

Parks Victoria Technical Series

Number 119

An integrated monitoring program for Cape Howe Marine National Park

January 2023



Healthy Parks
Healthy People



Published by Parks Victoria

Level 10, 535 Bourke St, Melbourne Vic 3000

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National Library of Australia

Cataloguing-in-publication data

Includes bibliography

ISSN 1448-4935

Citation: Whitmarsh SK, Porskamp P, Tinkler P, Gray S, Howe S, Ierodiaconou D, Sams MA and Young MA (2023) *An integrated monitoring program for Cape Howe Marine National Park*, Parks Victoria Technical Series No 119, Parks Victoria, Melbourne.

Front cover: Cape Howe Marine National Park

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Executive summary

Parks Victoria has established the Signs of Healthy Parks (SHP) program for its network of marine national parks and sanctuaries with the aim of both improving baseline knowledge of Victoria's marine protected areas (MPAs) and addressing applied management questions. The program aims to ensure systematic, robust and integrated ecological monitoring across the breadth of Victoria's marine national parks and sanctuaries. Parks Victoria has implemented subtidal and intertidal reef monitoring programs in a large number of its MPAs from as far back as 1998; however, they only cover a small proportion of the key habitats in the parks. Using advances in ocean technology, and through collaborative research partnership with Deakin University, the SHP program can now monitor the health of the entire extent of a park, including the full gamut of protected features. Building on Parks Victoria's Conservation Action Planning process and historical monitoring programs, the SHP program aims to monitor the health of protected areas using a range of environmental indicators that provide information about natural values and ecological processes occurring within the parks and potential threats. This study implemented a suite of monitoring approaches to assess the health and condition of the no-take Cape Howe Marine National Park (MNP) in eastern Victoria.

Cape Howe MNP is Victoria's most easterly marine national park. It encompasses a diversity of habitats including long sandy beaches, intertidal reef along its eastern shore, shallow and deep subtidal reefs, extensive subtidal soft sediment and expansive areas of deep open water. It supports a high diversity of species due to its location at the mixing of warm eastern and cool southern waters that allow eastern temperate, southern cosmopolitan and temperate species to co-occur. Threats to the park include illegal fishing, particularly for commercially important species such as abalone and Southern Rock Lobster (*Jasus edwardsii*), and increased sea-surface temperatures and ocean acidification in response to climate change. The native herbivorous Longspined Sea Urchin (*Centrostephanus rodgersii*) occurs in abundance on reefs and can remove all erect algae to create 'urchin barrens'.

The aims of this project were to:

- understand changes in sea-surface temperature at Cape Howe MNP
- compile seabed bathymetry data to develop full-coverage maps of the Cape Howe MNP and adjacent waters to assist in developing species distribution models and predicting the occurrence of urchin barrens
- compare abundance and diversity of fish and invertebrate populations and canopy-forming algae through time using Reef Life Survey methods at historical Subtidal Reef Monitoring Program (SRMP) marine protected area (MPA) and reference sites on shallow subtidal reefs
- compare abundance and diversity of fish populations using baited remote underwater video station (BRUVS) survey of demersal fish assemblages within and adjacent to the MPA to
 - initiate time series monitoring of demersal fish assemblages across the entire depth range of the park
 - understand the effect of seafloor structure on abundance and diversity of fish using spatially explicit distribution modelling techniques

- predict abundance and diversity of fish using species distribution models
- understand differences in fish assemblages via species richness, biomass, abundance and relationships with seafloor structure in areas sampled by BRUVS
- assess the abundance and size of Southern Rock Lobster within and outside Cape Howe MNP to understand the effectiveness of protection within the park
- model urchin barren coverage inside Cape Howe MNP using maximum entropy (MaxEnt) species distribution models.

Analyses of annual and seasonal patterns in sea-surface temperature found that all seasons are experiencing an increasing trend in temperature, although winter is very close to experiencing no trend.

The most abundant species of fish observed across years using underwater visual census (UVC) methods were Mado Sweep (*Atypichthys strigatus*), Eastern Hulafish (*Trachinops taeniatus*), One-spot Puller (*Chromis hypsilepis*) and Maori Wrasse (*Ophthalmolepis lineolata*). Common species of invertebrates and cryptic fish observed using UVC methods across years were the Longspined Sea Urchin (*Centrostephanus rodgersii*), Blacklip Abalone (*Haliotis rubra*) and Tent Turban (*Astraliu tentoriiforme*).

Long-term UVC data found the abundance of all key mobile fish species to be healthy and in good condition, with the exception of 4 species. Old Wife (*Enoplosus armatus*), Six-Spine Leatherjacket (*Meuschenia freycineti*) and Eastern Blue Groper (*Achoerodus viridis*) have not been observed using UVC in the park since 2011, 2012 and 2015, respectively, and Crimsonband Wrasse (*Notolabrus gymnogenis*) abundances are below the lower limit of acceptable change. The size of individual fish species was generally similar in MPA and reference sites. However, White-Ear (*Parma microlepis*) were smaller outside the MPA, and Banded Morwong (*Chirodactylus spectabilis*) and Senator Wrasse (*Pictilabrus laticlavus*) have fallen below the lower limit of acceptable change for mean length within the MPA. The drivers for declines in these particular species are not well understood, and further investigation is required to understand the causes and implications of changes in their abundance.

Populations of the key mobile invertebrates the Longspined Sea Urchin, Triton Shell (*Cabestana spengleri*) and Blacklip Abalone were found to be in good condition. However, consistent declines over the past 15 years were observed in Turban Shell (*Lunella undulata*), Purple Sea Urchin (*Heliocidaris erythrogramma*) and Dog Whelk (*Dicathais orbita*). Declines in these species could be related to increases in urchin barrens or loss of habitat at greater depths, but further investigation is required to understand what is causing these declines and whether they are related to declines in fish and algal species detected in this study.

The cover of encrusting coralline algae has declined within the MPA since 2014; it was below the lower control limit in the 2008, 2015 and 2021 surveys. In contrast, cover of the major canopy-forming species Crayweed (*Phyllospora comosa*) had been relatively stable until 2015, when it was not observed. Since then, Crayweed has been on a positive trend upwards with mean percentage cover in the MPA well above the lower limit of acceptable change in 2021. The increase in Crayweed may indicate reduced urchin grazing at shallower depths where UVC was undertaken or successful recruitment of a high number of individuals between 2015 and 2021.

The MaxEnt model predicting the relationship between urchin barren presence and environmental variables found distance to reef to be the most important explanatory variable followed by reef

slope, depth and, finally, MPA status (inside or outside the marine national park). Urchin barrens were most likely to occur on reef habitat, on more complex (sloping) terrain, in shallower depths and in areas outside the MPA. Overall, it was predicted that the study area contains 6.3 km² of likely urchin barrens. This is a substantial area of habitat; it represents about 15% of the total area of the park and a significantly higher proportion of subtidal rocky reef.

The most abundant species observed using BRUVS methods were the Yellowtail Scad (*Trachurus novaezelandiae*), Velvet Leatherjacket (*Meuschenia scaber*), Mado Sweep (*A. strigatus*) and hermit crabs in the Diogenidae family. The most widespread species were Velvet Leatherjacket, Ocean Jacket (*Nelusetta ayraud*), Grey Morwong (*Nemadactylus douglasii*), Jackass Morwong (*Nemadactylus macropterus*) and hermit crabs. The highest contributors to biomass were the Velvet Leatherjacket, Draughtboard Shark (*Cephaloscyllium laticeps*), Ocean Jacket, Southern Eagle Ray (*Myliobatis tenuicaudatus*), Smooth Stingray (*Bathytoshia brevicaudata*) and Gummy Shark (*Mustelus antarcticus*).

As depth increased, the habitat types observed with BRUVS changed across the sampling area. Shallow sites (less than 40 metres (m) deep) were dominated by fine sand with a moderate component of bare rock along with some invertebrate complex and erect macroalgae. The middle depth zones (40 to 60 m) showed a transition zone from fine sand to biogenic pebble and coarse sand-dominated habitats and also had the highest cover of sponges, erect macroalgae and invertebrate-dominated habitats. Deep sites (greater than 60 m) were predominantly low-relief soft-sediment habitats consisting of coarse sand and various forms of pebbles.

Between 2006 and 2021 there was a decline in bare rock cover and an increase in encrusting algae in the lowest depth stratum (10 to 20 m). The prevalence of sponge habitat and coarse sand in the 50 to 60 m stratum also increased. Deeper depth strata in 2006 were almost entirely dominated by biogenic pebble substrate, whereas in 2021 they had biogenic pebbles, coarse sand and larger pebbles. Although the sampling effort in 2021 was less than in 2006, the number of urchin barrens observed was similar. Slightly more barrens were observed in deeper areas in 2021, with barrens observed extending to 50 m in depth.

Southern Rock Lobster were found in higher abundance inside Cape Howe MNP than outside: 63 individuals were caught inside and 13 outside. Modelling of relationships between Southern Rock Lobster and habitat inside and outside the marine national park found strong habitat relationships within the park but not outside, likely due to the low number of individuals caught outside. Results from generalised additive models (GAMs) used to associate Southern Rock Lobster population characteristics with habitat and protection status inside and outside Cape Howe MNP found that lobster numbers increased with depth, eastness and location inside the park. The difference in abundance inside and outside the park may be driven by the lack of suitable habitat, and the greater number of urchin barrens, outside the park. Low numbers of rock lobsters were generally observed in shallow reefs with urchin barrens; most rock lobster observations were associated with sponge-dominated circalittoral reefs. The higher numbers of rock lobster found within Cape Howe MNP compared with outside suggests it protects potentially important source populations whose larvae can disperse beyond park boundaries to replenish distant populations.

Overall, most fish and macroinvertebrate species were found to be in good health in Cape Howe MNP, and changes in habitat types were relatively small, although urchin barrens are a prevalent feature on reefs. However, there have been clear declines in the abundance and condition of key fish and invertebrate species. The underlying driver for these declines remains unclear, and there are not obvious commonalities between many of the species in decline. Given Cape Howe's location at

the interface of warm eastern and cool southern waters, it is an important marine national park to monitor for climate-driven changes. It is not clear whether the changes in some of the species detected in this study are related to increasing sea-surface temperatures or a range of other drivers. Ongoing long-term monitoring and further investigations of potential drivers are needed to better understand what these changes signify.

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1. Introduction

Parks Victoria manages a system of 13 marine national parks and 11 marine sanctuaries, making up approximately 5.3% of Victoria's state waters. Established in 2002, the network of marine national parks and sanctuaries was designed to represent the diversity of Victoria's marine environment, its habitats, and associated flora and fauna (VEAC 2014). In order to reliably manage these areas, an understanding of the condition and trends of natural values that occur within the parks, sanctuaries and reserves is essential (Devillers et al. 2015). Parks Victoria has established extensive marine research and monitoring programs that address significant management challenges for its marine protected areas (MPAs). Such challenges include improving baseline knowledge of the MPAs and addressing new applied management questions. Parks Victoria's research program is guided by the research themes outlined in Parks Victoria's Environmental Research Strategy 2012–2025 and based on priorities identified through the conservation planning process for the marine national parks and sanctuaries.

1.1 Signs of Healthy Parks program

Parks Victoria has established the Signs of Healthy Parks (SHP) program for its network of marine national parks and sanctuaries with the aim of both improving baseline knowledge of Victoria's marine protected areas and addressing applied management questions. The program aims to ensure systematic, robust and integrated ecological monitoring across the breadth of Victoria's marine national parks and sanctuaries.

The SHP program continues and enhances Parks Victoria's historical Subtidal Reef Monitoring Program (SRMP; 1998–2015) (Edmunds and Hart 2003, Lindsay and Edmunds 2006), which used diver underwater visual census methods to describe patterns and trends in macroalgae, fish and macroinvertebrate communities within shallower subtidal sites within MPAs. The SHP program also continues the Intertidal Reef Monitoring Program (IRMP; 2002–2014) (Stewart et al. 2007), which monitored invertebrates and macroalgae present in the intertidal zone on reefs within 9 targeted MPAs and at 9 matched reference sites outside the MPAs. These programs were designed using the best scientific practices of the time; however, they only covered a small proportion of the key habitats in the parks. Using advances in ocean technology, and through collaborative research partnership with Deakin University, the SHP program now monitors the health of the entire extent of a park, allowing the full gamut of protected features to be monitored.

The SHP program monitors indicators of key ecological attributes and threats identified through Parks Victoria's Conservation Action Planning process in at least one of the large marine national parks within each bioregion, including Discovery Bay MNP (Otway bioregion), Point Addis MNP (Central Victoria bioregion), Port Phillip Heads MNP (Victorian Embayments bioregion), Wilsons Promontory MNP (Flinders bioregion) and Cape Howe MNP (Twofold Shelf bioregion). These indicators provide information about the patterns and trends in natural values and ecological processes occurring within the parks as well as identifying potential threats to conservation assets. This study presents a suite of monitoring approaches implemented to assess the ecological health and condition of the no-take Cape Howe MNP.

1.2 Cape Howe Marine National Park

Cape Howe MNP is Victoria's most easterly marine national park. It lies within the Victorian section of the Twofold Shelf bioregion, which is one of the Integrated Marine and Coastal Regionalisation of

Australia (IMCRA) meso-scale regions. Important natural values of Cape Howe MNP arise from its diversity of habitats, which include long sandy beaches, intertidal reef along its eastern shore, shallow and deep subtidal reefs, extensive subtidal soft sediment, and expansive areas of deep open water (Parks Victoria 2006). Its rocky reefs have a complex structure, including eroded low-profile sandstone reef and high-profile granite reef (Carey et al. 2007). The shallow subtidal reef is dominated by a mixture of Crayweed (*Phyllospora comosa*) and Southern Bull Kelp (*Durvillaea potatorum*); the reef further offshore tends to be dominated by *P. comosa* (Ball and Blake 2007, Edmunds et al. 2010).

On these reefs the herbivorous Longspined Sea Urchin (*Centrostephanus rodgersii*) can remove all erect algae to create 'urchin barrens' (Williams et al. 2007). Common invertebrates include the Blacklip Abalone (*Haliotis rubra*), Turban Shell (*Lunella undulata*) and Tent Turban (*Astrarium tentoriiforme*) (Williams et al. 2007, Edmunds et al. 2010). Common fish are Herring Cale (*Olisthops cyanomelas*), Six-Spine Leatherjacket (*Meuschenia freycineti*), Mado Sweep (*Atypichthys strigatus*), Banded Morwong (*Chirodactylus spectabilis*) and the damselfishes White-Ear (*Parma microlepis*) and One-Spot Puller (*Chromis hypsilepis*) (Williams et al. 2007, Edmunds et al. 2010). Its deep (30 to 50 metres (m)) sandstone reefs are heavily covered with a diverse array of sponges, ascidians and gorgonians (Carey et al. 2007, Holmes et al. 2007).

The extensive area of subtidal soft sediments is mainly a fine and medium sand, with a low carbonate content, which becomes dominated by shells in depths greater than 50 m (Holmes et al. 2007). Sponges dominate sediment at depths greater than 40 m, with orange ball sponges (*Tethya* spp.) dominant in 40 to 60 m depth (Holmes et al. 2007). Crustaceans such as amphipods, cumaceans, isopods and ostracods, and polychaetes are the dominant infauna in sediments at 10 to 20 m depth (Heislors and Parry 2007). Common fish over sediment and sediment-covered reef are Yellowtail Scad (*Trachurus novaezelandiae*), Ocean Jacket (*Nelusetta ayraud*), Whiting (*Sillago*), Grubfish (*Parapercis* sp.), Bluespotted Flathead (*Platycephalus caeruleopunctatus*), Velvet Leatherjacket (*Meuschenia scaber*) and Butterfly Perch (*Caesioperca lepidoptera*) (Moore et al. 2009). The Draughtboard Shark (*Cephaloscyllium laticeps*) is also common and can be found down to the deepest depths (105 m) of the marine national park (Moore et al. 2009).

Cape Howe MNP has a high diversity of intertidal and shallow subtidal invertebrates (Edmunds et al. 2005). Eastern temperate, southern cosmopolitan and temperate species co-occur as a result of the mixing of warm eastern and cool southern waters (Edmunds et al. 2005). Thirty-eight species of marine flora and fauna are believed to be at their eastern, western or northern distributional limits within the planning area (O'Hara and Poore 2000).

1.2.1 Threats to natural values in Cape Howe MNP

Illegal resource extraction (i.e. fishing) has been identified as a key threat to natural values in the park. Illegal harvest of abalone has been an issue for many years, especially close to the New South Wales border. Abalone are a keystone species on subtidal reefs, and illegal fishing has the potential to significantly impact on the health of these reefs. This can also compound the effect of Longspined Sea Urchins (*C. rodgersii*) on abalone populations and subtidal reef systems (see below). Illegal fishing also has the potential to impact the abundance of rock lobster and Eastern Blue Groper (*Achoerodus viridis*); these species are predators of Longspined Sea Urchins and can help to mitigate the impacts of urchins on subtidal reefs. Anecdotal reports suggest rock lobster populations have declined in East Gippsland over the past few decades, possibly related to climate change. Illegal fishing has the potential to worsen this effect. Southern Rock Lobster (*Jasus edwardsii*) is also a keystone species on subtidal reefs and a key predator of other mobile invertebrates.

South-east Australia is considered a national climate change hotspot in terms of increases in sea-surface temperature (SST) and ocean acidification. A major impact of marine climate change on Cape Howe MNP has been the range expansion of Longspined Sea Urchins with a strengthening of the East Australian Current. The Longspined Sea Urchin presents a growing threat to marine biodiversity in Victoria. The distribution of *C. rodgersii* was historically restricted to subtidal reefs along the coasts of New South Wales, eastern Victoria and the Flinders Island Group in the Bass Strait (Andrew and O'Neill 2000). However, over the past few decades, the range of *C. rodgersii* has expanded southwards to the east coast of Tasmania and westward along the coastline of Victoria towards Wilsons Promontory. This phenomenon is likely due to strengthening of the East Australian Current, bringing warmer water containing urchin larvae further southwards (Ridgway 2007, Suthers et al. 2011). Range expansion of the Longspined Sea Urchin is resulting in extensive kelp bed loss and barren formation with important consequences for species that rely on kelp for habitat and food. Monitoring and management of overabundant urchins across Victoria is thus a key priority for protecting biodiversity within Victoria's marine protected areas, including within the Cape Howe MNP. Other range-expanding species, such as the Common Sydney Octopus (*Octopus tetricus*), also pose a significant threat to native mobile invertebrate species including Southern Rock Lobster.

Climate stressors are likely to also be having direct effects on other key values in the park. For example, increased SST is implicated in the decline of several species of canopy-forming brown macroalgae such as Giant Kelp (*Macrocystis pyrifera*), Southern Bull Kelp (*Durvillaea potatorum*) and possibly Golden Kelp (*Ecklonia radiata*). Recent research on *Durvillaea* has led to the formal identification of an eastern species, *Durvillaea amatheiae* (Weber et al. 2017). *Durvillaea amatheiae* has a limited range in eastern Victoria (east of Wilsons Promontory); with limited reef habitat available for this species, it is particularly vulnerable to climate change.

Invasive exotic species are also identified as a key threat to natural values in the Cape Howe MNP. Early detection of invasive species is critical as eradication becomes extremely difficult or impossible once marine pests are established. The plankton-feeding New Zealand Screw Shell (*Maoricolpus roseus*) has the ability to blanket the ocean floor and occurs within the park in high densities, resulting in modified seafloor characteristics and competition with native flora and fauna. While very little can be done to manage New Zealand Screw Shells, understanding the extent of this invasive species will help us assess how effectively we can manage other threats.

Marine pollution has also been identified as a potential risk to natural values in the Cape Howe MNP, particularly oil spills due the amount of boating traffic that pass around Cape Howe and sometimes through the park.

1.3 Monitoring priorities of the Cape Howe MNP

Monitoring priorities have been identified for Cape Howe MNP based on Parks Victoria's Conservation Action Planning process. Monitoring priorities focus on understanding the status and trends of key ecological attributes for the 2 major ecosystems found in Cape Howe MNP: subtidal reefs and unvegetated soft sediments. These priorities are outlined below.

1.3.1 Subtidal reefs

Several species and communities associated with both shallow and deep subtidal reefs have been targeted for monitoring in the Cape Howe MNP.

- **Brown macroalgae-dominated beds** (*Durvillaea amatheiae*, *Phyllospora comosa*), which provide information about the condition of dominant habitat types and impacts of urchin grazing and climate change.
- **Large mobile fish (including sharks and rays)**, especially those that are 'site attached' rather than transitory. Large mobile fish species are indicators of local ecosystem health. Changes in numbers and distributions are used to assess the health of ecosystems within and outside the marine national park.
- **Mobile macroinvertebrates**, especially abalone and Southern Rock Lobster as these are keystone species for which illegal fishing has been identified as a major threat in the park. Keystone mobile macroinvertebrates provide information on the overall health of the ecosystem and as well as the effectiveness of protection against illegal harvest.
- **Sessile invertebrate communities** (e.g. sponges, ascidians, gorgonians, sea whips), which provide information about the health of dominant habitats found at deeper depths where macroalgae no longer dominate.

1.3.2 Unvegetated soft sediments

Sessile invertebrate communities have also been assessed within the soft-sediment communities of the park. These communities support complex habitat for a range of fish and mobile invertebrates and provide information about the health of unique soft-sediment habitats.

1.4 Objectives

The overarching objective of this project is to undertake a targeted monitoring program for the Cape Howe MNP based on the key conservation, management and monitoring priorities established for the park. Specific objectives related to this goal include to:

- understand changes in sea-surface temperature at Cape Howe MNP
- compile seabed bathymetry data to develop full-coverage maps of the Cape Howe MNP and adjacent waters to assist in developing species distribution models and predicting the occurrence of urchin barrens
- compare abundance and diversity of fish and invertebrate populations and canopy-forming algae through time using Reef Life Survey methods at historical Subtidal Reef Monitoring Program (SRMP) marine protected area (MPA) and reference sites on shallow subtidal reefs
- compare abundance and diversity of fish populations using baited remote underwater video station (BRUVS) survey of demersal fish assemblages within and adjacent to the MPA to
 - initiate time series monitoring of demersal fish assemblages across the entire depth range of the park

- understand the effect of seafloor structure on abundance and diversity of fish using spatially explicit distribution modelling techniques
- predict abundance and diversity of fish using species distribution models
- understand differences in fish assemblages via species richness, biomass, abundance and relationships with seafloor structure in areas sampled by BRUVS
- assess the abundance and size of Southern Rock Lobster within and outside Cape Howe MNP to understand the effectiveness of protection within the park
- model urchin barren coverage inside Cape Howe MNP using maximum entropy (MaxEnt) species distribution models.

2. Methods

2.1 Sea-surface temperature

Sea-surface temperature data were sourced from the Integrated Marine Observing System (IMOS) (IMOS 2018). IMOS is a national collaborative research infrastructure supported by the Australian Government. These data were downloaded in NetCDF format at monthly intervals and converted into individual ArcGIS rasters for analysis. Annual and seasonal (summer and winter) SST means were computed from 1993 to 2018 from the monthly SST datasets. To assess patterns and trends in seasonal SST within the Cape Howe MNP, a univariate state-space model was run in the MARSS package (Holmes et al. 2018) for annual and seasonal patterns in SST. The MARSS model was chosen because it is capable of determining trends from stochastic datasets like the SST observations in the Cape Howe MNP.

2.2 Reef Life Survey for shallow subtidal reefs

Using the methods developed by the Reef Life Survey (RLS) citizen science program, surveys were completed within the Cape Howe MNP in 2008 (1 site), 2009 (2), 2011 (9), 2012 (1), 2014 (1), 2015 (2) and 2021 (6) (Figure 2.1). The data collected were in addition to data collected in the historic SRMP program, which conducted surveys in 2001 (7 sites), 2004 (8 sites), 2008 (8 sites), 2009 (9 sites), 2010 (8 sites), and 2014 (8 sites). In recent years, Reef Life Survey has become a common method for completing underwater visual census (UVC) surveys of shallow subtidal reefs. Globally, the RLS technique has allowed approximately 2,000 sites to be surveyed using a standard set of survey methods, which are described in detail in an online methods manual (Reef Life Survey 2015).

RLS surveys encompass a UVC technique in which scuba divers swim along a 50 m transect line laid on hard substrate, along the depth contour of the site being surveyed (depths of Cape Howe MNP sites range between 1 and 11 m). All fish species observed within 5 m of the transect line are recorded on a waterproof datasheet as the diver swims slowly along the line (at approximately 2 metres per minute). On the same transect, all macroinvertebrates and cryptic fish larger than 2.5 cm and within 1 m of the transect tape are recorded. At each site, divers complete two 50 m transects in opposite directions along the same depth contour. This means that an area of 1,000 m² is surveyed for fish and 200 m² for invertebrates at each location.

Additionally, 20 digital photo quadrats were collected at 2.5 m intervals along each transect. Photo quadrats are taken from approximately 50 cm above the seabed (usually sufficient to encompass an area of approximately 0.3 m × 0.3 m). These photo quadrats are later annotated using SQUIDLE+ software, with 5 to 9 evenly distributed points on a 5 × 5 grid. Each point is classified to the lowest taxonomic level possible using an adapted Collaborative and Annotation Tools for Analysis of Marine Imagery and video (CATAMI) classification scheme (Althaus et al. 2015). This classification allows data on algae, sessile invertebrates and substratum type to be recorded and stored for later analysis. The main differences in this survey approach compared with previous SRMP methods are a reduced survey effort (from a 200 m total transect distance in SRMP to 100 m at each site in RLS) and the replacement of in situ quadrat surveys with photo quadrats. To account for the differences in quadrat scoring, this study compares only canopy-forming species to previous SRMP observations, as understory communities are likely to be obscured in photo quadrats.

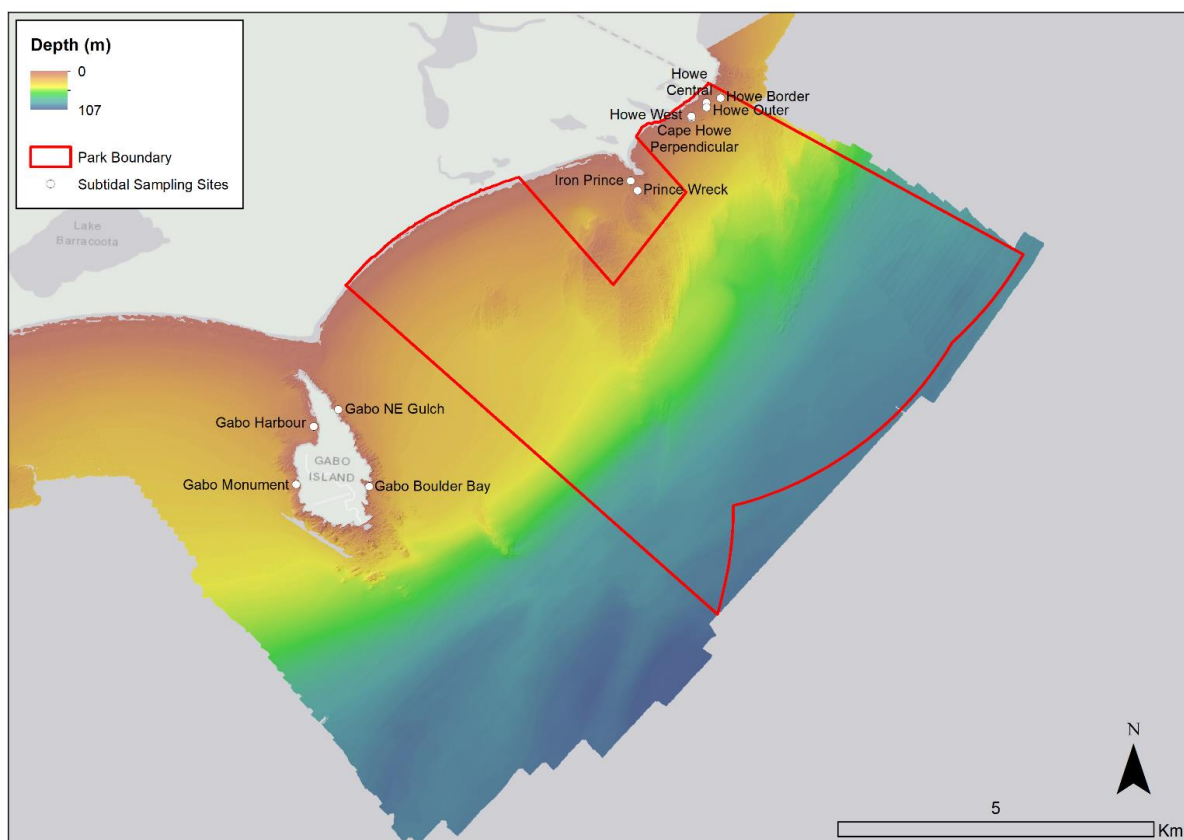


Figure 2.1: Map of the Cape Howe region showing the locations of all SRMP/RLS monitoring sites

2.3 Control charts

Parks Victoria, in collaboration with the University of Melbourne and Deakin University, has developed a series of control charts to provide timely, accurate and reliable information on the condition of natural assets, level of threats and management effectiveness. These charts form part of Parks Victoria's State of the Parks evaluation. They depict a simple line graph tracking an indicator through time, identifying a zone of acceptable change and upper and/or lower control limits to flag values where a management response should be considered.

Control limits around the zone of acceptable change are calculated to reflect the statistical properties of the chart (Montgomery 2007) as follows:

$$UCL = ULAC + \frac{Z_{\alpha}\sigma}{\sqrt{n}}$$

$$LCL = LLAC + \frac{Z_{\alpha}\sigma}{\sqrt{n}}$$

where UCL = upper control limit, LCL = lower control limit, $ULAC$ = upper limit of acceptable change, $LLAC$ = lower limit of acceptable change, Z_{α} = multiplier for number of standard deviations to correspond with desired Type I error rate α , σ = weighted average standard deviation, and n = modal sample size to allow for possibly unequal sample sizes.

The lower limit of acceptable change (LLAC) was initially identified as the minimum value inside the MPA from SRMP surveys before the marine national park was declared (in 2002). For some indicators, modification of this LLAC was needed to ensure the lower control limit remained above zero. Each chart was then plotted over the entire survey period including SRMP surveys (2001–2015) and RLS surveys (2001). The lower control limit (LCL) was then calculated from the LLAC, taking into account the variation in the data (standard deviation) and number of sites surveyed (n) over the entire survey period. For ease of interpretation, zones of acceptable change on charts were shaded green, control limits were indicated by dashed horizontal lines, and areas outside the control limits were shaded red.

One of the current limitations of the control chart approach is how we choose to define the LLAC. Ideally, these limits relate to different management scenarios and actions. However, this often requires a sound understanding of threats to the ecosystem and the resilience of the ecosystem to disturbance and change. While we have made our best efforts in this report to choose LLACs, ultimately this is a process that requires further thought and consideration on a case-by-case basis.

2.4 Baited remote underwater video stations

2.4.1 Sampling design

The baited remote underwater video stations (BRUVS) sampling design was based on previous surveys completed in the park and an extension of coverage both in the park and into an adjacent area outside the park. In the years 2006 and 2021, a total of 237 BRUVS were completed within and around Cape Howe MNP. In 2006, all BRUVS occurred inside the park, while in 2021 sampling occurred both inside and outside the park. A subsample of these 237 BRUVS was selected to help track changes through time. These locations were subsampled by first stratifying sites by depth and complexity and using a minimum distance of 300 m between sites. Additional sites were also added in 2021 to target macroalgae-dominated reef within and adjacent to the marine national park in order to understand the effects of *Centrostephanus rodgersii* barrens on fish assemblages. These sites were also subsampled from previous BRUVS locations, where possible, and areas outside the park were chosen by identifying areas of similar habitat based on bathymetry and complexity information. The coverage of BRUVS within the park was also extended to an unsurveyed wedge of the park in the north-east. These new sites were stratified by depth and complexity. The new sites selected to extend coverage of BRUVS outside the park were also stratified by depth and complexity. The survey design is unbalanced so that a higher density of samples could be taken from inside the park; it has 64 sites within the park and 42 outside.

2.4.2 Video collection and annotation

Video collection 2006

Two Sony HC15 digital camcorders were fitted on each BRUVS frame (Moore 2010). BRUVS were baited with 800 grams (g) of crushed Australian Sardine (*Sardinops sagax*) suspended 1.2 m in front of the cameras. Each BRUVS drop was separated by 500 m to avoid plume overlap, and each was allowed to soak (record) for 60 minutes. All deployments occurred between 0800 and 1700 local time to optimise light for video analysis and minimise the effects that different types of lighting and light environment can have on diurnal fish behaviour, which may affect the consistency of species counts.

Post-processing of BRUVS footage was completed using a custom video and computer interface developed by the Australian Institute of Marine Science (BRUVS1.5.mdb, Australian Institute of Marine Science, 2006). For each video *MaxN* (maximum number of individuals of a particular species in the frame at any given time) of each species was recorded. To analyse fish lengths, PhotoMeasure stereo-photo comparator software was used (Moore et al. 2010).

Video collection 2021

Two high-definition video cameras (GoPro Hero7 Black) were fitted on each BRUVS frame. The pairs of cameras were mounted 0.7 m apart and angled in at 8 degrees to allow for stereo imaging. Filming in stereo adds the capability of making accurate measurements of individual fish, informing estimates of biomass (Harvey et al. 2003, Langlois et al. 2018). Each BRUVS frame was calibrated in a pool prior to fieldwork commencing. BRUVS were baited with 1 kilogram (kg) of Australian Sardine to attract fish into the field of view for abundance and diversity estimates and length measurements. They were then left to soak on the seafloor for a minimum of 60 minutes.

Post-processing of BRUVS footage was completed using the program EventMeasure (SeaGIS 2020) by trained observers specialised in temperate fish analyses. For each video, the *MaxN* (maximum number of individuals of a particular species in the frame at any given time) was recorded, providing a conservative measure of relative abundance. *MaxN* is a widely recognised way of obtaining fish population data from BRUVS and means that no individual is double-counted (Cappo et al. 2004). The total length of each individual fish counted in the *MaxN* for each species (where applicable) in each video was then measured to within 10 mm precision (with most measurements within 5 mm precision) using standard metrics in EventMeasure (SeaGIS 2020). Weights of individuals of each species were estimated using length–weight relationships obtained from FishBase (Froese and Pauly 2010). Where length–weight relationships were not available for a species, that of a close relative was used. Total relative biomass for each site was calculated as the sum of individual weights of each species. For any individuals that could not be measured at the *MaxN*, the mean weight of that species was assigned. Total species richness, family richness, total relative abundance and total relative biomass were subsequently calculated for each deployment.

Species were identified to the lowest taxonomic level possible. When similar species were indistinguishable from the video footage, they were classified to family or genus level where possible, for example, trevally (*Pseudocaranx* spp.) and butterfly perch (*Caesioperca* spp.). Morid cods were grouped into *Pseudophycis* spp., although may have also been from the genus *Lotella*.

2.4.3 Habitat classification from BRUVS

Broad classifications of habitat were derived from the field of view of each BRUVS deployment in 2006 and 2021, allowing comparison of observations from different habitats. Standard methodology from Langlois et al. (2018) was used to enable direct comparison between studies. To do this, benthic composition and structural relief were classified for each cell in a 20-cell grid (4 rows × 5 columns) overlaid on a still image grab of the site. Classifications were made for the benthic class and relief score that predominated in each cell. A simplified adaptation of the CATAMI classification scheme (Althaus et al. 2015) was used to classify the benthic composition and relief of each site. This method used the TransectMeasure program from SeaGIS for annotations, and output was processed for percentage cover of benthos types using scripts from an open-access GitHub repository (Langlois et al. 2010). For the purpose of this study, each set of 20 classifications from each site was used to derive a single habitat class for that site, along with estimates of the percentage cover for each habitat component. Broad habitat classes were defined as follows:

- urchin barren – primarily consisting of consolidated rock, boulder or cobbles, either bare or covered in encrusting macroalgae. Visible urchins may or may not have been present. Minimal to nil erect or canopy-forming macroalgae present
- circalittoral reef – invertebrate-dominated habitats, primarily occurring at depths greater than 30 m
- infralittoral reef – shallow (less than 30 m) habitats with erect or canopy-forming macroalgae as the dominant habitat type
- soft sediment – unconsolidated sand or pebble habitats.

Relief type was also classified as it is representative of complexity or the height and angle of substrate. A mean relief score was provided for each BRUVS location. Distinct categories were adapted from Wilson et al. (2007), including:

0. Flat substrate, sandy, rubble with few features. ~0 substrate slope.
1. Some relief features amongst mostly flat substrate/sand/rubble. <45 degree substrate slope.
2. Mostly relief features amongst some flat substrate or rubble. ~45 substrate slope.
3. Good relief structure with some overhangs. >45 substrate slope.
4. High structural complexity, fissures and caves. Vertical wall. ~90 substrate slope.
5. Exceptional structural complexity, numerous large holes and caves. Vertical wall. ~90 substrate slope.

2.4.4 Statistical analysis of the BRUVS data

Univariate and multivariate statistics

To understand drivers of differences in fish assemblages, univariate (single variable) and multivariate (multiple variable) analyses of differences in fish assemblages were carried out using PRIMER v7 (Clarke and Gorley 2015) with the PERMANOVA+ add-on (Anderson et al. 2008).

Within each year, all sites were assessed for the influence of habitat type (fixed factor, 4 levels: urchin barren, circalittoral reef, infralittoral reef, soft sediment) and for the 2021 dataset for the factor status (fixed factor, 2 levels: inside the MPA, outside the MPA). When significant differences were detected, pairwise tests were used to distinguish between levels of the factor with Monte Carlo adjustments where unique permutations were below 100. For comparison of assemblages between years, the sites that geographically matched to within 100 m were included, resulting in 36 sites across years. This matched dataset was analysed using the factors of year (random factor, 2 levels: 2006 and 2021) and habitat type.

For univariate analyses, Euclidean distance matrices were constructed for total relative abundance (*MaxN* summed across species per deployment), species richness and total biomass (summed across species able to be measured per deployment).

For the multivariate assemblage data (i.e. the abundance of all observed species for each deployment for both seasons), a Bray–Curtis similarity matrix was constructed on standard dispersion (by habitat) weighted data to account for the schooling nature of some fish species (Clarke et al. 2006). Shade plots were viewed as a guide for the appropriate level of transformation (Clarke et al. 2014). Data were portrayed visually using canonical analysis of principal coordinates (CAP) ordination plots. CAP is a specific hypothesis-testing analysis which aims to find differences between levels of a chosen factor. Along with a PERMANOVA on the above factors, CAP analysis was used to

test for differences in fish assemblages between habitats across and within years. As part of the CAP routine, a leave-one-out allocation test, which assigns each deployment to a factor level based on similarity values, is also conducted. The success rate of this test (i.e. how many deployments were assigned to the correct level) can then be reported. SIMilarity PERcentages (SIMPER) analysis was used to assess species contributions to the observed assemblages and differences across factors.

The influence of environmental variables (percentage cover of habitat components, mean relief and depth) on the variation observed in the fish assemblages data was assessed using distance-based linear models (DISTLM) (McArdle and Anderson 2001). This approach seeks to analyse and model the relationship between one or more predictor variables and the resemblance matrix of a desired biotic dataset. Draftsman plots were first viewed to assess for potential inter-correlations between environmental variables; one of a pair of variables that had strong correlations (>0.7) was excluded from further analysis. Combinations of variables were tested using the 'best' selection method with AICc (Akaike information criterion (AIC) with correction for small sample sizes) being used to select the final model. This method tests all combinations of variables for each number of variables (i.e. the best combination for one variable, 2 variables and so on) to determine the most parsimonious model that incorporates the 'best' combination of variables (Anderson et al. 2008). This 'best' combination of variables was then plotted as vector overlays over the multivariate assemblage using a constrained ordination plot created using a distance-based redundancy analysis (dbRDA) performed with the best model (Anderson et al. 2008).

Fish modelling approach

Predictive models of fish richness, biomass and abundance increase the coverage of observational data from BRUVS surveys by using seafloor characteristics to predict richness, biomass and abundance across the entirety of the marine national park. The machine learning technique random forests (RF) was selected because it provides good predictive performance with noisy data (Díaz-Uriarte and De Andres 2006) and reduces the risk of over fitting (Breiman 2001). Additionally, RF models have been used to model various species richness distributions, and they show high performance when compared to other machine learning methods (Dai et al. 2020, Li et al. 2017, Olaya-Marín et al. 2013). Although species richness values are count data, using the RF model to predict 28 classes across the state would result in low class accuracies for rare classes. To overcome this, species richness was treated as a continuous variable for the machine learning approach. RF models were created using the 'randomForest' package in R (Liaw and Wiener 2002). Predictive species richness and abundance maps were created in ArcGIS Pro using the spatial statistics package 'Forest-based Classification and Regression' tool, which is based on the random forest algorithm from Breiman (2001). This is the same algorithm used in the 'randomForest' package in R. Variables were selected through the 'variable importance' measurement values retained by the 'randomForest' model in R. Any variables with negative importance were omitted from the model. Correlation between variables was checked using Pearson's cross-correlation values. If any variables had a Pearson's cross-correlation value below -0.7 or above 0.7 , the variable with the lower importance was dropped from the model.

For use in this study, a high-resolution (1 m horizontal resolution) bathymetric grid was obtained for the Cape Howe MNP. Four derivatives of bathymetry were calculated from this grid to further our understanding of the species distribution models: bathymetry, slope, rugosity (surface roughness) and distance to reef.

For all sites, derivatives characterising the seabed terrain were extracted using the 'raster' and 'spatialEco' packages from the programming language R. Derivatives used included bathymetry,

standard deviation of depth, slope, aspect (northness and eastness), vector ruggedness measure (VRM) and complexity (Table 2.1).

Table 2.1: Description of bathymetric variables tested in this study. All derivatives of bathymetry were made using the ‘Benthic Terrain Modeller’ tool for ArcGIS (Walbridge et al. 2018)

Derivatives	Description
Bathymetry	Elevation of a plane passed through its closest grid point.
Rugosity	A measure of topographic roughness that relates ratio of surface area to planar area.
Slope	Maximum change in elevation between each cell and cells in a specified surrounding neighbourhood.
MPA status	Within the MPA or outside the MPA.
Distance to reef	Distance to reef calculated using the ‘Euclidean Distance’ tool in ArcGIS Pro.

2.5 Fisheries-independent Southern Rock Lobster survey

Rock lobster have previously been identified as an essential indicator of the health of subtidal reefs, as well as a proxy for the effect of poaching in marine national parks and sanctuaries. In Cape Howe MNP, knowledge of rock lobster abundance is also important for understanding the predatory controls on urchin barrens within the park. Rock lobster surveys can further offer valuable information about ‘secondary’ ecosystem services provided by the park to the rock lobster fishery, as well as informing ecosystem-based management of the fishery. Southern Rock Lobster (SRL) have previously been surveyed as part of the Subtidal Reef Monitoring Program; however, it has been difficult to accurately sample rock lobster populations as monitoring sites rarely overlapped with prime rock lobster habitat. As an alternative, this study used standardised fishery assessment trapping methods to provide fine-scale SRL population information within and adjacent to Cape Howe MNP. Lobster pots were baited with 1 kg of locally available bait and escape gaps were wired shut (Woods and Edmunds 2013, Young et al. 2016, Ierodiaconou et al. 2018). A total of 145 pots were deployed during this survey: 85 placed inside the park and 60 outside. For each day’s sampling, pots outside the park were deployed in areas of similar habitat to those deployed inside the park to accurately test the effect the park has on the local SRL population. All captured SRL were counted and sexed, females were assessed for reproductive condition, and all lobsters were measured for carapace length (CL). Bycatch were recorded for each pot location before promptly being returned to the water, and genetic samples of SRL were stored in ethanol for future analyses. To calculate SRL biomass, we used the length–weight relationship provided in Punt (2003): $W = a CL^b$, where W is the weight in kilograms, CL is carapace length and a and b are coefficients related to sex and size class (females: $a = 0.000271$, $b = 3.135$; males: $a = 0.000285$, $b = 3.114$). For each pot lift, a pre-recruit index was developed by calculating the percentage of lobsters below the sex-specific legal size limits for this region (females: $CL = 105$ mm, males: $CL = 110$ mm) (Bentley et al. 2005).

The purpose of this SRL survey was to compare populations inside and outside the MPA and determine the role of environmental drivers on the size, weight, sex and number of SRL caught. Generalised additive models (GAMs) were further used to investigate the effect of environmental variables on SRL at this site. GAMs were run using various groupings of abundance and biomass (sex and legal size limits) as response variables. Deployments from inside and outside the MPA were included in a single model ($n = 145$ deployments) to determine differences inside and outside the park. We randomly selected 75% ($n = 101$) of deployments for training the models, reserving the

remaining 25% ($n = 44$) for testing the accuracy of predictions. Root mean squared error (RMSE) was used to assess the accuracy of the predicted data compared with observed data.

2.6 Modelling of urchin barren coverage

To model the distribution of urchin barrens in Cape Howe MNP and the area adjacent to it, we first collated all available ground truthing datasets collected since 2006 in the region, including the habitat observations from the BRUVS used in this study. From these datasets, we extracted all points that were associated with urchin barrens. We then extracted the habitat variables associated with each of those points, including depth, distance to reef, protection status (inside or outside the MPA) and slope.

Once all the data points and their environmental information were collated, we used the SDMtoolbox in ArcGIS 10.7.1 to run a maximum entropy (MaxEnt) model. MaxEnt is an ecological method for creating predictive distribution maps based on presence-only data. This method was chosen because the time series of data and lack of recent spatial coverage in ground truthing data do not allow us to adequately determine locations of true urchin barren absence. Using the 'MaxEnt' tool in SDMtoolbox, we ran a model to determine the probability of occurrence of urchin barrens across the study area. Model performance was assessed using the area under curve value with values of 0.5 indicating a model is no better than random and values greater than 0.75 indicating the 'best' model. Within the SDMtoolbox, a spatial jackknifing procedure automatically chooses the best model and also quantifies the explanatory power of each environmental variable.

3. Results

3.1 Trends in sea-surface temperature

Figure 3.1 shows summer, winter and annual sea-surface temperatures (SSTs) from 1985 to 2018. The trend analyses on the annual and seasonal patterns in SST show that all seasons are experiencing an increasing trend in temperature, although winter is very close to experiencing no trend (Table 3.1, Figure 3.2).

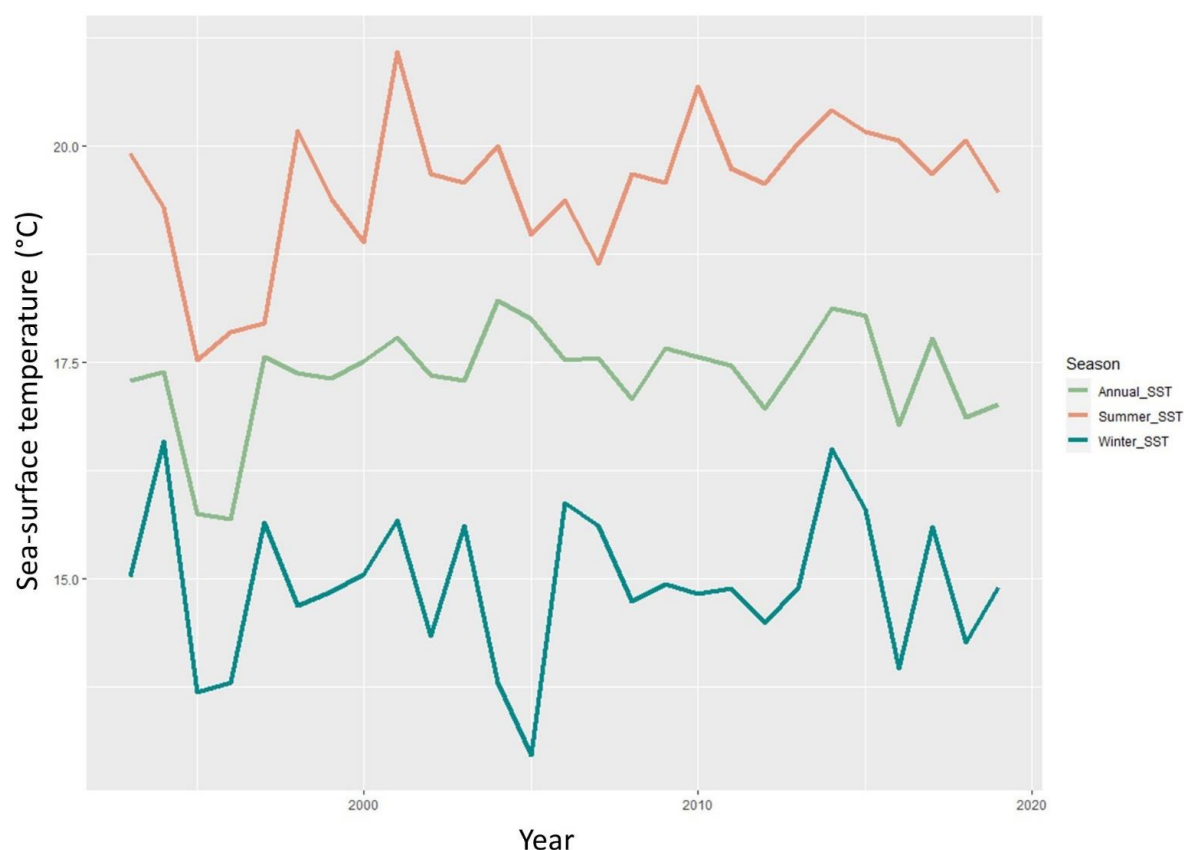


Figure 3.1: Annual and seasonal sea-surface temperatures within Cape Howe MNP

Table 3.1: Results from the state-space model used to assess trends in temperature. Maximum likelihood estimate (the trend term – positive values mean increasing trends), the standard error of that estimate, and the lower and upper confidence intervals associated with those trends

Season	Maximum likelihood estimate	Standard error	Lower CI	Upper CI
Annual	0.0139	0.0156	-0.0168	0.0445
Summer	0.0624	0.0567	-0.0486	0.1730
Winter	0.0001	0.1664	-0.0158	0.0161

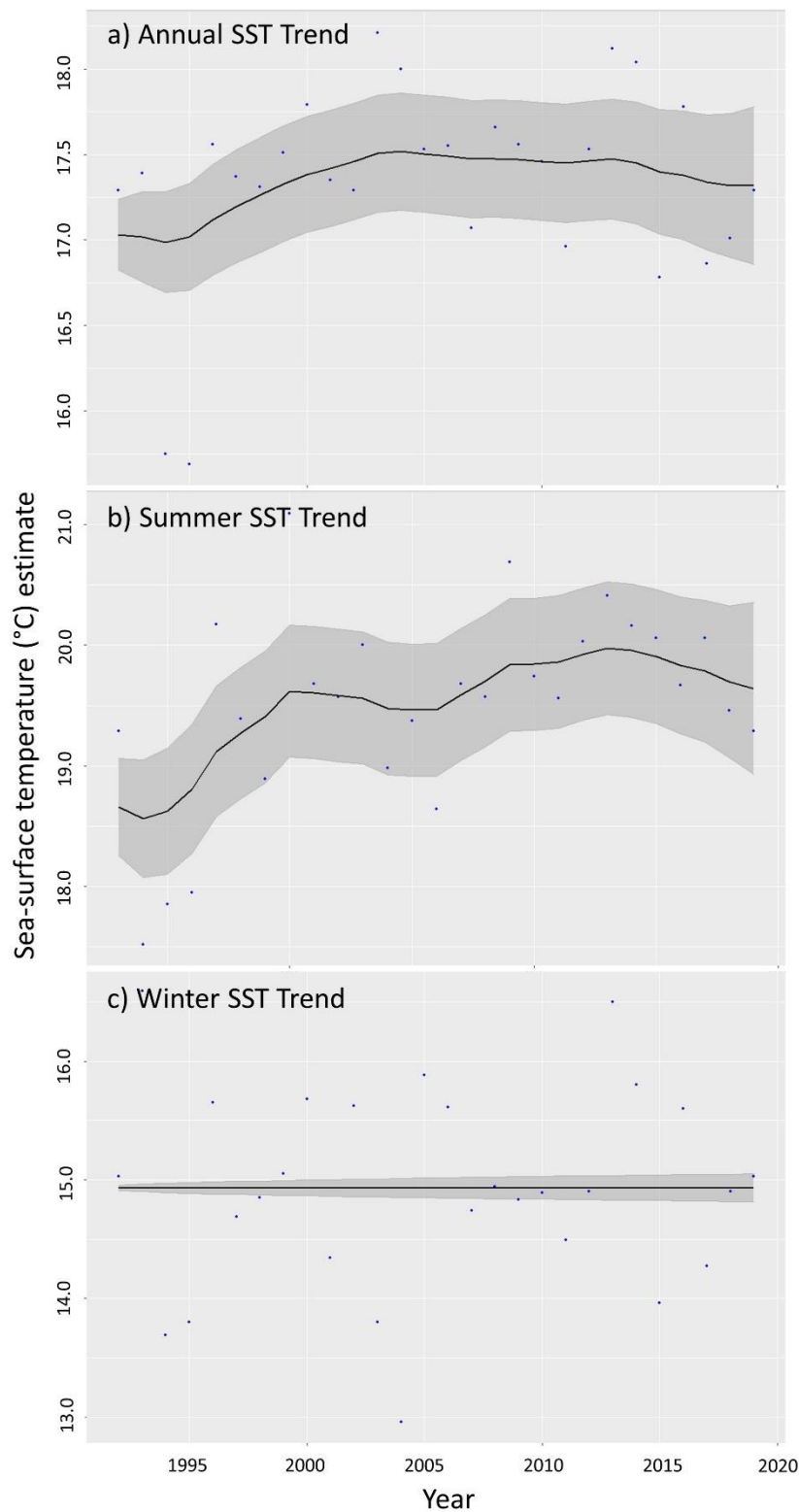


Figure 3.2: The (a) annual, (b) summer and (c) winter sea-surface temperature values (points) and the model-estimated SST (solid line). The solid lines are the means of the stochastic levels; the confidence intervals of that model are shown as 2 standard deviations (the shaded area)

3.2 Reef Life Survey for shallow subtidal reefs

When combined with the previous SRMP program, data from shallow subtidal reefs in Cape Howe MNP now span over 20 years. A total of 48,659 observations of 80 individual species of fish and 11,076 observations of 56 individual species of invertebrates and cryptic fish were made from RLS data capture over 13 years from 2008 to 2021 (Table 3.2). In addition 5,799 points were classified for benthic habitat from a total of 1,116 photo quadrats, allowing canopy cover metrics to be extracted for comparison with previous SRMP data. Common species of fish observed across years included Mado Sweep (*Atypichthys strigatus*), Eastern Hulafish (*Trachinops taeniatus*), One-spot Puller (*Chromis hypsilepis*) and Maori Wrasse (*Ophthalmolepis lineolata*) (Figure 6). Common species of invertebrates and cryptic fish observed across years included Longspined Sea Urchin (*Centrostephanus rodgersii*), Blacklip Abalone (*Haliotis rubra*) and Tent Turban (*Astridium tentoriiforme*) (Figure 3.4).

There is a lot of variability in habitat forming species between years; however, there are some trends like the decline of Golden Kelp (*Ecklonia radiata*) since 2008 (Figure 3.5). Crustose coralline algae has been present in all years except 2008. Bare rock had its largest proportions of observations in 2009 and 2015. Crayweed (*Phyllospora comosa*) was observed in 2009, 2011 and 2021 with at least 25% coverage for each year it was observed. Filamentous brown algae was observed in 2008, 2009 and 2011 but not in 2014, 2015 or 2021.

Table 3.2: Summary of Reef Life Survey sampling and observations across all years

	2008	2009	2011	2012	2014	2015	2021	Total
Number of RLS sites completed (number of SRMP sites completed)	1	2 (9)	9	1	1 (8)	2	6	22
Area surveyed for fish (m²)	1,000	2,000	9,000	1,000	1,000	2,000	6,000	22,000
Area surveyed for invertebrates (m²)	200	400	1,800	200	200	400	2,400	5,600
Number of photo quadrats classified for benthic habitat	40	54	992	0	20	60	0	1,166
Number of points classified for benthic habitat	200	270	4,960	0	99	300	0	5,799
Number of fish observations [observation values of schooling fish; typically estimates by divers]	340	225	39,128 [28,533]	243	866	1,557	6,300	48,659
Number of fish species observed	24	19	62	22	24	36	46	80
Number of invertebrates and cryptic fish observations	27	550	8,327	116	127	529	1,400	11,076
Number of invertebrate and cryptic fish species observed	3	15	43	4	5	24	24	56

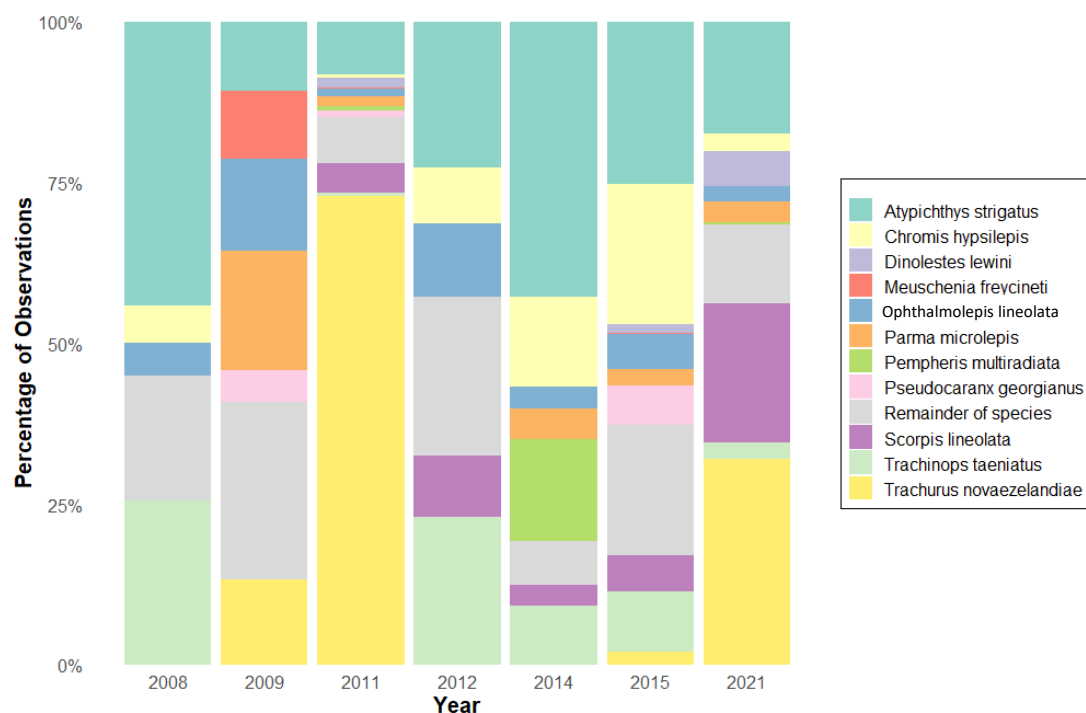


Figure 3.3: Proportions (as percentages) of the total observations of fish made in Cape Howe Reef Life Surveys between 2008 and 2021. 'Remainder of species' combines all species whose proportion of the total observations is less than 4%

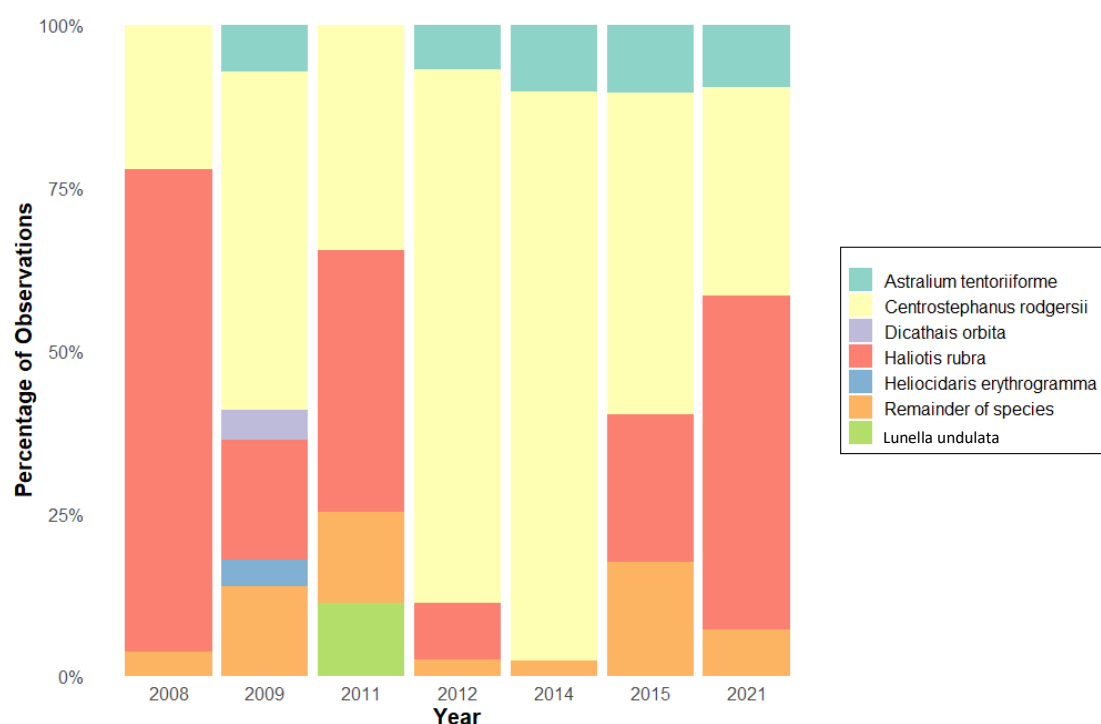


Figure 3.4: Proportions (as percentages) of the total observations of invertebrates made in Cape Howe Reef Life Surveys between 2008 and 2021. 'Remainder of species' combines all species whose proportion of the total observations is less than 4%

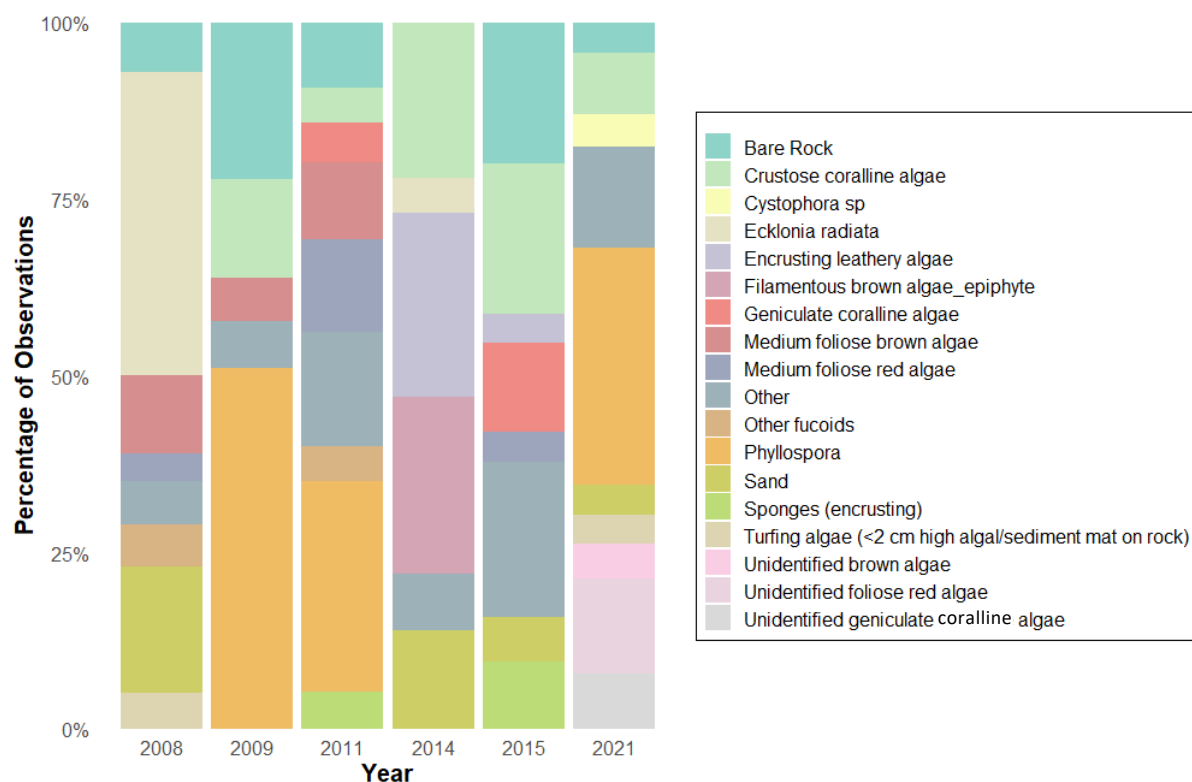
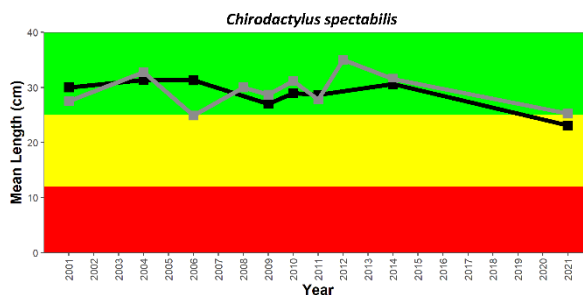
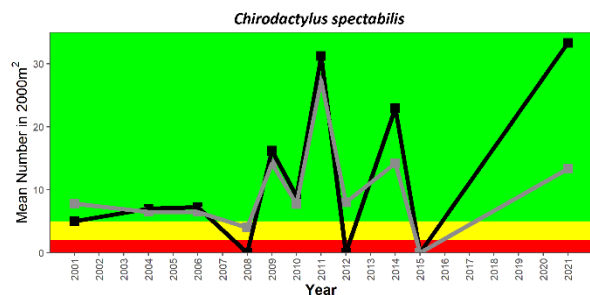
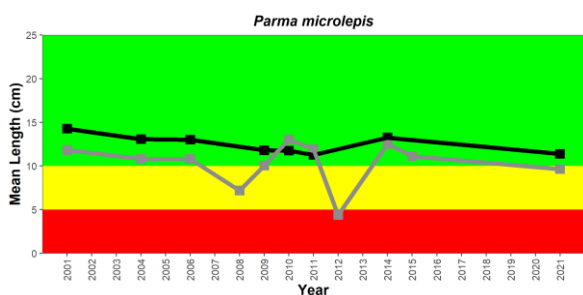
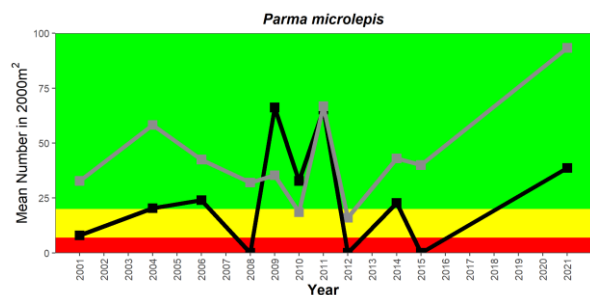
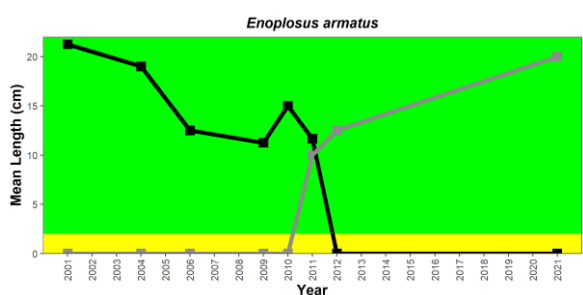
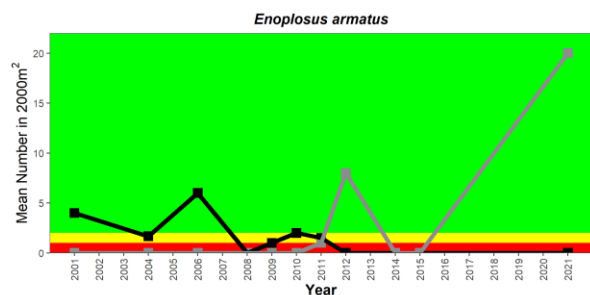
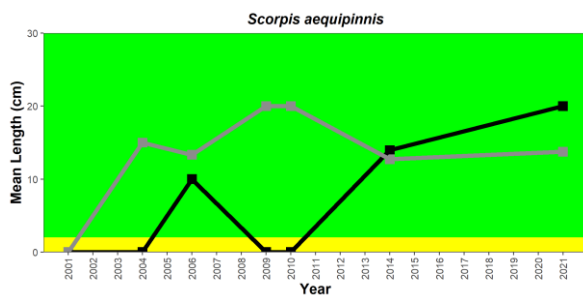
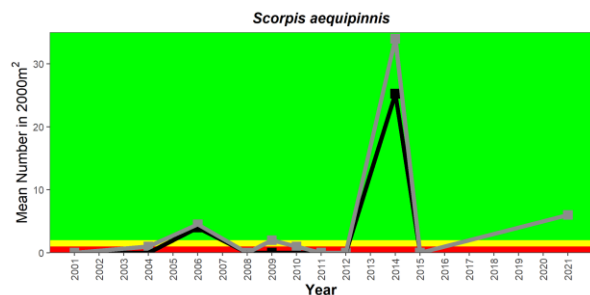
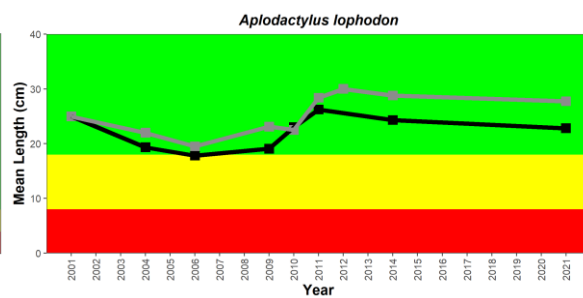
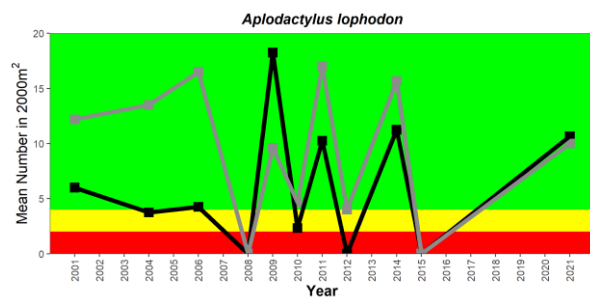
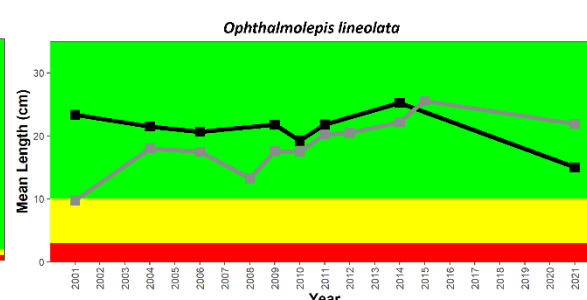
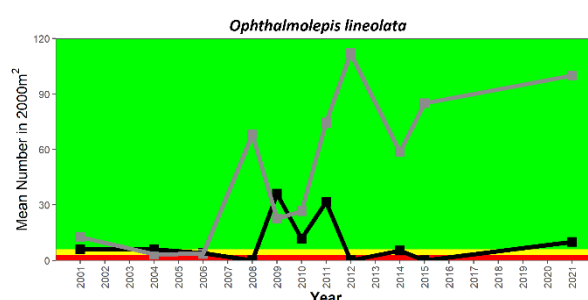
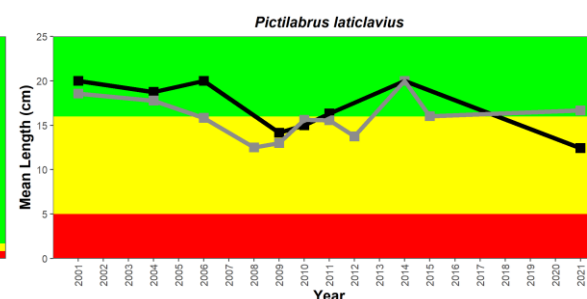
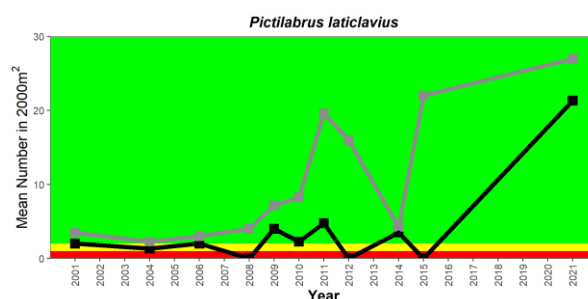
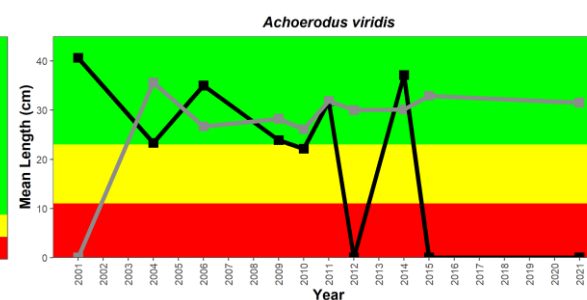
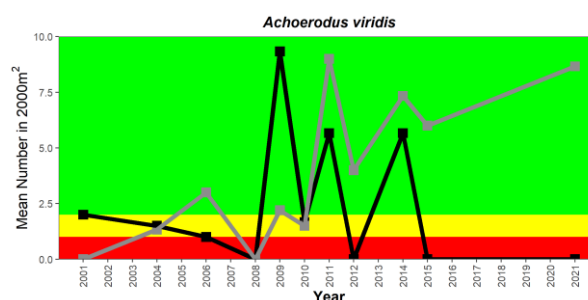
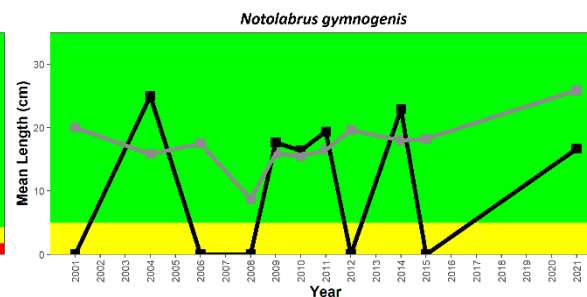
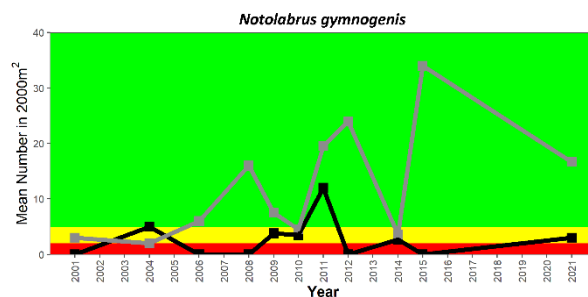
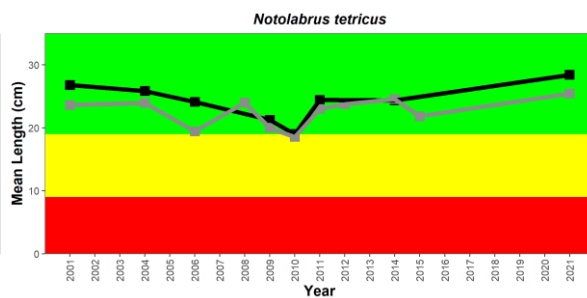
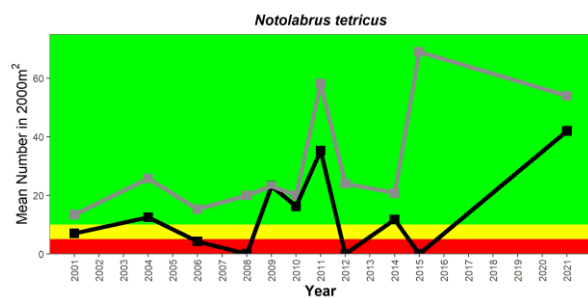


Figure 3.5: Proportions (as percentages) of the total observations of habitat made in Cape Howe Reef Life Surveys between 2008 and 2021. Other represents all species with a proportion of the total observations less than 4%

3.2.1 Key mobile fish species

Control charts indicate that the abundance of all key mobile fish species fall within the zone of 'good' condition inside the MNP, with the exception of 4 species (Figure 3.6). Old Wife (*Enoplosus armatus*) abundance is currently below the lower control limit because they have not been observed within the MPA since 2011. Crimsonband Wrasse (*Notolabrus gymnogenis*) abundance within the MPA is currently above the lower control limit but below the lower limit of acceptable change. Eastern Blue Groper (*Achoerodus viridis*) has not been observed within the MPA since 2015 and Six-Spine Leatherjacket (*Meuschenia freycineti*) has not been observed within the MPA since 2012. The mean length of individual fish species was generally similar between MPA and reference sites over time, with 6 out of the 12 species above the lower limit of acceptable change, and the other 6 within or below the lower limit of acceptable change. The lack of observations within the MPA for *E. armatus*, *A. viridis*, and *M. freycineti* means that all fell below the lower control limit for size. White-Ear (*Parma microlepis*) size fell below the lower limit of acceptable change in reference sites. During the most recent survey, Banded Morwong (*Chirodactylus spectabilis*) and Senator Wrasse (*Pictilabrus laticlavius*) fell below the lower limit of acceptable change for mean length within the MPA.





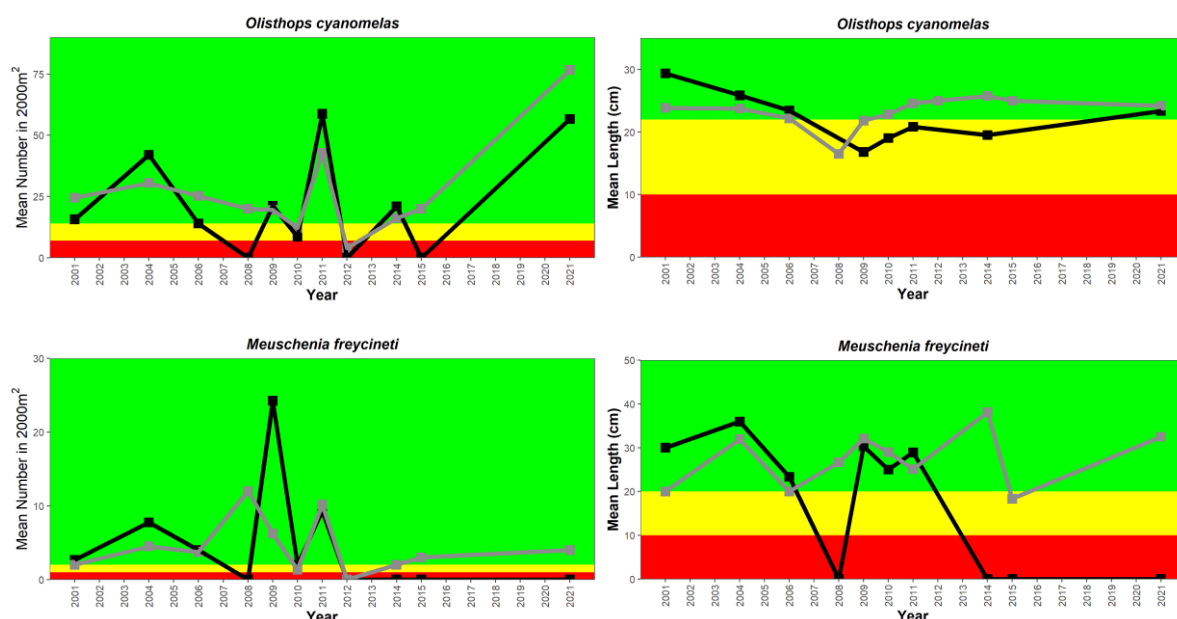
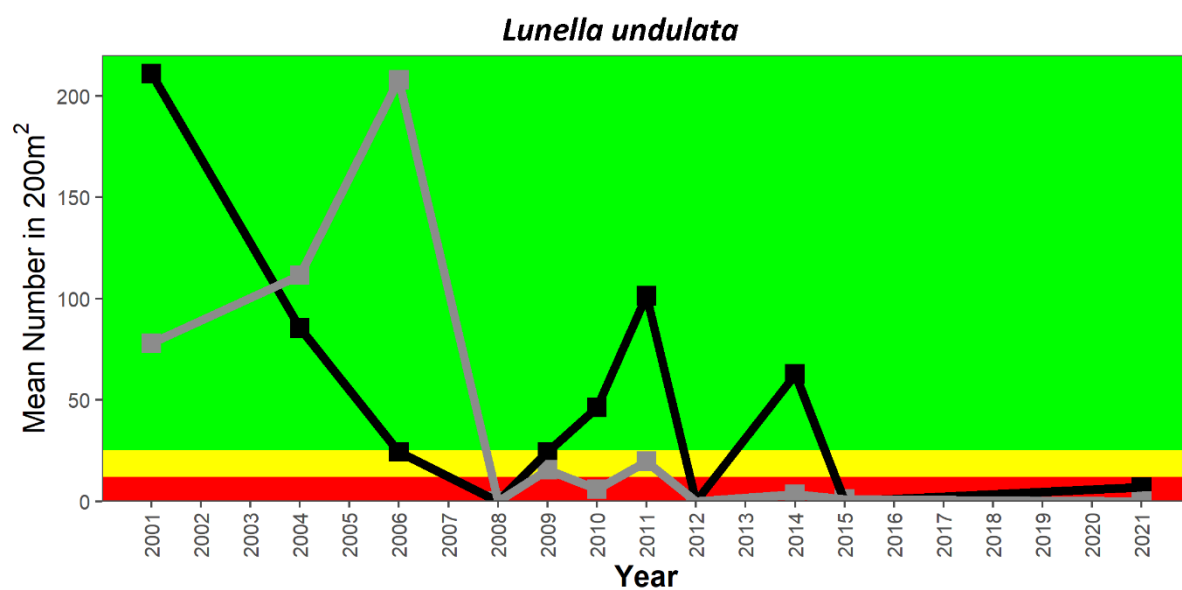
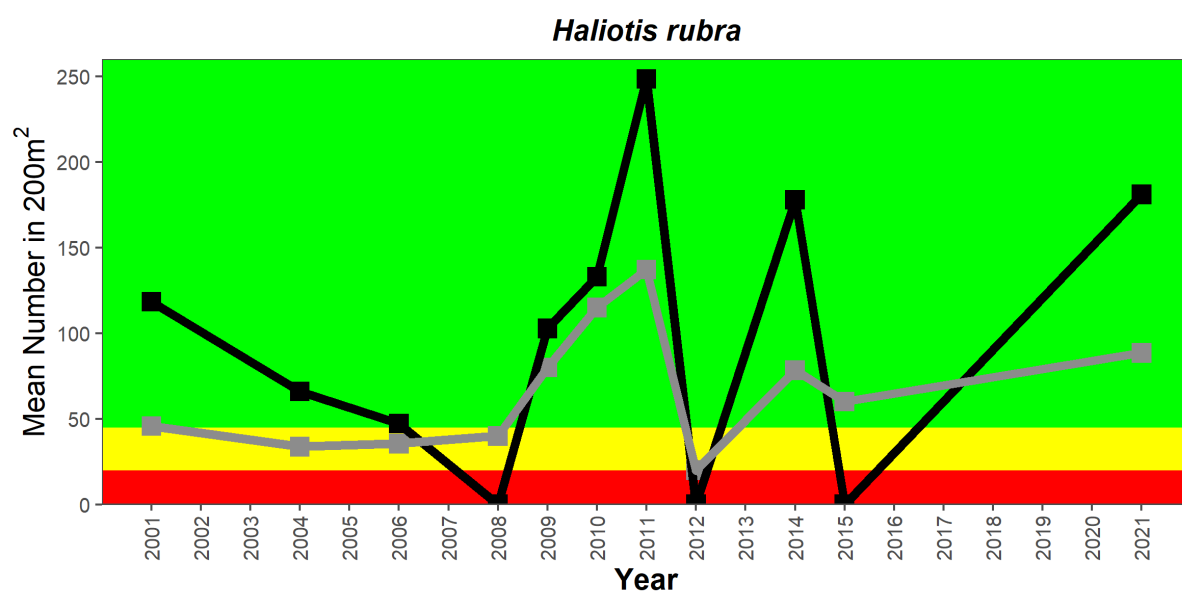
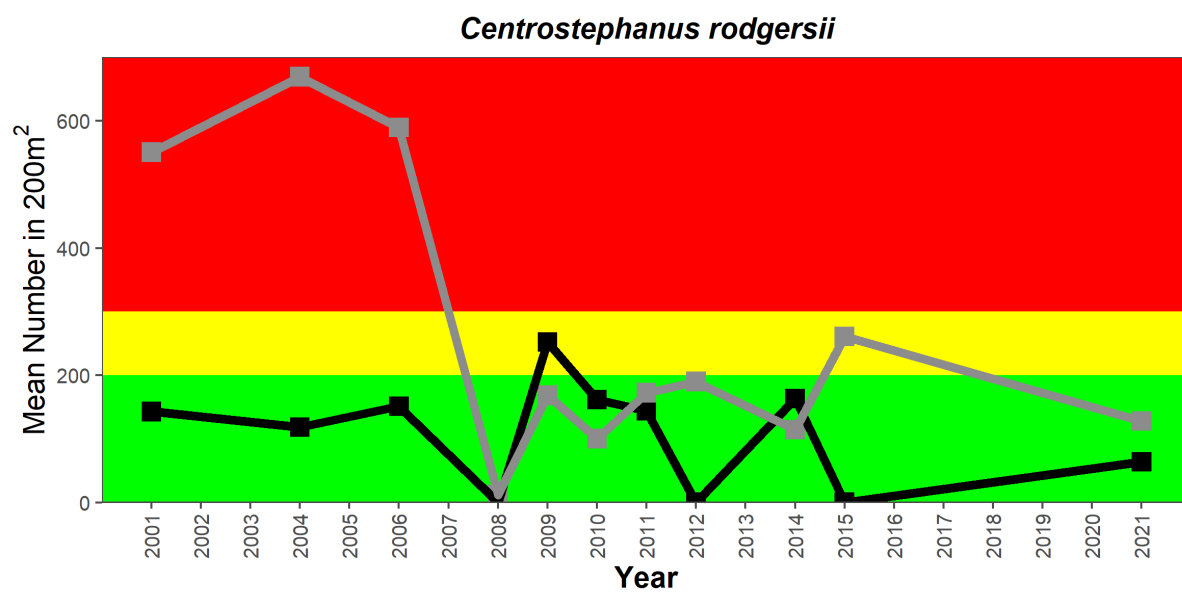
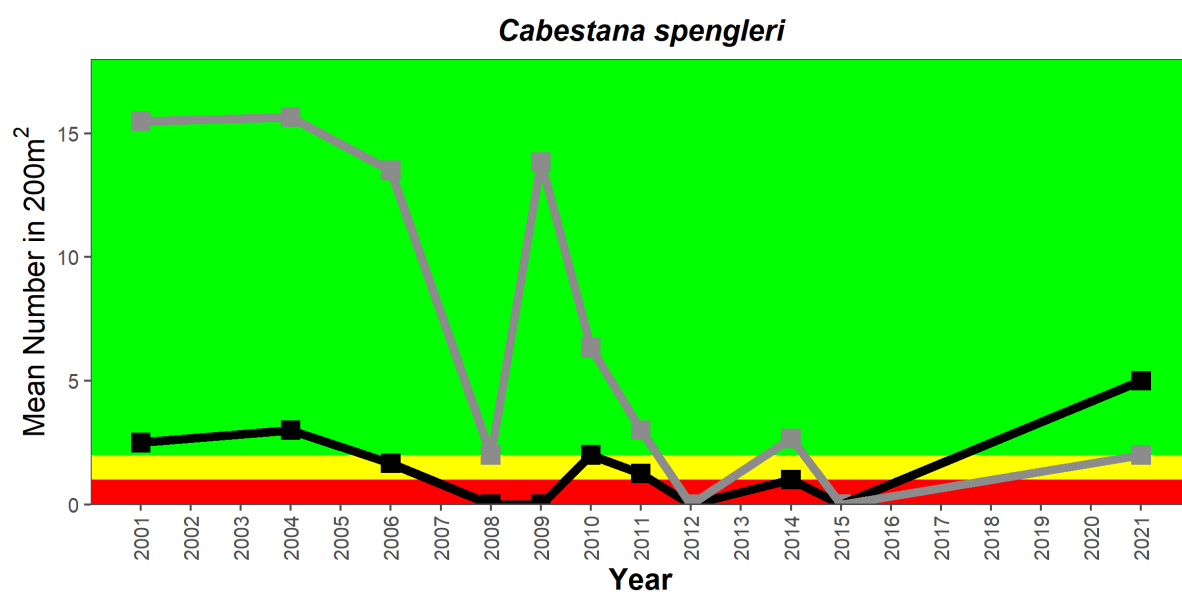
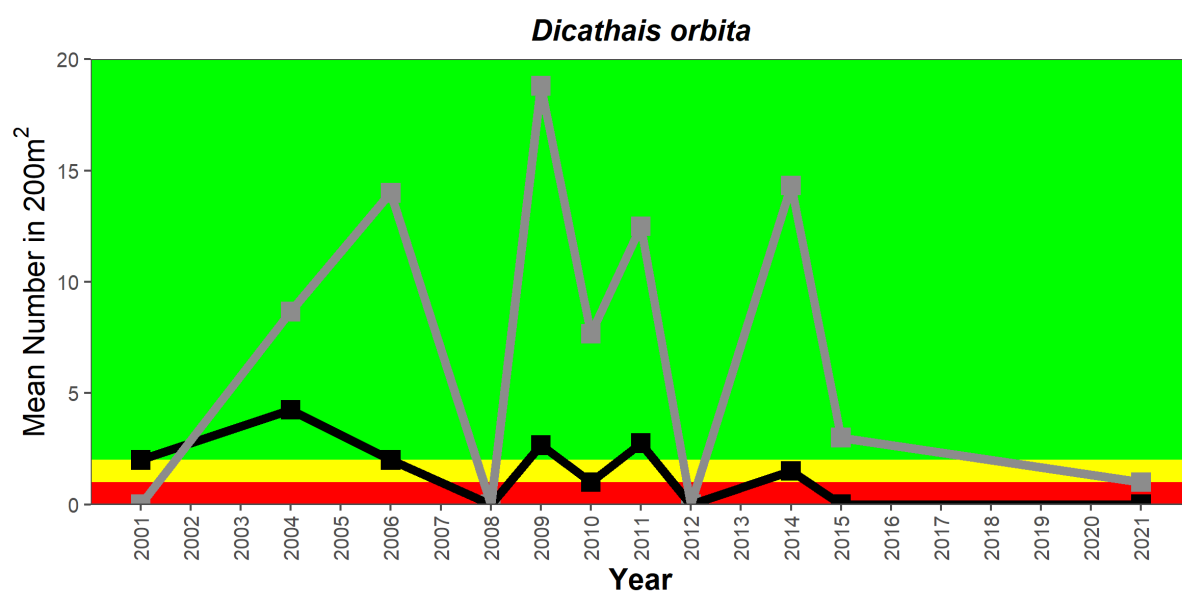
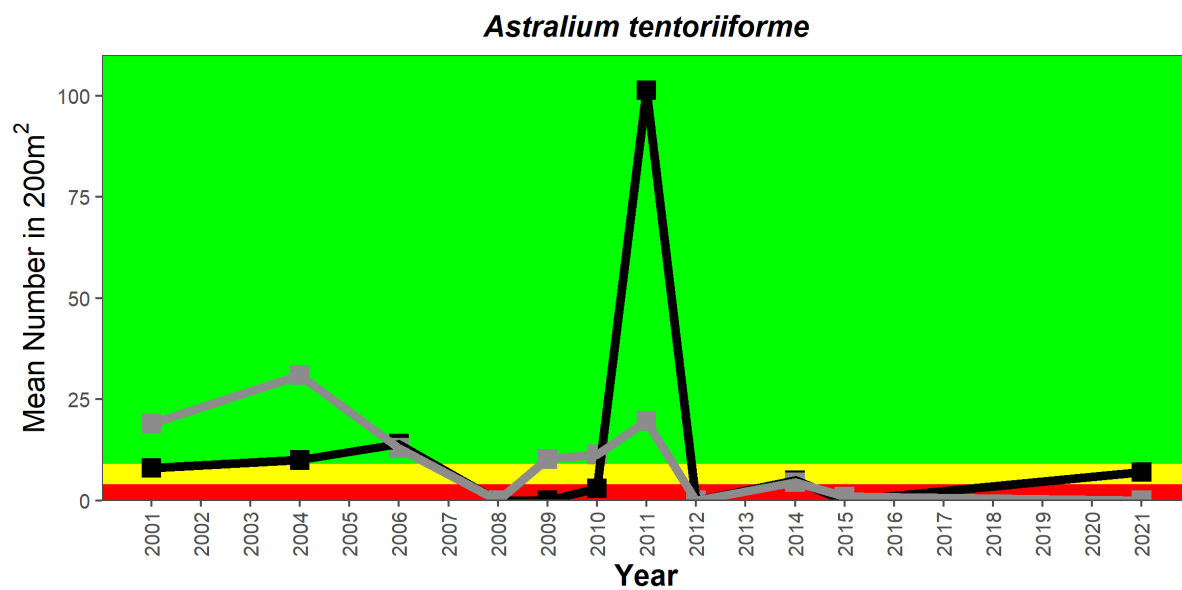


Figure 3.6: Control charts showing change in abundance and length of key mobile fish species within Cape Howe MNP (black line) and at reference sites outside the park (grey line). Observations for these charts were sourced from SRMP and Reef Life Survey diver transects. These charts show the lower limit of acceptable change (LLAC, top of yellow band – set as the minimum value inside the MPA from SRMP surveys from 1998 to 2002) and lower control limit (LCL, top of red band) based on the variation from surveys, which indicate the level at which conditions are sufficiently poor that some management response is required. Surveying effort was relatively low in 2008, 2012 and 2015, which could result in more records of no sightings for some species

3.2.2 Key macroinvertebrates

The current (2021) status of Longspined Sea Urchin (*Centrostephanus rodgersii*), Triton Shell (*Cabestana spengleri*), and Blacklip Abalone (*Haliotis rubra*) was found to be ‘good’ condition. Triton Shell and Blacklip Abalone were well above the lower control limit and Longspined Sea Urchin, which is considered undesirable at high densities as it can create urchin barrens and reduce macroalgal cover, was in the zone of ‘good’ condition (i.e. low densities) (Figure 3.7). However, consistent declines over the past 15 years were observed in Turban Shell (*Lunella undulata*) and Purple Sea Urchin (*Heliocidaris erythrogramma*). *Lunella undulata* has been below its lower control limit of 23 individuals per 200 m² since 2008 at reference sites and since 2015 within the MPA; from 2001 to 2006, the average abundance inside the MPA reduced from 214 to 23. A similar trend was also seen for *H. erythrogramma*, with its highest abundances within the MPA occurring between 2001 and 2006. Since then, with the exception of 2009, it has been below its lower limit of acceptable change of 7.5 individuals per 200 m². Abundance of Dog Whelk (*Dicathais orbita*) within the MPA has been on the decline since 2006 and is currently below its lower limit of acceptable change of 2.1. Tent Turban (*Astraliu tentoriiforme*) abundance was highest in 2011, with the other years showing relatively low values. In 2021 there were no observations of *Astraliu tentoriiforme* at reference sites, and the abundance within the MPA was above the lower limit of acceptable change.





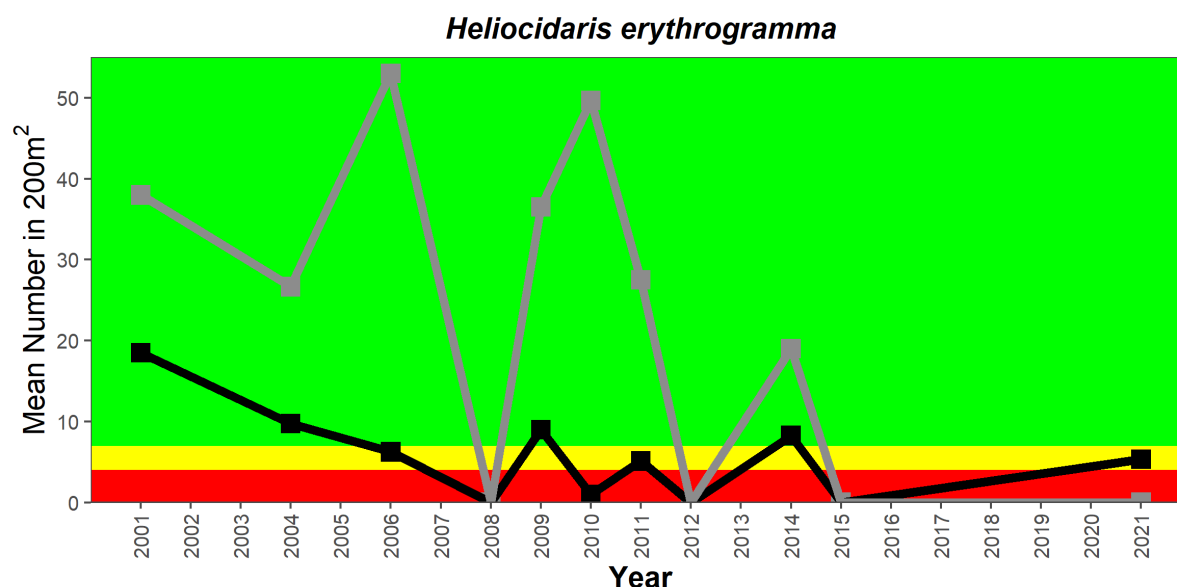


Figure 3.7: Control charts showing change in abundance of key motile macroinvertebrate species within Cape Howe MNP (black line) and at reference sites outside the park (grey line). Observations for these charts were sourced from SRMP and Reef Life Survey diver transects. These charts have a lower limit of acceptable change (LLAC, top of the yellow band – set as the minimum value inside the MPA from SRMP surveys from 2001 to 2008) and lower control limit (LCL, top of red band) based on the variation from surveys, which indicate the level when conditions are sufficiently poor that some management response is required. Surveying effort was relatively low in 2008, 2012 and 2015, which could result in more records of no sightings for some species

3.2.3 Macroalgal beds

The most notable change in the subtidal reef macroalgal communities of Cape Howe MNP has been the decline of encrusting coralline algae within the MPA since 2014 (Figure 3.8); it was below the lower control limit in the park in the 2008, 2015 and 2021 surveys. The other species of interest, Crayweed (*Phyllospora comosa*), had been relatively stable until 2015, when it was not observed. Since 2015, *P. comosa* has been on a positive trend upwards with mean percentage cover in the MPA well above the lower limit of acceptable change in 2021 and just below the lower limit of acceptable change at the reference sites. Although present, *Ecklonia radiata* was not present in high enough abundance in these shallow transects to determine patterns.

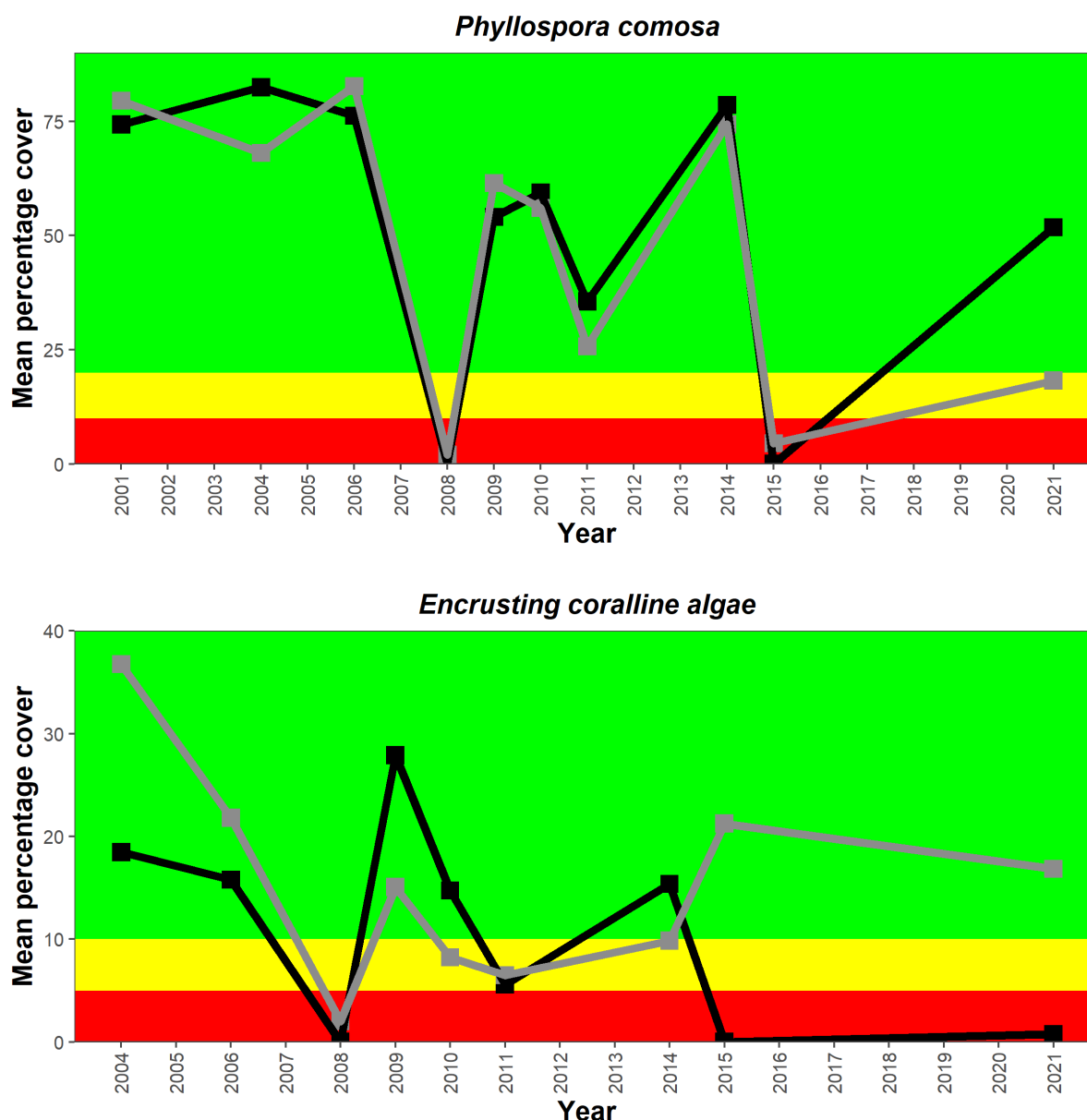


Figure 3.8: Control charts showing change in percentage cover of algal species within Cape Howe MNP (black line) and at reference sites (grey line). These charts show the lower limit of acceptable change (LLAC, top of yellow band – set as the minimum value inside the MPA from SRMP surveys from 1998 to 2002) and lower control limit (LCL, top of red band) based on the variation from surveys, which indicate the level at which conditions are sufficiently poor that some management response is required

3.3 Baited remote underwater video stations

3.3.1 Habitat from BRUVS

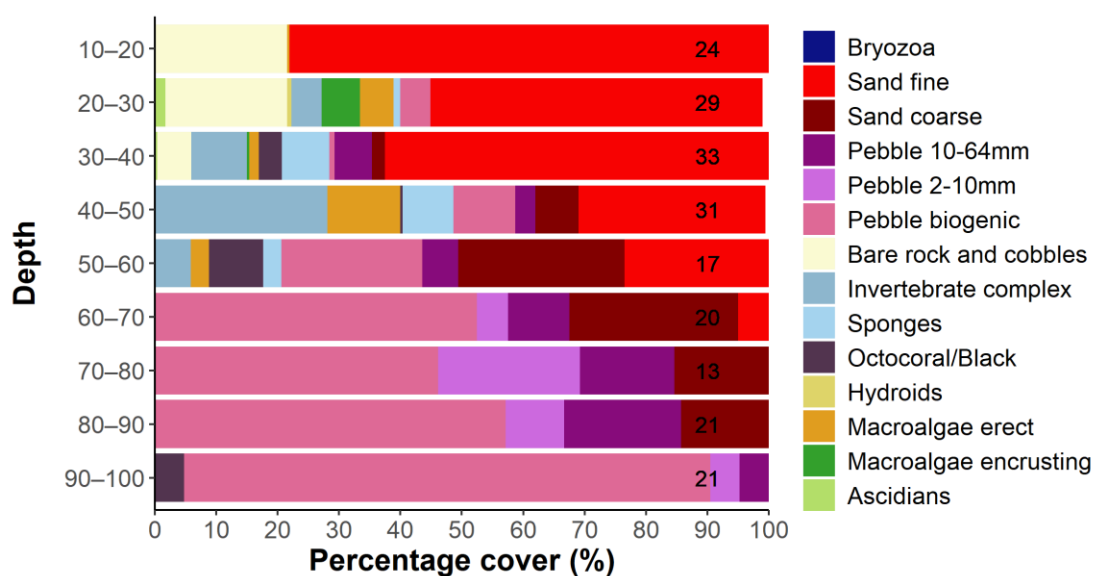
Habitat types changed with depth across the sampling area (Figure 3.9). In 2006, shallow sites (less than 40 m) were dominated by fine sand with a moderate component of bare rock with some invertebrate complex and erect macroalgae. The middle depth zones (40 to 60 m) showed a transition from fine sand to habitats dominated by biogenic pebble and coarse sand. These depths

also had the highest cover of sponges, erect macroalgae and invertebrate-dominated habitats. Sites at depths greater than 60 m were predominantly low-relief soft-sediment habitats consisting of coarse sand and various forms of pebbles. Similar trends were observed for all sites in 2021; however, fine sand was less predominant and the cover of bare rock and encrusting macroalgae increased in shallow areas. There was also greater prevalence of octocorals/black corals in the middle depth zones and extending into deeper areas (up to 80 m) in 2021.

For the matched sites across both years (Figure 3.10), there was a shift in classification from bare rock cover to encrusting algae in the lowest depth stratum (10 to 20 m). The prevalence of sponge habitat and coarse sand in the 50 to 60 m stratum also increased in 2021. Deeper depth strata in 2006 were almost entirely dominated by biogenic pebble substrate whereas in 2021 they were dominated by biogenic pebbles, coarse sand and larger pebbles.

Despite the larger number of samples analysed in 2006 (around twice the effort of that in 2021), the number of barrens observed overall (17) was similar to the number observed in 2021 (21) (Figure 3.11). The fact that a similar number of barrens was observed despite a reduced sampling effort in 2021 could indicate that barren areas had expanded, but this does not appear to be the case. The similarity between the 2006 and 2021 counts is likely due to the limited amount of reef habitat available in the sampling area; the bigger sampling effort in 2006 would not have included a greater area of reef habitat. Also, some additional barren reefs were detected in 2021 because the BRUVS surveys were extended beyond the marine national park, which included additional barren reefs. Thus, the slight increase from 17 to 21 does not reflect an increase in barren area inside the park. In 2021, two barrens were observed at 40 to 50 m depth; no barrens were observed at that depth in 2006.

a)



b)

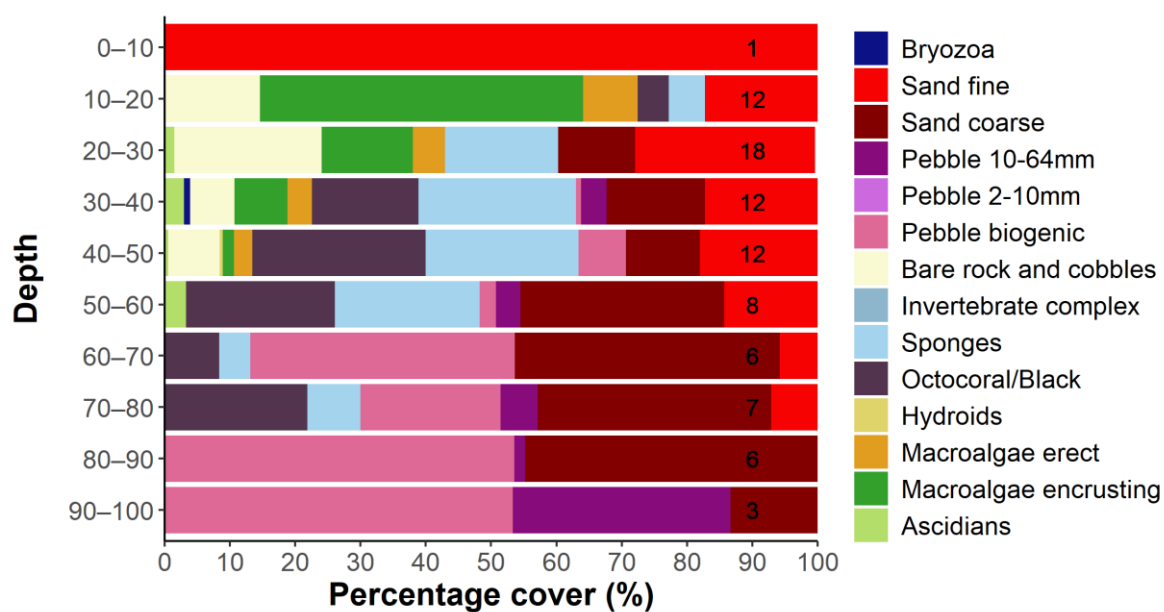
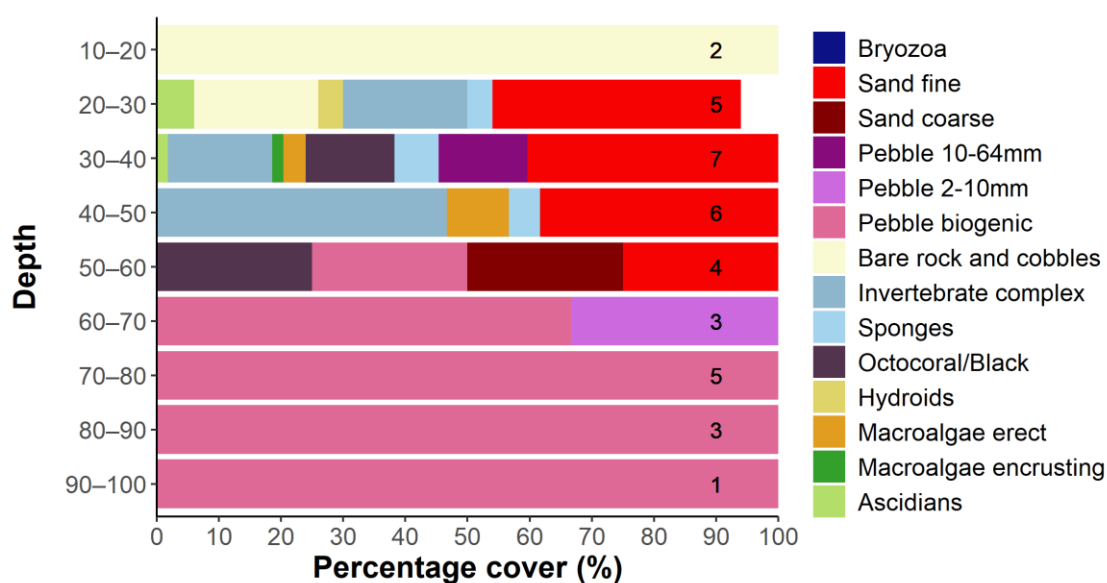


Figure 3.9: Percentage cover of different habitat components calculated from the field of view of each BRUVS deployment for all sites in (a) 2006 and (b) 2021. The percentage cover of ‘unknown’ is not shown. The numbers in each bar indicate the number of samples included

a)



b)

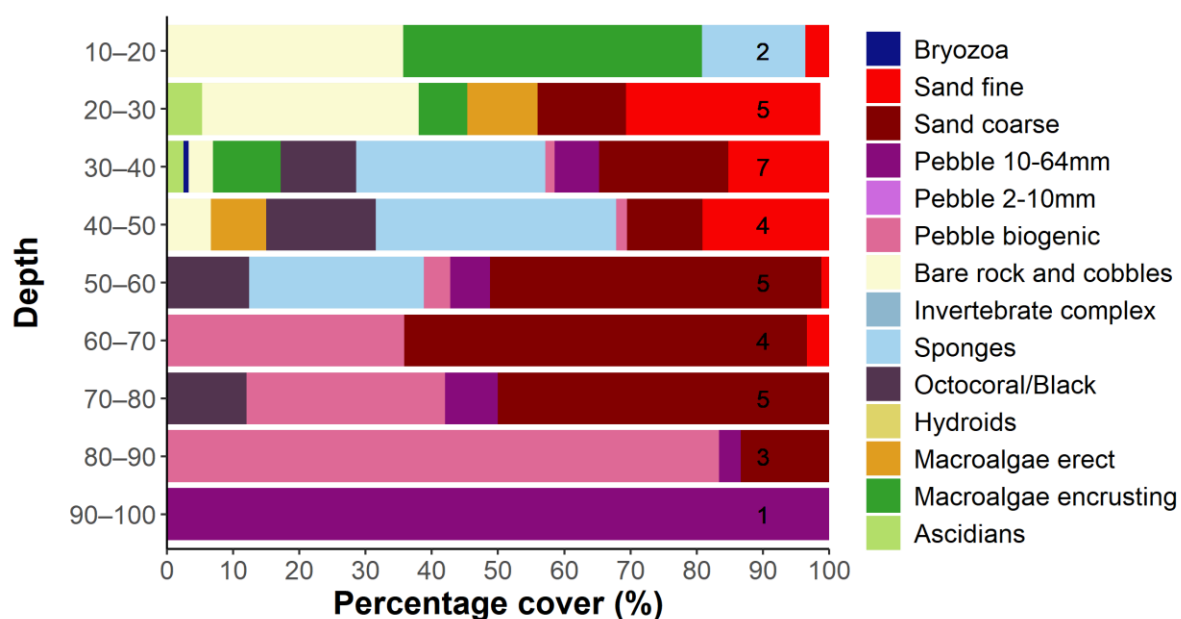


Figure 3.10: Percentage cover of different habitat components calculated from the field of view of each BRUVS deployment for the matched sites in (a) 2006 and (b) 2021. The percentage cover of 'unknown' is not shown. The numbers in each bar indicate the number of samples included

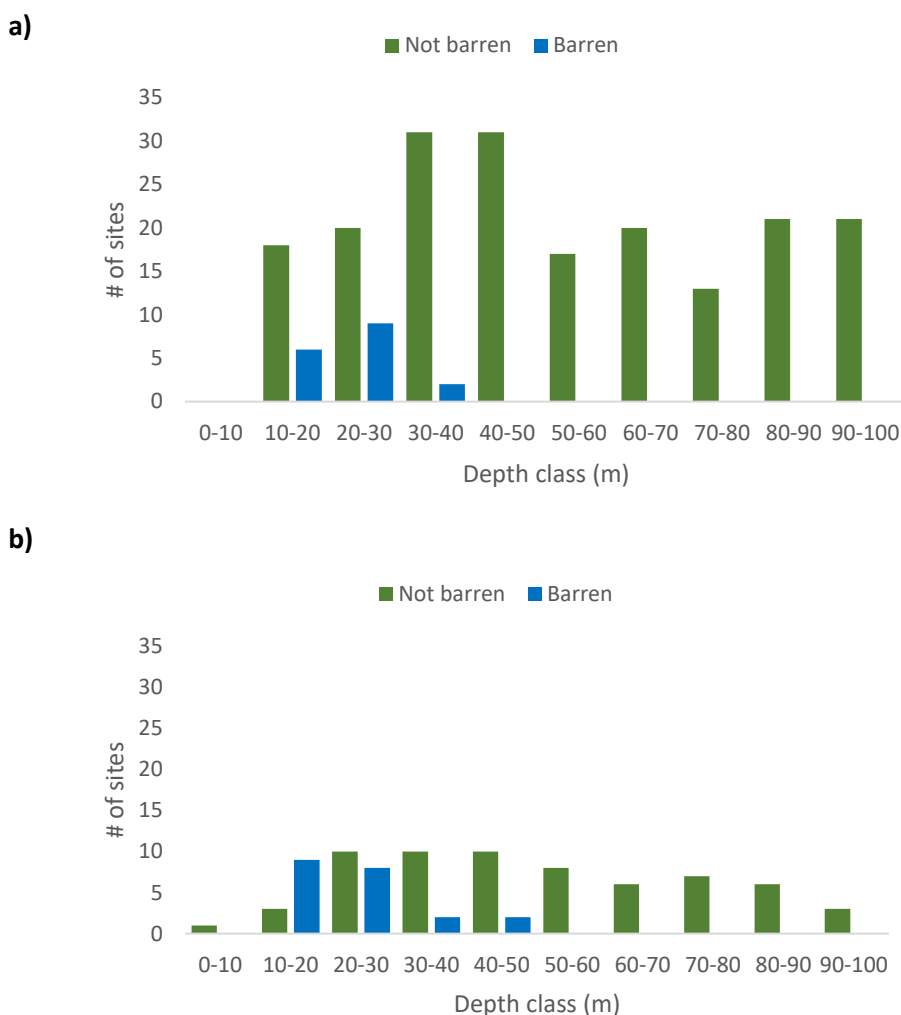


Figure 3.11: Number of deployments classified as barren or not barren for all sites in (a) 2006 and (b) 2021

3.3.2 Key mobile fish groups or species from BRUVS

Key mobile fish groups or species in 2006

In total, 16,241 individuals across 79 species were observed via 208 BRUVS in 2006. Teleost fishes were the most diverse group, consisting of 58 species, followed by 11 chondrichthyans, 9 invertebrates and 1 marine mammal, the Australian Fur Seal (*Arctocephalus pusillus*). The most abundant species was the Yellowtail Scad (*Trachurus novaezelandiae*) followed by the Velvet Leatherjacket (*Meuschenia scaber*) and hermit crabs in the Diogenidae family (Figure 3.12a). *M. scaber* was also the most widespread species, occurring in 164 out of 208 deployments. The Ocean Jacket (*Nelusetta ayraud*) and the hermit crabs were the next most widespread species or groups, occurring in 130 and 128 deployments, respectively. The Velvet Leatherjacket had the highest biomass, followed by Draughtboard Shark (*Cephaloscyllium laticeps*) and Ocean Jacket (Figure 3.12b).

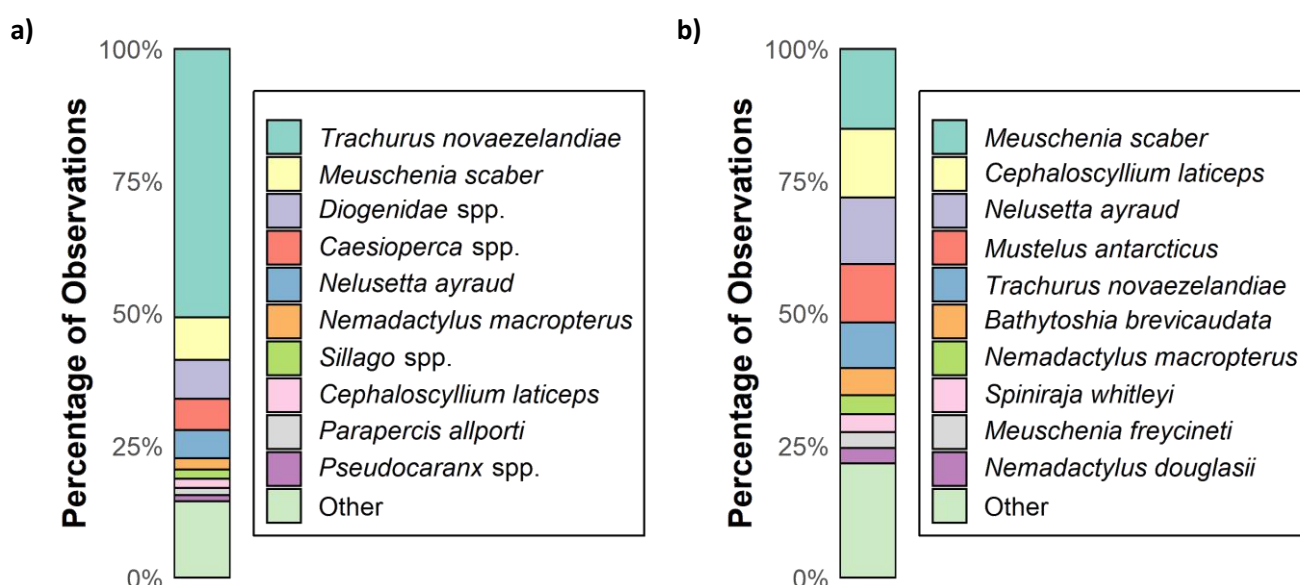


Figure 3.12: Proportions (as percentages) of top species for (a) abundance and (b) total fish lengths (proxy for biomass) across the complete fish assemblage in 2006. 'Other' represents families or species with total observation of less than 2% of the total

Two species of conservation significance were observed from the BRUVS deployments: Eastern Blue Groper (*Achoerodus viridis*) (classified as Near Threatened by the IUCN) and Australian Fur Seal (*Arctocephalus pusillus*) (listed in the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act)). Sampling in 2006, which took place only within the park, included 16 sightings of Eastern Blue Groper and 7 of Australian Fur Seals.

Key mobile fish groups or species in 2021

In total, 6,967 individuals across 94 species were observed via 85 BRUVS deployments in 2021. Teleost fishes were the most diverse group (consisting of 71 species) followed by chondrichthyans (13 species), invertebrates (9 species or groups) and 1 marine mammal, the fur seal (*A. pusillus*). The most abundant species was the Velvet Leatherjacket (*Meuschenia scaber*) followed by Mado Sweep (*Atypichthys strigatus*) and Yellowtail Scad (*Trachurus novaezelandiae*) (Figure 3.13). *M. scaber* was also the most widespread species, occurring in 61 out of 85 deployments, with Grey Morwong (*Nemadactylus douglasii*) and Jackass Morwong (*Nemadactylus macropterus*) the next most widespread species, occurring in 57 and 50 deployments, respectively. Southern Eagle Rays (*Myliobatis tenuicaudatus*) had the highest biomass followed by Smooth Stingrays (*Bathytoshia brevicaudata*) and Gummy Shark (*Mustelus antarcticus*) (Figure 3.13).

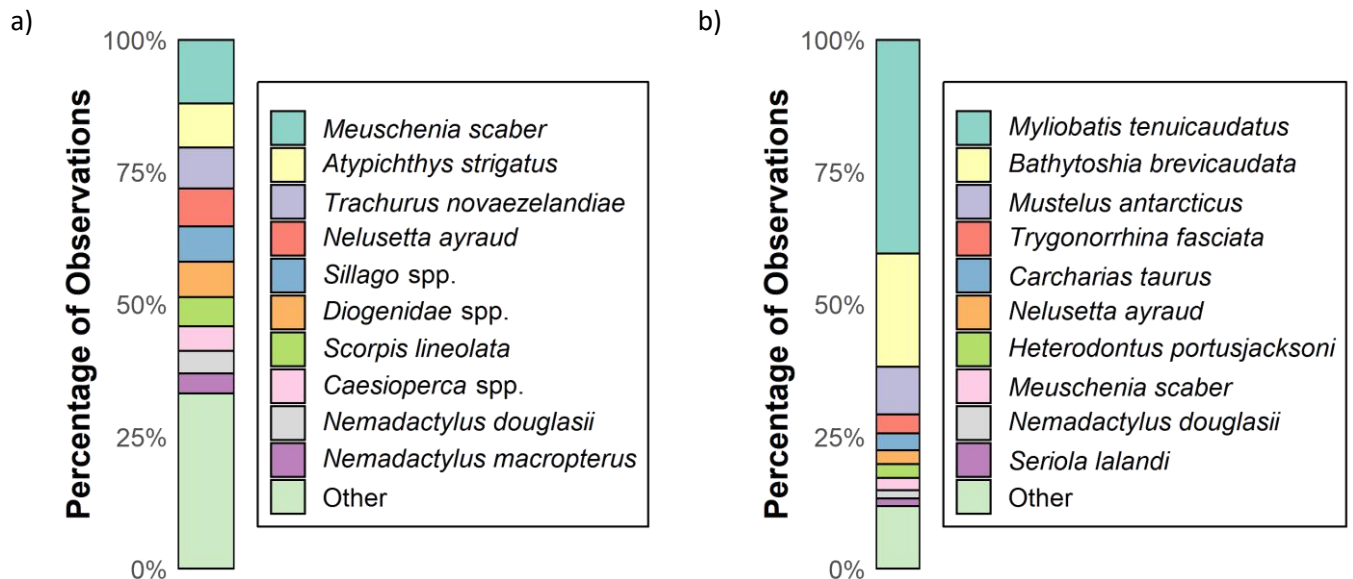


Figure 3.13: Proportions (as percentages) of top species for (a) abundance and (b) total fish lengths (proxy for biomass) across the complete fish assemblage in 2021. ‘Other’ represents families or species with total observation of less than 2% of the total

Three species of conservation significance were observed from the BRUVS deployment in 2021: Eastern Blue Groper (*A. viridis*) (IUCN Near Threatened), Australian Fur Seal (*A. pusillus*) (listed in the EPBC Act) and Grey nurse Shark (*Carcharias taurus*) (IUCN Critically Endangered). Eastern Blue Groper were observed both inside and outside the park; however, more observations were recorded inside (11 vs. 3, respectively). Australian Fur Seals were observed only within the park boundaries at one site, while the single observation of a Grey nurse Shark was outside the park boundaries.

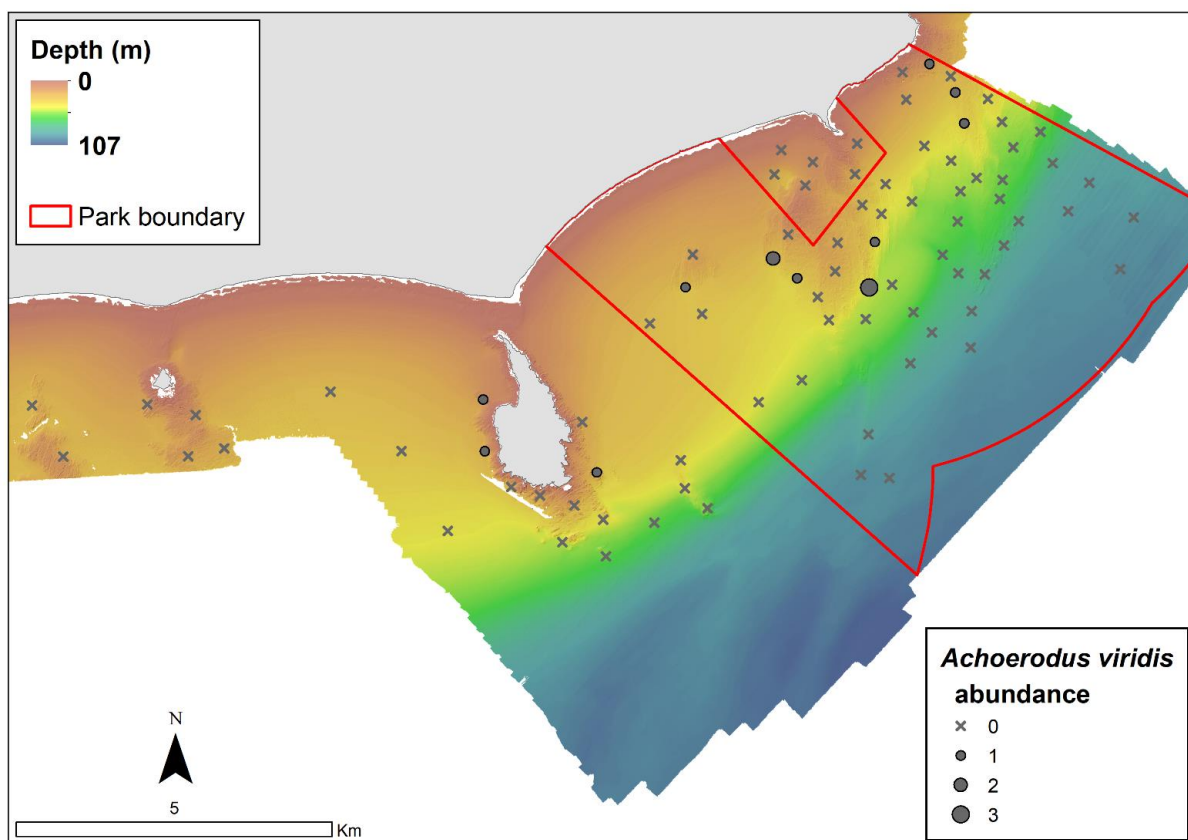


Figure 3.14: Abundance of *Achoerodus viridis* for all BRUVS deployments from 2021 sampling efforts. The size of each site marker corresponds to the number of *A. viridis* observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

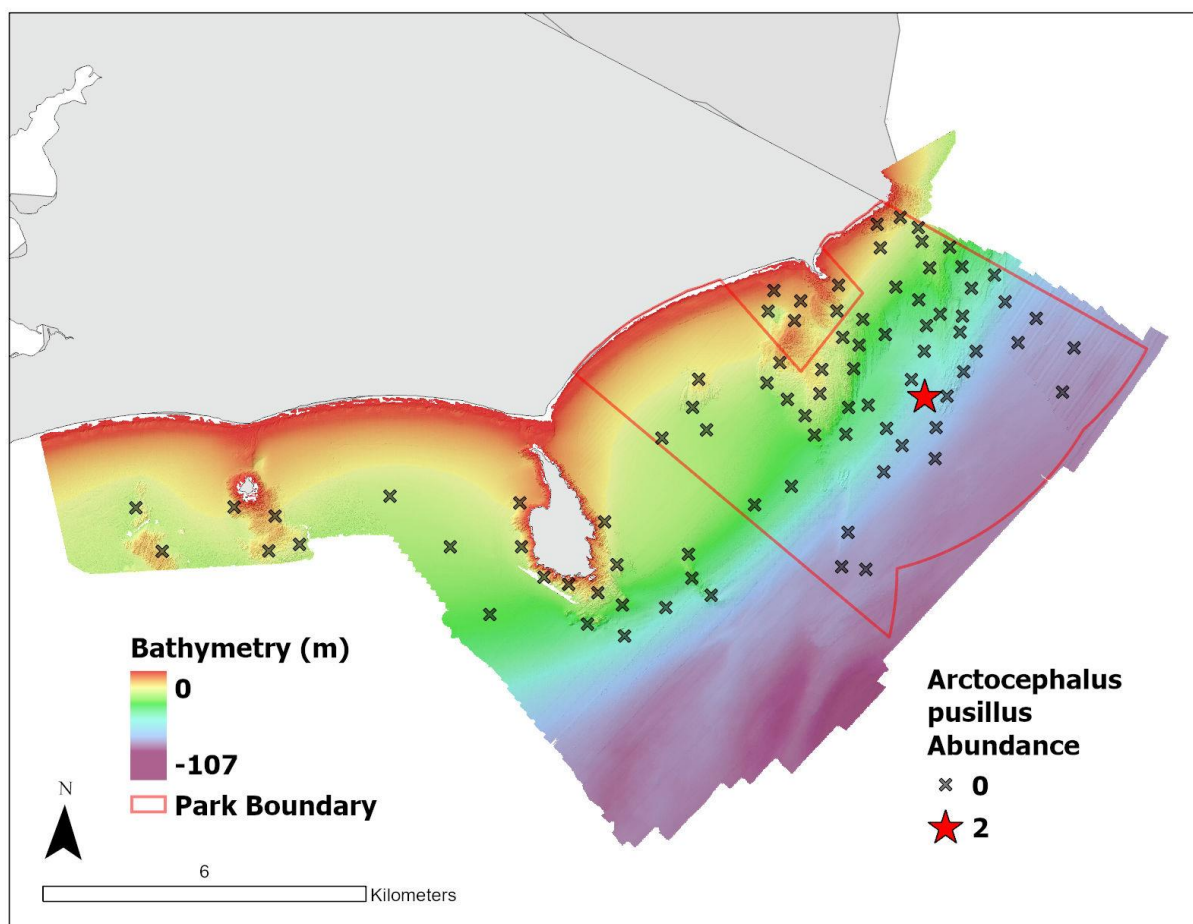


Figure 3.15: Presence of *Arctocephalus pusillus* for all BRUVS deployments from 2021 sampling efforts. The red star corresponds to the site where *A. pusillus* was observed. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

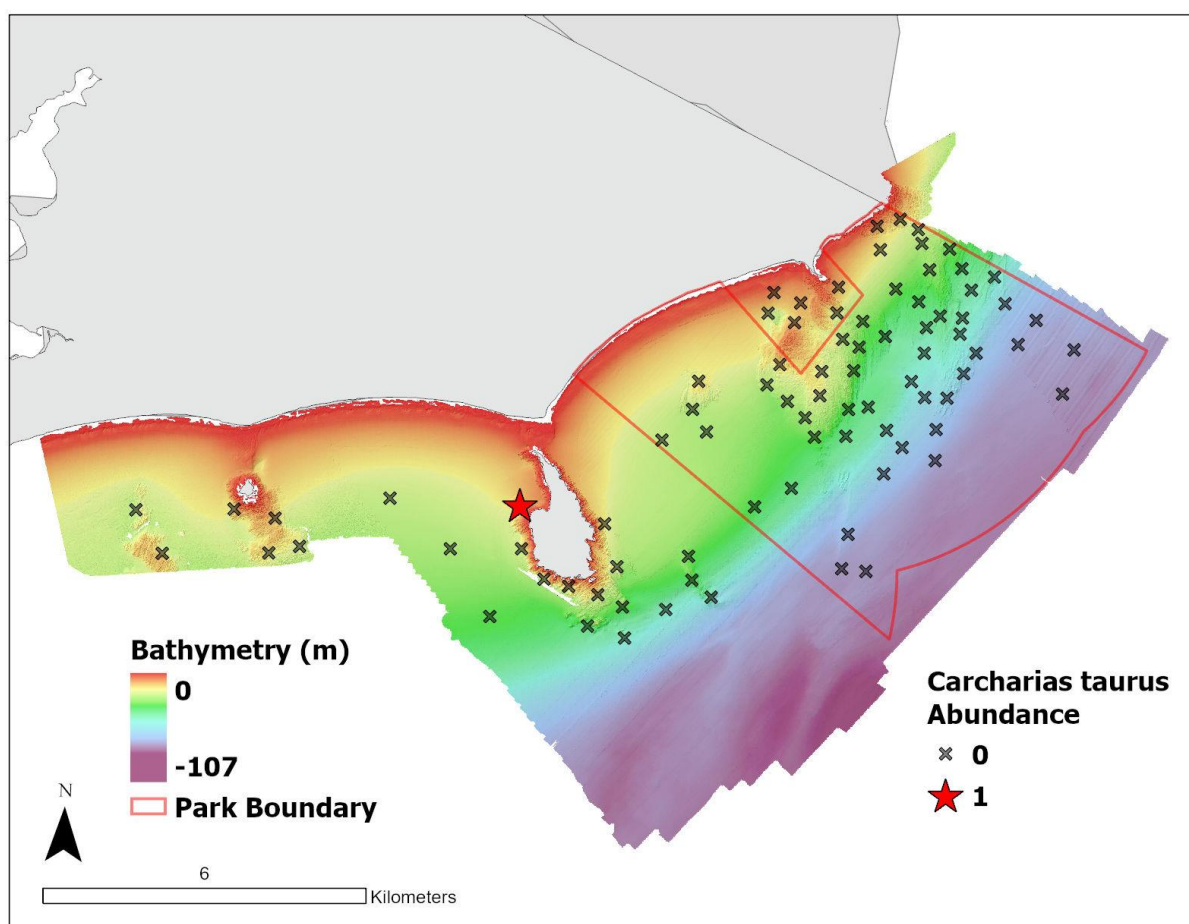


Figure 3.16: Presence of *Carcharias taurus* for all BRUVS deployments from 2021 sampling efforts. The red star corresponds to the site where *C. taurus* was observed. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

3.3.3 Comparison of species abundance and biomass between 2006 and 2021

Thirty-six BRUVS deployments matched geographically within 100 m in 2006 and 2021. In these 72 deployments, 5,398 individuals were observed across 104 species or groups. The number of individuals observed in 2006 (2,612) was similar to the number observed in 2021 (2,786); the average number of individuals observed per deployment was 72.5 in 2006 and 77.4 in 2021 (Table 3.3). Species richness was slightly higher in 2021 at 14.5 species on average per deployment compared to 11.9 in 2006. The broad habitat types observed were similar between years; however, slightly more circalittoral reef was observed in 2006. More urchin barrens were observed in 2021 (5 sites) than in 2006 (3 sites) (Table 3.3).

Table 3.3: Summary table showing the difference between the 2006 and 2021 fish assemblages observed via BRUVS at the matched sites ($n = 36$ each year)

Variable	Year		Total
	2006	2021	
Number of individuals observed	2,612	2,786	5,398
Mean number of individuals per deployment	72.5	77.4	75.0
Mean number of species per deployment	11.9	14.5	13.2
Habitat type			
Circalittoral reef	9	7	16
Infralittoral reef	1	1	2
Soft sediment	23	23	46
Urchin barren	3	5	8
Barren status			
Not barren	33	31	64
Barren	3	5	8

The fish assemblages observed differed significantly between years and for different habitat types within and between years (Table 3.4, Figure 3.17). Fish assemblages in soft-sediment habitat were significantly different from assemblages in both circalittoral reef and urchin barrens in both years (Table 3.4). Fish assemblages in circalittoral reef also differed significantly from those in urchin barrens in 2021 (Table 3.4). Fish assemblages in infralittoral reef did not differ from other habitat types. The low number of infralittoral sites sampled likely influenced the non-significant result in these habitats. Distinct clustering of assemblages into low-complexity soft-sediment habitats or higher complexity habitats with some hard substrate (e.g. reef and barren habitats) can be observed, as can a distinct clustering by year (Figure 3.17). This is further confirmed by a significant P value for the CAP analysis ($P = 0.001$) and a high level of allocation success (79% of sites correctly allocated to the right group). Species typical of soft-sediment areas, such as hermit crabs (*Diogenidae* spp.) and Ocean Jacket (*Nelusetta ayraud*), were strongly correlated with the soft-sediment habitats. In contrast, those typical of hard-bottom areas were more closely correlated with circalittoral and barren habitats, such as Green Moray (*Gymnothorax prasinus*), White-Ear (*Parma microlepis*), Blue-Throat Wrasse (*Notolabrus tetricus*) and Six-Spine Leatherjacket (*Meuschenia freycineti*), among others (Figure 3.17).

The species influencing the differences between years were analysed using SIMPER. The 10 most influential species and groups included *N. ayraud*, *Diogenidae* spp. and Maori Wrasse (*Ophthalmolepis lineolata*). Of the top 10, one species was more abundant in 2006 (*M. freycineti*), 2 had similar abundances in both years (*O. lineolata* and Barred Grubfish (*Parapercis allporti*)), and the remaining 7 species were more abundant in 2021 (Table 3.5). The top 5 species influencing differences between years for the soft-sediment habitat included 2 that were more abundant in 2006 (Velvet Leatherjacket (*Meuschenia scaber*) and *P. allporti*) and 3 species and groups that were more abundant in 2021 (*N. ayraud*, *Diogenidae* spp. and Gummy Shark (*Mustelus antarcticus*)) (Table 3.5). Overall, the dissimilarity in assemblages between years for soft-sediment habitats was high at 80%. For circalittoral reef between years, the 5 most influential species included Maori Wrasse (*Ophthalmolepis lineolata*), which was more abundant in 2006, and *G. prasinus*, Halfbanded Seaperch (*Hypoplectrodes maccullochi*), *M. scaber* and Jackass Morwong (*Nemadactylus*

macropterus), which were more abundant in 2021 (Table 3.5). The dissimilarity between years for circalittoral reef, at 66%, was not as high as for soft-sediment habitats.

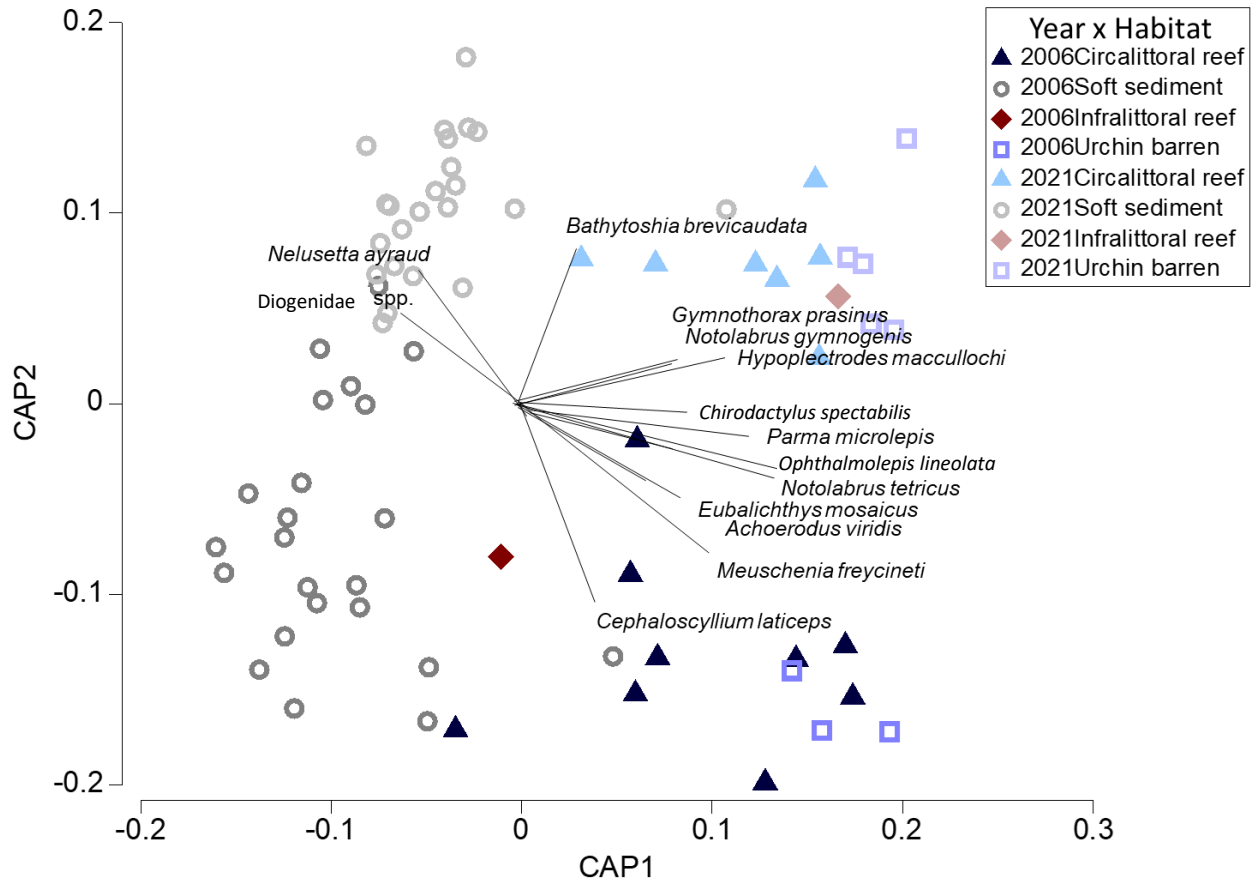


Figure 3.17: CAP ordination plot showing the influence of year and habitat type on the fish assemblages observed via BRUVS for the matched sites in 2006 and 2021. Each point represents the assemblage structure (of all species) of that deployment in relation to the other deployments (similarity); points that are closer together are more similar than those further apart. The overlay shows the species correlated with the first 2 axes (>0.5)

Table 3.4: Analyses of the multivariate fish assemblage observed via BRUVS for the factors of year and habitat type. Significant values ($p < 0.05$) are shown in bold and an asterisk indicates a Monte-Carlo p-value was used

Factor	df	MS	Pseudo- <i>F</i>	<i>P</i>	Unique perms	
Year	1	5,210.1	2.1072	0.021	998	
Habitat	3	17,176	4.2423	0.012	977	
Year × habitat	3	4,048.8	1.6375	0.016	998	
Residual	64	2,472.6				
	<i>t</i>	<i>P</i> (MC)	perms	<i>t</i>	<i>P</i> (MC)	perms
	2006			2021		
Circalittoral reef vs. soft sediment	2.44	0.001	997	2.95	0.001	997
Circalittoral reef vs. infralittoral reef	1.41	0.10	10	1.26	0.16	8
Circalittoral reef vs. urchin barren	0.99	0.50	215	1.51	0.036	559
Soft sediment vs. infralittoral reef	1.05	0.38	24	1.45	0.067	24
Soft sediment vs. urchin barren	2.04	0.001	829	2.66	0.001	995
Infralittoral reef vs. urchin barren	2.01	0.10	4	1.02	0.49	6
	2006 vs. 2021					
Circalittoral reef	1.76	0.001	946			
Soft sediments	2.28	0.001	998			
Infralittoral reef	-	-	-			
Urchin barren	1.49	0.081*	56			

Table 3.5: Results from SIMPER analysis of the matched assemblages observed in 2006 and 2021 using BRUVS showing the top 10 species contributing to differences between years and the top 5 species contributing to differences within habitat types between years

Species	Average abundance		Average abundance		Average dissimilarity	Diss/SD	% contribution	Cumulative %
	2006		2021					
2006 vs. 2021								
Average dissimilarity = 82.56								
<i>Nelusetta ayraud</i>	0.37	<	0.99		5.53	0.80	6.70	6.70
<i>Diogenidae spp.</i>	0.46	<	0.83		4.32	0.87	5.23	11.93
<i>Ophthalmolepis lineolata</i>	0.89	=	0.89		4.11	0.92	4.97	16.90
<i>Mustelus antarcticus</i>	0.61	<	0.92		3.82	0.97	4.63	21.53
<i>Meuschenia scaber</i>	0.69	<	0.79		3.78	0.91	4.58	26.11
<i>Bathytoshia brevicaudata</i>	0.14	<	0.69		2.87	0.90	3.48	29.59
<i>Notolabrus tetricus</i>	0.47	<	0.50		2.42	0.75	2.94	32.53
<i>Meuschenia freycineti</i>	0.69	>	0.25		2.42	0.86	2.94	35.47
<i>Parapercis allporti</i>	0.29	=	0.28		2.29	0.64	2.77	38.24
<i>Gymnothorax prasinus</i>	0.23	<	0.65		2.21	0.83	2.67	40.91
Soft sediment								
Average dissimilarity = 80.3								
<i>Nelusetta ayraud</i>	0.46	<	1.45		9.34	1.10	11.64	11.64
<i>Diogenidae spp.</i>	0.70	<	1.29		6.74	1.15	8.39	20.03
<i>Meuschenia scaber</i>	0.79	>	0.57		5.48	1.03	6.83	27.86
<i>Mustelus antarcticus</i>	0.48	<	0.87		4.92	1.00	6.13	32.99
<i>Parapercis allporti</i>	0.46	>	0.41		3.87	0.84	4.83	37.82
Circalittoral reef								
Average dissimilarity = 65.8								
<i>Ophthalmolepis lineolata</i>	2.44	>	2.29		3.70	1.39	5.62	5.62
<i>Gymnothorax prasinus</i>	0.46	<	1.86		3.28	1.29	4.98	10.60
<i>Hypoplectrodes maccullochi</i>	0.22	<	1.43		3.07	1.25	4.67	15.26
<i>Meuschenia scaber</i>	0.62	<	1.75		2.73	1.75	4.15	19.42
<i>Nemadactylus macropterus</i>	0.25	<	1.05		2.46	0.50	3.74	23.16

3.3.4 Environmental drivers of fish assemblage

Environmental drivers in 2021

Various environmental drivers influenced the fish assemblages observed via BRUVS in 2021. For all sites sampled, there was a significant effect of status (i.e. inside the park (no-take) vs. outside the park (fished)) and habitat type (Table 3.6). The species driving the differences in the assemblages observed in fished and no-take areas included Blue-Throat Wrasse (*Notolabrus tetricus*), Maori

Wrasse (*Ophthalamolepis lineolata*), White-Ear (*Parma microlepis*), Green Moray (*Gymnothorax prasinus*), Eastern Fiddler Ray (*Trygonorrhina fasciata*) and Halfbanded Seaperch (*Hypoplectrodes maccullochi*), which were more abundant in fished areas, and Ocean Jackets (*Nelusetta ayraud*), hermit crabs (Diogenidae spp.), Gummy Shark (*Mustelus antarcticus*) and Smooth Stingray (*Bathytoshia brevicaudata*), which were more abundant in no-take areas (Table 3.7).

Fish assemblages differed significantly between all habitat type pairs within no-take areas, except for the pair of infralittoral reef habitats and urchin barrens, whose fish assemblages were similar (Table 3.6). All habitat type pairs in fished areas showed significant differences for fish assemblages (infralittoral sites could not be compared because there were no samples) (Table 3.6). Significant differences in the fish assemblage were observed between fished and no-take areas for soft-sediment habitats and urchin barrens, but not for circalittoral reefs (Table 3.6). These patterns can also be observed in the clustering shown in the CAP analysis (Figure 3.18): soft sediments, urchin barrens and circalittoral reefs show distinct clusters.

The CAP analysis was also significant ($P = 0.001$) and had a moderate allocation success rate of 67%. The fine-scale environmental variables driving these differences were investigated using DISTLM analysis, which included 5 variables in the top model (percentage cover of fine sand, sponges and octocoral/black coral; depth; and mean relief) (Figure 3.19a). The top model explained 42% of the variation within the fish assemblage, with the first 2 axes of the dbRDA plot explaining 26% and 9% of the total variation, respectively. Species correlated with the variables selected in the top model included Bluespotted Flathead (*Platycephalus caeruleopunctatus*), which increased in abundance as the percentage cover of fine sand increased, *N. ayraud*, which increased in abundance as the depth increased, and various reef-associated species that increased as the percentage cover of sponges, octocorals, and relief increased (Figure 3.19b).

In 2021, no differences were observed between fished and no-take areas for species richness, total abundance and total biomass alone ($P > 0.28$). Species richness in and total biomass were, however, influenced by habitat type, while total abundance was not (Figure 3.20b,d). For species richness, soft-sediment habitats had significantly lower numbers than both urchin barrens and circalittoral reef; circalittoral reef had the highest species richness (Figure 3.20b). Soft-sediment habitats also had significantly lower biomass than circalittoral reef and urchin barrens but infralittoral reef did not differ from either urchin barrens, circalittoral reef or soft sediment (Figure 3.20f).

Table 3.6: Analysis of the multivariate fish assemblage observed via BRUVS in 2021 for the factors of status (fished vs. no-take) and habitat type. Significant values ($p < 0.05$) are shown in bold and an asterisk indicates a Monte-Carlo p-value was used

Factor	df	MS	Pseudo- <i>F</i>	<i>P</i>	Unique perms	
Status	1	6,987.5	2.96	0.003	998	
Habitat	3	19,317	8.18	0.001	997	
Status × habitat	2	5,158.5	2.18	0.002	998	
Residual	78	2,362.5				
	t	<i>P</i>	perms	t	<i>P</i>	perms
Pairs		No-take			Fished	
Circalittoral reef vs. soft sediment	3.67	0.001	998	1.95	0.001	768
Circalittoral reef vs. infralittoral reef	1.47	0.017	120			
Circalittoral reef vs. urchin barren	1.86	0.001	997	1.51	0.031	819
Soft sediment vs. infralittoral reef	1.90	0.005	450			
Soft sediment vs. urchin barren	3.17	0.001	999	2.71	0.001	997
Infralittoral reef vs. urchin barren	0.85	0.669*	45			
Fished vs. no-take						
Circalittoral reef	1.28	0.056	862			
Soft sediments	2.26	0.001	997			
Infralittoral reef	-	-	-			
Urchin barren	1.43	0.013	996			

Table 3.7: Results from SIMPER analysis of the assemblage observed in 2021 using BRUVS showing the top 10 species contributing to the dissimilarity between fished and no-take areas

Species	Average abundance No-take		Average abundance fished	Average dissimilarity	Diss/SD	% contribution	Cumulative %
<i>Notolabrus tetricus</i>	0.52	<	1.66	5.07	0.99	6.14	6.14
<i>Nelusetta ayraud</i>	0.99	>	0.00	4.95	0.69	5.99	12.13
<i>Ophthalamolepis lineolata</i>	0.65	<	1.36	4.32	1.05	5.23	17.37
Diogenidae spp.	0.68	>	0.11	3.24	0.66	3.92	21.29
<i>Parma microlepis</i>	0.52	<	0.93	3.07	0.93	3.72	25.01
<i>Gymnothorax prasinus</i>	0.55	<	0.90	2.94	1.03	3.56	28.56
<i>Trygonorrhina fasciata</i>	0.35	<	0.45	2.57	0.61	3.11	31.67
<i>Mustelus antarcticus</i>	0.66	>	0.30	2.42	0.88	2.93	34.60
<i>Bathytoshia brevicaudata</i>	0.66	>	0.59	2.41	0.87	2.91	37.51
<i>Hypoplectrodes maccullochi</i>	0.38	<	0.59	2.28	0.79	2.76	40.28

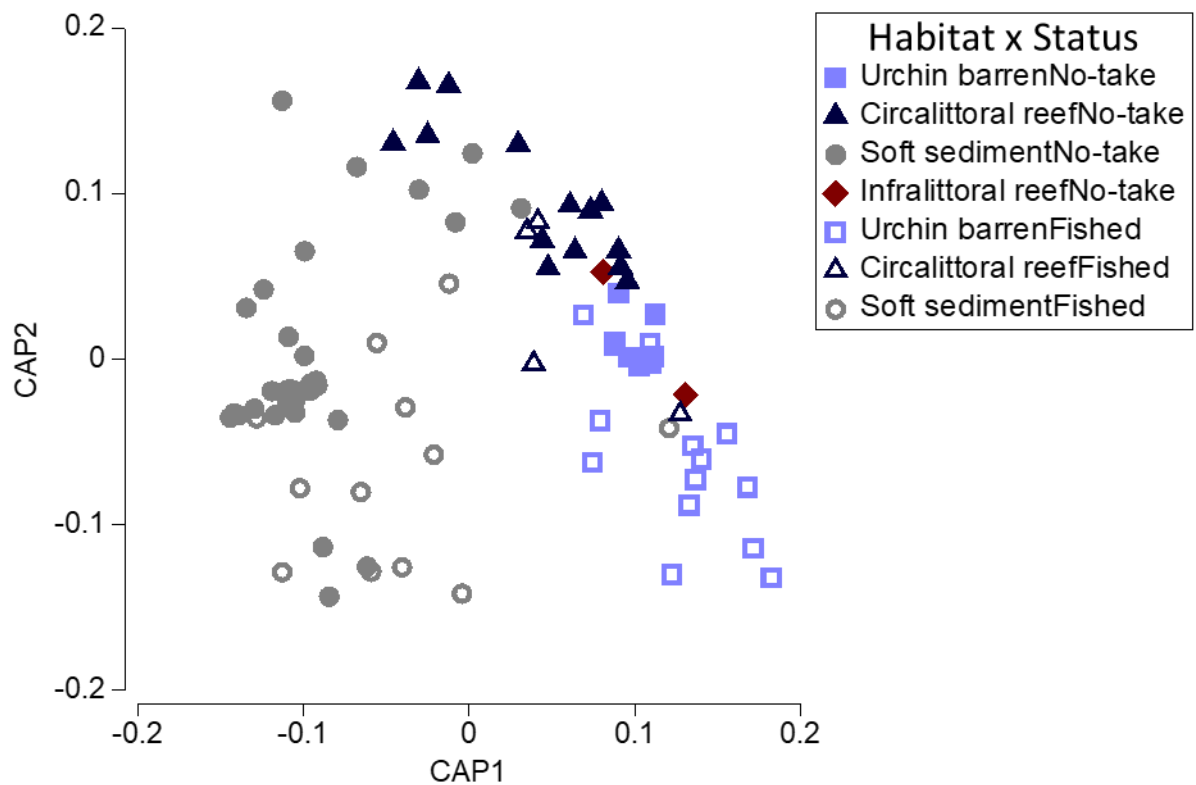


Figure 3.18: CAP ordination plot showing the influence of habitat type and status (fished vs. no-take) on the fish assemblages observed via BRUVS in 2021. Each point represents the assemblage structure (of all species) of that deployment in relation to the other deployments (similarity); points that are closer together are more similar than those further apart

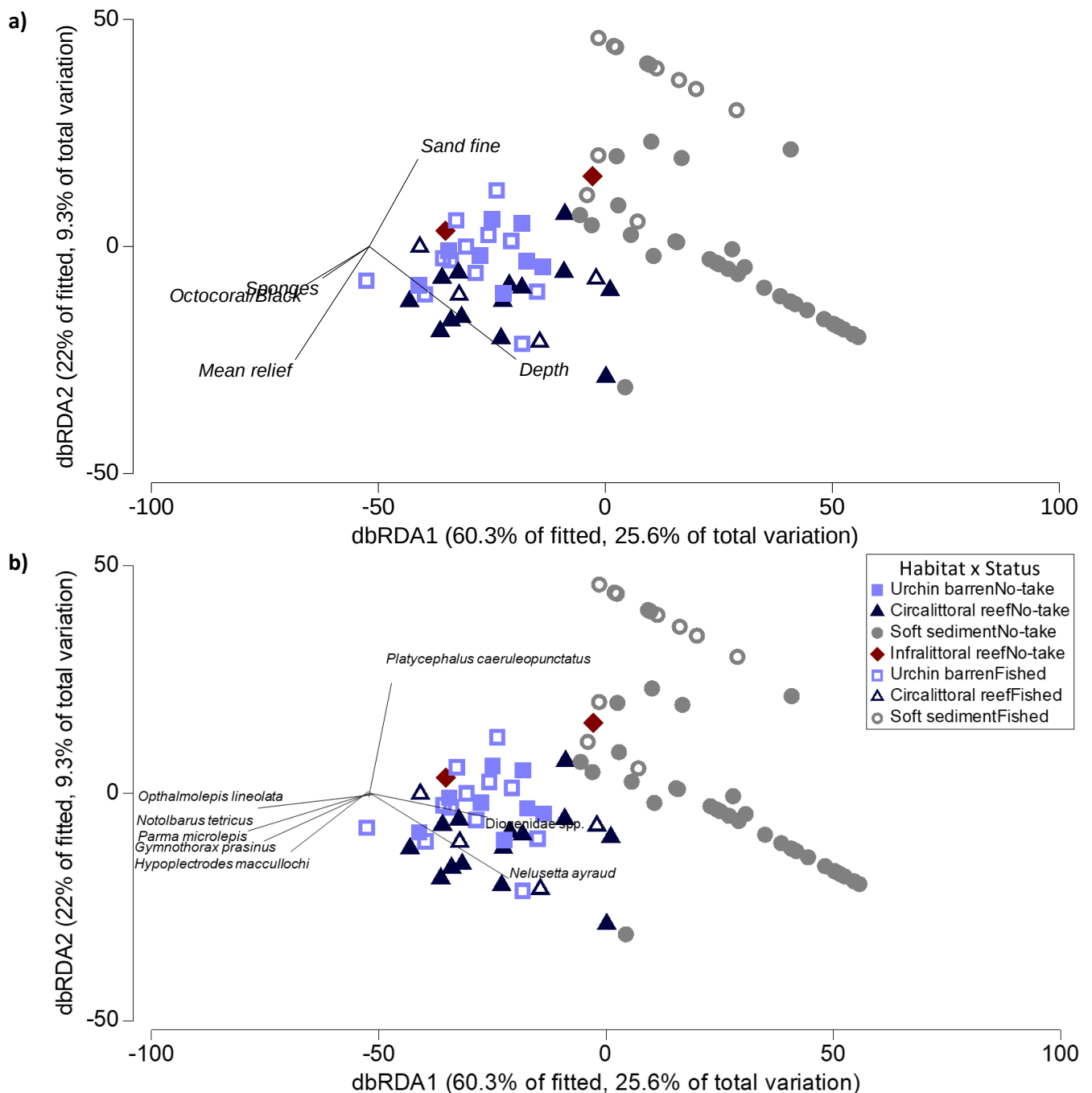


Figure 3.19: Distance-based redundancy analysis plots showing the influence of environmental variables (percentage cover of fine sand, sponges, and octocoral/black corals, mean relief, and depth (m)) based on the top model output from DISTLM for the 2021 assemblage. Species with correlations >0.5 with the dbRDA axes are shown in (b)

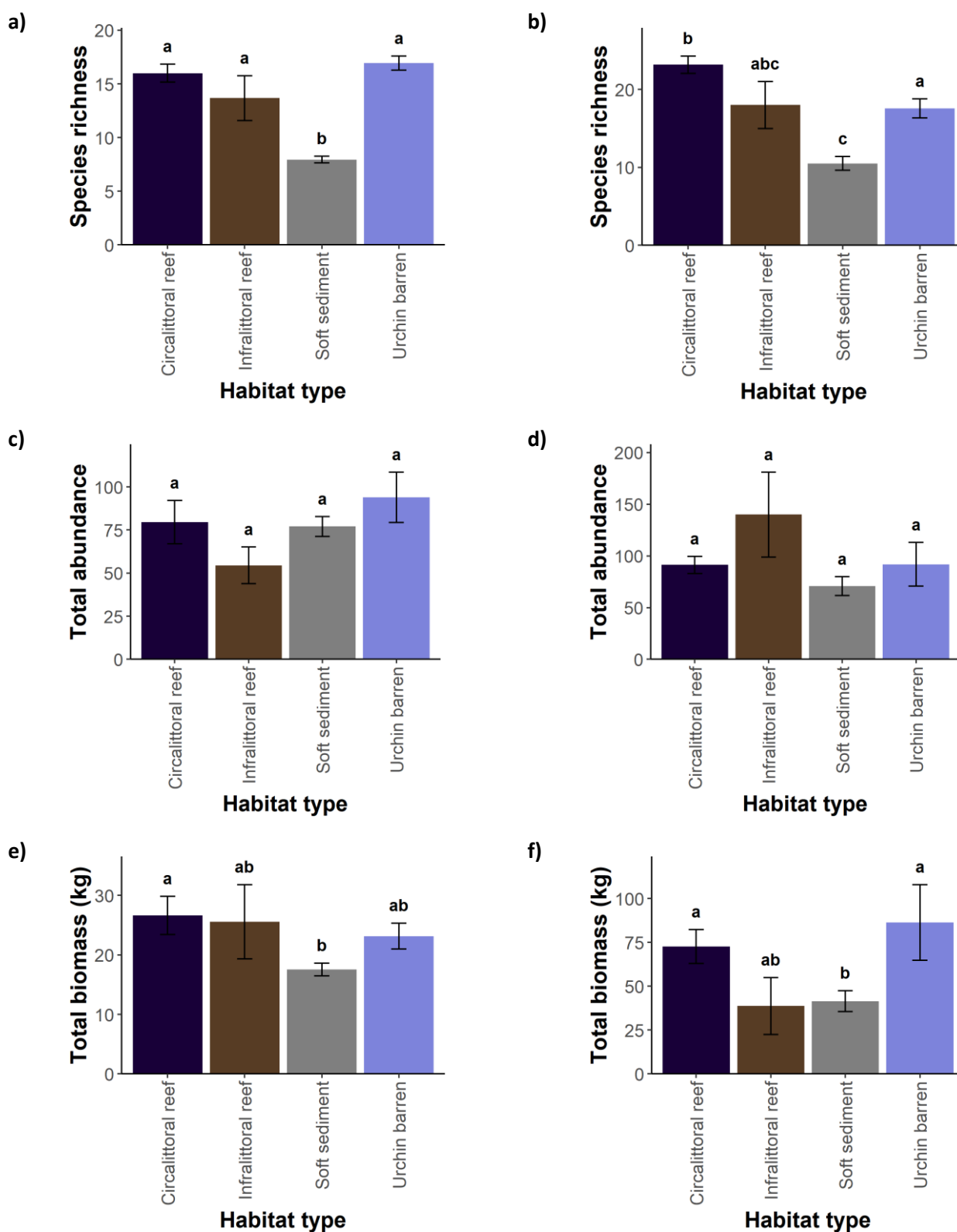


Figure 3.20: Average ($\pm$ SE) species richness, total abundance, and total biomass (kg) for habitat type in 2006 (graphs a, c, e) and 2021 (graphs b, d, f). Letters denote significant differences among pairs according to PERMANOVA analyses (i.e. bars with the same letter within the same graph are not significantly different)

Environmental drivers in 2006

Habitat type had a significant effect on the assemblages observed via BRUVS in 2006 (pseudo- $F = 10.42$, $P = 0.001$). All habitat types were significantly different from one another with the exception of circalittoral and infralittoral reefs (Table 3.8). Distinct clustering by habitat type can also be observed in the CAP analysis, particularly between soft-sediment habitats and urchin barrens (Figure 3.21). CAP also showed significant differences between habitat types ($P = 0.001$) and had a high allocation success rate of 82%. DISTLM analysis showed the variables driving the differences observed in the assemblage, with 5 variables included in the top model (percentage cover of fine sand, bare rock and sponges; mean relief; and depth) (Figure 3.22a). Bluespotted Flathead (*Platycephalus caeruleopunctatus*) were associated with an increase in the percentage cover of fine sand, while hermit crabs (Diogenidae spp.), Barred Grubfish (*Parapercis allporti*) and Ocean Jackets (*Nelusetta ayraud*) were associated with increased depths (Figure 3.22b). Common reef-associated species, such as Six-Spine Leatherjacket (*Meuschenia freycineti*) and White-Ear (*Parma microlepis*), were correlated with increased relief, and cover of bare rock and sponges (Figure 3.22b). Species richness and total biomass were also influenced by habitat type; however, total abundance was not significantly different among habitat types (Figure 3.20a,c,e). Soft-sediment habitats had significantly lower species richness than the other habitat types, and also had significantly lower biomass than circalittoral reef (Figure 3.20a,e).

Table 3.8: Pairwise analysis of the multivariate fish assemblage observed via BRUVS in 2021 for the factor habitat type. Significant values are shown in bold

Pairs	t	<i>P</i>	Unique perms
Circalittoral reef vs. soft sediment	3.59	0.001	999
Circalittoral reef vs. infralittoral reef	1.27	0.102	996
Circalittoral reef vs. urchin barren	2.35	0.001	998
Soft sediment vs. infralittoral reef	1.40	0.036	998
Soft sediment vs. urchin barren	4.31	0.001	998
Infralittoral reef vs. urchin barren	2.22	0.001	994

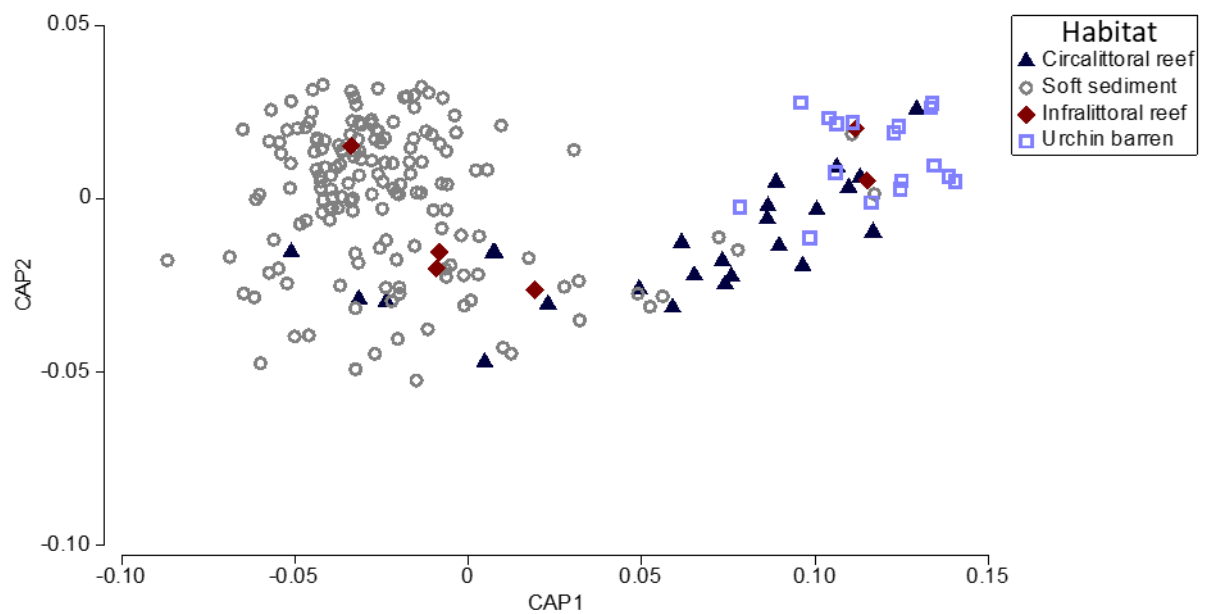


Figure 3.21: CAP ordination plot showing the influence of habitat type on the fish assemblage observed via BRUVS for 2006. Each point represents the assemblage structure (of all species) of that deployment in relation to the other deployments (similarity); points that are closer together are more similar than those further apart

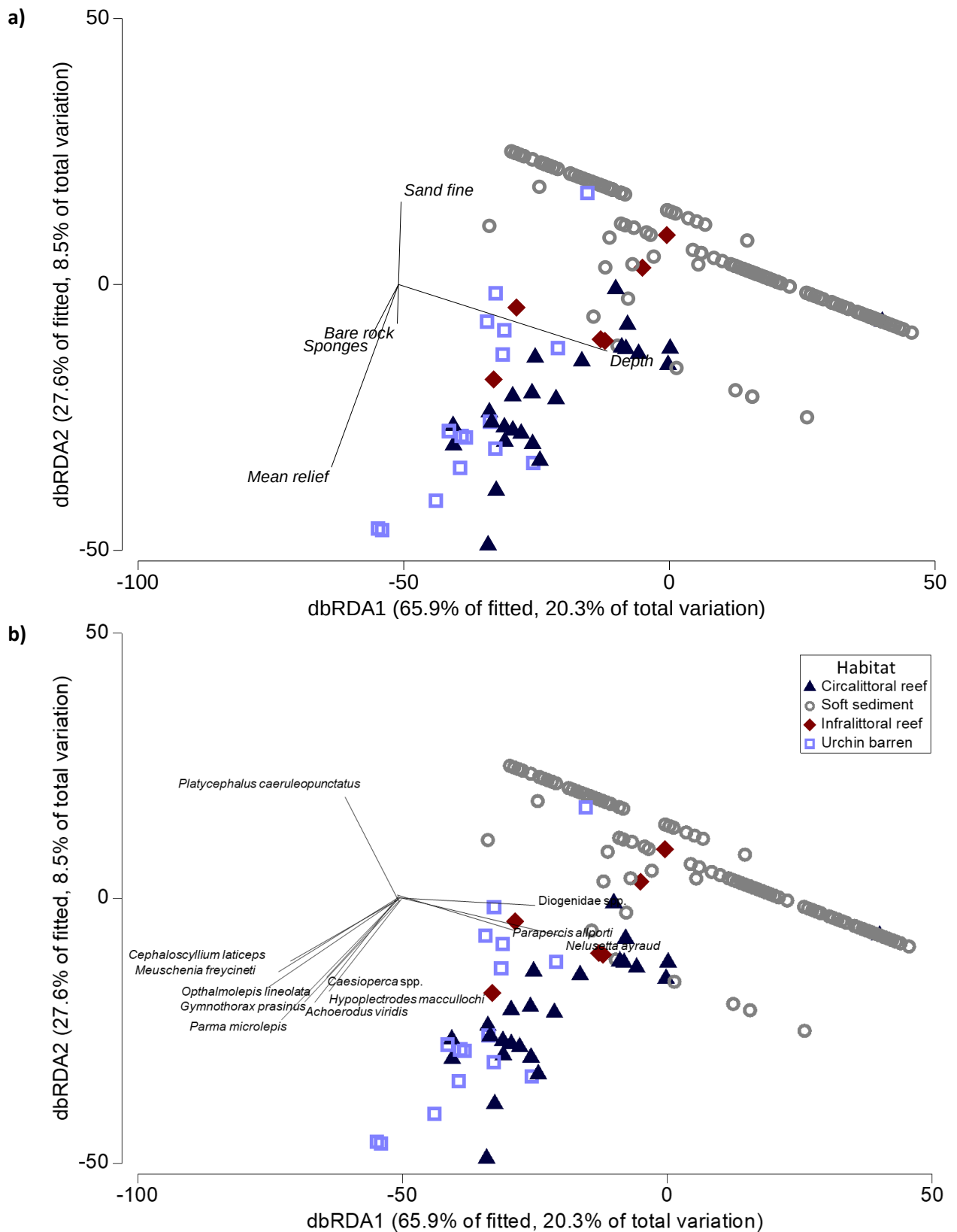


Figure 3.22: Distance-based redundancy analysis plots showing the influence of environmental variables (percentage cover of fine sand, sponges and bare rock; mean relief; and depth (m)) based on the top model output from DISTLM for the 2006 assemblage. Species with correlations >0.5 with the dbRDA axes are shown in (b)

3.3.5 Power analysis

Power analysis showed a moderate ability to detect change for the 2021 sampling design used ($n = 85$) with adequate power to detect large effects (0.75) and some ability to detect medium effects (0.5) (Figure 3.23), although using this design may make it difficult to detect small effects (0.25). This was calculated based on the sample sizes used inside and outside the park.

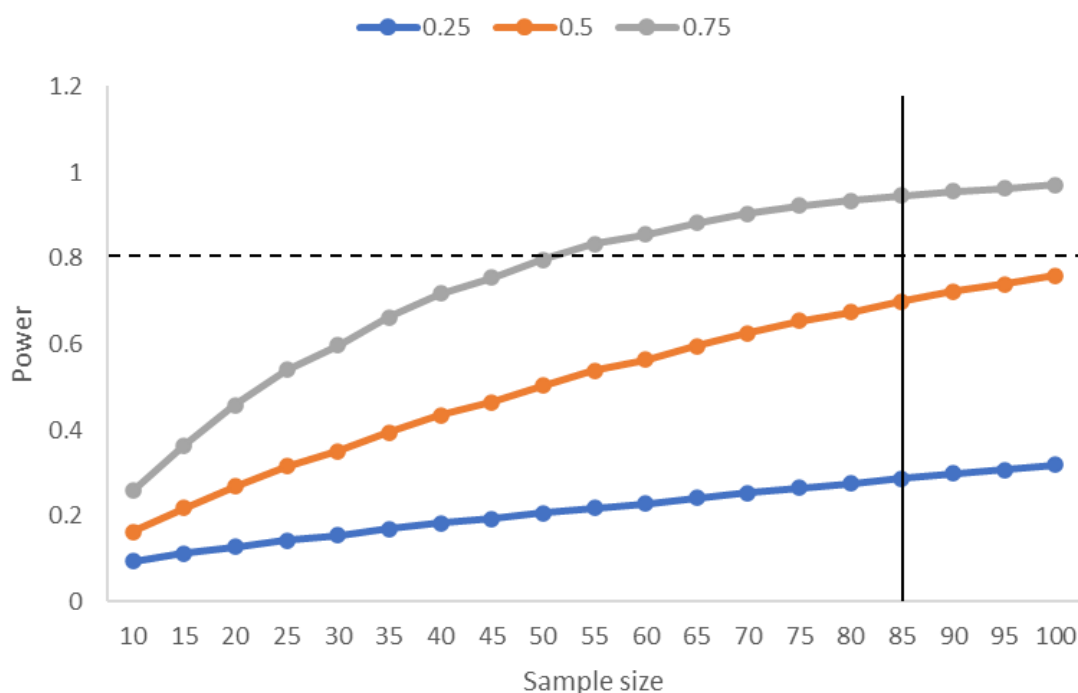


Figure 3.23: Power analysis showing the achieved power given the sample size (number of BRUVS drops = 85, shown by black line) for 3 effect sizes (0.25, 0.5, 0.75). The dashed line shows the recommended minimum power

3.3.6 BRUVS vs. Reef Life Survey and Subtidal Reef Monitoring Program

The fish assemblage observed via BRUVS differed from the assemblages observed via RLS/SRMP surveys. For fish species alone (i.e. teleosts and chondrichthyans), 49 species were common to both methods, 40 species were unique to BRUVS and 35 species were unique to RLS/SRMP surveys (Figure 3.24). The species unique to BRUVS included large, more mobile species, such as sharks and rays (e.g. fiddler rays, Grey nurse Shark (*Carcharias taurus*) and Gummy Shark (*Mustelus antarcticus*)) and species that are more common in deep waters, such as Red Gurnard Perch (*Helicolenus percooides*), Ocean Jacks (*Nelusetta ayraud*) and Common Gurnard Perch (*Neosebastes scorpaenoides*). The species unique to RLS/SRMP included small, cryptic and habitat-associated species such as weedfish (*Heteroclinus* spp.), Common Seadragon (*Phyllopteryx taeniolatus*) and Little Siphonfish (*Siphamia cephalotes*).

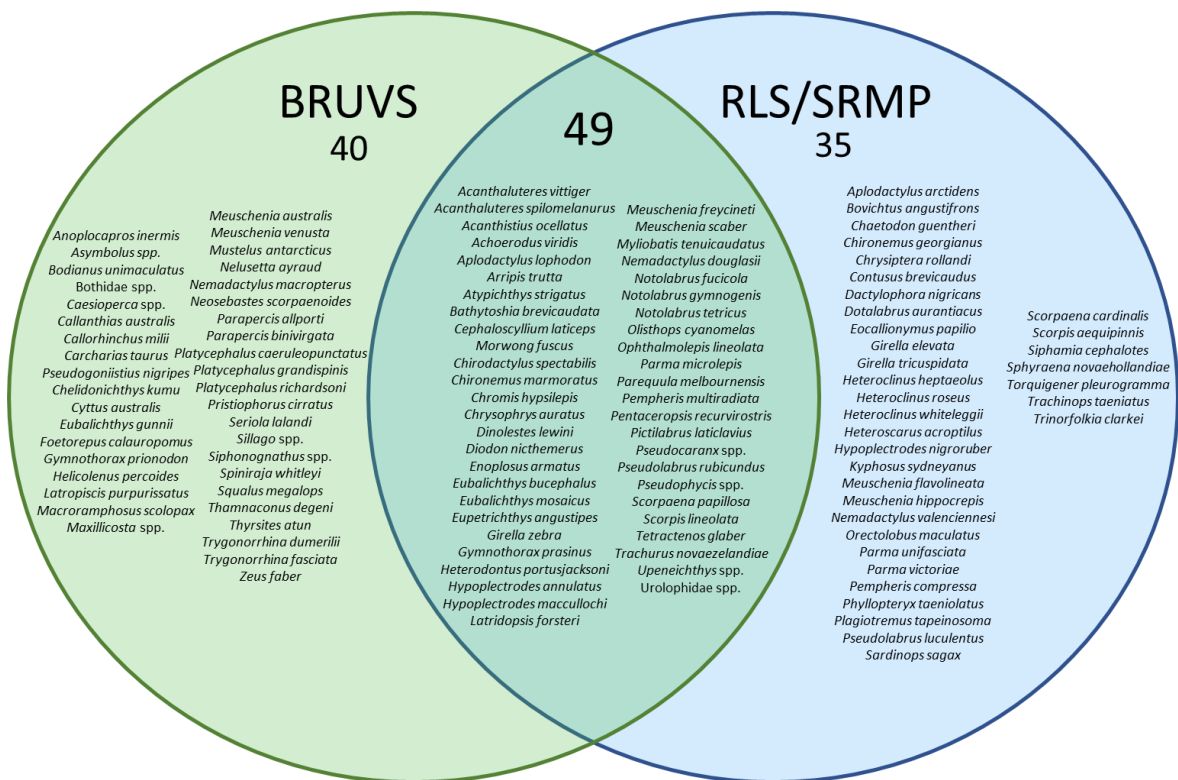


Figure 3.24: Overlap and unique species observed in BRUVS and RLS/SRMP surveys for the fish-only assemblage combined across all years sampled

3.3.7 Performance of species distribution models using BRUVS data

Large variation in model performance was present across community metrics and individual species modelled (Figures 3.25, 3.27, 3.29, 3.31, 3.33, 3.35, 3.37, 3.39, 3.41). Species richness, herbivore abundance and *Ophthalmolepis lineolata* abundance were found to have the highest deviations explained at 47.9%, 32.1% and 38.8%, respectively. Depth (bathymetry), distance to reef and rugosity were found to best explain trends in species richness and *O. lineolata* abundance (Table 3.9). Distance to reef and rugosity had positive relationships with species richness and *O. lineolata* abundance and a negative relationship with depth. Herbivore abundance was best explained by a positive relationship with distance to reef and rugosity. Of individual species modelled, relative abundance was predicted for *Mustelus antarcticus*, *Nemadactylus douglasii* and *O. lineolata*, which were selected based on sample size and fisheries importance (Figures 3.37, 3.39, 3.41). Models of *M. antarcticus* abundance and *N. douglasii* abundance explained 17.6% and 17.0% of deviation, respectively. *M. antarcticus* and *N. douglasii* were best explained by depth and rugosity, both having a Gaussian relationship. These findings were reflected by trends observed in bubble plots of BRUVS deployment results, where the most evident trend was that highest species richness was over or near reef (Figures 3.26, 3.28, 3.30, 3.32, 3.34, 3.36, 3.38, 3.40, 3.42). Furthermore, some differences were observed between sites within the MPA and reference sites. For the 2021 survey, species richness and total abundance were generally higher inside the MPA than outside.

Table 3.9: Importance of predictor variables (columns) used in each of the model subsets (rows) used in this study. Colour gradient denotes the relative importance of each variable in its respective model. Values are on a scale from red (low importance) to dark green (high importance)

	Depth	Distance to reef	Slope	MPA	Rugosity
Species richness	26.16	25.65	14.05	16.61	20.1
Total abundance	6.1	4.1	1.2	11.3	5.5
Total biomass		2.8			1.4
Herbivore abundance	6.8	27.4	8.9		19.4
Carnivore abundance	4.9	8.9	6		6.5
Invertivore abundance	2.6	9.5	3.5		6
<i>Nemadactylus douglasii</i>	9.9	1.4	0.9		8.4
<i>Mustelus antarcticus</i>	10.4	3.8	3.7	17.9	7.5
<i>Ophthalamolepis lineolata</i>	17.1	24.8	8.1		15.4

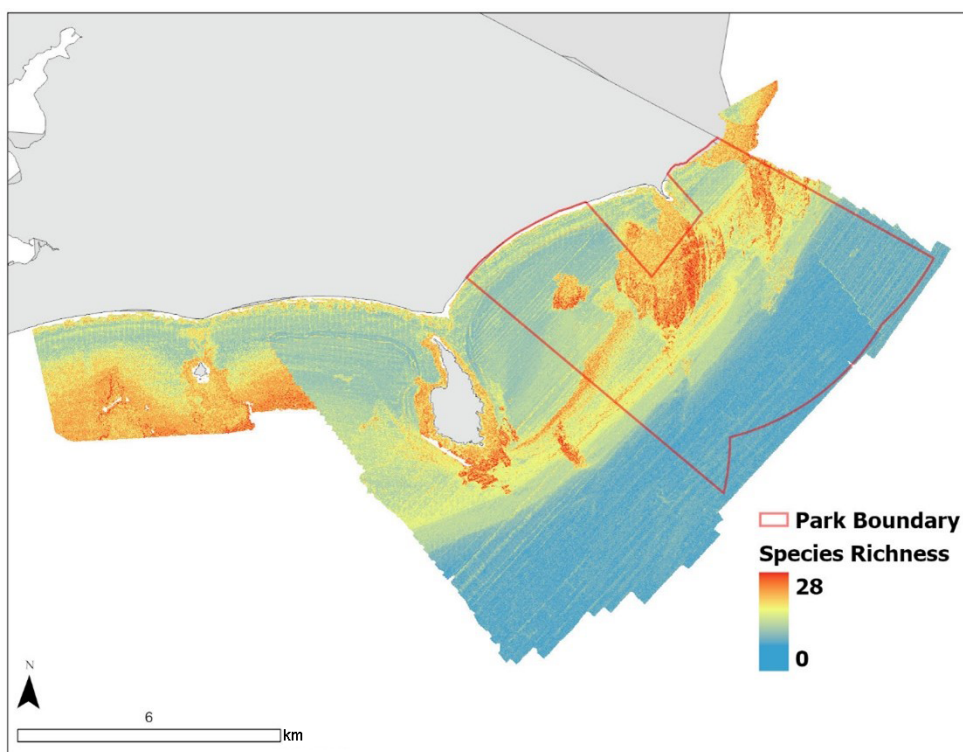


Figure 3.25: Predicted species richness in 2021 from BRUVS sampling across the whole study site. The random forest model explained 47.9% of the variance and had a pseudo- R^2 value of 0.75

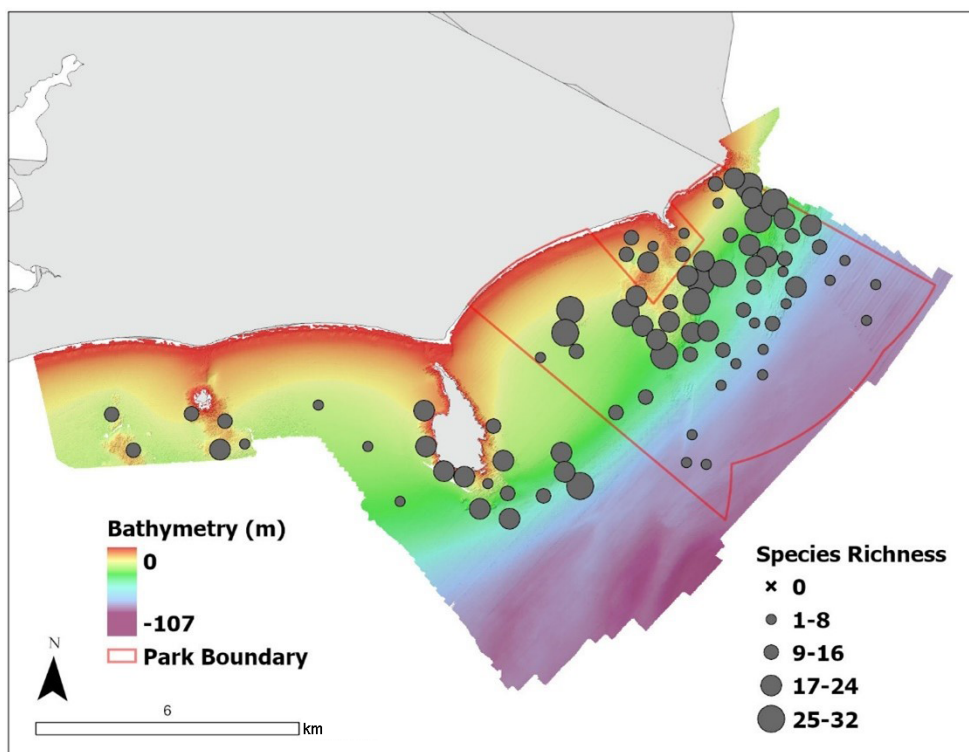


Figure 3.26: Species richness of fish for all BRUVS deployments in 2021. The size of each site marker corresponds to the species richness of fish observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

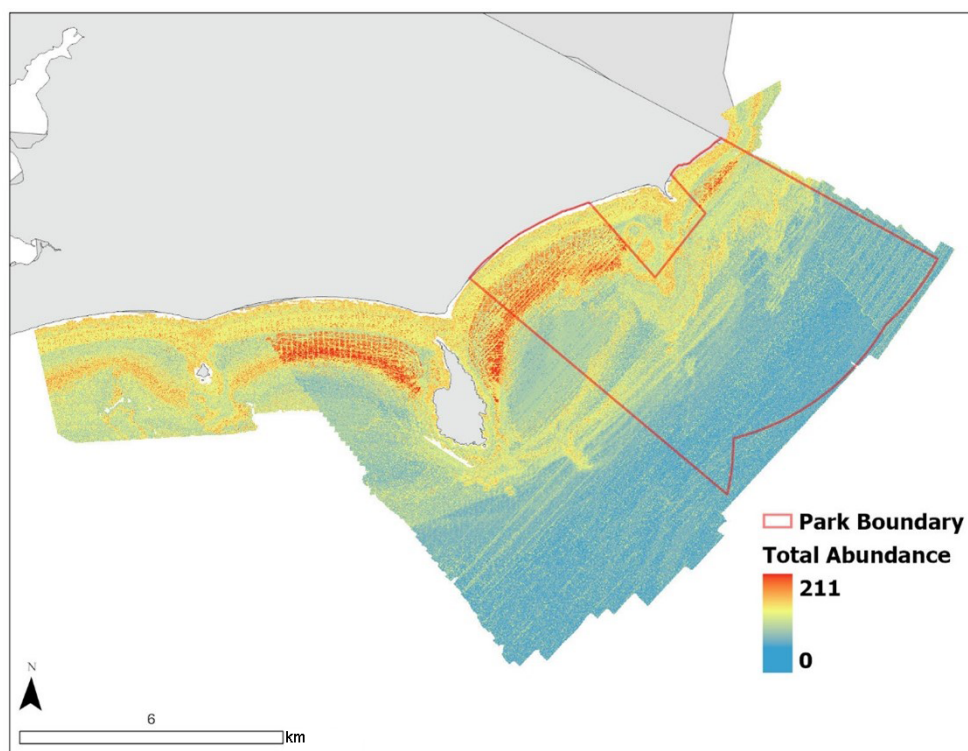


Figure 3.27: Predicted total abundance of all species in 2021 from BRUVS sampling across the whole study site. The random forest model explained 31.3% of the variance and had a pseudo- R^2 value of 0.85

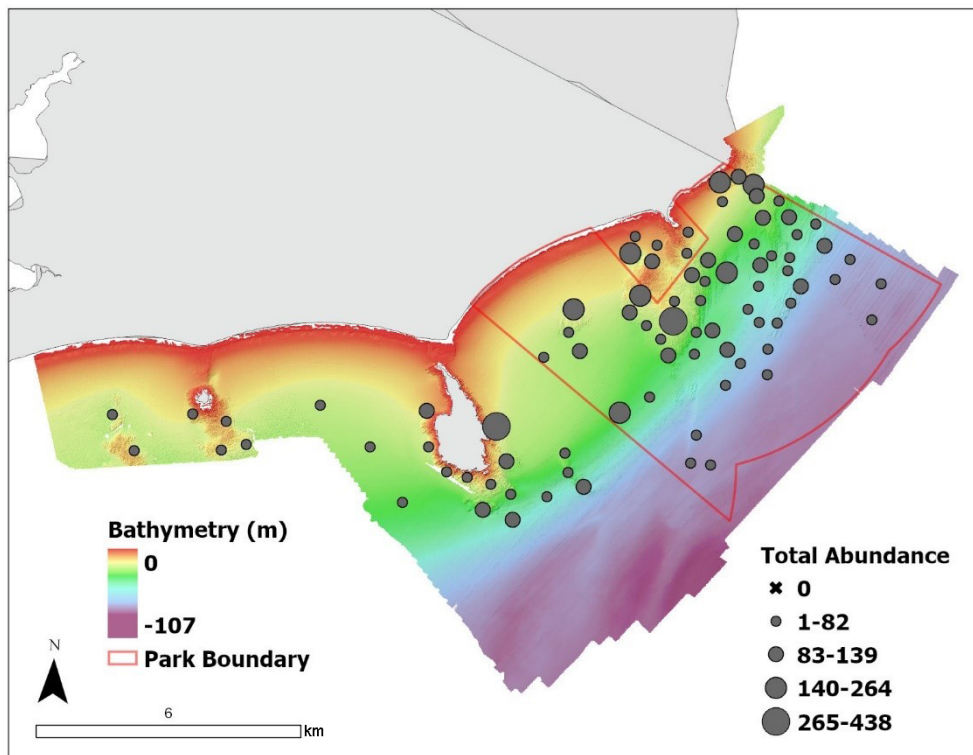


Figure 3.28: Total abundance of fish for all BRUVS deployments in 2021. The size of each site marker corresponds to the total abundance of fish observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

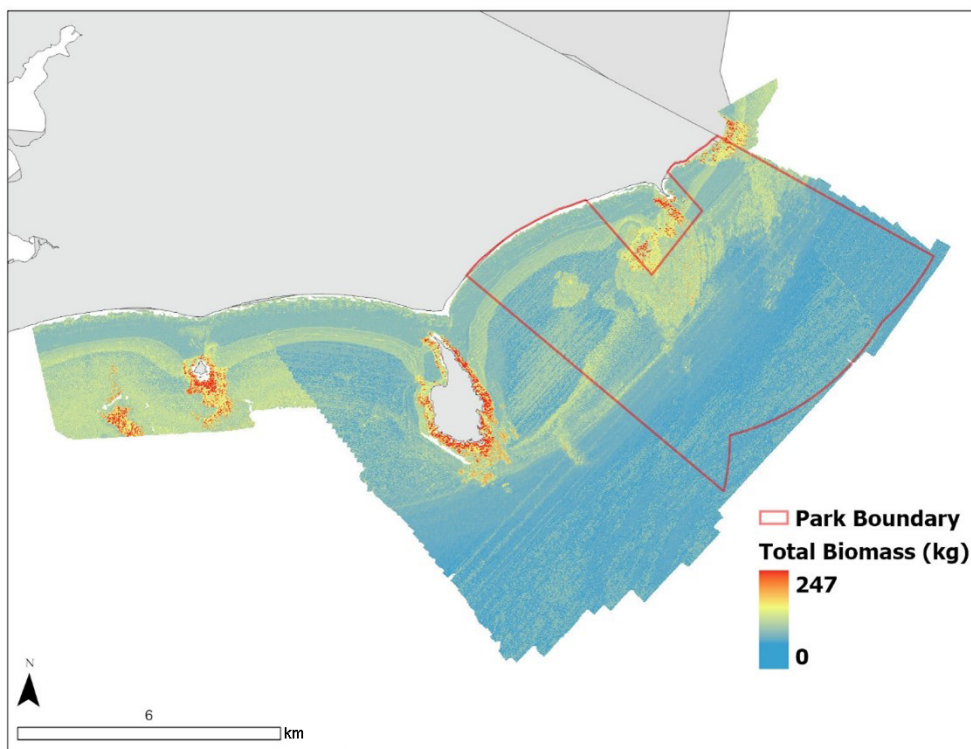


Figure 3.29: Predicted biomass of all species in 2021 from BRUVS sampling across the whole study site. The random forest model explained 22.0% of the variance and had a pseudo- R^2 value of 0.80

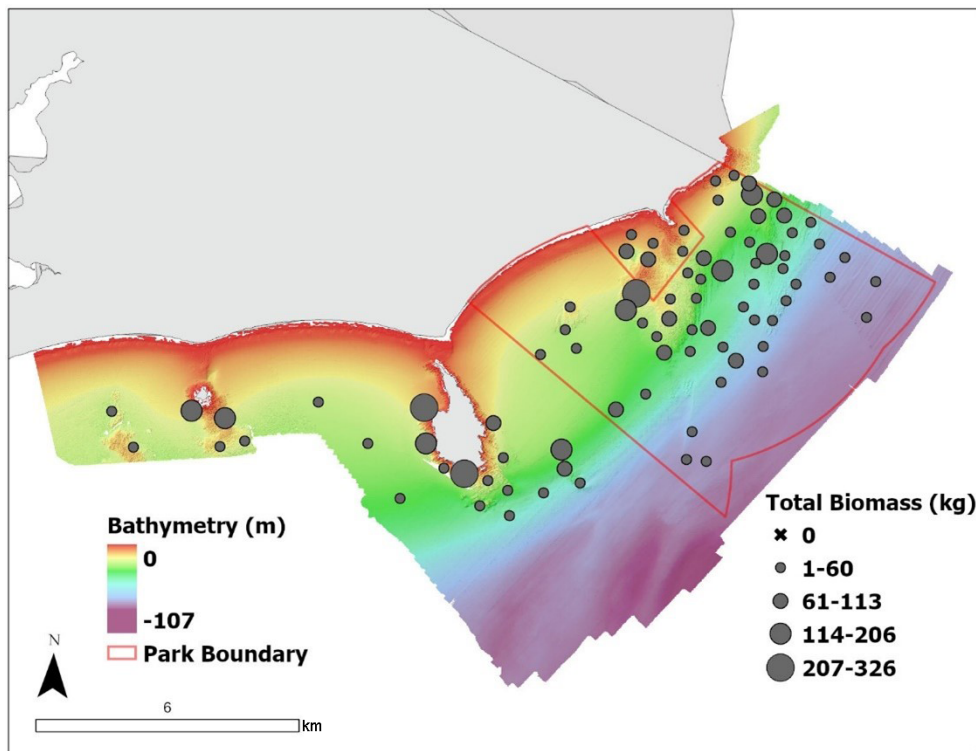


Figure 3.30: Total biomass for all BRUVS deployments in 2021. The size of each site marker corresponds to the total biomass in kilograms of fish observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

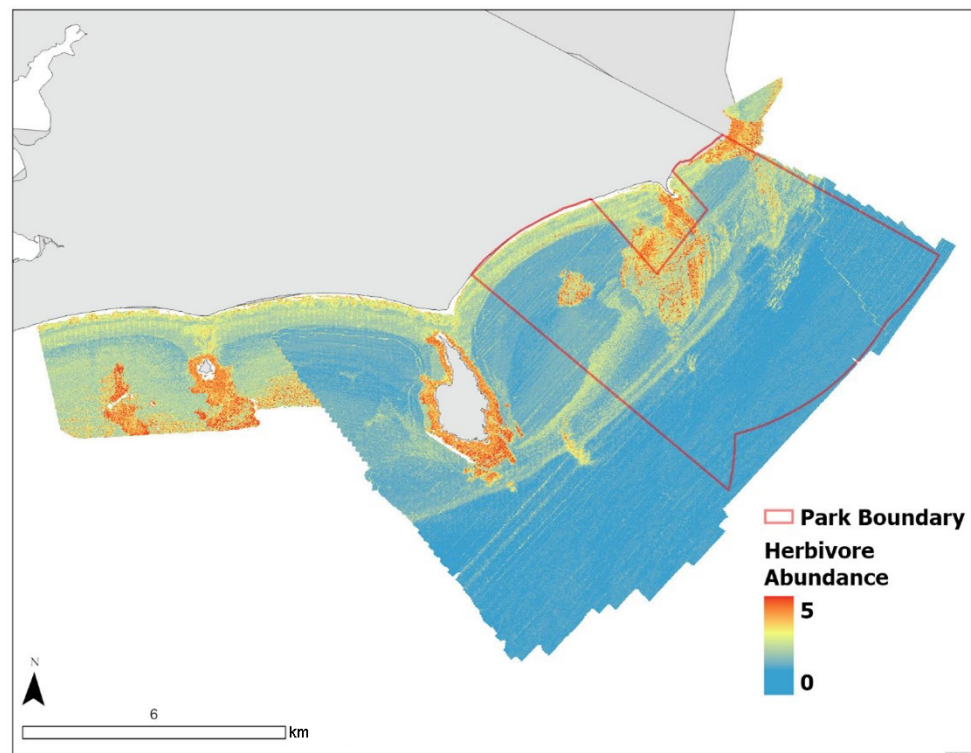


Figure 3.31: Predicted herbivore abundance in 2021 from BRUVS sampling across the whole study site. The random forest model explained 32.1% of the variance and had a pseudo- R^2 value of 0.87

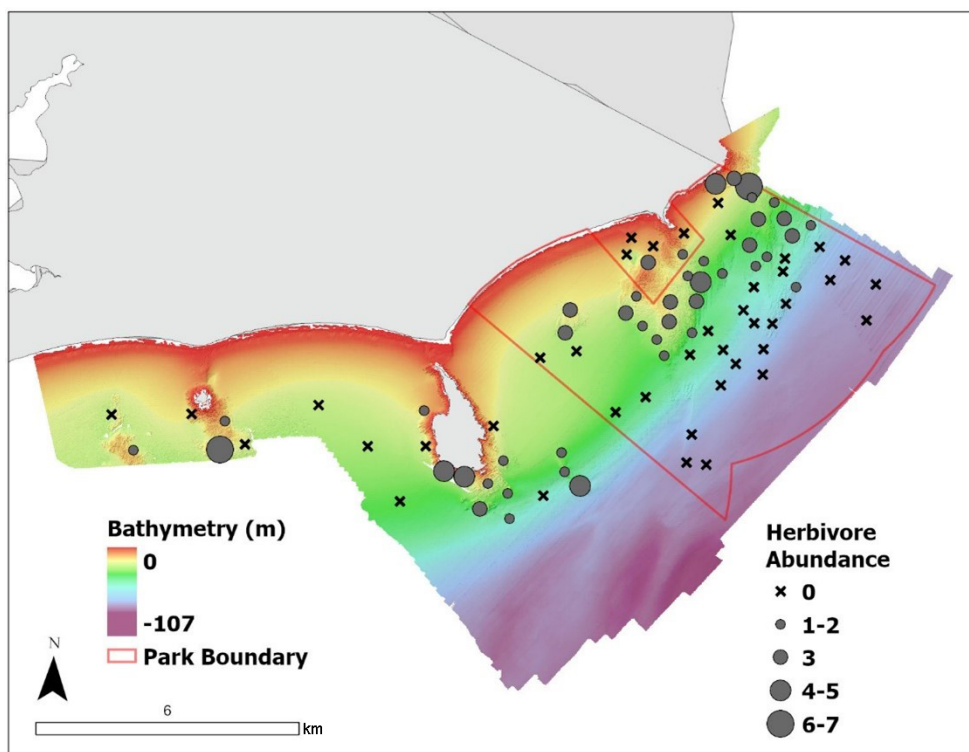


Figure 3.32: Herbivore abundance for all BRUVS deployments in 2021. The size of each site marker corresponds to the herbivore abundance of fish observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

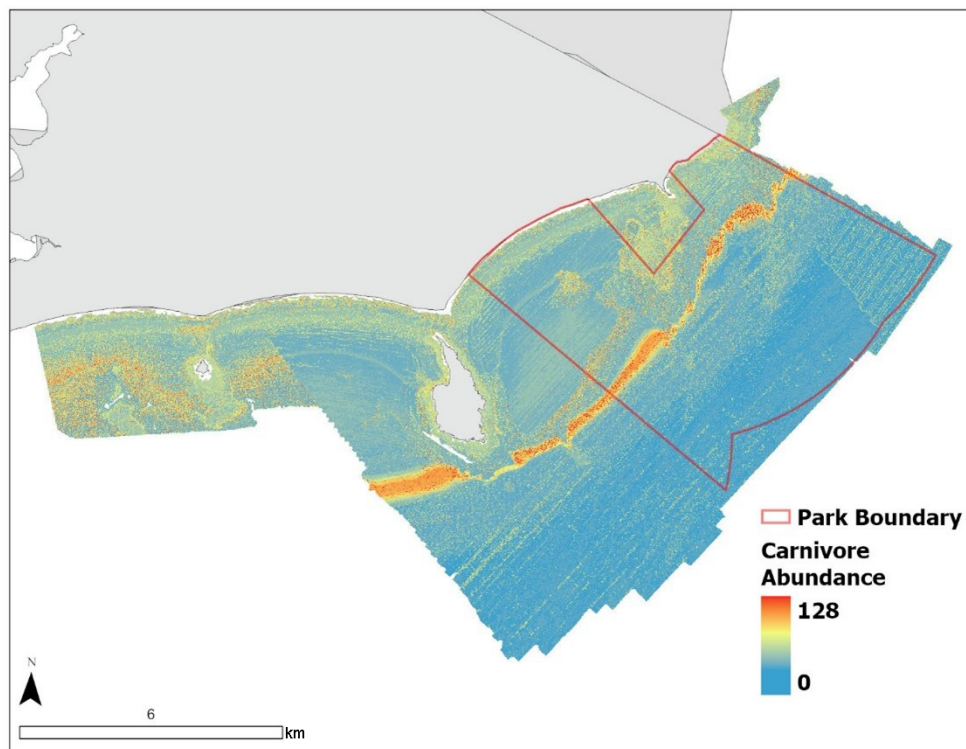


Figure 3.33: Predicted carnivore abundance in 2021 from BRUVS sampling across the whole study site. The random forest model explained 29.5% of the variance and had a pseudo- R^2 value of 0.03

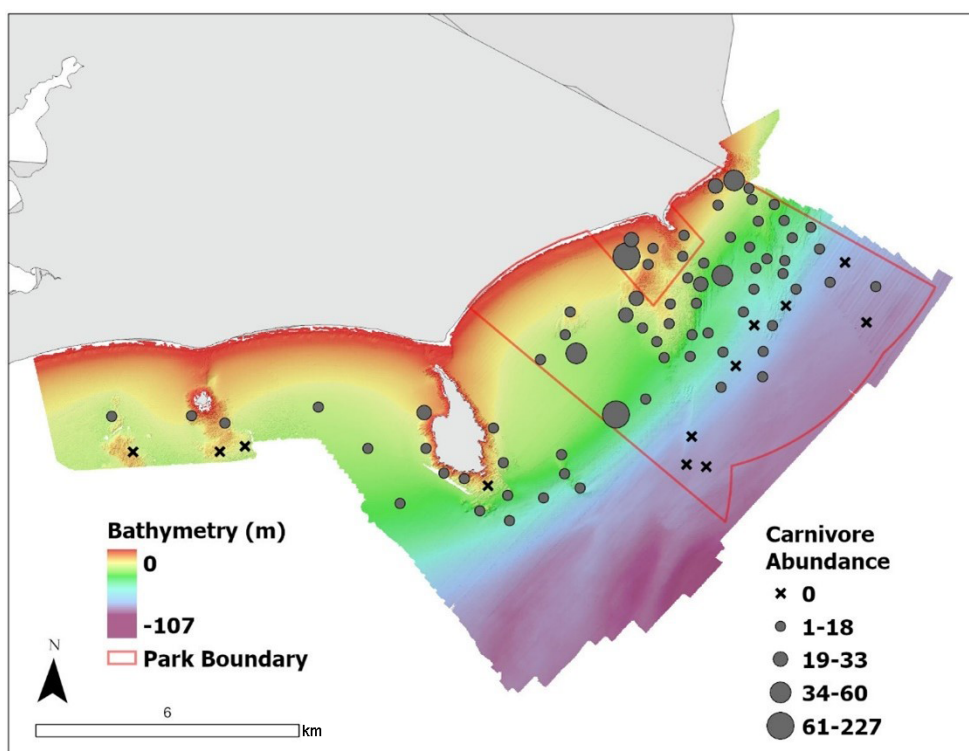


Figure 3.34: Carnivore abundance of fish for all BRUVS deployments in 2021. The size of each site marker corresponds to the carnivore abundance of fish observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

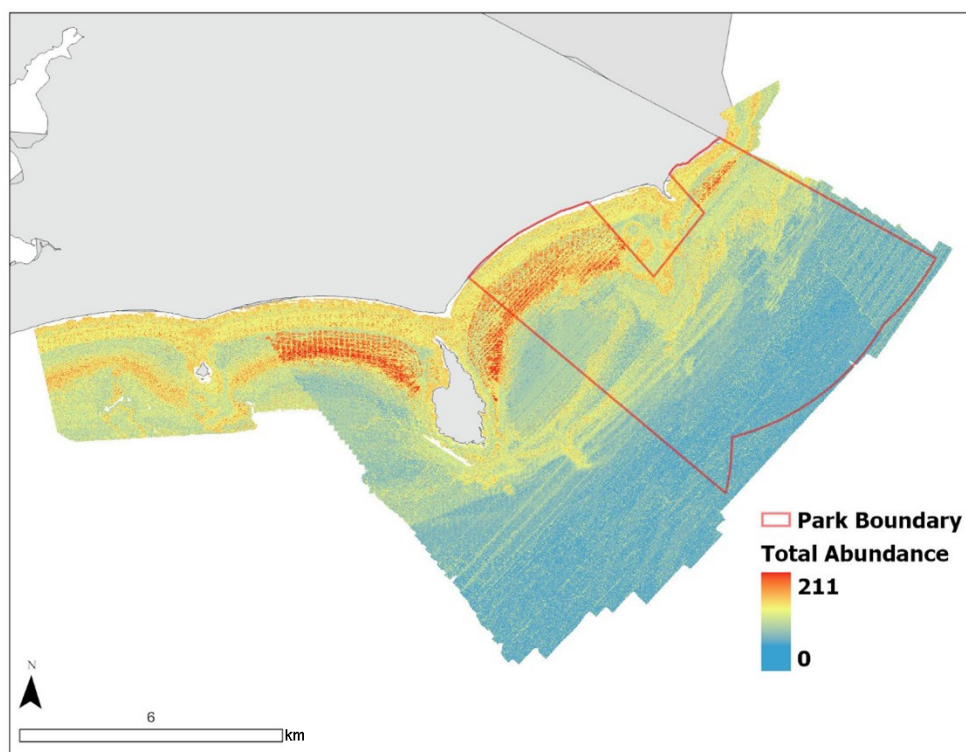


Figure 3.35: Predicted invertivore abundance in 2021 from BRUVS sampling across the whole study site. The random forest model explained 29.5% of the variance and had a pseudo- R^2 value of 0.83

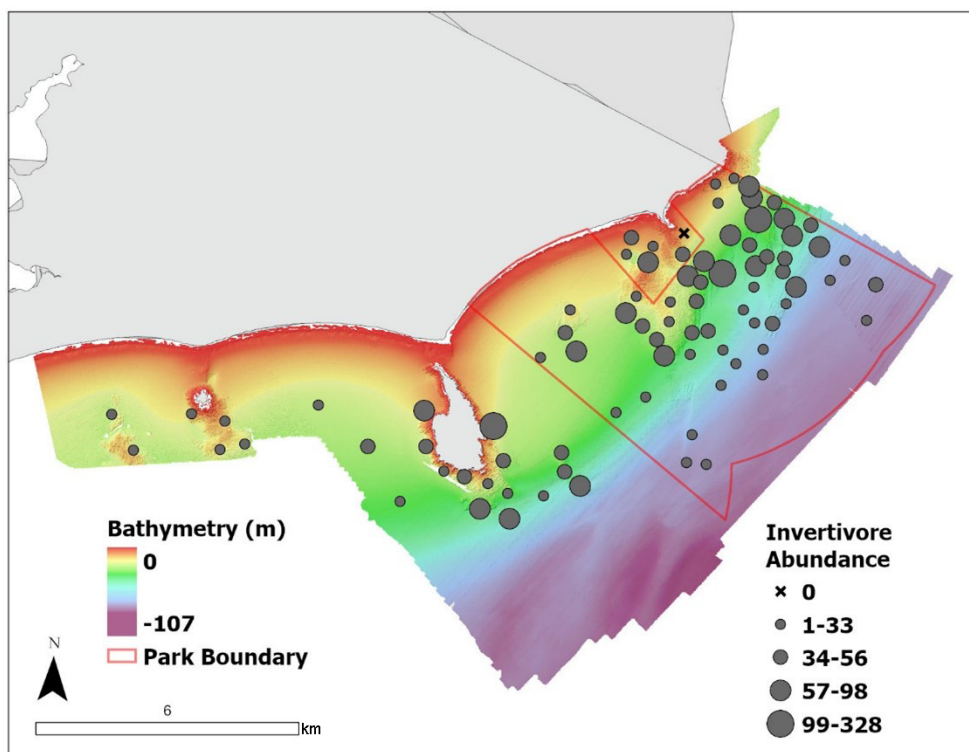


Figure 3.36: Invertivore abundance of fish for all BRUVS deployments in 2021. The size of each site marker corresponds to the invertivore abundance of fish observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

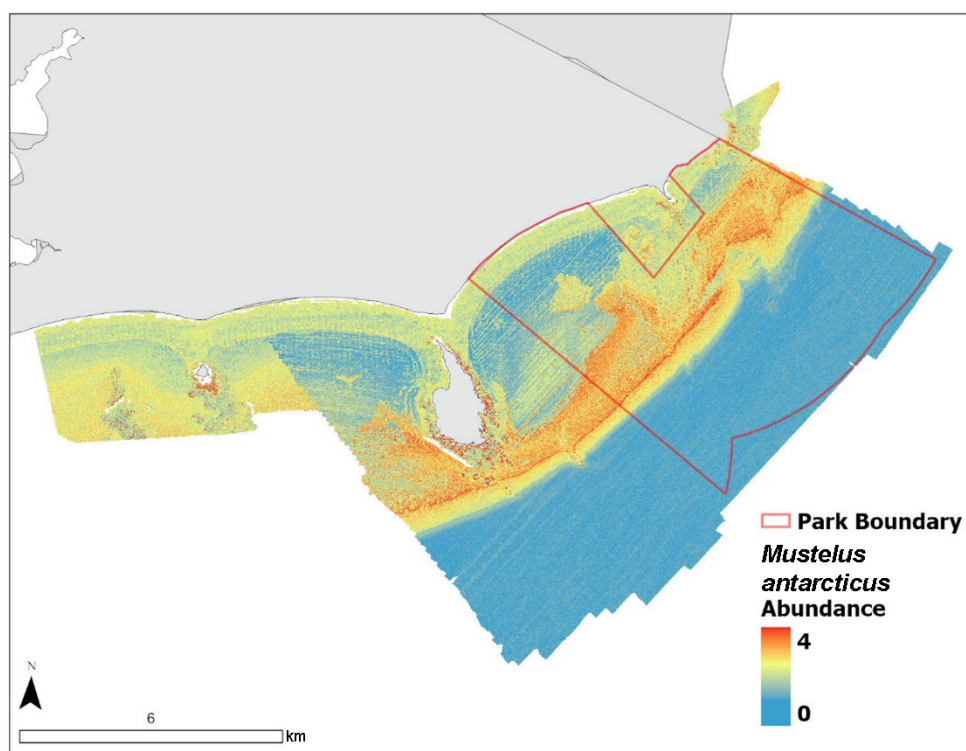


Figure 3.37: Predicted abundance of *Mustelus antarcticus* in 2021 from BRUVS sampling across the whole study site. The random forest model explained 17.6% of the variance and had a pseudo- R^2 value of 0.13

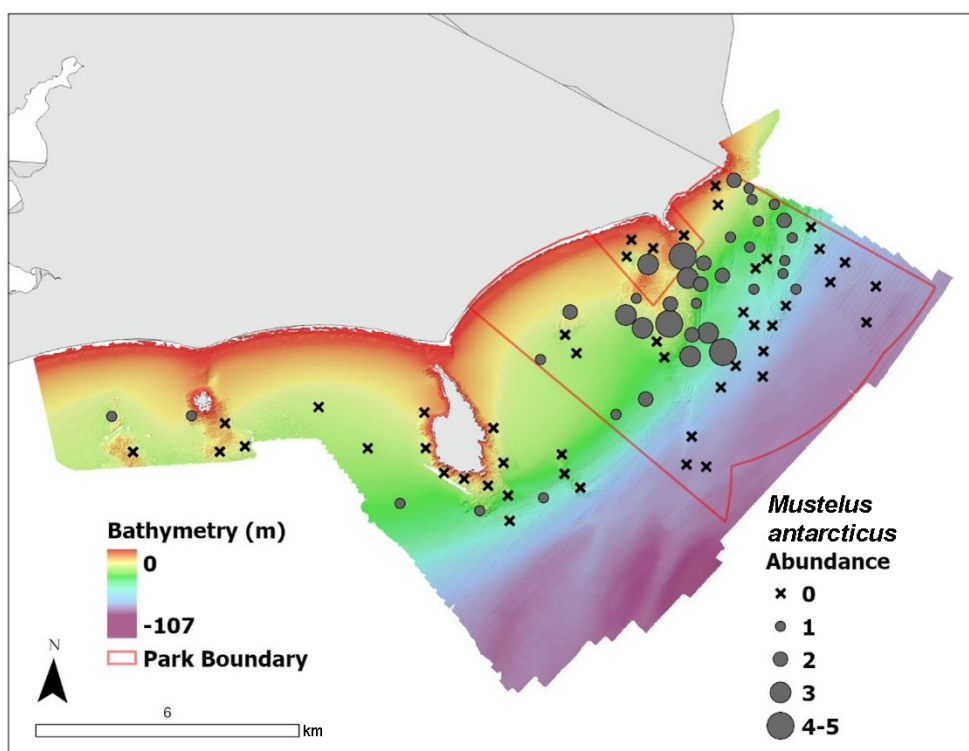


Figure 3.38: Abundance of *Mustelus antarcticus* for all BRUVS deployments in 2021. The size of each site marker corresponds to the abundance of *M. antarcticus* observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

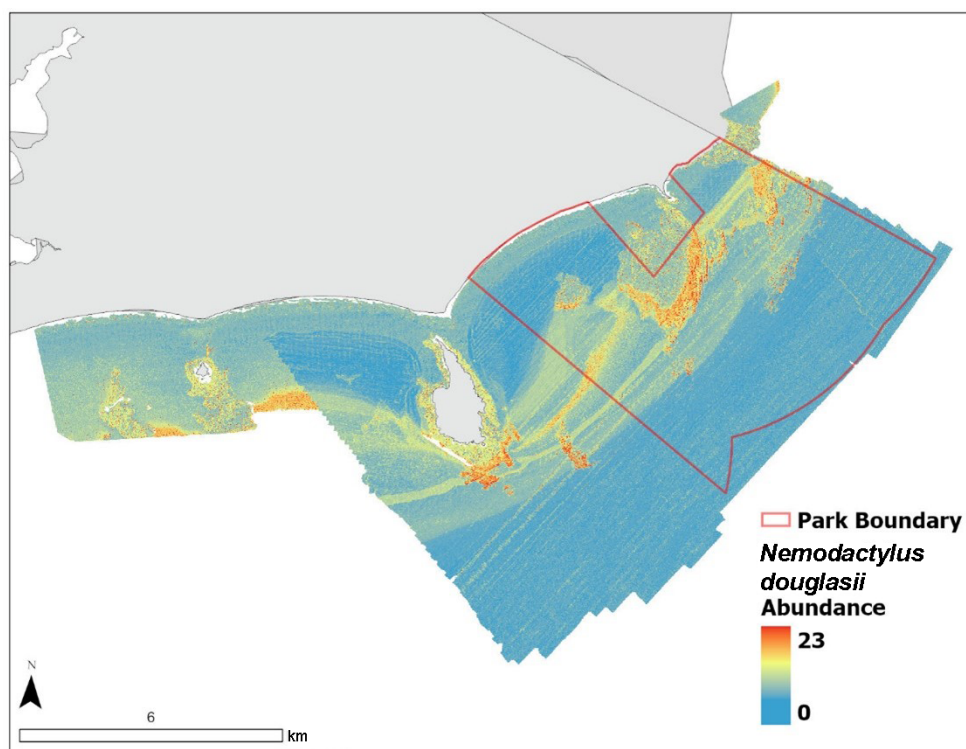


Figure 3.39: Predicted abundance of *Nemadactylus douglasii* in 2021 from BRUVS sampling across the whole study site. The random forest model explained 17.0% of the variance and had a pseudo- R^2 value of 0.07

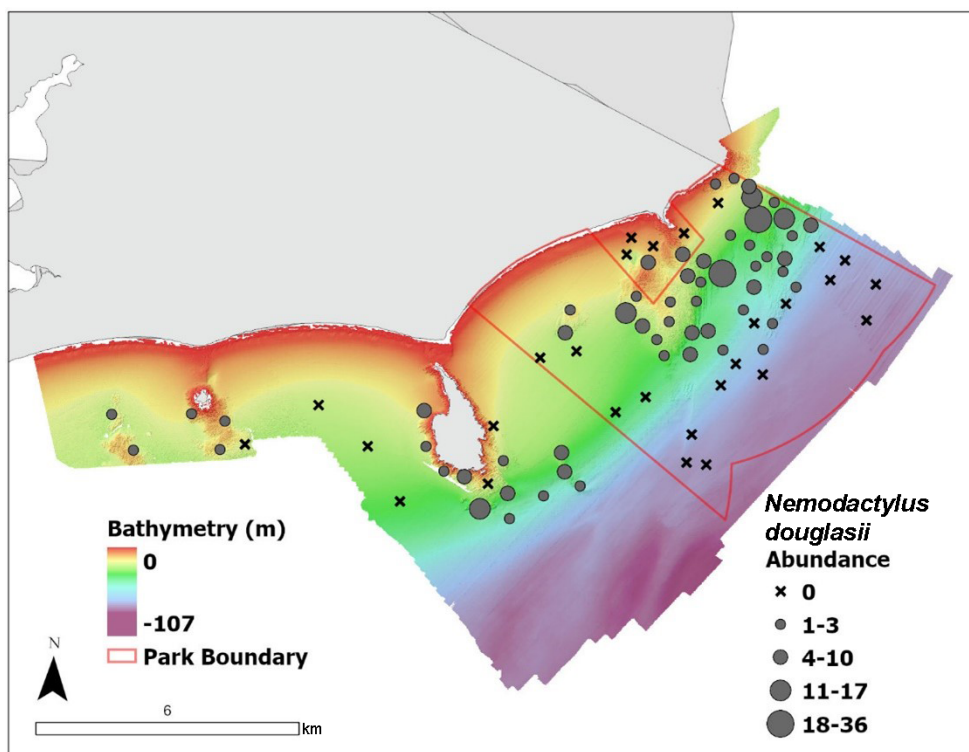


Figure 3.40: Abundance of *Nemadactylus douglasii* for all BRUVS deployments in 2021. The size of each site marker corresponds to the abundance of *N. douglasii* observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

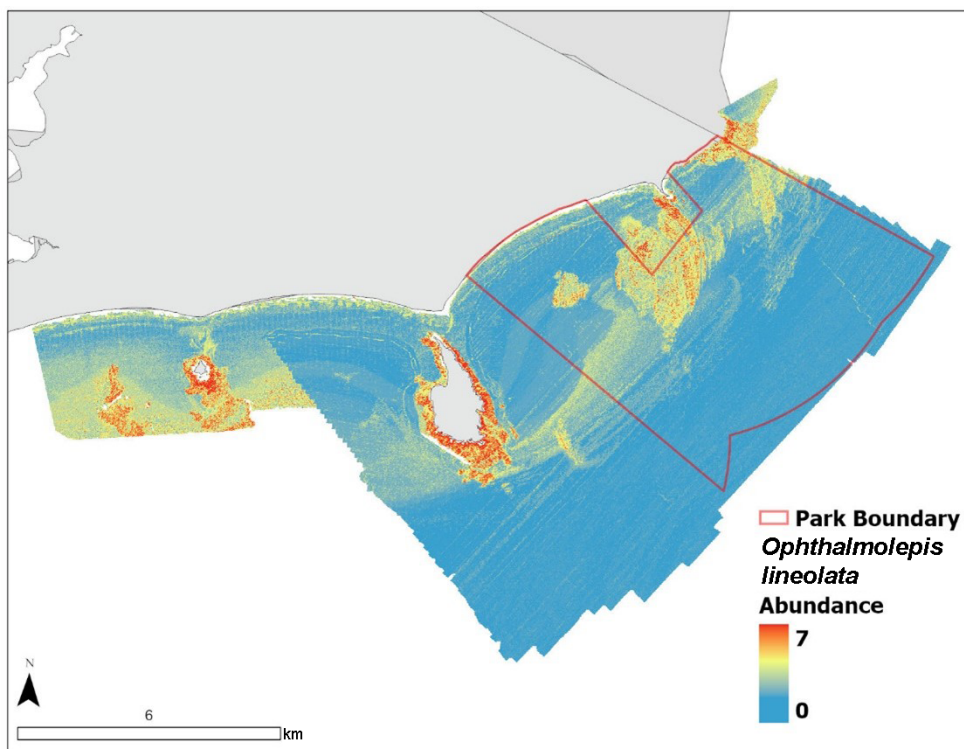


Figure 3.41: Predicted abundance of *Ophthalmolepis lineolata* in 2021 from BRUVS sampling across the whole study site. The random forest model explained 38.8% of the variance and had a pseudo- R^2 value of 0.51

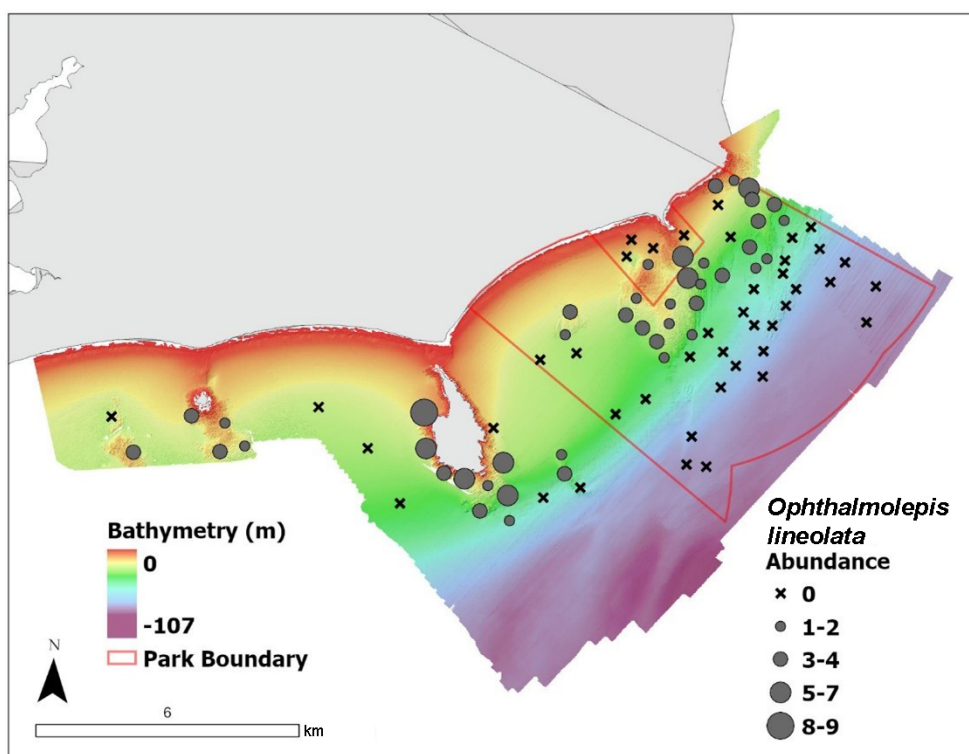


Figure 3.42: Abundance of *Ophthalepis lineolata* for all BRUVS deployments in 2021. The size of each site marker corresponds to the abundance of *O. lineolata* observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

3.4 Fisheries-independent Southern Rock Lobster survey

A total of 145 spatially referenced lobster pots were placed inside (85) and outside (60) the marine national park between 28 February and 4 March 2021. Seventy-six Southern Rock Lobster (*Jasus edwardsii*), with an estimated total biomass of 79.5 kg, were caught and examined during this study. Sixty-three lobsters were caught within the MPA compared with 13 caught in fished waters (Figure 3.43, Table 3.10). Lobsters caught within the park were not found to be larger on average than those caught in fished areas (Figures 3.44, 3.45).

In all 4 bubble plots of male and female abundance and biomass, locations where no lobsters were caught were typically relatively low-profile and potentially sand-inundated reef (Figures 3.46–3.49). Clear differences can also be observed inside and outside the MPA: pot locations inside the MPA caught more lobsters than did those outside the MPA.

Results of generalised additive models showed varying success in understanding the environmental drivers of various groupings of *J. edwardsii* (Table 3.11). Deviances of approximately 69% and 64%, respectively, were explained by models of the number of individuals per pot lift and total biomass per pot lift. Number of female individuals per pot lift had the highest deviance explained at 72%. Performance of models of number of male lobsters per pot lift was poor, with a deviance explained of 54%. Depth and eastness were significant in explaining the variation in all groups tested. Inside MPA was also significant in explaining variation in number of individuals and total biomass per pot lift, for both all lobsters and subsets of female lobsters. These findings were strongly reflected by trends observed in the bubble plots (Figures 3.46–3.49).

Overall, bycatch numbers were relatively high in this study, with an average of 2.67 organisms being caught per pot. Hermit crabs (*Diogenidae* spp., 20.9%), Velvet Leatherjacket (*Meuschenia scaber*, 18.6%) and seaperch (*Helicolenus* spp.) (12.9%), made up the vast majority of all bycatch (Table 3.12). All species were returned to the water without harm. Slightly more bycatch were caught inside the MPA ($n = 228$) than outside ($n = 160$).

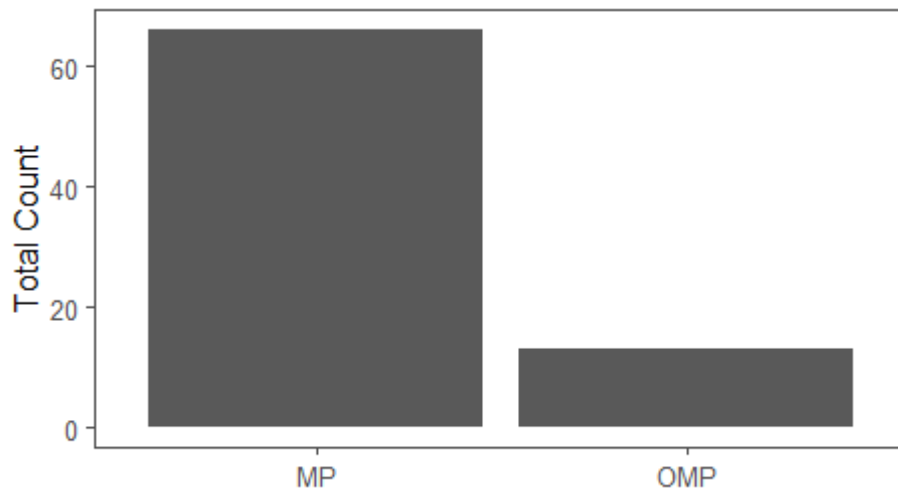


Figure 3.43: Total lobster counts from potting surveys within the Cape Howe MNP in protected (MP) and non-protected (OMP) areas

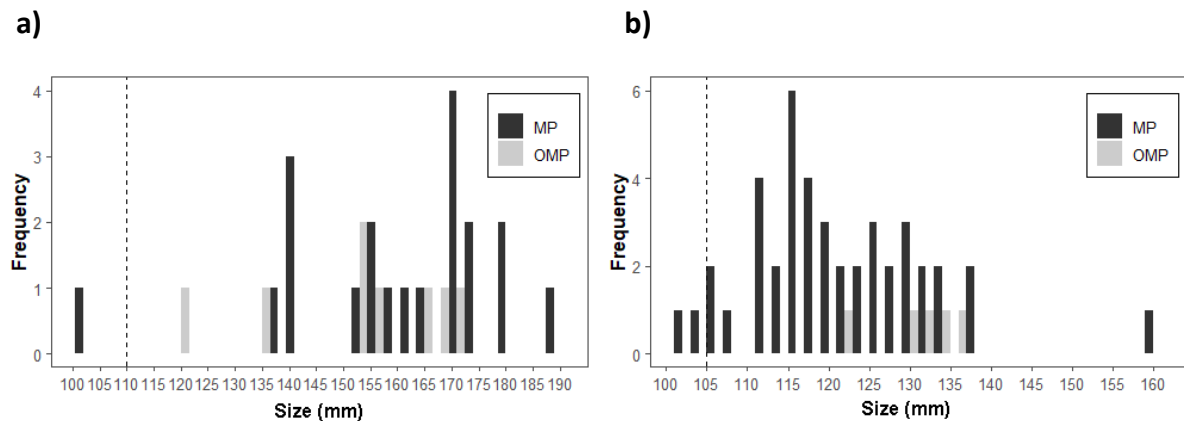


Figure 3.44: Size distributions (carapace length in mm) of (a) male and (b) female *Jasus edwardsii* inside (MP) and outside (OMP) Cape Howe MNP. The length of legal size for males and females is displayed as a dashed grey line in each distribution plot

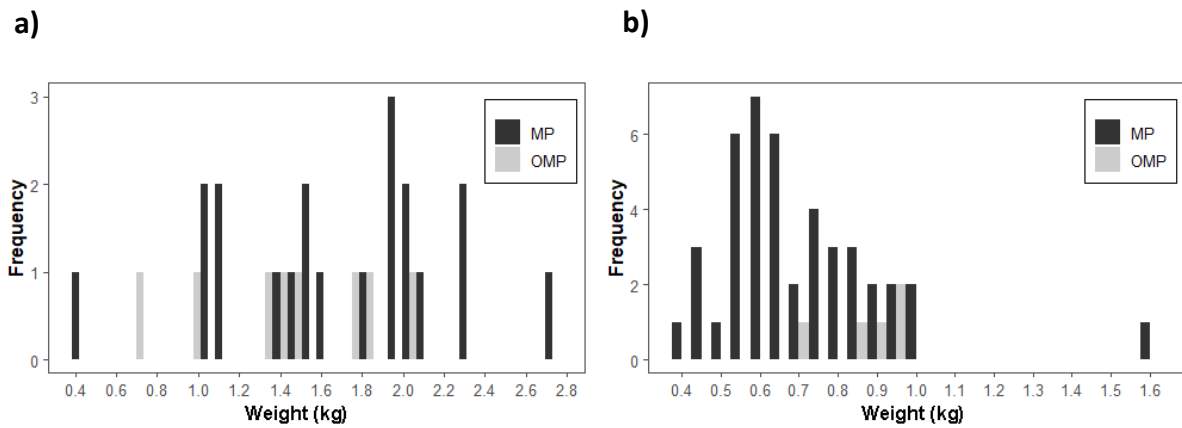


Figure 3.45: Weight distributions of (a) male and (b) female *Jasus edwardsii* inside (MP) and outside (OMP) Cape Howe MNP (total weight in kg)

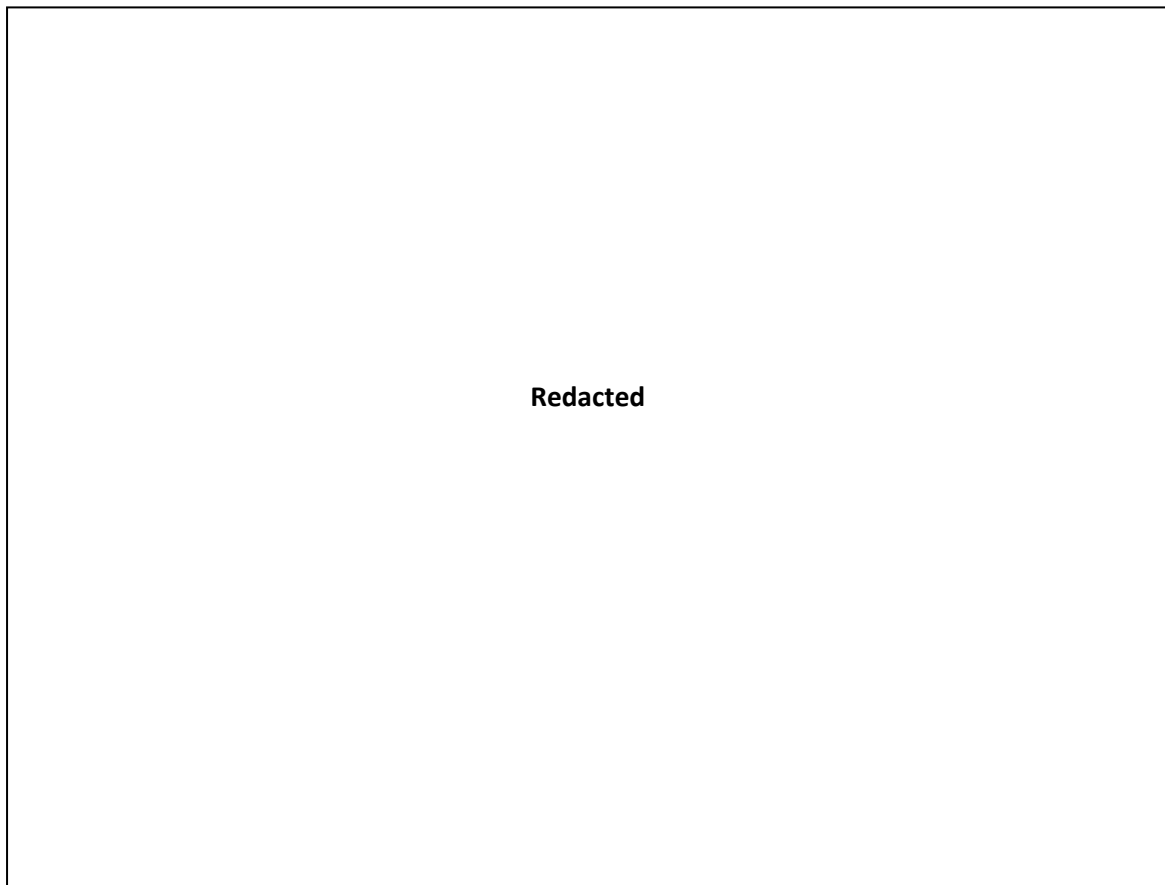


Figure 3.46: Total abundance of *Jasus edwardsii* from all lobster pot deployments. The size of each site marker corresponds to the total abundance of *J. edwardsii* observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

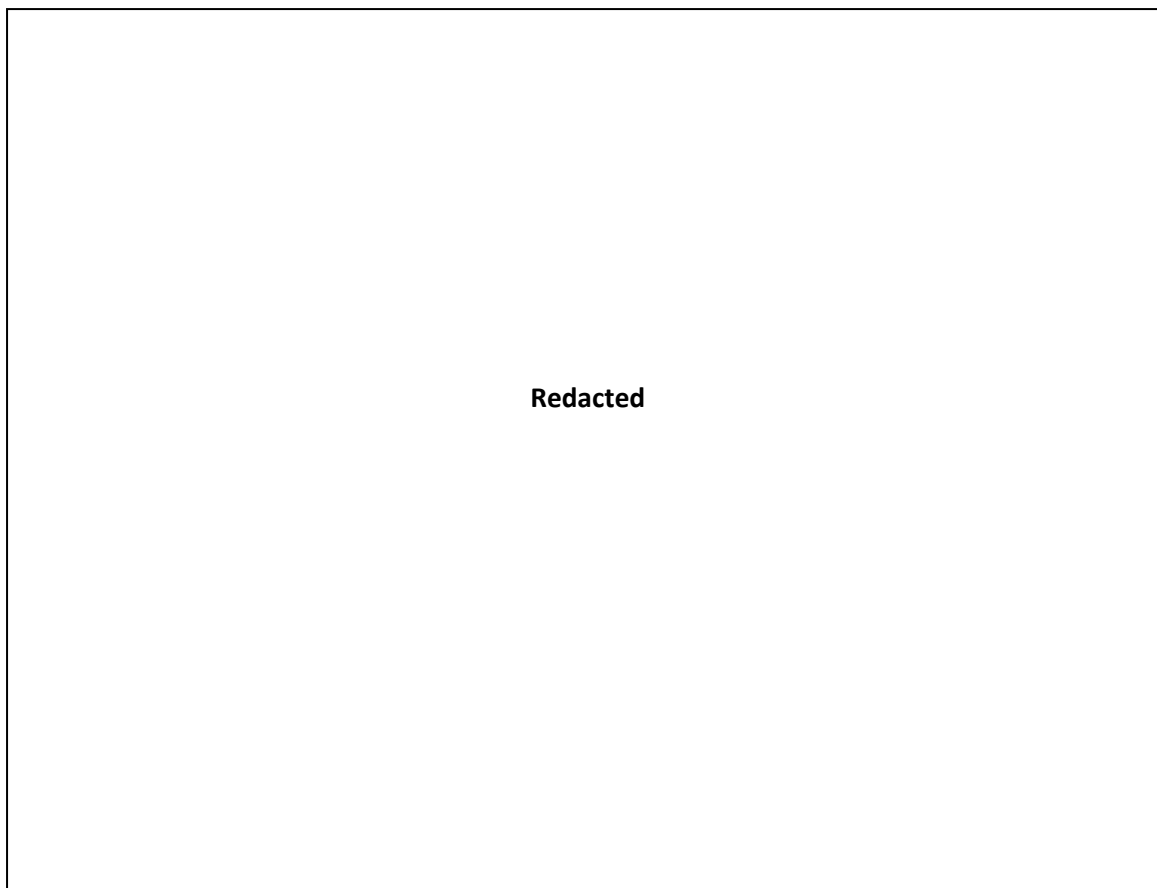


Figure 3.47: Abundance of female *Jasus edwardsii* for all lobster pot deployments. The size of each site marker corresponds to the abundance of female *J. edwardsii* observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

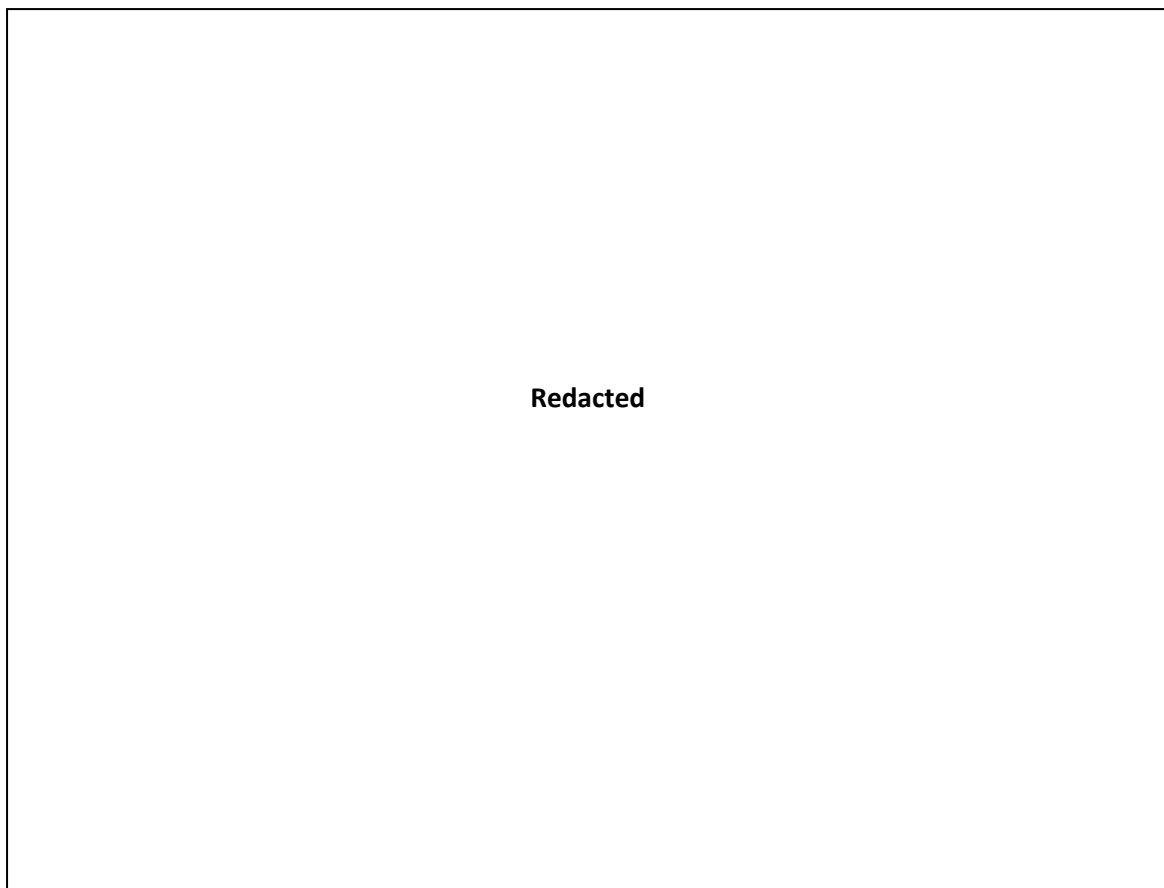


Figure 3.48: Abundance of male *Jasus edwardsii* from all lobster pot deployments. The size of each site marker corresponds to the abundance of male *J. edwardsii* observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

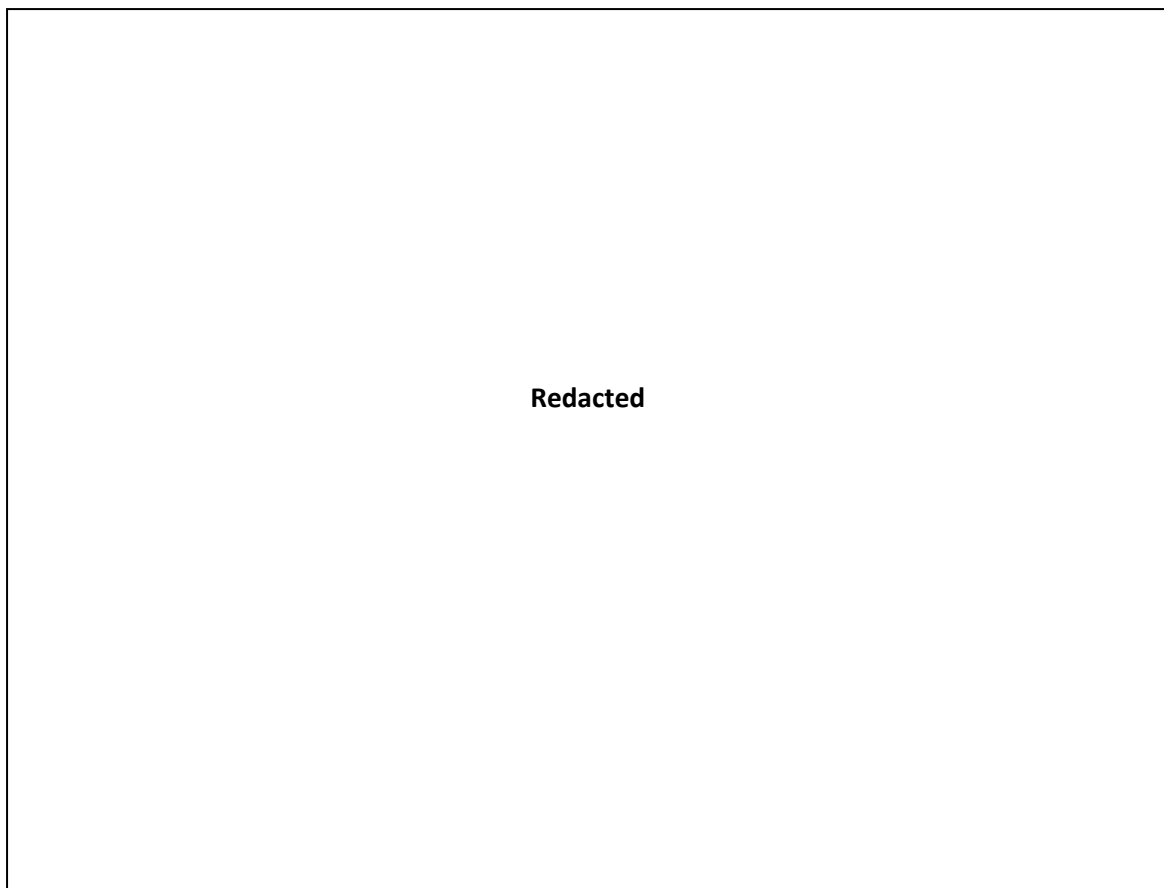


Figure 3.49: Total biomass of *Jasus edwardsii* for all lobster pot deployments. The size of each site marker corresponds to the total biomass (kg) of *J. edwardsii* observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

Table 3.10: Comparison of Southern Rock Lobster statistics inside ($n = 63$) and outside ($n = 13$) Cape Howe MNP

Protection status	Sex	Number of individuals	Number of legal size individuals	Pre-recruit index	Average carapace length (mm)	SD of average carapace length	Biomass (kg)	Average weight (kg)	SD of average weight	Number of reproductive females
Inside park	Male	20	19	0.05	160.0	19.7	33.4	1.7	0.6	
	Female	43	41	0.03	121.8	10.8	30.3	0.7	0.2	0
	Total	63	60	0.05	133.9	22.8	63.7	1.0	0.6	
Outside park	Male	8	8	0	152.5	16.9	11.4	1.4	0.4	
	Female	5	5	0	131.0	5.5	4.3	0.9	0.1	0
	Total	13	13	0	144.2	17.2	15.7	1.2	0.4	

Table 3.11: Summary statistics from generalised additive models (GAMs) used to associate Southern Rock Lobster population characteristics with habitat and protection status inside (MP) and outside (OMP) Cape Howe MNP. Best descriptor variables are identified by +

Group	Depth	Distance to reef	Curvature	Rugosity	Slope	BPI ¹	Eastness	Northness	OMP	MP	AIC ²	Deviance explained ³ (%)	RMSE ⁴
Number of individuals per pot lift	+	+					+			+	145.0	69	5.0
Total biomass per pot lift	+	+	+				+			+	158.3	64	4.4
Number of females per pot lift	+						+			+	107.9	72	5.8
Number of males per pot lift	+						+			+	86.0	54	4.3

Notes:

1. BPI = bathymetric position index
2. AIC = Akaike information criterion
3. Deviance explained is an indication of the model's goodness-of-fit
4. RMSE = root mean squared error

Table 3.12: Bycatch observed from lobster potting using research pots with no escape gaps in the Cape Howe MNP and adjacent fished reference locations. All bycatch observed was identified, measured and promptly return to the water

Bycatch species	MPA	Fished
Six-Spine Leatherjacket (<i>Meuschenia freycineti</i>)	1	2
Old Wife (<i>Enoplosus armatus</i>)	1	0
Wobbegong (<i>Orectolobus</i> spp.)	3	5
Port Jackson Shark (<i>Heterodontus portusjacksoni</i>)	2	3
Octopus (Octopodidae spp.)	3	0
Jackass Morwong (<i>Nemadactylus macropterus</i>)	7	4
Starfish (Echinodermata spp.)	0	1
Velvet Leatherjacket (<i>Meuschenia scaber</i>)	41	31
Grey Morwong (<i>Nemadactylus douglasii</i>)	4	0
Ocean Jacket (<i>Nelusetta ayraud</i>)	20	11
Degen's Leatherjacket (<i>Thamnaconus degeni</i>)	2	0
Toothbrush Leatherjacket (<i>Acanthaluteres vittiger</i>)	0	1
Eastern Wirrah (<i>Acanthistius ocellatus</i>)	8	7
Rosy Wrasse (<i>Pseudolabrus rubicundus</i>)	1	1
Mado Sweep (<i>Atypichthys strigatus</i>)	2	2
Maori Wrasse (<i>Ophthalmolepis lineolata</i>)	6	2
Cobbler or Southern Red Scorpion (<i>Scorpaena</i> spp.)	1	3
Green Moray (<i>Gymnothorax prasinus</i>)	2	8
Bearded Cod (<i>Pseudophycis barbata</i>)	8	1
Queen Snapper (<i>Nemadactylus valenciennesi</i>)	1	0
Red Bait Crab (<i>Guinusia chabrus</i>)	1	0
Snapper (<i>Pagrus auratus</i>)	16	1
Sweep (<i>Scorpiis</i> spp.)	3	4
Blue-Throat Wrasse (<i>Notolabrus tetricus</i>)	15	15
Black Banded Seaperch (<i>Hypoplectrodes annulatus</i>)	2	3
Barber Perch (<i>Caesioperca rasor</i>)	0	1
Seaperch (<i>Helicolenus</i> spp.)	30	20
Cod species (<i>Moridae</i> spp.)	0	1
Hermit crab (Diogenidae spp.)	48	33

3.4.1 Urchin barren predictive model

The MaxEnt model predicts well the relationship between urchin barren presence and environmental variables with an area under curve value of 0.875. Distance to reef was found to be the most important variable explaining the distribution of urchin barrens (influence of 77%) followed by slope (12%), depth (10%) and, finally, MPA status (1%). Urchin barrens were most likely to occur on reef habitat, on more complex (sloping) terrain, in shallower depths and in areas outside the MPA. The probability of occurrence and predictive distribution of urchin barrens across the Cape Howe MNP and the adjacent area are shown in Figure 3.50. Overall, it was predicted that the study area contains 6.3 km² of likely urchin barrens.

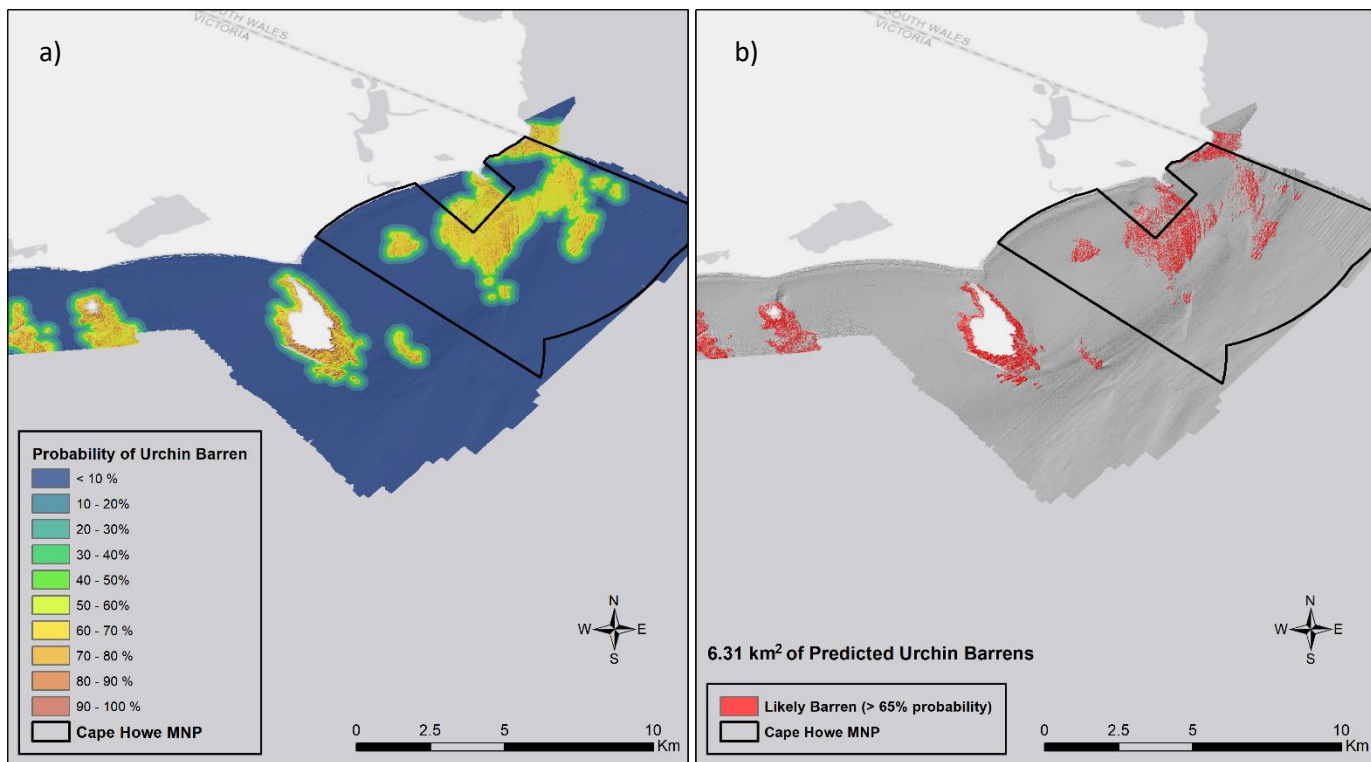


Figure 3.50: (a) Probability of occurrence of urchin barrens and (b) distribution of urchin barrens (based on >65% probability of occurrences)

4. Discussion

4.1 Benthic communities

This study observed benthic communities using a range of data sources. An underwater visual census (UVC) approach was previously implemented as part of the Shallow Reef Monitoring Program (SRMP) across 11 time series from 2001 to 2014 (Woods et al. 2014) and continued using the Reef Life Survey (RLS) underwater visual census methodology in 2021. These surveys have provided long-term data spanning 20 years that allow an understanding of trends through time. BRUVS surveys undertaken in 2006 and 2021 provide a limited time series to understand changes in time but allow assessment of biotic habitat in deeper water inaccessible to UVC.

Overall, control charts based on UVC methods showed that macroalgae are in fair condition. In contrast, encrusting coralline algae has had low percentage cover since 2015 within the MPA and is below the lower limit of acceptable change. Bare stipes of Crayweed (*Phyllospora comosa*) were observed in 2008 and 2015. SRMP surveys previously revealed that coverage of *P. comosa* has declined since 2001, although there was some evidence of increased cover in 2015. Since then, there have been some increases in *P. comosa*, with percentage cover within the MPA well above the LLAC in 2021. The driver of this increase is unclear but may indicate reduced action of urchin grazing in shallower waters or greater recruitment success between 2015 and 2021.

BRUVS surveys found a change in habitat structure from 2006 to 2021 with an increase in biogenic pebbles, sponges and coarse sand and a decrease in bare rock and cobble. These changes may be the result of a true habitat shift (e.g. an increase in sponge cover) or may be a shift to biogenic pebbles and coarse sand because of changes to seafloor composition (e.g. through movements via currents). Alternatively, the changes may reflect a shift in the viewpoint of the BRUVS. Although BRUVS were placed on the seafloor in geographically the same area, they may have had different viewpoints, for example, south-facing or north-facing. In heterogeneous habitats, this can result in different habitats being assessed.

The implications of the habitat change on the fish and invertebrate communities is currently not well understood. The DISTLM analysis identified sponge cover as an important factor that influences the composition of fish assemblages, likely differentiating between soft-sediment and hard-bottom (reef) fish communities. This could help explain the increase in abundance and species richness observed from 2006 to 2021.

Although around twice as many BRUVS samples were analysed in 2006 than in 2021, the number of urchin barrens observed was similar (17 vs. 21). The similarity in number is probably due to the prioritisation of reef habitat in the sampling design in 2021, the overall limited amount of reef habitat within the sampling area that can be surveyed, and the extension of BRUVS surveys outside the MPA in 2021, including likely barren reefs. All these factors increase the likelihood of observing barrens with less sampling effort. The matched sampling effort showed a slight increase in the number of barrens observed (3 in 2006 vs. 5 in 2021). This may indicate that the urchin barrens have become more extensive in deeper waters over the 15-year period, or it could be another effect of BRUVS viewpoint differences. Conversely, the RLS surveys show a decrease in abundance of urchins and increased canopy-forming macroalgae cover in recent years. This pattern may indicate that the shallower areas serve as a refuge from the impacts of urchins.

The BRUVS and RLS imaging methods implemented in this study played different roles in providing an understanding of the distribution and change in benthic communities within Cape Howe MNP.

Photo quadrats captured during underwater visual census helped to achieve the highest image clarity of all methods; however, the constraints of diving meant that the spatial extent of these surveys was limited. The UVC approach also had the least spatially explicit position information, as it is identified by a centroid of the site locality rather than individual referenced images, as well as limited spatial coverage across the site.

4.2 Large mobile fish (including sharks and rays)

Populations of 6 of the 12 key mobile fish species are currently considered to be healthy; their abundances were found to be within the zone of 'good' condition. The remaining 6 are in poor condition, 4 due to a low number of observations within the MPA and 2 due to the small size of individuals observed. Three of the species in poor condition due to low abundance were not observed using UVC within the MPA from 2015: Old Wife (*Enoplosus armatus*), Eastern Blue Groper (*Achoerodus viridis*) and Six-Spine Leatherjacket (*Meuschenia freycineti*). These species were all observed in the 2021 BRUVS sample; however, *E. armatus* was the only species whose abundance increased from 2006 to 2021. Crimsonband Wrasse (*Notolabrus gymnogenis*), the other species rated in poor condition due to low abundance, was also observed in the 2021 BRUVS samples; it increased in abundance from only one observation in 2006 to 6 observations in 2021. Two other species, Banded Morwong (*Chirodactylus spectabilis*) and Senator Wrasse (*Pictilabrus laticlavus*), were classified as poor condition due to the low mean length measurements recorded in recent UVC surveys.

The species identified to be contributing to the differences between years from the BRUVS analysis included 7 species that had higher abundance in 2021, 2 species which were similarly abundant in both years and one species, *M. freycineti*, that declined in abundance from 2006 to 2021. This trend for *M. freycineti* was also observed in the UVC surveys, in which it has not been observed since 2012. Six out of the 10 species that influenced the difference between inside and outside the marine park for the 2021 BRUVS sampling had higher abundances outside the marine park, while 4 species were more abundant within the protected area. Two of those 4 species, Ocean Jacket (*Nelusetta ayraud*) and Gummy Shark (*Mustelus antarcticus*), are considered fisheries-targeted species.

Species that are pressured from extractive fishing activities are generally expected to benefit from marine protected areas (Halpern and Warner 2002, Lester et al. 2009). Flow-on effects can also impact the abundance of non-target species, which can either increase or decrease (McLaren et al. 2015). Only 2 of the 6 species in poor condition due to size or abundance are likely to experience fishing pressure when outside protected areas: *A. viridis* and *C. spectabilis*. These species would be expected to have increased size and abundance in the protected area. However, they were both found to have lower size inside the park, and *A. viridis* also had lower abundance. These results are inconsistent with a beneficial effect of protection. We would also expect similar declines for these species in reference locations if they were being influenced by widespread environmental changes or natural abundance fluctuations, but this is not observed in the UVC datasets. Overall, the declines in these species in the park could be due to the park having unfavourable habitat conditions, generally unfavourable environmental conditions within the region, natural variations in species abundance or ineffective protection provided by the protected area because of either park size, lack of connectivity between parks, or lack of enforcement of the protected area. We have insufficient ecological information for many of the declining species to explain why the changes to abundance are occurring, and this is an area requiring further investigation.

This study let us use baited remote underwater video stations to establish time series fish assemblage monitoring over the entire depth range of Cape Howe MNP. While UVC techniques are

known to sample a different set of species to those sampled by BRUVS, and in particular to sample cryptic species (Colton and Swearer 2010, Lowry et al. 2012), using BRUVS in this study greatly increased the depth range of observations. This greater depth range increased the number of species that could be observed by also incorporating species only found outside diver depths (BRUVS added 40 species not detected by UVC methods in this study). Continuing to use BRUVS for monitoring MPAs through time will allow time series analyses similar to those performed for diver surveys, including creating control charts. While there are not currently enough data to complete those types of analyses, this sampling will serve as baseline information for future exploration.

Including BRUVS in monitoring also allowed the creation of spatially explicit species distributions for various diversity metrics and abundance of key groups and species throughout the park, aided by the distribution of sample locations across observed environmental gradients. Reaching the same density of field locations using UVC methods would not be feasible. Our approach, driven by a solid foundation understanding of the seabed structure provided by MBES/LiDAR technologies, allows for a spatially balanced design across the environmental variability of the site. An advantage of distribution modelling approaches, such as those used in this study, is the ability to predict patterns in abundance and biodiversity beyond sampled locations if relationships with environmental drivers can be inferred (Araújo and Guisan 2006, Sequeira et al. 2016). Distribution modelling can be relevant for a range of applications such as tracking invasive species and researching effects of climate change (Elith and Graham 2009). This study, for example, predicted hotspots of species richness and abundance across an area of over 107 km², including across the entire extent of Cape Howe MNP. Our findings identified strong relationships with environmental drivers for some species and community groups, but not for all groups. For example, the models for species richness and *Ophthalepis lineolata* abundance explained a significant amount of the variance in the relative abundance of that species and predicted its distribution with fairly good accuracy. On the other hand, some models, such as those for total biomass and *Nemadactylus douglasii* abundance, did not explain much of the variation and were not able to create high confidence predictions. These differences in results could relate to the habitat affinities of the different species, the prevalence of the species, the size of the samples, and/or the representativeness of the samples. Despite the shortcomings of some of these models, they can be used to help plan sampling in the future to target areas that are likely to improve the modelling capacity for the different species and/or groups. Baseline knowledge obtained from BRUVS surveys in this study can now be used by park authorities to target diversity assessments and to identify sites of high biodiversity or public interest not previously visited.

4.3 Mobile macroinvertebrates

4.3.1 Southern Rock Lobster

Results from generalised additive models (GAMs) used to associate Southern Rock Lobster (*Jasus edwardsii*) population characteristics with habitat and protection status inside and outside Cape Howe MNP indicate a significant association with being in the MPA. The results showed no significant association with being outside the MPA, likely due to the low number of individuals caught outside the MPA. Differences in abundance inside and outside the park may be driven by the lack of suitable habitat outside the park. Low numbers of rock lobsters were observed in shallow reefs with urchin barrens. Most rock lobster observations were associated with sponge-dominated circalittoral reefs. To better understand these spatial differences in productivity across the *J. edwardsii* populations in Victoria, and the potential benefit of MPAs for increasing productivity, we need more, and higher resolution, spatially explicit observation data across the network. Access to

these types of datasets would allow us to better understand how the environment and protection by MPAs affect *J. edwardsii* productivity.

Jasus edwardsii has a complex early life cycle and can spend 12 to 24 months within oceanic waterbodies undergoing 11 larval stages before settling (Thomas et al. 2000, Linnane et al. 2013). Previous studies have suggested that Australian *J. edwardsii* populations are responsible for some trans-Tasman larval flow that contributes to and possibly maintains New Zealand populations (Chiswell et al. 2003, Morgan et al. 2013). Our study, however, found a lack of sub-legal-size lobsters both inside and outside the MPA. This could suggest a reduced supply of recruits to this population, which could be monitored by repeating these surveys through time. Future monitoring, which could introduce juvenile-specific sampling methods such as puerulus collectors, could confirm this. The low number of pre-recruit rock lobsters could also arise from larger lobsters excluding the smaller lobsters from traps (Miller 1990, Frusher and Hoenig 2001). This may be unlikely given that low sample sizes meant many empty pots were available for occupancy by any juveniles present.

MPA are important for protecting source populations of Southern Rock Lobster. Studying populations inside MPAs allows us to better understand recruitment dynamics to assess how well they protect source populations and benefit areas outside MPA boundaries. The results from this study will act as an appropriate baseline for future studies documenting changes in this MNP to assess how the population responds to protection, recruitment and a changing climate over time. Continued sampling of this and other MPAs will provide a more conclusive understanding of how *J. edwardsii* is responding to complete removal of fishing pressure within MPA boundaries across the state.

4.3.2 Other key macroinvertebrates

Cape Howe MNP also provides refuge for a number of key macroinvertebrates such as the commercially important Blacklip Abalone (*Haliotis rubra*) and Turban Shell (*Lunella undulata*) (Woods et al. 2014). One possible cause of the consistent declines in these species over the past 20 years may be macroalgae habitats transitioning to urchin barrens in the deeper sections. This reduction in offshore macroalgae would reduce the drift algae available as a food source for the abalone. However, further investigation is required to better understand the drivers of declines in these macroinvertebrates and whether they are related to declines detected for other species in this study.

Overall, Tent Turban (*Astraliu tentoriiforme*), Dog Whelk (*Dicathais orbita*) and Purple Sea Urchin (*Heliocidaris erythrogramma*) conditions have been declining since 2001; all 3 species are now in poor condition. Given that 4 of the 7 characteristic macroinvertebrates have shown substantial decline, it is important to better understand the drivers of this decline, including the link between their decline and the decline in encrusting coralline algae and whether any management action could aid in the recovery of these species. Reasons for the reduction in macroinvertebrate abundance are currently unclear. Three of those 4 species are considered herbivorous while one, *D. orbita*, is predatory. It could be that a lack of suitable habitat or food supply is affecting the abundance of the grazing species, which in turn could affect *D. orbita*. A reduction in recruitment and changing recruitment patterns may also explain the trends in abundance. The east coast of Australia is experiencing changes in currents due to fluctuations in the East Australian Current as a result of climate change (Suthers et al. 2011, Cetina-Heredia et al. 2014). These changes may affect recruitment processes for macroinvertebrates, which often have long planktonic larval phases (Pechenik 1999). Further research into these factors is required to better understand the processes influencing abundance. This research could include collecting spat, using transects to determine

microhabitat availability and food supply for grazing invertebrates, timing searches to better understand current abundance, and setting up exclusionary cages to assess predation rates.

The MaxEnt model predicting urchin barren presence found that distance to reef was the most important variable to explain the distribution of urchin barrens. It was followed by reef slope, depth and finally inside/outside the marine national park. The highest probability of occurrence was on reef habitat, on more complex (sloping) terrain, in shallower depths and in areas outside the MPA. Overall, it was predicted that study area contains 6.3 km² of likely urchin barrens. This is a substantial area of habitat, representing about 15% of the total area of the park and a significantly higher proportion of subtidal rocky reef.

Management actions to reduce urchin numbers have been effective for restoring barrens back to kelp beds in other regions (Ling et al. 2010, Layton et al. 2020), and similar actions could benefit Cape Howe MNP. Information provided by the MaxEnt model could help inform target areas. However, many barren areas occur at depths below 20 m; these areas are inaccessible for most methods of urchin reduction, which require divers. Anecdotally, urchin barrens caused by Longspined Sea Urchin (*Centrostephanus rodgersii*) have been present at Cape Howe at least since the park was created in 2002, and they may form part of the natural character of the park. Monitoring of urchin barrens should continue to help identify whether barrens are at levels that require intervention.

4.4 Conclusion

Overall, most fish and macroinvertebrate species were found to be in good health in Cape Howe MNP, and changes in habitat types were relatively small, although urchin barrens are a prevalent feature on reefs. However, there have been clear declines in the abundance and condition of key fish and invertebrate species. The underlying driver for these declines remains unclear, and there are not obvious commonalities between many of the species in decline. Given Cape Howe's location at the interface of warm eastern and cool southern waters, it is an important marine national park to monitor for climate-driven changes. It is not clear whether the changes in some of the species detected in this study relate to increasing sea-surface temperatures or a range of other drivers. Ongoing long-term monitoring and further investigations of potential drivers are needed to better understand what these changes signify.

5. Acknowledgements

We sincerely thank the Parks Victoria (PV) staff and volunteers who supported fieldwork and Reef Life Survey monitoring in Cape Howe: Tess Hoinville (PV project coordinator, skipper and support diver), Monique Bregman (PV skipper and support diver); Shaun Davis (PV dive coordinator and dive attendant), Gary Summers (PV RLS diver), Peter Hay (PV RLS diver), Jonathon Stevenson (PV RLS diver), Nicole Filby (PV RLS diver) and Andrew Green (RLS volunteer). We thank Phil Reichelt and the PV Mallacoota team for their assistance with local knowledge and equipment storage. We also thank former PV Marine Science Manager Jacqui Pocklington for contract development and management, planning and logistics of the initial stages of the Cape Howe monitoring program detailed in this report.

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7. Appendix 1 – Relative abundance of all species across years

Appendix 1.1. Summary of species observed using BRUVS. Total observed refers to the total number of individuals recorded in *MaxN* across all sites. Total lengths refers to the total number of length measurements made across all sites. Total biomass observed refers to the total estimated biomass of each species (measured in grams) observed across the whole study. This measurement includes the biomass of all individuals measured, as well as an estimate of biomass of all individuals unable to be measured (average individual biomass multiplied by number of individuals not measured). Relative abundance was calculated as the mean and standard error of the mean for the *MaxN* of each species, across all sites (including sites where the species was not observed). Relative total biomass was measured in grams and was calculated as the mean and standard error of the mean for the total biomass of each species, across all sites (including sites where the species was not observed). Mean length denotes the mean length (and standard error of the mean) of each species across all sites and is given in centimetres.

Family	Taxon	Common name	Total observed	Relative abundance (mean ±SE)	Total lengths	Mean length (cm) (±SE)	Total biomass (g)	Relative total biomass (g) (mean ±SE)	Year
Teleosts									
Aplodactylidae	<i>Aplodactylus lophodon</i>	Rock Cale	2	0.02 ± 0.02	1	26.58 ± 0	275.71	3.24 ± 2.28	2021
Aracanidae	<i>Anoplocapros inermis</i>	Eastern Smooth Boxfish	1	0 ± 0	NA	NA	NA	NA	2006
Arripidae	<i>Arripis trutta</i>	Australian Salmon	1	0.01 ± 0.01	1	18.57 ± 0	131.44	1.55 ± 1.55	2021
			150	1.76 ± 1.31	24	38.49 ± 1.16	85,453.77	1,005.34 ± 757.47	2021
Aulopidae	<i>Latropiscis purpurissatus</i>	Sergeant Baker	9	0.04 ± 0.02	NA	NA	NA	NA	2006
			8	0.09 ± 0.03	3	40.79 ± 3.51	3,914.65	46.05 ± 16.09	2021
Bothidae	Bothidae spp.	Flounder	1	0.01 ± 0.01	NA	NA	NA	NA	2021
Callanthiidae	<i>Callanthias australis</i>	Splendid Perch	8	0.04 ± 0.02	6	21.46 ± 1.39	1,005.09	4.83 ± 2.48	2006
			4	0.05 ± 0.04	2	20.12 ± 0.05	393.02	4.62 ± 3.65	2021
Callionymidae	<i>Foetorepus calauropomus</i>	Common Stinkfish	2	0.01 ± 0.01	NA	NA	NA	NA	2006
Carangidae	<i>Pseudocaranx</i> spp.	Trevally	186	0.89 ± 0.29	81	23.62 ± 0.85	46,767.85	224.85 ± 79.6	2006
			239	2.81 ± 0.83	107	17.82 ± 0.81	29,308.73	344.81 ± 97.75	2021
			1	0 ± 0	NA	NA	NA	NA	2006
	<i>Seriola lalandi</i>	Yellowtail Kingfish	32	0.38 ± 0.21	14	58.9 ± 3.8	95,871.07	1,127.89 ± 602.62	2021

Family	Taxon	Common name	Total observed	Relative abundance (mean ±SE)	Total lengths	Mean length (cm) (±SE)	Total biomass (g)	Relative total biomass (g) (mean ±SE)	Year
Centriscidae	<i>Trachurus novaezelandiae</i>	Yellowtail Scad	8,239	39.61 ± 4.87	757	16.82 ± 0.29	1,031,274	4,958.05 ± 584.82	2006
			540	6.35 ± 2.59	78	14.93 ± 0.57	48,014.17	564.87 ± 223.25	2021
	<i>Macroramphosus scolopax</i>	Common Bellowsfish	21	0.25 ± 0.24	3	11.95 ± 0.44	209.55	2.47 ± 2.35	2021
	<i>Cheilodactylus spectabilis</i>	Banded Morwong	11	0.05 ± 0.02	5	33.09 ± 1.82	16,781.86	80.68 ± 28.01	2006
			16	0.19 ± 0.05	12	35.95 ± 1.67	31,987.34	376.32 ± 109.29	2021
	<i>Nemadactylus douglasii</i>	Grey Morwong	168	0.81 ± 0.11	104	29.59 ± 0.83	71,734.4	344.88 ± 45.76	2006
			301	3.54 ± 0.69	126	28.76 ± 0.7	111,776.4	1,315.02 ± 238.14	2021
	<i>Nemadactylus macropterus</i>	Jackass Morwong	342	1.64 ± 0.26	196	26.97 ± 0.3	103,409.5	497.16 ± 74.23	2006
			260	3.06 ± 0.74	99	27.82 ± 0.56	83,435.52	981.59 ± 209.8	2021
	<i>Chironemus marmoratus</i>	Kelpfish	11	0.13 ± 0.06	9	22.63 ± 0.87	1,534.36	18.05 ± 7.33	2021
Cyttidae	<i>Cyttus australis</i>	Silver Dory	3	0.01 ± 0.01	2	21.77 ± 2.15	629.52	3.03 ± 1.79	2006
			7	0.08 ± 0.03	2	21.92 ± 1.58	1,476.62	17.37 ± 7.28	2021
Dinolestidae	<i>Dinolestes lewini</i>	Longfin Pike	33	0.16 ± 0.12	23	31.68 ± 0.86	6,415.44	30.84 ± 22.33	2006
			75	0.88 ± 0.33	20	32.43 ± 0.94	15,313.67	180.16 ± 66.58	2021
Diodontidae	<i>Diodon nicthemerus</i>	Globe Fish	12	0.06 ± 0.02	NA	NA	NA	NA	2006
			3	0.04 ± 0.02	1	23.85 ± 0	887.24	10.44 ± 5.95	2021
Enoplosidae	<i>Enoplosus armatus</i>	Old Wife	2	0.01 ± 0.01	NA	NA	NA	NA	2006
			19	0.22 ± 0.07	8	19.28 ± 0.96	773.8	9.1 ± 2.78	2021
Gempylidae	<i>Thyrsites atun</i>	Barracouta	91	0.44 ± 0.1	14	54.66 ± 5.53	94,541.18	454.52 ± 102.5	2006
Gerreidae	<i>Parequula melbournensis</i>	Silverbelly	35	0.17 ± 0.11	20	11.32 ± 0.6	842.75	4.05 ± 2.12	2006
			3	0.04 ± 0.02	1	19.55 ± 0	343.95	4.05 ± 2.31	2021
Kyphosidae	<i>Girella zebra</i>	Zebra Fish	5	0.06 ± 0.03	1	32.95 ± 0	3,149.7	37.06 ± 22	2021
	<i>Atypichthys strigatus</i>	Mado Sweep	77	0.37 ± 0.34	31	12.2 ± 0.27	3,184.17	15.31 ± 13.37	2006
			578	6.8 ± 4.86	63	11.35 ± 0.46	19,485.05	229.24 ± 160.28	2021
	<i>Scorpi lineolata</i>	Silver Sweep	74	0.36 ± 0.18	68	18.87 ± 0.42	6,213.65	29.87 ± 15.99	2006
			384	4.52 ± 2.25	72	17.62 ± 0.44	24,365.71	286.66 ± 137.7	2021
Labridae	<i>Achoerodus viridis</i>	Eastern Blue Groper	15	0.07 ± 0.02	5	47.17 ± 12.34	38,065.54	183.01 ± 60.09	2006
			14	0.16 ± 0.05	9	52.89 ± 4.05	43,426.25	510.9 ± 160.64	2021
	<i>Bodianus unimaculatus</i>	Eastern Pigfish	2	0.02 ± 0.02	1	48.79 ± 0	3,391.93	39.91 ± 28.05	2021

Family	Taxon	Common name	Total observed	Relative abundance (mean ±SE)	Total lengths	Mean length (cm) (±SE)	Total biomass (g)	Relative total biomass (g) (mean ±SE)	Year
Latridae	<i>Eupetrichthys angustipes</i>	Snake-Skin Wrasse	2	0.01 ± 0.01	1	11.97 ± 0	37.92	0.18 ± 0.13	2006
	<i>Notolabrus fucicola</i>	Purple Wrasse	1	0 ± 0	NA	NA	NA	NA	2006
	<i>Notolabrus gymnogenis</i>	Crimsonband Wrasse	4	0.05 ± 0.02	3	30.85 ± 0.77	1,597.91	18.8 ± 9.28	2021
			6	0.03 ± 0.01	3	23.05 ± 3.91	1,537.85	7.39 ± 3.27	2006
	<i>Notolabrus tetricus</i>	Blue-Throat Wrasse	16	0.19 ± 0.04	11	24.89 ± 1.29	5,038.58	59.28 ± 15.04	2021
			59	0.28 ± 0.05	1	32.86 ± 0	6,112.54	29.39 ± 4.97	2006
	<i>Ophthalmolepis lineolata</i>	Maori Wrasse	104	1.22 ± 0.2	72	34.64 ± 0.76	12,654.8	148.88 ± 22.59	2021
			117	0.56 ± 0.08	83	28.76 ± 0.65	21,890.86	105.24 ± 15.99	2006
	<i>Pictilabrus laticlavus</i>	Senator Wrasse	135	1.59 ± 0.23	74	29.99 ± 0.52	27,254.3	320.64 ± 43.74	2021
			1	0 ± 0	1	24.51 ± 0	168.83	0.81 ± 0.81	2006
	<i>Pseudolabrus rubicundus</i>	Rosy Wrasse	118	0.57 ± 0.09	88	19.64 ± 0.5	12,206.37	58.68 ± 9.39	2006
			31	0.36 ± 0.08	22	22.2 ± 1.1	4,292.04	50.49 ± 10.02	2021
	<i>Latridopsis forsteri</i>	Bastard Trumpeter	39	0.19 ± 0.14	9	43.26 ± 2.31	38,799.03	186.53 ± 134.34	2006
			19	0.22 ± 0.08	9	46.93 ± 0.65	23,866.16	280.78 ± 101.87	2021
	<i>Morwong fuscus</i>	Red Morwong	13	0.15 ± 0.05	11	33.32 ± 1.61	6,751.67	79.43 ± 28.76	2021
Monacanthidae	<i>Pseudogoniistius nigripes</i>	Magpie Perch	3	0.01 ± 0.01	1	29.28 ± 0	968.73	4.66 ± 2.68	2006
			5	0.06 ± 0.03	5	31.99 ± 1.78	2,185.7	25.71 ± 14.9	2021
	<i>Acanthaluteres spilomelanurus</i>	Bridled Leatherjacket	1	0.01 ± 0.01	NA	NA	NA	NA	2021
	<i>Acanthaluteres vittiger</i>	Toothbrush Leatherjacket	9	0.04 ± 0.02	5	26.04 ± 1.49	2,643.14	12.71 ± 5.35	2006
			3	0.04 ± 0.02	1	23.34 ± 0	628.8	7.4 ± 4.22	2021
	<i>Eubalichthys bucephalus</i>	Black Reef Leatherjacket	6	0.03 ± 0.02	1	20.11 ± 0	826.3	3.97 ± 2.08	2006
			10	0.12 ± 0.04	4	32.43 ± 1.29	5,615.77	66.07 ± 23.52	2021
	<i>Eubalichthys gunnii</i>	Gunn's Leatherjacket	1	0 ± 0	1	20.71 ± 0	100.27	0.48 ± 0.48	2006
			8	0.09 ± 0.03	4	28.84 ± 2.76	2,294.98	27 ± 9.95	2021
	<i>Eubalichthys mosaicus</i>	Mosaic Leatherjacket	13	0.06 ± 0.02	11	22.03 ± 1.29	2,490.34	11.97 ± 3.87	2006
			13	0.15 ± 0.05	8	26.45 ± 2.33	4,265.03	50.18 ± 19.75	2021
	<i>Meuschenia australis</i>	Brown-Striped Leatherjacket	1	0.01 ± 0.01	1	23.82 ± 0	222.29	2.62 ± 2.62	2021
	<i>Meuschenia freycineti</i>	Six-Spine Leatherjacket	163	0.78 ± 0.09	102	29.55 ± 0.7	74,764.61	359.45 ± 42.89	2006

Family	Taxon	Common name	Total observed	Relative abundance (mean ±SE)	Total lengths	Mean length (cm) (±SE)	Total biomass (g)	Relative total biomass (g) (mean ±SE)	Year
Moridae	<i>Meuschenia scaber</i>	Velvet Leatherjacket	41	0.48 ± 0.07	29	34.36 ± 1.3	28,511.81	335.43 ± 52.91	2021
			1,309	6.29 ± 0.49	819	19.46 ± 0.1	385,506.7	1,853.4 ± 145.16	2006
			838	9.86 ± 1.46	265	19.45 ± 0.15	240,703.6	2,831.81 ± 411.7	2021
	<i>Meuschenia venusta</i>	Stars and Stripes Leatherjacket	2	0.02 ± 0.02	2	17.88 ± 0.18	193.09	2.27 ± 1.6	2021
	<i>Nelusetta ayraud</i>	Ocean Jacket	865	4.16 ± 0.39	613	26.03 ± 0.22	270,508.8	1,300.52 ± 127.44	2006
	<i>Thamnaconus degeni</i>	Degen’s Leatherjacket	502	5.91 ± 1.01	162	31.78 ± 0.33	266,331.6	3,133.31 ± 549.19	2021
			94	0.45 ± 0.11	58	16.32 ± 0.38	7,401.14	35.58 ± 8.74	2006
			19	0.22 ± 0.06	13	19.32 ± 0.9	2,413.74	28.4 ± 7.15	2021
	<i>Pseudophycis</i> spp.	Cod	5	0.02 ± 0.01	2	37.17 ± 6.27	2,390.73	11.49 ± 6.21	2006
	Mullidae	<i>Upeneichthys</i> spp.	Goatfishes	20	0.24 ± 0.06	12	27.61 ± 1.84	4,013.3	47.22 ± 13.8
60				0.29 ± 0.05	39	23.21 ± 1.38	13,603.82	65.4 ± 17.2	2006
Muraenidae	<i>Gymnothorax prasinus</i>	Green Moray	32	0.38 ± 0.07	22	23.29 ± 0.62	5,664.78	66.64 ± 12.52	2021
			60	0.29 ± 0.05	10	28.17 ± 5.46	2,110.5	10.15 ± 2.16	2006
Neosebastidae	<i>Gymnothorax prionodon</i>	Sawtooth Moray	159	1.87 ± 0.27	6	47.17 ± 2.37	23,243.36	273.45 ± 39.12	2021
			2	0.02 ± 0.02	NA	NA	NA	NA	2021
	<i>Maxillicosta</i> spp.	Gurnard scorpionfishes	1	0.01 ± 0.01	NA	NA	NA	NA	2021
	<i>Neosebastes scorpaenoides</i>	Common Gurnard Perch	81	0.39 ± 0.07	40	26.63 ± 1.43	36,482.37	175.4 ± 29.16	2006
Odacidae	<i>Olisthops cyanomelas</i>	Herring Cale	30	0.35 ± 0.07	21	30.81 ± 0.97	18,868.3	221.98 ± 47.98	2021
			1	0 ± 0	NA	NA	NA	NA	2006
			5	0.06 ± 0.04	2	30.72 ± 3.61	1,342.68	15.8 ± 11.16	2021
Pempheridae	<i>Siphonognathus</i> spp.	Weed whiting	2	0.02 ± 0.02	NA	NA	NA	NA	2021
	<i>Pempheris multiradiata</i>	Common Bullseye	5	0.06 ± 0.06	NA	NA	NA	NA	2021
Pentacerotidae	<i>Pentaceropsis recurvirostris</i>	Long-Snouted Boarfish	4	0.02 ± 0.02	NA	NA	NA	NA	2006
Pinguipedidae	<i>Parapercis allporti</i>	Barred Grubfish	23	0.27 ± 0.1	6	33.3 ± 2.55	17,969.97	211.41 ± 88.22	2021
			219	1.05 ± 0.17	135	13.95 ± 0.37	6,204.21	29.83 ± 5.2	2006

Family	Taxon	Common name	Total observed	Relative abundance (mean ±SE)	Total lengths	Mean length (cm) (±SE)	Total biomass (g)	Relative total biomass (g) (mean ±SE)	Year
Platycephalidae			51	0.6 ± 0.16	26	16.39 ± 0.96	2,187.64	25.74 ± 6.11	2021
	<i>Parapercis binivirgata</i>	Redbanded Grubfish	2	0.01 ± 0.01	NA	NA	NA	NA	2006
	<i>Platycephalus caeruleopunctatus</i>	Bluespotted Flathead	122	0.59 ± 0.08	61	29.27 ± 1.08	22,488.14	108.12 ± 15.69	2006
			108	1.27 ± 0.36	66	29.59 ± 1.02	20,883.3	245.69 ± 63.6	2021
	<i>Platycephalus grandispinis</i>	Longspine Flathead	12	0.14 ± 0.07	10	21.78 ± 1.03	798.88	9.4 ± 5.63	2021
	<i>Platycephalus richardsoni</i>	Tiger Flathead	44	0.21 ± 0.03	24	34.3 ± 2.28	13,585.43	65.31 ± 11.77	2006
Pomacentridae			2	0.02 ± 0.02	1	38.22 ± 0	736.66	8.67 ± 6.09	2021
	<i>Chromis hypsilepis</i>	One-Spot Puller	110	0.53 ± 0.25	43	12.13 ± 0.4	3,050.77	14.67 ± 6.82	2006
			10	0.12 ± 0.1	4	15.11 ± 1.29	520.85	6.13 ± 5.35	2021
	<i>Parma microlepis</i>	White-Ear	43	0.21 ± 0.03	27	14.44 ± 0.58	3,062.75	14.72 ± 2.66	2006
Pristiophoridae			56	0.66 ± 0.1	39	15.36 ± 0.33	4,579.11	53.87 ± 8.06	2021
	<i>Pristiophorus cirratus</i>	Common Sawshark	1	0 ± 0	NA	NA	NA	NA	2006
Scorpaenidae	<i>Scorpaena papillosa</i>	Southern Red Scorpionfish	3	0.01 ± 0.01	NA	NA	NA	NA	2006
Sebastidae			40	0.47 ± 0.2	12	15.06 ± 0.48	1,841.67	21.67 ± 9.04	2021
	<i>Helicolenus percoides</i>	Red Gurnard Perch	147	0.71 ± 0.17	74	20.31 ± 0.71	35,671.75	171.5 ± 34.33	2006
			81	0.95 ± 0.26	37	20.41 ± 0.74	17,905.07	210.65 ± 57.45	2021
Serranidae	<i>Acanthistius ocellatus</i>	Eastern Wirrah	5	0.02 ± 0.01	5	31.23 ± 3.42	1,456.4	7 ± 4.31	2006
			22	0.26 ± 0.08	15	30.8 ± 1.24	5,665.8	66.66 ± 19.7	2021
	<i>Caesioperca</i> spp.	Perch	970	4.66 ± 0.96	369	15.99 ± 0.21	37,463.35	180.11 ± 35.7	2006
			321	3.78 ± 1.22	71	16.4 ± 0.43	12,816.18	150.78 ± 46.14	2021
	<i>Hypoplectrodes annulatus</i>	Blackbanded Seaperch	1	0 ± 0	1	17.58 ± 0	46.52	0.22 ± 0.22	2006
			10	0.12 ± 0.05	7	17.14 ± 1.62	474	5.58 ± 2.38	2021
	<i>Hypoplectrodes maccullochi</i>	Halfbanded Seaperch	25	0.12 ± 0.03	10	12.51 ± 0.87	446.25	2.15 ± 0.53	2006
			38	0.45 ± 0.09	25	12.03 ± 0.5	617.74	7.27 ± 1.52	2021
Sillaginidae	<i>Sillago</i> spp.	Whiting	290	1.39 ± 0.5	10	14.26 ± 1.39	4,885.41	23.49 ± 8.42	2006
			468	5.51 ± 3.62	42	10.35 ± 0.36	2,905.65	34.18 ± 21.77	2021
Sparidae	<i>Pagrus auratus</i>	Snapper	98	0.47 ± 0.08	59	24.69 ± 0.72	33,229.07	159.76 ± 26.43	2006
			124	1.46 ± 0.29	78	27.29 ± 0.8	57,845.27	680.53 ± 150.29	2021
Tetraodontidae	<i>Tetractenos glaber</i>	Smooth Toadfish	1	0.01 ± 0.01	NA	NA	NA	NA	2021

Family	Taxon	Common name	Total observed	Relative abundance (mean ±SE)	Total lengths	Mean length (cm) (±SE)	Total biomass (g)	Relative total biomass (g) (mean ±SE)	Year
Triglidae	<i>Chelidonichthys kumu</i>	Red Gurnard	30	0.14 ± 0.03	14	28.81 ± 2.93	8,383.43	40.3 ± 9.52	2006
			9	0.11 ± 0.04	8	28.22 ± 4.37	2,936.57	34.55 ± 19.92	2021
Zeidae	<i>Zeus faber</i>	John Dory	1	0.01 ± 0.01	NA	NA	NA	NA	2021
			1	0 ± 0	1	48.19 ± 0	1,310.39	6.3 ± 6.3	2006
Chondrichthyans									
Callorhinchidae	<i>Callorhinchus milii</i>	Elephantfish	3	0.04 ± 0.02	3	73.49 ± 4.45	8,515.72	100.18 ± 59.07	2021
Dasyatidae	<i>Bathytoshia brevicaudata</i>	Smooth Stingray	16	0.08 ± 0.02	4	117.06 ± 6.58	327,594.3	1,574.97 ± 402.29	2006
			54	0.64 ± 0.07	33	110.14 ± 5.68	1,061,792	12,491.67 ± 1,924.01	2021
Heterodontidae	<i>Heterodontus portusjacksoni</i>	Port Jackson Shark	19	0.09 ± 0.02	15	66.9 ± 8.08	54,757.86	263.26 ± 85.99	2006
			29	0.34 ± 0.06	22	77.64 ± 4.69	105,406.8	1,240.08 ± 274.04	2021
Myliobatidae	<i>Myliobatis tenuicaudatus</i>	Southern Eagle Ray	14	0.07 ± 0.02	1	46.04 ± 0	60,270.06	289.76 ± 74.97	2006
			19	0.22 ± 0.05	15	118.16 ± 8.47	1,624,422	19,110.84 ± 5,162.54	2021
Odontaspidae	<i>Carcharias taurus</i>	Greynurse Shark	1	0.01 ± 0.01	1	200.86 ± 0	104,122.8	1,224.97 ± 1,224.97	2021
Pentanchidae	<i>Asymbolus</i> spp.	Catsharks	54	0.26 ± 0.04	37	41.73 ± 1.26	20,849.96	100.24 ± 15.91	2006
			13	0.15 ± 0.05	6	50.02 ± 4.07	8,603	101.21 ± 30.24	2021
Rajidae	<i>Spiniraja whitleyi</i>	Melbourne Skate	21	0.1 ± 0.02	5	98.51 ± 17.47	172,875.1	831.13 ± 221	2006
			2	0.02 ± 0.02	1	113.97 ± 0	23,591.59	277.55 ± 195.08	2021
Rhinobatidae	<i>Trygonorrhina dumerilii</i>	Southern Fiddler Ray	6	0.07 ± 0.03	6	65.93 ± 4.96	12,929.17	152.11 ± 66.7	2021
	<i>Trygonorrhina fasciata</i>	Eastern Fiddler Ray	20	0.1 ± 0.02	7	85.07 ± 5.29	87,326.66	419.84 ± 101.56	2006
			55	0.65 ± 0.14	37	76.22 ± 1.6	174,002.2	2,047.08 ± 449.25	2021
Scyliorhinidae	<i>Cephaloscyllium laticeps</i>	Draughtboard Shark	283	1.36 ± 0.12	134	64.96 ± 1.13	421,775.5	2027.77 ± 180.73	2006
			4	0.05 ± 0.02	4	73.15 ± 8.41	9,107.84	107.15 ± 57.82	2021
Squalidae	<i>Squalus megalops</i>	Spikey Dogfish	17	0.08 ± 0.03	8	32.95 ± 3.54	3,383.75	16.27 ± 6.17	2006
Triakidae	<i>Mustelus antarcticus</i>	Gummy Shark	102	0.49 ± 0.05	42	80.64 ± 3.83	392,610.3	1,887.55 ± 227.1	2006

Family	Taxon	Common name	Total observed	Relative abundance (mean ±SE)	Total lengths	Mean length (cm) (±SE)	Total biomass (g)	Relative total biomass (g) (mean ±SE)	Year
Urolophidae			69	0.81 ± 0.12	46	95.41 ± 3.25	427,987.6	5,035.15 ± 843.56	2021
	Urolophidae spp.	Stingarees	12	0.06 ± 0.02	2	37.73 ± 5.45	7,535.91	36.23 ± 10.35	2006
			21	0.25 ± 0.05	12	37.43 ± 4.39	16,693.67	196.4 ± 62.74	2021
	<i>Urolophus cruciatus</i>	Banded Stingaree	5	0.06 ± 0.03	4	29.4 ± 2.44	1,531.49	18.02 ± 8.55	2021
Invertebrates									
Asteriidae	Asteriidae spp.	Seastars	1	0.01 ± 0.01	NA	NA	NA	NA	2021
	<i>Coscinasterias muricata</i>	Eleven-Arm Star	5	0.06 ± 0.03	NA	NA	NA	NA	2021
Cidaridae	<i>Phyllacanthus parvispinus</i>	Eastern Slate-Pencil Urchin	1	0.01 ± 0.01	NA	NA	NA	NA	2021
Diadematidae	<i>Centrostephanus rodgersii</i>	Longspined Sea Urchin	26	0.12 ± 0.07	NA	NA	NA	NA	2006
Diogenidae			107	1.26 ± 0.51	NA	NA	NA	NA	2021
	Diogenidae spp.	Hermit crabs	1,195	5.75 ± 0.53	NA	NA	NA	NA	2006
			466	5.48 ± 1.14	NA	NA	NA	NA	2021
Loliginidae	<i>Sepioteuthis australis</i>	Southern Calamary	9	0.04 ± 0.02	NA	NA	NA	NA	2006
Luidiidae	<i>Luidia australiae</i>	Southern Sand Star	1	0.01 ± 0.01	NA	NA	NA	NA	2021
Majidae	<i>Leptomithrax gaimardii</i>	Spider Crab	1	0 ± 0	NA	NA	NA	NA	2006
Octopodidae	<i>Macroctopus maorum</i>	Maori Octopus	3	0.01 ± 0.01	NA	NA	NA	NA	2006
			2	0.02 ± 0.02	NA	NA	NA	NA	2021
Ommastrephidae	<i>Nototodarus gouldi</i>	Gould's Squid	1	0 ± 0	NA	NA	NA	NA	2006
Ovalipidae	<i>Ovalipes australiensis</i>	Common Sand Crab	2	0.01 ± 0.01	NA	NA	NA	NA	2006
			5	0.06 ± 0.03	NA	NA	NA	NA	2021
Palinuridae	<i>Jasus edwardsii</i>	Southern Rock Lobster	2	0.01 ± 0.01	NA	NA	NA	NA	2006
			6	0.07 ± 0.04	NA	NA	NA	NA	2021
Sepiidae	<i>Sepia apama</i>	Giant Cuttle	1	0 ± 0	NA	NA	NA	NA	2006
Mammals									
Otariidae	<i>Arctocephalus pusillus</i>	Australian Fur Seal	6	0.03 ± 0.01	NA	NA	NA	NA	2006
			2	0.02 ± 0.02	NA	NA	NA	NA	2021

Appendix 1.2. Relative abundances (mean $\pm$ SE) of all fish species observed in Reef Life Surveys, for each year sampled

Taxon	2008	2009	2011	2012	2014	2015	2021
<i>Acanthaluteres vittiger</i>	0	0 $\pm$ 0.00	6 $\pm$ 0.68	0	0	0 $\pm$ 0.00	1 $\pm$ 0.29
<i>Acanthistius ocellatus</i>	2	0 $\pm$ 0.00	6 $\pm$ 0.44	0	4	0 $\pm$ 0.00	2 $\pm$ 0.39
<i>Achoerodus viridis</i>	0	0 $\pm$ 0.00	62 $\pm$ 2.60	1	0	6 $\pm$ 1.29	13 $\pm$ 1.56
<i>Aplodactylus arctidens</i>	2	0 $\pm$ 0.00	9 $\pm$ 1.17	0	0	0 $\pm$ 0.00	23 $\pm$ 4.60
<i>Aplodactylus lophodon</i>	0	0 $\pm$ 0.00	109 $\pm$ 5.25	1	0	0 $\pm$ 0.00	26 $\pm$ 2.44
<i>Arctocephalus pusillus</i>	0	0 $\pm$ 0.00	2 $\pm$ 0.23	0	0	0 $\pm$ 0.00	0 $\pm$ 0.00
<i>Arripis trutta</i>	0	0 $\pm$ 0.00	0 $\pm$ 0.00	0	0	0 $\pm$ 0.00	1 $\pm$ 0.29
<i>Atypichthys strigatus</i>	150	24 $\pm$ 1.41	3,117 $\pm$ 86.61	55	371	394 $\pm$ 15.80	1,100 $\pm$ 59.21
<i>Bathytoshia brevicaudata</i>	0	0 $\pm$ 0.00	0 $\pm$ 0.00	0	0	1 $\pm$ 0.50	0 $\pm$ 0.00
<i>Bovichtus angustifrons</i>	1	0 $\pm$ 0.00	0 $\pm$ 0.00	0	0	0 $\pm$ 0.00	0 $\pm$ 0.00
<i>Cephaloscyllium laticeps</i>	1	2 $\pm$ 1.41	3 $\pm$ 0.27	0	0	2 $\pm$ 1.00	0 $\pm$ 0.00
<i>Chaetodon guentheri</i>	1	0 $\pm$ 0.00	0 $\pm$ 0.00	0	0	0 $\pm$ 0.00	0 $\pm$ 0.00
<i>Chirodactylus spectabilis</i>	1	12 $\pm$ 2.83	262 $\pm$ 5.22	2	2	0 $\pm$ 0.00	70 $\pm$ 6.95
<i>Chironemus georgianus</i>	0	0 $\pm$ 0.00	0 $\pm$ 0.00	0	1	0 $\pm$ 0.00	0 $\pm$ 0.00
<i>Chironemus marmoratus</i>	0	14 $\pm$ 1.41	207 $\pm$ 5.23	0	0	0 $\pm$ 0.00	65 $\pm$ 5.84
<i>Chromis hypsilepis</i>	20	0 $\pm$ 0.00	184 $\pm$ 11.85	21	121	339 $\pm$ 86.38	162 $\pm$ 32.85
<i>Chrysiptera rollandi</i>	0	0 $\pm$ 0.00	149 $\pm$ 17.07	0	0	0 $\pm$ 0.00	0 $\pm$ 0.00
<i>Dinolestes lewini</i>	2	0 $\pm$ 0.00	626 $\pm$ 43.89	0	0	20 $\pm$ 7.57	344 $\pm$ 68.04
<i>Diodon nicthemerus</i>	0	4 $\pm$ 1.41	5 $\pm$ 0.41	0	0	1 $\pm$ 0.50	0 $\pm$ 0.00
<i>Dotalabrus aurantiacus</i>	0	0 $\pm$ 0.00	11 $\pm$ 0.87	0	0	0 $\pm$ 0.00	3 $\pm$ 0.62
<i>Enoplosus armatus</i>	0	0 $\pm$ 0.00	5 $\pm$ 0.34	2	0	0 $\pm$ 0.00	10 $\pm$ 2.89
<i>Eocallionymus papilio</i>	0	0 $\pm$ 0.00	0 $\pm$ 0.00	0	0	1 $\pm$ 0.50	0 $\pm$ 0.00
<i>Eubalichthys bucephalus</i>	0	0 $\pm$ 0.00	2 $\pm$ 0.23	0	0	0 $\pm$ 0.00	0 $\pm$ 0.00
<i>Eubalichthys mosaicus</i>	0	0 $\pm$ 0.00	1 $\pm$ 0.16	0	0	0 $\pm$ 0.00	0 $\pm$ 0.00
<i>Eupetrichthys angustipes</i>	4	0 $\pm$ 0.00	6 $\pm$ 0.49	3	3	7 $\pm$ 1.50	6 $\pm$ 1.00
<i>Girella elevata</i>	0	0 $\pm$ 0.00	0 $\pm$ 0.00	0	0	1 $\pm$ 0.50	0 $\pm$ 0.00
<i>Girella tricuspidata</i>	0	0 $\pm$ 0.00	18 $\pm$ 1.89	0	0	0 $\pm$ 0.00	0 $\pm$ 0.00
<i>Girella zebra</i>	0	0 $\pm$ 0.00	14 $\pm$ 0.71	0	1	34 $\pm$ 15.70	11 $\pm$ 1.44
<i>Gymnothorax prasinus</i>	0	2 $\pm$ 1.41	6 $\pm$ 0.59	0	5	0 $\pm$ 0.00	1 $\pm$ 0.29
<i>Heteroclinus heptaeolus</i>	0	0 $\pm$ 0.00	0 $\pm$ 0.00	0	0	0 $\pm$ 0.00	1 $\pm$ 0.29
<i>Heteroclinus roseus</i>	0	0 $\pm$ 0.00	4 $\pm$ 0.51	0	0	0 $\pm$ 0.00	0 $\pm$ 0.00
<i>Heteroclinus whiteleggii</i>	0	0 $\pm$ 0.00	0 $\pm$ 0.00	0	0	1 $\pm$ 0.50	0 $\pm$ 0.00
<i>Heterodontus portusjacksoni</i>	1	0 $\pm$ 0.00	18 $\pm$ 1.67	0	0	1 $\pm$ 0.50	3 $\pm$ 0.62
<i>Heteroscarus acroptilus</i>	0	0 $\pm$ 0.00	7 $\pm$ 0.39	0	1	1 $\pm$ 0.50	0 $\pm$ 0.00
<i>Hypoplectrodes annulatus</i>	0	0 $\pm$ 0.00	0 $\pm$ 0.00	0	0	0 $\pm$ 0.00	1 $\pm$ 0.29
<i>Hypoplectrodes maccullochi</i>	9	1 $\pm$ 0.71	35 $\pm$ 2.73	3	2	7 $\pm$ 2.36	4 $\pm$ 0.89
<i>Kyphosus sydneyanus</i>	0	0 $\pm$ 0.00	7 $\pm$ 0.73	0	0	0 $\pm$ 0.00	1 $\pm$ 0.29
<i>Latridopsis forsteri</i>	0	1 $\pm$ 0.71	15 $\pm$ 1.15	0	0	0 $\pm$ 0.00	0 $\pm$ 0.00
<i>Lotella rhacina</i>	0	0 $\pm$ 0.00	8 $\pm$ 0.53	1	0	0 $\pm$ 0.00	5 $\pm$ 0.90

Taxon	2008	2009	2011	2012	2014	2015	2021
<i>Meuschenia flavolineata</i>	0	0 ± 0.00	11 ± 0.90	0	0	0 ± 0.00	1 ± 0.29
<i>Meuschenia freycineti</i>	3	24 ± 15.56	79 ± 2.80	0	0	3 ± 0.50	2 ± 0.39
<i>Meuschenia scaber</i>	0	0 ± 0.00	1 ± 0.16	0	0	0 ± 0.00	0 ± 0.00
<i>Morwong fuscus</i>	4	0 ± 0.00	54 ± 2.59	5	7	9 ± 3.30	13 ± 2.84
<i>Myliobatis tenuicaudatus</i>	0	0 ± 0.00	1 ± 0.16	0	0	0 ± 0.00	0 ± 0.00
<i>Nemadactylus douglasii</i>	0	0 ± 0.00	7 ± 0.73	0	0	1 ± 0.50	0 ± 0.00
<i>Nemadactylus valenciennesi</i>	0	0 ± 0.00	2 ± 0.32	0	0	0 ± 0.00	0 ± 0.00
<i>Notolabrus fucicola</i>	1	1 ± 0.71	493 ± 10.59	3	10	18 ± 2.08	16 ± 1.72
<i>Notolabrus gymnogenis</i>	4	5 ± 0.71	126 ± 5.59	6	2	34 ± 7.33	28 ± 5.63
<i>Notolabrus tetricus</i>	5	11 ± 4.95	432 ± 8.58	6	11	69 ± 4.57	144 ± 3.98
<i>Olisthops cyanomelas</i>	5	4 ± 1.41	449 ± 6.75	1	2	10 ± 2.89	200 ± 12.52
<i>Ophthalmolepis lineolata</i>	17	32 ± 22.63	424 ± 12.69	28	29	85 ± 10.47	155 ± 18.99
<i>Orectolobus maculatus</i>	0	0 ± 0.00	1 ± 0.16	0	5	0 ± 0.00	0 ± 0.00
<i>Pagrus auratus</i>	0	0 ± 0.00	10 ± 1.62	0	0	0 ± 0.00	0 ± 0.00
<i>Parma microlepis</i>	8	42 ± 22.63	595 ± 10.62	4	42	40 ± 6.68	198 ± 18.59
<i>Parma unifasciata</i>	0	0 ± 0.00	1 ± 0.16	0	0	0 ± 0.00	0 ± 0.00
<i>Pempheris compressa</i>	0	0 ± 0.00	0 ± 0.00	5	0	72 ± 22.05	35 ± 6.89
<i>Pempheris multiradiata</i>	5	0 ± 0.00	224 ± 20.11	0	136	1 ± 0.50	21 ± 4.09
<i>Pentaceropsis recurvirostris</i>	0	0 ± 0.00	0 ± 0.00	0	0	0 ± 0.00	7 ± 2.02
<i>Pictilabrus laticlavus</i>	1	4 ± 0.00	117 ± 2.77	4	1	22 ± 6.45	59 ± 6.73
<i>Plagiotremus tapeinosoma</i>	0	0 ± 0.00	0 ± 0.00	0	0	0 ± 0.00	1 ± 0.29
<i>Pseudocaranx georgianus</i>	0	11 ± 7.78	446 ± 38.45	0	0	95 ± 20.56	0 ± 0.00
<i>Pseudolabrus luculentus</i>	0	0 ± 0.00	3 ± 0.27	0	0	1 ± 0.50	0 ± 0.00
<i>Pseudolabrus rubicundus</i>	0	0 ± 0.00	0 ± 0.00	1	2	0 ± 0.00	3 ± 0.87
<i>Scorpaena cardinalis</i>	0	1 ± 0.71	0 ± 0.00	0	0	0 ± 0.00	0 ± 0.00
<i>Scorpaena papillosa</i>	0	0 ± 0.00	2 ± 0.23	0	0	0 ± 0.00	0 ± 0.00
<i>Scorpius aequipinnis</i>	0	0 ± 0.00	0 ± 0.00	0	0	0 ± 0.00	9 ± 1.60
<i>Scorpius lineolata</i>	0	0 ± 0.00	1,752 ± 67.16	23	27	89 ± 29.80	1,369 ± 123.36
<i>Tetractenos glaber</i>	0	0 ± 0.00	18 ± 0.73	0	0	0 ± 0.00	3 ± 0.62
<i>Torquigener pleurogramma</i>	0	0 ± 0.00	1 ± 0.16	0	0	0 ± 0.00	0 ± 0.00
<i>Trachinops taeniatus</i>	87	0 ± 0.00	191 ± 16.21	56	80	147 ± 42.44	154 ± 31.54
<i>Trachurus novaezelandiae</i>	0	30 ± 0.00	28,533 ± 856.55	0	0	30 ± 15.00	2020 ± 179.49
<i>Trinorfolkia clarkei</i>	0	0 ± 0.00	11 ± 0.80	0	1	4 ± 1.41	3 ± 0.62
<i>Trygonoptera mucosa</i>	0	0 ± 0.00	0 ± 0.00	0	0	2 ± 1.00	0 ± 0.00
<i>Upeneichthys lineatus</i>	0	0 ± 0.00	0 ± 0.00	0	0	0 ± 0.00	3 ± 0.62
<i>Upeneichthys vlamingii</i>	6	0 ± 0.00	8 ± 0.47	12	0	7 ± 2.22	2 ± 0.39
<i>Urolophus kapalensis</i>	0	0 ± 0.00	0 ± 0.00	0	0	2 ± 0.58	1 ± 0.29

Taxon	2008	2009	2011	2012	2014	2015	2021
<i>Urolophus paucimaculatus</i>	0	0 ± 0.00	3 ± 0.27	0	0	0 ± 0.00	0 ± 0.00
<i>Urolophus</i> spp.	0	0 ± 0.00	1 ± 0.16	0	0	0 ± 0.00	0 ± 0.00

Appendix 1.3. Relative abundances (mean $\pm$ SE) of all macroinvertebrate and cryptic fish species observed in Reef Life Surveys, for each year sampled (2011, 2012, 2021)

Taxon	2011	2012	2021
<i>Acanthistius cinctus</i>	1 $\pm$ 0.27	0	0 $\pm$ 0.00
<i>Acanthistius ocellatus</i>	2 $\pm$ 0.36	0	0 $\pm$ 0.00
<i>Enneapterygius rufopileus</i>	8 $\pm$ 2.14	0	0 $\pm$ 0.00
<i>Gymnothorax prasinus</i>	6 $\pm$ 0.85	0	4 $\pm$ 0.73
<i>Heteroclinus nasutus</i>	1 $\pm$ 0.27	0	0 $\pm$ 0.00
<i>Heteroclinus tristis</i>	1 $\pm$ 0.27	0	0 $\pm$ 0.00
<i>Heteroclinus whiteleggii</i>	0 $\pm$ 0.00	0	3 $\pm$ 1.00
<i>Heterodontus portusjacksoni</i>	1 $\pm$ 0.27	0	0 $\pm$ 0.00
<i>Hypoplectrodes maccullochi</i>	9 $\pm$ 1.65	0	5 $\pm$ 1.33
<i>Lotella rhacina</i>	6 $\pm$ 0.51	0	6 $\pm$ 0.87
<i>Orectolobus halei</i>	0 $\pm$ 0.00	0	1 $\pm$ 0.33
<i>Pempheris compressa</i>	0 $\pm$ 0.00	4	0 $\pm$ 0.00
<i>Pempheris multiradiata</i>	0 $\pm$ 0.00	0	25 $\pm$ 6.59
<i>Pempheris</i> spp.	0 $\pm$ 0.00	0	1 $\pm$ 0.33
<i>Scorpaena cardinalis</i>	0 $\pm$ 0.00	0	1 $\pm$ 0.33
<i>Scorpaena papillosa</i>	7 $\pm$ 1.16	0	4 $\pm$ 1.01
<i>Trinorfolkia clarkei</i>	36 $\pm$ 5.21	0	8 $\pm$ 2.03

