



REEF LIFE
SURVEY

Lord Howe Island Marine Park Shallow Reef Ecological Assessment 2022

Rick D. Stuart-Smith, Antonia T. Cooper, Elizabeth S. Oh, Ella
Clausius, Joanna Schmid, Graham J. Edgar

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All images

Antonia Cooper



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List of acronyms

ACRONYM	EXPANDED
AMP	Australian Marine Park/ Commonwealth Marine Reserve
RLSF	The Reef Life Survey Foundation
MPA	Marine Protected Area
SZ	Sanctuary Zone
HPZ	Habitat Protection Zone
IUCN	International Union for Conservation of Nature
RLS	Reef Life Survey
LHI	Lord Howe Island
LHIMP	Lord Howe Island Marine Park
LRFI	Large Reef Fish Index
RFTI	Reef Fish Thermal Index



Executive summary

This report describes the results of 17 years of monitoring shallow reefs (0-30m depth) in the Lord Howe Island Marine Park (LHIMP) based on data collected on 988 individual surveys over 10 monitoring campaigns, representing 45 monitoring sites across all zones of the LHIMP. These include data from reef fish surveys which include species, size and biomass information; finer-scale targeted surveys of large mobile invertebrates and cryptic fishes closely associated with the substrate; and analysis of percentage cover of coral, macroalgae and other sessile organisms from photoquadrats of the substrate. Surveys undertaken by RLS represent a highly cost-effective means to collect data on shallow reef biodiversity that is both rich in detail and facilitates ecosystem-scale analysis, while also maintaining a high standard of data quality consistent with a globally-standardised methodology.

This report builds on previous analysis to investigate spatial patterns and temporal trends in shallow reef biodiversity and ecology in the LHIMP. The inclusion of recent data not previously analysed (from surveys in 2020 and 2022) provides a longer times series and more robust context to assess patterns and trends, underscoring the high value of this continuous long-term monitoring data for informing management of the LHIMP. In particular, patterns and trends relating to several ongoing threats are examined including climate change, fishing pressure, and sea urchin grazing impacts. Other threats are discussed along with opportunities for management and future research.

Species across all groups examined (fishes, mobile invertebrates and benthic) cluster into similar assemblages on shallow reefs in the LHIMP, hereafter termed Ecological Communities. This includes the Lagoon, Algal Holes, Lord Howe Island (LHI) Offshore, and Balls Pyramid Ecological Communities. The Lagoon Ecological Community is characterised by a diverse benthic assemblage including branching coral and associated habitats (sand and rubble), solitary anemones, and seagrass. It has a predominantly tropical fish assemblage and supports among the highest fish and cryptic fish diversity in the LHIMP, particularly in the Lord Howe Island Lagoon Sanctuary Zone. It also supports the greatest biomass of benthic invertivore and corallivore fish, and provides critical habitat or nursery area for endemic, threatened and protected species including McCulloch's Anemonefish (*Amphiprion mccullochi*), Doubleheader (*Coris bulbifrons*), Bluefish (*Girella cyanea*) and Black Rockcod (*Epinephelus daemeli*).

The LHI Offshore Ecological Community includes offshore reefs surrounding LHI and is characterised by high benthic cover of crustose coralline algae, sponges, and encrusting and sub-massive coral species. It supports the highest species richness of fish and invertebrates and greatest biomass of fish, particularly on the diverse rheophilic (current-loving) reefs in the Neds Beach & Admiralty Islands Sanctuary Zone. The Balls Pyramid Ecological Community has similar benthic forms and supports similarly high diversity and biomass of fish, particularly in the Observatory Rock Sanctuary Zone and isolated rocks in the Balls Pyramid Sanctuary Zone, but with unique assemblages of species. The deeper offshore reefs of these two Ecological Communities are also key habitat for regionally endemic species such as the Halfbanded Angelfish (*Geniacanthus semicinctus*) and Wideband Anemonefish (*Amphiprion latezonatus*).

The Algal Holes Ecological Community is only known from a small area on the outer reef of the southern lagoon and has uniquely high benthic cover of macroalgae including red foliose algae. It also supports distinct assemblages of fish and invertebrates, particularly herbivorous fish such as the protected Bluefish (*Girella cyanea*) and grazing invertebrates such as the endemic Onion Turban sea snail (*Turbo cepoides*). Within most Ecological Communities, hotspots for species richness, fish biomass (including of large reef fish

and key species) generally occur within Sanctuary Zones (SZs), highlighting their importance for ongoing conservation of biodiversity and ecosystem function. The exception to this is the Algal Holes Ecological Community which is not represented within a SZ. Each Ecological Community is faced with different threats and management opportunities, warranting specific consideration during management planning.

Fish assemblages changed very little from 2006 to 2022 and no overall changes in response to coral bleaching events or topicalization effects were detected. However, declines in some species such as corallivores and endemic McCulloch's Anemonefish (*Amphiprion McCullochi*) in the lagoon are likely indicative of climate change impacts. There has been little evidence of increased fish biomass in response to reduced fishing pressure in SZs, with no net increase in total biomass from 2006 to 2022 or evidence of greater biomass in SZs relative to Habitat Protection Zone (HPZ). This may indicate low fishing pressure outside SZs, continued illegal fishing pressure within SZs, high mobility of target species across zone types, longer time frames required for ecological effects to become apparent, or these effects being obscured by other sources of variability. Biomass of target fished species similarly showed little net change over 17 years including Bluefish (*Girella cyanea*) and Doubleheader (*Coris bulbifrons*).

Mobile invertebrates also showed little overall change in assemblage structure from 2006 to 2022, although some important trends were observed in sea urchins. Densities of Long-spined Sea Urchin (*Centrostephanus rodgersii*) continued to increase throughout the monitoring period, and several population booms of Lamington Sea Urchin (*Tripneustes australiae*) have also occurred. These include around the Admiralty Islands in 2008 and around Balls Pyramid in 2022. A more widespread population boom occurred on offshore reefs surrounding LHI in 2024 during the most recent surveys and this data will be included in future reports. Increases in sea urchin density can impact benthic assemblages, particularly through rapid declines in macroalgae cover from grazing as previously documented around the Admiralty Islands. The potential loss of unique and endemic macroalgal on LHIMP reefs is therefore an ongoing concern.

Key changes in benthic assemblages included loss of coral cover following mass coral bleaching events in the lagoon in 2010 and 2019. The former event had greatest impact at North Bay and the latter at Horseshoe Reef, resulting in a decline in coral cover of around 10% across lagoon sites overall. While only representing a small proportion of benthic cover, the long-term persistence of this decline indicates that repeated coral bleaching events present a high risk of cumulative impact, which is likely as coral bleaching events are predicted increase in frequency and severity with climate change. Impacts from a 2024 bleaching event were still unfolding during the most recent surveys and have not been analysed for this report, but anecdotal observations suggest they were more widespread than previous events including the first documented occurrence of widespread bleaching on reefs outside the lagoon.





1 Introduction

BACKGROUND

Shallow reefs surrounding Lord Howe Island (LHI) and Balls Pyramid support globally significant natural values, recognised by the World Heritage status and implementation of the Lord Howe Island Marine Park (LHIMP) in the NSW state waters which encompass them. A marine park in the surrounding Commonwealth waters also includes isolated shallow reefs north of LHI (Elizabeth and Middleton Reefs) which are considered in separate reports (Edgar et al. 2019). Key natural values of shallow reefs in the LHIMP include the unique combination of tropical and temperate marine species; the occurrence of many species at their distributional limits including the world's southernmost coral reef; high diversity of marine macro-algae, fishes and invertebrates; and high levels of endemism.

Due to the influence of both tropical and temperate water masses along the Tasman front, these near pristine habitats include both coral reefs characteristic of tropical habitats to the north, and macroalgal and soft coral covered rocky reefs more typical of sub-tropical habitats on mainland Australia. These distinct reef communities occur in closer proximity than at other locations, making the LHIMP globally unique. Overall, tropical species contribute most to diversity on these reefs while temperate species contribute most to biomass and abundance.

These shallow reefs are also home to many endemic species, with few islands worldwide exceeding the levels of endemism they support. This includes fishes restricted to the LHIMP or surrounding region (i.e. Elizabeth and Middleton Reef, Norfolk Island, New Caledonia and the Kermadec Islands) such as McCulloch's Anemonefish (*Amphiprion mccullochi*), Three-striped Butterflyfish (*Chaetodon tricinctus*), Lord Howe Moray (*Gymnothorax annasona*), Halfbanded Angelfish (*Geniakanthus semicinctus*), Blacktip Morwong (*Goniistius francisi*), Norfolk Cardinalfish (*Ostorhinchus norfolcensis*) and secretive Island Longfin (*Plesiops insularis*). There are also 'insular' species predominantly found in the LHIMP or regional islands, but with vagrants also occasionally recorded elsewhere along the Australian mainland and/or northern New Zealand (such as Doubleheader *Coris bulbifrons*, Wideband Anemonefish *Amphiprion latezonatus* and Splended Hawkfish *Cirrhitis splendens*). Other iconic fish species include Bluefish (*Girella cyanaea*), Galapagos Shark (*Carcharhinus galapagensis*) and Lord Howe Island Butterflyfish (*Amphichaetodon howensis*) which have broader distributions but are particularly abundant in the LHIMP. There are also many endemic marine invertebrates on these shallow reefs including the Onion Turban sea snail (*Turbo cepoides*).

Diverse and endemic species of coral and macroalgae grow on these reefs too, with the LHIMP being one of the richest subtropical localities worldwide for benthic macroalgae with more than 305 species present and a high degree of endemism. Further descriptions of the natural values on shallow reefs in the LHIMP can be found in Edgar et al. (2008, 2011) and Harasti et al. (2022). Many of the widespread species which occur in the LHIMP have also been found to rely on self-recruitment to sustain local populations, including some of the most common and ecologically important species of coral, fish and invertebrates (Miller and Ayre 2004, Noreen et al. 2009, Harasti et al. 2022). Together with the high rates of endemism, this indicates that shallow reef species and ecosystems in the LHIMP are highly vulnerable to impacts including local extinction from local or global threats as described below.

Being among the most accessible and appealing habitats for a wide range of activities, shallow reefs also support many of the social and economic values of the LHIMP as documented by a recent community survey report (Heller 2024). The report highlighted the importance of shallow reef habitats and biodiversity to residents and the broader community, directly supporting five of the top six environmental values (corals,

habitats and ecosystems, marine biodiversity, threatened and protected species, and fish), the top three economic values (uniqueness, world heritage status and tourism), top two social values (something to leave for future generations and connection to nature) and top two cultural values (iconic/symbolic animals and sustainability/eco-centrism). The local community also expressed a desire to be more engaged with research and monitoring outcomes, and so this report also aims to assist with education and awareness of the ecology and management of shallow reef biodiversity within the LHIMP.

THE LORD HOWE ISLAND MARINE PARK

The primary purpose of the LHIMP, as stated in the *Marine Estate Management Act 2014* is to conserve biological diversity and maintain ecosystem integrity and ecosystem function of the bioregion, while providing for a range of ecologically sustainable uses. The LHIMP was declared in 1999 and a zoning plan has been in place since 2004. The park consists of seven Sanctuary Zones (SZs) and three small Special Purpose Zones to provide for managed fish feeding at Neds Beach, North Bay and Erscotts Hole. All remaining waters occur in Habitat Protection Zone (HPZ). SZs are no-take areas and, in accordance with the *Marine Estate Management (Management Rules) Regulation 1999*, aim to provide the highest level of protection for biological diversity, habitat, ecological processes, and natural features. SZs limit impacts from a wide range of activities including fishing and collecting, anchoring, and development. Some methods of fishing are permitted in HPZ and charter and recreational fishing occurs in these areas, including operation of a small artisanal fishery to supply fish for sale to local businesses. Fish cannot be commercially exported from the Island, and fishing pressure has been relatively low both prior to and since implementation of the marine park (Edgar et al., 2011). Further description of fishing activities in the LHIMP can be found in Figueira and Harianto (2022).

Monitoring of the shallow reef biodiversity in the LHIMP was established in 2006 and 2008 by a consulting company Aquenal Pty Ltd, in such a way to: (a) assess the suitability of the zoning plan, (b) assess the performance of SZs with respect to protection and recovery of ecological processes and conservation values, and (c) provide general condition reporting for shallow reef biodiversity around LHI (Valentine et al. 2008). Monitoring was continued from 2009 through a collaboration between the Reef Life Survey (RLS) program and LHIMP staff with support from the local dive operators. This arrangement has enabled the cost-effective monitoring of 45 sites (following consolidation of nearby sites in 2018; Fig. 1). These sites span a large number of reefs throughout the LHIMP and are monitored by RLS and LHIMP teams every second year. The time series for most sites now includes 10 biennial survey events (with the 11th recently completed in 2024, not included in data analyses here).

RLS is also used by other Marine Protected Area (MPA) management teams, state management agencies and the Australian Government, and contributes to national State of the Environment reporting. The RLS global dataset has provided numerous scientific insights and valuable direction for management, particularly for marine protected areas (see Edgar et al. 2014). Thus, RLS monitoring of the LHIMP both informs local management by identifying ecological patterns and trends in context of local threats and management opportunities, while also contributing to broader management, research and public engagement outcomes.

KEY THREATS TO SHALLOW REEF BIODIVERSITY

There are many threats to the shallow reef biodiversity and values in the LHIMP including those associated with climate change, nutrient input, pollution, aquatic biosecurity, vessel activities, coastal development, fishing, other commercial and recreational activities, and biological threats (such as sea urchin grazing). Brief overviews are given below but further descriptions of these threats and current LHIMP management

arrangements can be found in recent reports including Harasti et al. (2022) and Heller (2024). Several of these were recently found to be of high concern to the community (Heller 2024).

Many of these threats are likely to be interactive and act as compounding stressors on shallow reefs. Current LHIMP management arrangements address a wide range of localised threats to support reef resilience, and also support targeted research to address knowledge gaps about these threats. Complimentary research and monitoring programs also occur on the deeper reefs and other habitats in the LHIMP. Despite these efforts there are several ongoing knowledge gaps highlighted in this report, including the interaction between multiple stressors on shallow reefs.

Climate change

There are a range of both realised and potential impacts from climate change on shallow reefs in the LHIMP, the greatest being coral bleaching and mortality with flow-on effects to reef assemblages. Mass bleaching events have been documented in the LHIMP in 1998, 2010, 2019, and 2024. As noted by Edgar et al. (2008) these have resulted from the coincidence of several factors including: elevated regional sea surface temperatures due to El Nino climate conditions; locally elevated sea surface temperatures due to warm-core eddies from the EAC; prolonged periods of light winds and low swell reducing flushing of heated water in the lagoon; and spring low tides or negative sea surface anomalies which increase the effects of solar heating in the lagoon and which can also result in aerial exposure of coral.

Impacts of the 2010 bleaching event were recorded during RLS monitoring and have been previously described (Stuart-Smith et al. 2019) while impacts of the 2019 event detected by RLS monitoring are included here for the first time. The 2024 bleaching event was occurring during the most recent RLS surveys and these impacts will be included in following reports. Documented impacts in the LHIMP include loss of coral cover and subsequent changes in benthic assemblages, as well as declines in key susceptible species. The most severe impacts from these events have to date been restricted to reefs in the Lagoon Ecological Community, although they are predicted to increase in frequency and severity with climate change and may increasingly impact offshore reefs.

Bleaching events and marine heatwaves can also cause a rapid and substantial reshuffling of fish assemblage structure (Day et al. 2018) via direct effects on fishes through metabolism, ecological interactions and population dynamics, or indirectly through changes in habitat (Pratchett et al. 2008, Pratchett et al. 2011, Stuart-Smith et al. 2018). No major overall changes in fish or mobile invertebrate assemblages have yet been documented in the LHIMP, but these may occur in the future.

Increasing marine heat stress associated with climate change is also likely to impact macroalgae on shallow reefs in the LHIMP. Given many of these species occur at their northern distribution limits, they may decline in abundance and diversity in response to increasing sea temperatures. Local extinctions of temperate macroalgae species have been previously documented in the LHIMP (Edgar et al. 2008, 2011) and mass diebacks associated with increasing water temperatures documented elsewhere (Wernberg et al. 2016). Endemic macroalgae species and the Algal Holes Ecological Community and Lagoon Ecological Community may therefore be at particular risk from marine heat stress.

These impacts cannot be prevented by local management alone, however LHIMP management have developed and implemented a marine heatwave response plan including protocols for forecasting, assessing and communicating impacts. Targeted monitoring during the most recent 2024 bleaching event included repeated drone and in-situ visual surveys throughout the lagoon, maintenance of an array of water temperature loggers, and targeted studies of vulnerable species such as McCulloch's Anemonefish including strategies for recovery. Management arrangements which mitigate other stressors to shallow reefs also help improve reef resilience in the face of climate change impacts.

Nutrient input

Nutrient input into coastal waters surrounding LHI is another key threat noted since earliest ecological surveys (Edgar et al. 2008). This can be from a range of anthropogenic sources including groundwater flow and upwellings which transport nutrients from septic and other human inputs, and runoff from urban and agricultural areas which carry nutrients including from cattle manure (Davis 2022a, 2022b). Ecological indicators of these impacts have been documented in northern lagoon since earliest monitoring such as cyanobacteria and epiphytic growth on coral and seagrass (Harriot 1995, Edgar et al., 2008, Valentine et al. 2008). It is likely that nutrient inputs drive a number of other ecological patterns on shallow reefs, as has been documented on Norfolk Island where nutrient inputs were linked to shifts from coral to macroalgae dominated reefs and coral disease outbreaks (Ainsworth et al. 2021).

This threat is managed through local wastewater management arrangements, and LHIMP management is facilitating research to discriminate between different nutrient inputs and their ecological effects in the LHIMP. The greatest impacts from nutrient input are likely to occur in the Lagoon Ecological Community due to the strong influence of groundwater including upwellings, runoff from urban and agricultural areas including at creeks and ICOLLs, and lower rates of water flushing in the lagoon.

Other pollution

Other pollution events also pose a high risk of impact to shallow reefs as also noted since earliest ecological monitoring (Edgar et al., 2008). These could include vessel or land-based spills of fuel, oil or chemicals. There is also a risk of cumulative impact from minor leaks of fuel, chemicals or oils associated with regular vessel use at the boat ramp and jetty. The threat of pollution impacts is greatest in the Lagoon Ecological Community due to the concentration of vessel and land-based activities and infrastructure in or adjacent to the lagoon which pose risk of pollution events, as well as lower rates of water flushing in the lagoon.

Pollution is managed in accordance with the Protection of the *Environment Operations Act 1997*, MARPOL regulations for vessel discharge, along with other LHIMP management arrangements. As noted by Edgar et al. (2008) the long-term monitoring presented in this report provides important baseline information against which to assess the impacts of any future pollution events. Targeted monitoring of impact and control sites has also been undertaken by LHIMP management, including in response to a large diesel fuel spill in the northern lagoon in 2022 (Woods 2023).

Aquatic biosecurity

Threats to aquatic biosecurity (i.e. the introduction of non-native marine pests or diseases) pose among the greatest risk to shallow reefs in the LHIMP as noted since earliest ecological surveys (Edgar et al., 2008). While there are several potential vectors for such incursions, biofouling and ballast water from visiting vessels pose the greatest risk. These are currently managed in the LHIMP through ballast water restrictions, biofouling requirements, and hull inspections of visiting vessels with associated response guidelines. A long-term marine pest surveillance strategy has also been implemented in the LHIMP which includes regular monitoring and education. As for the other risks described above, shallow reefs in the Lagoon Ecological Community are at highest risk of impact from marine pest or disease incursions.

Vessel activities

There are a range of other direct impacts on shallow reefs from commercial and recreational vessel use, as also noted since earliest ecological surveys of these reefs in the LHIMP (Edgar et al. 2008). These include anchoring, vessel scour and other mechanical damage, sedimentation and turbidity impacts. For example, sedimentation from the commencement of regular commercial shipping in the northern lagoon in 1984 may have contributed to the mortality of branching pocilliporid corals at an adjacent inshore reef (Sylphs Hole) and subsequent ecological shift to more stress tolerant massive *Porites spp.* documented in this period

(Harriott et al. 1995, Denis et al. 2017). Sedimentation impacts were also noted in this area during initial ecological monitoring (Edgar et al. 2008).

Current management arrangements in the LHIMP to address these impacts include a range of anchoring restrictions, permit requirements and conditions for all commercial vessel activities in the LHIMP, provision of courtesy moorings in the lagoon to minimise anchoring, prohibition of personal watercraft, among other arrangements. As for other threats listed above these impacts pose greatest risk to shallow reefs in the Lagoon Ecological Community due to the concentration of vessel activity and proximity to shallow reefs in the lagoon.

Coastal development

Coastal development also presents a range of threats to shallow reefs in the LHIMP, including habitat loss and disturbance, sedimentation and turbidity impacts, hydrological changes, pollution including runoff, and increase in impacts from human activities associated with infrastructure. These impacts are managed through permit and approval processes for development and works in the LHIMP. For similar reasons to those described for other threats above, these present highest risk to shallow reefs in the Lagoon Ecological Community.

Fishing

Fishing and collection activities present a direct threat to shallow reef ecosystems through the removal of target species with potential flow-on effects to fish, invertebrate and benthic assemblages. A small artisanal fishery operates in the LHIMP, as further described by Figueira and Harianto (2022). Fishing pressure is low compared with other coastal regions of Australia (Edgar et al., 2008), however, concerns about impacts from fishing pressure including illegal fishing in SZs have been raised (Stuart-Smith et al. 2015, 2019).

Previous reports have documented trends in fish biomass (including total fish biomass, biomass of large reef fish, and target species) which may have been driven by fishing impacts (Edgar et al. 2011, Stuart-Smith et al. 2015, 2019). Particular concerns have been raised about fishing impacts on Doubleheader and Bluefish which are likely to be especially vulnerable to overexploitation and play key ecological roles in shallow reef ecosystems (Stuart-Smith et al. 2015, 2019). Doubleheader are regionally endemic so susceptible to local exploitation, and are a benthic invertivore so may play an ecological role in controlling sea urchin densities and associated grazing impacts (Edgar et al. 2008). Bluefish are a long-lived species protected elsewhere in NSW and are also susceptible to local fishing pressure. As large schooling herbivores, they likely play an important ecological role in controlling macroalgae cover on coral reefs and thereby supporting reef resilience and recovery from coral bleaching and other impacts (Donovan et al. 2022). Further description of fishing impacts on target species can also be found in Harasti et al. (2022).

Fishing impacts are currently managed in the LHIMP through bag and size limits (including for vulnerable species such as Doubleheader and Bluefish), prohibition of fishing methods which directly harm habitat in all areas of the LHIMP (such as trawling, dredging, long-lining and gill net fishing), SZs (representing approximately 27% of the LHIMP) in which all fishing is prohibited, prohibition of commercial export of fish from LHI, and permit and catch reporting requirements for charter fishing operators with ongoing catch data management and analysis. Prior to the establishment of the LHIMP some informal SZs and other fishing prohibitions were also maintained by the local community for over 40 years (as noted in Edgar et al., 2008).

Other activities

A wide range of other commercial, recreational and research activities are undertaken on shallow reefs in the LHIMP, which carry some risk of impacts. These activities include fish feeding, diving and snorkelling, organised events, photography and filming, kayaking, surfing, etc. Some of these are managed in the LHIMP through activity restrictions and/or permit requirements and conditions. They are also monitored using data

collected from submitted reports and surveys during regular land and vessel-based patrols with associated analysis of patterns and trends.

Sea urchin grazing

The threat of increasing sea urchin densities and associated grazing impacts have been noted since early ecological monitoring (Valentine et al., 2008). In particular, these impacts have been raised in relation to population booms of Lamington Sea Urchin (*Tripneustes australiae*) as well as increasing densities of Long-spined Sea Urchin (*Centrostephanus rodgersii*).

A population boom of Lamington Sea Urchin occurred in 2008 which resulted in large declines in macroalgae cover at affected sites due to grazing impacts (Valentine et al. 2010). This species is known to have boom and bust population dynamics characterised by sudden and dramatic increases in abundance followed by mass mortality. Population booms have significant benthic impacts although the ecological drivers and long-term effects are still poorly understood. Long-spined Sea Urchin have also been observed to occur in high densities and form small urchin barrens in the LHIMP since earliest ecological monitoring, with increases in the extent of these barrens noted as a potential threat (Edgar et al. 2008). Subsequent reports have confirmed sustained increases in the density of this species (Edgar et al. 2011). Grazing impacts from these species present a particular threat to endemic macroalgae species and the unique Algal Holes Ecological Community in the LHIMP, as well as impacting the commercial and recreational value of sites for SCUBA diving and snorkelling (as noted by Valentine and Edgar 2010). Localised increases in Purple Sea Urchin (*Heliocidaris tuberculata*) have also been noted as posing a threat of grazing impacts in some areas (Stuart-smith et al. 2015). These impacts are assessed and discussed further throughout this report.

As noted by Edgar et al. (2011) management strategies may include reducing fishing pressure on sea urchin predators (such as Doubleheader), or direct control through removal of sea urchins. The efficacy of these actions remain to be identified, but are likely subject to the broader ecological drivers of these population trends. Given these ecological drivers are not yet understood, LHIMP management is currently collaborating in and supporting research to help address this knowledge gap. This includes genetic research to understand population connectivity for these species throughout their range and help predict future trends in larval supply and population dynamics, as well as population and benthic monitoring to better understand ecological drivers and impacts of sea urchin grazing.

AIMS

The primary aims of this report are to provide an overview of condition of shallow reef biodiversity in the LHIMP, and identify ecological patterns and trends including differences between management zones. These include measures of reef fish diversity and biomass, invertebrate diversity and densities, trends in threatened, endemic or targeted fish species, and in the cover of benthic organisms such as corals and macroalgae. In particular, assessment of change in biodiversity values is investigated and discussed in relation to the ongoing threats of climate change, fishing pressure, and sea urchin grazing impacts.

LHIMP staff also identified information gaps which were included as specific aims for this report:

1. Assessment of the impacts of coral bleaching events in 2010 and 2019 (and anecdotal observations from the 2024 bleaching event)
2. Assessment of sea urchin densities and dynamics, particularly with respect to impacts on macroalgal cover
3. Assessment of trends in the green macroalgae *Caulerpa spp.* at key sites in response to reported concerns

2 Methods

Following the field survey activities described in previous reports (Stuart-Smith et al. 2015b, Stuart-Smith RD 2019), Reef Life Survey teams undertook monitoring over 2 week periods in 2020, (between February 9th to 22nd) and 2022 (between March 13th to March 25th, plus an additional site on 5th May 2022). The teams consisted of experienced trained RLS volunteers from the mainland and local LHIMP staff. The teams surveyed 93 transects from 36 sites in 2020 and 110 transects from 36 sites in 2022 taking the total number of transects surveyed during LHI UVC reef monitoring from 2006 to 2022 to 988. Surveys in 2006 and 2008 were undertaken by Aquenal Pty Ltd, with those from 2009 onward undertaken by RLS and collaborators as above, resulting in some minor differences in methodology which are noted where relevant throughout the results. All sites with years surveyed are shown in Table 1 and the distribution of sites in Figure 1.

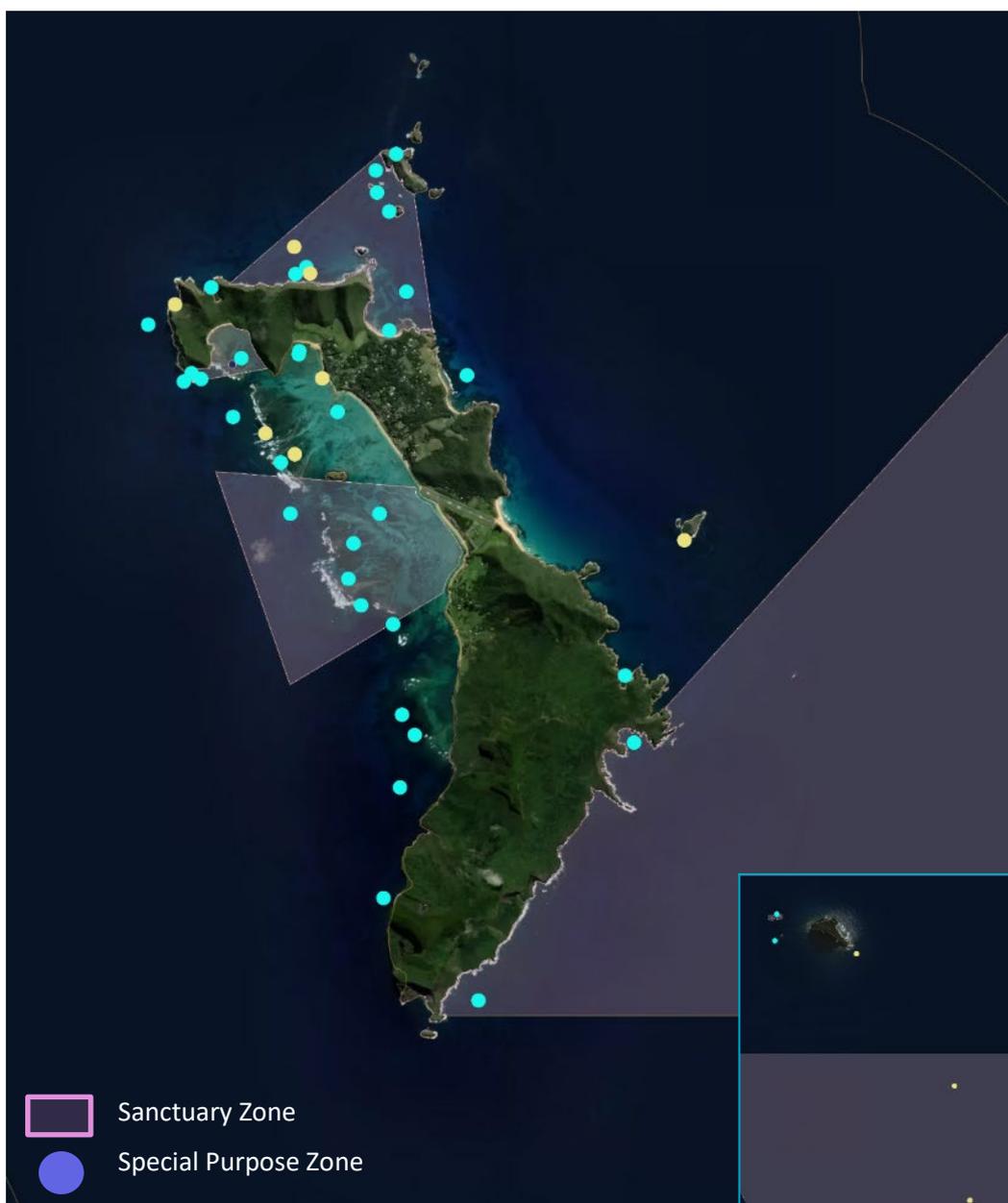


Figure 1. Map of reef monitoring sites and zones in the Lord Howe Island Marine Park with Balls Pyramid inset. Blue sites are core long-term sites, surveyed > 7 times since 2006. Yellow sites have been surveyed < 7 times. Three small Special Purpose Zones for fish feeding occur inside Sanctuary Zones (pink) with all other areas being Habitat Protection Zone.

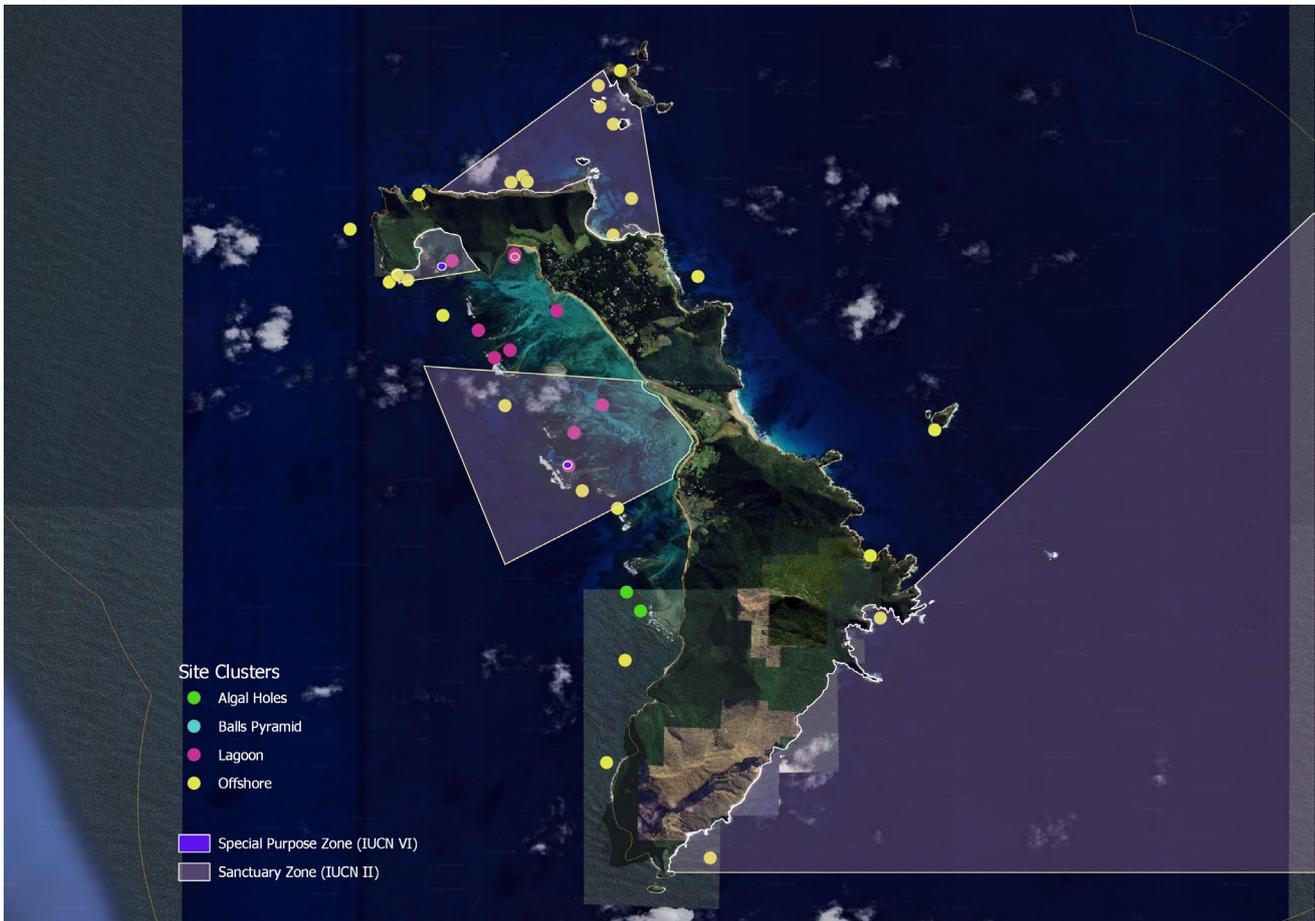


Figure 2 Map of reef monitoring sites and zones in the Lord Howe Island Marine Park with Balls Pyramid inset. Sites have been grouped into ecological communities (Lagoon, Algal Holes, Lord Howe Island (LHI) Offshore, and Balls Pyramid) based on habitat type and reef structure.

FISH SURVEYS (METHOD 1)

Fish census protocols involved a diver laying out a 50 m transect line along a depth contour on reef. All fish species sighted within 5 m x 50 m blocks either side of the transect line were recorded on waterproof paper as divers swam slowly along the line (Figure 2). The number and estimated size category of each species were also recorded. Size categories used were 25, 50, 75, 100, 125, 150, 200, 250, 300, 350, 400, 500, 625 mm, and 125 mm categories above, which represent total fish length (from snout to tip of tail). All species sighted within the blocks were recorded, including those with unknown identity. Photographs were used to later confirm identities with appropriate taxonomic experts, as necessary. In occasional circumstances when no photograph was available, taxa were recorded to the highest taxonomic resolution for which there was confidence (e.g. genus or family, if not species). Other large pelagic animals such as mammals, sea snakes, turtles and cephalopods were also recorded during RLS Method 1 fish surveys, but not considered in analyses focusing on fishes. Species observed outside the boundaries of the survey blocks or after the fish survey had been completed were recorded as 'Method 0'. Such records are a presence record for the time and location but were not used in quantitative analyses at the site level. 'Method 0' sightings were also made of invertebrates and any other notable species.

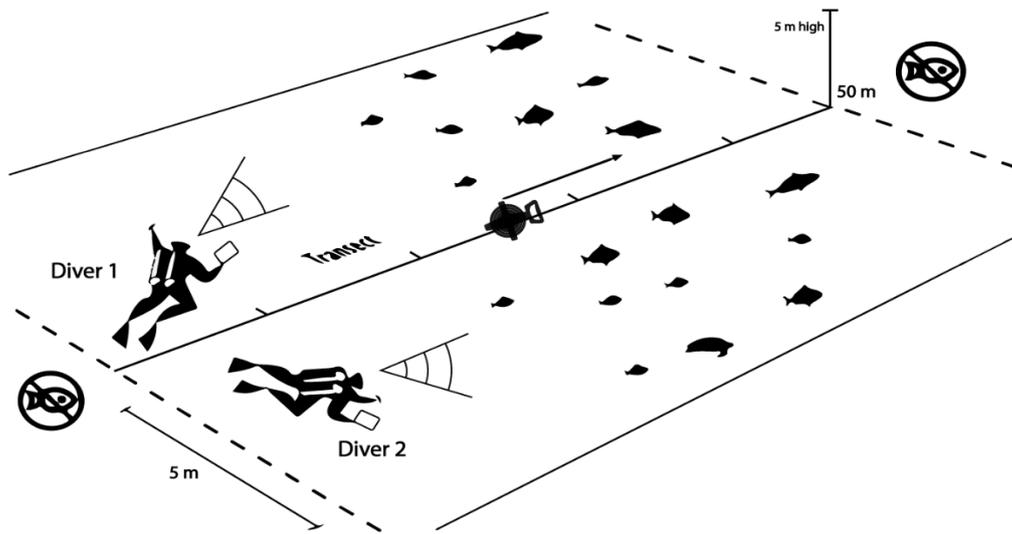


Figure 3. Stylised representation of Method 1 survey technique

MACROALGAL AND SESSILE INVERTEBRATE SURVEYS (METHOD 2 AND METHOD 3)

Large macroinvertebrates (echinoderms, molluscs and crustaceans > 2.5 cm) and cryptic fishes were surveyed along the same transect lines set for fish surveys. Divers swam near the seabed, up each side of the transect line, recording all mobile macroinvertebrates and cryptic fishes on the reef surface within 1 m of the line (Figure 3). This required searching along crevices and undercuts, but without moving rocks or disturbing corals. Cryptic fishes include those from pre-defined families that are inconspicuous and closely associated with the seabed (and are thus disproportionately overlooked during general Method 1 fish surveys). The global list of families defined as cryptic for the purpose of RLS surveys can be found in the online methods manual (www.reeflifesurvey.com/methods). As data from Method 2 were collected in blocks of a different width to protocols applied for Method 1, and were analysed separately from those data, individuals of cryptic fishes known to already be recorded on Method 1 were also recorded as part of Method 2. Sizes were estimated for cryptic fishes within the same size bins as for Method 1.

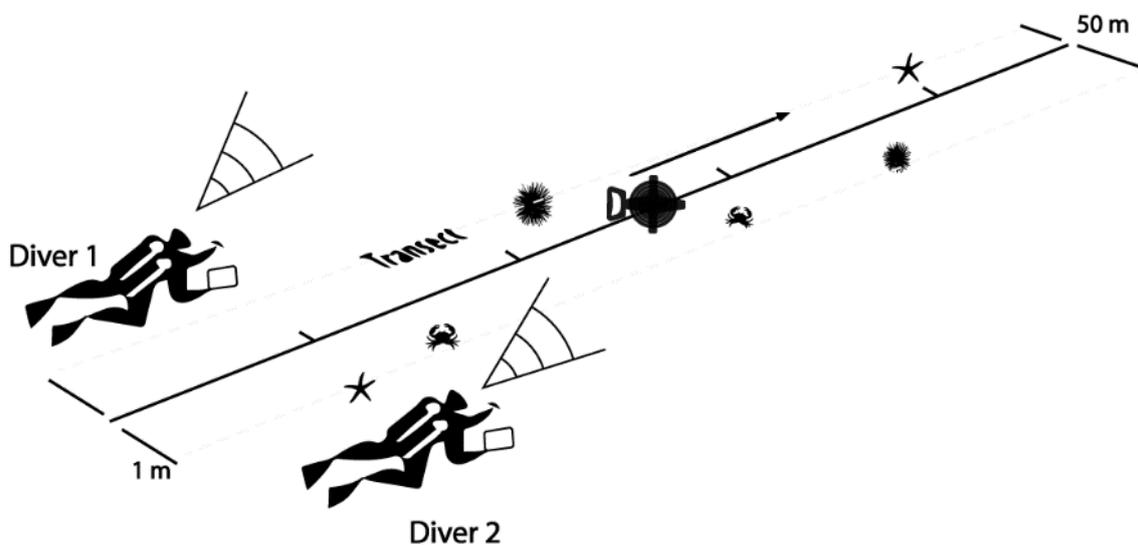


Figure 4. Stylised representation of Method 2 survey technique

Information on the percentage cover of sessile animals and seaweeds along the transect lines set for fish and invertebrate censuses was recorded using photoquadrats taken sequentially each 2.5 m (or 5 m, see below) along the 50 m transect. Digital photoquadrats were taken vertically downward from a height sufficient to encompass an area of at least 0.3 m x 0.3 m. When a wide-angle lens was used and the photoquadrats encompassed at least 0.5 m x 0.5 m, sometimes only 10 images were taken (one every 5 m). The percentage cover of different macroalgal, coral, sponge and other attached invertebrate species in photoquadrats was digitally quantified in the laboratory using the Coral Point Count with Excel extensions (CPCe) software (Kohler and Gill, 2006) and Squidle+ (www.squidle.org). A grid of 5 points was overlaid on each image and the taxon lying directly below each point recorded. Identification was to the lowest possible taxonomic resolution, with taxa for which identification was uncertain grouped with congeners or other members of the family or order.

DATA PREPARATION AND STATISTICAL ANALYSES

Data on fish densities and size structure were used to estimate species-specific biomass values. Species' length-weight relationships provided in Fishbase were used, following the same calculation as for previous papers and reports (Edgar et al. 2011, Stuart-Smith et al. 2015c, Stuart-Smith et al. 2019). Fish biomass and abundance were generally correlated, so biomass was used as a proxy for both when assessing trends in fish assemblages. Raw fish abundance data was, however, used to calculate Reef Fish Thermal Index (described below). Abundance data was also used to assess trends in cryptic fish given they show less size variability, and invertebrates (described as 'density') for which individual sizes were not recorded. For some species such as large highly mobile schooling fish, presence absence data was used to assess trends given the low ecological signal-to-noise ratio in their abundance and biomass data. Similarly, for some invertebrate species such as sea urchins temporal trends were further analysed for a subset of sites where the species was present only, to identify absolute magnitude of change in density without zero-inflation effects on the data. Additional covariate data for fishes included thermal affinity values based on their realised temperature distributions (Stuart-Smith et al. 2015a, Stuart-Smith et al. 2017b) and trophic group, based on the major food sources for each species (Stuart-Smith et al. 2013).

Multivariate analysis of each of the fishes, mobile invertebrates and benthic photoquadrat data was undertaken using non-metric multidimensional scaling (nMDS). Fish biomass and invertebrate densities were $\log(X+1)$ transformed, and benthic percentage cover values square-root transformed, and Bray-Curtis Similarity calculated to relate the assemblage structure in each sampling unit to the others. Previous reports showed clear divisions in assemblage structure between sites in the lagoon, Algal Holes area, Balls Pyramid area and other offshore reefs around LHI (Stuart-Smith et al., 2015). Thus, to visualise general trends through time, sites were aggregated within SZ and HPZ in each of these Ecological Communities.

For fishes, additional univariate analyses were used to investigate metrics representing assemblage level change (species richness, total fish biomass, Large Reef Fish Index, Reef Fish Thermal Index), changes within trophic groups (biomass of herbivores, planktivores, corallivores, benthic invertivores, higher carnivores), and the biomass of key species individually. For invertebrates, univariate metrics included species richness, total sea urchin densities (minus the small burrowing species *Echinostrephus*), and densities of each of the sea urchin species individually.

The Reef Fish Thermal Index (RFTI) is the community temperature index (CTI) calculated for the reef fish assemblage, which is endorsed and supported by the Biodiversity Indicators Partnership (<https://www.bipindicators.net/indicators/reef-fish-thermal-index>) and has been applied as a way to measure biodiversity change specifically related to changing sea temperatures (Stuart-Smith et al. 2015a, Stuart-Smith et al. 2017a, Stuart-Smith et al. 2022). Calculation details are provided in those references, but it is basically the mean of the thermal affinity of fish species recorded on transects, weighted by the log of

their abundance. Thermal affinity values are a species level trait calculated from species' global thermal distributions and are curated by Reef Life Survey. The Large Reef Fish Index (LRFI or also known as B20) is also endorsed by the Biodiversity Indicators Partnership and is the sum of biomass of all individual fishes recorded on a transect that are 20 cm or larger. It is used here instead of the previous metric which used 30 cm as a cut-off, following recommendations from a national analysis (Stuart-Smith et al. 2017a).

Time series analyses used GAMMs, where the metric of interest was modelled among sites in SZs and HPZ separately as a function of time, with site as a random effect. This approach differed to that used for evaluation of trends in previous reports, and importantly, allowed the inclusion of all monitoring sites for time series, rather than the subset of sites most consistently monitored since the 2006 and 2008 Aquenal surveys. Although nearly all sites have been surveyed in recent monitoring years (see Appendix 1), this modelling approach accounted for the fact that some sites were missing in some years. As a consequence of this different modelling approach, some trends may differ slightly from those presented in previous reports. This, in part, reflects that more sites have been included in those previous years, as well as because the longer time series spanning 10 occasions in 17 years means that small jumps in some sites and years are now better anchored by more persistent trends. Thus, the trends presented here better reflect those across the entire monitoring series and the full set of investigated sites. Included in the GAMM plots are 95% confidence intervals, which when not overlapping with those for the initial year are interpreted to indicate a significant trend through time, and when not overlapping between grouped variables (such as marine park zoning), interpreted as significantly different value between groups.

Trends in the cover of key benthic biota such as coral and macroalgal were assessed using LOESS (locally weighted running line smoother) curves fitted to raw percent cover values (averaged per site), given that insufficient data were available to fit GAMMs at the subset of sites or years assessed. Species observations from photoquadrats were grouped into standardised categories from the multiple datasets available from previous reports and new digitising efforts for un-annotated photoquadrats, retaining the highest taxonomic resolution possible. All available photoquadrats (from 2009 onwards) were re-analysed for key lagoon sites by coral taxonomist Emre Turak, to achieve consistency in coral species-identification and retain persistent records of these annotations in Squidle+. Benthic assemblage composition was assessed at key sites affected by marine heatwaves (coral dominated lagoon sites) and previously reported outbreaks of Lamington Sea Urchin (*Tripneustes australiae*) using stacked box plots color-coded for major functional groups of benthos. For these figures, dominant coral genera remained separated, given they are foundational species providing reef structure through their different morphologies, and that they vary in their life-histories and susceptibility to bleaching.

Heatmaps were created using Inverse distance weighted (IDW) interpolation. Areas spanning yellow to red represent higher values and white to blue areas represent low to zero values.

3 Results

A total of 203 surveys were undertaken in 2020 and 2022, taking the total number of surveys undertaken during reef monitoring by Aquenal, RLS and LHIMP teams to 988 since 2006.

THREATENED AND INVASIVE SPECIES

No new records of threatened and protected species were made during 2020 or 2022 surveys reported here. Of those previously recorded, Green Turtles (*Chelonia mydas*; listed as Endangered on the IUCN Red List and Vulnerable under both the NSW *Biodiversity Conservation Act 2016* and Commonwealth *Environment Protection and Biodiversity Conservation Act 1999*) were recorded again in both years, blotched fantail ray (*Taeniurops meyeri*, listed Vulnerable on the IUCN Ref List) were recorded in 2020, and three observations of Black Cod (*Epinephelus daemeli*; listed as Vulnerable and protected in NSW waters under the NSW *Fisheries Management Act 1994*) were made in 2020. No Black Cod were observed in 2022. None of the above threatened species were recorded in sufficient numbers for analysis of population trends but additional targeted monitoring has been undertaken for Black Cod (Harasti et al. 2022).

Hawksbill turtles (*Eretmochelys imbricata*, listed as Critically Endangered on the IUCN Red List and Vulnerable under the *Environment Protection and Biodiversity Conservation Act 1999*) are also known to commonly occur in the LHIMP and Ballina Angelfish (*Chaetodontoplus ballinae*) and sygnathids such as Booths Pipefish (*Halicampus boothae*) which are protected in NSW waters under the NSW *Fisheries Management Act 1994* are also occasionally observed in the LHIMP but none of these species were recorded on survey transects in 2020 or 2022. Doubleheader (*Coris bulbifrons*; Vulnerable, IUCN), Galapagos Sharks (*Carcharhinus galapagensis*; Near Threatened, IUCN), Elegant Wrasse (*Anampses elegans*, protected in NSW waters under the NSW *Fisheries Management Act 1994*) and Bluefish (*Girella cyanea*, also protected in NSW waters other than the LHIMP where a daily bag limit of 5 applies) were the most common threatened and protected fishes present on LHI reefs, and trends were examined individually in the section below.

The Chevron Butterflyfish (*Chaetodon trifascialis*) is listed as Near Threatened on the IUCN Red List, due to reliance on Acropora corals as a food source, which are also in decline throughout its range as a result of bleaching and other impacts. This species has consistently been recorded on reefs in the LHIMP, but with fewer records in recent surveys consistent with a general decline of corallivores. The endemic McCulloch's Anemonefish (*Amphiprion mccullochi*) has also been recently listed as Critically Endangered under the NSW *Fisheries Management Act 1994* and *Environment Protection and Biodiversity Conservation Act 1999*, due to declining populations and highly restricted geographic range. Declines in this species are likely driven by climate change and heat stress impacts on its host anemones. Targeted monitoring of this species has been ongoing since 2009 (Harasti et al. 2022) and results presented here confirm the declining trend for this species.

Many of the coral species recorded during RLS surveys are also likely to be threatened. In November 2024, IUCN Red List of Threatened Species Corals Assessment revealed that 44% of reef building coral species in the world were threatened with extinction. This included 66 species of hard coral species known to occur in LHIMP, which were assessed as 'Endangered'. Research is also underway using new molecular techniques to reclassify coral species, which will likely reveal new endemic species and a distinct coral assemblage in the LHIMP, which therefore may have even higher vulnerability to extinction than currently documented.

No introduced fish or invertebrate species were observed during surveys. Previously *Caulerpa taxifolia* has been reported as cryptogenic in the LHIMP (meaning it's status as introduced or native is unknown, Aquenal

2006). However, it is most likely to be native in the LHIMP given there are early records of it occurring in high abundance in the LHIMP since 1933 (Lucas, 1935) and the LHIMP is within the indigenous tropical/subtropical range of this species in Australian waters (Phillips, 2002 and also noted by Stuart-Smith et al., 2015). Nevertheless, in response to reports and concerns about increased abundances, trends in this species are shown in the results for *benthic assemblages* below.

REEF FISH ASSEMBLAGES

No previously unrecorded species of fish (at the time of the surveys) were recorded in the LHIMP in 2020 or 2022, so the total number of fish species recorded during RLS LHIMP surveys remains at 405 (see Appendix 4). All previous new distribution records have been included in Malcom Francis' checklist of 654 coastal fishes recorded in the LHIMP (Francis 2022) with new records from RLS divers represent almost 9% (at least 57) of these species, highlighting the significant contribution of RLS divers to the knowledge of LHIMP biodiversity.

Overall assemblage structure

Fish assemblages differ in composition and abundance between shallow reefs of the Lagoon, Algal Holes, Balls Pyramid area and LHI Offshore ('other') sites (Figure 4). These groups are referred to as Ecological Communities throughout this report. As per Stuart-Smith et al. (2019) the Lagoon Ecological Community is characterised by high abundances of corallivores (such as Butterflyfish, *Chaetodon spp.*), coral-dwelling fishes (such as *Dascyllus* species), parrotfishes, and omnivorous and farming damselfishes (including McCulloch's Anemonefish, *Amphiprion McCullochi*). The Balls Pyramid Ecological Community is characterised by a combination of planktivorous damselfishes (such as One Spot Puller, *Chromis hypsilepis*) and large herbivores (such as Drummer, *Kyphosus spp.*), several fishes from the Serranidae family, and the deeper-water Conspicuous Angelfish (*Chaetodontoplus conspicillatus*). The LHI Offshore Ecological Community shares these several species along with high numbers of the Bronze Bullseye (*Pempheris analis*), whilst the Algal Holes Ecological Community along with high abundances of Green Moon Wrasse (*Thalassoma lutescens*), the coral sea Gregory (*Stegastes gascoeni*) and the Three-band Butterflyfish (*Chaetodon tricinctus*) (Stuart-Smith et al. 2019). A further distinction is clear between fishes at HPZ and SZ sites in the Lagoon Ecological Community.

Surveys in 2020 and 2022 generally sit in the same multidimensional space as previous years (see darker symbols in Figure 4), indicating that fish assemblage structure has not changed much through time in contrast to the clear separation of Ecological Communities (i.e. temporal trends are smaller than spatial patterns). Some changes in assemblage structure were noted from 2006 to 2010 driven by sites in SZs, and therefore interpreted to potentially indicate effects from reduced fishing pressure (Edgar et al., 2011). However, when examined over the longer time series fish assemblage structure does not show strong or sustained temporal trends in SZs compared with HPZ.

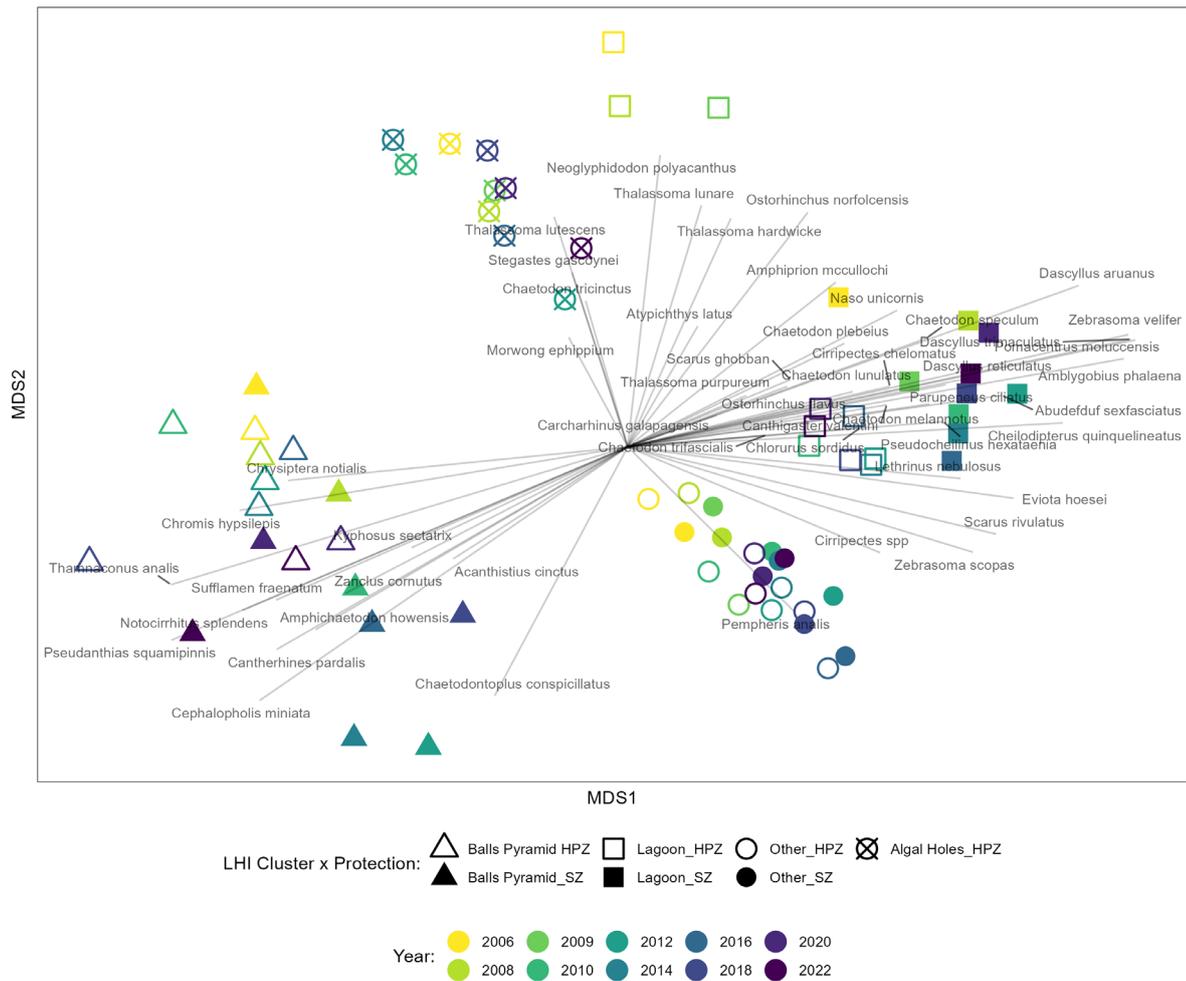


Figure 5. Non-metric multidimensional scaling plot of reef fish assemblage structure recorded on RLS surveys in the LHIMP. Symbols represent the composition and biomass of species averaged among sites within each of the four major Ecological Communities previously identified (shown by different symbol shapes), and surveys in different years represented by a colour gradient from light (oldest surveys) to dark (latest surveys).

Fish richness (the number of fish species recorded per 500 m² transect) fluctuated over the monitoring period, but there was a net increase in both SZ and HPZ from 2006 to 2022 (Figure 5). Fluctuations also occurred in total fish biomass, and despite previous concerns about declines in SZ (Stuart-Smith et al., 2015) there was no net decrease in SZ from 2006 to 2022 and a sustained increase in HPZ over this time. The Large Reef Fish Index (LRFI) which represents the biomass of exploitable sized fishes (>20 cm) also showed fluctuations which mirrored total fish biomass. Previous LRFI increases in SZ were interpreted as accumulation of large fish biomass in response to reduced fishing pressure (Stuart-Smith et al. 2015, 2019). However, again there was not net change in SZ from 2006 to 2022 over and above these fluctuations.

There was some synchronicity in fluctuations between fish richness and biomass metrics (total biomass and LRFI) which may indicate ecosystem-wide responses to unidentified ecological drivers. Fluctuations in biomass are also likely driven by stochastic detections of large bodied and/or schooling species as described below. It has also been previously suggested that fish biomass may fluctuate according to cycles in plankton delivery to the island, influencing planktivore biomass with flow-on effects to overall reef productivity (Stuart-Smith et al. 2019).

There was a lack of clear, sustained trends in fish richness or biomass associated with reduced fishing pressure in SZ over and above these fluctuations. The modelled trends also indicate that fish richness, total biomass, and LRFI remain comparable (or slightly lower) in SZ than HPZ. Potential reasons for this include low fishing pressure in HPZ, continued illegal fishing pressure in SZ, high mobility of target species across zones, insufficient time to detect delayed ecological effects, or effects being obscured by other sources of variation. These are considered further in the report discussion.

The most recent (2022) data show that almost all hotspots for fish richness, total biomass and LRFI occur in SZs. Hotspots of species richness occur around the Admiralty Islands (in the Neds Beach & Admiralty Island Sanctuary Zone) and in the Lord Howe Island Lagoon Sanctuary Zone. Hotspots for both total biomass and LRFI occur in both around the Admiralty Islands and Muttonbird Island, and in the Observatory Rock Sanctuary Zone adjacent to Balls Pyramid (Figure 5 heatmaps). Although not surveyed in 2020 or 2022, the Balls Pyramid Sanctuary Zone has also previously been found to support exceptionally high biomass of fish from all trophic levels (at South East Rock, Edgar et al., 2011).

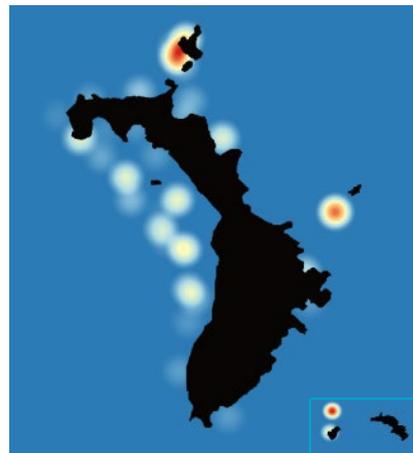
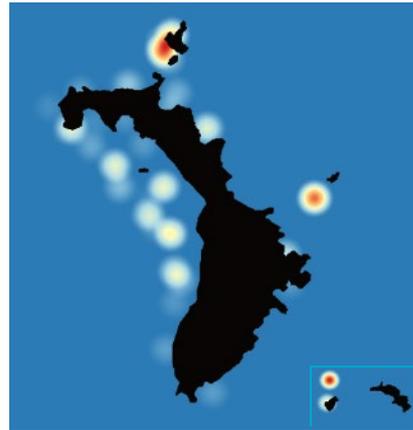
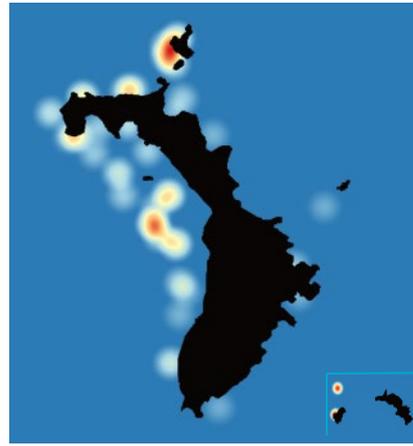
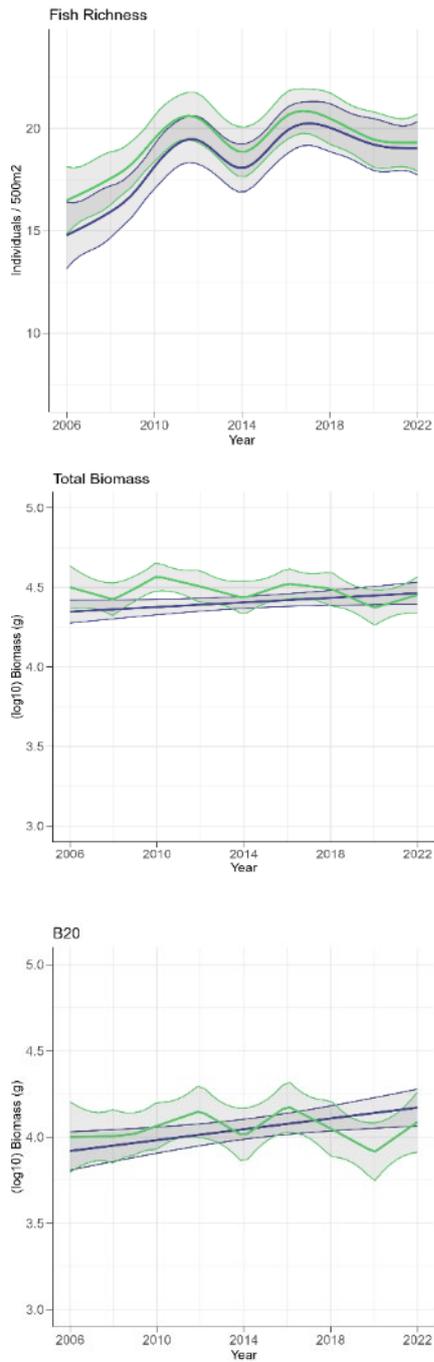


Figure 6. Trends in reef fish assemblage level metrics through time from RLS surveys in HPZ sites (purple) and SZ sites (green). Richness on the top plot is the number of fish species per 500 m² transect, and biomass values in the middle and bottom plot are per 500 m² and have been log(x+1) transformed. B20 denotes the Large Reef Fish Index. Trend lines represent mean trends from Generalised Additive Mixed Models using 'Site' as a random effect. Confidence bands represent 95% confidence intervals. Heatmaps showing hotspots for richness, biomass and large fishes around LHI based on 2022 surveys, with Balls Pyramid inset.

Climate change

There have been several mass coral bleaching events in the LHIMP since 2006 (see *benthic assemblages* results below) which may have been expected to affect reef fishes through habitat change or degradation (see *trophic structure* results below). However overall changes in fish assemblage structure have not been detected. Similarly, ocean warming driven by climate can cause topicalization of reef fish assemblages, whereby tropical species increase in diversity and abundance and temperate species are displaced or decrease in abundance. However, the Reef Fish Thermal Index (RFTI) has remained remarkably stable in the LHIMP since 2006, both inside and outside SZs (Figure 6). Thus, there is little evidence that turnover in species composition or changing abundances of individual species has related to their thermal affinities.

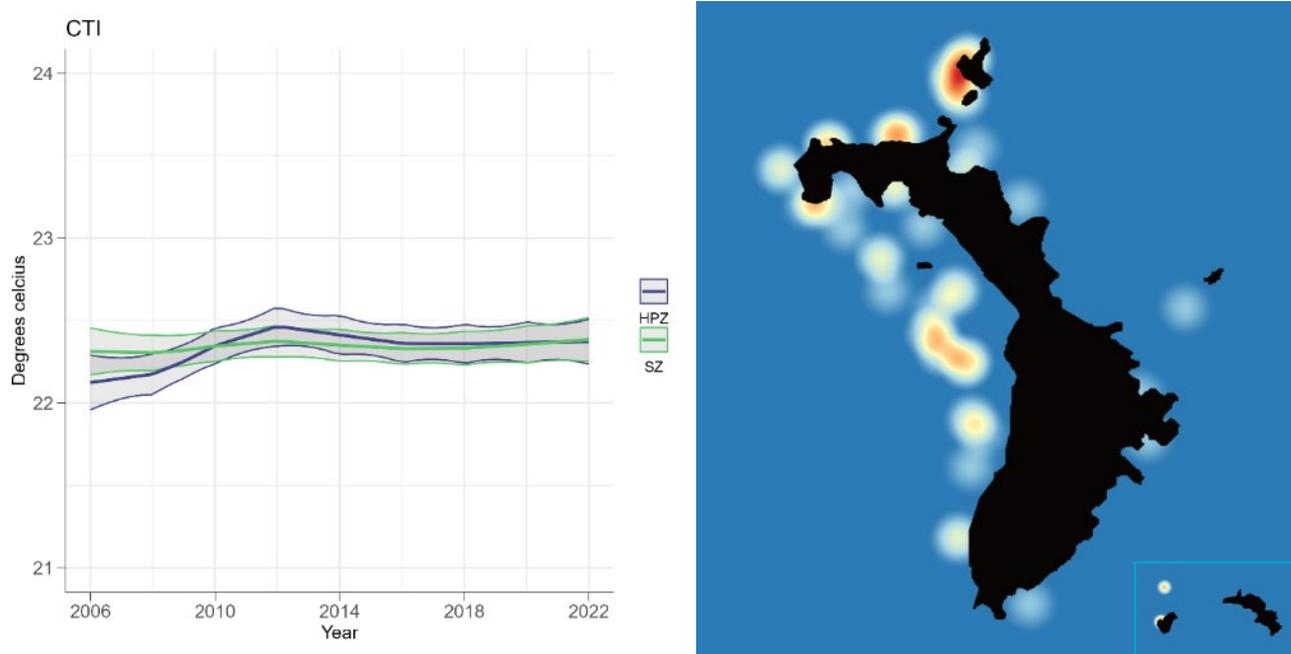


Figure 7. Trends in the Reef Fish Thermal Index (the community temperature index for fishes) through time from RLS surveys in HPZ sites (purple) and SZ sites (green). Trend lines represent mean trends from Generalised Additive Mixed Models using 'Site' as a random effect. Confidence bands represent 95% confidence intervals. Heatmap showing hotspots of RFTI values in 2022 in the LHIMP based on 2022 surveys (with Balls Pyramid inset).

Trophic structure

The biomass of fishes in each major trophic groups was further investigated given they perform different ecological functions, so specific changes may have flow-on effects (for example 'top-down' effects from changes in predation pressure) or be indicators of ecological change (for example 'bottom-up' effects from changes in benthic cover). Any changes in specific trophic groups may also therefore provide insights into broader ecological trends documented over time.

Benthic invertivores such as Doubleheader (*Coris bulbifrons*) consume benthic invertebrates including sea urchins. Corallivores such as Butterflyfish (*Chaetodon spp.*) feed on coral and are therefore indicators of healthy coral reefs. Higher carnivores feed on other fish and include common target species such as Kingfish (*Seriola lalandi*) as well as the most common apex predator in the LHIMP the Galapagos Shark (*Carcharhinus galapagensis*). Planktivores feed on plankton and include abundant species such as One Spot Puller (*Chromis*

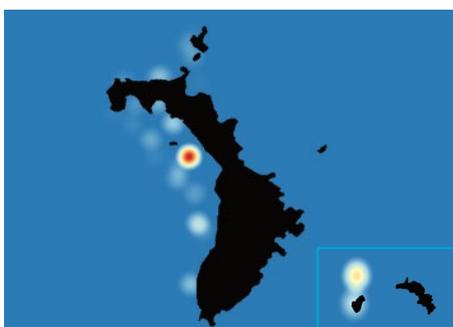
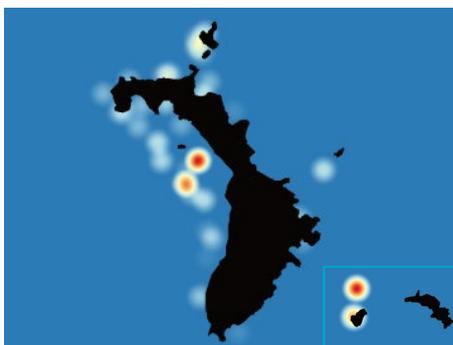
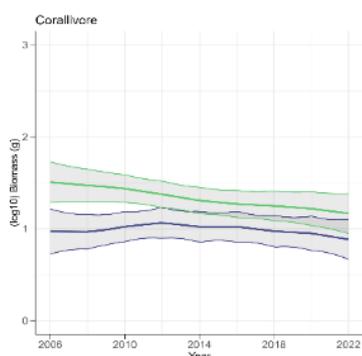
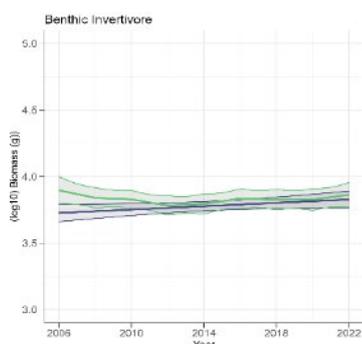
hypsilepis), which can be sensitive to broad oceanic processes influencing plankton supply. Herbivores feed on macroalgae and include large schooling species such as Sawtail (*Prionurus maculatus*) and Drummer (*Kyphosus spp.*), as well as target species such as Bluefish (*Girella cyanea*).

No large net changes in fish biomass were observed in these trophic groups from 2006 to 2022, although a gradual increase in higher carnivores was observed across both HPZ and SZ sites, and a gradual decrease observed in corallivore biomass (Figure 7). The former is heavily influenced by Galapagos Shark observations (see *species of interest* results below). Being large, highly mobile predators with low site fidelity and often swimming more than 5m above the reef, records of Galapagos Sharks on transect surveys are somewhat stochastic and so trends driven by this species should be interpreted conservatively.

Corallivores, in contrast, have high site fidelity and generally stay within 5m above the reef so changes in their biomass can more confidently be attributed to detection of an ecological trend. They are also directly affected by changes in benthic cover (loss of coral) so are robust indicators of coral reef health. The observed decline in their biomass may be linked to declines in coral cover in the lagoon, particularly the Lord Howe Island Lagoon Sanctuary Zone where they are most abundant (see *benthic assemblage* results below).

Cyclical trends in planktivores have previously been reported with peaks at eight-year intervals. These may have been driven by temporal dynamics in plankton supply or recruitment success, and potentially contributed to trends in total biomass noted by Stuart-Smith et al. (2019). However, over the longer time series these cyclical trends are not as apparent.

Herbivore biomass at SZ sites fluctuated from 2006 to 2022, and while this has previously been interpreted as ecological instability associated with interactions between herbivores and macroalgae (particularly *Caulerpa spp.*, as per Stuart-Smith et al. 2015, 2019) it may also reflect stochastic observations of large bodied and highly mobile schooling species including Sawtails (*Prionurus maculatus*), Bluefish (*Girella cyanea*) and other Drummer (*Kyphosus spp.*) which can dominate biomass in this trophic group.



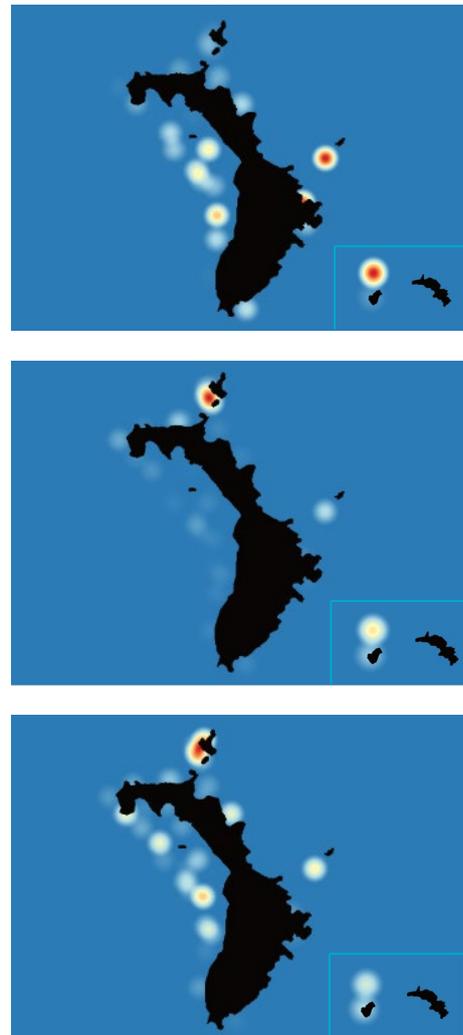
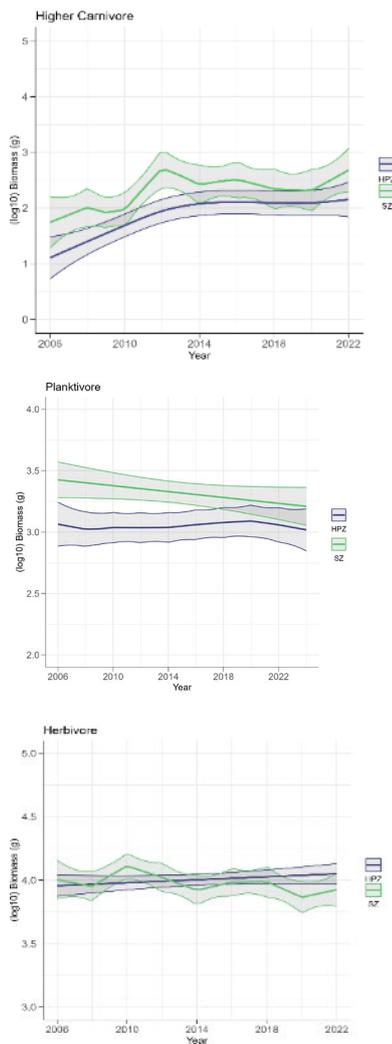


Figure 8. Trends in the biomass of major reef fish trophic groups through time from RLS surveys in HPZ sites (purple) and SZ sites (green). Biomass values on the Y-axis are per 500 m² and have been log(x+1) transformed. Trend lines represent mean trends from Generalised Additive Mixed Models using 'Site' as a random effect. Confidence bands represent 95% confidence intervals. Heatmaps showing hotspots for each trophic group in 2022.

Other species of interest

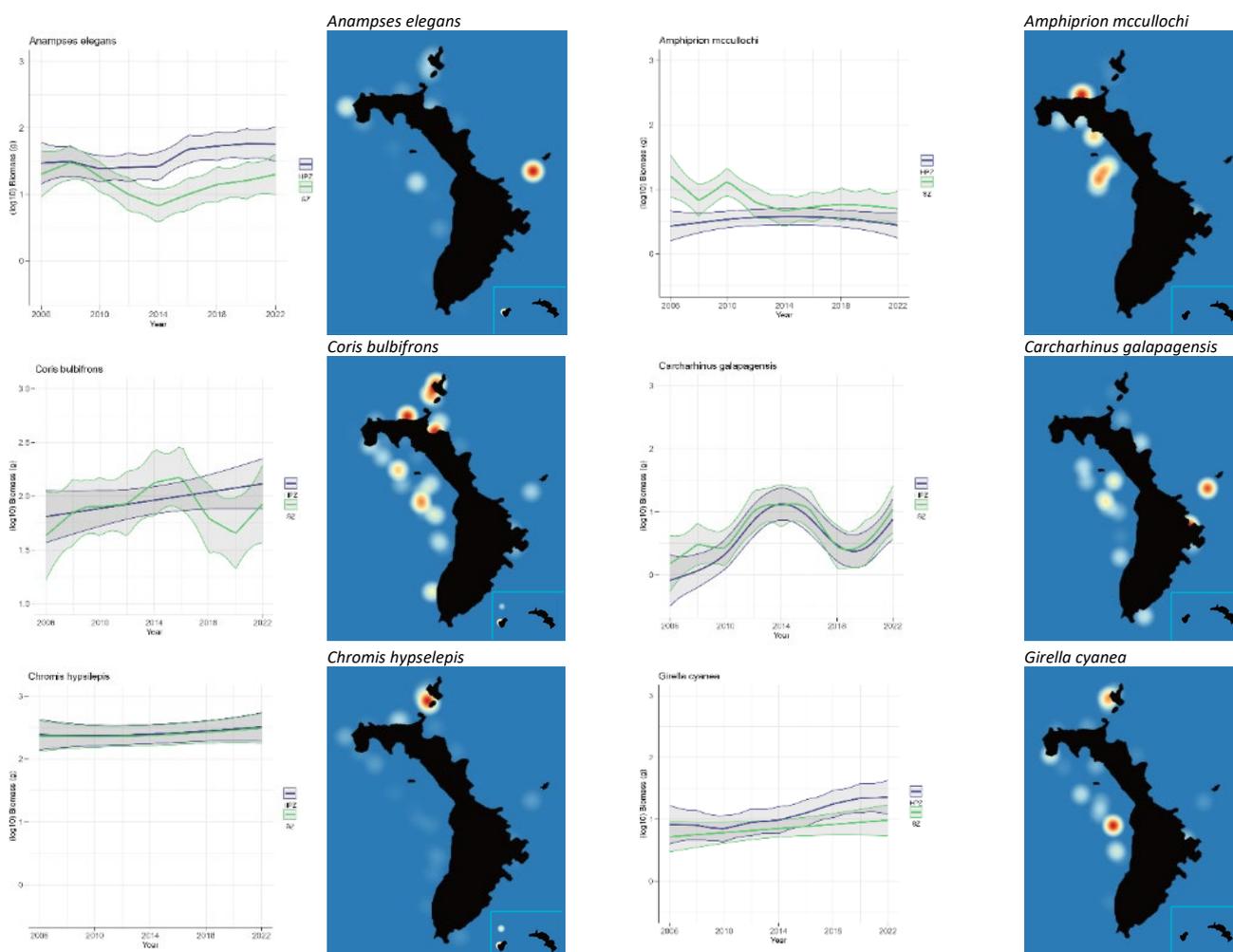
Trends in the biomass of a number of other key species were considered individually (Figure 8). These include species which are protected, endemic or contribute disproportionately to total biomass so are most likely to drive overall trends. The Galapagos Shark (*Carcharhinus galapagensis*) and endemic Doubleheader (*Coris bulbifrons*) are large species which therefore represent a large proportion of total fish biomass. Both showed large decreases in biomass in 2020 and subsequent increases in 2022 that corresponded to, and may have driven, similar overall trends in LRFI and total fish biomass. Over and above these fluctuations, there was a small net increase in Galapagos Shark over the time series but no large net change in Doubleheader from 2006 to 2022.

Other species that can represent a large proportion of total fish biomass include Bluefish (*Girella cyanaea*) which is a large herbivore, Onespot Puller (*Chromis hypsilepis*) which is a small but highly abundant planktivore, Luculent Wrasse (*Pseudolabrus luculentus*) which is a small but highly abundant benthic invertivore, and Sawtail (*Prionurus maculatus*) which is a large schooling herbivore. These species either had relatively stable biomass in both HPZ and SZs (in the case of Bluefish and Onespot Puller), or fluctuating biomass that did not correspond with trends in total fish biomass. Previous declines in Bluefish and Doubleheader were noted as potential impacts from fishing pressure including illegal fishing in SZs (Edgar et

al. 2011, Stuart-Smith et al. 2015, 2019). However, the long-term trends show these declines have not been sustained and these species show little net change across the full monitoring period other than slight increases in HPZ.

The endemic McCulloch’s Anemonefish (*Amphiprion mccullochi*) and protected Elegant Wrasse (*Anampses elegans*) showed declines in SZ from 2006 to 2014. A subsequent increase in *Elegant Wrasse* biomass was recorded from 2014 to 2022 but biomass of McCulloch Anemonefish did not show similar indications of recovery. This species shows extremely high site fidelity, remaining in or close to the host anemone they inhabit so, as for corallivores, declining trends are more likely to be indicative of important ecological changes than for other more mobile species. SZ sites previously supported greater biomass of McCulloch’s Anemonefish than HPZ but this is no longer the case. When analysed according to abundance rather than biomass (not shown here) a persistent decline for this species becomes even more evident with a net decrease of around 30% in SZs.

Schools of Kingfish (*Seriola lalandi*) represented large biomass values but were stochastic in the data, being mobile pelagic predators with low site fidelity on shallow reefs. Trends in their biomass were therefore highly variable, reflecting artefacts of sampling rather than ecological changes. Expressing trends for this species as frequency of occurrence based on presence or absence (i.e. the proportion of surveys on which any individuals were observed) provides a clearer indication of any population trends, at the expense of within-year replication. In the last report, Kingfish frequency had dropped from 9.4% to 6.5% of surveys (from 2014 to 2018) and it dropped further to 2.2% in 2020 but increased again to 6.4% in 2022. Trends in Kingfish are also monitored through catch data submitted by charter fishing operators (Figueira and Harianto 2022).



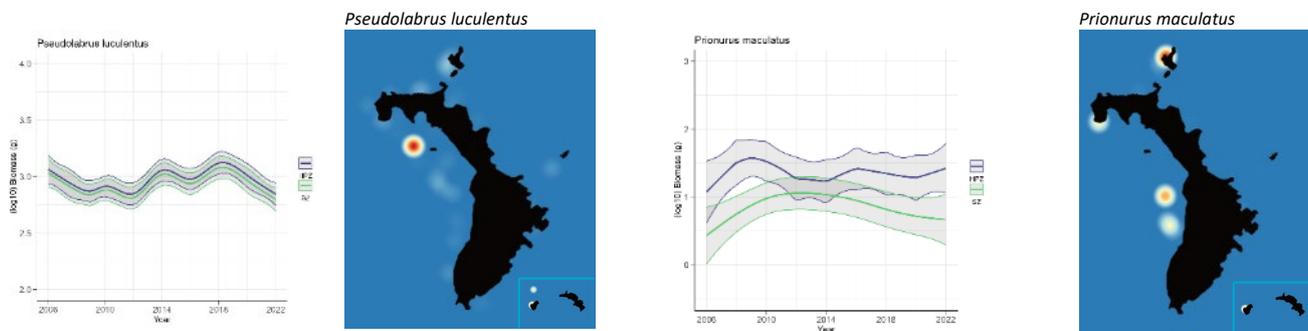


Figure 9. Trends in the biomass of key reef fish species through time from RLS surveys in HPZ sites (purple) and SZ sites (green). Biomass values on the Y-axis are per 500m² and have been log(x+1) transformed. Trend lines represent mean trends from Generalised Additive Mixed Models using ‘Site’ as a random effect. Confidence bands represent 95% confidence intervals. Heatmaps showing biomass hotspots of particular fish species in 2022.

MOBILE INVERTEBRATE ASSEMBLAGES

Overall assemblage structure

Mobile invertebrate assemblages show similar (albeit less distinct) separation into the four Ecological Communities in the LHIMP (Figure 9). The Lagoon Ecological Community is characterised by low abundance of most mobile invertebrates, apart from the sea hare *Aplysia dactylomela* several crabs species, and the presence of the giant clam species *Tridacna maxima*. The Ball’s Pyramid Ecological Community is characterised by high abundances of several echinoderm species including Savigny’s Longspine Sea Urchin (*Diadema savignyi*) and the Orange Long-armed Sea Star (*Ophidiaster confertus*) and high abundances of Long-spined Sea Urchin (*Centrostephanus rodgersii*), while the LHI Offshore Ecological Community is characterised by similar species along with several species of holothurians and feather stars. The Algal Holes Ecological Community is characterised by high abundances of Purple Sea Urchin (*Heliocidaris tuberculata*) and the endemic Onion Turban sea snail (*Turbo cepoides*) (Stuart-Smith et al. 2019). Surveys in 2020 and 2022 confirmed assemblage structures similar to previous years. The Lagoon Ecological Community showed greatest variability, showing similarities to the Algal Holes Ecological Community in some years.

The number of mobile invertebrate species recorded per survey increased from ~5 species per 50 m² in 2006 to ~7 species in 2022 (Figure 10). However, experience of the authors indicates that this may be an artefact of increased surveyor skill and experience in distinguishing more species of invertebrates including different feather stars and nudibranchs. Differences in the breadth of invertebrates included in surveys undertaken in 2006 and 2008 may also contribute to the early trend of increasing species richness. Hotspots for species richness occur around the Admiralty Islands (in the Ned’s Beach and Admiralty Island Sanctuary Zone) and around Wheatsheaf Islet adjacent to Balls Pyramid.

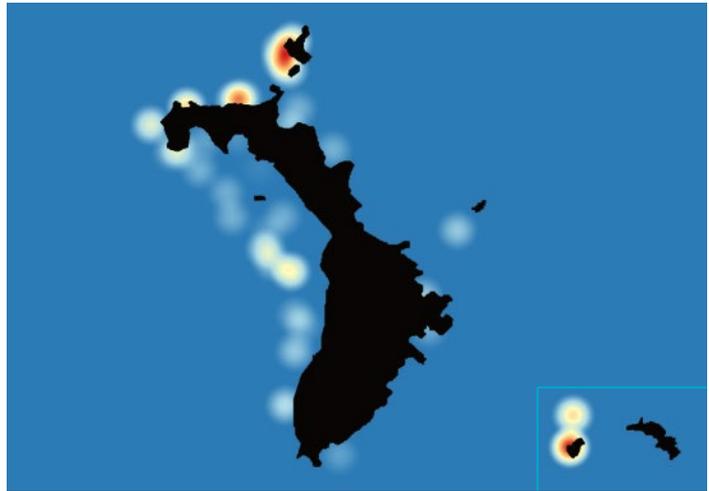
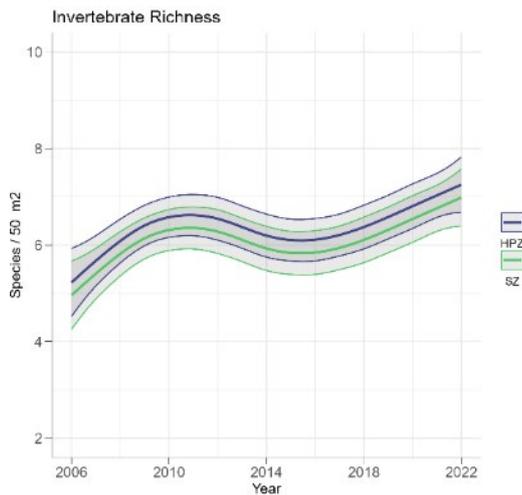


Figure 11. Trends in mobile invertebrate species richness (per 50 m²) through time from RLS surveys in HPZ sites (purple) and SZ sites (green). Trend lines represent mean trends from Generalised Additive Mixed Models using ‘Site’ as a random effect. Confidence bands represent 95% confidence intervals. Heatmap shows invertebrate richness hotspots from surveys in 2022.

Sea urchin populations

Sea urchins represent a large proportion of the LHIMP mobile invertebrate fauna, and play an extremely important ecological role by grazing benthic assemblages, particularly macroalgae. This poses a threat to the endemic macroalgae species and unique Algal Holes Ecological Community in the LHIMP. However, it can also clear suitable substrate for other benthic species to establish including hard coral and crustose coralline algae (CCA) which can in turn facilitate recruitment of other larvae as discussed further below.

Total sea urchin densities have gradually increased across the 17 years of analysis (Figure 11), although this trend is not driven by a uniform increase across all species or in all locations. Previously it has been suggested that benthic invertivores such as Doubleheader may help control sea urchin density, but when the relationship between Doubleheader biomass, benthic invertivore biomass, and total sea urchin density from 2006 to 2022 was assessed no inverse relationship was detected. Similarly, sea urchin density did not appear to be influenced by the abundances of predatory fish during sea urchin population boom events (Valentine et al 2008).



Figure 12. Trends in total sea urchin densities (per 50 m²) through time from RLS surveys in HPZ sites (purple) and SZ sites (green). Densities have been log(x+1) transformed. The left plot is using all data from the LHIMP, while the right plot only uses sites from the Lagoon Ecological Community. Trend lines represent mean trends from Generalised Additive Mixed Models using ‘Site’ as a random effect. Confidence bands represent 95% confidence intervals. *Echinostrephus* spp. burrowing sea urchins are excluded from total urchin counts. Heatmap shows hotspots of urchin density from 2022 surveys.

Overall, the most abundant sea urchin species in the LHIMP is the Long-spined Sea Urchin (*Centrostephanus rodgersii*) which has a regionally endemic subtropical to temperate distribution, followed by the Needle-spined Sea Urchin (*Echinostrephus* spp.) which has a tropical distribution and Purple Sea Urchin (*Heliocidaris tuberculata*) which also has a regionally endemic subtropical to temperate distribution (Edgar et al., 2008, Figure 12). These species tend to occupy different depth ranges, for example Purple Sea Urchin are more common on shallow reefs to 8m depth while Long-spined Sea Urchins are most abundant from 8-14m (Edgar et al. 2008). Based on data from 2022, all sea urchin species occur in highest densities in the LHI Offshore Ecological Community either at Neds Beach (Purple Sea Urchin) or around the Admiralty Islands (all other species). The most common species in the Algal Holes Ecological Community is the Needle-spined Sea Urchin which is a small filter feeder and largely remains within bio-eroded burrows (Westlake et al. 2021) therefore does not itself pose a threat of grazing impacts. Other urchin species currently occur at relatively low densities in this Ecological Community but still pose a high risk of grazing impact if densities increase as discussed below.

Species specific trends in sea urchin density are important to consider given they may have different ecological effects and drivers (Figure 12). Long-spined Sea Urchin (*Centrostephanus rodgersii*) densities continually increased in both SZ and HPZ, with periods of particular increase from 2006 to 2010 and 2014 to 2022. This trend (Figure 12) is presented here on a log transformed y axis which also accounts for zeros and site variability, however, the absolute magnitude of this increase is very large. Considering only surveys where Long-spined Sea Urchin occurred (i.e. removing the down weighting effect of zeros in the data) mean density increased from 40 urchins per 50 m² in 2006, to 140 per 50 m² in 2022. This trend is likely driving an increase in ‘bare’ benthic cover as further described in *benthic assemblages* results. The increasing abundance of this species and expansion of ‘urchin barrens’ associated with it have been highlighted as a concerning trend since early monitoring (Edgar et al. 2008, 2011).

Densities of Burrowing Sea Urchin (*Echinometra mathaei*) also almost doubled from 2006 to 2022, and while this smaller species may have less impact on benthic cover than the larger species mentioned above, there may still be corresponding grazing impacts. A rapidly increasing trend in densities of Needle-spined Sea Urchin (*Echinostrephus* spp.) also occurred, however as mentioned above this species is less likely to impact benthic assemblages. Densities of Savigny’s Longspine Sea Urchin (*Diadema savignyi*) have also steadily increased from 2006 to 2022, and previous analysis noted large increases corresponded with increases in coral cover (Stuart-Smith et al. 2015) which may therefore be ecologically related to densities of this species.

There was no overall increase in Purple Sea Urchin (*Heliocidaris tuberculata*) however it is likely responsible for a localised increase of sea urchins at Neds Beach as described below, and be related to decreasing cover of macroalgae at this site (see *benthic assemblages* results). Overall densities of Lamington Sea Urchin (*Tripneustes australiae* - which has a regionally endemic subtropical to warm temperate distribution, see McLaren et al. 2023) peaked in 2008 before rapidly declining and remaining at low abundances. A localised population boom also occurred around Balls Pyramid in 2022 but did not drive a detectable increase across all sites. As noted previously a more widespread population boom also occurred in 2024 which will be considered in future reports. Given the ecological importance of these populations booms they are analysed further below and grazing impacts assessed in *benthic assemblages* results.

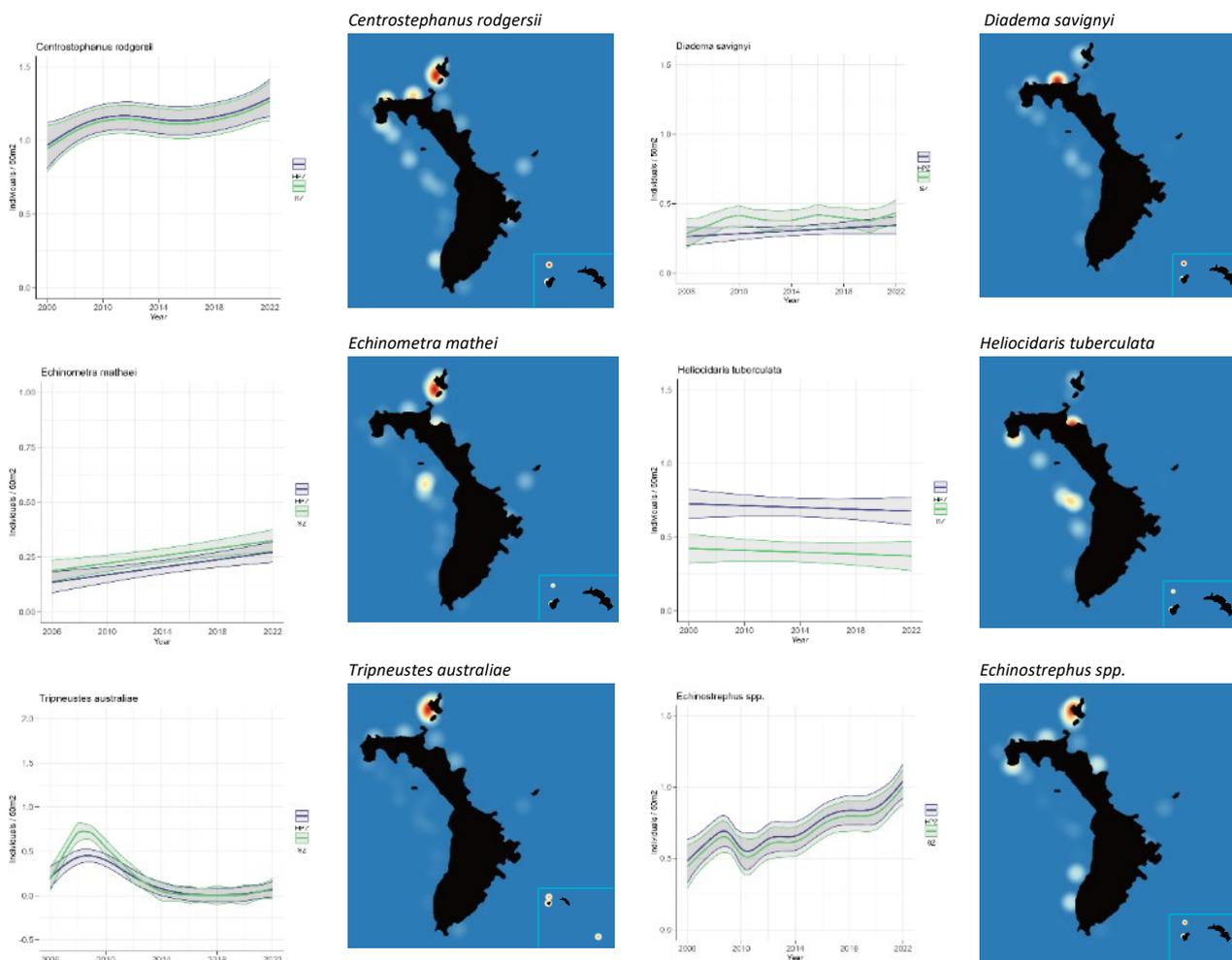


Figure 13. Trends in densities (per 50 m²) of individual sea urchin species through time from RLS surveys in HPZ sites (purple) and SZ sites (green). Densities have been log(x+1) transformed. Trend lines represent mean trends from Generalised Additive Mixed Models using ‘Site’ as a random effect. Confidence bands represent 95% confidence intervals. Heatmaps show hotspots of densities of different urchin species based on 2022 surveys.

Trends in sea urchin density were also examined in three areas individually (Figure 13) based on concerns about localised populations of urchins and/or grazing impacts including Neds Beach, Erscotts Hole, and the Algal Holes Ecological Community. Urchin densities were highly variable at Algal Holes and Erscotts Hole, with no overall trend through time. Previous reports have raised concerns about species-specific trends in sea urchins at the Algal Holes (Stuart-Smith et al. 2015, 2019) including fluctuations in Long-spined Sea Urchin (*Centrostephanus rodgersii*) and Purple Sea Urchin (*Heliocidaris tuberculata*) but when modelled over

the longer time series sea urchin populations do not show sustained increases at these sites. As previously noted (Stuart-Smith et al. 2015) the recorded densities of sea urchins at the Algal Holes are also influenced by transect placement, indicated by high within-year variability in the data (Figure 13). Previous reports have also raised concern about declines in macroalgae cover due to grazing impacts at the Algal Holes (Stuart-Smith et al. 2019) however, the longer time series shows only a slight decline (<10%) over the monitoring period. Continued monitoring of macroalgae cover and analysis of species-specific trends in sea urchin density are nonetheless important in this unique Ecological Community.

A large increase in urchin densities occurred at Neds Beach from 2014 to 2022 and was likely driven by Purple Sea Urchin (*Heliocidaris tuberculata*) being the most common species at this site. Previous analysis (Stuart-Smith et al. 2015) found large but short-lived increases in populations of Purple Sea Urchin at sites where greatest loss of coral cover occurred following the 2010 mass bleaching event. This included Neds Beach, which also experienced among the highest coral mortality during the 2024 bleaching event. It is therefore possible that coral bleaching and mortality is related to the continued increases of Purple Sea Urchins at Neds Beach. This population increase also corresponded with a ~12% decline in macroalgae cover overall, and an even steeper decline in the green algae *Caulerpa taxifolia* as described further below in *benthic assemblages* results. Further monitoring of this trend, and investigation of flow-on ecological impacts and potential drivers is recommended.

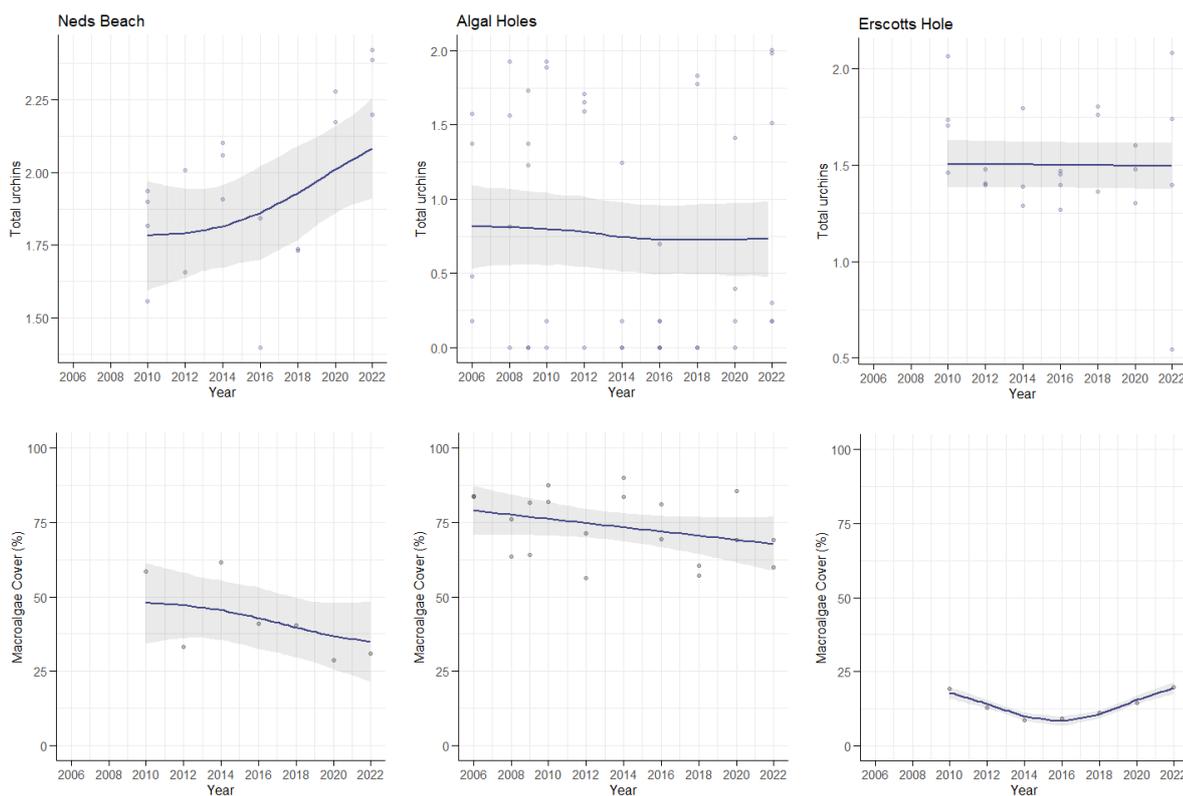


Figure 14. Trends in total sea urchin densities (per 50 m²) and the cover of benthic macroalgae from Neds beach, Algal Holes and Erscotts Hole sites. Urchin densities have been log(x+1) transformed and macroalgal cover represents % cover from photoquadrats. Individual points represent raw data from transects, and confidence bands represent 95% confidence intervals.

Trends in the density of Lamington Sea Urchin (*Tripneustes australiae*) and impacts on macroalgae cover were also specifically investigated in the two areas where population booms were documented as mentioned above, including sites around the Admiralty Islands and Balls Pyramid (Figure 14). As reported previously, densities of Lamington Sea Urchin around the Admiralty Islands were low in 2006 and peaked dramatically during a population boom in 2008, before decreasing gradually over the following six years and remained low from 2014 - 2022 (Figure 14). A corresponding rapid decrease in macroalgae cover occurred from 2006 to 2008 around the Admiralty Islands as described further below in *benthic assemblages* results. At one of the Admiralty Island sites (Ruperts Reef), elevated densities of Lamington Sea Urchin were noted again in 2022 preceding a more widespread population boom throughout the LHI Offshore Ecological Community in 2024, which will be analysed in future reports.

A population boom of Lamington Sea Urchins also occurred at sites around Balls Pyramid in 2022, continuing in to 2024, but benthic impacts of this event are not yet apparent in the analysed data. Further analysis will be included in future reports following full annotation of the benthic photoquadrats taken during 2024 surveys.

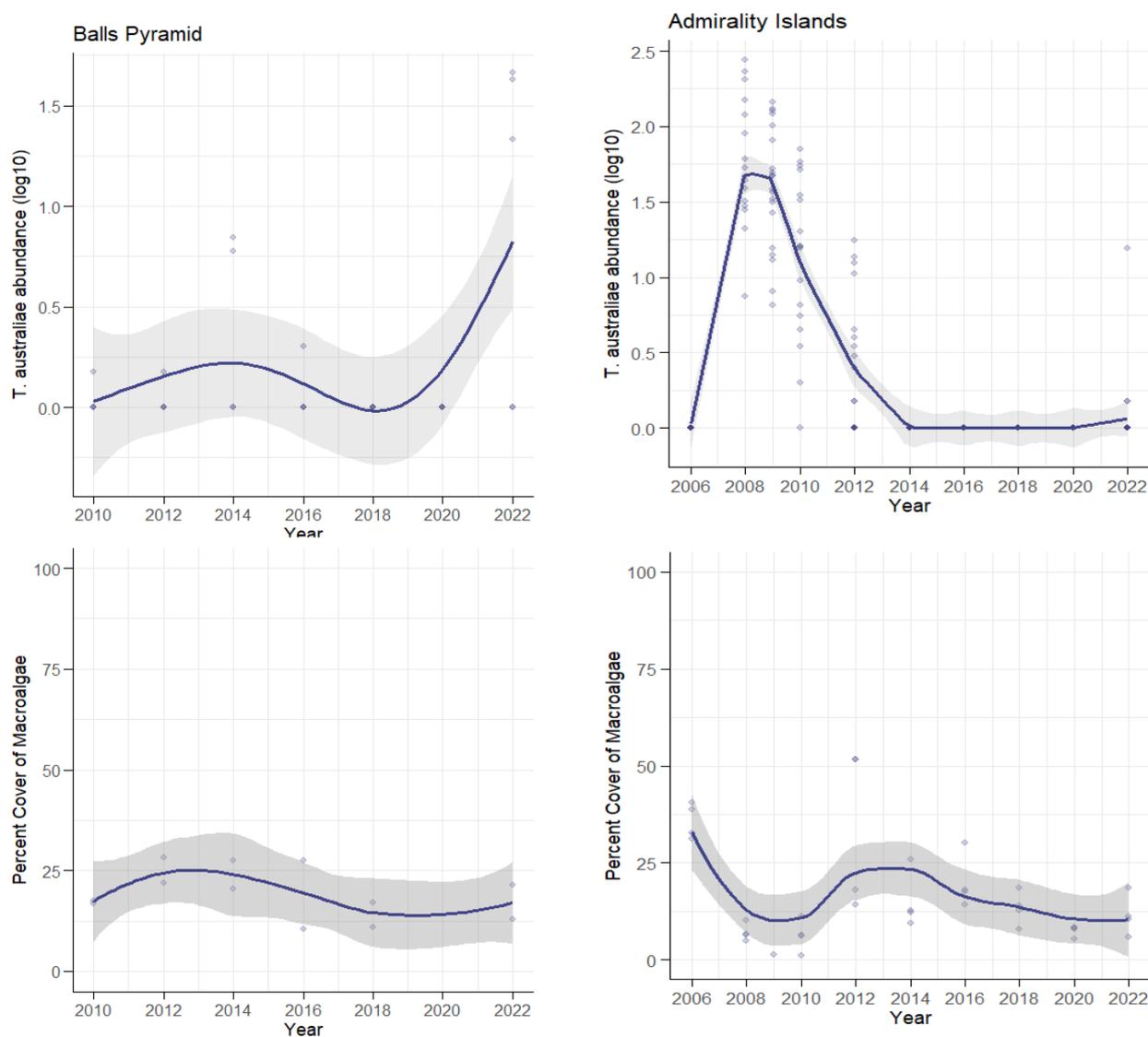


Figure 15. Trends in *Tripneustes australiae* densities (per 50 m²) and the cover of benthic macroalgae from Admiralty Islands and Balls Pyramid sites. Urchin densities have been log(x+1) transformed and macroalgal cover represents % cover from photoquadrats. Individual points represent raw data from transects, and confidence bands represent 95% confidence intervals.

CRYPTIC FISHES

The abundance and richness of cryptic fishes increased through time, following similar trajectories in SZs and HPZ (Figure 15). Surveys from 2006 and 2008 were not included in analysis given ‘cryptic’ fish species were not specifically recorded using the Method 2 surveys in these years. As for invertebrates, the increase in species richness of cryptic fish from 2010 to 2022 is likely to be partially driven by increased surveyor skill and experience in detecting and distinguishing more species of cryptic fish, with improved knowledge of their taxonomy supported by increased resources. The increasing trend in abundance may similarly reflect increased detection due to surveyor knowledge and experience, or may reflect an ecological trend similar to that observed elsewhere in the region following a series of warm years from 2016, including the 2018/19 marine heatwave (Holbrook et al. 2020). Ongoing monitoring and additional analysis may help identify the drivers of this trend.

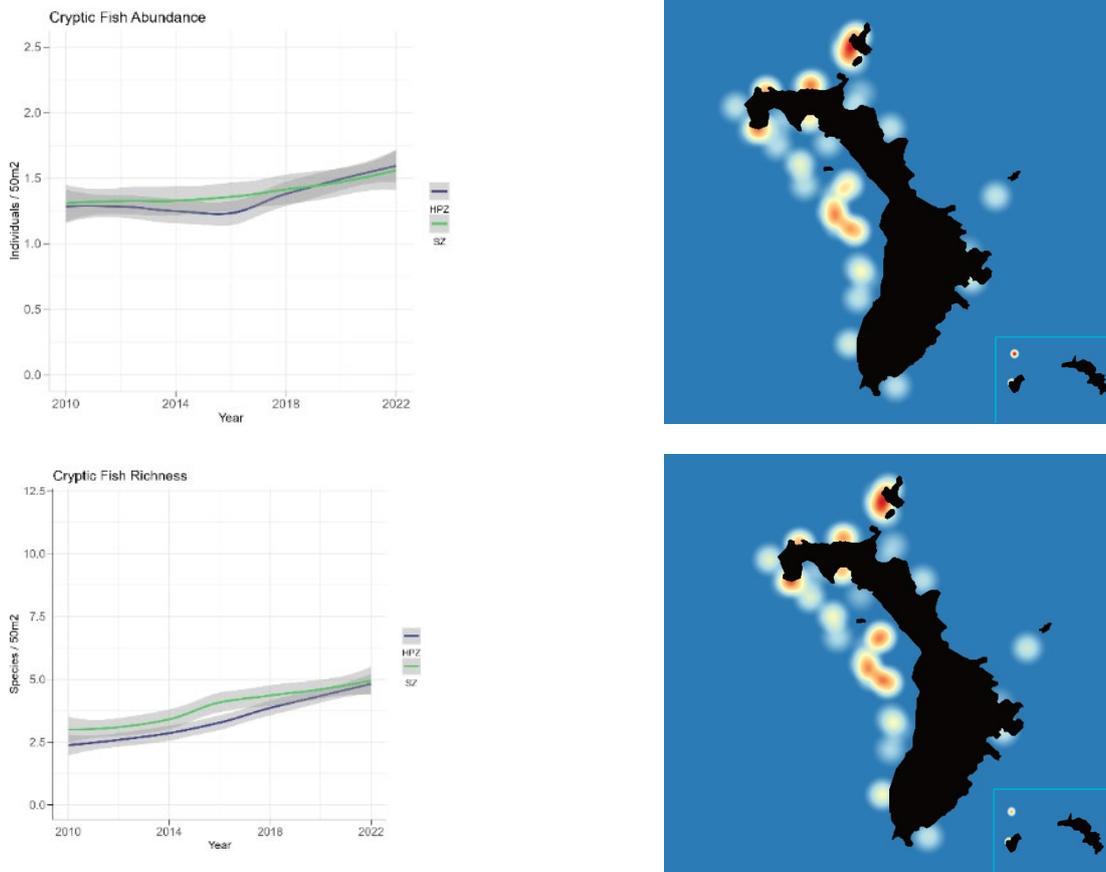


Figure 16. Trends in cryptic fish species abundance and richness (per 50 m²) through time from RLS surveys in HPZ sites (purple) and SZ sites (green). Trend lines are LOESS smoothers, rather than GAMMs, as used for other figures, and time-series start at 2010. Heatmaps showing hotspots of cryptic fish abundance and richness in the LHIMP in 2022.

BENTHIC ASSEMBLAGES

The composition of sessile benthic species is also distinct in each Ecological Community (Figure 16), and has previously been related to environmental conditions including wave exposure (Edgar et al. 2011). The Algal Holes Ecological Community is differentiated by higher cover of red foliose algae such as *Plocamium hamatum*, *Sarcodia ciliata*, *Euptilota articulata*, and *Pterocladia lucida*. Wave and current exposed sites in the LHI Offshore Ecological Community and Balls Pyramid Ecological Community have higher cover of crustose coralline algae, sponges, the coral *Isopora spp.*, and other encrusting and sub-massive coral species. The sheltered Lagoon Ecological Community have more species of branching corals, solitary anemones, the brown algae *Padina spp.*, coral rubble, and patches of sand and seagrass.

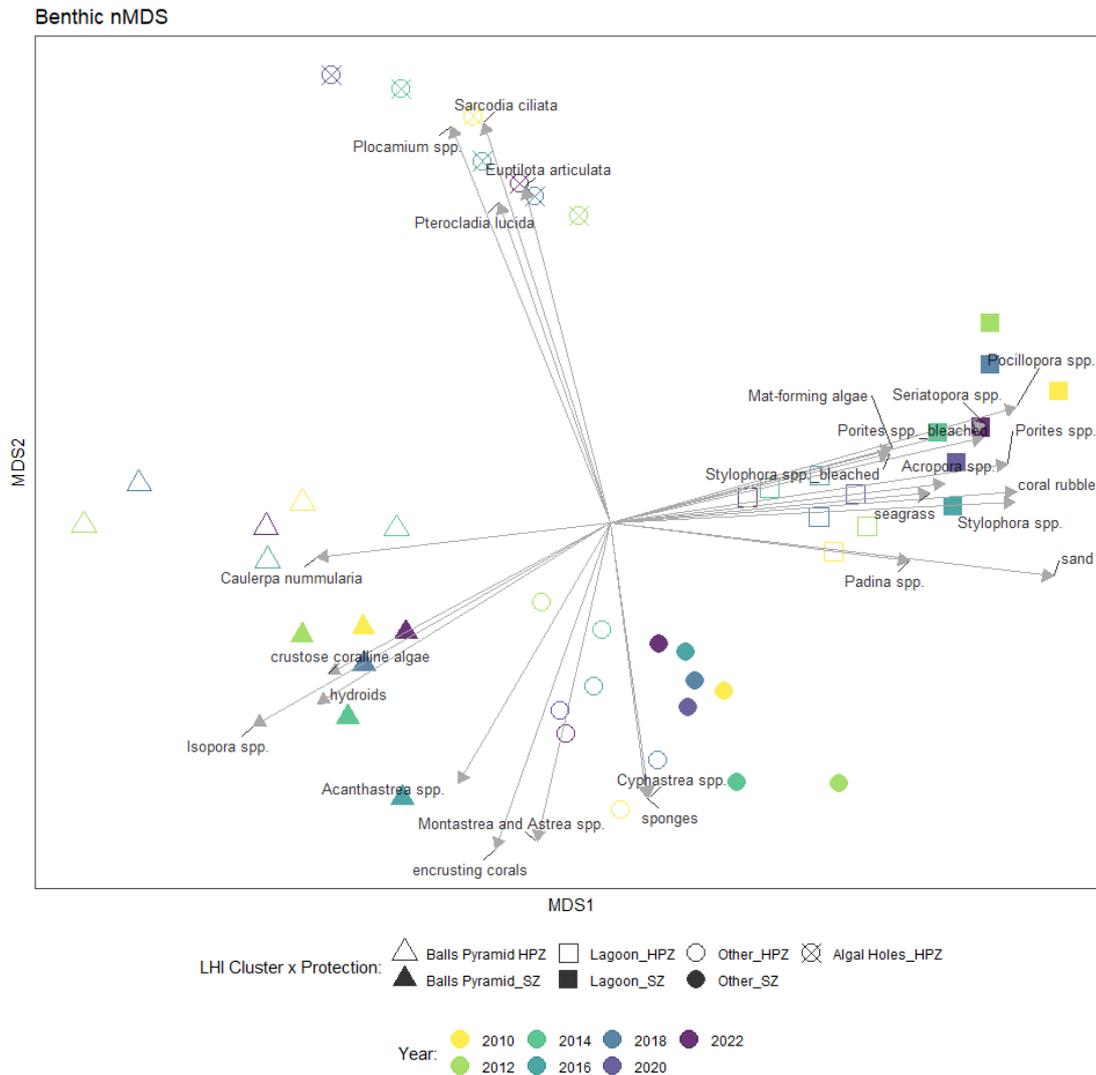


Figure 17. Non-metric multidimensional scaling (nMDS) plot of benthic assemblages scored from photoquadrats associated with RLS surveys in the LHIMP. Symbols represent the composition and percent cover of benthic taxa averaged among sites within each of the four major Ecological Communities previously identified (shown by different symbol shapes), and surveys in different years represented by a colour gradient from light (oldest surveys) to dark (latest surveys). Corals were scored to the highest taxonomic resolution possible, but shown at genus level for this plot (macroalgae are species level). Vectors represent taxa showing a correlation with axes >0.5.

Lagoon benthic assemblages

Overall, 43 distinct taxa of coral have been recorded in the Lagoon Ecological Community during RLS surveys (see Appendix 3). Within the Lagoon Ecological Community, benthic assemblage composition varied most according to site with some additional variation according to year (Figure 17). Sites toward the outer edges of the lagoon (Erscotts Hole, Stephen's Hole and Horseshoe Reef) which are exposed greater wave and current action were generally distinct from the more sheltered inner lagoon sites (North Bay, Comets Hole and Sylphs Hole). Erscotts Hole and Stephen's Hole had greater cover of the corals *Isopora* spp., soft corals and the green macroalgae *Caulerpa* spp., whilst Horseshoe Reef was characterized by different coral species and ascidians. North Bay was characterised by high cover of seagrass and the corals *Acropora* spp., while Comets Hole and Sylphs Hole were characterised by other sessile benthic species.

While assemblage composition remained fairly distinct between each individual lagoon site (Figures 19-24) the 2010 mass bleaching event in the lagoon resulted in further distinctions with four sites (Horseshoe Reef, North Bay, Comets Hole and Sylphs Hole) which were characterised by high cover of bleached hard coral in 2010. Further description of this bleaching event can be found in Edgar et al. (2011). There were no strong patterns of variation in other years, however, surveys were not conducted during the 2019 coral bleaching event when similar patterns may have also been apparent. As mentioned previously, data from 2024 was also collected during a bleaching event but has not yet been analysed.

There was most variation in composition at North Bay, much of which was driven by variation in cover of *Acropora* spp. coral. This forms extensive thickets at North Bay which were subject to severe bleaching and mortality in 2010 (Edgar et al. 2011). Extreme low tides due to prolonged negative sea surface anomalies also periodically result in aerial exposure of fast growing and shallow *Acropora* spp. coral thickets, often resulting in widespread mortality of large areas in North Bay as noted by Harrison and Carroll (2002) and also documented by LHIMP staff and LHI residents in other years. Being effectively a benthic 'monoculture' with few other species, differences in transect placement in relation to these thickets can also cause large variation in recorded benthic cover. Further variation in benthic assemblages was related to variable seagrass cover which may similarly be driven by differences in transect placement on or between reef patches.

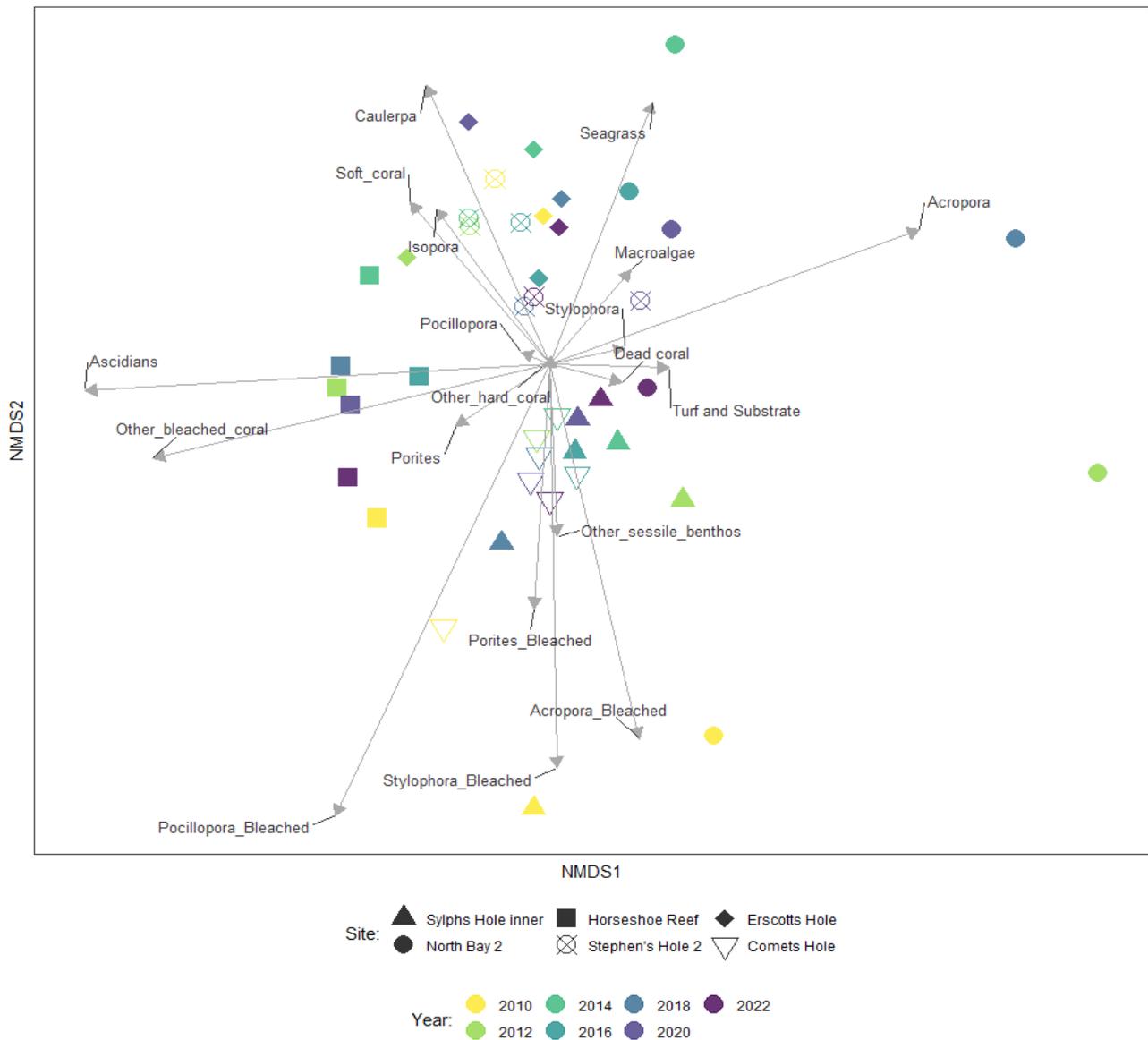


Figure 18. Non-metric multidimensional scaling (nMDS) plot of benthic assemblages at Lagoon sites only. Symbols represent individual sites within the lagoon, and surveys in different years represented by a colour gradient from light (oldest surveys) to dark (latest surveys).

Overall, the average total cover of live corals in the lagoon decreased from ~40% in 2006 to ~30% in 2022 (Figure 18), however there was large variation between sites and species. The highest occurrences of bleached coral were recorded in 2010 during a mass bleaching event (see Edgar et al. 2011), with bleaching also recorded at lower levels (<5%) in other years. Significant bleaching was also observed at sites both within and outside the lagoon in 2024 as described further below but has not yet been fully analysed. This data complements a targeted coral bleaching monitoring and response plan implemented by LHIMP management during the 2024 bleaching event.

Where bleaching was documented, the columnar *Porites heronensis* was the most commonly bleached coral recorded. A large proportion of these were noted by a coral taxonomist as a possible local variant of *Porites heronensis*, appearing discrete from colonies of that species found in other regions of Australia, and is particularly susceptible to bleaching at the growing edges (such as the top surface of columnar structures).

Pocillopora damicornis, *Stylophora pistillata* and species of *Acropora*, were found at most sites in the lagoon and were also recorded as bleaching extensively in 2010 (Edgar et al. 2011). Several other species of coral were recorded as bleaching, especially at Comets Hole, of which *Seriatopora caliendrum* was the most abundant and most frequently bleached. *Isopora spp.* corals in the lagoon did not experience bleaching on any transects.

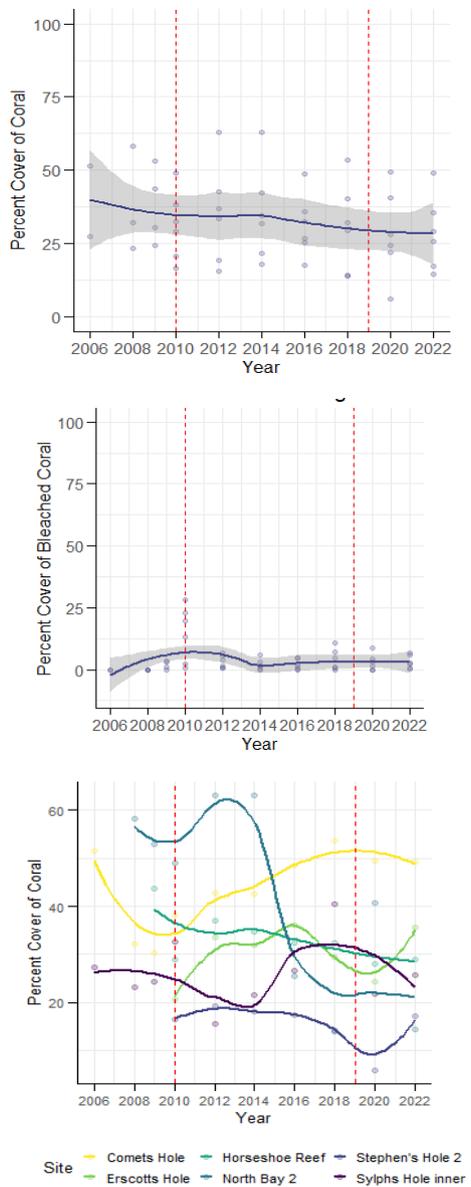


Figure 19. Temporal lagoon trends in the percentage of living hard corals (top) bleached corals (all hard corals bleached at the time of survey; middle) and all hard corals by individual site (bottom). Trend lines are from LOESS smoothers (+SE) fit to the raw data on average percent cover per site. Dashed vertical red lines represent the two major bleaching events in the lagoon during this time period (2010 and 2019).

Differences in levels of bleaching between sites is likely related to differences in species composition. Comets Hole and Sylphs Hole had the most similar assemblage composition (Figures 19 – 24), dominated by coral species susceptible to bleaching including *Porites heronensis*, *Pocillopora damicornis*, and some *Acropora spp.* and *Stylophora pistillata*. Comets Hole, however, also harboured more other diverse species of hard

coral and less macroalgae cover than Sylphs Hole, supporting more than double the cover of corals overall. Both sites experienced bleaching of *Porites heronensis*, *Pocillopora damicornis*, and *Stylophora pistillata* in 2010, with coral cover appearing to recover between the 2010 and 2019 bleaching events, after which a declining trend in coral cover was again detected. There was a large increase in macroalgae cover at Sylphs Hole following the 2019 bleaching event, however this was only slightly greater than macroalgae cover recorded in the earliest years of monitoring (2006 and 2008). Sustained increases in macroalgae cover were also recorded at North Bay, Stephens Hole, and Escotts Hole following the 2019 bleaching event but again in most cases to levels comparable or only slightly greater than in 2006 and/or 2008.

From 2008 to 2012 North Bay had the highest recorded coral cover of lagoon sites, characterised by very high cover of *Acropora spp.* coral (dominantly *Acropora yongei*, *Acropora abrotanoides*, *Acropora tortuosa* and *Acropora lovelli*), but from 2016 to 2022 had among the lowest recorded coral cover of lagoon site and benthic assemblages were characterized by higher cover of macroalgae. As mentioned previously, this may be attributed to impacts from coral bleaching in 2010, including documented bleaching of *Acropora spp.*, *Pocillopora damicornis*, *Stylophora pistillata* and *Porites heronensis* which had lower recorded benthic cover in subsequent years. It may also be driven by mortality from low tide aerial exposure events. Some of the high variation between years at this site may also be due to variable transect placement. Causes for the decline in coral cover documented at this site warrant further analysis and monitoring.

Horseshoe Reef had similar coral assemblages to Comets and Sylphs but higher cover of *Isopora spp.* There has been little compositional change in this assemblage since the 2010 bleaching event, other than the appearance of the green ascidian (*Diplosoma virens*) in 2010 which has persisted since. Anecdotally, this species is known to currently occur at other sites throughout the lagoon including extensive areas of dead *Acropora spp.* coral following bleaching and mortality at North Bay in 2010 (not represented in RLS data). Further investigations into its ecological role and characteristics are therefore warranted. Both Stephens and Escotts Holes experienced less bleaching than sites in the inner lagoon in 2010, although both experienced some reduction in overall coral cover in 2020, a year after the 2019 marine heatwave. Of the lagoon sites, Stephens Hole had the highest average cover of macroalgae, including *Caulerpa spp.* which was >25% cover in 2012 and 2014. Trends in *Caulerpa spp.* are analysed further below.

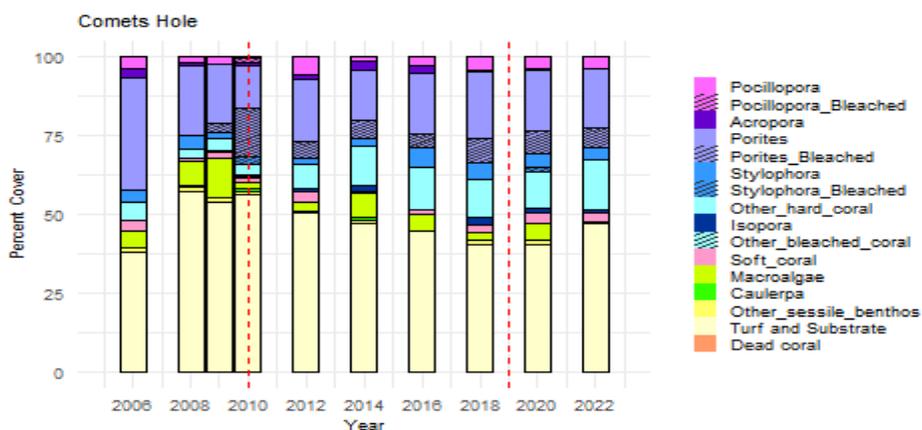


Figure 20. Changes in benthic composition through time at Comets Hole.

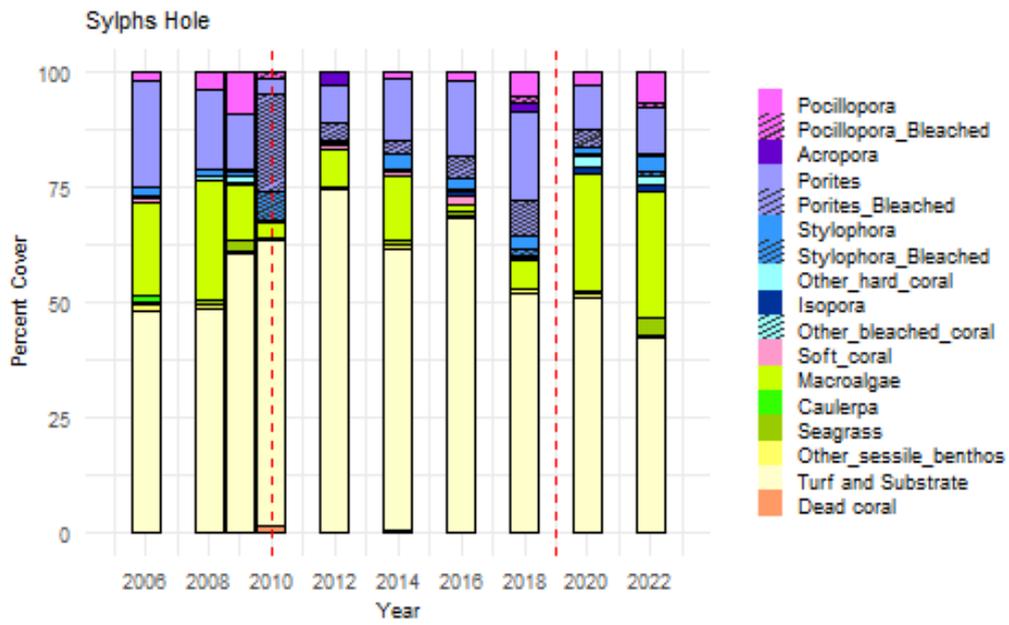


Figure 21. Changes in benthic composition through time at Sylphs Hole

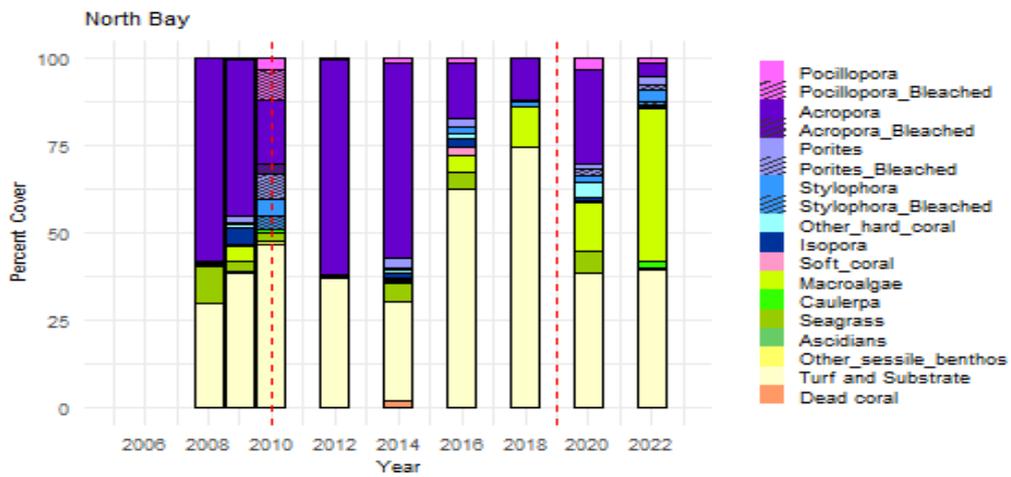


Figure 22. Changes in benthic composition through time at North Bay

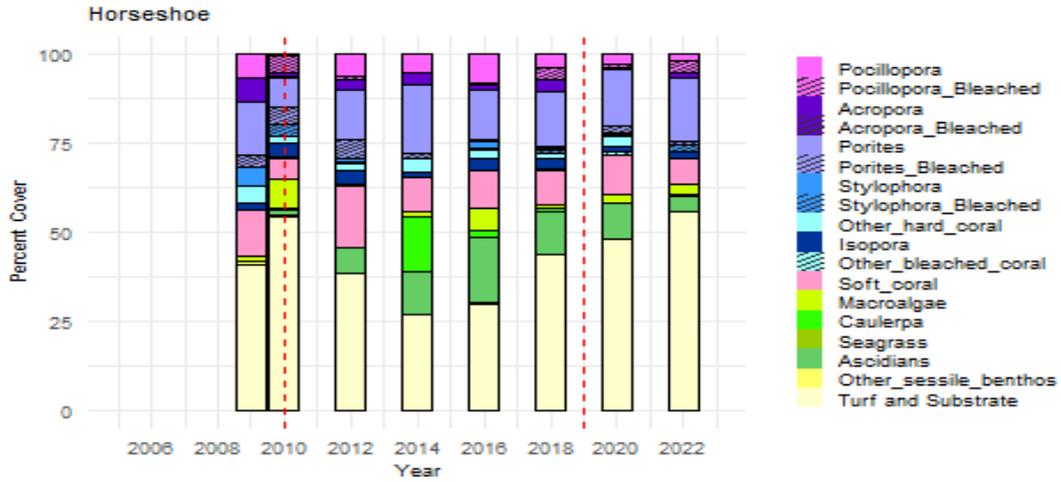


Figure 23. Changes in benthic composition through time at Horseshoe Reef

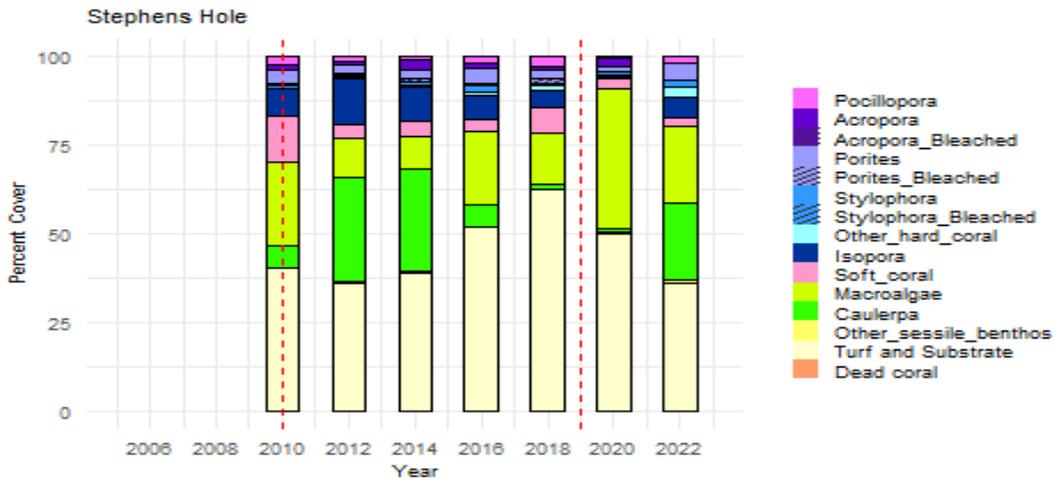


Figure 24. Changes in benthic composition through time at Stephens Hole

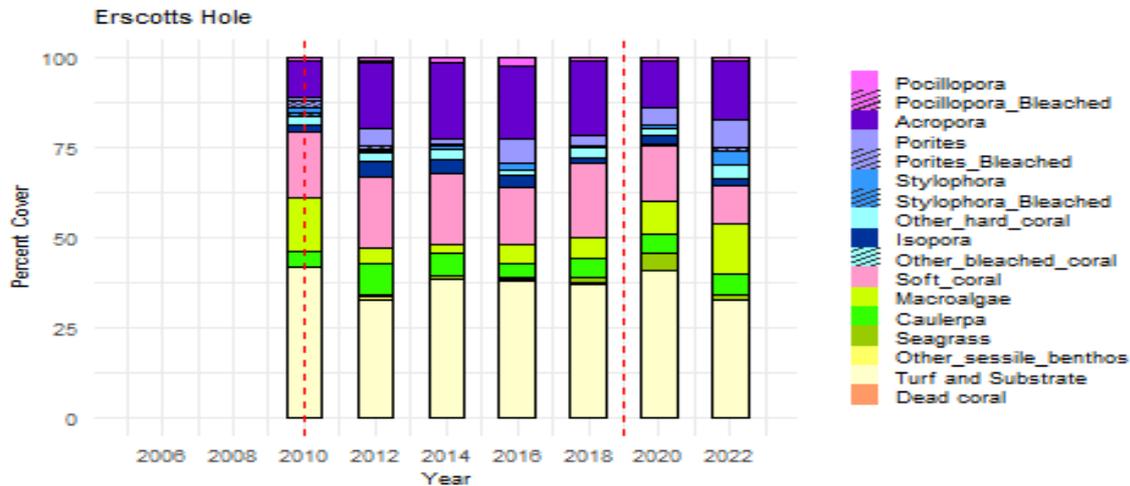


Figure 25. Changes in benthic composition through time at Ercotts Hole

Caulerpa spp. trends

Species of green algae in the genus *Caulerpa* readily propagates from fragments and are known for their ability to achieve very high coverage on reefs, with the potential to smother or outcompete other benthic species including coral. Being generally tropical species, climate change may also drive increasing abundances of these species (Stuart-Smith et al. 2019). Concerns have specifically been raised about increases in *Caulerpa spp.* at sites such as Neds Beach, the Algal Holes and lagoon sites (Stuart-Smith et al. 2015). Therefore, trends in cover of *Caulerpa spp.* were specifically investigated at these sites (including the lagoon site Ercott’s Hole) (Figure 25).

As previously noted, *Caulerpa taxifolia* is most likely native to the LHIMP but invasive strains have caused ecological impacts elsewhere. Large fluctuations in *Caulerpa taxifolia* at Algal Holes and in the Lagoon have previously been recorded (Stuart-Smith et al. 2015, 2019) and concern raised about a potential ecological shifts from coral to algae dominated reefs in relation to this species (Stuart-Smith et al. 2015). However, this trend has not been sustained and there has been little net change at these sites from 2006 to 2022.

Caulerpa taxifolia covered over half of the reef substrate at Neds Beach in 2010-2014 (52% in 2010 and 59% in 2014) but subsequent surveys show a substantial decline to low levels comparable to Ercotts Hole and Algal Holes. The decline in in cover since 2014 may have been driven by the localized increase in urchin densities (specifically the Purple Sea Urchin *Heliocidaris tuberculata*) as previously described. Overall, trends in this species did not clearly correlate with years with higher marine heat stress (i.e. coral bleaching events).

Caulerpa racemosa can spread rapidly across reefs forming sprawling mats and competing with other benthic species for light and space. It has previously been noted to have replaced coral cover at Sylphs Hole (Edgar et al., 2008) and show large increases in cover at Algal Holes (Stuart Smith et al. 2015). However, this species is also known to have large seasonal fluctuations, including mass annual die-backs after sexual reproduction and release of gametes (Clifton, 2013). The timing of these seasonal fluctuations relative to RLS survey campaigns may therefore confound documented trends for this species, although over the longer time series there were no large net changes in cover.

Coverages of two other *Caulerpa* species, *Caulerpa nummularia* (reported as synonym *Caulerpa peltata* in previous reports) and *Caulerpa chemnitzia* were too low and variable across LHIMP sites to distinguish

trends, although large increases in *Caulerpa nummularia* at Balls Pyramid have previously been reported (Stuart-Smith et al. 2015). These two species both have an often-sparse abundance of small dish shaped fronds that are also difficult to distinguish from images. As for other macroalgae species, the temporal dynamics of *Caulerpa spp.* and ecological drivers should be investigated further to better understand the baseline variability of these species and facilitate detection of important trends over and above this.

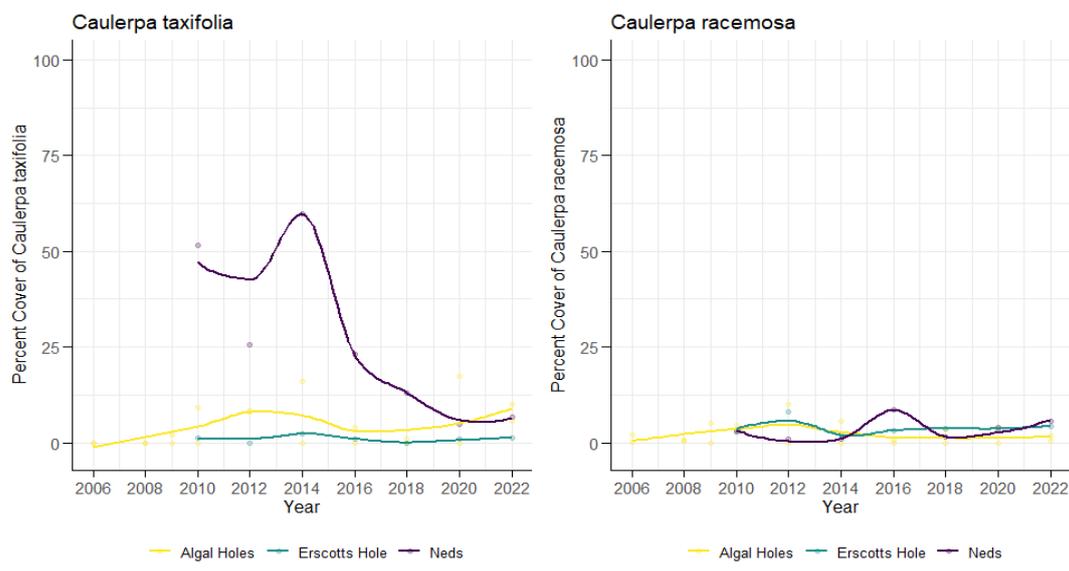


Figure 26. Percent cover of *Caulerpa* species at three key locations, Erscotts Hole (blue), Algal Holes (yellow) and Neds Beach (purple – shallow transects only) over time in the LHIMP.

Benthic impacts from sea urchin grazing

As described in more detail in Valentine et al. (2008) and Valentine and Edgar (2010) there was a large boom of Lamington Sea Urchins (*Tripneustes australiae*) around the Admiralty Islands in 2008, with high densities of sea urchins persisting to 2010 before rapidly declining. There was a large corresponding decrease in macroalgae cover due to grazing impacts, which raised concerns about the possibility of increasing threats to macroalgae species, long-term changes to benthic assemblages, and potential future flow on effects.

Trends in benthic assemblage composition are presented here for the four sites where highest recruitment densities of Lamington Sea Urchin were recorded during the 2008 population boom (Sugarloaf, Keyhole, Noddy, and Ruperts Reef) as well as two sites where moderate levels of recruitment were recorded (Malabar and Old Gulch) (Figures 26 – 31). Although recovery of macroalgae cover following the 2008 population boom has previously reported (Stuart-Smith et al. 2015) the longer time-series data with compositional detail not previously provided presents an opportunity to re-examine these trends and better understand potential trajectories of recovery or impact.

As described in previous reports, large declines in macroalgae were recorded across all impacted sites from 2006 to 2008, generally replaced by relatively bare areas characteristics of grazing impacts (i.e. the ‘turf and substrate’ benthic category in Figures 26 - 31). At several sites there was subsequently a large increase in macroalgae in 2012-2014 which was interpreted as recovery from grazing impacts (Stuart-Smith et al. 2015, 2019). In 2012 macroalgae cover was comparable or greater than in 2006 at two sites (Noddy Island and Ruperts Reef) representing over 35% of benthic cover (although differences in classification may influence the magnitude of increase documented in earlier reports as per Stuart-Smith et al. 2015). The drivers of these large increases are not known, however, these they were not sustained and decreased again to

relatively low levels in subsequent years, which did not correspond with sea urchin population booms or other known impacts.

The large increases in macroalgae cover in 2012-2014 may therefore represent fluctuations influenced by other ecological drivers rather than a sustained recovery from grazing impacts. For example, marine heat stress which resulted in a coral bleaching event in 2010 may have driven rapid macroalgae growth given macroalgae increases were also documented at some affected lagoon sites following bleaching events (described above). Other possible ecological drivers of macroalgae fluctuations on offshore reefs include changes in water chemistry, temperature, nutrient levels, or other oceanographic influences linked to movements of the East Australian Current.

While the lack of sustained increase to levels of macroalgae recorded in 2006 has previously been interpreted as signs of long-term impact from sea urchin grazing (Stuart-Smith et al. 2015, 2019) trends over the longer time series indicate this may not necessarily be the case. Despite densities of Lamington Sea Urchins remaining low for almost a decade, data from 2014 to 2022 show that macroalgae cover has generally stabilised at much lower levels across most impacted sites, commonly representing between 5 – 15% of benthic cover. This may therefore represent the baseline ecological state for these reefs, and the high levels of macroalgae cover recorded in 2006 prior to the population boom in Lamington Sea Urchins may have represented a fluctuation peak similar to 2012-2014.

No impacts on coral were detected as a result of the 2008 Lamington Sea Urchin population boom (Valentine and Edgar 2010) and there have been no consistent or large trends in coral cover at these sites throughout the monitoring period. Macroalgae with large fronds (in contrast to turfing algae) can compete with coral for space and light and cause physical disturbance to coral (Ainsworth et al. 2021). Reductions in macroalgae from grazing impacts may therefore be expected to benefit coral growth, and increases in soft coral (and to a lesser extent hard coral) were subsequently suggested in replacement of macroalgae at sites impacted by sea urchin grazing (Stuart-Smith et al. 2015). However consistent or sustained changes in cover of soft or hard corals have not been documented across these sites. Increases in crustose coralline algae (CCA) were also documented following grazing impacts in 2008 (Valentine and Edgar 2010) which can induce and support settlement of coral and other invertebrate larvae. However, again no corresponding increases in coral cover or changes in invertebrate assemblages were documented. No other immediate or long-term flow-on effects to fish (including herbivores) or invertebrates were detected as a result of the 2008 Lamington Sea Urchin population boom, with impacted sites currently supporting high species richness for fish and invertebrates, as well as hotspots for fish biomass as described above.

There is also likelihood that ecological drivers of sea urchin population density affect several species simultaneously, and result in compounding grazing impacts. For example, increases in bare areas of reef (i.e. the 'turf and substrate' benthic category) following the Lamington Sea Urchin Population boom may have also be driven by grazing impacts of Long-spined Sea Urchin (*Centrostephanus rodgersii*). Increases in bare reef occurred at many sites from 2006 to 2010, and 2014 to 2022, coinciding with periods of population increase in Long-spined Sea Urchin. The more sustained ecological changes caused by grazing impacts of this species may be obscured by large fluctuations in benthic cover associated with booms of Lamington Sea Urchin and other ecological drivers as described above. As noted by (Valentine and Edgar 2010), population dynamics of Lamington Sea Urchin and Long-spined Sea Urchin are likely to be correlated given they may have synchronised responses to environmental conditions favourable to sea urchin recruitment. Analysis in Stuart-Smith et al. (2019) also indicated that trends in Purple Sea Urchin (*Heliocidaris tuberculata*) mirrored those of Long-spined Sea Urchin, with corresponding increases from 2006 to 2010 and 2014 to 2018. To a lesser extent, similar trends were also documented for Burrowing Sea Urchin (*Echinometra mathei*), Needle Spine Sea Urchin (*Echinostrephus spp.*) and Savigny's Longspine Sea urchin (*Diadema savignyi*) (Stuart-Smith et al. 2019) although these trends are less apparent in the models presented over the longer time series in this report.

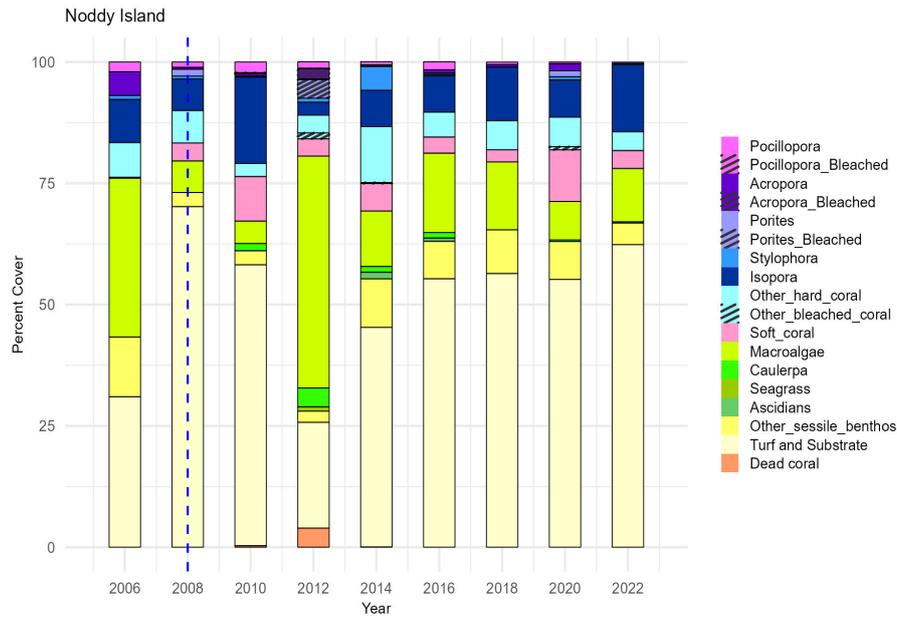


Figure 27 Changes in benthic composition through time at Noddy Island. Peak *Tripneustes australiae* urchin density at this site was 433 per 100 m².

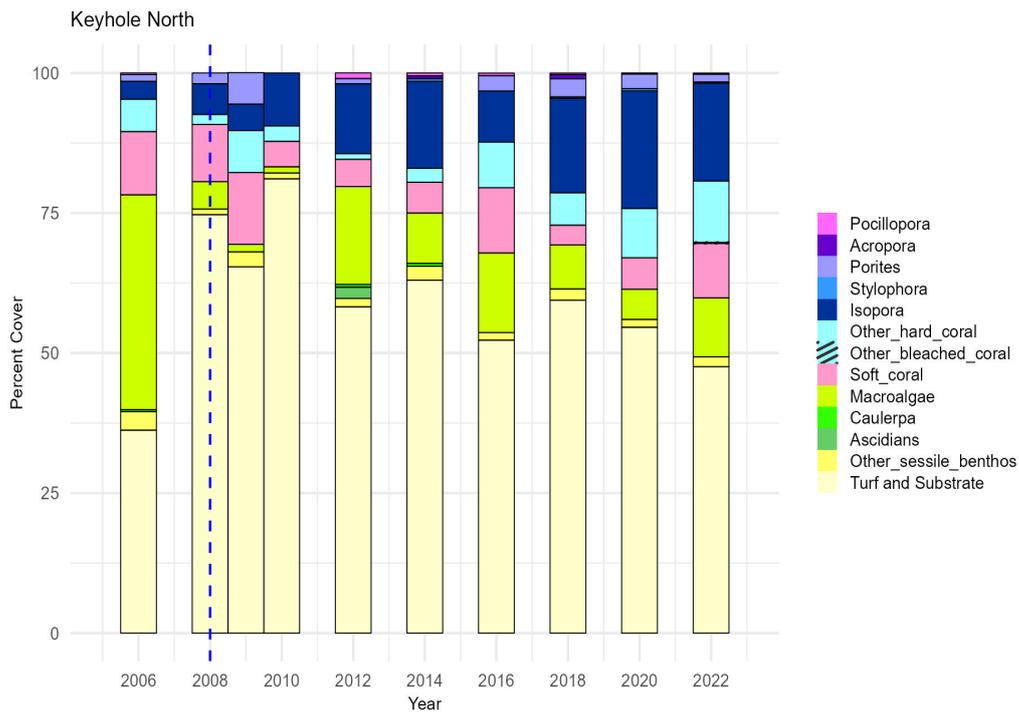


Figure 28 Changes in benthic composition through time at Keyhole North. Peak *Tripneustes australiae* urchin density at this site was 268 per 100 m².

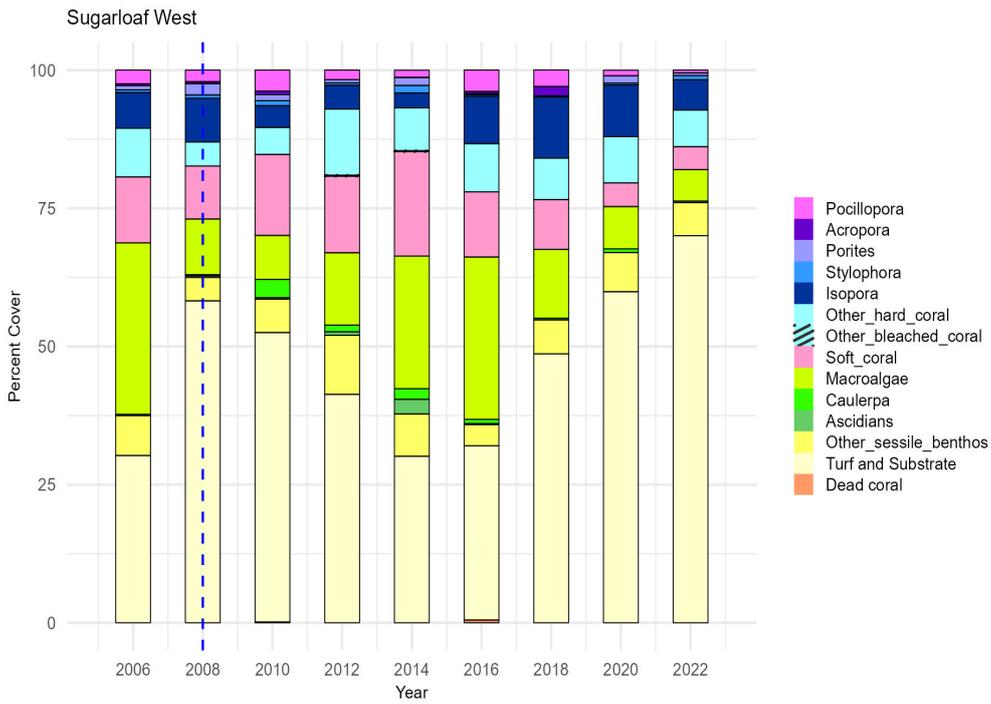


Figure 29 Changes in benthic composition through time at Sugarloaf West. Peak *Tripneustes australiae* urchin density at this site was 329 per 100 m².

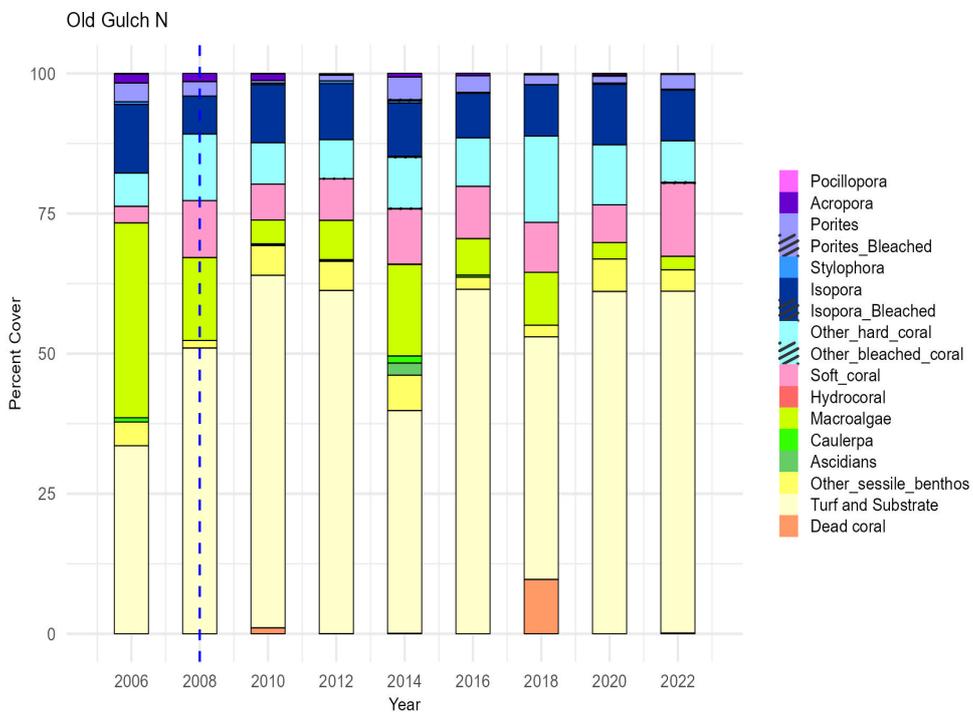


Figure 30 Changes in benthic composition through time at Old Gulch N. Peak *Tripneustes australiae* urchin density at this site was 69 per 100 m².

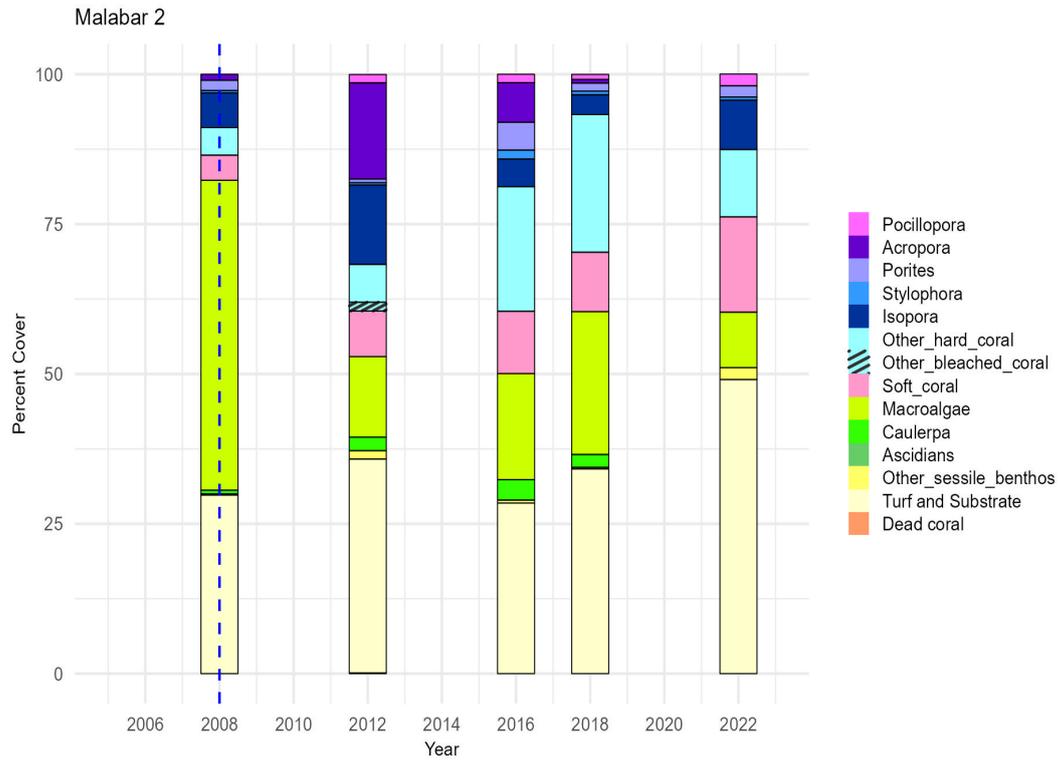


Figure 31 Changes in benthic composition through time at Malabar 2. Peak *Tripneustes australiae* urchin density at this site was 88 per 100 m².

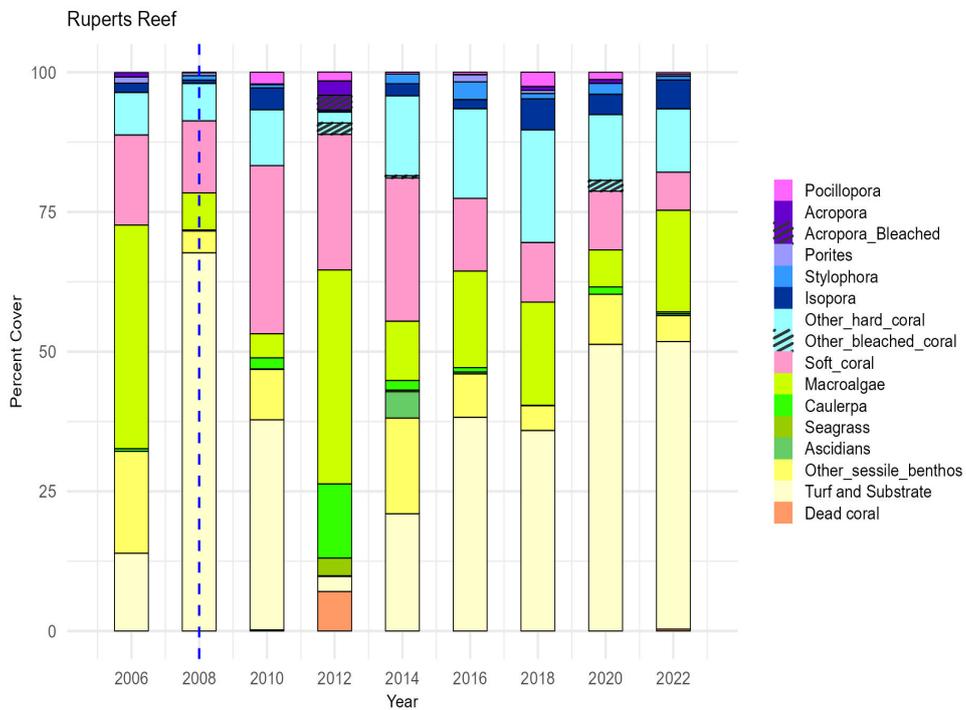


Figure 32 Changes in benthic composition through time at Ruperts Reef. Peak *Tripneustes australiae* urchin density at this site was 148 per 100 m².

4 Discussion

Shallow reefs in the LHIMP support many unique and important environmental, social and economic values as highlighted in recent reports (Harasti et al. 2022, Heller 2024). Overall, results from the long-term monitoring presented here indicate that these reefs have remained in generally similar ecological condition since 2006. Nonetheless, this report builds on previous findings in regard to ongoing threats including climate change, fishing pressure, and sea urchin grazing impacts which warrant continued monitoring and additional research. It also highlights further opportunities for management and research including increasing awareness of the values supported by SZs, improving understanding of macroalgae dynamics, investigating ecological drivers of the distinct LHIMP shallow reef Ecological Communities, and assessing patterns and trends in a broader spatial and temporal context.

CLIMATE CHANGE IMPACTS

As noted in the introduction, potential impacts to shallow reefs in the LHIMP from climate change include coral bleaching and associated flow-on effects, overall changes to fish and invertebrate assemblage structures, and changes in macro-algae assemblages. Only the former has been clearly documented in the LHIMP so far, but others may occur as climate change effects escalate.

Coral bleaching and mortality impacts have so far been largely isolated to shallow reefs in the Lagoon Ecological Community, which are subject to greater marine heat stress from additional solar heating and reduced flushing of warm water. Recovery has occurred following previous bleaching events (including in 2010 and 2019 as captured in this report) however there has also been a sustained decline in live coral cover on lagoon reef sites of approximately 10% from 2006 to 2022. While currently representing a relatively small proportion of total benthic cover, this indicates there is a high risk of further cumulative loss of coral cover as bleaching events become more frequent and severe with climate change. This represents a severe risk to the environmental, social, and economic values these lagoon reefs support.

Coral bleaching and mortality can also result in overall changes to benthic assemblages, such as shifts in coral composition favouring stress-tolerant species or increases in non-coral species. These kinds of changes have not yet been detected in response to coral bleaching events, other than increased cover of macroalgae at some sites in the lagoon following the 2019 bleaching event which warrant continued monitoring as discussed further below. Conversely, climate change also presents a high risk of declines in many macroalgae species in the LHIMP through heat stress impacts on subtropical, temperate and endemic species with lower thermal tolerances. This present a particular threat to the globally unique Algal Holes Ecological Community, therefore minimising additionally stressors to this community where possible is a key management opportunity.

Impacts of climate change to reef fishes in the LHIMP appear so far to be limited to flow-on effects to a few species which rely directly on corals or anemones (which have similar thermal bleaching responses). Sustained declines in corallivores and the endemic McCulloch's anemonefish have been detected and are likely driven by declines in coral and anemones from bleaching and mortality on shallow reefs in the lagoon. Further monitoring and analysis of bleaching impacts and flow-on effects is critical, with additional targeted monitoring of McCulloch's anemonefish and recovery actions are being investigated with support from the LHIMP.

Changes in fish assemblages can also occur as a more general response to climate change, driven by changes in the recruitment and survivorship of different species with warming waters and strengthening of the East Australian Current (known as 'topicalization'). No such changes have been detected in the LHIMP as indicated by relatively stable RFTI values, unlike locations on the mainland coasts of Australia where assemblage structure appears to be more dynamic between warm and cool years (Day et al. 2018). As previously noted, (Stuart-Smith et al. 2015), the geographical isolation of reefs in the LHIMP means there is very limited opportunity for adult immigration for most reef species, or recruitment of new species with short larval stages. Only long-distance larval dispersal and recruitment events may add new species to the assemblage. The observed stability in RFTI in the LHIMP is probably in large part due to these effects of isolation and may possibly also be the reason for the high stability in fish assemblage structure, although this remains to be tested. However, as for benthic and invertebrate species, the lower connectivity and fewer opportunities for recruitment mean species and assemblages in the LHIMP are more vulnerable to local extinction from any impacts.

In the face of escalating climate change impacts, it is especially important to understand how other stressors interact and can be managed to maximise reef resilience. For example, coral bleaching and mortality can be exacerbated by stressors such as high nutrient input, sedimentation and turbidity (Edgar et al. 2008, Davis 2022b). This is particularly important for shallow reefs in the Lagoon Ecological Community which are subject to greatest stress from almost all identified impacts including climate change and coral bleaching, nutrient input, sedimentation, pollution, vessel activities and development.

2024 mass bleaching event

A mass bleaching event occurred in 2024, during which Sea Surface Temperatures surrounding LHI reached the highest magnitude of Degree Heating Weeks (DHW) since 2010 and record high temperatures were recorded in the lagoon on the array of loggers managed by the LHIMP. Impacts of the 2024 bleaching event were recorded during RLS surveys and also through separate targeted and repeated monitoring which included monthly in-situ and aerial surveillance throughout the lagoon. Data from RLS surveys complemented this monitoring by documenting the first occurrence of widespread bleaching on offshore reefs (see below) which makes an important contribution to our understanding of this and future events.

While not yet formally analysed, RLS surveys from 2024 recorded bleached and stressed corals at sites both inside and outside the lagoon. Outside the lagoon, encrusting corals such as *Montipora* sp. And *Porites* sp. were observed bleached across all depths, including up to 33 m at dive sites near the Admiralty Islands, while branching corals such as *Pocillopora* sp. appeared most affected in the lagoon and at shallower depths. Visible signs of stress and bleaching were observed at most sites in the lagoon. Recent mortality was also observed at sites inside the lagoon during surveys, but results from ongoing monitoring will provide more insight into the extent of coral mortality and associated effects.



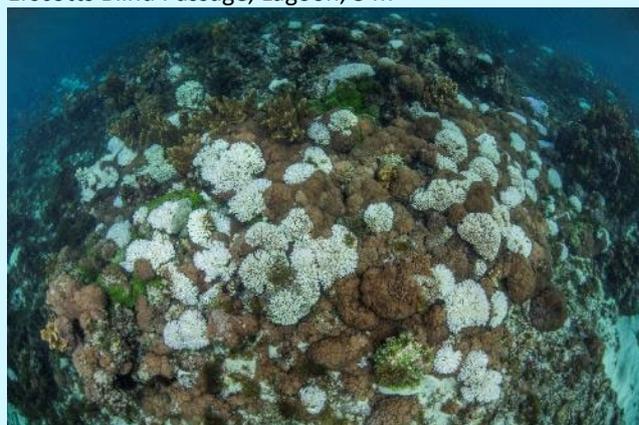
Sylphs Hole, Lagoon, 2 m



Erscotts Blind Passage, Lagoon, 3 m



Middle Beach, 15 m



Neds Beach, 6 m

Plate 1 Image plate highlighting observed stressed and bleached corals in the Lord Howe Island Marine Park from 2024 surveys.

FISHING IMPACTS

While in previous reports some trends in fish biomass were detected and attributed to potential impacts of fishing pressure, over the long time series there has been no detectable accumulation of fish biomass in SZs in response to reduced fishing pressure compared with HPZ. Data from 988 surveys from 2006 to 2022 across a comprehensive set of sites around the LHIMP indicated no significant difference or divergence in the Large Reef Fish Indicator (i.e. the biomass of large fishes counted on transects), or total fish biomass, between HPZ and SZs. Reasons for this may include: low fishing pressure in HPZ; continued (illegal) fishing pressure in SZs; fish movements across zones preventing accumulation of biomass within SZs; slow accumulation of biomass and a delayed response to reduced fishing pressure; or other sources of variation obscuring trends.

As noted since earliest ecological reports there is relatively low fishing pressure in the LHIMP compared with many other coastal areas, and this is likely to make fishing impacts on biomass more difficult to detect (Edgar et al., 2008, 2011). Impacts on biomass may also be less evident in the shallow reef monitoring data given much of the fishing pressure in the LHIMP occurs on deeper reefs offshore. Although long term BRUV monitoring of deeper reefs have also detected few effects of differences in fishing pressure, and similarly attributed this to low fishing pressure in HPZ (Rees *et al.*, 2021). Fishing effort and impacts throughout the LHIMP and surrounding waters are also considered further through commercial charter catch data collection and reporting (Figuera and Harianto 2022).

It has also been noted in previous reports that continued (illegal) fishing activity in SZs may prevent fish biomass from accumulating to detectable levels. There has been strong community support for SZs in the LHIMP as indicated by results of an attitudinal survey (NSW MPA 2010) and the fact that informal SZs were established and managed by the community for over 40 years prior to formal establishment of the LHIMP zoning plan (noted in Edgar et al. 2008). However, some illegal fishing in SZs is still reported, and fish biomass and the effectiveness of SZs can be compromised by even small levels of fishing pressure (Jennings and Polunin 1996, Edgar et al. 2014). Increased education and compliance capacity are key management opportunities previously suggested to address this (Stuart-Smith et al. 2015, 2019).

As noted by Edgar et al. (2011) the movement of fish across different zones in the LHIMP may also prevent detectable accumulation of biomass in SZs. Many target species of fish are highly mobile and have low site fidelity, including Silver Trevally (*Pseudocaranx* sp.) and Kingfish (*Seriola lalandi*) so reduced fishing pressure in SZs may not increase biomass in these same areas. Current research is underway to identify the life history and movement patterns of Kingfish in the LHIMP and throughout the range of this species, to better inform stock trends and management. As mentioned above fish biomass throughout the LHIMP is also monitored through charter fisher catch data and reporting.

Accumulation of biomass in response to reduced fishing pressure may also occur over a longer time period than the current monitoring data shows. Edgar et al (2008, 2011) noted that some ecological effects from reduced fishing pressure in SZs can take more than two decades to become detectable. The isolation and low levels of recruitment for local fish populations (discussed above in *climate change impacts*) may further delay these ecological effects in the LHIMP. Lastly, it is possible that other sources of variation in the data are obscuring trends related to fishing pressure. For example, different Ecological Communities support different levels of fishing pressure and different target species, so biomass trends may vary between them. Reduction in fishing pressure has also not been uniform or synchronised in LHIMP SZs, given some areas were more heavily fished prior to implementation of the LHIMP zoning plan (see Edgar et al. 2011) and others were already informal SZs established by the community for several decades. Further analysis might therefore reveal accumulation of biomass within specific sites, SZs or Ecological Communities.

Despite the lack of detectable accumulation of biomass in SZs, they still provide important refuges for species likely to be disproportionately impacted by even low fishing pressure. These include Black Rockcod, Bluefish

and Doubleheader for which SZs are likely to help sustain local populations. Black Rockcod are a protected species in NSW waters however have low survival rates from accidental bycatch so SZs provide important refuges from these fishing impacts. This includes lagoon reefs which are key nursery areas for juveniles but where bycatch of juveniles is also reported. The low number of Black Rockcod recorded in RLS data concur with the low population densities of these species recorded on shallow reefs during targeted monitoring (Harasti et al. 2022) and recorded on deeper reefs in long-term BRUV data (Rees et al. 2021). The low population density of this species in the LHIMP has been raised as a key concern relating to fishing impacts (Harasti et al. 2022). Along with maintenance of SZs, research and education around methods to increase bycatch survival are management opportunities to address threats to this species.

Bluefish are also vulnerable to exploitation in the LHIMP, due to their restricted geographic distributions and life history traits (Lewis 2012). Previous declines in this species were detected in RLS data and concerns raised about potential impacts of fishing pressure (Edgar et al. 2011, Stuart-Smith et al. 2019). Declines have also previously been detected on deeper reefs in BRUV data (Reeds et al. 2021) and charter fishing catch data (Figueira and Harianto, 2022). However, the longer time series of RLS data presented here indicates there has been no net decline in Bluefish biomass on shallow reefs from 2006 to 2022. Regardless, continued monitoring of this species is important given its vulnerability to impacts, high value as a recreational target species, and potential ecological flow-on effects from loss of herbivore biomass. As noted in previous reports, additional targeted research to better understand population dynamics of this species is recommended (Stuart-Smith et al. 2019). Maintenance of existing SZs and consideration of additional protection of macroalgae dominated reefs which support herbivore biomass (such as the Algal Holes Ecological Community) are also management opportunities to support local populations of this species into the future.

Doubleheader are also particularly susceptible to fishing pressure, due to their restricted geographic distribution, slow growth rate, the small proportion of large mature individuals in their population, and ease of capture by line-fishing. Similar concerns have been raised for this species about fishing impacts driving declines in biomass (Stuart-Smith et al. 2019). However, as for Bluefish, the longer time series indicates no net decline on shallow reefs from 2006 to 2022. Charter fishing catch data also do not indicate clear trends with regards to impacts on Doubleheader, however future increases in fishing effort in the lagoon may disproportionately affect this species (Figueira and Harianto 2022). Maintenance of SZs, continued education around the conservation significance of this species, and further research in to its life history and movement patterns are nonetheless recommended as strategies for its conservation (as per Stuart-Smith et al. 2019).

SEA URCHIN GRAZING IMPACTS

Grazing impacts from increases in sea urchin populations present another ongoing threat to shallow reefs in the LHIMP. In particular there is risk of decline or local extinctions of macroalgae (including endemic species and the unique Algal Holes Ecological Community) with potential ecological flow-on effects to herbivorous fish (such as Bluefish) and invertebrates (such as the endemic sea snail *Turbo cepoides*). They also impact commercial and recreational values of sites for SCUBA diving and snorkelling through reduced aesthetic quality (as noted by Valentine and Edgar 2010, and also reported during recent sea urchin population booms). Grazing impacts in the LHIMP are caused by population booms of Lamington Sea Urchin (*Tripneustes australiae*) as well as sustained increases in other grazing and barren-forming species such as Long-spined Sea Urchin (*Centrostephanus rodgersii*) and Purple Sea Urchin (*Heliocidarus tuberculata*). These species all have regionally endemic subtropical to temperate distributions, extending from the NSW coastline of mainland Australia to northern New Zealand and including remote islands in the region (Gall 2016, Byrne et al. 2022). Importantly, more targeted research is required to understand their current and ongoing impacts in the LHIMP, particularly with regards to climate change and future population trends.

Tripneustes spp. populations are known to boom and bust due to synchronised recruitment events, fast growth rates, and subsequent mass mortality (Valentine and Edgar 2010). As outlined earlier in the report a population boom of Lamington Sea Urchin (*Tripneustes australiae*) in 2008 caused large declines in macroalgae cover at impacted sites. Despite ongoing monitoring of these sites, trends in long-term impact or recovery are difficult to infer due to other potential sources of variability in macroalgae cover, and uncertainty around the baseline levels or temporal dynamics of macroalgae on these reefs (discussed further below). Population booms of Lamington Sea Urchin have also been recorded during RLS surveys around Balls Pyramid in 2022 and widespread offshore reefs surrounding LHI in 2024. Data from 2024 relating to sea urchin density and grazing impacts from these events will be included in future reports, but in the interim will also contribute to targeted research on this species both in the LHIMP and throughout its range to better understand its ecological impacts and population trends. While it is likely that the species will ‘bust’ and return to low densities in coming years, population booms nonetheless present a high risk of rapid impacts on benthic communities.

The drivers of these population booms are poorly understood, and the effect of climate change on this species difficult to predict (Bronstein et al. 2019). Previously the species was believed to be *Tripneustes gratilla* which has a tropical distribution and therefore potential for increasing impacts with climate change (Valentine and Edgar 2010). For example, warming water and strengthening of the East Australian Current could be expected to increase larval supply and fecundity in the LHIMP, resulting in increasing frequency and severity of population booms (Edgar et al. 2008, Valentine et al. 2008). It is now known, however, that the species is *Tripneustes australiae* which has a regionally endemic subtropical and temperate distribution (McLaren et al. 2023). It is therefore less clear how it may respond to climate change, although it is likely to be vulnerable itself to climate change impacts due to its restricted geographical range and competitive exclusion or hybridized by expanding populations of *Tripneustes gratilla* (Bronstein et al. 2019). To try and address this knowledge gap, population genetic analysis is also underway to better understand the regional connectivity of this species, identify the larval sources which cause population booms in the LHIMP, and therefore predict how these population booms may be influenced by climate change and strengthening East Australian Current.

Unlike the boom and bust nature of the Lamington Sea Urchin densities, Long-spined Sea Urchin (*Centrostephanus rodgersii*) have shown sustained increases in the LHIMP across a larger number of sites during the 17 years of monitoring. This species has been noted to form ‘barrens’ since earliest ecological monitoring (Edgar et al., 2008) which are areas where grazing impacts preclude many sessile benthic species including macroalgae, sponges, ascidians and soft coral (Andrew and Underwood 1989, Johnson et al. 2005, Ling 2008). These barrens are also known, though, to support unique and diverse assemblages of invertebrates and fish which depend on them as part of a natural mosaic of habitats (Davis et al. 2023, Kingston & Byrne 2023). The southern range extension of this species due to climate change has resulted in significant declines and local extinctions of kelp forests and macroalgae in Tasmania, however there are also indications from RLS data that its populations are declining in the northern extent of its range (Davis et al. 2023). As noted by Davis et al. (2023) and Kingston & Byrne (2023), impacts of this species and management responses should therefore be considered separately within each region and not rely on ecological assumptions from other areas. This is especially important in the LHIMP which occurs in the northern extent of the species range but also supports an isolated population which may be subject to unique ecological drivers and trends.

In the LHIMP concerns have been raised since earliest monitoring about increasing densities of Long-spined Sea Urchin and resulting expansion of urchin barrens posing a threat to endemic and unique macroalgae (Edgar et al. 2008, 2011). In line with these predictions, populations have consistently increased by over 300% since 2006, and are likely contributing to increases in ‘bare’ reef documented at some sites such as around the Admiralty Islands. The effect of these population increases on macroalgae in the LHIMP is more

difficult to identify due to confounding influences on fluctuating macroalgae cover (as discussed below) including grazing impacts from other species. Regardless, targeted analysis and research into the ecological drivers and impacts from the expansion of urchin barrens is recommended as a priority (as per Edgar et al. 2011). There is also strong evidence that multiple sea urchin species show synchronised population trends in the LHIMP and therefore may respond to the same ecological drivers. Therefore, future research and analysis should also aim to distinguish between the grazing impacts of different species where possible, for example through stratification of results by site or depth, given species tend to occupy different depth ranges (Edgar et al. 2008) or mapping current boundaries of urchin barrens and populations in vulnerable areas such as the Algal Holes Ecological Community.

Another research priority relates to how climate change, and a strengthening East Australian Current, may influence population dynamics of these three large grazing sea urchin species in the LHIMP (Lamington Sea Urchin, Long-spine Sea Urchin, and Purple Sea Urchin). Being regionally endemic with subtropical to temperate distributions, these three species may show future population declines in the northern extent of their range and be susceptible to mortality from marine heatwaves (Davis et al. 2023, Gall 2016, Byrne et al. 2022). Furthermore, warming water is likely to shorten their larval duration resulting in reduced recruitment to the LHIMP from Eastern Australia, making these isolated populations further vulnerable to impacts and declines (Gall 2016, Bronstein et al. 2021, McLaren et al. 2023). These predictions contrast with the increasing densities and population booms documented in the LHIMP from 2006 to 2022, highlighting the need for targeted research to better understand the drivers of local populations dynamics including larval sources and population connectivity. Genetic analysis is currently being undertaken for all three species to help address this knowledge gap and complement ecological studies into their population dynamics.

Other actions to address sea urchin grazing impacts have been previously suggested including manual removal/control and further reducing fishing pressure on benthic invertivore predators (Edgar et al. 2011, Stuart-Smith et al. 2015, 2019). The potential success of these actions is not known and likely subject to broader ecological drivers including oceanic processes, climate change, and larval sources. Manual removal of sea urchins may present a short-term and small-scale opportunity to preserve aesthetic value of key dive and snorkel sites, or areas supporting vulnerable species such as endemic macroalgae (Edgar et al. 2011). The latter would therefore need to be complemented by targeted mapping of endemic or vulnerable macroalgae species and assemblages at a higher spatial and taxonomic resolution than the current RLS monitoring data. Consideration of any larger-scale harvesting or control strategies should take in to account the unknown implications of climate change on local population of these sea urchin species as described above, and note lessons learnt from other sea urchin fisheries or control programs including previous overfishing of Purple Sea Urchin elsewhere in its range (Gall 2016).

Reducing fishing impacts on benthic invertivores including Doubleheader have also previously been suggested (for example through increased compliance activity, additional species protection, or increases in SZ area) given these predators may help control sea urchin densities on a broader scale (Edgar et al. 2011, Stuart-Smith et al. 2015, 2019). However, analysis of data from 2006 to 2022 showed no detectable relationship between sea urchin density and benthic invertivores or Doubleheader specifically. This concurs with previous findings that predator biomass did not influence sea urchin density during the 2008 population boom of Lamington Sea Urchin (Edgar et al. 2011). These fish have also not shown any consistent trends of reduced biomass attributable to fishing pressure. Nonetheless, further research into the life history and ecological role of Doubleheader is highly recommended along with ongoing monitoring of these trends and further consideration of management opportunities.

SANCTUARY ZONE VALUES

A key management opportunity for the LHIMP and LHI community is increasing education and awareness of the natural, social and economic values supported by SZs in the LHIMP. Shallow reefs in the LHIMP are globally unique and SZs support almost all hotspots for their natural values including species diversity, biomass, and populations of threatened or protected species. In these areas shallow reef ecosystems are afforded maximum protection from a range of localised human impacts including fishing, collecting, anchoring, development and other disturbances. SZs in the LHIMP therefore preserve some of the world's best examples of shallow reef ecosystems in relatively pristine condition, qualities which are highly valued by the community (Heller 2024). They also support a wide range of low-impact recreational and commercial activities, making highly important contributions to social values and the local tourism economy (Heller 2024). In a world of shifting baselines where we rarely experience or research unimpacted reefs, SZs in the LHIMP therefore provide an increasingly valuable opportunity to appreciate and understand pristine and protected shallow reef ecosystems.

MACROALGAE DYNAMICS

The largest documented changes in benthic assemblages on shallow reefs in the LHIMP were driven by fluctuations in macroalgae cover. These dynamics appear to be highly complex and respond strongly to several ecological drivers over different temporal scales. These include rapid declines in macroalgae on offshore reefs (particularly brown foliose macroalgae) due to population booms of grazing sea urchins, sustained declines in macroalgae (including *Caulerpa taxifolia* at Neds Beach) corresponding with gradual increases in grazing sea urchins, increases in macroalgae cover on lagoon reefs following coral bleaching and mortality, and other large fluctuations which cannot yet be clearly explained. There are also likely to be seasonal fluctuations including annual reproductive cycles of green algae such as *Caulerpa spp.*, and potential responses to other drivers such local nutrient inputs and broader oceanographic processes (Stuart-Smith et al. 2015).

This therefore represents a key research opportunity to better document and understand the baseline dynamics of macroalgae on shallow reefs in the LHIMP, to help identify potential drivers, flow-on effects, and signs of negative impacts or ecological phase shifts over and above baseline dynamics (as per Stuart-Smith et al. 2015, 2019). To do this, trends in different functional groups of macroalgae should be investigated separately, given they are likely to show different temporal dynamics and respond to different ecological drivers and impacts (as per Ainsworth et al. 2021). This may also help inform the temporal dynamics of flow-on effects to other species including herbivorous fish (such as Bluefish) and invertebrates (such as the endemic Onion Turban sea snail and sea urchins) as noted in previous reports (Stuart-Smith et al. 2019). Similarly, ecological drivers and trends in the benthic ascidian *Diplosoma virens* which has increasingly high cover on some reefs should be investigated to better understand how it interacts with other benthic species and any flow-on effects.

ECOLOGICAL COMMUNITIES

Another key research and management opportunity is to further consider how each unique Ecological Community in the LHIMP is sustained and impacted by different ecological drivers and threats. Along with supporting unique natural values as outlined in the introduction of this report, the four distinct shallow reef Ecological Communities in the LHIMP contribute in unique ways to the social, cultural and economic values of the marine park by facilitating different kinds of activities and uses. Each Ecological Community is also uniquely susceptible to different threats, and likely to benefit from different management opportunities.

The close proximity of these distinct Ecological Communities in the LHIMP make it globally unique, however, the environmental drivers which result in this co-occurrence are not yet fully understood. While depth, wave exposure and turbidity likely drive many differences between Ecological Communities (Edgar et al. 2008) there may be other confounding factors. In particular, environmental conditions resulting in formation of the unique Algal Holes Ecological Community on the outer edge of the southern lagoon are not yet understood and may include differences in water temperatures, salinity, exposure to wave energy, shading effects of the adjacent mountains, or nutrient runoff from dense seabird populations in the mountains. Edgar et al., (2008) suggested that localised nutrient enrichment as the most likely factor responsible for the development of this Ecological Community, and LHIMP staff are currently facilitating research to help discriminate the sources and ecological consequences of different nutrient inputs into the LHIMP to help address this (as recommended by Valentine et al. 2008 and subsequent reports). There may also be differences in current and outflow in this area, indicated by the uniquely well-developed spur-and-groove formations along the reef crest here (known as the 'Potholes'). The geomorphology of the reef crest in this area, being slightly deeper and occurring between wave-deposited rubble banks, may also be an ecological driver. Given the close proximity of different Ecological Communities in the LHIMP, investigating their ecological drivers represents a significant research opportunity to better understand them both in the LHIMP and elsewhere (as also noted by Edgar et al., 2008).

While long-term maintenance of SZ boundaries provides maximum benefits to natural and other values (Edgar et al. 2014) there is an outstanding management opportunity to represent all Ecological Communities in LHIMP SZs. To do so would necessitate the extension or addition of a SZ encompassing the Algal Holes Ecological Community as recommended previously (Edgar et al. 2008, 2011, Stuart-Smith et al 2015, 2019). Minimising impacts to this unique macroalgae dominated Ecological Community is particularly important given: the global and local conservation significance of macroalgae in the LHIMP; the role it plays in supporting local populations of herbivorous fish including Bluefish and endemic sea snail *Turbo Cepoides*; the increasing vulnerability of this characteristically temperate community to climate change into the future; and importance of maintaining healthy population of benthic invertivore fish in these areas to control sea urchin populations and grazing impacts (Edgar et al. 2008, 2011; Stuart-Smith et al. 2015). All other Ecological Communities in the LHIMP are currently represented within SZs, and these areas tend to support the highest natural values such as species diversity and fish biomass.

BROADER CONTEXT

Better insight into the ecological trends and trajectories described here may also be gained from comparisons with other nearby or ecologically similar locations. For example, macroalgae trends and dynamics could be compared with other locations in temperate-tropical transition zones which have both reef and coral dominated shallow reefs. Trends in sea urchin populations could be compared with other nearby locations with the species ranges, with particular regard to larval dispersal and thermal tolerances. Many threats and trends are also likely to be comparable with those on the shallow reefs of Norfolk Island, including macroalgae and benthic ascidian dynamics, impacts from bleaching and local nutrient input (Ainsworth et al. 2021). RLS data is freely available for download via the Australian Ocean Data Network data portal, and temporal trends are also presented via the RLS Reef Life Explorer website, to facilitate these comparisons. Such comparisons not only present an opportunity to better understand shallow reef ecology in the LHIMP, but also across reefs regionally or globally.

Important ecological context is also facilitated through continued long-term monitoring. As recommended in previous reports, monitoring should continue on a two-yearly basis into the future. As shown in this report, the long time series provides invaluable context to assess and re-examine key ecological patterns and trends. For example, some trends documented in previous reports were found to show fluctuating patterns over a

longer time series, but little net change from 2006 to 2022 over and above this variability. In contrast, some trends emerged over the longer time series which had smaller magnitude of change but were persistent, therefore being difficult to detect over shorter monitoring periods but resulted in ecologically important net differences from 2006 to 2022. Some interactions between different components of each Ecological Community have also become more apparent as synchronised or inverse trends over the longer time series, providing key insights and hypotheses for further monitoring, additional targeted research, and recommended management opportunities.

5 Recommendations

Many of the recommendations made in previous reports have been, or continue to be, addressed by LHIMP management and researchers, including:

- Continued monitoring of shallow reefs using the same methods at 2 yearly intervals
- Expansion of monitoring sites as per Edgar et al. (2008, 2011), Valentine et al. (2008) and Stuart-Smith et al. (2015)
- Long-term maintenance of established SZs to maximise biodiversity and ecosystem benefits
- Establishment of an ongoing marine pest monitoring program
- Research on the effects of nutrient input and enrichment
- Additional monitoring of intertidal reefs in the LHIMP
- Additional surveys of impacted and reference sites following impact events such as spills or mass bleaching
- Further research on threats of fishing pressure to Bluefish in the LHIMP
- Ongoing community education about the conservation importance of Bluefish and Doubleheader, including fishing impacts on these species
- Investigation of trends in the green macroalgae *Caulerpa spp.* and grazing pressure by herbivorous fishes
- Targeted research into the population dynamics, movements and bycatch of Galapagos Shark
- Investigations of illegal fishing activity in SZs

Further recommendations supported by findings of this study and ongoing from previous reports are:

- Further assessment of climate change impacts on shallow reefs in the LHIMP including
 - Ongoing monitoring of coral bleaching and mortality impacts, including flow-on effects to corallivores and McCulloch's Anemonefish
 - Additional research into the interactions between climate change and other stressors, to help identify management actions which can improve reef resilience
- Ongoing assessment of fishing impacts including
 - Continued monitoring of Bluefish biomass and fishing pressure on this species
 - Research to better understand the ecological role and population dynamics of Doubleheader, including impacts of fishing pressure
- Further research on ecological drivers and impacts of sea urchin grazing in the LHIMP including
 - Factors contributing to the formation of Long-spined Sea Urchin (*Centrostephanus rodgersii*) barrens and their flow-on effects to other species
 - Drivers and trends in boom-and-bust population dynamics of Lamington Sea Urchin (*Tripneustes australiae*) and associated impacts
 - Predicted impacts of climate change and strengthening of the East Australian Current on LHIMP populations of these species as well as the Purple Sea Urchin (*Heliocidaris tuberculata*)

- Further investigation of potential management actions including targeted removal or reduced fishing pressure on benthic invertivore predators
- Increasing education and awareness of values supported by SZs in the LHIMP
- Investigate trends and ecological drivers of macroalgae dynamics including
 - Assessing and mapping the spatial patterns of macroalgae to a higher taxonomic resolution, to identify key strongholds for endemic species and areas most threatened by grazing impacts
 - Further investigations of baseline fluctuations and trends in the growth and abundance of different functional groups of macroalgae, with the aim of being able to detect impacts over and above these
- Research into the ecological drivers of, and threats to, distinct shallow reef Ecological Communities (including coral and macroalgae dominated reefs) in the LHIMP, including consideration of a SZ which includes the Algal Holes Ecological Community
- Comparison of trends in the LHIMP with other nearby or ecologically relevant locations



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Appendices

APPENDIX 1 –Number of surveys by site by year.

site_code	site_name	latitude	longitude	Zone	2006	2008	2009	2010	2012	2014	2016	2018	2020	2022
LHI1	North Bommie	-31.52379	159.03913	HPZ	2	2	2	3	2	3	3	2	2	2
LHI10	Noddy Island	-31.50197	159.06513	SZ	2	2	2	2	3	4	5	2	3	4
LHI11	Little Slope	-31.58355	159.06596	SZ	4	4	5	4	4	4	5	4	4	4
LHI13	Little Island	-31.57082	159.06824	HPZ	2	2	4	2	2	2	6	2	2	2
LHI14	Algal Hole North	-31.56235	159.06843	HPZ	2	2	2	2	2	2	3	2	2	4
LHI15	Algal Hole South	-31.56469	159.07015	HPZ	2	2	3	2	2	2	3	2	2	2
LHI16	Rabbit Island	-31.53915	159.05341	SZ	2	2	3	2	4	4	4	2	2	2
LHI17	North Head inside	-31.52289	159.04014	SZ	2	2	3	2	4	2	5	2	2	2
LHI18	Keyhole North	-31.49747	159.06767	HPZ	2	2	5	2	2	2	3	4	5	6
LHI19	Sugarloaf West	-31.50414	159.06679	SZ	4	4	5	5	4	4	4	4	3	4
LHI2	North Bommie 2	-31.52352	159.04141	HPZ	2	2	6	2	2	2	6	2	2	3
LHI21	Big Slope	-31.5954	159.07875	SZ	2	2		2	2	2	2	2	2	2
LHI22	Georges Bay	-31.56557	159.09975	SZ	2	2		2	2	2	2	2	2	2
LHI23	Boat Harbour NW	-31.55782	159.09852	HPZ	2	2		2	2	2		2	2	2
LHI24	Phillip Rock	-31.51721	159.0343	HPZ	4	4		4	4	4	4	4	4	4
LHI26	Sylphs Hole inner	-31.52032	159.05466	SZ	2	2	2	3	2	2	2	2	4	4
LHI27	Sylphs Hole	-31.5207	159.05458	SZ	2	2	2	3	2	2	2	2		
LHI28	Old Gulch	-31.51293	159.0428	HPZ	4	4	4	6	6	3	5	6	4	6
LHI3	Erscotts Blind Passage	-31.54974	159.06295	SZ	4	4	4	6	4	4	4	4	4	4
LHI30	Malabar	-31.51059	159.0556	SZ	2	2	5	3	2	2	2	2	2	2
LHI31	Wheatsheaf	-31.75636	159.23627	HPZ	2	2		2	2	2	2	2	2	3
LHI32	Observatory	-31.75067	159.23682	SZ	2	2		2	2	2	2	2	2	2
LHI33	Signal Point	-31.52736	159.05983	HPZ	2	2		2	2	2	2	2	2	2
LHI34	Neds Beach Deep	-31.5134	159.06903	SZ		2		3	2	2	2	2	2	2
LHI35	Middle Beach	-31.5231	159.07723	HPZ		2		5	2	2	2	2	2	2
LHI36	Stephen's Hole	-31.53225	159.05403	HPZ		2			2	2	2	2		
LHI37	Malabar 2	-31.5113	159.05615	SZ		2			2		2	2		2
LHI38	North Bay 2	-31.52113	159.04688	SZ		2	2	5	2	3	4	2	2	2
LHI39	Yellow Rock Slope	-31.52794	159.04575	HPZ			6		2	4	2	2	2	2
LHI40	Horseshoe Reef	-31.54252	159.06194	HPZ			2	4	2	3	2	4	2	2
LHI41	Stephen's Hole 2	-31.5332	159.05212	HPZ				5	4	4	4	4	2	4
LHI43	South East Rock	-31.7875	159.28145	SZ				2	3			2		
LHI45	Malabar Deep	-31.50823	159.05395	SZ				3		2				
LHI46	Mutton Bird Island	-31.54218	159.10646	HPZ				2	2	2		2	2	2
LHI47	Neds Beach	-31.51793	159.06675	SZ				4	2	3	2	2	2	3
LHI48	Malabar West	-31.51139	159.05416	SZ				4	2	2	5	2	2	4
LHI49	Erscotts Hole	-31.54666	159.06128	HPZ				4	3	3	4	3	3	4
LHI5	Comets Hole	-31.53908	159.06543	SZ	4	4	5	4	4	4	4	4	4	4
LHI50	Le Merthe Hole	-31.52979	159.05013	HPZ					4	3		3	2	
LHI51	Pyramid South Bommie	-31.7592	159.25679	HPZ					3					
LHI52	Sunken Rock	-31.81209	159.2853	SZ								2		
LHI53	Pot O' Gold	-31.52346	159.05777	HPZ								1		
LHI54	New Gulch	-31.51492	159.03793	HPZ							2			
LHI7	Erscotts Passage South	-31.55193	159.06731	HPZ	4	4	5	4	4	4	4	4	4	6
LHI9	Ruperts Reef	-31.49935	159.06494	SZ	2	2	4	2	3	3	4	3	3	3

Appendix 2. - Average % cover of coral species (or genus) at lagoon sites (average of data for all years).

Coral species	LHI26	LHI38	LHI40	LHI41	LHI49	LHI5	mean
Porites heronensis	17.63	2.44	17.26	3.12	4.72	24.56	11.62
Acropora yongei	0.04	27.07	0.06	0	0	0	4.53
Pocillopora damicornis	3.55	2.3	6.04	1.51	1.16	3.11	2.94
Stylophora pistillata	2.67	2.25	2.16	1.12	1.72	3.8	2.29
Seriatopora caliendrum	0.3	0.37	1.51	0	0.69	5.45	1.39
Isopora spp.	0.4	0.25	2.42	3.54	1.23	0.2	1.34
Acropora austera	0	0.5	0.12	0	6.88	0	1.25
Acropora lovelli	0	1.22	0.04	0	5.92	0	1.20
Isopora cuneata	0.07	0.66	0	2.88	1.23	0.66	0.92
Acropora glauca	0	0.2	2.2	1.16	1.74	0.05	0.89
Acropora abrotanoides	0.42	1.89	0	0.05	0.74	0.07	0.53
Acropora spp.	0.16	0.53	0.03	0.26	0.71	0.57	0.38
Seriatopora hystrix	0	0.04	0	0	0.08	2.12	0.37
Porites spp.	0.16	0.07	0	0.3	0.09	1.34	0.33
Acropora tortuosa	0	1.67	0	0	0	0	0.28
Isopora palifera	0.08	0.17	0.16	0.3	0.42	0.07	0.20
Acropora digitifera	0	0	0	0	1.16	0	0.19
Cyphastrea spp.	0.25	0.14	0	0	0.08	0.1	0.10
Acropora hyacinthus	0	0	0	0	0	0.51	0.09
Cyphastrea serailia	0.08	0	0.43	0	0.04	0	0.09
Goniastrea australensis	0	0	0	0.37	0.14	0	0.09
Acropora solitaryensis	0	0.14	0	0.27	0	0	0.07
Astrea curta	0	0.06	0.16	0.04	0.1	0	0.06
Montipora aequituberculata	0	0	0	0	0	0.38	0.06
Acanthastrea hillae	0	0	0	0	0.28	0	0.05
Goniastrea favulus	0	0	0	0.3	0	0	0.05
Cyphastrea chalcidicum	0	0	0.21	0	0	0	0.03
Montipora spongodes	0	0	0	0	0	0.17	0.03
Porites lichen	0	0	0	0.11	0.05	0.03	0.03
Astreopora spp.	0	0	0	0	0	0.12	0.02
Seriatopora spp.	0	0	0	0	0.05	0.07	0.02
Acropora polystoma	0	0	0.12	0	0	0	0.02
Echinophyllia aspera	0	0	0.03	0	0.04	0.04	0.02
Montipora mollis	0	0	0.12	0	0	0	0.02
Montipora spp.	0	0	0.03	0	0	0.07	0.02
Acanthastrea spp.	0	0.08	0	0	0	0	0.01
Goniopora spp.	0	0.03	0	0	0.05	0	0.01
Acanthastrea lordhowensis	0	0	0	0	0.05	0	0.01
Favia fava	0	0	0	0.04	0.04	0	0.01
Favia rotumana	0	0	0	0	0.09	0	0.01
Favia speciosa	0	0	0	0	0	0.07	0.01
Favites halicora	0	0	0	0	0.05	0	0.01
Goniopora norfolkensis	0	0	0	0	0.09	0	0.01
Hydnophora exesa	0	0.06	0	0	0	0	0.01
Merulina ampliata	0	0.06	0	0	0	0	0.01
Montipora turgescens	0	0	0.06	0	0	0.03	0.01
Plesiastrea versipora	0	0.06	0	0	0	0	0.01
Pocillopora aliciae	0	0	0	0.07	0	0	0.01
Psammocora profundacella	0	0	0.04	0	0	0	0.01
Favia spp.	0	0	0	0	0.05	0	0.01
Goniastrea spp.	0	0	0	0	0.05	0	0.01
Acropora cytherea	0	0	0	0	0	0.03	0.00
Acropora donei	0	0	0.03	0	0	0	0.00
Astrea annuligera	0	0.03	0	0	0	0	0.00

APPENDIX 3

LHI_report_coral_species	LHI26	LHI38	LHI40	LHI41	LHI49	LHI5	mean
Porites heronensis	7.72	1.02	12.5	2.52	4.22	13.62	6.93
Porites cf heronensis (Lord Howe)	9.91	1.43	4.76	0.6	0.49	10.94	4.69
Acropora yongei	0.04	27.07	0.06	0	0	0	4.53
Pocillopora damicornis	3.55	2.3	6.04	1.51	1.16	3.11	2.94
Stylophora pistillata	2.67	2.25	2.16	1.12	1.72	3.8	2.29
Seriatopora caliendrum	0.3	0.37	1.51	0	0.69	5.45	1.39
Isopora Submassive	0.4	0.25	2.42	3.54	1.23	0.17	1.33
Acropora austera	0	0.5	0.12	0	6.88	0	1.25
Acropora lovelli	0	1.22	0.04	0	5.92	0	1.2
Isopora cuneata	0.07	0.66	0	2.88	1.23	0.66	0.92
Acropora glauca	0	0.2	2.2	1.16	1.74	0.05	0.89
Acropora abrotanoides	0.42	1.89	0	0.05	0.74	0.07	0.53
Seriatopora hystrix	0	0.04	0	0	0.08	2.12	0.37
Acropora tortuosa	0	1.67	0	0	0	0	0.28
Porites Submassive	0	0	0	0.26	0.04	1.05	0.22
Isopora palifera	0.08	0.17	0.16	0.3	0.42	0.07	0.2
Acropora digitifera	0	0	0	0	1.16	0	0.19
Sub-massive corals	0.08	0.06	0.04	0	0.43	0.19	0.13
Cyphastrea Submassive	0.25	0.14	0	0	0.08	0.1	0.1
Acropora hyacinthus	0	0	0	0	0	0.51	0.09
Cyphastrea serailia	0.08	0	0.43	0	0.04	0	0.09
Encrusting corals	0.11	0.03	0	0.22	0.14	0.03	0.09
Goniastrea australensis	0	0	0	0.37	0.14	0	0.09
Tabular Acropora corals	0	0	0.03	0	0.52	0	0.09
Acropora solitaryensis	0	0.14	0	0.27	0	0	0.07
Astrea curta	0	0.06	0.16	0.04	0.1	0	0.06
Montipora aequituberculata	0	0	0	0	0	0.38	0.06
Acanthastrea hillae	0	0	0	0	0.28	0	0.05
Acropora Solid plate	0	0	0	0	0	0.32	0.05
Goniastrea favulus	0	0	0	0.3	0	0	0.05
Porites	0.13	0.07	0	0	0	0.1	0.05
Acropora	0.05	0.03	0	0.04	0	0.14	0.04
Acropora Staghorn	0	0.22	0	0	0	0	0.04
Corymbose Acropora corals	0	0.08	0	0	0.05	0.08	0.04
Acropora Arborescent table	0	0.17	0	0	0	0	0.03
Cyphastrea chalcidicum	0	0	0.21	0	0	0	0.03
Montipora spongodes	0	0	0	0	0	0.17	0.03
Porites Columnar or digitate	0	0	0	0	0	0.16	0.03
Porites lichen	0	0	0	0.11	0.05	0.03	0.03
Tubipora musica	0	0	0	0.04	0.05	0.06	0.03
Acropora polystoma	0	0	0.12	0	0	0	0.02
Astreopora	0	0	0	0	0	0.12	0.02
Echinophyllia aspera	0	0	0.03	0	0.04	0.04	0.02
Foliose/Plate corals	0	0	0	0.07	0.05	0	0.02
Montipora mollis	0	0	0.12	0	0	0	0.02
Porites Encrusting	0.03	0	0	0.04	0	0.03	0.02
Seriatopora	0	0	0	0	0.05	0.07	0.02
Acanthastrea Submassive	0	0.06	0	0	0	0	0.01

LHI_report_coral_species	LHI26	LHI38	LHI40	LHI41	LHI49	LHI5	mean
Acanthastrea lordhowensis	0	0	0	0	0.05	0	0.01
Favia Submassive	0	0	0	0	0.05	0	0.01
Favia fava	0	0	0	0.04	0.04	0	0.01
Favia rotumana	0	0	0	0	0.09	0	0.01
Favia speciosa	0	0	0	0	0	0.07	0.01
Favites halicora	0	0	0	0	0.05	0	0.01
Goniastrea Hemispherical	0	0	0	0	0.05	0	0.01
Goniopora Hemispherical	0	0.03	0	0	0.05	0	0.01
Goniopora norfolkensis	0	0	0	0	0.09	0	0.01
Hydnophora exesa	0	0.06	0	0	0	0	0.01
Merulina ampliata	0	0.06	0	0	0	0	0.01
Montipora	0	0	0	0	0	0.04	0.01
Montipora turgescens	0	0	0.06	0	0	0.03	0.01
Plesiastrea versipora	0	0.06	0	0	0	0	0.01
Pocillopora aliciae	0	0	0	0.07	0	0	0.01
Porites Massive	0	0	0	0	0.05	0	0.01
Psammocora profundacella	0	0	0.04	0	0	0	0.01
Acanthastrea	0	0.02	0	0	0	0	0
Acropora cytherea	0	0	0	0	0	0.03	0
Acropora donei	0	0	0.03	0	0	0	0
Astrea annuligera	0	0.03	0	0	0	0	0
Branching corals	0	0	0	0	0	0	0
Isopora Encrusting	0	0	0	0	0	0.03	0
Massive corals	0.01	0	0	0	0	0	0
Montipora Foliose	0	0	0	0	0	0.03	0
Montipora Submassive	0	0	0.03	0	0	0	0
Stony corals	0	0	0	0	0	0	0

APPENDIX 4 –

Fish species observed on surveys during RLS expeditions

Species Name	Species Name	Species Name	Species Name
Abudefduf bengalensis	Chrysiptera flavipinnis	Limnichthys fasciatus	Scarus altipinnis
Abudefduf sexfasciatus	Chrysiptera notialis	Lutjanus bohar	Scarus chameleon
Abudefduf sordidus	Cirrhilabrus punctatus	Lutjanus fulviflamma	Scarus dimidiatus
Abudefduf vaigiensis	Cirrhichthys aprinus	Lutjanus kasmira	Scarus flavipectoralis
Acanthistius cinctus	Cirrhichthys falco	Macrodonatogobius wilburi	Scarus frenatus
Acanthurus albipectoralis	Cirripectes alboapicalis	Macropharyngodon meleagris	Scarus ghobban
Acanthurus blochii	Cirripectes chelomatus	Macropharyngodon negrosensis	Scarus globiceps
Acanthurus dussumieri	Cirripectes filamentosus	Malacanthus brevisrostris	Scarus longipinnis
Acanthurus nigrofuscus	Cirripectes stigmaticus	Meiacanthus atrodorsalis	Scarus niger
Acanthurus olivaceus	Coris aygula	Microcanthus joyceae	Scarus oviceps
Acanthurus pyroferus	Coris bulbifrons	Monotaxis grandoculis	Scarus psittacus
Acanthurus triostegus	Coris dorsomacula	Morwong ephippium	Scarus rivulatus
Aluterus scriptus	Coris picta	Mulloidichthys flavolineatus	Scarus rubroviolaceus
Amblygobius nocturnus	Coris sandeyeri	Mulloidichthys vanicolensis	Scarus schlegeli

Amblygobius phalaena	Ctenochaetus striatus	Myripristis berndti	Scobinichthys granulatus
Amphichaetodon howensis	Cyprinocirrhites polyactis	Myripristis kuntee	Scolopsis bilineata
Amphiprion latezonatus	Dascyllus aruanus	Myripristis murdjan	Scorpaena cardinalis
Amphiprion mccullochi	Dascyllus reticulatus	Naso brevirostris	Scorpaenodes evides
Anampses caeruleopunctatus	Dascyllus trimaculatus	Naso lituratus	Scorpaenopsis oxycephala
Anampses elegans	Dendrochirus zebra	Naso unicornis	Scorpius violacea
Anampses femininus	Diagramma pictum	Naso vlamingii	Seriola dumerili
Anampses geographicus	Diodon holocanthus	Neoglyphidodon polyacanthus	Seriola lalandi
Anampses neoguinaicus	Diodon hystrix	Neoniphon sammara	Seriola rivoliana
Aplodactylus etheridgii	Echeneis naucrates	Notocirrhites splendens	Siganus fuscescens
Apogon limenus	Echidna nebulosa	Notolabrus gymnogenis	Sillago ciliata
Arothron hispidus	Elagatis bipinnulata	Notolabrus inscriptus	Sphyræna barracuda
Arothron nigropunctatus	Enchelycore ramosa	Novaculichthys taeniourus	Stanulus talboti
Arripis trutta	Enneapterygius howensis	Novaculoides macrolepidotus	Stegastes apicalis
Arripis xylabion	Enneapterygius rufopileus	Octopus cyanea	Stegastes fasciolatus
Aspidontus taeniatus	Epibulus insidiator	Ostorhinchus aureus	Stegastes gascoynei
Asterropteryx semipunctata	Epinephelus cyanopodus	Ostorhinchus doederleini	Stegastes lacrymatus
Atherinomorus vaigiensis	Epinephelus daemeli	Ostorhinchus flavus	Stethojulis bandanensis
Atypichthys latus	Epinephelus fasciatus	Ostorhinchus norfolcensis	Stethojulis interrupta
Aulostomus chinensis	Epinephelus maculatus	Ostracion cubicus	Stethojulis strigiventer
Balistoides conspicillum	Epinephelus melanostigma	Ostracion meleagris	Suezichthys arquatus
Bathystethus cultratus	Epinephelus merra	Oxycheilinus digramma	Sufflamen chrysopterum
Bathytoshia brevicaudata	Epinephelus polyphkadion	Oxymonacanthus longirostris	Sufflamen fraenatum
Bathytoshia lata	Epinephelus rivulatus	Paracaesio xanthura	Synodus dermatogenys
Bodianus axillaris	Epinephelus tauvina	Paracirrhites arcatus	Synodus doaki
Bodianus perditio	Eretmochelys imbricata	Paracirrhites forsteri	Synodus jaculum
Bothus mancus	Eviota hoesei	Paraluteres prionurus	Synodus similis
Brachaluteres taylori	Eviota readerae	Parapercis australis	Synodus variegatus
Cantherhines dumerilii	Eviota sigillata	Parapercis queenslandica	Taeniurops meyeri
Cantherhines fronticinctus	Exallias brevis	Parapriacanthus elongatus	Teixeirichthys jordani
Cantherhines pardalis	Fistularia commersonii	Parapriacanthus ransonneti	Thalassoma amblycephalum
Canthigaster bennetti	Forcipiger flavissimus	Parma alboscaphularis	Thalassoma hardwicke
Canthigaster callisterna	Fusigobius duospilus	Parma polylepis	Thalassoma lunare
Canthigaster janthinoptera	Fusigobius neophytus	Parupeneus barberinus	Thalassoma lutescens
Canthigaster valentini	Genicanthus semicinctus	Parupeneus ciliatus	Thalassoma nigrofasciatum
Carangoides ferdau	Girella cyanea	Parupeneus cyclostomus	Thalassoma purpureum
Carangoides orthogrammus	Girella elevata	Parupeneus multifasciatus	Thalassoma quinquevittatum
Caranx lugubris	Gnatholepis anjerensis	Parupeneus pleurostigma	Thalassoma trilobatum
Caranx melampygus	Gnatholepis cauerensis	Parupeneus spilurus	Thamnaconus analis
Caranx sexfasciatus	Gobiodon citrinus	Pempheris affinis	Trachinotus baillonii
Carcharhinus galapagensis	Gomphosus varius	Pempheris analis	Trachurus novaezelandiae
Centropyge bispinosa	Goniistius francisi	Pempheris oualensis	Trachypoma macracanthus
Centropyge flavissima	Goniistius vestitus	Pentapodus paradiseus	Valenciennea strigata
Centropyge tibicen	Grammistes sexlineatus	Pervagor alternans	Zanclus cornutus
Centropyge vrolikii	Gymnothorax annasona	Pervagor janthinosoma	Zebrosoma scopas
Cephalopholis argus	Gymnothorax eurostus	Petroscirtes fallax	Zebrosoma velifer

Cephalopholis miniata	Gymnothorax favagineus	Plagiotremus rhinorhynchus	Salarias fasciatus
Chaetodon auriga	Gymnothorax meleagris	Plagiotremus tapeinosoma	Eviota sp. [green]
Chaetodon bennetti	Gymnothorax nubilus	Platax teira	Gymnothorax undulatus
Chaetodon citrinellus	Gymnothorax prionodon	Plectorhinchus flavomaculatus	Priolepis cincta
Chaetodon ephippium	Gymnothorax thyrsoideus	Plectorhinchus picus	Norfolkia squamiceps
Chaetodon flavirostris	Halicampus boothae	Plectroglyphidodon dickii	Gymnothorax thyrsoideus
Chaetodon guentheri	Halichoeres biocellatus	Plectroglyphidodon johnstonianus	Sargocentron cornutum
Chaetodon kleinii	Halichoeres hortulanus	Plesiops insularis	Trygonorrhina fasciata
Chaetodon lineolatus	Halichoeres margaritaceus	Plotosus lineatus	Dicotylichthys punctulatus
Chaetodon lunula	Halichoeres marginatus	Pomacanthus imperator	Aseraggodes bahamondei
Chaetodon lunulatus	Halichoeres melanurus	Pomacanthus semicirculatus	Gymnothorax porphyreus
Chaetodon melannotus	Halichoeres nebulosus	Pomacentrus australis	Hypoplectrodes maccullochi
Chaetodon mertensii	Halichoeres trimaculatus	Pomacentrus bankanensis	Istigobius decoratus
Chaetodon ornatissimus	Hemigymnus fasciatus	Pomacentrus coelestis	Bothus pantherinus
Chaetodon pelewensis	Hemigymnus melapterus	Pomacentrus moluccensis	Eviota fasciola
Chaetodon plebeius	Heniochus acuminatus	Priacanthus hamrur	Koumansetta rainfordi
Chaetodon rainfordi	Heniochus chrysostomus	Prionurus maculatus	Ecsenius fourmanoiri
Chaetodon speculum	Heniochus monoceros	Prionurus microlepidotus	Fusigobius inframaculatus
Chaetodontoplus conspicillatus	Heniochus varius	Pseudanthias pictilis	Paragobiodon echinocephalus
Chaetodontoplus meredithi	Heteropriacanthus cruentatus	Pseudanthias squamipinnis	Scorpaenopsis cirrosa
Chaetodon tricinctus	Hipposcarus longiceps	Pseudocaranx dentex	Lepadichthys frenatus
Chaetodon trifascialis	Hologymnosus annulatus	Pseudocaranx georgianus	Eviota teresae
Chaetodon ulietensis	Hologymnosus doliatus	Pseudocaranx sp. [dentex]	Scorpaenopsis diabolus
Chaetodon unimaculatus	Hologymnosus sp. [dark]	Pseudocheilinus hexataenia	Pherallodus indicus
Chaetodon vagabundus	Hypoplectrodes annulatus	Pseudocoris yamashiroi	Diplogrammus goramensis
Cheilinus chlorourus	Hypoplectrodes sp. [Lord Howe]	Pseudojuloides elongatus	Ecsenius bicolor
Cheilinus trilobatus	Istigobius rigilius	Pseudolabrus luculentus	Sebastapistes tinkhami
Cheilio inermis	Kuhlia mugil	Ptereleotris evides	Parascorpaena aurita
Cheilodipterus macrodon	Kyphosus bigibbus	Ptereleotris monoptera	Heraldia nocturna
Cheilodipterus quinquelineatus	Kyphosus cinerascens	Ptereleotris zebra	Cirripectes castaneus
Chelonia mydas	Kyphosus sectatrix	Pterocaesio digramma	Acanthocybium solandri
Chilomycterus reticulatus	Kyphosus sydneyanus	Pterois antennata	Aprion virescens
Chironemus marmoratus	Kyphosus vaigiensis	Pterois volitans	Chaetodontoplus ballinae
Chlorurus frontalis	Labracoglossa nitida	Pycnochromis agilis	Chanos chanos
Chlorurus microrhinos	Labrichthys unilineatus	Pycnochromis amboinensis	Diodon liturosus
Chlorurus sordidus	Labroides bicolor	Pycnochromis iomelas	Eviota sp. [trans white & red streaks]
Chromis atripectoralis	Labroides dimidiatus	Pycnochromis margaritifera	Gobiesocid sp. (LHI brown)
Chromis chrysurus	Leururus versicolor	Pycnochromis vanderbilti	Iniistius celebicus
Chromis hypsilepis	Lethrinus atkinsoni	Rhinecanthus lunula	Malacanthus latovittatus
Chromis kennensis	Lethrinus miniatus	Sargocentron rubrum	Oxycheilinus bimaculatus
Chromis nitida	Lethrinus nebulosus	Saurida nebulosa	Plectorhinchus schotaf
Chromis viridis			



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Reef Life Survey