorous fishes (e.g., some damselfishes) were the most abundant functional groups on Elizabeth and Middleton Reefs accounting for 34.0% and 21.1% of all fish recorded, respectively, across the 14 sites surveyed in February 2024 (Figure 4.13a). In contrast, these two groups of predominantly small-bodied fishes collectively accounted for less than 4% of total fish biomass, with large-bodied carnivores (31.3%), grazing herbivores (24.8%), and browsing herbivores (23.8%) being the dominant groups in terms of fish biomass (Figure 4.13b). The biomass of grazing herbivores (e.g., some surgeonfishes) tended to be greatest on the reef front, while the biomass of browsing herbivores (e.g., kyphosids) was greatest on the reef front and back reef habitats. The biomass of carnivores was relatively evenly distributed across reefs and habitats, the only exception being the lagoon habitat on Middleton Reef where a large number of Galapagos reef sharks (*Carcharhinus galapagensis*) were recorded (Figure 4.13b).

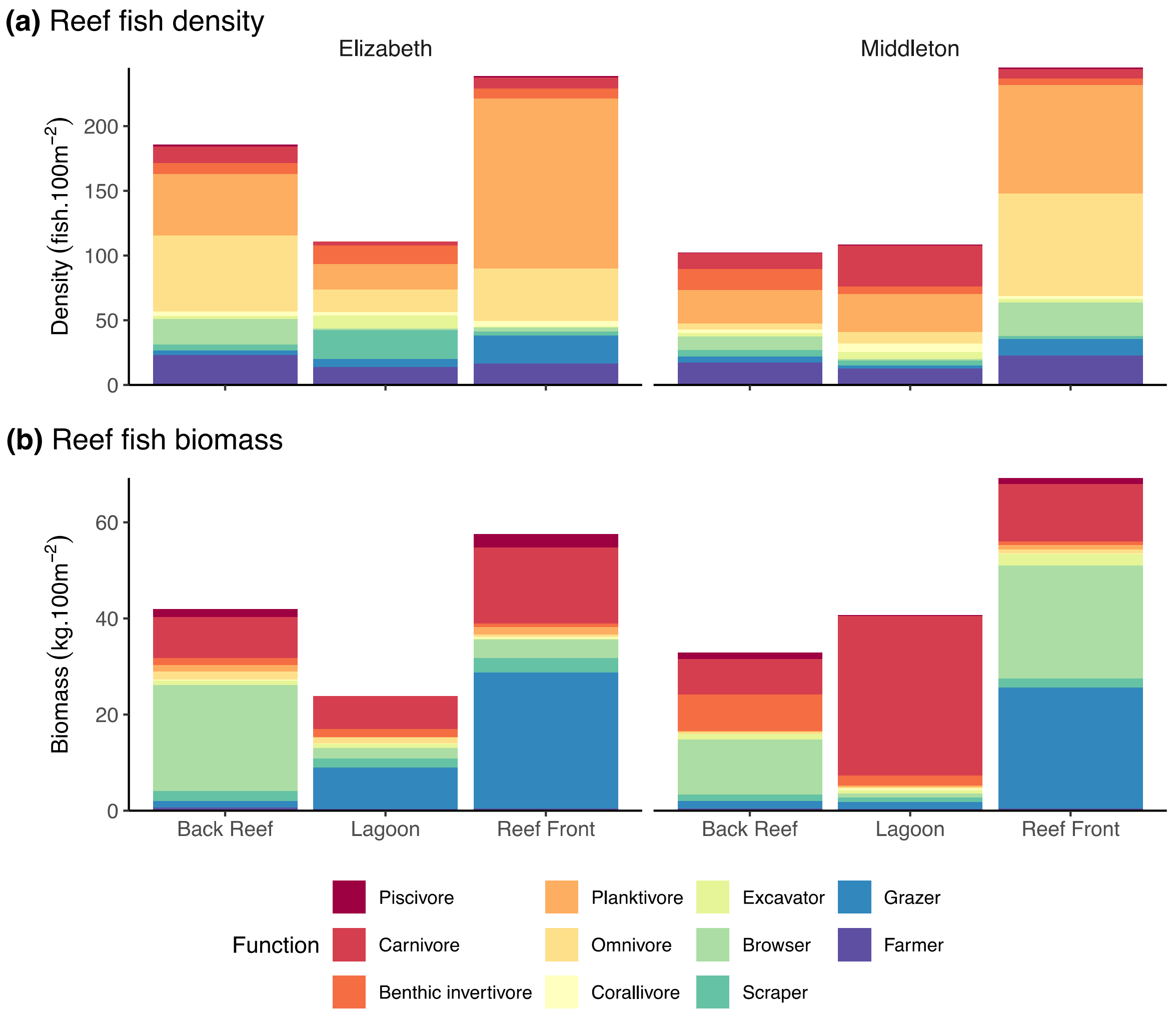

Figure 4.13 Variation in functional composition of reef fish assemblages among habitats on Elizabeth and Middleton Reefs in February 2024. Data are based on replicate 50m belt transects at 14 sites (i.e., pooled across transects and reef zones). Colours represent the eleven functional groups identified

4.3.1 Herbivorous fishes

The average abundance of herbivorous fishes ranged from 23.1 (± 5.4 SE) individuals per 100m^2 on Middleton Reef to 30.52 (± 5.6 SE) individuals per 100m^2 on Elizabeth Reef (Figure 4.14a), while average biomass of herbivorous fishes was broadly comparable between reefs (Middleton Reef: 19.7 ± 6.6 kg per 100m^2; Elizabeth Reef: 22.0 ± 4.4 kg per 100m^2; Figure 4.14b). Both the abundance and biomass of herbivorous fishes varied among habitats and between reef zones, with greater fish abundance on the reef crest than the reef slope, and generally higher biomass on the reef front than the lagoon or back reef habitats (Figure 4.14). The moderate abundance, but low biomass, of herbivorous fishes within the lagoon habitats is reflective of the predominance of smaller-bodied individuals (especially juvenile fishes) in this habitat and suggests that this environment likely provides an important nursery habitat.

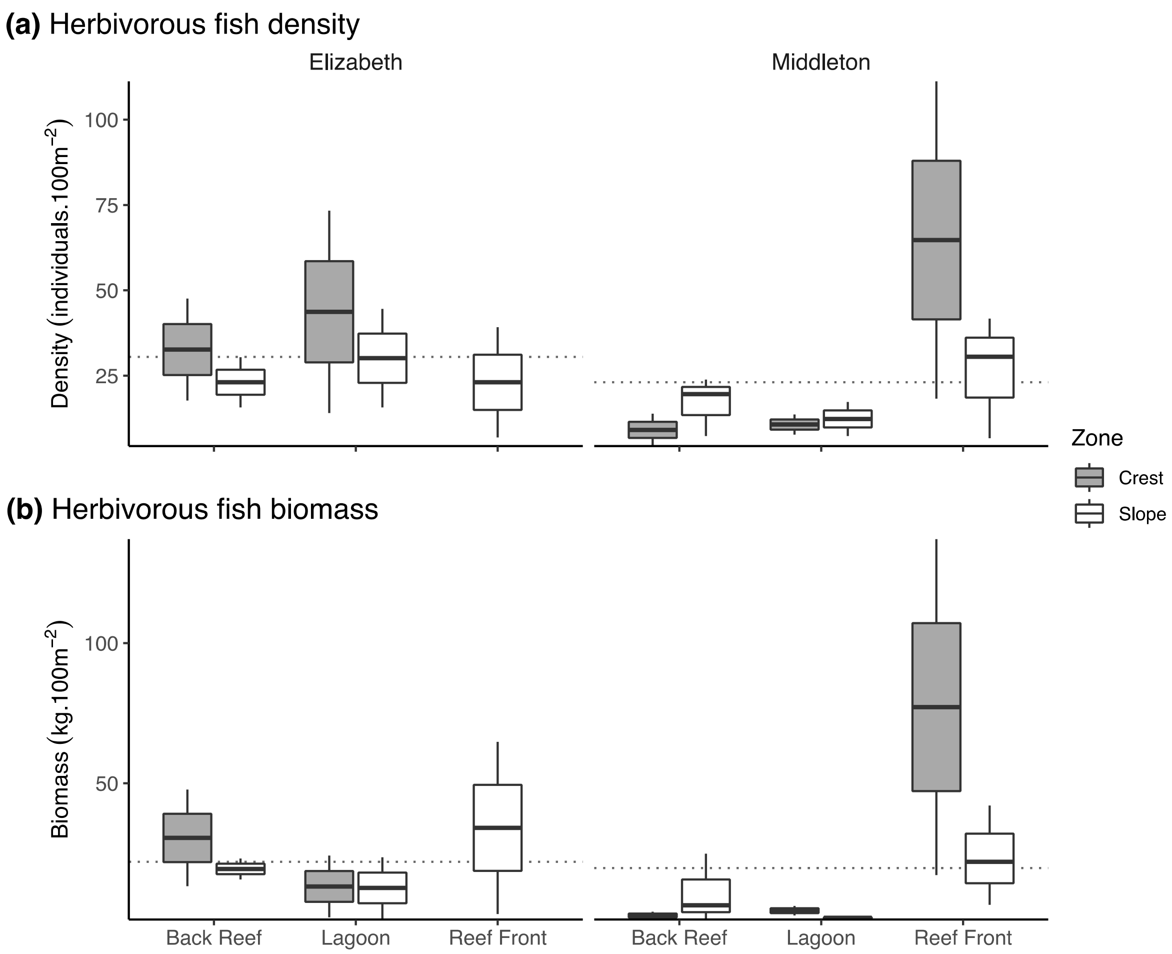

Figure 4.14 Variation in **(a)** density, and **(b)** biomass of herbivorous reef fish assemblages among habitats (reef front, lagoon and back reef) and between zones (reef slope vs reef crest) on Elizabeth and Middleton Reefs in February 2024. Data are based on replicate 50m belt transects within each zone at each site. Dashed lines represent reef averages.

Temporal changes in herbivorous reef fish assemblages

The density of herbivorous has remained relatively stable on Elizabeth Reef from 2011 to 2024 (2011: 27.4 individuals per 100m^2, 2014: 22.2 individuals per 100m^2, 2018: 26.6 individuals per 100m^2, 2024: 29.3 individuals per 100m^2), and initially declined (2011-2014) and then stabilised (2014-2024) on Middleton Reef over the same time period (2011: 31.0 individuals per 100m^2, 2014: 25.6 individuals per 100m^2, 2018: 23.3 individuals per 100m^2, 2024: 22.2 individuals per 100m^2; Figure

4.15b). While the density of herbivorous fishes was relatively stable across the three habitats on Elizabeth Reef, decreases in the density of herbivorous fishes within the lagoon and back reef habitats on Middleton Reef, were offset by an increase on the reef front (Figure 4.15b).

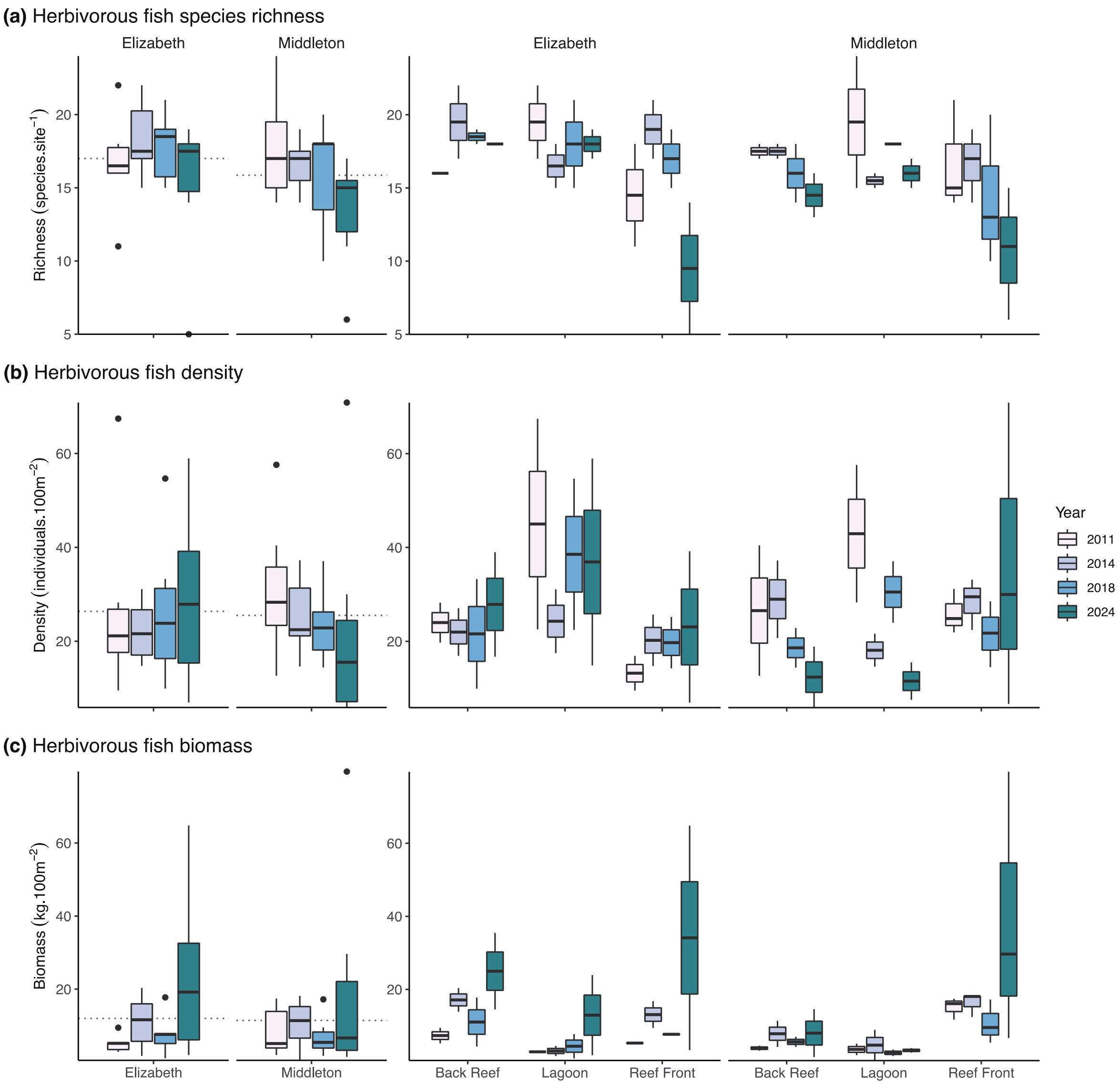

Figure 4.14 Temporal variation (2011-2024) in **(a)** richness, **(b)** density, and **(c)** biomass of herbivorous reef fish assemblages among habitats on Elizabeth and Middleton Reefs. Data are based on 50m belt transects at matched sites and pooled across reef zones (i.e., reef slope and reef crest). Dashed lines on the left-hand panels represent reef averages.

Despite the density of herbivorous fishes being relatively stable from 2011 to 2024, there was a large increase in the biomass of herbivorous fishes from 2018 to 2024

(Elizabeth: 2011: 5.1 kg per 100m^2; 2014: 11.1 kg per 100m^2; 2018: 7.7 kg per 100m^2; 2024: 24.0 kg per 100m^2; Middleton: 2011: 8.6 kg per 100m^2; 2014: 10.5 kg per 100m^2; 2018: 6.9 kg per 100m^2; 2024: 19.8 kg per 100m^2; Figure 4.15c). These increases in biomass were largely attributable to increases in the biomass of browsing fishes (primarily the Pacific Chub *Kyphosus sectatrix* and the Spotted Sawtail (*Prionurus maculatus*) on the reef front of each reef (Figure 4.15c).

4.3.2 Endemic fishes

McCulloch's anemonefish – McCulloch's anemonefish, *Amphiprion mccullochi*, (Figure 4.16) was only observed in the lagoon habitat at both reefs in 2024, with the average density being higher at Elizabeth (1.75 ± 1.26 individuals per 100m^2) than Middleton Reef (0.75 ± 0.59 individuals per 100m^2; Figure 4.17a). *A. mccullochi* is a habitat specialist, being only found in close association with the bubble-tip anemone *Entacmaea quadricolor*. The distribution of *A. mccullochi* among habitats is, therefore, likely driven by the availability of its host anemone.

The average density of *A. mccullochi* has remained relatively stable or increased on Elizabeth and Middleton Reefs from 2011 to 2024 (Elizabeth 2011: 0.23 individuals per 100m^2; 2014: 0.29 individuals per 100m^2; 2018: 0.52 individuals per 100m^2; 2024: 0.70 individuals per 100m^2; Middleton 2011: 0.03 individuals per 100m^2; 2014: 0.33 individuals per 100m^2; 2018: 0.04 individuals per 100m^2; 2024: 0.23 individuals per 100m^2; Figure 4.18a). It should be noted that some *A. mccullochi* individuals were recorded on the back reef of both reefs in 2014, however they haven't been recorded in this habitat in subsequent surveys.

Three-striped butterflyfish – The average density of the endemic three-striped butterflyfish *Chaetodon tricinctus* (Figure 4.16) was approximately 3-fold greater on Elizabeth Reef (1.42 ± 0.40 individuals per 100m^2) than on Middleton Reef (0.46 ± 0.14 individuals per 100m^2) in 2024 (Figure 4.17b). There was considerable variation in the density of *C. tricinctus* among habitats on each reef, ranging from 0.07 individuals per 100m^2 on the back reef of Middleton Reef (coinciding with low coral cover and high macroalgal cover) to 3.33 individuals per 100m^2 on the reef front of Elizabeth Reef (coinciding with the highest coral cover recorded in 2024;

Figure 4.2). Although there is some debate over the diet of *C. tricinctus* (Pratchett et al. 2014), its distribution among habitats suggests it relies on live coral for food and/or shelter. Like many other butterflyfish species, such reliance on live coral will likely render populations of *C. tricinctus* to be susceptible to changes in live coral cover.

Figure 4.16 Photographs of McCulloch's anemonefish *Amphiprion mccullochi* (top) and the three-striped butterflyfish *Chaetodon tricinctus* (bottom). Image credits: Victor Huertas

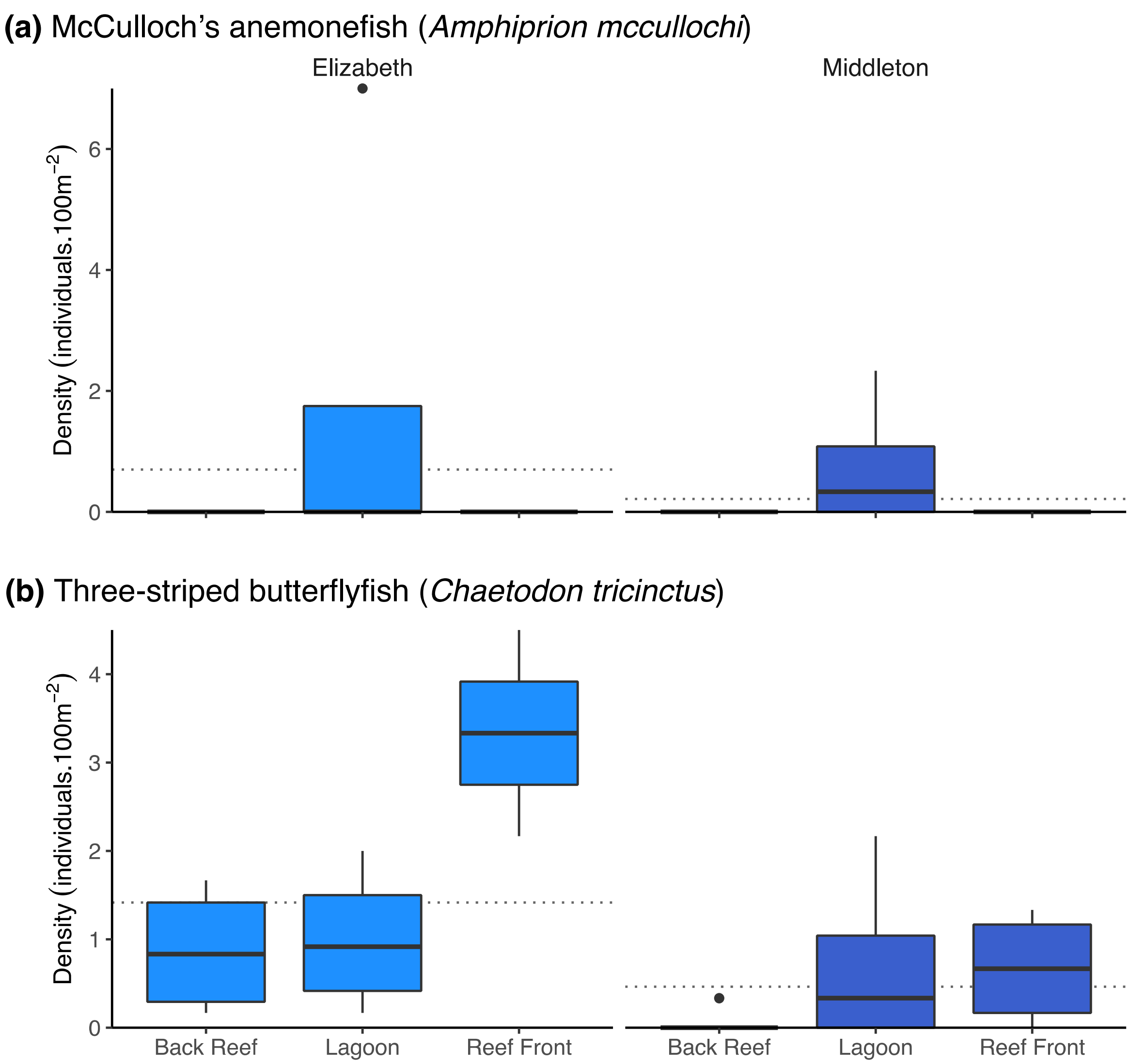

Figure 4.17 Variation in the density of **(a)** McCulloch's anemonefish *Amphiprion mccullochi*, and **(b)** three-striped butterflyfish *Chaetodon tricinctus* among habitats on Elizabeth and Middleton Reefs in February 2024. Data are based on replicate 50m belt transects within each site. Dashed lines represent reef averages.

There have been contrasting temporal trends in the density of *C. tricinctus* on Elizabeth and Middleton Reefs between 2011 and 2024. The average density of *C. tricinctus* has steadily increased from 2011 (0.80 individuals per 100m^2) to 2024 (1.42 individuals per 100m^2) on Elizabeth Reef, and was largely due to increases in density on the reef front habitat (Figure 4.18b). In contrast, the average density of *C. tricinctus* initially increased from 2011 to 2014 (0.38 and 1.13 individuals per 100m^2, respectively), before declining to 0.47 individuals per 100m^2 in 2024. This decline was evident across all three habitats and was most pronounced in the back reef (Figure 4.18b).

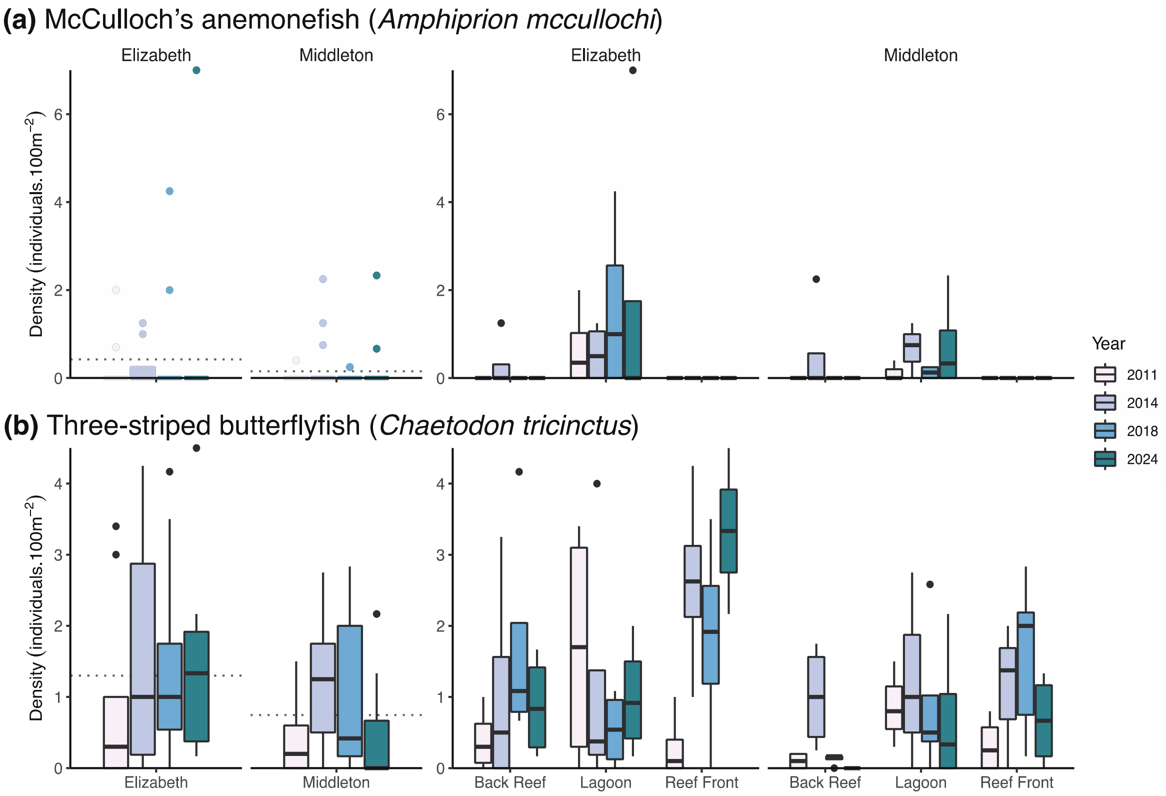

Figure 4.18 Temporal variation (2011-2024) in the density of **(a)** McCulloch's anemonefish *Amphiprion mccullochi*, and **(b)** three-striped butterflyfish *Chaetodon tricinctus* among habitats on Elizabeth and Middleton Reefs. Data are based on 50m belt transects at matched sites and pooled across reef zones (i.e., reef slope and reef crest). Dashed lines on the left-hand panels represent reef averages.

Doubleheader wrasse (*Coris bulbifrons*)

Adult doubleheader wrasse *Coris bulbifrons* (Figure 4.19) were relatively abundant across both Elizabeth and Middleton Reefs in 2024, with 90 individuals being recorded across all transects. Juvenile *C. bulbifrons* were, however, relatively rare with only 3 individuals recorded in 2024, compared to 39 juveniles in 2018 and 14 juveniles in 2014. Juvenile *C. bulbifrons* were recorded in the back reef and lagoon habitats of Middleton Reef (Figure 4.20a), while adult *C. bulbifrons* were recorded across all habitats at both reefs (Figure 4.20b). Overall, the density of adult *C. bulbifrons* was greater on Middleton (1.7 individuals per 100m^2) than Elizabeth

Reef (0.6 individuals per 100m^2), and higher within the lagoon and back reef habitats than the reef front within each reef (Figure 4.20b). Exceptionally high densities of adult *C. bulbifrons* (average: 17.9 individuals per 100m^2; range: 8.4 - 29.6 individuals per 100m^2) were recorded on the reef slope at Middleton site 10 (Figure 4.19; a ‘new’ site that had not previously been surveyed in 2011, 2014, or 2018) and likely represented a spawning aggregation for this species.

Figure 4.19 Photographs of an adult doubleheader wrasse *Coris bulbifrons* (top), and the aggregation of adult *C. bulbifrons* on the reef slope at Middleton site 10 (bottom). Image credits: Victor Huertas

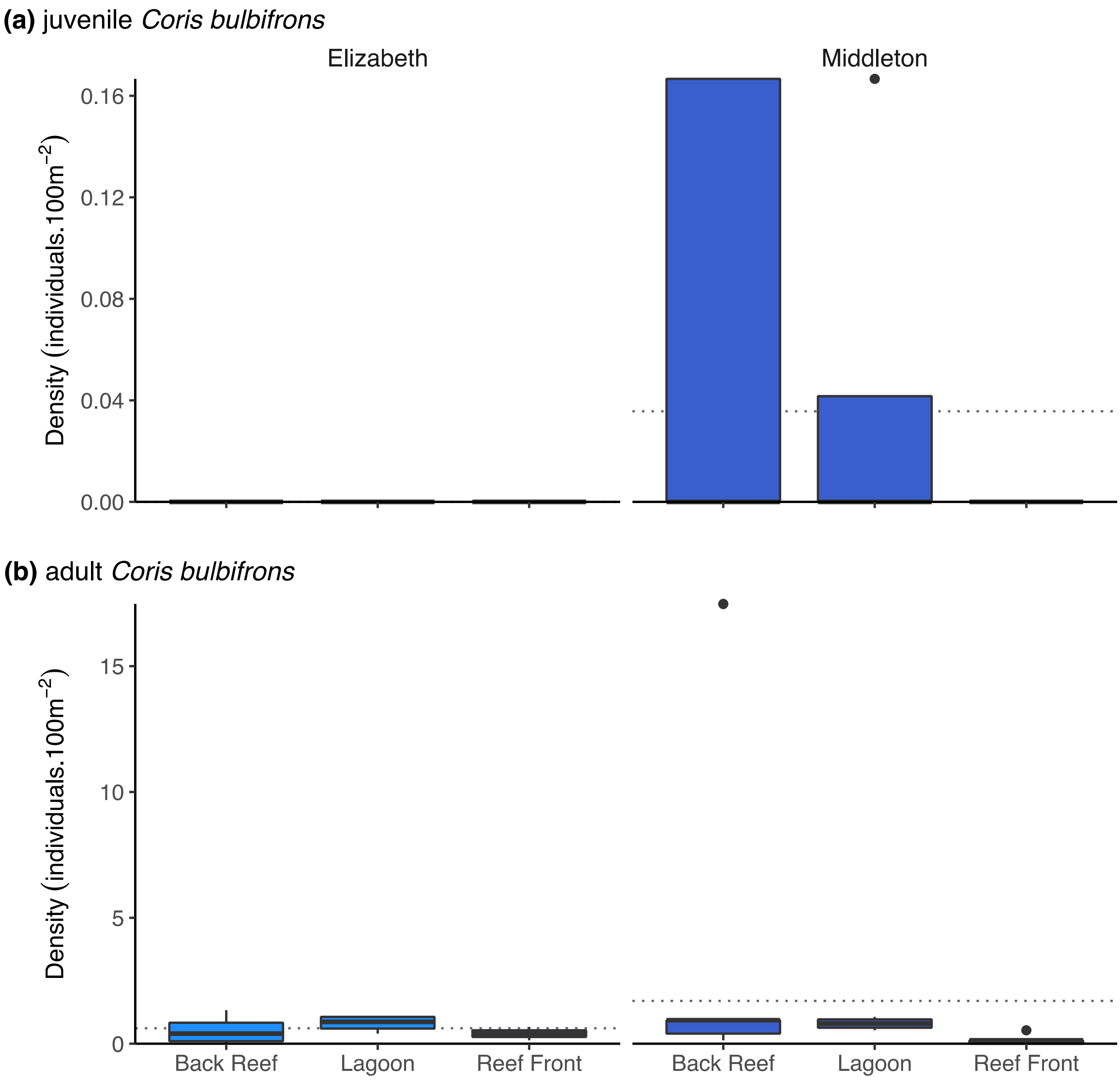

Figure 4.20 Variation in the density of **(a)** juvenile and **(b)** adult doubleheader wrasse *Coris bulbifrons* among habitats on Elizabeth and Middleton Reefs in February 2024. Data are based on replicate 50m belt transects within each site. Note the difference scales on the y-axis of each plot, and the extreme outlier in **(b)** showing high density of adult *C. bulbifrons* at one back reef site on Middleton Reef. Dashed lines represent reef averages.

Comparisons of sites that have been surveyed in 2011, 2014, 2018 and 2024 (i.e., excluding Middleton site 10) show that the average densities of juvenile *C. bulbifrons* have been low and variable among years on each reef (Elizabeth: 0.0 – 0.64 individuals per $100m^2$; Middleton: 0.03 - 0.16 individuals per $100m^2$) and within each habitat (Figure 4.21a). This variability likely reflects the naturally temporal variation in the recruitment of reef fish (e.g., Sale et al. 1984). In contrast, the average densities of adult *C. bulbifrons* have been remarkably stable on both reefs, and within habitats on each reef, over the past 13 years, ranging from 0.34 to 0.61 individuals per $100m^2$ on Elizabeth Reef and 0.49 to 0.85 individuals per $100m^2$ on Middleton Reef (Figure 4.21b).

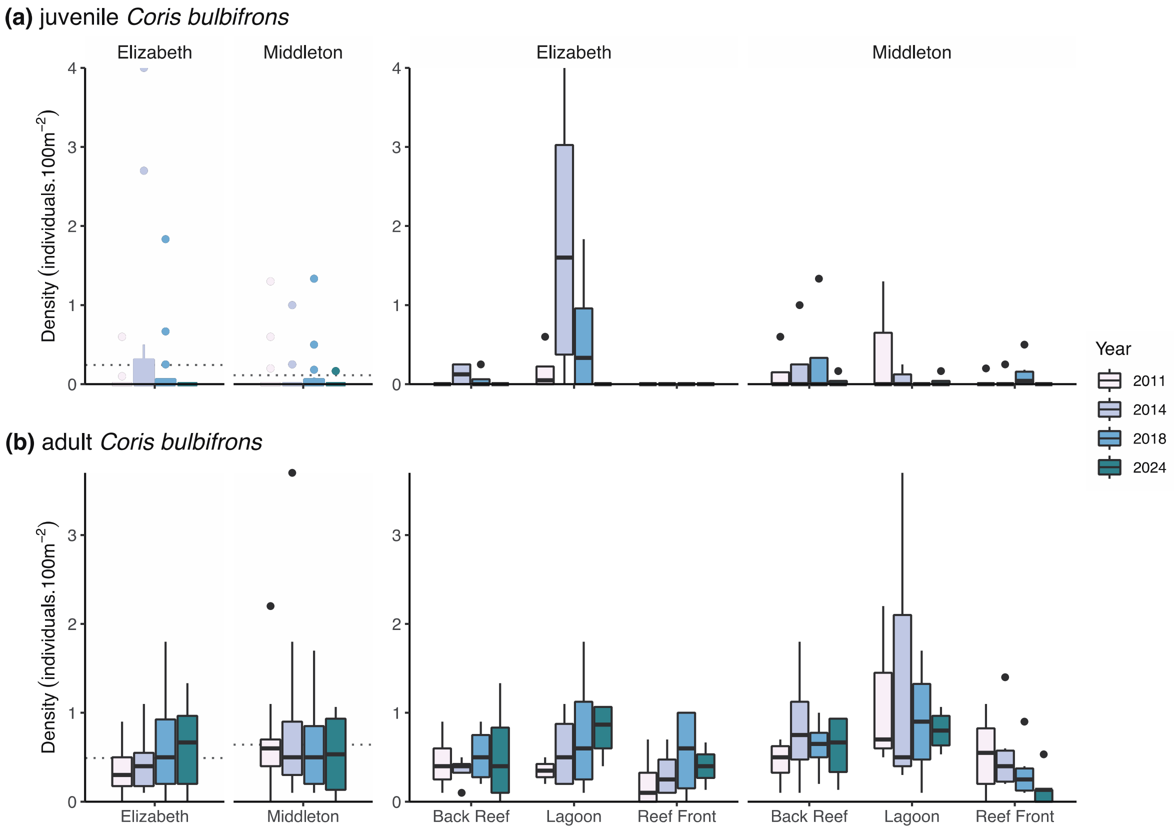

Figure 4.21 Temporal variation (2011-2024) in the density of **(a)** juvenile and **(b)** adult doubleheader wrasse *Coris bulbifrons* among habitats on Elizabeth and Middleton Reefs. Data are based on 50m belt transects at matched sites and pooled across reef zones (i.e., reef slope and reef crest). Dashed lines on the left-hand panels represent reef averages.

Galapagos shark (*Carcharhinus galapagensis*)

High densities of the Galapagos shark (*Carcharhinus galapagensis*) were again recorded at Elizabeth (13.1 ± 3.1individuals per hectare) and Middleton Reefs (24.5 ±12.9 individuals per hectare) in 2024 (Figures 4.22a, 4.23). These densities represent a small (~5%) increase on Elizabeth Reef, and a 2.5-fold increase on Middleton Reef since the previous surveys (i.e., from 2018 to 2024; Figure 4.22a). The large increase in density of *C. galapagensis* on Middleton Reef was driven by increases in the lagoon (2018: 30.0 individuals per hectare; 2024: 48.3 individuals per hectare) and reef front habitats (2018: 6.3 individuals per hectare; 2024: 23.3 individuals per hectare). The vast majority of *C. galapagensis* recorded in 2024 were relatively small (average total length 108 cm, range: 50 – 160 cm) with the largest individual recorded being 160 cm long, and is consistent with previous surveys. Given that *C. galapagensis* are born at 60-80 cm, do not mature until 210-250 cm, and reach up to 300 cm in length (Last and Stevens 1994), the length range observed at Elizabeth and Middleton indicates that the large populations in shallow reef habitats at Elizabeth and Middleton are immature individuals, and that these reefs are likely to represent an important nursery area for this species.

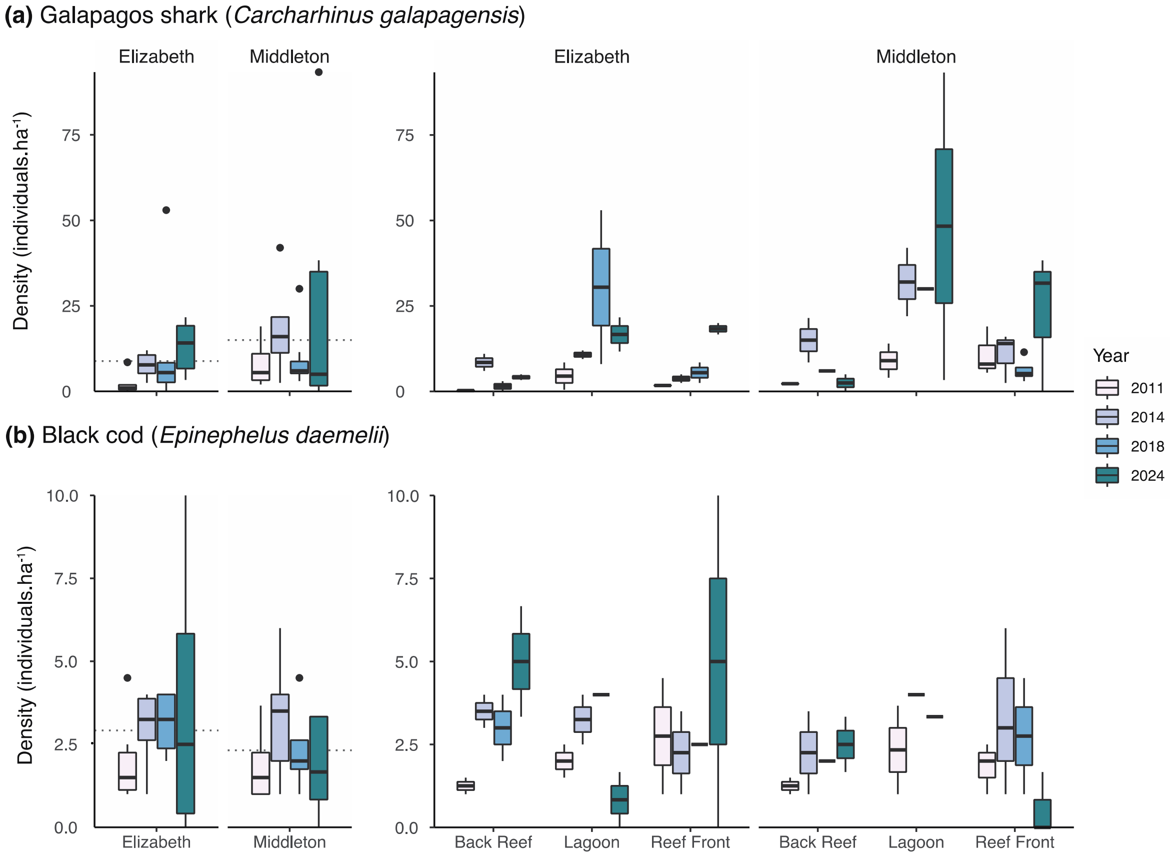

Figure 4.22 Temporal variation (2011-2024) in the density of **(a)** Galapagos shark *Carcharhinus galapagensis* and **(b)** Black cod *Epinephelus daemelii* among habitats on Elizabeth and Middleton Reefs. Data are based on 50m belt transects at matched sites and pooled across reef zones (i.e., reef slope and reef crest). Dashed lines on the left-hand panels represent reef averages.

Figure 4.23 Photgraphs of Galapagos sharks *Carcharhinus galapagensis* (top) and Black cod *Epinephelus daemelii* (bottom). Image credits: Victor Huertas

Black cod (*Epinephelus daemelii*)

Black cod (*Epinephelus daemelii;* Figure 4.23), listed as 'vulnerable' under Commonwealth environmental legislation and 'near threatened' on the IUCN Red List, were recorded across all habitats at Elizabeth and Middleton Reefs (Figure 4.22b). The average abundance of *E. daemelii* across all transects in 2024 was 2.69 (± 0.81 SE) individuals per hectare, and was directly comparable to density estimates across the same sites in 2018 (2.75 ± 0.44 individuals per hectare). There were differences in the density of *E. daemelii* between reefs (Elizabeth: 3.61 individuals per hectare; Middleton: 1.9 individuals per hectare) and among habitats (Figure 4.22b). Interestingly, the density of *E. daemelii* increased from 3.13 to 3.61 individuals per hectare on Elizabeth Reef from 2018 to 2024, but decreased from 2.38 to 1.90 on Middleton Reef over the same period. These changes likely reflect natural local variation in the density of *E. daemelii* (e.g., local movements or variation in the detectability of *E. daemelii*), rather than systematic declines and we do not consider this variation is cause for concern.

4.3.3 Regional variation in fish assemblages

Comparisons with surveys conducted on five reefs in the southern Coral Sea Marine Park (CSMP) during February – March 2024 show that the average taxonomic richness of reef fish assemblages was lower on Elizabeth and Middleton Reefs (52.2 species per site) compared to the five reefs in the southern CSMP (70.8 species per site; Figure 4.24a). In contrast to differences in species richness, the density and biomass of reef fishes was 1.9- and 3.7-fold greater on Elizabeth and Middleton Reefs (density: 154.0 individuals per $100m^2$; biomass: 38.8 kg per $100m^2$) than the southern CSMP reefs (density: 83.8 individuals per $100m^2$; biomass: 10.5 kg per $100m^2$; Figure 4.24b, c). These differences were largely attributed to the higher density of omnivorous fishes, and higher biomass of piscivores and carnivores (primarily *C. galapagensis*), and grazing and browsing herbivorous fishes (primarily *Kyphosus* spp., and *Prionurus* spp.) on Elizabeth and Middleton Reefs compared to those of the southern CSMP (Figure 4.25).

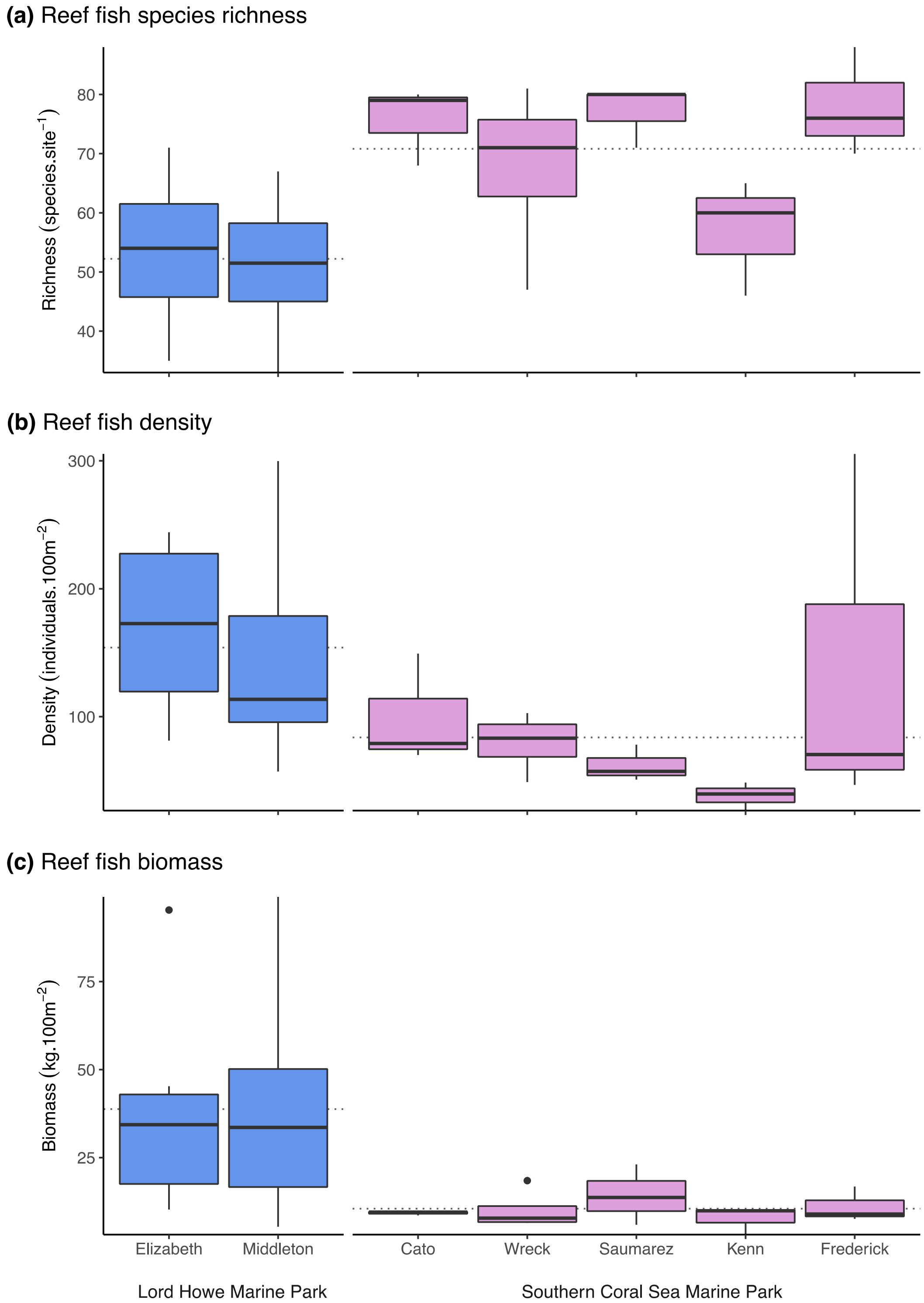

Figure 4.24 Variation in the **(a)** species richness, **(b)** density, and **(c)** biomass of reef fish assemblages among reefs and regions (Lord Howe Marine Park and southern Coral Sea Marine Park) in February-March 2024. Data are based on 50m point-intercept transects at three to eight sites per reef (i.e., pooled across transects and reef zones). Dashed lines indicate region averages.

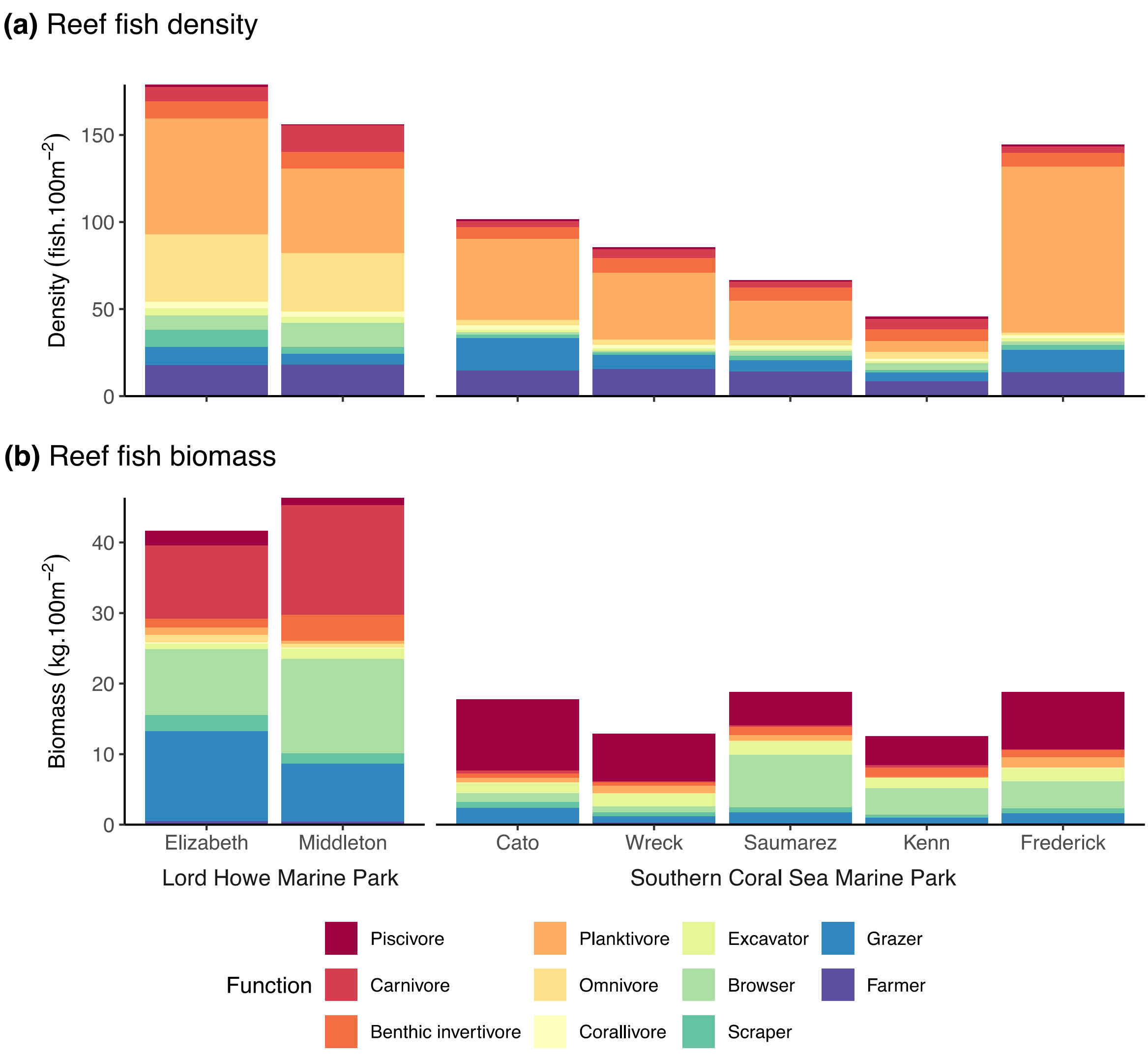

Figure 4.25 Variation in the functional composition of reef fish assemblages among reefs and regions (Lord Howe Marine Park and southern Coral Sea Marine Park) in February – March 2024. Data are based on 50m point-intercept transects at three to eight sites per reef (i.e., pooled across transects and reef zones). Colours show the relative contribution of the eleven functional groups.

4.4 Deep reef habitats

Deep water surveys (> 30m) using Baited Remote Underwater Video systems (BRUVs) or Remotely Operated Vehicles (ROV) were not possible during voyage time at Elizabeth and Middleton due to strong winds (> 25 knots) and large swells that made deploying and piloting the ROV from a tender unsafe. This restricted the area of operation for both BRUV and ROV surveys conducted from small tenders. Where possible, sampling was undertaken at depths between 15-25m but this was limited to sheltered lagoon and channel sites in close proximity to the *MV Iron Joy*.

4.4.1 Remotely Operated Vehicle (ROV) surveys

Sites in Elizabeth north-east lagoon channel and northern outer shelf surveyed by ROV were characterised by isolated patch reefs surrounded by low complexity unconsolidated (largely sand) substrata (Figure 4.26). These patch reefs consisted mostly of massive, sub-massive and encrusting hard coral morphologies. Soft corals and sponges were also prevalent in these areas. Notably, there was a relatively high cover of cyanobacterial mats at these sites, which covered dead coral skeletons as well as forming biofilms on sandy substrate surfaces (Figure 4.27).

Figure 4.26 Photographs from the Remotely Operated Vehicle (ROV) surveys of small patch reefs in the lagoon of Elizabeth reef, 25m depth.

Cyanobacteria are oxy-photosynthetic bacteria and are ubiquitous in coral reef ecosystems (Charpy et al. 2012). They are an important organic food source for planktonic and benthic heterotrophic organisms but are also associated with habitat degradation and declining reef health (e.g., Ford et al. 2018). Most of the cyanobacteria recorded by ROV surveys at Elizabeth reef were observed to be growing on already dead coral skeletons. Although some cyanobacteria can cause tissue necrosis in scleractinian corals (Burgo and Hoey 2024), it is unlikely that the cyanobacteria observed at Elizabeth Reef contributed to coral mortality. Rather, it is more likely that other agents contributed to colony mortality (e.g., predation, thermal stress) and elevated water temperatures have promoted extensive cyanobacterial growth at the time of our surveys. Irrespective of the cause of these cyanobacterial mats, if they persist they will likely inhibit coral recruitment and hinder the recovery of coral populations (Titlyanov et al. 2007; Burgo and Hoey 2024).

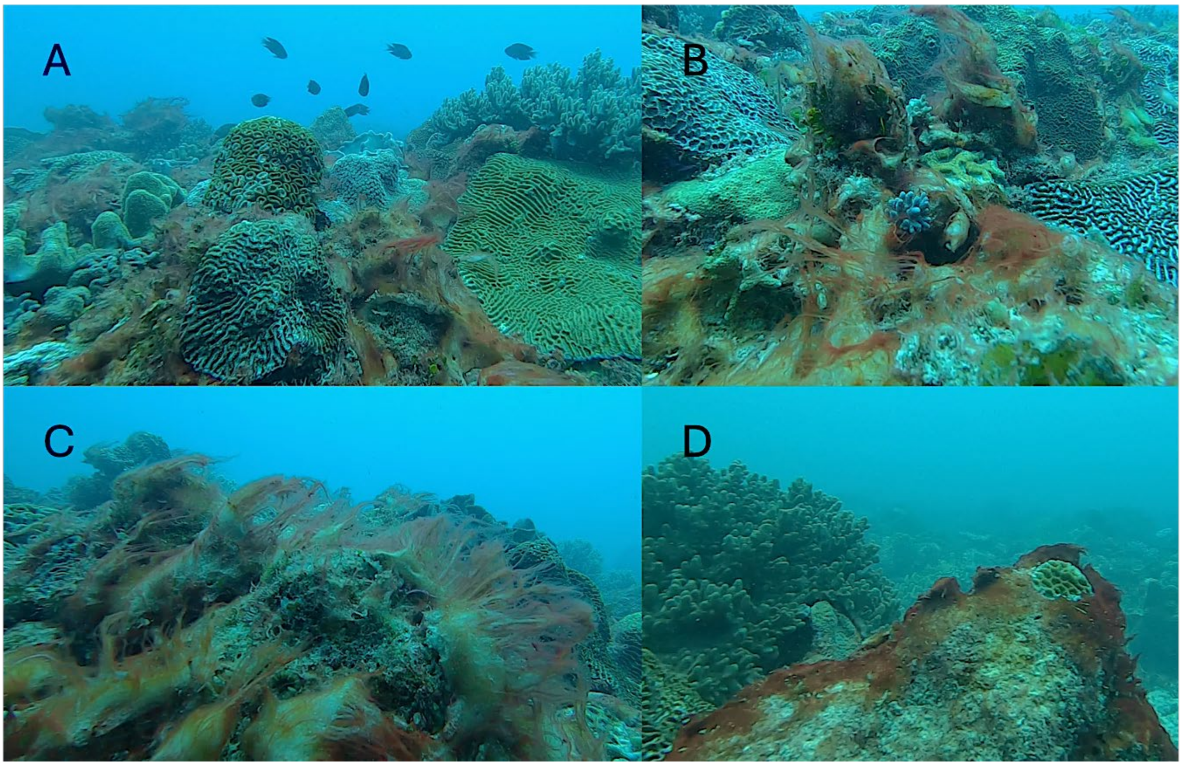

Figure 4.27 Photographs from the Remotely Operated Vehicle (ROV) surveys of the north-east lagoon channel and northern outer shelf on Elizabeth Reef showing cyanobacterial mats growing over hard substrata.

4.4.2 Baited Remote Underwater Video stations (BRUVs)

Ten species of fish from 3 families were recorded across all BRUV drops in the western lagoon at Middleton Reef (Figure 4.28). The white trevally, *Pseudocaranx dentex,* was the most abundant species recorded by BRUVs (relative abundance mean MaxN = 8), followed by the Galapagos shark *C. galapagensis* (Mean MaxN = 4) and Japanese sea bream *Gymnocranius euanus* (mean MaxN = 3). The round ribbon tail ray *Taeniura meyeni* was observed on 5 BRUV drops, often accompanied by yellowtail kingfish *Seriola lalandi.* A juvenile tiger shark *Galeocerdo cuvier* was observed on two of the BRUV drops yet was not recorded during the diver-based surveys of both reefs.

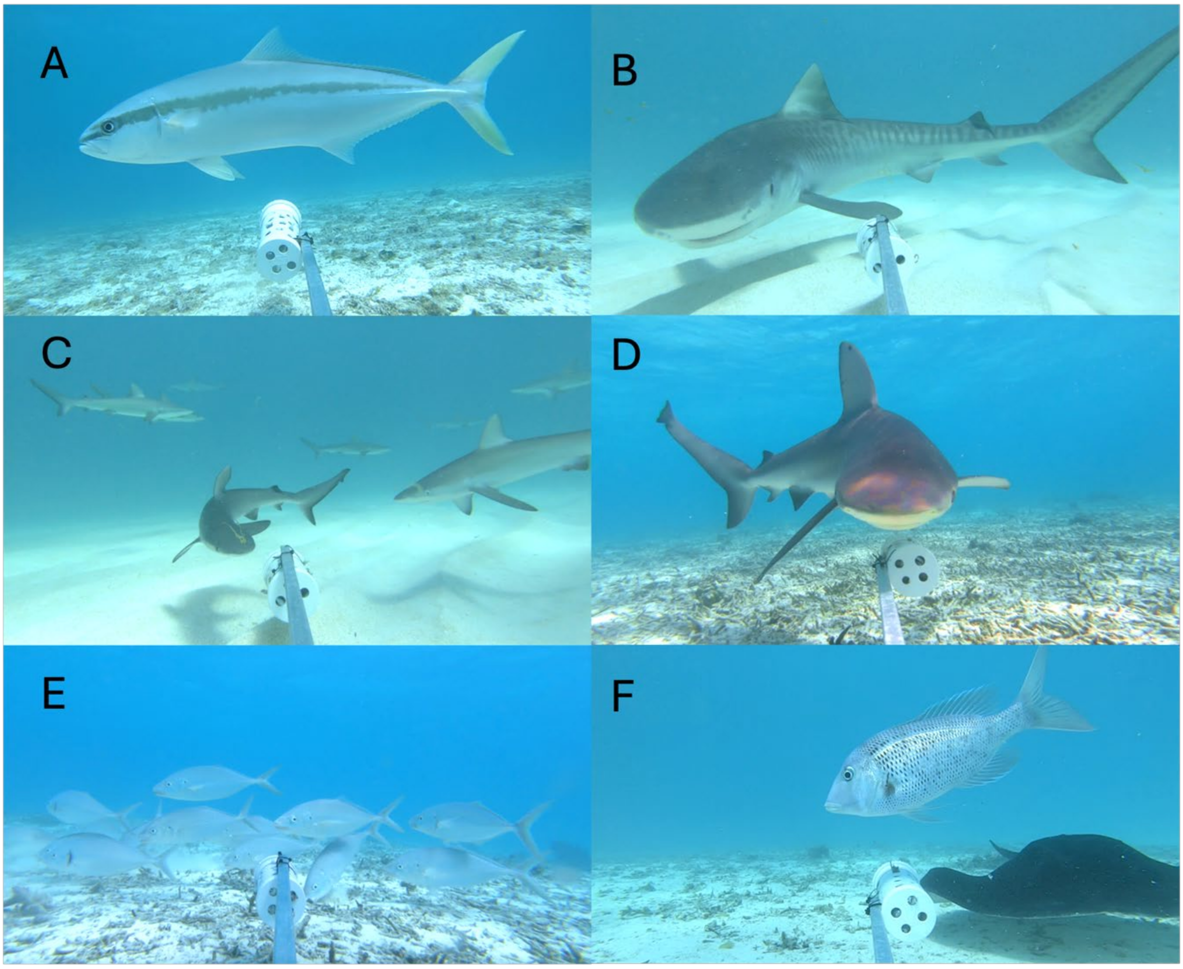

Figure 4.28 Photos of the predominant fish species captured by Baited Remote Underwater Video (BRUV) drops at Middleton Reef in February 2024. **(a)** yellowtail kingfish, *Seriola lalandi,* (**b)** juvenile tiger shark, *Galeocerdo cuvier,* **(c, d)** Galapagos shark, *Carcharhinus galapagensis*, **(e)** White trevally, *Pseudocaranx dentex*, **(f)** Japanese sea bream, *Gymnocranius euanus* and round ribbon tail ray, *Taeniura meyeni*.

4.5 Shipwrecks

There are numerous shipwrecks around the margins of both Elizabeth and Middleton Reefs. In 2011, the hull and structure of several of these wrecks were clearly visible above water and were often the first thing sighted when approaching the reefs. These wrecks have broken down considerably over the past 13 years, such that some are no longer visible above water, while the structures of others that were once above the water have decreased considerably in size. The *SS Runic,* a 171m long refrigerated cargo ship that ran aground on Middleton Reef on 19th February 1961, was the largest visible wreck on Middleton in 2011 with much of its hull and deck visible above the surrounding sea, and could be seen from

>1km from the reef (Figure 4.29). By 2024 the only visible structures were the engine block and a small section of the bow (Figure 4.29); the rest of the hull has seemingly rusted and been broken down by wave action.

Similarly, the *Fuku Maru No. 7*, a 239 tonne Japanese tuna fishing boat ran aground on Middleton Reef on 2nd November 1963. Despite being reported to be burnt out, the hull was used as a food cache and shelter for many years (https://www.wrecksite.eu/wreck.aspx?186500). In 2011 a large section of the midship and stern were visible above water, however only the rudder post and engine block were visible in February 2024 (Figure 4.30).

Other wrecks that were clearly recognisable above water in 2011, are no longer visible. The wreckage of the *Annasona*, a 73m steel Barquetine that wrecked on Middleton Reef on 18th January 1907, could be clearly seen in 2011 (Figure 4.31) but could not be relocated during the 2024 voyage. The *Monray Frontier,* a fibreglass longliner ran aground on Middleton Reef in 1998 some 200m from the *SS Runic*, and could be seen with the hull largely intact in 2011 (Figure 4.231). The *Monray Frontier* was observed again during the 2014 surveys, however by 2018 it was gone, presumably refloated and washed off the reef during a severe storm.

Figure 4.29 Photos of the wreck of the *SS Runic* showing the deterioration of the ship from 2011 to 2024. The wreck of the *SS Runic* sits on the reef crest of the north-western aspect of Middleton Reef. Images: Andrew Hoey

Figure 4.30 Photos of the wreck of the *Fuku Maru No. 7* showing the deterioration of the ship from 2011 to 2024. The wreck of the *Fuku Maru No. 7* sits on the shallow reef flat on the south-southeastern aspect of Middleton Reef. Images: Andrew Hoey

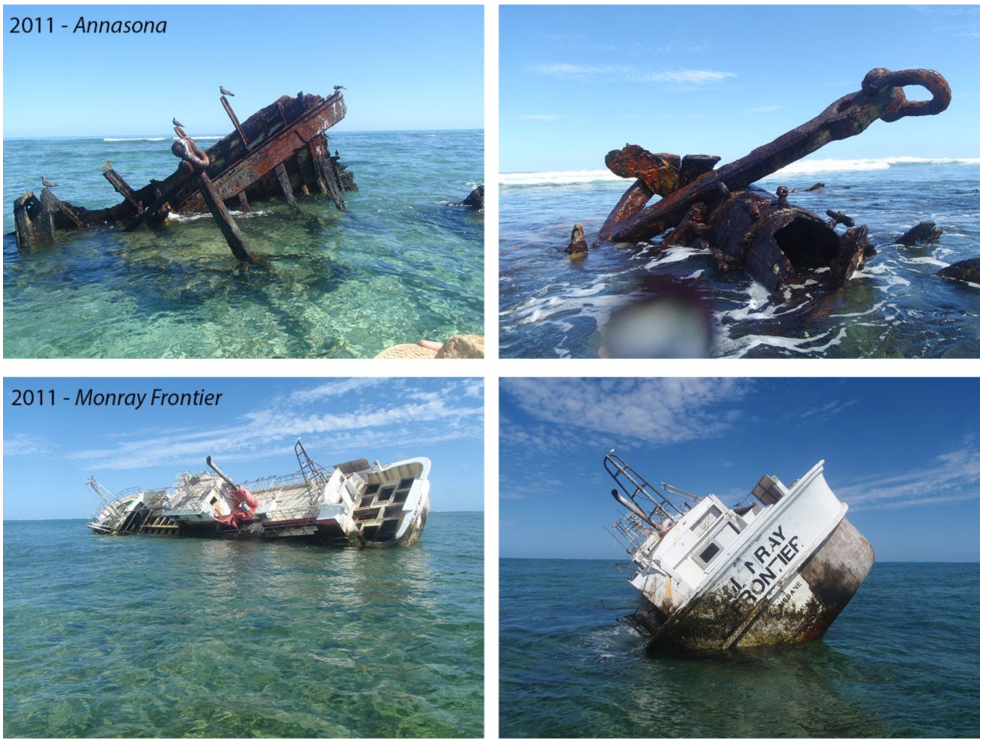

Figure 4.31 Photos of the wrecks of the *Annasona* (top) and *Monray Frontier* (bottom) on Middleton Reef in 2011. The wreck of the *Annasona* was on the shallow reef flat on the southwestern aspect of Middleton Reef, while the *Monray Frontier* was on the shallow lagoon, 200-300m east of the wreck of the *SS Runic*. Images: Andrew Hoey

5 Conclusions

Elizabeth and Middleton Reefs are the world's southernmost platform reefs and support a unique mix of tropical and subtropical species, including several endemic species (e.g., Hobbs et al. 2008; Pratchett et al. 2011; Hoey et al. 2014, 2018; Edgar et al. 2018). Isolated reefs such as Elizabeth and Middleton Reefs are often described as 'near pristine' due to the distance to human populations and hence the limited direct anthropogenic pressures (e.g., declining water quality, pollution, fishing) to which they are exposed. However, isolated reefs are extremely vulnerable to acute disturbances that cause widespread coral mortality (e.g., coral bleaching, severe storms, outbreaks of coral predators) as their recovery potential is constrained by their reliance on self-recruitment (i.e., larvae that are spawned from the reef, rather than other nearby and connected reefs; Gilmour et al. 2013). The recovery potential of isolated sub-tropical reefs is likely to be further constrained by lower growth rates of corals compared to low latitude reefs (Anderson et al. 2015), and is evidenced by the slow and ongoing recovery of coral populations on Elizabeth and Middleton Reefs following an outbreak of the corallivorous crown-of-thorns starfish in the early- to mid-1980's (Harriott 1998).

5.1 Benthic communities

Average coral cover recorded across Elizabeth and Middleton Reefs in February 2024 was 31.3%, the highest coral cover recorded since 2011 (Elizabeth: 35.9%; Middleton 27.4%), while macroalgal cover has remained low (9.1%) and stable over the past 6 years. Importantly, current coral cover at Elizabeth and Middleton Reefs represents an increase, albeit small (*ca.* 4%), from 30.1% in 2018, whereas coral cover has decreased markedly throughout the Coral Sea Marine Park (CSMP) over the same period due to the effects of multiple coral bleaching events (Hoey et al. 2020, 2024). Current coral cover on Elizabeth and Middleton Reefs is 2- to 3-fold greater than that of reefs in the southern and central CSMP (15.8% and 12.2%, respectively), some 780 – 1,800km to the north (Hoey et al. 2024). Despite the currently high coral cover on Elizabeth and Middleton Reefs, the limited increase in coral cover over the previous 6 years (*ca.* 4%) compared to the 4 years between 2014 and 2018 (a 34% increase) indicates there was likely some mortality of corals between 2018 and 2024. Given the time between successive surveys the cause of this mortality is difficult, if not impossible, to identify. While no crown-of-thorns starfish (*Acanthaster* cf. *solaris*) were observed during the 2024 surveys, some bleaching of corals was recorded on these reefs in 2018 and the accumulated heat stress shortly after the 2018 surveys was among the highest recorded on these reefs at that time (Hoey et al. 2018), however it is unknown if this led to any mortality of corals.

Climate change and associated disturbances are increasingly shaping the composition and state of coral reefs globally (e.g., Hughes et al. 2017a, 2018a, b, 2019; Pratchett et al. 2020). Thus, it is becoming critical to understand the patterns of disturbance, as well as the responses, recovery and resilience of individual reefs and reef systems. While Elizabeth and Middleton Reefs were largely unaffected by the spikes in global ocean temperatures that caused widespread and severe coral bleaching during 2015-2016 (Hughes et al. 2017a), low to moderate levels of bleaching were recorded on both reefs during the February 2024 voyage (13.6% of coral colonies were pale or bleached). However, the 2024 marine heatwave in the Tasman Sea and Coral Sea was still building at the time of the surveys and did not reach its peak until late March (Figure 4.9). Consistent with the building intensity of the marine heatwave the lowest incidence of bleaching was recorded on Elizabeth Reef (11% of colonies), with higher levels being recorded on Middleton Reef (17.6% of colonies) just a few days later. At the time of our last surveys on Middleton Reef (14th February 2024) large areas of the Lord Howe Marine Park and southern CSMP were exposed to > 8 DHW (Figure 4.9), levels of heat stress where severe bleaching and mortality may be expected (Hughes et al. 2018). Importantly, the marine heat wave continued to build through March with large areas in the southern Coral Sea and Tasman Sea exposed to >16 DHW and up to 20 DHW in some areas. This level of heat stress is unprecedented, and it is substantially greater than what reefs experienced during the 2020 bleaching event that led to *ca.* 40% decline in shallow water coral cover across the CSMP (Hoey et al. 2021), or the 2016 and 2017 bleaching events on the Great Barrier Reef (Hughes et al. 2017a, 2018a, b). Future monitoring (ideally in late 2024 or early 2025, and prior to any potential future heat stress) will be critical to assess the impacts of this marine heat wave on the coral communities of Elizabeth and Middleton Reefs.

Densities of juvenile corals at Elizabeth and Middleton Reefs (18.0 juveniles per $10m^2$) are low compared to low latitude reef systems (e.g., CSMP: 36 juveniles per $10m^2$, Hoey et al. 2024; GBR: 61-82 juveniles per $10m^2$, Trapon et al. 2013; New Caledonia: 20-116 juveniles per $10m^2$; Adjeroud et al. 2010), but greater than those from Lord Howe Island (8 juveniles per $10m^2$; Hoey et al. 2011). These low rates of replenishment appear to be characteristic of high latitude reefs (Harriott 1992; Bauman et al. 2014) and coupled with the lower coral growth rates on subtropical reefs (Harriott 1999; Anderson et al. 2015) are likely to limit the recovery potential of coral populations on these reefs following disturbance. Collectively these factors are likely to have contributed to the protracted recovery of coral communities following the crown-of-thorns starfish outbreak in the late 1980’s and early 1990’s (Harriott 1998).

5.2 Reef fish

Our surveys in February 2024 indicate that reef fish assemblages on Elizabeth and Middleton Reefs are healthy, and have displayed limited change over the past 6 years. The density and biomass of reef fishes on Elizabeth and Middleton Reefs (density: 154.0 individuals per $100m^2$; biomass: 3,880 kg per ha) are 2- to 4- fold greater than on reefs in the southern CSMP (density: 83.8 individuals per $100m^2$; biomass: 1,050 kg per ha), and high relative to coral reef environments globally (Cinner et al. 2016). Importantly, reef fish biomass at Elizabeth and Middleton Reefs is greater than estimates of unfished biomass for coral reefs globally (1,000-1,250 kg per hectare; MacNeil et al. 2015; McClanahan 2018), and together with the biomass of Galapagos sharks and large-bodied piscivores (e.g., Black cod), likely reflects the isolation and limited fishing pressure on these reefs.

The density of endemic and vulnerable/threatened fish species was generally stable or increased from 2018 to 2024. Importantly, there were small increases in the populations of McCulloch’s anemonefish, *Amphiprion mccullochi*, on Elizabeth and Middleton Reefs over the past 6 years. Although populations of *A. mccullochi* on Elizabeth and Middleton Reefs are restricted to the lagoon habitat and considerably smaller than those on reefs surrounding Lord Howe Island, the increases in density from 2018 to 2024 contrasts with the reported 50% decline in *A. mccullochi* population on Lord Howe Island (Hobbs 2022). The declines of *A.*

mccullochi on Lord Howe Island have been attributed to the reductions in the abundance of its host anemone (*Entacmaea quadricolor*) due to bleaching-induced mortality (Hobbs 2022). This reduction in abundance at Lord Howe Island, and the reported local extinction of this species at Norfolk Island (Hobbs 2022) has led to proposed determination of *A. mccullochi* as 'critically endangered'. While the small increase in the Elizabeth and Middleton populations of *A. mccullochi* is positive, the high habitat dependence of this species on a single anemone species (*E. quadricolor*) and the susceptibility of that anemone to thermally-induced bleaching, makes *A. mccullochi* vulnerable to ongoing and future marine heatwaves. Given the extreme seawater temperatures experienced in late March 2024 future monitoring of *A. mccullochi* populations on Elizabeth and Middleton Reefs (ideally in late 2024 or early 2025) is critical.

The density of the 'near threatened' Black cod, *Epinephelus daemelii*, recorded across Elizabeth and Middleton Reefs was consistent between the 2018 and 2024 surveys (2.7 – 2.8 individuals per hectare). Black cod (*E. daemelii*) has been extensively fished and experienced population declines throughout much of its geographic range. While the overall density of *E. daemelii* was unchanged, there was a small decline in density on Middleton Reef from 2018 to 2024 (2.4 and 1.9 individuals per hectare, respectively). Although this decline in density was relatively small (ca 20%), it is within the range of historic densities (Oxley et al. 2003; Choat et al. 2006), and may reflect natural variation due to local movements, variation in detectability, and/or the limited surveys conducted on the shallow reef crest of exposed reef front sites in 2024, continued monitoring is critical to assess its population status at Elizabeth and Middleton Reefs; one of the last strongholds for Black cod. Moreover, dedicated research is required to assess the natural dynamics of these local populations (e.g., recruitment rates, recruitment/nursery habitats, movement patterns), which will be critical for assessing the overall vulnerability of these populations. For example, juvenile *E. daemelii* have been shown to use rock pools and shallow intertidal habitats along the New South Wales coast (Harasti et al. 2014), however the juvenile habitat/s of this species on Elizabeth and Middleton Reefs are unknown, with the smallest individual recorded during our surveys being 60cm total length.

Fishes on coral reefs are being increasingly viewed in terms of their ecological roles, or functions, rather than their taxonomic affinities (e.g., Richardson et al. 2018). This shift in focus has been driven by the realisation that reductions in, or the loss of, key functional groups of fishes through anthropogenic activities (primarily fishing) has underpinned the degradation of some coral reef systems (e.g., Hughes 1994; Rasher et al. 2013). Reductions in predatory fishes have been suggested to release their prey populations from top-down control (e.g., Dulvy et al. 2004; Boaden and Kingsford 2015), and changes in the foraging behaviour due to predation risk (Rizzari et al. 2014; Rasher et al. 2017; Sherman et al. 2020). Perhaps the most pervasive effect of fishing on the functioning of coral reefs is the reduction in herbivorous fish populations that have underpinned shifts from coral- to macroalgal-dominated reefs in many regions (Hughes 1994; Rasher et al. 2013; Graham et al. 2015), although this top-down role of herbivorous fishes is increasingly debated (Russ et al. 2015; Clements et al 2017).

The herbivorous fish assemblages of Elizabeth and Middleton Reefs represent a unique mix of both tropical and temperate species, and have been relatively stable since 2011 (Pratchett et al. 2011, Hoey et al. 2014, 2018). The densities of herbivorous fishes on Elizabeth and Middleton Reefs were broadly comparable to lower latitude reefs of the central and northern Great Barrier Reef (Wismer et al. 2009) and substantially greater than those of Lord Howe Island, 260 km to the south (Hoey et al. 2011), and reefs of the southern CSMP >700km to the north (Hoey et al. 2024). There were, however, marked differences in the functional composition of herbivorous fish assemblages between these different reef systems. Herbivorous fish assemblages on Elizabeth and Middleton reefs were dominated by macroalgal browsing species, in particular *Kyphosus sectatrix* and *Prionurus maculatus* that together accounted for >70 % of the total herbivorous fish biomass. We currently know very little of the diet, feeding behaviour, growth or longevity of these two species, and should be the focus of dedicated research to better understand the importance of herbivory in structuring these unique reef systems.

5.3 Recommendations

Regular comprehensive monitoring of coral reef environments, using consistent and standardised protocols, is essential to understand the structure, function, ecological significance, and changing health and condition of Elizabeth and Middleton Reefs, especially in light of the increasing incidence of heat stress events. Monitoring of coral and fish communities (in particular McCulloch's anemonefish, *A. mccullochi*) on Elizabeth and Middleton Reefs in late 2024 or early 2025 (i.e., before any future heat stress) is critical to quantify the effects of the 2024 marine heat wave on these unique ecosystems. Without future monitoring, changes in the population status of endemic and threatened species would be largely unknown, severely limiting the capacity of managers to make informed decisions.

To effectively monitor the potential changes in coral, fish and invertebrate communities following major disturbances, and their potential recovery, we recommend regular monitoring of benthic, fish, and macro-invertebrate communities using the same methods and sites as previous (2011-24) surveys. This series of surveys represents one of the longest, if not the longest, running monitoring programs of any Australian Marine Park, and is invaluable in providing contemporary baselines and detecting change. 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 3-5 years, however more frequent and responsive monitoring is recommended to assess any effects of future disturbances, and in particular the predicted increases in the frequency of marine heatwaves (Hughes et al. 2018; Hoegh-Guldberg et al. 2018).

- In the absence of frequent and responsive monitoring, the utility of occasional visits to Elizabeth and Middleton Reefs should be maximised. This could provide managers with a more timely alert regarding any apparent concerns and/or threats to the ecosystem health of these two unique reefs. At present there is a requirement for recreational fishing permit holders to Elizabeth Reef to submit post-trip reports including observations of environmental conditions (e.g., bleaching) and there is a voluntary report for visitors to Middleton Reef. These requirement for these reports should be maintained and/or expanded.

- Maintain collaboration and regular communication between the managers of the Lord Howe Marine Park (Commonwealth waters) and the Lord Howe Island Marine Park (NSW waters) to ensure insights are gained from monitoring and visitors observations in more accessible reefs around Lord Howe Island.
- As well as monitoring the current status of benthic, fish and macro-invertebrate population, dedicated research to quantify the ecology and demographic processes of key taxa (e.g., recruitment, growth and mortality of corals, nursery habitats of Black cod, and diet, behaviour and demographics of browsing herbivores) will greatly improve our understanding of the vulnerability, recovery potential, and resilience of Elizabeth and Middleton Reefs.

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6 APPENDIX 1 – Sites surveyed

List of sites surveyed across Elizabeth and Middleton Reefs in February - March 2024.

Reef	Site	Date	Survey Method	Habitat	Depth	Latitude	Longitude
Elizabeth	Elizabeth 7	9/2/2024	UVC	Lagoon	3-9m	-29.93617	159.05162
Elizabeth	Elizabeth 6d	9/2/2024	UVC	Lagoon	6-8m	-29.93465	159.09326
Elizabeth	Elizabeth 6s	9/2/2024	UVC	Lagoon	3m	-29.93580	159.08910
Elizabeth	Elizabeth 4	10/2/2024	UVC	Back reef	3-9m	-29.92693	159.04010
Elizabeth	Elizabeth 5a	10/2/2024	UVC	Back reef	3-9m	-29.91825	159.06030
Elizabeth	Elizabeth 3	11/2/2024	UVC	Reef front	9-10m	-29.93283	159.02316
Elizabeth	Elizabeth 2	11/2/2024	UVC	Reef front	9-10m	-29.94333	159.01833
Middleton	Middleton 2	12/2/2024	UVC	Back reef	3-9m	-29.45295	159.08435
Middleton	Middleton 3a	12/2/2024	UVC	Lagoon	2-6m	-29.45993	159.06650
Middleton	Middleton 1	12/2/2024	UVC	Back reef	3-9m	-29.44350	159.09756
Middleton	Middleton 4	13/2/2024	UVC	Lagoon	3-9m	-29.44290	159.11502
Middleton	Middleton 10	13/2/2024	UVC	Back reef	7-9m	-29.42767	159.10692
Middleton	Middleton 5a	13/2/2024	UVC	Reef front	3-9m	-29.45257	159.04985
Middleton	Middleton 8	14/2/2024	UVC	Reef front	3-9m	-29.48477	159.07661
Middleton	Middleton 6	14/2/2024	UVC	Reef front	3-9m	-29.47728	159.12160
Middleton	MIDD-01	13/2/2024	BRUV	Lagoon	7m	-29.45760	159.06898
Middleton	MIDD-01	13/2/2024	BRUV	Lagoon	8m	-29.45927	159.06915
Middleton	MIDD-01	13/2/2024	BRUV	Lagoon	10m	-29.46083	159.06970
Middleton	MIDD-01	13/2/2024	BRUV	Lagoon	6m	-29.46253	159.07060
Middleton	MIDD-01	13/2/2024	BRUV	Lagoon	5m	-29.46238	159.06833
Middleton	MIDD-02	14/2/2024	BRUV	Lagoon	6m	-29.46392	159.06765
Middleton	MIDD-02	14/2/2024	BRUV	Lagoon	10m	-29.46233	159.06538
Middleton	MIDD-02	14/2/2024	BRUV	Lagoon	8m	-29.45998	159.06484
Middleton	MIDD-02	14/2/2024	BRUV	Lagoon	7m	-29.45724	159.06583
Middleton	MIDD-02	14/2/2024	BRUV	Lagoon	5m	-29.45444	159.06732
Middleton	MIDD-03	14/2/2024	BRUV	Lagoon	11m	-29.45498	159.06232
Middleton	MIDD-03	14/2/2024	BRUV	Lagoon	8m	-29.45760	159.06198
Middleton	MIDD-03	14/2/2024	BRUV	Lagoon	12m	-29.45947	159.06112
Middleton	MIDD-03	14/2/2024	BRUV	Lagoon	7m	-29.46208	159.06112
Middleton	MIDD-03	14/2/2024	BRUV	Lagoon	6m	-29.46411	159.06309
Elizabeth	ELIZ-01	10/2/2024	ROV	Lagoon	25m	-29.9325	159.09233
Elizabeth	ELIZ-02	10/2/2024	ROV	Lagoon	25m	-29.91853	159.05664
Elizabeth	ELIZ-03	10/2/2024	ROV	Lagoon	25m	-29.92210	159.05351
Elizabeth	ELIZ-04	10/2/2024	ROV	Lagoon	25m	-29.93280	159.09760

7 APPENDIX 2 – Fish species surveyed

List of fish species recorded across Elizabeth and Middleton Reefs during the 2024 surveys.

Species	
Acanthurus albipectoralis	*Heniochus chrysostomus*
Acanthurus dussumieri	*Kyphosus bigibbus*
Acanthurus nigrofuscus	*Kyphosus sectatrix*
Acanthurus nigroris	*Kyphosus vaigiensis*
Acanthurus triostegus	*Labracoglossa nitida*
Amphiprion mccullochi	*Labrichthys unilineatus*
Anampses caeruleopunctatus	*Labroides bicolor*
Anampses elegans	*Labroides dimidiatus*
Anampses femininus	*Labropsis australis*
Anampses geographicus	*Lutjanus bohar*
Anampses neoguinaicus	*Lutjanus kasmira*
Aprion virescens	*Lutjanus quinquelineatus*
Bodianus axillaris	*Macropharyngodon meleagris*
Bodianus perditio	*Monotaxis grandoculis*
Carcharhinus galapagensis	*Naso brevirostris*
Centropyge tibicen	*Naso lituratus*
Cephalopholis argus	*Naso sp.*
Cephalopholis miniata	*Naso tonganus*
Chaetodon auriga	*Naso unicornis*
Chaetodon citrinellus	*Neoglyphidodon polyacanthus*
Chaetodon flavirostris	*Oxycheilinus digramma*
Chaetodon guentheri	*Oxycheilinus unifasciatus*
Chaetodon lineolatus	*Paracaesio xanthura*
Chaetodon lunula	*Parma polylepis*
Chaetodon lunulatus	*Parupeneus ciliatus*
Chaetodon melannotus	*Parupeneus multifasciatus*
Chaetodon mertensii	*Parupeneus pleurostigma*
Chaetodon pelewensis	*Parupeneus spilurus*
Chaetodon plebeius	*Plectorhinchus picus*
Chaetodon tricinctus	*Plectroglyphidodon dickii*
Chaetodon trifascialis	*Plectroglyphidodon gascoynei*
Chaetodon unimaculatus	*Plectroglyphidodon johnstonianus*
Chaetodon vagabundus	*Pomacentrus coelestis*
Chaetodontoplus conspicillatus	*Prionurus maculatus*
Cheilinus chlorourus	*Pseudocaranx sp*
Cheilinus trilobatus	*Pseudocheilinus hexataenia*
Cheilinus undulatus	*Pseudojuloides cerasinus*
Cheilodactylus ephippium	*Pseudolabrus luculentus*

Cheilodactylus francisi	*Scarus altipinnis*
Chlorurus frontalis	*Scarus chameleon*
Chlorurus microrhinos	*Scarus dimidiatus*
Chlorurus spilurus	*Scarus flavipectoralis*
Chromis atripectoralis	*Scarus frenatus*
Chromis chrysura	*Scarus ghobban*
Chromis flavomaculata	*Scarus globiceps*
Chromis hypsilepis	*Scarus niger*
Chromis vanderbilti	*Scarus oviceps*
Chrysiptera notialis	*Scarus psittacus*
Cirrhilabrus laboutei	*Scarus rivulatus*
Cirrhilabrus punctatus	*Scarus schlegeli*
Coris aygula	*Scarus sp*
Coris bulbifrons	*Scolopsis bilineatus*
Coris picta	*Seriola dumerili*
Ctenochaetus striatus	*Seriola lalandi*
Dascyllus aruanus	*Sphyraena barracuda*
Dascyllus reticulatus	*Stegastes fasciolatus*
Dascyllus trimaculatus	*Stethojulis bandanensis*
Epinephelus cyanopodus	*Stethojulis strigiventer*
Epinephelus daemelii	*Sufflamen chrysopterus*
Epinephelus fasciatus	*Sufflamen fraenatum*
Epinephelus merra	*Thalassoma amblycephalum*
Epinephelus rivulatus	*Thalassoma hardwicke*
Forcipiger flavissimus	*Thalassoma lunare*
Girella cyanea	*Thalassoma lutescens*
Gnathodentex aureolineatus	*Thalassoma nigrofasciatum*
Gomphosus varius	*Thalassoma purpureum*
Gymnocranius euanus	*Thalassoma quinquevittatum*
Halichoeres marginatus	*Variola louti*
Halichoeres trimaculatus	*Zanclus cornutus*
Hemigymnus fasciatus	*Zebrasoma scopas*
Hemigymnus melapterus	*Zebrasoma veliferum*