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An integrated monitoring program for Wilsons Promontory Marine National Park

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Front cover: Draughtboard Shark (*Cephaloscyllium laticeps*) at Wilsons Promontory Marine National Park

Photo: Chris Hayward, Parks Victoria

Executive summary

Parks Victoria has established the Signs of Healthy Parks (SHP) program for its network of marine national parks and sanctuaries with the dual aims of improving baseline knowledge of Victoria's marine protected areas (MPAs) and addressing applied management questions. The SHP program aims to monitor the health of protected areas using a range of environmental indicators that in turn provide information about the natural values and ecological processes within the parks, as well as potential threats and other drivers. This study implemented a suite of monitoring approaches to assess the health and condition of the no-take Wilsons Promontory Marine National Park (MNP) in eastern Victoria.

Wilsons Promontory MNP surrounds the southern tip of Wilsons Promontory, a peninsula forming the southernmost part of the Australian mainland. The marine national park protects important habitat, including shallow subtidal reefs, deep subtidal reefs, intertidal rocky shores, sandy beaches, seagrass, subtidal soft substrates and expansive areas of open water. Shallower reefs have extensive communities of abundant red and brown macroalgae, dominated by Crayweed (*Phyllospora comosa*) and Golden Kelp (*Ecklonia radiata*), which support diverse assemblages of macroinvertebrates and fish. The park's deep reefs have a dense cover of epifauna, especially sponges, stalked ascidians and sea whips, and abundant fish life, while soft sediments support diverse biota.

Major threats to the park's natural values include invasive pests, illegal fishing, oil spills and increasing sea-surface temperature (SST). Invasive pests and oil spills are currently absent from the marine national park, though the close proximity of pests and shipping lanes mean they are both a significant potential threat. The threat of illegal fishing is considered low due to the park's remoteness from the nearest port. South-eastern Australia is a hotspot for increasing sea-surface temperatures due to the predicted strengthening of the East Australian Current in response to climate change; this presents the most significant potential impact on the natural values of the park.

The overarching objective of this project was to design and deliver a targeted monitoring program for Wilsons Promontory MNP with the dual aims of understanding the health and condition of major conservation assets and identifying threats to those assets. Specific objectives related to this overarching goal were to:

- compile seabed bathymetry data to develop full-coverage maps of Wilsons Promontory MNP and adjacent waters to help characterise the seabed
- assess patterns in sea-surface temperature inside the park to better understand the influence of climate change
- assess trends in the abundance and diversity of fish and invertebrate populations and canopy-forming algae over time using underwater visual census methods in shallow subtidal reefs to understand long-term changes in the condition of key species and habitat
- assess the abundance and diversity of fish populations using baited remote underwater video station (BRUVS) surveys within and adjacent to the MPA to understand the effect of seafloor structure and the effectiveness of no-take protection on the abundance and diversity of fish

- assess habitat distribution using BRUVS deployed in 2016 and 2021 to begin to develop an understanding of changes in habitat over time
- develop distribution models for key species, species richness and species biomass using a combination of data from BRUVS and habitat mapping.

Analyses of annual and seasonal patterns in SST found that only summer SST experienced an overall increase; annual temperatures tended to fluctuate and there was no clear trend in winter temperatures. Wilsons Promontory MNP seems to be buffered from temperature increases, likely due to strong currents coming from the east and wrapping around the point as well as the movement and mixing resulting from increased wave energy due to the high winds the coast is subject to.

Common species of fish observed by diver census (Reef Life Survey) across years included Purple Wrasse (*Notolabrus fucicola*), Blue-Throat Wrasse (*Notolabrus tetricus*) and Old Wife (*Enoplosus armatus*). Common species of invertebrates observed across years included Blacklip Abalone (*Haliotis rubra*), Purple Sea Urchin (*Heliocidaris erythrogramma*), Orange Feather Star (*Cenolia trichoptera*), Gunns Six-Armed Star (*Meridiastra gunnii*) and Large-Plated Seastar (*Nectria macrobrachia*). Common species of macroalgae observed across all years include Golden Kelp, Crayweed, the brown macroalga *Seirococcus axillaris* and 2 species of red algae (*Phacelocarpus peperocarpos* and *Plocamium angustum*).

The 6 key mobile fish species monitored by diver census were found to have a healthy overall population abundance in 2020 inside and outside the MPA, falling within the zone of 'good' condition with 3 exceptions. In 2020, Purple Wrasse (*N. fucicola*) fell below the lower limit of acceptable change outside the MPA, but it seems to have been on an upward trajectory since 2015. Barber Perch (*Caesioperca rasor*) abundance within the MPA was also below the lower limit of acceptable change, and it is on a downward trend. Butterfly Perch (*Caesioperca lepidoptera*) was not observed within or outside the MPA in 2020.

Four species, Purple Wrasse (*N. fucicola*), Bluethroat Wrasse (*N. tetricus*), Old Wife *E. armatus* and Sea Sweep (*Scorpiis aequipinnis*) were more abundant inside the park than outside, while Barber Perch (*C. rasor*) was more abundant outside the park. The sizes of individual fish species were generally similar within and outside the park.

Abundances of Blacklip Abalone (*H. rubra*) within the park and Orange Feather Star (*C. trichopteran*) outside the park fell within the zone of good condition (i.e., within the range of acceptable change) in 2020. However, declining trends in abundance over the past 20 years were observed in Gunns Six-Armed Star (*Meridiastra gunnii*) and Large-Plated Seastar (*Nectria macrobrachia*). In 2020, Gunns Six-Armed Star (*M. gunnii*) was below the lower limit of acceptable change inside the park and below the lower control limit outside the park. Purple Urchin (*H. erythrogramma*) has been below its lower limit of acceptable change of 125 individuals per 200 m² since 2002 within and outside the MPA and was below the lower control limit inside the park in 2020. Three of the 5 macroinvertebrates - Blacklip Abalone (*H. rubra*), Gunns Six-Armed Star (*Meridiastra gunnii*) and Large-Plated Seastar (*Nectria macrobrachia*)- were more abundant inside the park than outside, while Orange Feather Star (*C. trichopteran*) and Purple Urchin (*H. erythrogramma*) were more abundant outside the park.

The most notable change in the subtidal reef macroalgal communities of the Wilsons Promontory MNP has been the decline in percentage cover of the macroalgae *S. axillaris*, *P. peperocarpos* and

P. angustum within and outside the MPA since 2010 and 2011. Percentage cover of all 3 species has been below the lower control limit since 2014 (i.e. in the surveys in 2014, 2015, 2017 and 2020) within the park. In contrast, percentage cover of the major canopy-forming brown alga Crayweed (*Phyllospora comosa*) was well within the range of acceptable change within the MPA; however, its percentage cover outside the MPA has declined into the lower limit of acceptable change. Percentage cover of Golden Kelp (*Ecklonia radiata*) was below the lower limit of acceptable change within and outside the MPA, as has been the case since 2015. Of the species studied, only Golden Kelp (*E. radiata*) and the macroalgae *S. axillaris* have higher percentage cover outside the park than inside, while Crayweed (*P. comosa*) has higher cover inside the park.

The most abundant fish species or groups found across the 2 sampling years (2016 and 2021) by BRUVS surveys were perch (*Caesioperca* spp.) and Yellowtail Scad (*Trachurus novaezelandiae*), while Degen's Leatherjacket (*Thamnaconus degeni*) was also one of the most abundant in 2016 and Rosy Wrasse (*Pseudolabrus rubicundus*) in 2021. Draughtboard Shark (*Cephaloscyllium laticeps*) and perch (*Caesioperca* spp.) were the most widespread species in both years. Blue-Throat Wrasse (*Notolabrus tetricus*) was the next most widespread in 2016 while Silverbelly (*Parequula melbournensis*) was in 2021. In 2016, Gummy Shark (*Mustelus antarcticus*) Draughtboard Shark (*C. laticeps*), and Port Jackson Shark (*Heterodontus portusjacksoni*) accounted for the highest biomass by area. In 2021, Smooth Stingray (*Bathytoshia brevicaudata*) had the highest biomass, followed by Gummy Shark (*M. antarcticus*) and Broadnose Sevengill Shark (*Notorynchus cepedianus*).

The BRUVS dataset did not show higher total abundance, species richness or total biomass in the marine park than outside, but the assemblage composition did differ between the areas sampled. Five species or groups were more abundant within the park, including Draughtboard Shark (*C. laticeps*), perch (*Caesioperca* spp.), Velvet Leatherjacket (*Meuschenia scaber*), Sand Flathead (*Platycephalus bassensis*) and Common Gurnard Perch (*Neosebastes scorpaenoides*). Four of these species, particularly *P. bassensis* and *N. scorpaenoides*, could be considered fisheries targets and are likely benefiting from protection within the park. Distinct fish assemblages were also recorded for each habitat type (soft sediment, infralittoral reef and circalittoral reef) within the BRUVS dataset. Low-complexity soft-sediment areas were characterised by Sand Flathead (*P. bassensis*) and Common Gurnard Perch (*N. scorpaenoides*), Degen's Leatherjacket (*T. degeni*), Yellowtail Scad (*T. novaezelandiae*) and Gummy Shark (*M. antarcticus*). Soft-sediment habitats also had the lowest species richness of any habitat type.

Circalittoral and infralittoral reef had higher species richness than did soft sediment. Similarities between the assemblages in the 2 reef areas included high abundances of *Caesioperca* spp., *P. rubicundus*, Six-Spine Leatherjacket (*Meuschenia freycineti*) and *N. tetricus* in both habitats. The habitat types differed in that circalittoral reef was characterised by high abundances of Jackass Morwong (*Nemadactylus macropterus*) and Longfin Pike (*Dinolestes lewini*) while infralittoral reef was characterised by Maori Wrasse (*Ophthalmolepis lineolata*) and Silver Sweep (*Scorpiis lineolata*).

The fish assemblages observed via BRUVS differed from the assemblages observed via Reef Life Survey (RLS) sampling. For fish species alone (i.e. bony fish and sharks/rays), 50 species were common to both methods, 40 species were unique to BRUVS and only 19 species were unique to RLS. The species unique to BRUVS included large, more mobile species, such as sharks and rays (e.g. Melbourne Skate (*Spiniraja whitleyi*), Southern Sawshark (*Pristiophorus nudipinnis*) and Gummy Shark (*M. antarcticus*)) and species that are more common in deep waters or sandy areas, such as Red Gurnard (*Chelidonichthys kumu*), flathead (*Platycephalus* spp.) and Red Gurnard Perch

(*Helicolenus percooides*). The species unique to RLS included those which are small, cryptic and associated with habitat, such as Little Rock Whiting (*Neoodax balteatus*), Weedy Seadragon (*Phyllopteryx taeniolatus*) and Common Threefin (*Trinorfolkia clarkei*).

Reef habitat was assessed using video from BRUVS surveys. Two BRUVS surveys have been conducted at Wilsons Promontory MNP, allowing for habitat change to be observed for the matching sites across both years. A shift in habitat type from pebble cover to coarse sand was observed in the deep depth stratum (>60 m); however, overall patterns in habitat type were similar between the years sampled. The shift in substratum type over time could be an indication of the sediment dynamics present in this park, which experiences high currents as water moves around the point.

Including BRUVS in monitoring also allowed for the creation of spatially explicit models for various diversity metrics and abundance of key groups and species throughout the park, aided by the distribution of sample locations across environmental gradients observed. Planktivore abundance, carnivore abundance and relative species richness were found by models to increase in shallower depths and in closer proximity to reef. Total abundance was also found to increase as depths became shallower, whereas total biomass increased with increasing depth.

Overall, many key species and habitat types in Wilsons Promontory MNP were found to be in good condition. Most key mobile fish species had healthy abundances, as did ecologically important macroinvertebrates like Blacklip Abalone (*Haliotis rubra*) and Orange Feather Star (*Cenolia trichoptera*), while major canopy-forming macroalgae had fair to good cover. However, there are signs of decline for several key species of fish, macroinvertebrates and, significantly, large macroalgal species. The underlying drivers of this decline are unclear.

The only major change in threats identified in this monitoring program was an increase in summer sea-surface temperatures. The impacts of increased temperatures on species in Wilsons Promontory MNP has not been studied in detail and warrants further investigation. More generally, ongoing long-term monitoring and further investigations of species in potential decline are needed to understand the drivers and guide management responses. Both underwater visual census and BRUVS surveys found that 10 species were more abundant inside the MPA than outside, which included some species commonly the target of fisheries. This suggests these species benefit from the no-take marine national park. However, species richness, total abundance and total biomass did not differ inside and outside the marine national park, which may also reflect a low impact of fishing pressure in areas surrounding the park.

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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 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 for its marine protected areas (MPAs) to improve baseline knowledge of the MPAs and address 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 their monitoring program is guided by priorities identified through the conservation planning process for the marine national parks and sanctuaries.

1.1 Signs of Healthy Parks marine protected area monitoring framework

Parks Victoria has established the Signs of Healthy Parks (SHP) program for its network of marine national parks and sanctuaries with the dual aims of improving baseline knowledge of Victoria's 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.

The SHP program continues and enhances Parks Victoria's historical Subtidal Reef Monitoring Program (SRMP) that ran from 1998 to 2014 (Edmunds and Hart, 2003; Lindsay and Edmunds, 2006). SRMP used diver underwater visual census methods to describe patterns and trends in macroalgal, fish and macroinvertebrate communities within shallow subtidal sites in MPAs. The SHP program also continues the Intertidal Reef Monitoring Program (IRMP) that ran from 2002 to 2014 (Stewart et al., 2007). IRMP monitored invertebrates and macroalgae in the intertidal zone on reefs within 9 targeted MPAs as well as at 9 matched reference sites outside the MPAs. These programs were designed using the best scientific practices of the time but only covered a small proportion of the key habitats in the parks. Using advances in ocean technology, and through a collaborative research partnership with Deakin University, the SHP program now monitors the health of the entire extent of parks, 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 Marine National Park (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 Wilsons Promontory MNP.

1.2 Wilsons Promontory Marine National Park

Wilsons Promontory MNP surrounds the southern tip of Wilsons Promontory, a peninsula forming the southernmost part of the Australian mainland. Most of the land adjoining Wilsons Promontory is protected by the Wilsons Promontory National Park. Wilsons Promontory MNP is the only marine national park in the Flinders bioregion of Victoria, one of the Integrated Marine and Coastal Regionalisation of Australia (IMCRA) meso-scale regions. Flinders bioregion encompasses the eastern entrance to Bass Strait and the Bass Strait islands.

Wilsons Promontory MNP protects important natural values, including shallow subtidal reefs, deep subtidal reefs, intertidal rocky shores, sandy beaches, seagrass, subtidal soft substrates and expansive areas of open water as well as biological communities with distinct biogeographic patterns. It has a wide range of habitat types ranging from low to high wave exposure. It has unusual granite habitats, with extensive reefs with smooth surfaces, boulders and rubble, and low-profile reefs. Shallower reefs have extensive communities of abundant red and brown macroalgae dominated by Crayweed (*Phyllospora comosa*) and Golden Kelp (*Ecklonia radiata*). Its deep reefs have a dense cover of epifauna, especially sponges, stalked ascidians and sea whips, and abundant fish life. Its soft sediment has diverse biotic assemblages. Communities of mobile invertebrates are characterised by abundant Blacklip Abalone (*Haliotis rubra*), Purple Sea Urchin (*Heliocidaris erythrogramma*) and Orange Feather Star (*Cenolia trichoptera*) as well the Large-Plated Seastar (*Nectria macrobrachia*) in the south-west and the seastars *Meridiastra gunnii* (Gunns Six-Armed Star) and *Petricia vernicina* (Velvet Star) in the north-west. Wilsons Promontory MNP is considered to have relatively high fish species richness and diversity along with higher-than-average abundances for most fish species compared to other areas in central Victoria.

1.3 Threats to natural values in Wilsons Promontory MNP

Marine invasive or overabundant species pose an extreme risk to a range of marine assets within Wilsons Promontory MNP. Marine pests are largely absent from the park itself, but the invasive Northern Pacific Seastar (*Asterias amurensis*), New Zealand Screw Shell (*Maoricolpus roseus*), Pacific Oyster (*Magallana gigas*) and Japanese Kelp (*Undaria pinnatifida*) are found in waters adjacent to the park.

The level of illegal fishing in the marine national park is considered low due to the distances needed to travel from the nearest port. However, it is still a recognised threat that has potential to affect fish and shark populations as well as invertebrates such as Southern Rock Lobster (*Jasus edwardsii*) and abalone.

The proximity of shipping routes through the park and the islands to the south of Wilsons Promontory means oil spills are a potential threat to the park, but no significant spills have occurred in this area to date.

Increasing sea-surface temperatures (SSTs) and the strengthening of the East Australian Current predicted in response to climate change have the potential to significantly impact the natural values of the park. South-eastern Australia is considered a national climate change hotspot for increases in SST and ocean acidification. Shifts into eastern Victoria of species historically found further north, such as the Longspined Sea Urchin (*Centrostephanus rodgersii*) and Sydney Octopus (*Octopus tetricus*), have the potential to significantly impact biota in Wilsons Promontory MNP.

1.4 Objectives

The overarching objective of this project was to design and deliver a targeted monitoring program for Wilsons Promontory MNP with the dual aims of understanding the condition of major conservation assets and identifying threats to those assets. Specific objectives related to this overarching goal were to:

- compile seabed bathymetry data to develop full-coverage maps of Wilsons Promontory MNP and adjacent waters to help characterise the seabed and develop species distribution models
- understand changes in sea-surface temperature inside the park to better understand risk from climate change
- assess trends in the abundance and diversity of fish and invertebrate populations and canopy-forming algae through time using underwater visual census (Reef Life Survey) methods at historical SRMP MPA and reference sites on shallow subtidal reefs to understand long-term changes in the condition of key species and habitat
- assess abundance and diversity of fish populations using baited remote underwater video station (BRUVS) surveys of demersal fish assemblages within and adjacent to the MPA to
 - develop time series monitoring of demersal fish assemblages across the entire depth range of the park
 - understand the effectiveness of no-take protection on the abundance and diversity of fish
 - understand the effect of seafloor structure on abundance and diversity of fish using spatially explicit distribution modelling techniques.

2. Methods

2.1 Seabed of Wilsons Promontory MNP and adjacent waters

A high-resolution bathymetric grid (horizontal resolution of 1 metre (m)) was obtained for Wilsons Promontory MNP (Figure 2.1). From this, 4 derivatives of bathymetry were calculated to further our understanding species distribution models: bathymetry, slope, rugosity (surface roughness) and distance to reef (Table 2.1). For more details on seabed analyses, see Ierodiaconou et al. (2020).

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)

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

2.2 Sea-surface temperature

SST 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 2019 from the monthly SST datasets. To assess patterns and trends in seasonal SST within Wilsons Promontory 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 can determine trends from stochastic datasets like the SST observations in Wilsons Promontory MNP.

2.3 Reef Life Survey for shallow subtidal reefs

Using the methods developed by the Reef Life Survey (RLS) citizen science program, surveys were completed at all historical Subtidal Reef Monitoring Program (SRMP) sites associated with Wilsons Promontory MNP (21 sites) in 2014–15, 2017 and 2020 (Figure 2.1). The data collected were in addition to the data collected in the SRMP surveys, which were conducted at 28 sites in 1999–2002, 2004–06, 2010 and 2011. In recent years, Reef Life Survey has become a common method for completing underwater visual census 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, described in detail in an online methods manual (Reef Life Survey, 2015).

RLS surveys encompass an underwater visual census 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 Wilsons Promontory 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, for each location, an area of 1,000 m² is surveyed for fish and 200 m² for invertebrates.

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 25 evenly distributed points on a 5 × 5 grid. Each point is classified to the lowest taxonomic level possible using an adapted 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 is 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 only compares canopy-forming species to previous SRMP observations, as understory communities are likely to be obscured in photo quadrats.

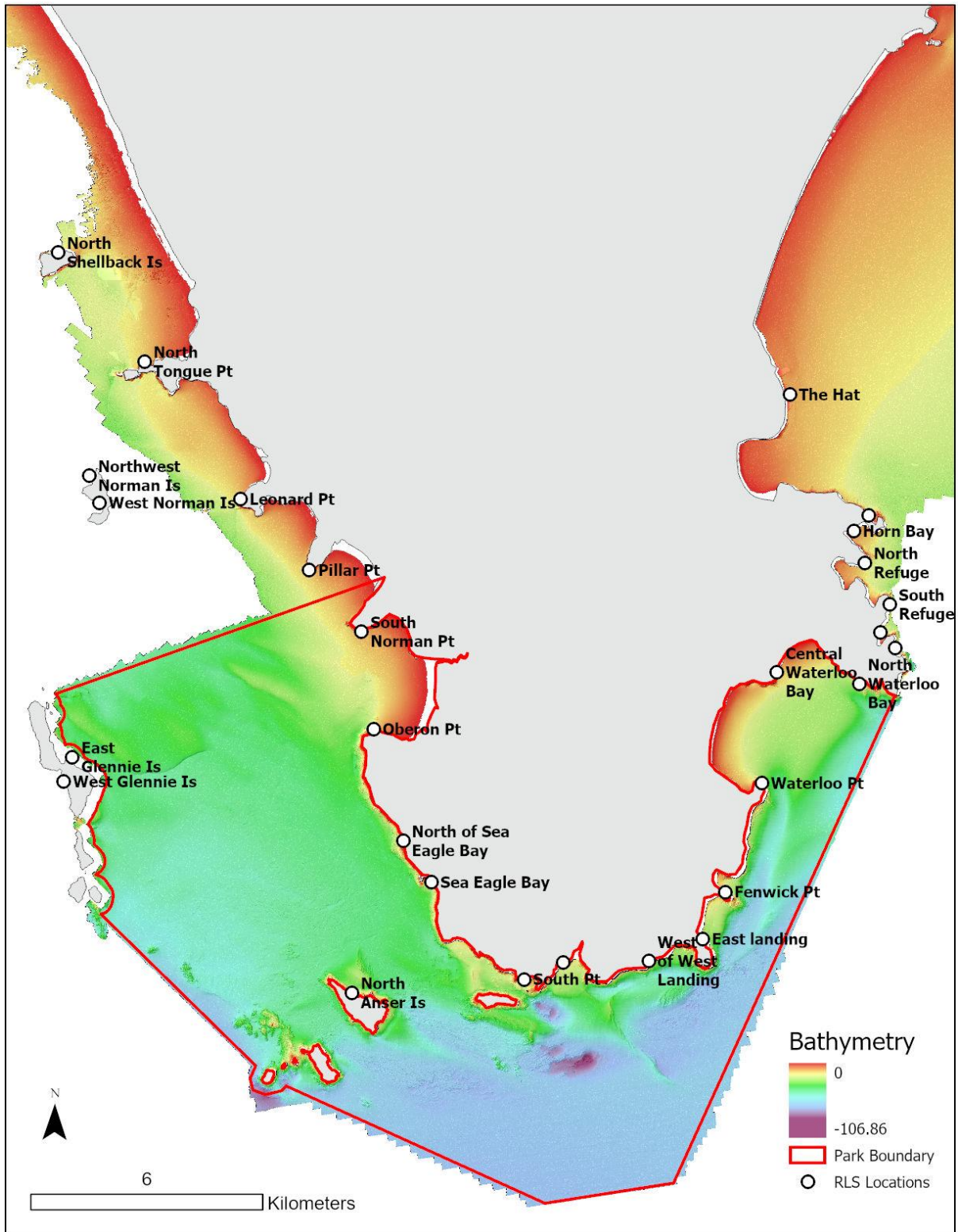


Figure 2.1: Map of the Wilsons Promontory region showing the locations of all monitoring sites (SRMP and RLS). 21 RLS sites were sampled for this project.

2.4 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 LCL is used to inform the management of marine protected areas and is derived from LLAC. The LCL indicates a limit that should trigger action by Parks Victoria which can include further investigation, research or monitoring to understand why species have fallen below an LCL and, where a threat had been identified that can be managed, appropriate management action to reduce the impact of threats. The lower limit of acceptable change (LLAC) was initially identified as the minimum value inside the MPA from SRMP surveys before the Wilsons Promontory Marine National Park was declared (1999–2002). 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 (1999–2002, 2004–2006, 2010 and 2011) and RLS surveys (2014–15, 2017 and 2020). 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 are shaded green, control limits were indicated by dashed horizontal lines, and areas outside the control limits are shaded red.

2.5 Baited remote underwater video stations

2.5.1 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. Over 2 years (2016 and 2021), 151 BRUVS surveys were completed within Wilsons Promontory MNP. A subsample of these 151 BRUVS was selected to help track changes through time. These locations were subsampled by first stratifying sites by depth and complexity and ensuring a minimum distance of 300 m between sites. These sites were also subsampled from previous BRUVS locations, where possible. 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 include an unsurveyed wedge of the park in the north-east. The new sites selected to extend coverage of BRUVS outside the park were also stratified by depth and complexity.

2.5.2 Video collection and annotation

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 (*Sardinops sagax*) to attract fish into the field of view for relative 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 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 millimetre (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. For each site, total relative biomass 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. Where similar species were indistinguishable from the video footage, they were classed to family or genus level where possible, for example, trevally (*Pseudocaranx* spp.) and perch (*Caesioperca* spp.). Morid cods were grouped into *Pseudophycis* spp., although they may have also been from the genus *Lotella*.

2.5.3 Habitat classification from BRUVS

Broad classifications of habitat were derived from the field of view of each BRUVS deployment, 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:

- 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.

2.5.4 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). Analyses were conducted to investigate the influence of sampling period (year), protection status (inside or outside the marine park) and habitat type (infralittoral reef, circalittoral reef or soft sediment) on the overall assemblage observed (multivariate) and on total abundance, total biomass and species richness individually (univariate).

Within each year (2016 and 2021), all sites were assessed for the influence of habitat type on fish assemblages (fixed factor, 3 levels: circalittoral reef, infralittoral reef, soft sediment). For the 2021 dataset, fish assemblages were also assessed for the influence of status (fixed factor, 2 levels: no-take (inside the park), fished (outside the park)). 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. Comparison of assemblages between years (2016 vs. 2021) included the 49 sites that geographically matched to within 100 m. This matched dataset was analysed using the factors of year (random factor, 2 levels) 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, was 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 a strong correlation (>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

To understand relationships between fish abundance and environmental variables, 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 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 17 classes across the region would result in low 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 all sites, derivatives characterising the seabed terrain were extracted using the 'raster' and 'spatialEco' packages from the programming language R. Derivatives used included bathymetry, slope, rugosity (surface roughness) and distance to reef (Table 2.1).

3. Results

3.1 Trends in sea-surface temperature

Figure 3.1 shows summer, winter and annual sea-surface temperatures (SSTs) from 1993 to 2018. The trend analyses on the annual and seasonal patterns in SST show that only summer SST experienced an overall increase; annual temperatures tended to fluctuate and there was no clear trend in winter temperatures (Table 3.1, Figure 3.2).

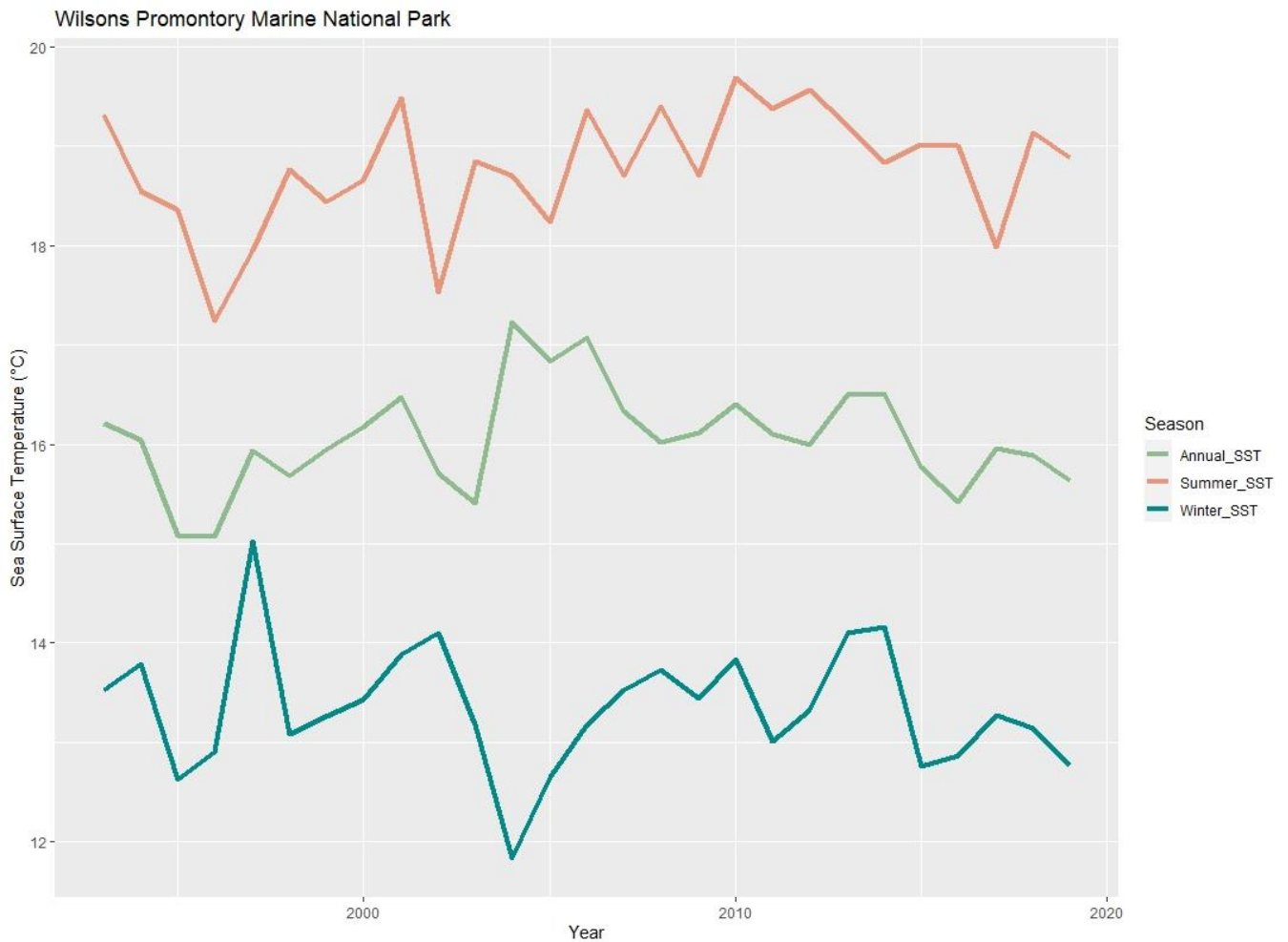


Figure 3.1: Annual and seasonal patterns in sea-surface temperature within Wilson's Promontory MNP

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	0.0330	0.0296	-0.0251	0.0911
Summer	0.0195	0.0210	-0.0216	0.0606
Winter	0.0000	0.0041	-0.0086	0.0086

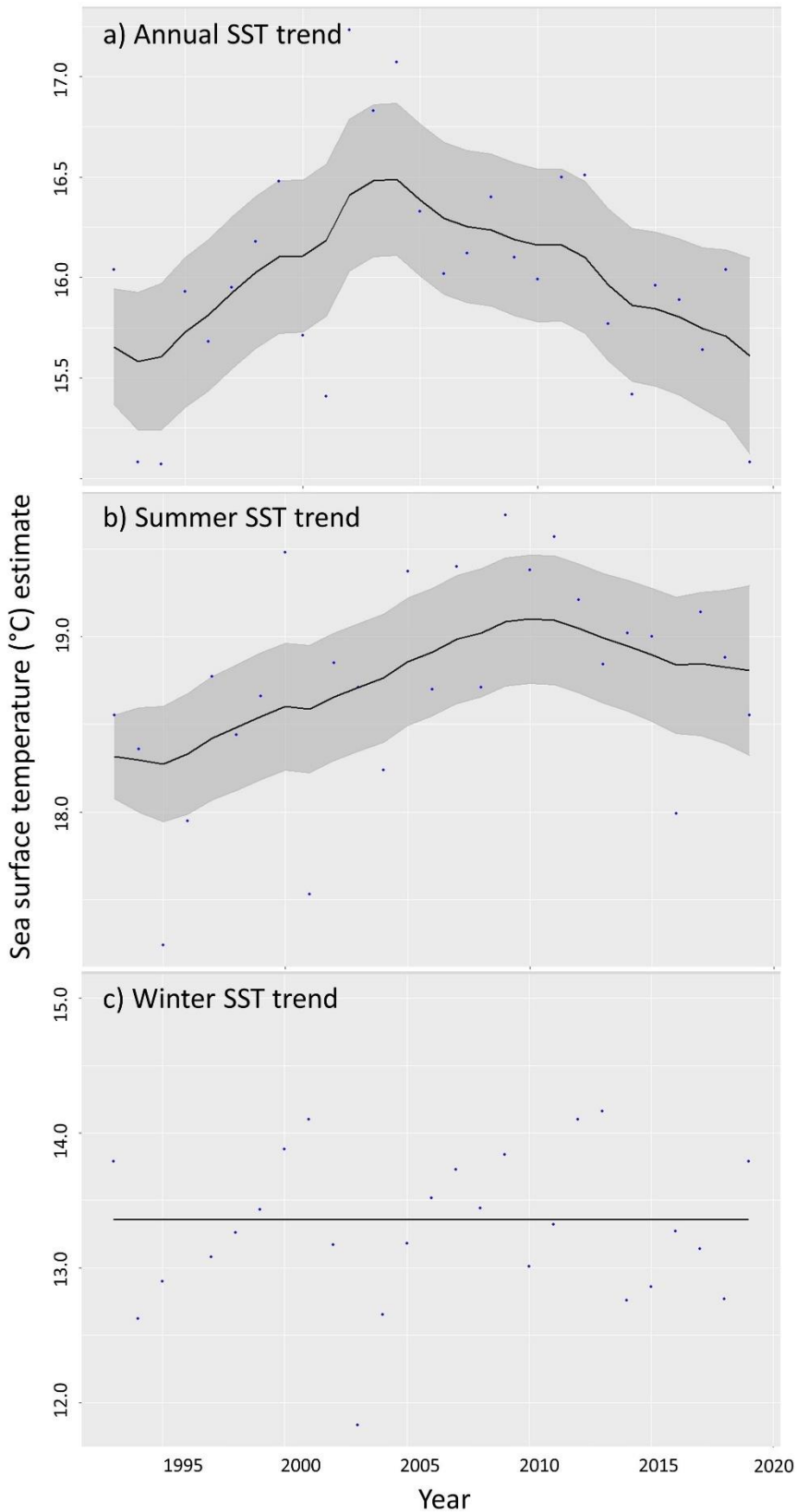


Figure 3.2: The (a) annual, (b) summer and (c) winter sea-surface temperature values (points) and the model-estimated SST (solid line). The solid lines are the means of the stochastic levels; the confidence intervals of that model are shown as 2 standard deviations (the shaded area)

3.2 Reef Life Survey for shallow subtidal reefs

When combined with results from the previous SRMP, data from shallow subtidal reefs in Wilsons Promontory MNP now span over 20 years. This is beneficial for understanding trends in fish communities, which previous research has shown can exhibit substantial differences in diversity and abundance between seasons and require longer data time series to identify trends (particularly summer and autumn compared to winter) (Jung et al., 2010).

From 2014 to 2020, RLS surveys captured a total of 54,992 observations of 167 individual species of fish (non cryptic) and 42,316 observations of 130 individual species of invertebrates and cryptic fish. In addition, 37,405 points were classified for benthic habitat from a total of 1,490 photo quadrats, allowing canopy cover metrics to be extracted for comparison with previous SRMP data. Common species of fish observed across years included Purple Wrasse (*Notolabrus fucicola*), Blue-Throat Wrasse (*N. tetricus*), Old Wife (*Enoplosus armatus*), Barber Perch (*Caesioperca rasor*), Butterfly Perch (*C. lepidoptera*) and Sea Sweep (*Scorpius aequipinnis*). Common species of invertebrates observed across years included Blacklip Abalone (*Haliotis rubra*), Purple Sea Urchin (*Heliocidaris erythrogramma*), Orange Feather Star (*Cenolia trichoptera*), Gunns Six-Armed Star (*Meridiastra gunnii*) and Large-Plated Seastar (*Nectria macrobrachia*). Common species of macroalgae observed across all years include Golden Kelp (*Ecklonia radiata*), Crayweed (*Phyllospora comosa*), the brown macroalga *Seirococcus axillaris* and 2 species of red algae (*Phacelocarpus peperocarpus* and *Plocamium angustum*).

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

	2014	2015	2017	2020	Total
Number of sites completed	2	2	20	20	44
Area surveyed for fish (m²)	2,000	2,000	20,000	20,000	44,000
Area surveyed for invertebrates (m²)	400	400	4,000	4,000	8,800
Number of photo quadrats classified for benthic habitat	70	70	559	791	1,490
Number of points classified for benthic habitat	1,800	1,800	13,975	19,830	37,405
Number of fish (non cryptic) observations	1,165	2,029	31,538	20,260	54,992
Number of fish species (non cryptic) observed	26	26	64	51	167
Number of invertebrates and cryptic fish observations	2,190	523	19,596	20,007	42,316
Number of invertebrate and cryptic fish species observed	20	12	55	43	130

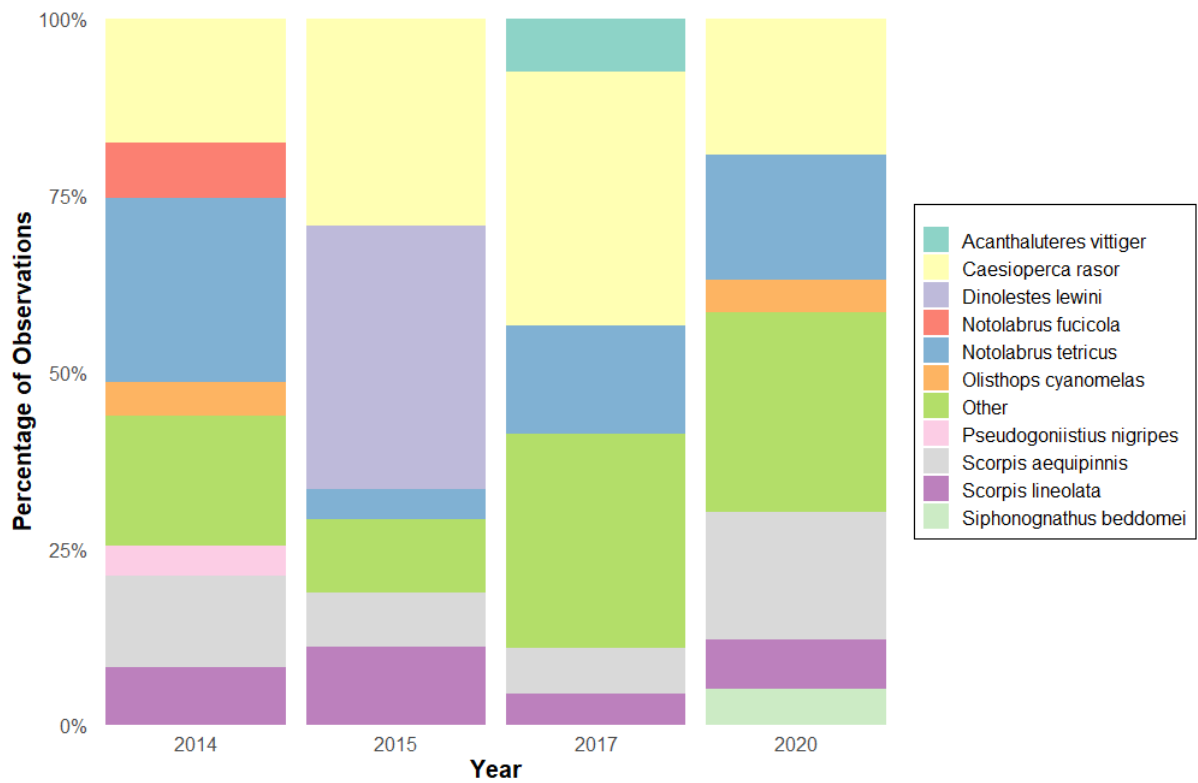


Figure 3.3: Proportions (as percentages) of the total observations of fish made in Wilsons Promontory Reef Life Surveys between 2014 and 2020. ‘Other’ combines all species whose proportion of the total observations is less than 4%

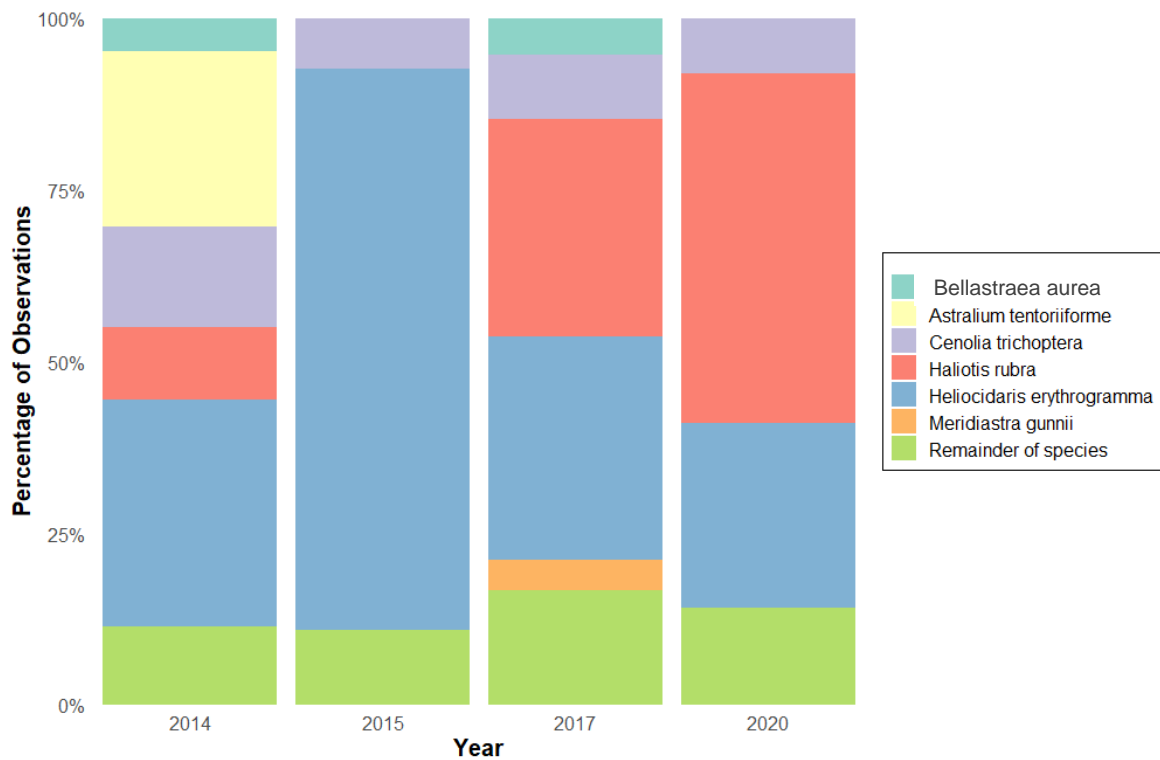


Figure 3.4: Proportions (as percentages) of the total observations of invertebrates made in Wilsons Promontory MNP between 2014 and 2020. ‘Remainder of species’ combines all species whose proportion of the total observations is less than 4%

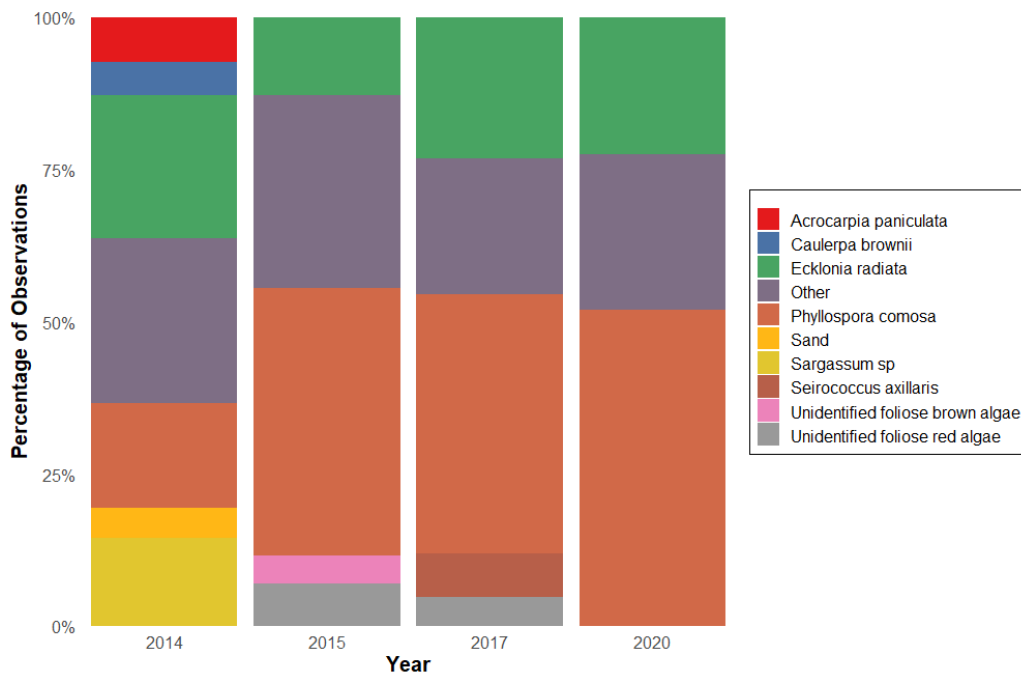


Figure 3.5: Proportions (as percentages) of the total observations of macroalgae and sand made in Wilsons Promontory MNP between 2014 and 2020. ‘Other’ represents all categories whose proportion of the total observations is less than 4%

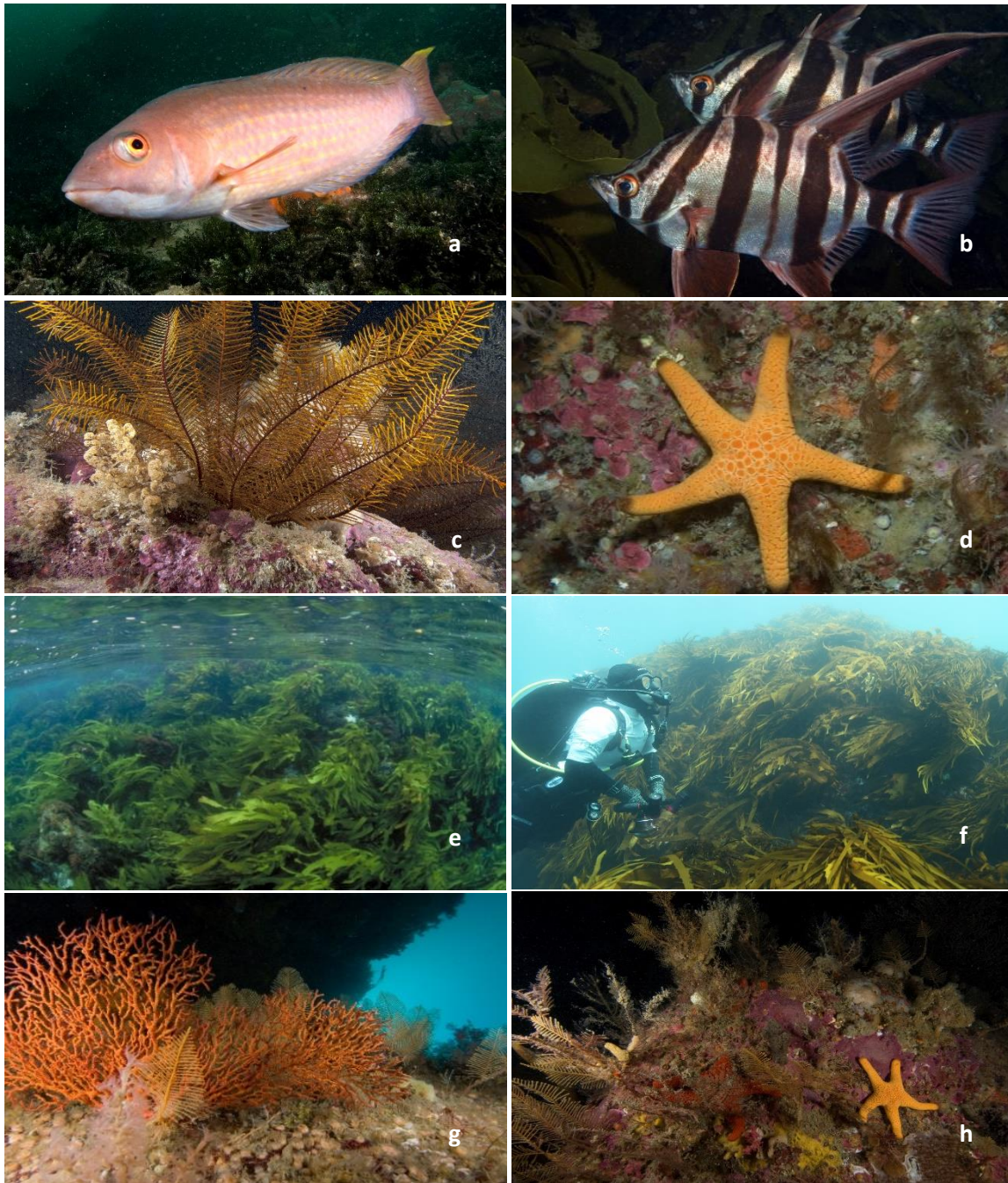
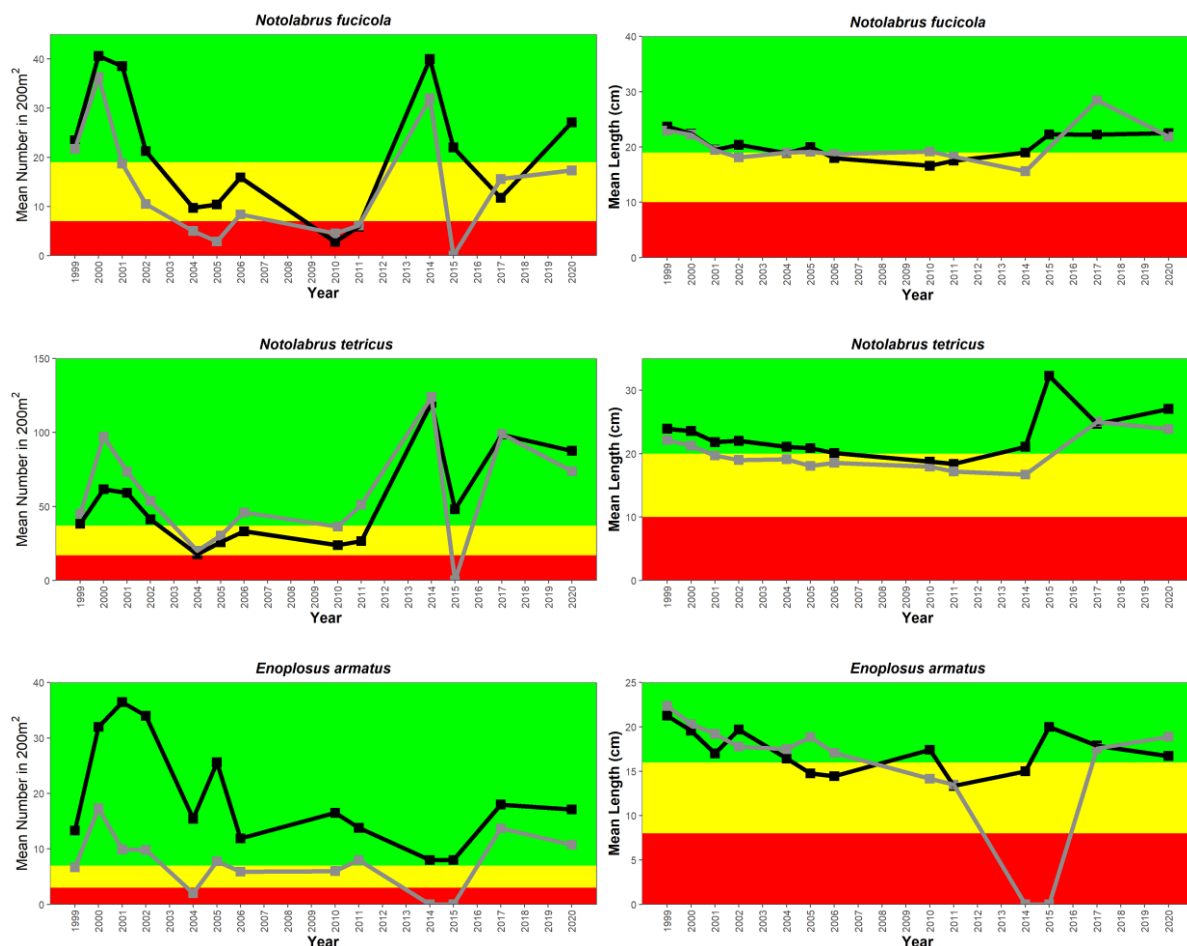


Figure 3.6. Example photos of key species and habitat types found in surveys of Wilsons Promontory Marine National Park. **(a)** Rosy Wrasse (*Pseudolabrus rubicundus*) **(b)** Old Wife (*Enoplosus armatus*) **(c)** Orange Feather Star (*Cenolia trichoptera*) **(d)** Large-Plated Seastar (*Nectria macrobrachia*) **(e)** macroalgal beds dominated by Golden Kelp (*Ecklonia radiata*) beds **(f)** macroalgal beds dominated Crayweed (*Phyllospora comosa*) beds **(g)** invertebrate complexes with gorgonian corals **(h)** invertebrate complexes with soft corals, sponges and seastars (*Nectria spp.*). Photo credits: Museum Victoria/Julian Finn (a, b, d), Museum Victoria/Mark Norman (c, g,h), Parks Victoria (e, f).

3.2.1 Key mobile fish species

Control charts show that abundances in 2020 of 3 key mobile fish species (*Notolabrus tetricus*, *Enoplosus armatus* and *Scorpius aequipinnis*) fell within the zone of ‘good’ condition both inside and outside the park, while 3 species fell below the lower limit of acceptable change in one or more locations (Figure 3.7). *Notolabrus fucicola* in 2020 was below the lower limit of acceptable change outside the MPA, but it has been on an upward trajectory since 2015. *Caesioperca rasor* abundance within the MPA in 2020 was below the lower limit of acceptable change and on a downward trend. *Caesioperca lepidoptera* was not observed within or outside the MPA in 2020. In 2020, 4 of the 6 species analysed (*N. fucicola*, *N. tetricus*, *E. armatus* and *S. aequipinnis*) were more abundant inside the park than outside, while *C. rasor* was more abundant outside the park.

The sizes of individual fish were generally similar within and outside the park. Sizes of 5 species fell within the zone of good condition (i.e. above the lower limit of acceptable change) while *C. lepidoptera* fell within the red zone (i.e. below the lower control limit).



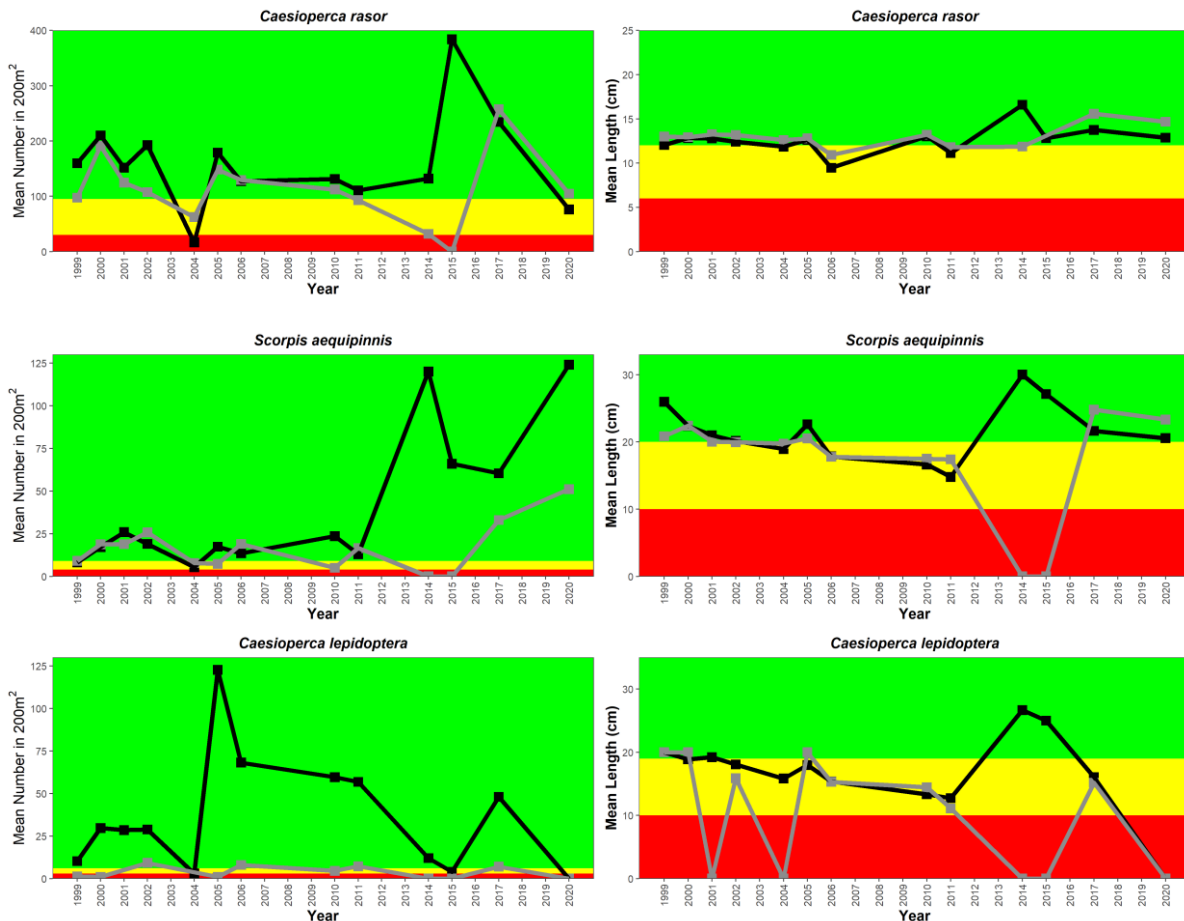
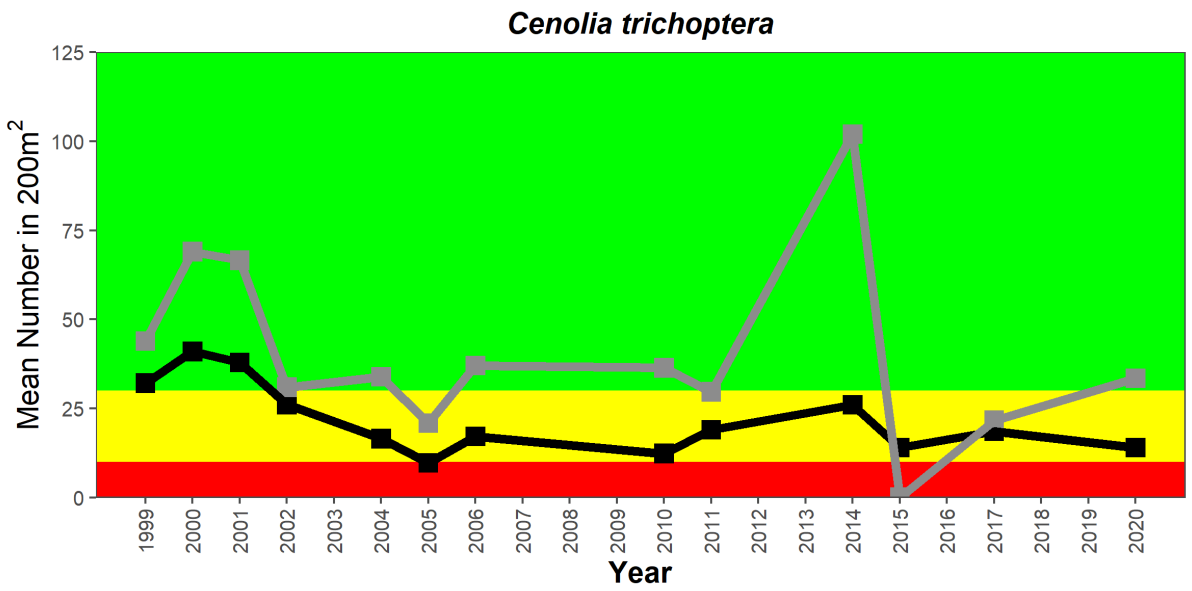
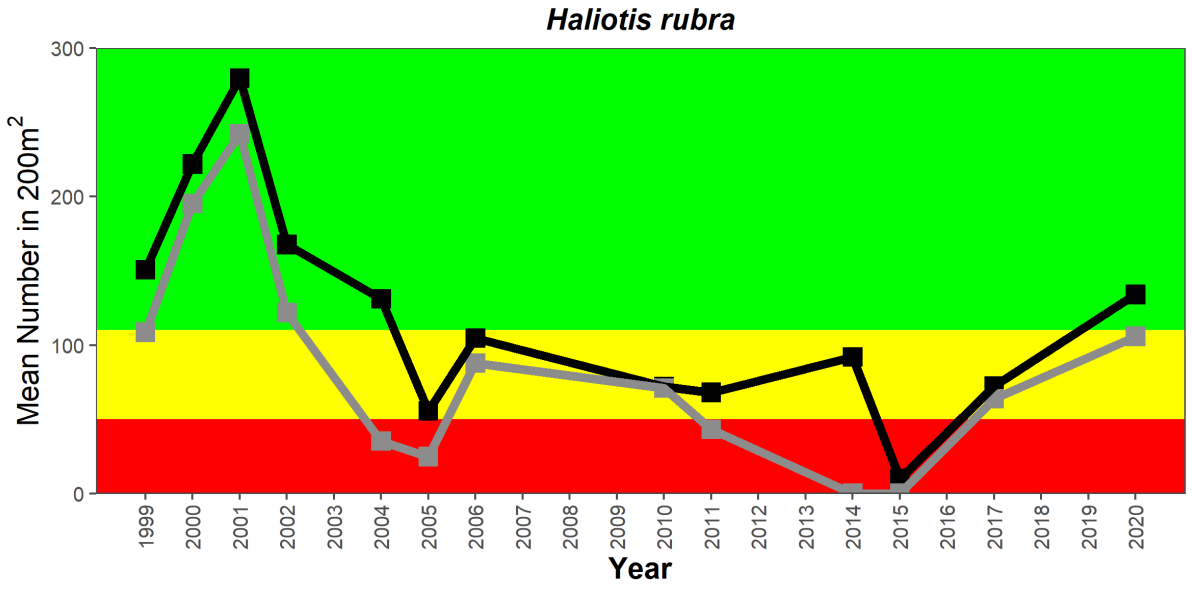
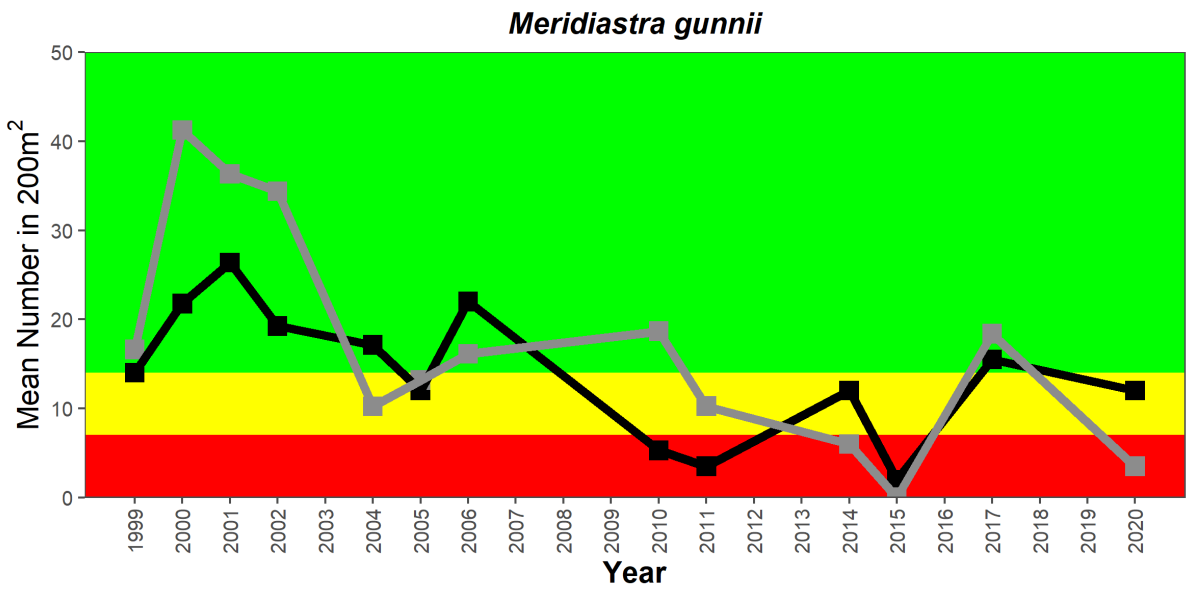
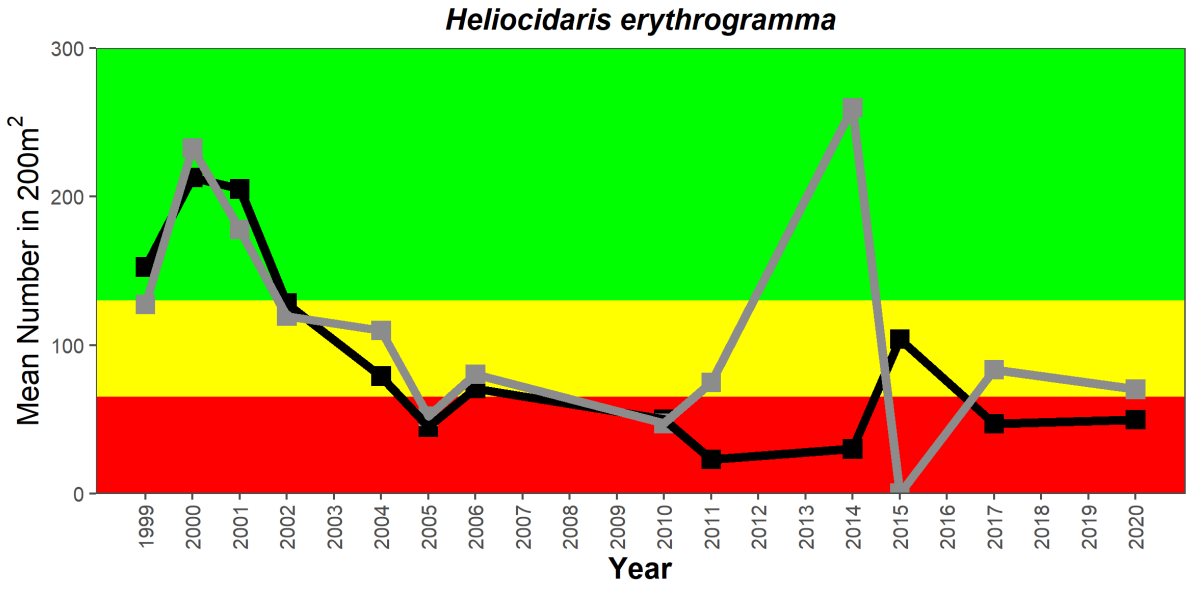


Figure 3.7: Control charts showing abundance (left) and length (right) of key mobile fish species within Wilsons Promontory MNP (black line) and reference sites outside the park (grey line). Observations for these charts were sourced from SRMP and Reef Life Survey diver transects. These charts show the lower limit of acceptable change (LLAC, top of the yellow band – set as the minimum value inside the MPA from SRMP surveys from 1999 to 2001) and lower control limit (LCL, top of the red band) based on the variation from surveys. The LCL indicates the level at which conditions are sufficiently poor that some management response is required

3.2.2 Key macroinvertebrates

Abundances of *Haliotis rubra* within the park and *Cenolia trichoptera* outside the park fell within the zone of good condition (i.e. above the lower limit of acceptable change) in 2020 (Figure 3.8). However, declining trends in abundance over the past 20 years were observed for *Meridiastra gunnii* and *Nectria macrobrachia*. In 2020, *M. gunnii* was below the lower limit of acceptable change inside the park and below the lower control limit outside the park. *Heliocidaris erythrogramma* has been below its lower limit of acceptable change of 125 individuals per 200 m² since 2002 within and outside the park and was below the lower control limit inside the park in 2020. Three of the 5 macroinvertebrates (*H. rubra*, *M. gunnii* and *N. macrobrachia*) were more abundant inside the park than outside, while *C. trichoptera* and *H. erythrogramma* were more abundant outside the park.





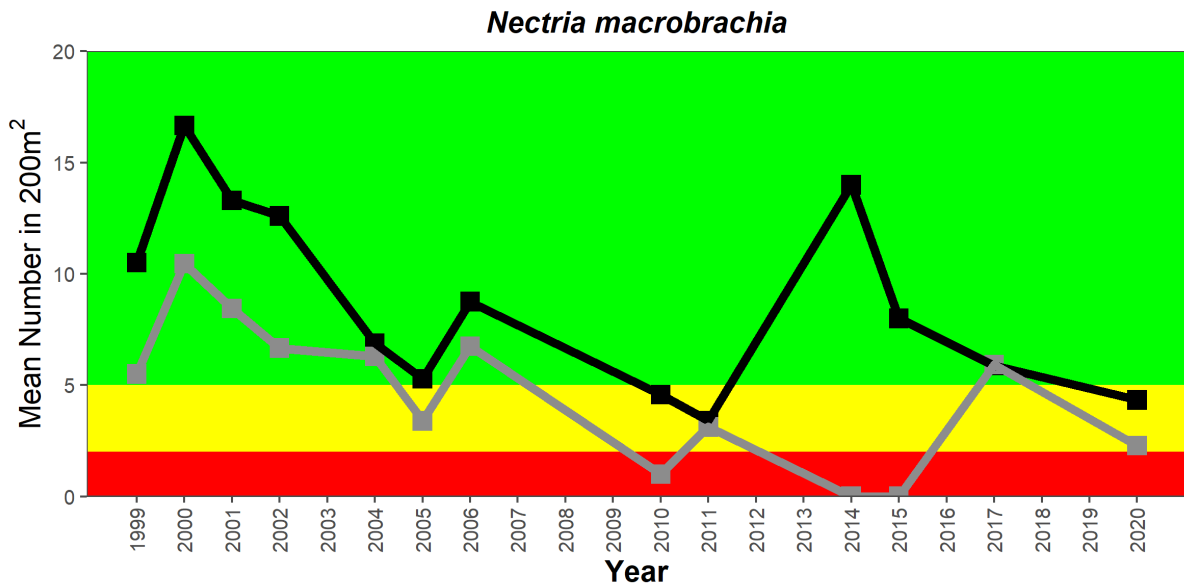
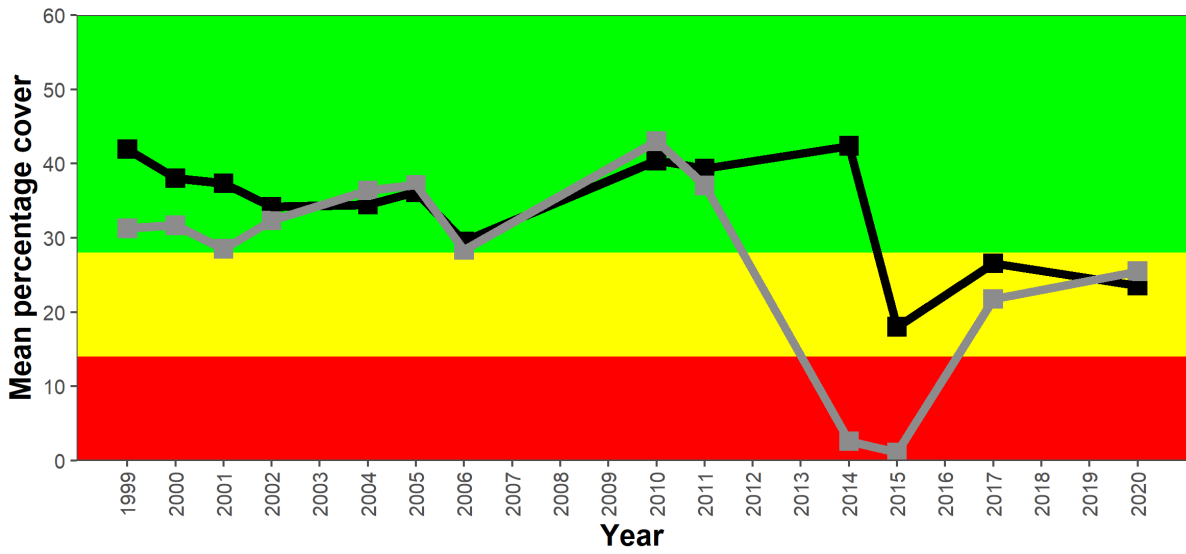


Figure 3.8: Control charts showing abundance of key motile macroinvertebrate species within Wilsons Promontory MNP (black line) and reference sites outside of the park (grey line). Observations for these charts were sourced from SRMP and Reef Life Survey diver transects. These charts have a lower limit of acceptable change (LLAC, top of the yellow band – set as the minimum value inside the MPA from SRMP surveys from 1999 to 2001) and lower control limit (LCL, top of red band) based on the variation from surveys. The LCL indicates the level at which conditions are sufficiently poor that some management response is required

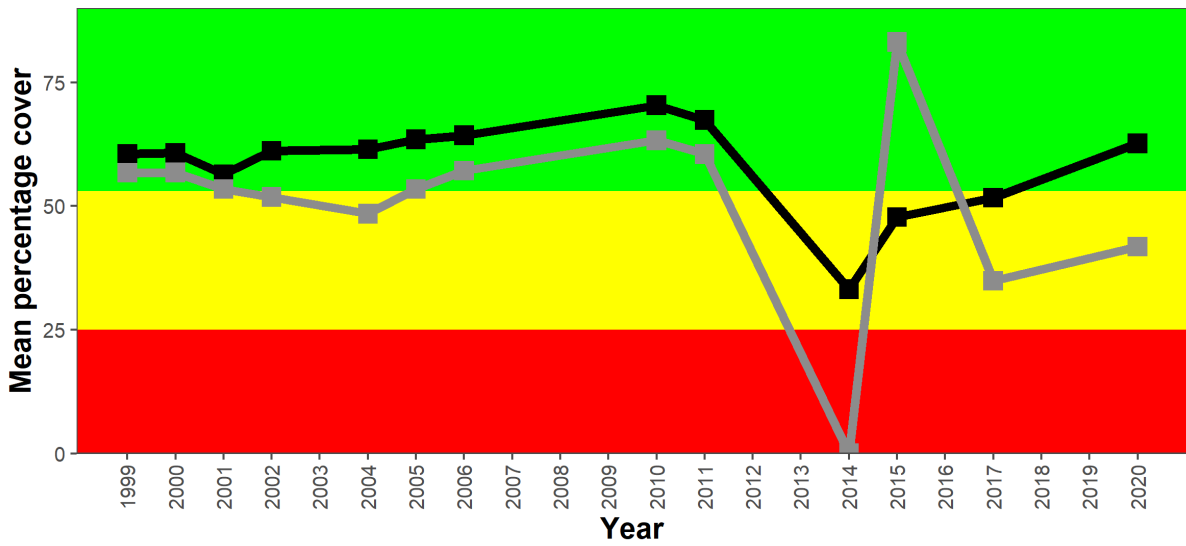
3.2.3 Macroalgal beds

The most notable change in the subtidal reef macroalgal communities of Wilsons Promontory MNP has been the decline in percentage cover of *Seirococcus axillaris*, *Phacelocarpus peperocarpos* and *Plocamium angustum* within and outside the MPA since 2010 and 2011 (Figure 3.9). Percentage cover of all 3 species has been below the lower control limit since 2014 (i.e. in the surveys in 2014, 2015, 2017 and 2020) within the park and for *P. peperocarpos* and *P. angustum* also outside the park. The percentage cover of *Phyllospora comosa* in 2020 was well above the lower limit of acceptable change within the MPA but below it outside. Percentage cover of *Ecklonia radiata* was below the lower limit of acceptable change within and outside the MPA, as has been the case since 2015. Of the species studied, only *E. radiata* and *S. axillaris* have higher percentage cover outside the park than inside, while *P. comosa* and *P. peperocarpos* have higher cover inside the park.

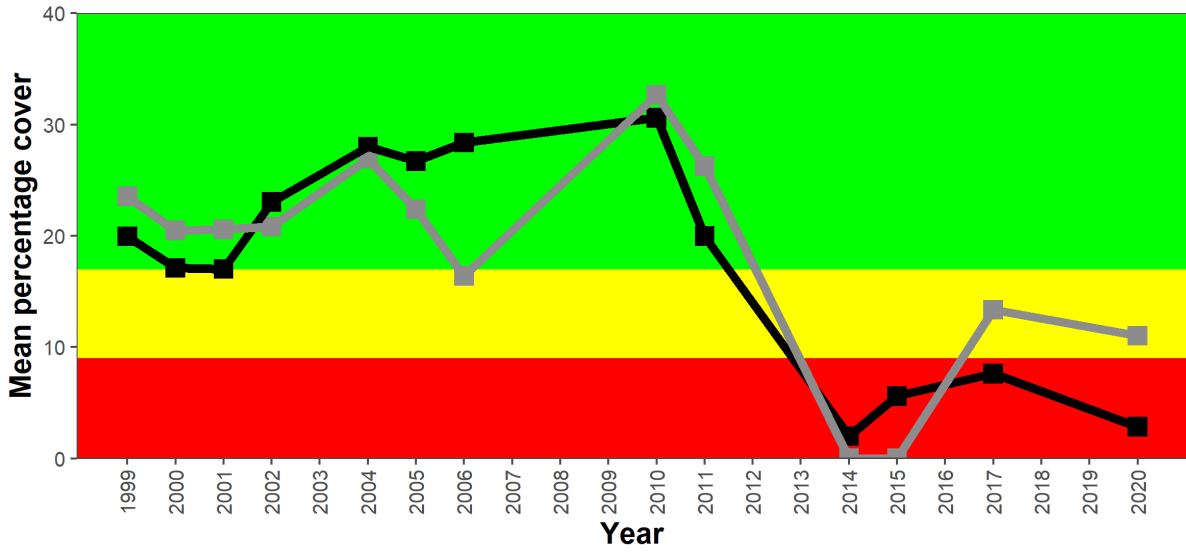
Ecklonia radiata



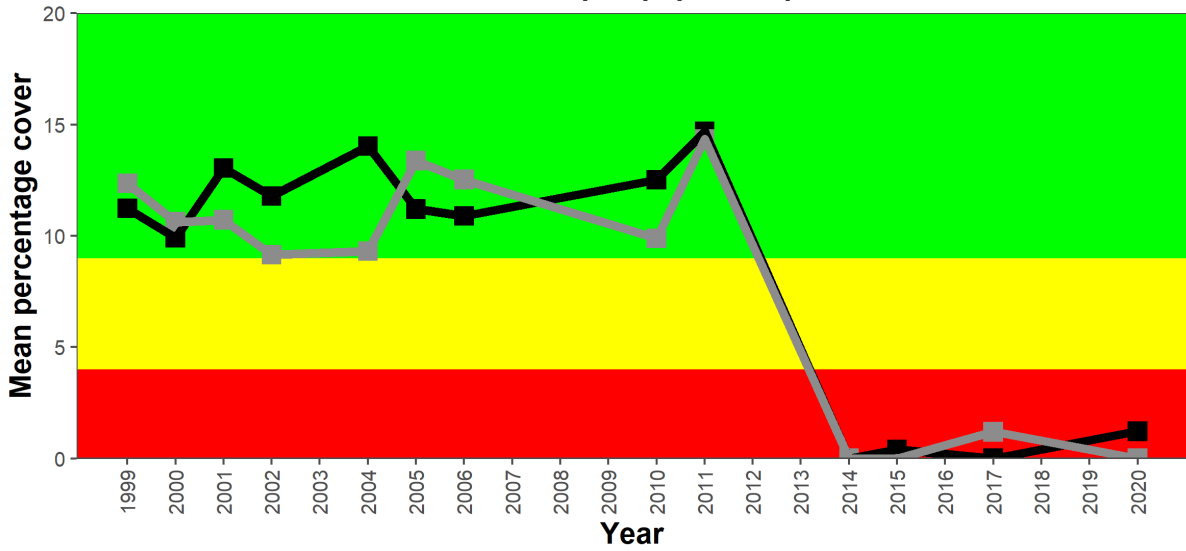
Phyllospora comosa



Seirococcus axillaris



Phacelocarpus peperocarpus



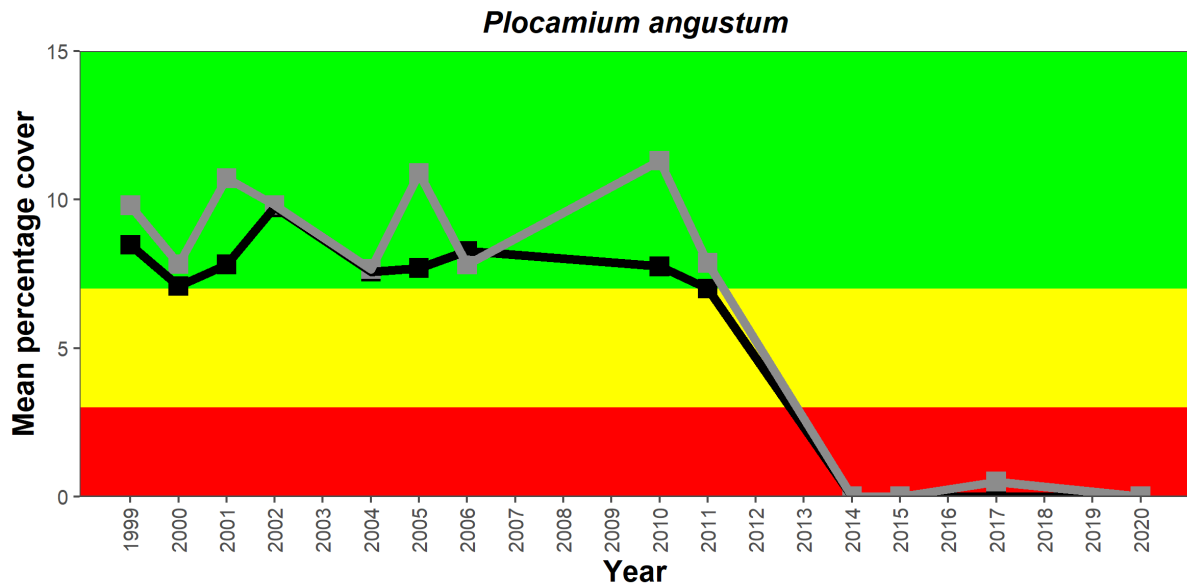


Figure 3.9: Control charts showing percentage cover of key macroalgae species within Wilsons Promontory MNP (black line) and reference sites outside of the park (grey line). Observations for these charts were sourced from SRMP and Reef Life Survey diver transects. These charts show the lower limit of acceptable change (LLAC, top of yellow band – set as the minimum value inside the MPA from SRMP surveys from 1999 to 2001) and lower control limit (LCL, top of red band) based on the variation from surveys. The LCL indicates the level at which conditions are sufficiently poor that some management response is required

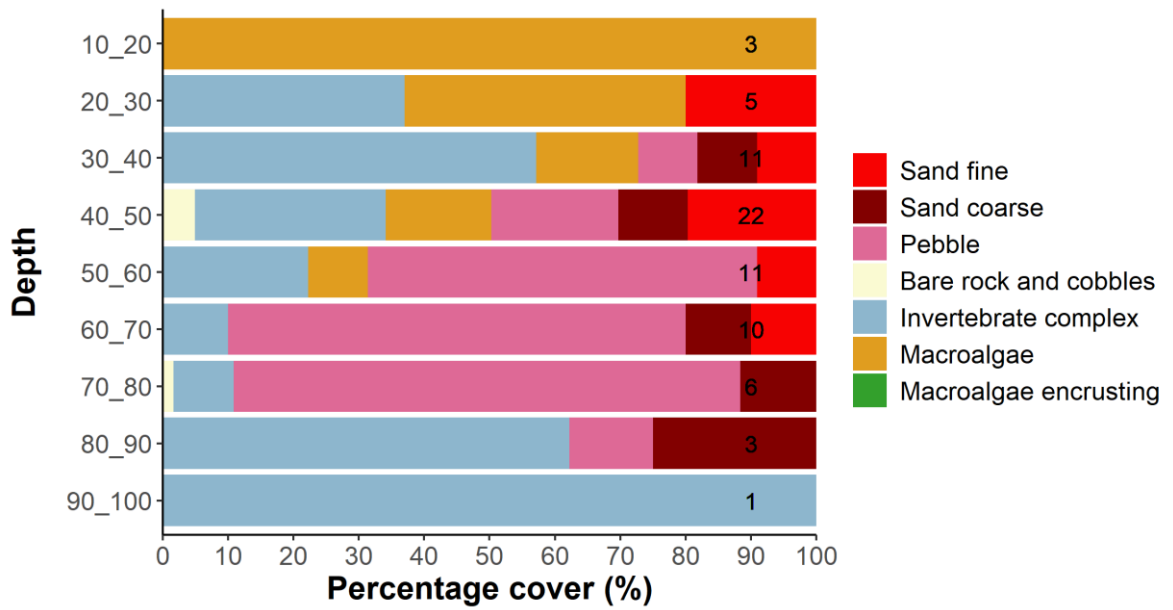
3.3 Baited remote underwater video stations (BRUVS) surveys

3.3.1 Habitat

Habitat types changed with depth across the sampling area (Figure 3.10). Across all sites in 2016, shallow sites (less than 40 m) were dominated by 2 habitat types: macroalgae and invertebrate complex. The middle and deep depth zones (40 to 100 m) had a mix of habitats dominated by soft sediments or pebble and by invertebrates. The results across all sites in 2021 were similar except that percentage cover of pebble was reduced.

The sites matched across both years showed a shift from pebble cover to coarse sand in the deep depth stratum (below 60 m) (Figure 3.11). However, patterns in habitat type were similar overall in 2016 and 2021.

a)



b)

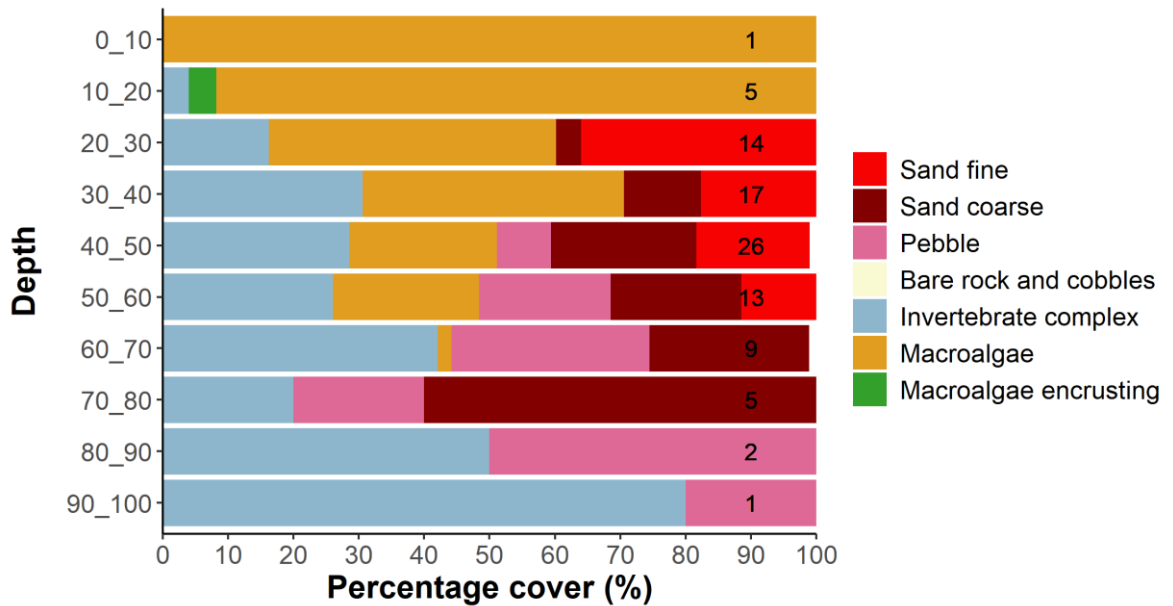
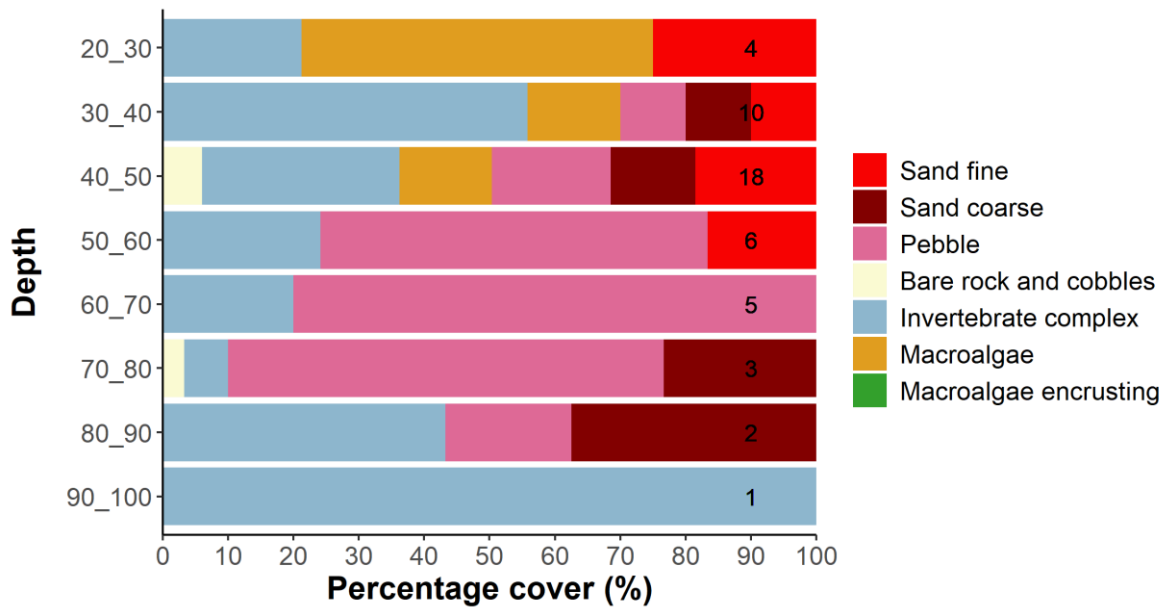


Figure 3.10: Percentage cover of different habitat components at different depths (m) calculated from the field of view of each BRUVS deployment for all sites in (a) 2016 and (b) 2021. The percentage cover of 'unknown' is not shown. The numbers on the chart represent the number of BRUVS in each depth zone

a)



b)

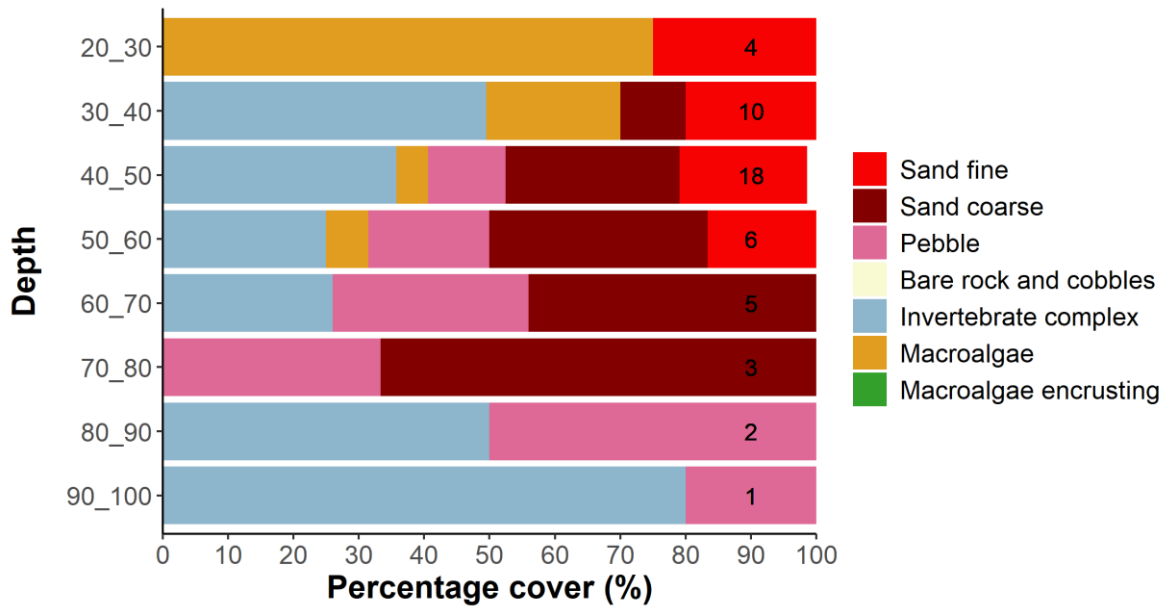


Figure 3.11: Percentage cover of different habitat components at different depths calculated from the field of view of each BRUVS deployment for the matched sites in (a) 2016 and (b) 2021. The percentage cover of 'unknown' is not shown. The numbers on the chart represent the number of BRUVS in each depth zone.



Figure 3.12. Screen grabs from high-definition BRUV video exhibiting the diversity of habitat and species found in Wilsons Promontory Marine National Park.

3.3.2 Key mobile large species

Key mobile large species in 2021

In total, 5,301 individuals across 88 species were observed via 89 BRUVS in 2021. Teleost fishes were the most diverse group (consisting of 67 species) followed by 13 chondrichthyans, 7 invertebrates, and one marine mammal (Australian Fur Seal (*Arctocephalus pusillus*)). The most abundant species or group was perch (*Caesioperca* spp.) followed by Degen's Leatherjacket (*Thamnaconus degeni*) and Yellowtail Scad *Trachurus novaezelandiae* (Figure 3.13).

Draughtboard Shark (*Cephaloscyllium laticeps*) was the most widespread species, occurring in 65 out of 89 deployments. *Caesioperca* spp. and Blue-Throat Wrasse (*Notolabrus tetricus*) were the next most widespread species, occurring in 60 and 56 deployments, respectively. Smooth Stingray (*Bathytoshia brevicaudata*) had the highest biomass, followed by Gummy Shark (*Mustelus antarcticus*) and Broadnose Sevengill Shark (*Notorynchus cepedianus*) (Figure 3.13).

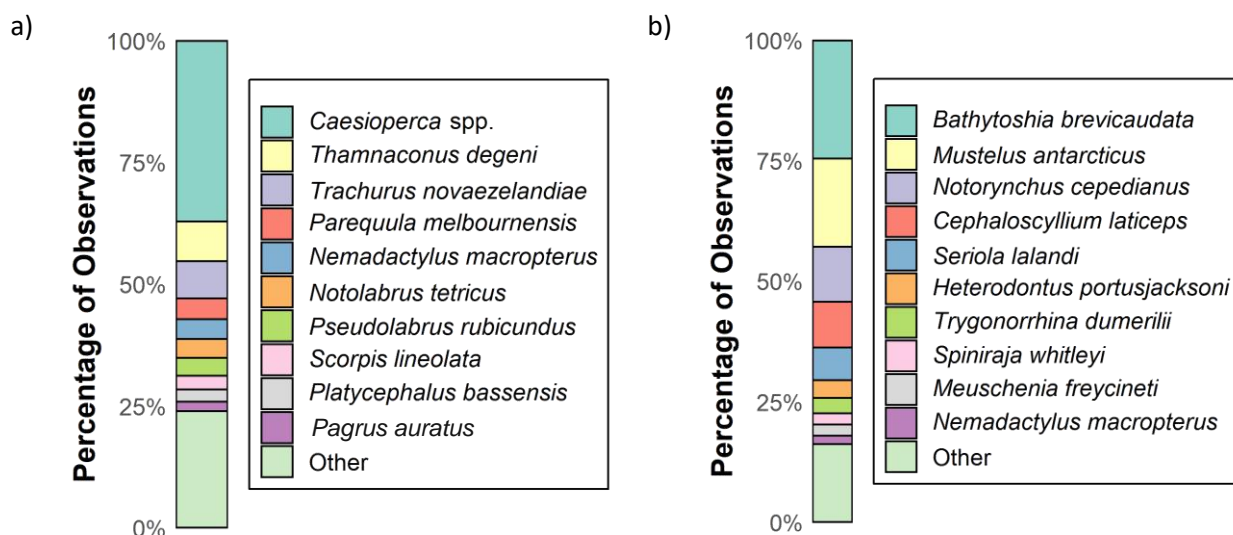


Figure 3.13: Proportions (as percentages) of top species for (a) abundance and (b) total fish lengths (proxy for biomass) across the complete fish assemblage in 2021. 'Other' represents families, genera or species whose proportion of the total observations is less than 2%

Three species of conservation significance were observed from the BRUVS deployments: Eastern Blue Groper (*Achoerodus viridis*) (IUCN Near threatened), Australian Fur Seal (listed under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act)) and Melbourne Skate (*Spiniraja whitleyi*) (IUCN Vulnerable). Eastern Blue Groper was observed on only 2 occasions, both inside the park. Australian Fur Seal was observed inside the park twice and outside the park on 3 occasions. Melbourne Skate was observed once inside the park boundaries and 3 times outside.

Key mobile large species in 2016

In total, 3,960 individuals across 77 species were observed via 62 BRUVS in 2016. Slightly more BRUVS deployments were included in this dataset than in the 2016 report due to the inclusion of deployments that flipped after the initial 30 mins or were partially obscured but still readable. Teleost fishes were the most diverse group (60 species) followed by 11 chondrichthyans and 5 invertebrates. The most abundant species or group was perch (*Caesioperca* spp.) followed by the

Yellowtail Scad (*T. novaezelandiae*) and Rosy Wrasse (*Pseudolabrus rubicundus*) (Figure 3.14). Draughtboard Shark (*C. laticeps*) was the most widespread species, occurring in 49 out of 62 deployments. Silverbelly (*Parequula melbournensis*) and *Caesioperca* spp. were the next most widespread species or group, occurring in 38 and 36 deployments, respectively. Gummy Shark (*M. antarcticus*) had the highest biomass, followed by *C. laticeps* and Port Jackson Shark (*Heterodontus portusjacksoni*) (Figure 3.14).

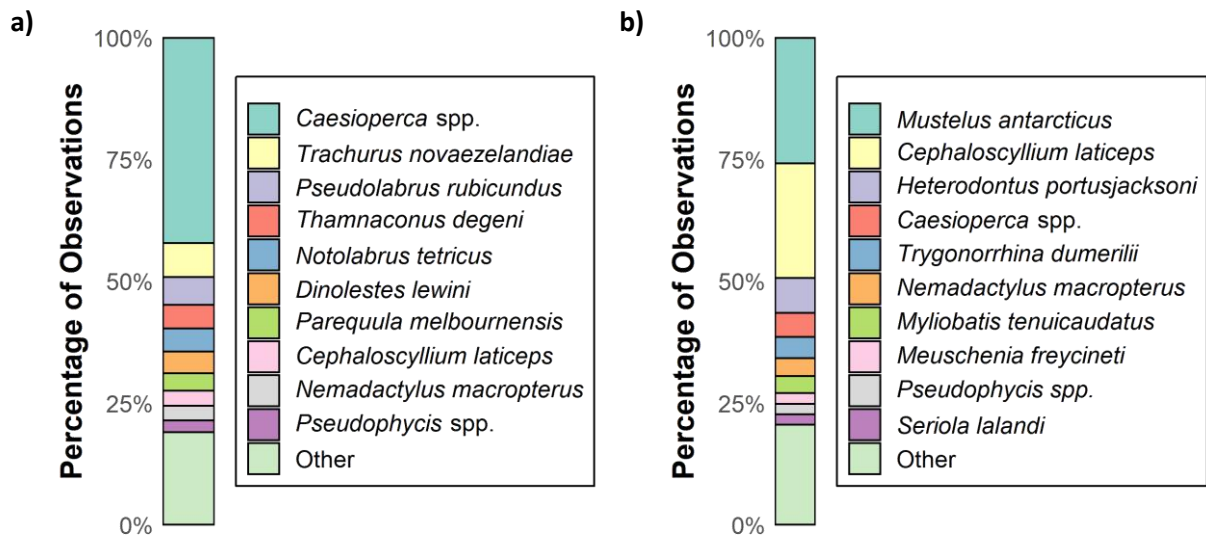


Figure 3.14: The top 10 species by number of observations for (a) abundance and (b) total fish lengths (proxy for biomass) measured for the fish assemblages observed via BRUVS in 2016 at all sites. ‘Other’ represents families, genera or species whose proportion of the total observations is less than 2%

Two species of conservation significance were observed from the BRUVS deployments: Eastern Blue Grouper (*A. viridis*) (IUCN Near threatened) and Melbourne Skate (*S. whitleyi*) (IUCN Vulnerable). Each species was sighted twice within the park; note that sampling in 2016 took place only within the park.

3.3.3 Comparison of species abundance and biomass between 2016 and 2021

Forty-nine BRUVS deployments matched geographically within 100 m between 2016 and 2021. From these 98 deployments, 6,483 individuals were observed across 99 species. A similar number of individuals was observed in each year sampled (3,327 in 2016 and 3,156 in 2021), resulting in an average number of individuals observed per deployment of 67.9 in 2016 and 64.4 in 2021 (Table 3.3). Species richness was equal in both years at 12.1 species on average per deployment. The numbers of species observed per deployment in each habitat type were also very similar between years (Table 3.3).

Table 3.3: Summary of results from BRUVS observations at the matched sites in 2016 and 2021 (n = 49)

Variable	Year		Total
	2016	2021	
Number of individuals observed	3,327	3,156	6,483
Mean number of individuals per deployment	67.9	64.4	66.1
Mean number of species per deployment	12.1	12.1	12.1
Mean number of species observed per deployment in:			
Circalittoral reef	16	17	33
Infralittoral reef	7	6	13
Soft sediment	26	26	52

The fish assemblages observed differed significantly between years and between habitat types within and between years (Table 3.4), Figure 3.15). Assemblages within all habitat types significantly differed from one another within years (Table 3.4). Assemblages in circalittoral and soft-sediment habitats differed significantly between years, but assemblages in infralittoral habitats did not (Table 3.4).

Figure 3.15 shows that assemblages cluster into low-complexity soft-sediment habitats distinct from higher complexity reef habitats with hard substrate; it also shows distinct clustering by year. This clustering is further confirmed by a significant *P* value for the CAP analysis (*P* = 0.001) and a moderate level of allocation success (61% of sites correctly allocated to the right group). One species typical of soft-sediment areas, Sand Flathead (*Platycephalus bassensis*), was strongly correlated with that habitat type, while species typical of hard-bottomed areas were more closely correlated with circalittoral and infralittoral reef habitats, including Blue-Throat Wrasse (*Notolabrus tetricus*) and Six-Spine Leatherjacket (*Meuschenia freycineti*) (Figure 3.15).

The species influencing the differences between years were analysed using SIMPER. The 10 most influential species included *Cephaloscyllium laticeps*, *M. freycineti* and *N. tetricus*. One of the top 10 species, Common Gurnard Perch (*Neosebastes scorpaenoides*), was more abundant in 2021 than in 2016. Four species had similar abundances in the 2 years, and the remaining 5 species – *C. laticeps*, *N. tetricus*, Velvet Leatherjacket (*Meuschenia scaber*), *Caesioperca* spp. and Rosy Wrasse (*Pseudolabrus rubicundus*) – were more abundant in 2016 (Table 3.5). Analysis of the top 5 species influencing differences between years for the soft-sediment habitat showed that *C. laticeps* and *M. scaber* were more abundant in 2016 than in 2021, while *N. scorpaenoides*, *Parequula melbournensis* and *Platycephalus bassensis* were more abundant in 2021 (Table 3.5). Overall, the dissimilarity in assemblages between years was high at 72% for soft-sediment habitats. For circalittoral reef habitats, the top 5 species influencing differences between years included *M. freycineti*, which was more abundant in 2021 than 2016, and *C. laticeps*, *Caesioperca* spp., *N. tetricus* and *P. rubicundus*, which were more abundant in 2016 (Table 3.5). The dissimilarity

between years for circalittoral reef habitat (58% dissimilar) was not as high as for soft-sediment habitats.

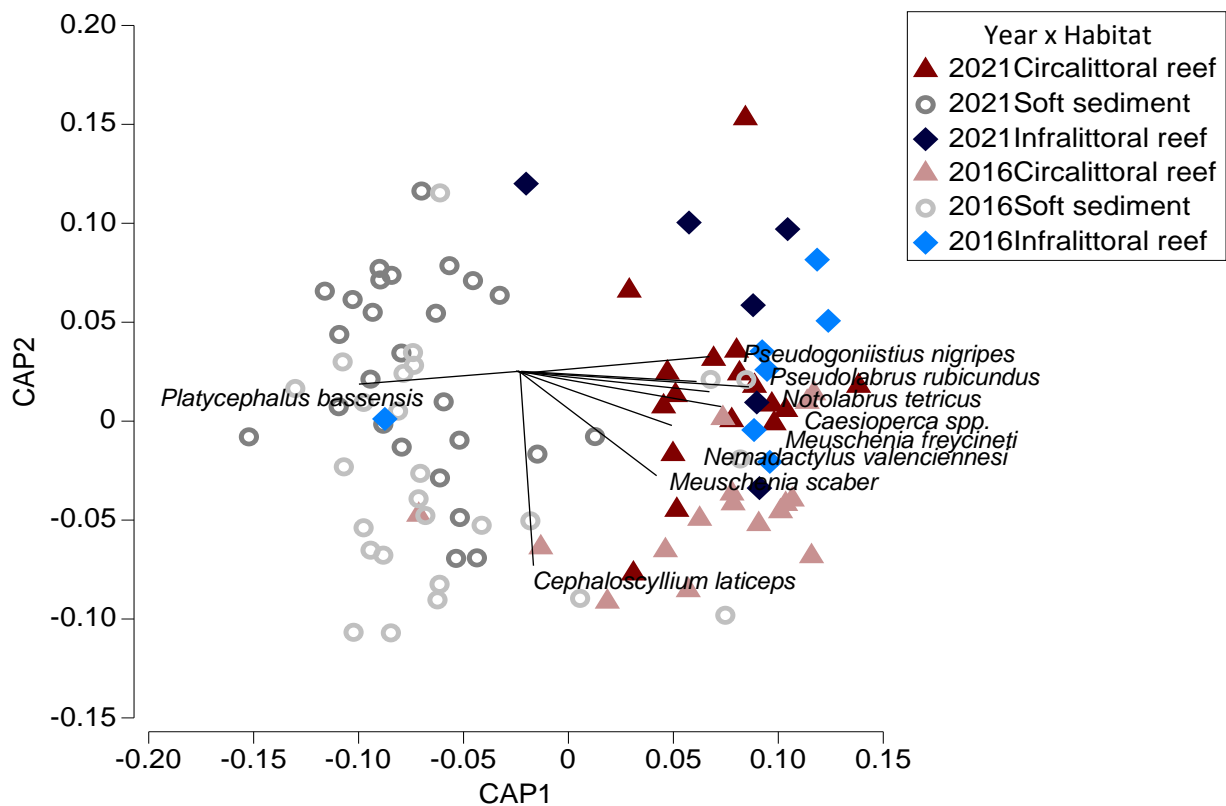


Figure 3.15: CAP ordination plot showing the influence of year and habitat type on the fish assemblages observed via BRUVS for the matched sites in 2016 and 2021. Each point represents the assemblage structure (of all species) of that deployment in relation to the other deployments (similarity); points that are closer together 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

Factor	df	MS	Pseudo-F	P	Unique perms
Year	1	4,694.9	2.10	0.009	999
Habitat	2	28,297	8.13	0.033	339
Year x habitat	2	3,478.9	1.55	0.028	997
Residual	92	2,240.1			

	t	P(MC)	perms	t	P(MC)	perms
	2016			2021		
Circalittoral reef vs. soft sediment	2.87	0.001	998	3.74	0.001	999
Circalittoral reef vs. Infralittoral reef	1.51	0.023	996	1.80	0.001	995
Soft sediment vs. Infralittoral reef	2.18	0.001	998	2.28	0.001	998

2016 vs. 2021						
Circalittoral reef	1.69	0.006	998			
Soft sediments	1.51	0.014	999			
Infralittoral reef	1.1	0.275	758			

Table 3.5: Results from SIMPER analysis of the matched assemblages observed in 2016 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 2016		Average abundance 2021	Average dissimilarity	Diss/SD	% contribution	Cumulative %
2016 vs. 2021							
Average dissimilarity = 74.4							
<i>Cephaloscyllium laticeps</i>	2.16	>	1.41	6.88	1.19	9.24	9.24
<i>Meuschenia freycineti</i>	0.80	~	0.82	3.90	0.99	5.24	14.48
<i>Notolabrus tetricus</i>	1.04	>	0.55	3.77	1.11	5.06	19.55
<i>Neosebastes scorpaenoides</i>	0.52	<	0.91	3.72	0.84	5.00	24.54
<i>Meuschenia scaber</i>	0.80	>	0.69	3.59	1.00	4.82	29.36
<i>Caesioperca</i> spp.	0.71	>	0.56	3.18	0.92	4.28	33.64

<i>Mustelus antarcticus</i>	0.45	~	0.42	2.62	0.89	3.52	37.16
<i>Parequula melbournensis</i>	0.38	~	0.41	2.53	0.58	3.41	40.57
<i>Pseudolabrus rubicundus</i>	0.63	>	0.36	2.53	0.80	3.40	43.97
<i>Platycephalus bassensis</i>	0.32	~	0.33	2.41	0.69	3.23	47.20
Soft sediment							
Average dissimilarity = 72.2							
<i>Cephaloscyllium laticeps</i>	2.08	>	1.46	8.74	1.18	12.10	12.10
<i>Neosebastes scorpaenoides</i>	0.53	<	1.12	5.08	1.05	7.04	19.14
<i>Parequula melbournensis</i>	0.55	<	0.70	4.39	0.72	6.08	25.21
<i>Platycephalus bassensis</i>	0.51	<	0.62	4.14	0.97	5.73	30.95
<i>Meuschenia scaber</i>	0.59	>	0.37	3.75	0.78	5.20	36.14
Circalittoral reef							
Average dissimilarity = 58.0							
<i>Cephaloscyllium laticeps</i>	2.88	>	1.29	5.86	1.57	10.11	10.11
<i>Caesioperca spp.</i>	1.61	>	1.34	3.78	1.29	6.52	16.62
<i>Notolabrus tetricus</i>	1.91	>	1.07	3.41	1.29	5.87	22.49
<i>Meuschenia freycineti</i>	1.61	<	1.74	3.31	1.04	5.71	28.20
<i>Pseudolabrus rubicundus</i>	1.22	>	0.50	2.56	1.21	4.40	32.60

3.3.4 Environmental drivers of fish assemblage

Environmental drivers in 2021

Various environmental drivers influenced the fish assemblages observed using BRUVS in 2021. For all sites sampled, there was a significant effect of protection status (i.e. inside the park (no-take) vs. outside the park (fished)) and habitat type (Table 3.6), but not their interaction. The top 10 species driving the differences in the assemblage between fished and no-take areas included 5 with higher abundances in the fished areas – Blue-Throat Wrasse (*Notolabrus tetricus*), morid cod (*Pseudophycis* spp.), Six-Spine Leatherjacket (*Meuschenia freycineti*), Maori Wrasse (*Ophthalmolepis lineolata*) and Magpie Perch (*Pseudogoniistius nigripes*) – and 5 showing the opposite trend – Draughtboard Shark (*Cephaloscyllium laticeps*), perch (*Caesioperca* spp.), Velvet Leatherjacket (*Meuschenia scaber*), Sand

Flathead (*Platycephalus bassensis*) and Common Gurnard Perch (*Neosebastes scorpaenoides*) (Table 3.7).

All habitat types showed significant differences for the assemblages observed between pairs based on PERMANOVA analysis (Table 3.6). CAP analysis (Figure 3.16) also showed significant differences ($P = 0.001$) and had a moderate allocation success rate of 70%, with 3 distinct clusters observable for circalittoral reef, infralittoral reef and soft sediment. The fine-scale environmental variables driving these differences were investigated using DISTLM analysis, which included 4 variables in the top model (depth and percentage cover of fine sand, invertebrate complex and macroalgae) (Figure 3.17). The top model explained 36% of the variation within the fish assemblage, with the first 2 axes of the dbRDA plot explaining 25% and 7% of the total variation, respectively. Species correlated with the variables selected in the top model included *P. bassensis*, which increased in abundance as the percentage cover of fine sand increased; *N. scorpaenoides* and Gummy Shark (*Mustelus antarcticus*), which increased in abundance as the depth increased; *M. freycineti*, which increased as the percentage cover of invertebrates increased; and *N. tetricus*, which increased as the percentage cover of macroalgae increased (Figure 3.17).

No differences were observed in 2021 between fished and no-take areas for species richness, total abundance and total biomass alone ($P > 0.17$). Species richness and total abundance were influenced by habitat type, while total biomass was not (Figure 3.18). Soft-sediment habitats had significantly lower species richness than both circalittoral and infralittoral reef (Figure 3.18a). Circalittoral reefs had significantly higher total abundance than both infralittoral and soft-sediment habitats (Figure 3.18c).

Table 3.6: Analyses 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

Factor	df	MS	Pseudo-F	P	Unique perms
Status	1	4,572.3	2.09	0.011	999
Habitat	2	16,409	7.51	0.001	999
Status x habitat	2	2,064.4	0.94	0.588	996
Residual	83	2,185.8			
		<i>t</i>	<i>P</i>	perms	
Circalittoral reef vs. soft sediment		2.60	0.001	999	
Circalittoral reef vs. infralittoral reef		1.64	0.005	999	
Soft sediment vs. infralittoral reef		3.28	0.001	998	

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>	0.95	<	1.87	5.60	1.40	7.25	7.25
<i>Cephaloscyllium laticeps</i>	1.34	>	0.63	3.90	0.93	5.05	12.30
<i>Pseudophycis</i> spp.	0.68	<	1.05	3.90	1.08	5.05	17.35
<i>Meuschenia freycineti</i>	0.73	<	1.15	3.74	1.21	4.84	22.20
<i>Caesioperca</i> spp.	0.86	>	0.77	3.50	1.24	4.53	26.72
<i>Meuschenia scaber</i>	0.84	>	0.52	3.28	0.90	4.25	30.97
<i>Ophthalmolepis lineolata</i>	0.21	<	0.79	2.88	1.22	3.72	34.70
<i>Platycephalus bassensis</i>	0.57	>	0.22	2.78	0.64	3.61	38.30
<i>Pseudogoniistius nigripes</i>	0.22	<	0.79	2.66	1.10	3.44	41.74
<i>Neosebastes scorpaenoides</i>	0.68	>	0.07	2.64	0.84	3.42	45.16

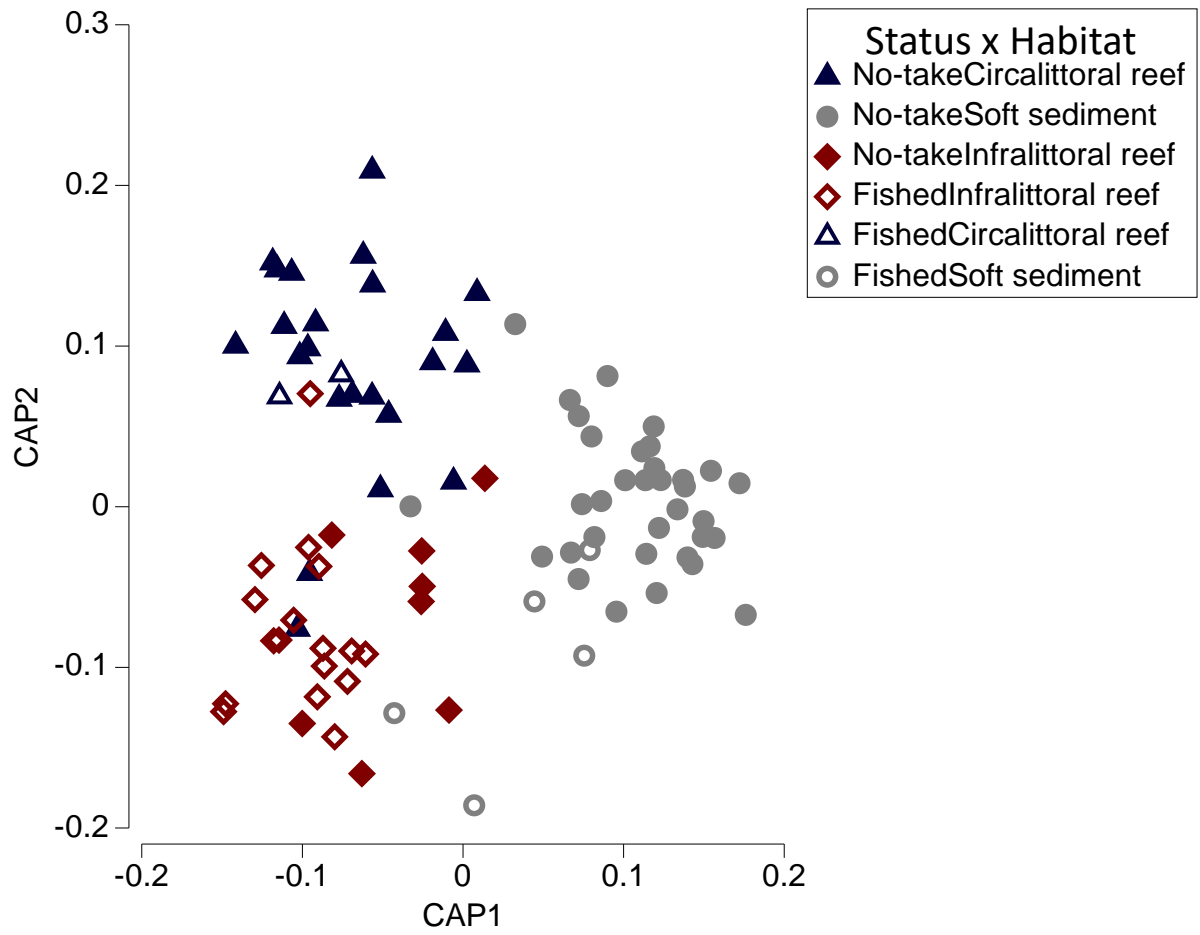


Figure 3.16: CAP ordination plot showing the influence of habitat type and status (fished vs. no-take) on the fish assemblage observed via BRUVS in 2021. Each point represents the assemblage structure (of all species) of that deployment in relation to the other deployments (similarity); points that are closer together are more similar than those further apart

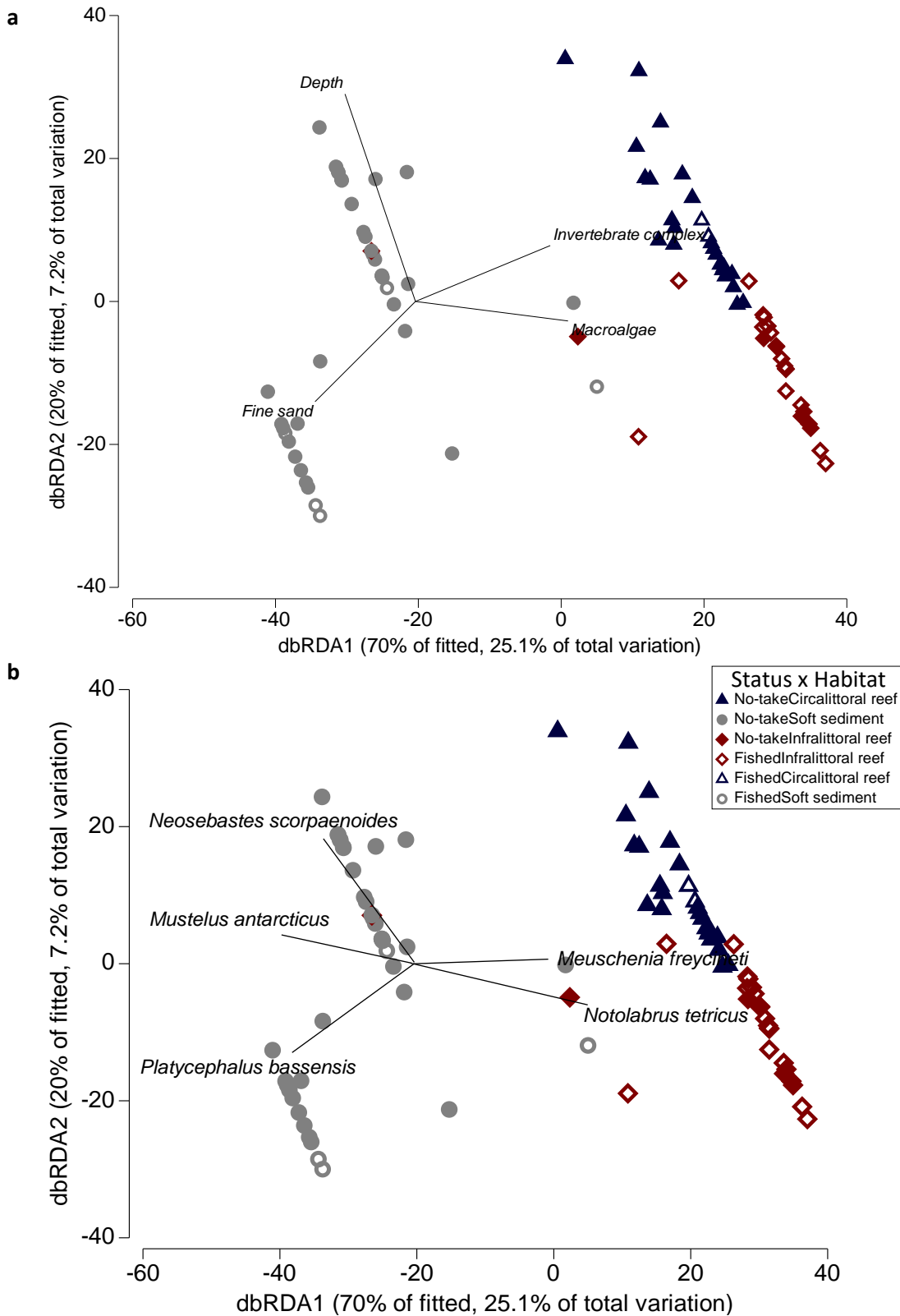


Figure 3.17: Distance-based redundancy analysis plots showing the influence of environmental variables (depth and percentage cover of fine sand, invertebrate complex and macroalgae) based on the top model output from DISTLM for the 2021 assemblage. Species with correlations >0.6 with the dbRDA axes are shown in (b)

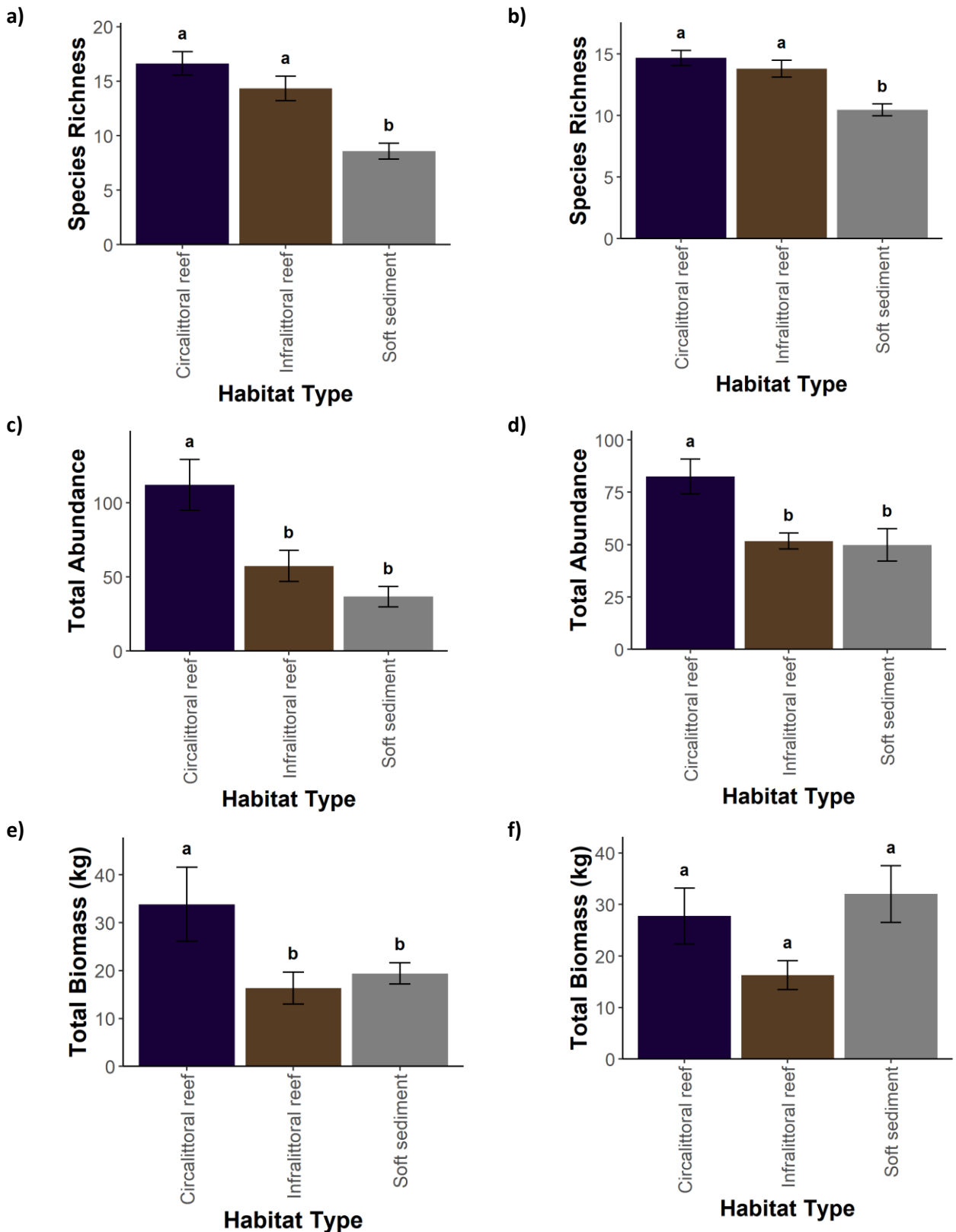


Figure 3.18: Average (\pm SE) (a) species richness, (b) total abundance and (c) total biomass (kg) for habitat type in 2016 (left) and 2021 (right). Letters denote significant differences between pairs according to PERMANOVA analyses (i.e. bars with the same letter within the same graph are not significantly different)

Environmental drivers in 2016

Habitat type had a significant effect on the assemblages observed via BRUVS in 2016 based on PERMANOVA analysis (Pseudo- $F = 7.72$, $P = 0.001$). All habitat types were significantly different from one another ($P < 0.002$). Distinct clustering by habitat type can also be observed in the CAP analysis (Figure 3.19). CAP also showed significant differences between habitat types ($P = 0.001$) and had a high allocation success rate of 76%. DISTLM analysis showed the variables driving the differences observed in the assemblage, with 3 variables included in the top model (percentage cover of invertebrate complex, mean relief and depth) (Figure 3.20). Common reef-associated species, such as Six-Spine Leatherjacket (*Meuschenia freycineti*) and Blue-Throat Wrasse (*Notolabrus tetricus*), were correlated with increased relief and cover of invertebrates (Figure 3.20). Sand Flathead (*Platycephalus bassensis*) showed the opposite trend. Draughtboard Shark (*Cephaloscyllium laticeps*) and Velvet Leatherjacket (*Meuschenia scaber*) were associated with increased depth (Figure 3.20).

Species richness, total abundance and total biomass were also influenced by habitat type (Figure 3.18). Soft-sediment habitats had significantly lower species richness than the other habitat types, and circalittoral reef habitats had significantly higher total abundance and total biomass than infralittoral reef and soft-sediment habitats (Figure 3.18).

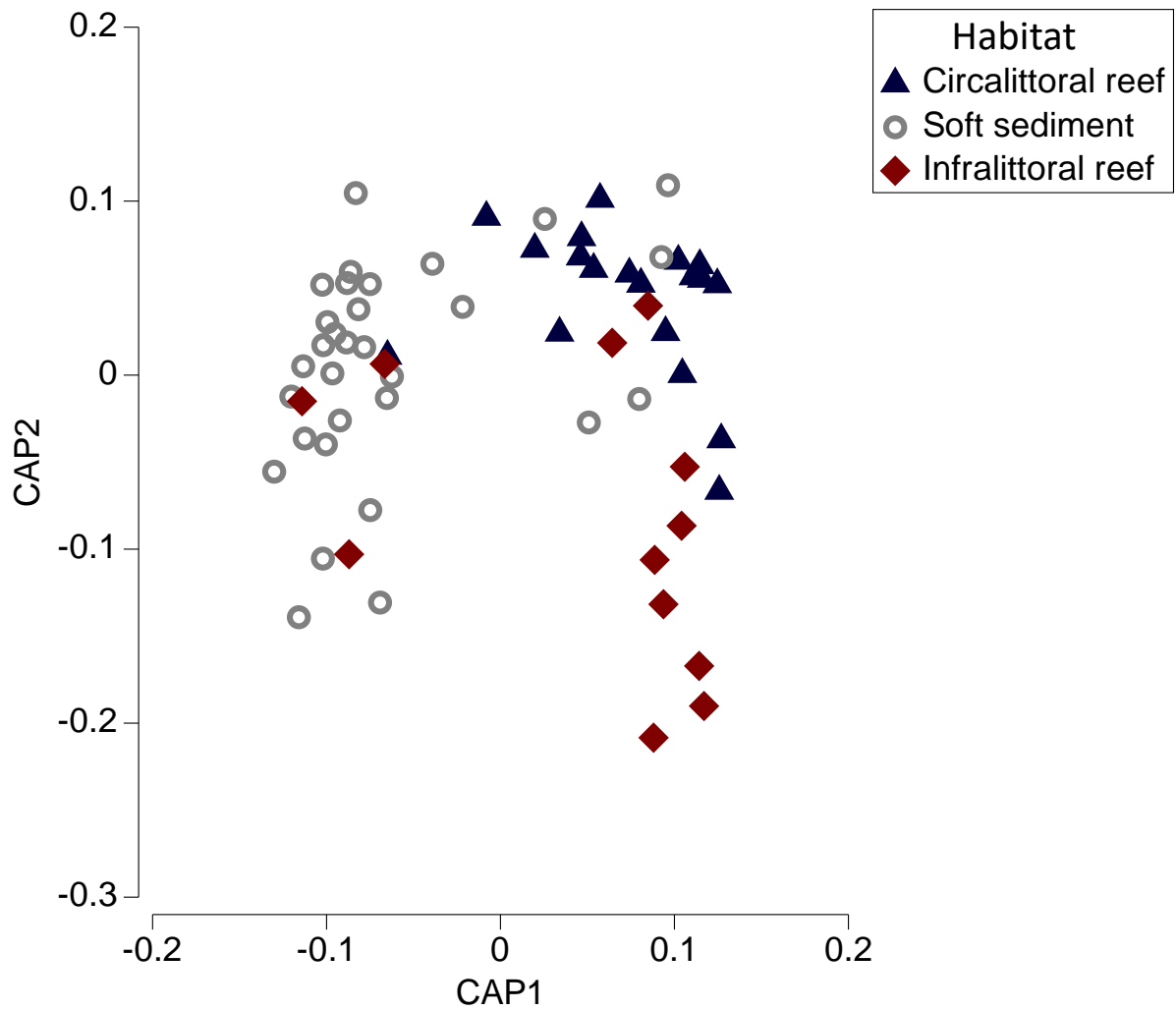


Figure 3.19: CAP ordination plot showing the influence of habitat type on the fish assemblage observed via BRUVS in 2016. Each point represents the assemblage structure (of all species) of that deployment in relation to the other deployments (similarity); points that are closer together more similar than those further apart

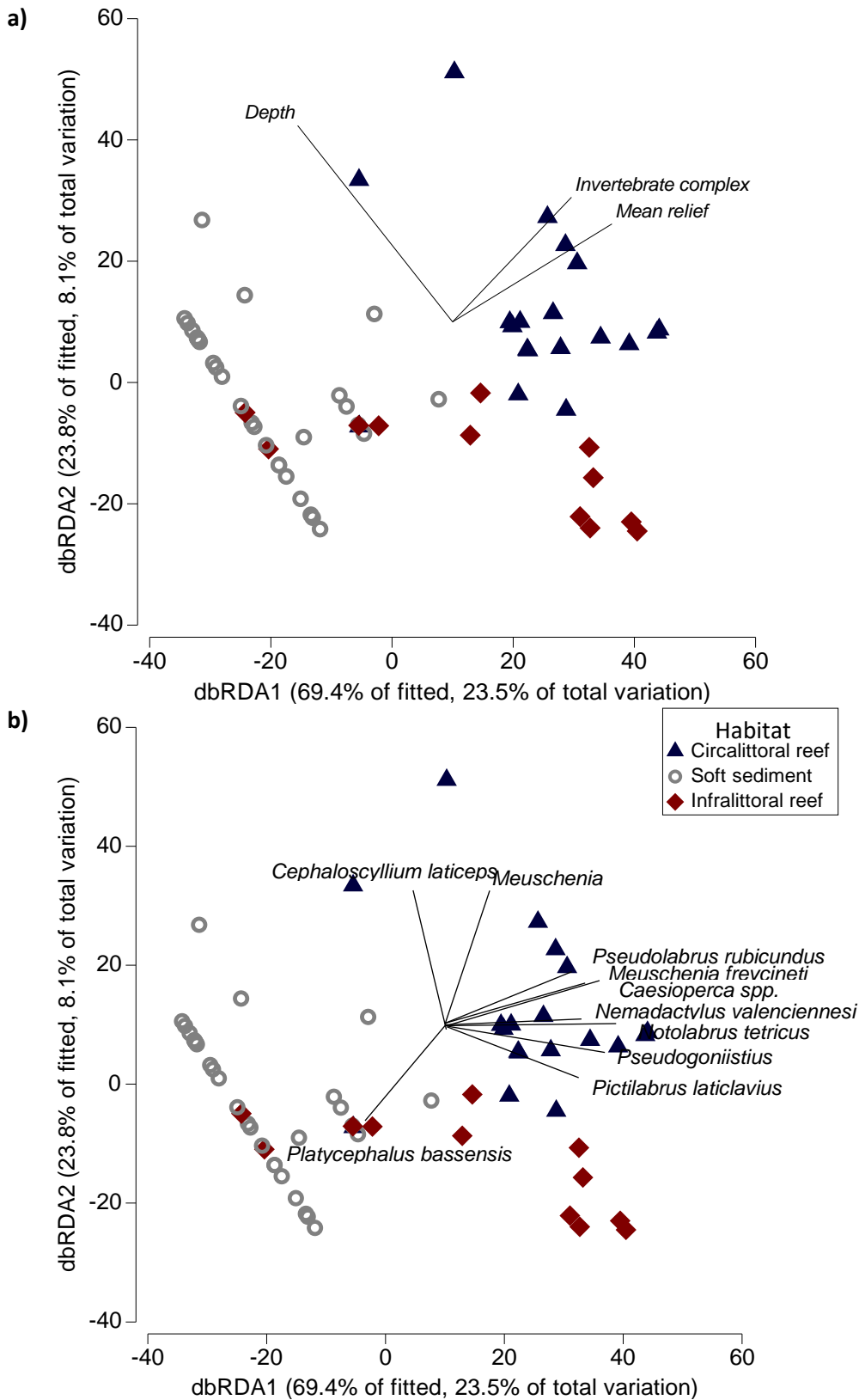


Figure 3.20: Distance-based redundancy analysis plots showing the influence of environmental variables (mean relief, depth, percentage cover of invertebrate complex) based on the top model output from DISTLM for the 2016 assemblage. Species with correlations >0.5 with the dbRDA axes are shown in (b)

3.3.5 Performance of species distribution models using BRUVS data

Large variation in model performance was present across community metrics and individual species modelled (Figures 3.21, 3.23, 3.25, 3.27, 3.29, 3.31, 3.34, 3.36, 3.38). Of the community metrics tested, planktivore abundance, carnivore abundance and relative species richness were found to have the highest deviations explained, at 36.8%, 24.3% and 16.9%, respectively. Depth (bathymetry) and distance to reef were found to best explain trends in planktivore abundance, carnivore abundance and relative species richness (Table 3.8). Planktivore abundance, carnivore abundance and relative species richness all increased in shallower depths and in closer proximity to reef. Of the 3 best-performing models, planktivore performed best, explaining 38.8% of deviation. Models of invertivore abundance, total abundance and total biomass performed worse, explaining 9.0%, 4.2% and 13.7%, respectively. Both total abundance and total biomass were found to be best explained by depth (bathymetry). Total abundance was also found to increase as depths became shallower, whereas total biomass increased with increasing depth. These findings were reflected by trends observed in bubble plots (Figures 3.22, 3.24, 3.26, 3.28, 3.30, 3.32, 3.33, 3.35, 3.37, 3.39), where the most evident trend was that highest species richness was over or near reef. A species distribution model was not created for herbivore abundance due to too few samples (10) across too few sites (8).

Table 3.8: Heatmap table showing predictor variables (columns) used in each of the model subsets (rows) 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	Rugosity
Species richness	23	29	21	27
Total abundance	27	25	24	24
Total biomass	28	21	27	24
Planktivore abundance	22	24	27	27
Carnivore abundance	21	43	18	18
Invertivore abundance	26	28	27	19
<i>Cephaloscyllium laticeps</i>	27	14	16	20
<i>Notolabrus tetricus</i>	25	28	26	21
<i>Caesioperca</i> spp.	22	30	23	25

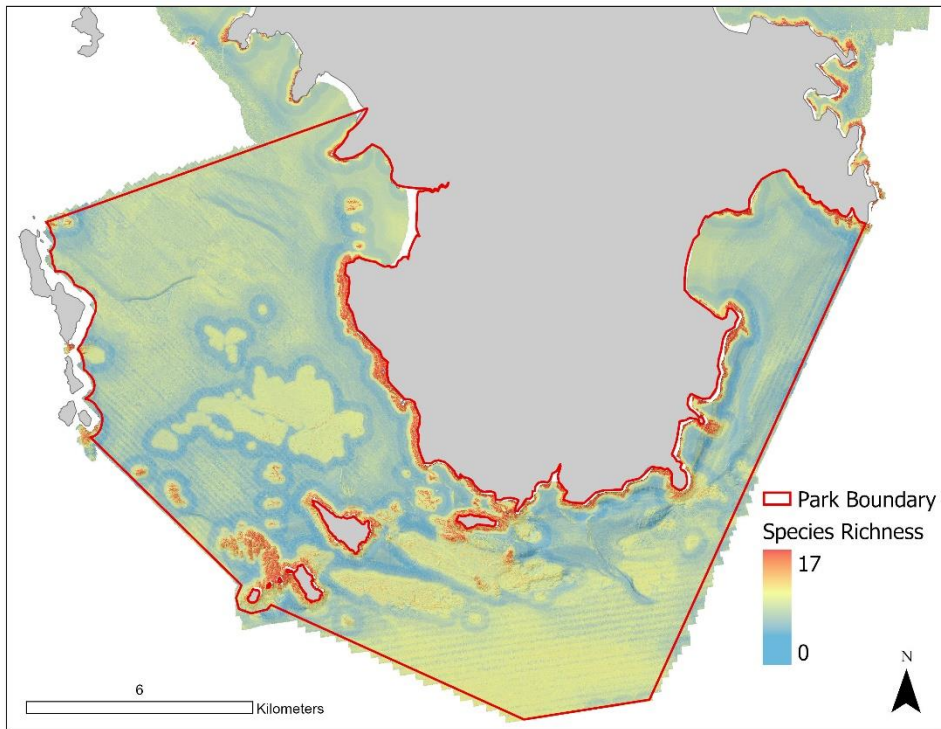


Figure 3.21: Predicted species richness in 2021 from BRUVS sampling across the whole study site. The random forest model explained 16.9% of the variance and had a pseudo- R^2 value of 0.87

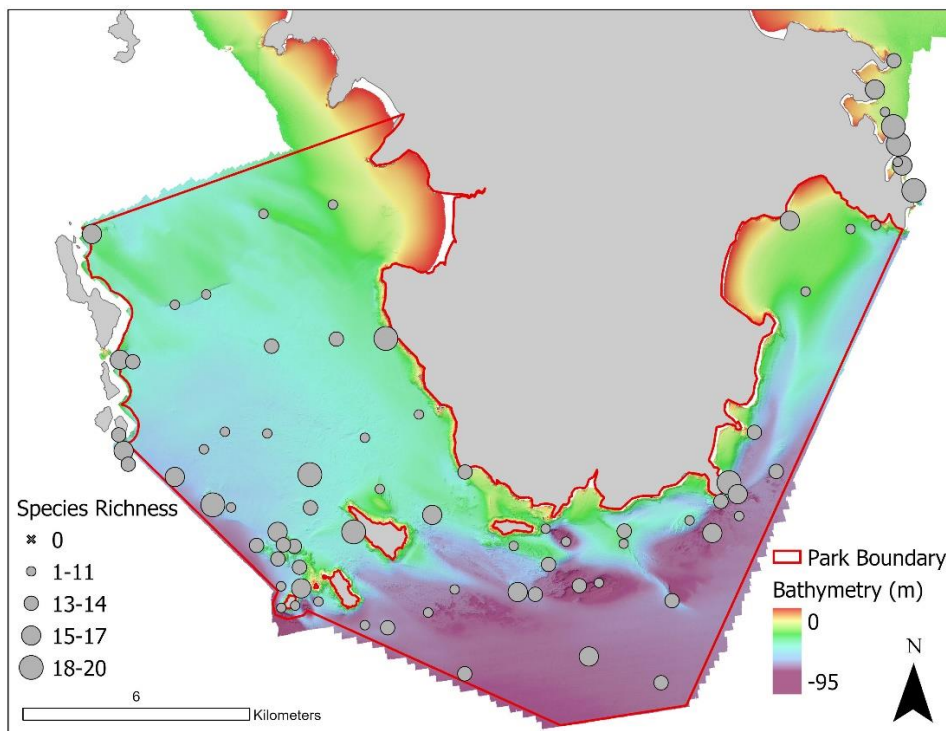


Figure 3.22: Species richness of fish for all BRUVS deployments in 2021. The size of each site marker corresponds to the relative species richness of fish observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

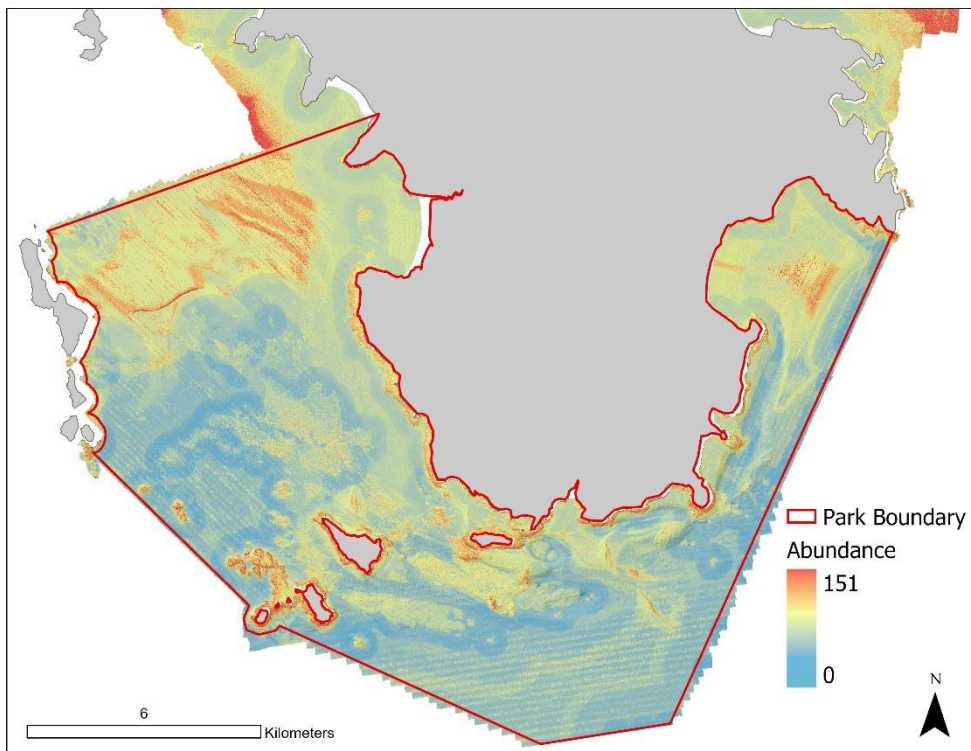


Figure 3.23: Predicted total abundance in 2021 from BRUVS sampling across the whole study site. The random forest model explained 4.2% of the variance and had a pseudo- R^2 value of 0.80

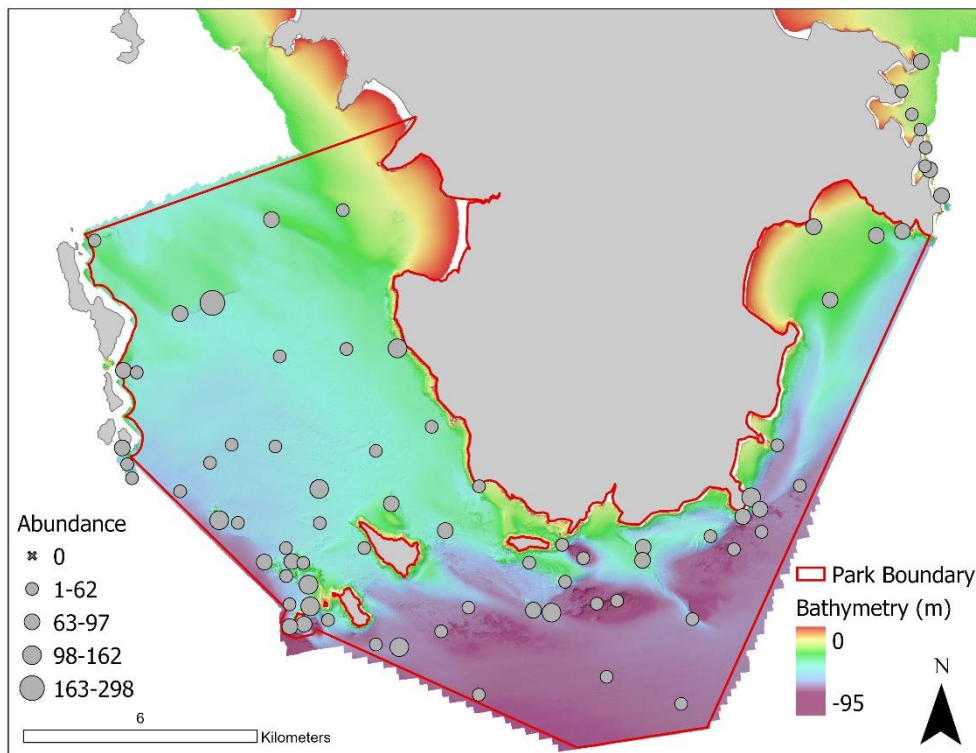


Figure 3.24: Total abundance of fish for all BRUVS deployments in 2021. The size of each site marker corresponds to the total abundance of fish observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

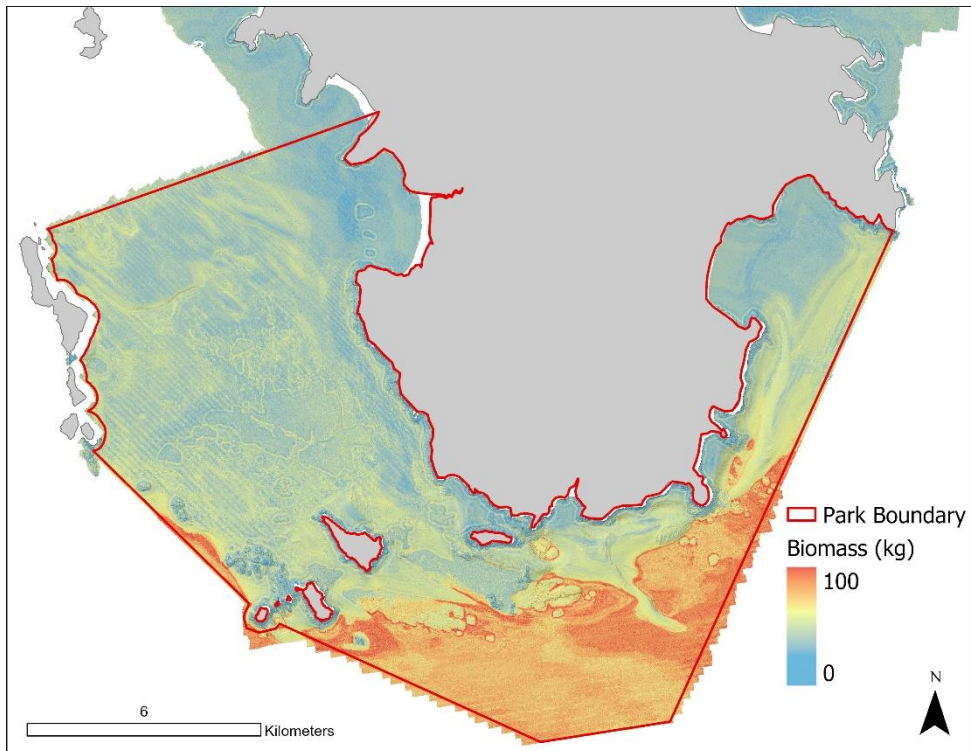


Figure 3.25: Predicted biomass of all species in 2021 from BRUVS sampling across the whole study site. The random forest model explained 13.7% of the variance and had a pseudo- R^2 value of 0.88

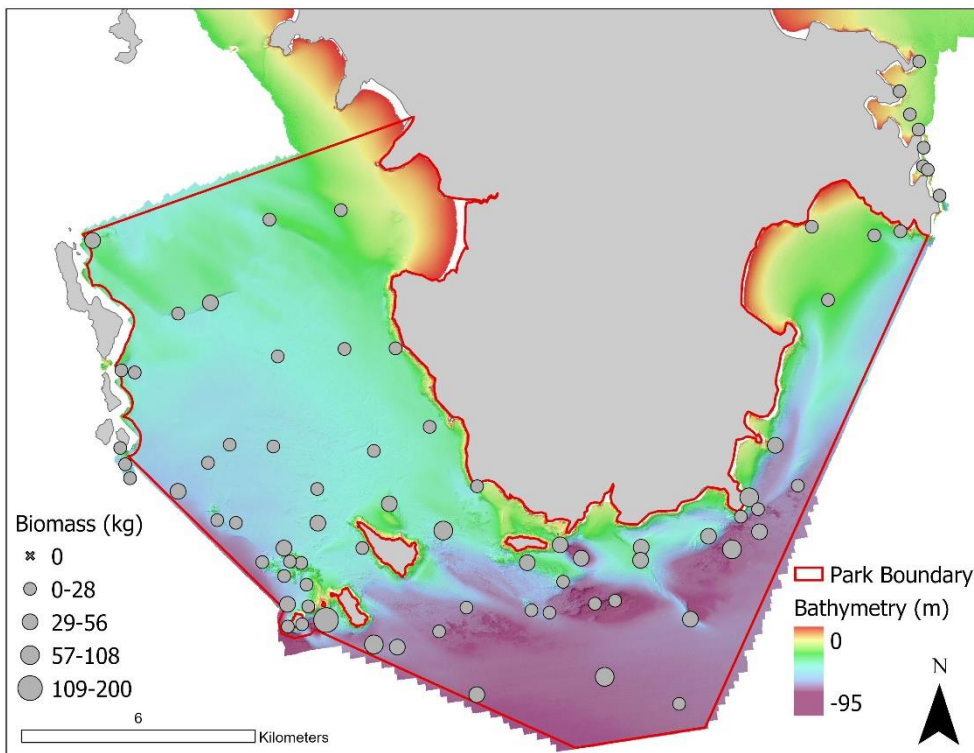


Figure 3.26: Total biomass for all BRUVS deployments in 2021. The size of each site marker corresponds to the total biomass in kilograms of fish observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

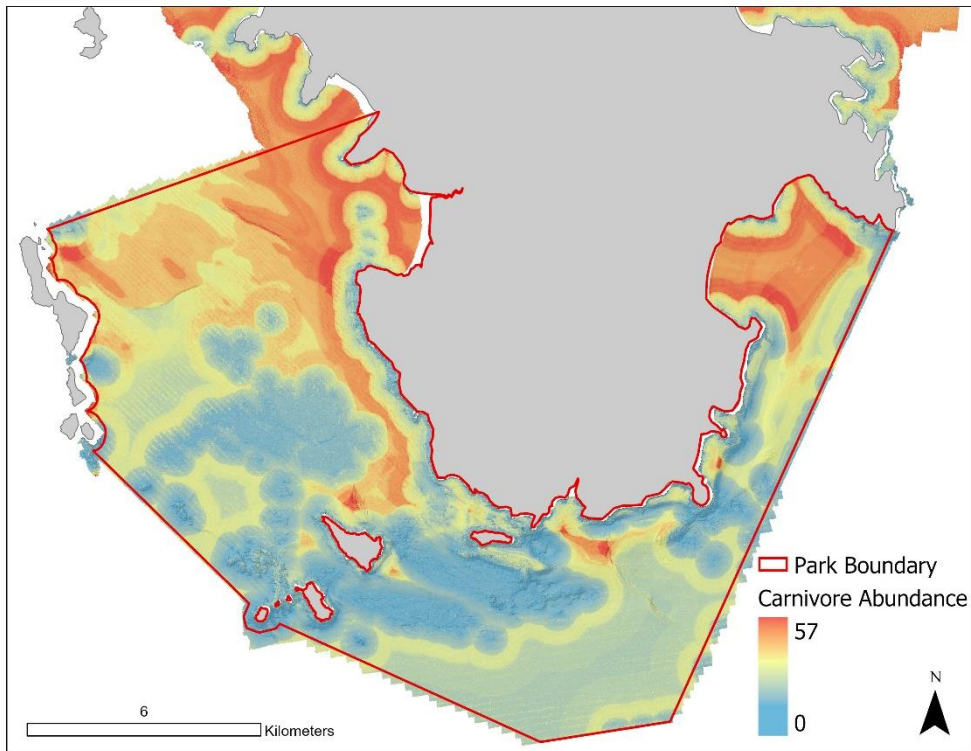


Figure 3.27: Predicted carnivore abundance in 2021 from BRUVS sampling across the whole study site. The random forest model explained 24.3% of the variance and had a pseudo- R^2 value of 0.93

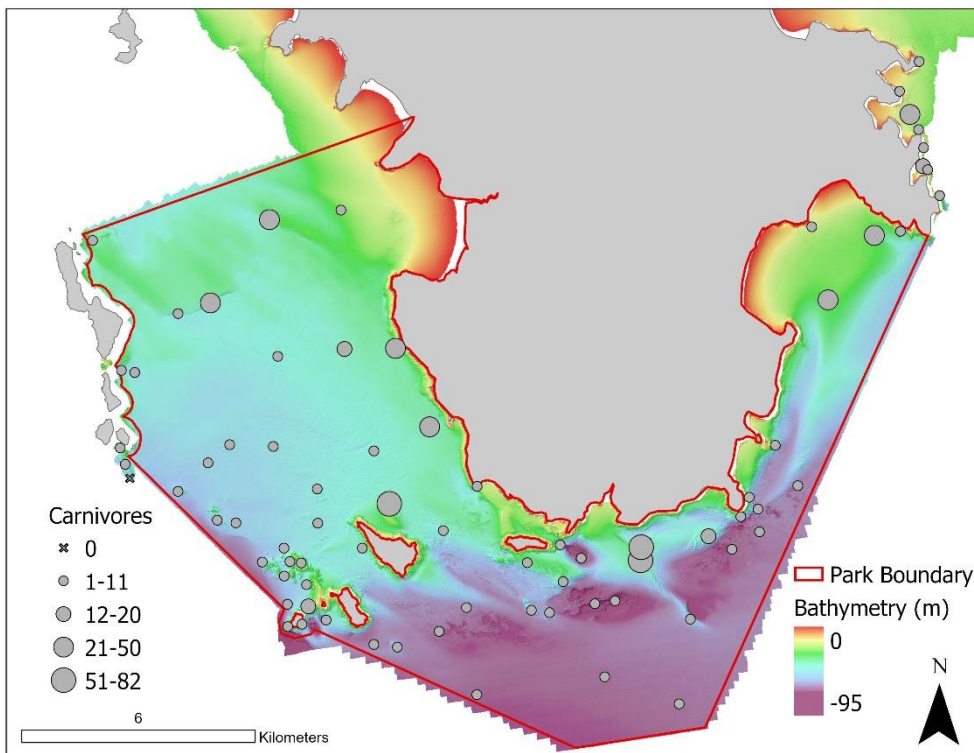


Figure 3.28: Carnivore abundance of fish for all BRUVS deployments in 2021. The size of each site marker corresponds to the total carnivore abundance of fish observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

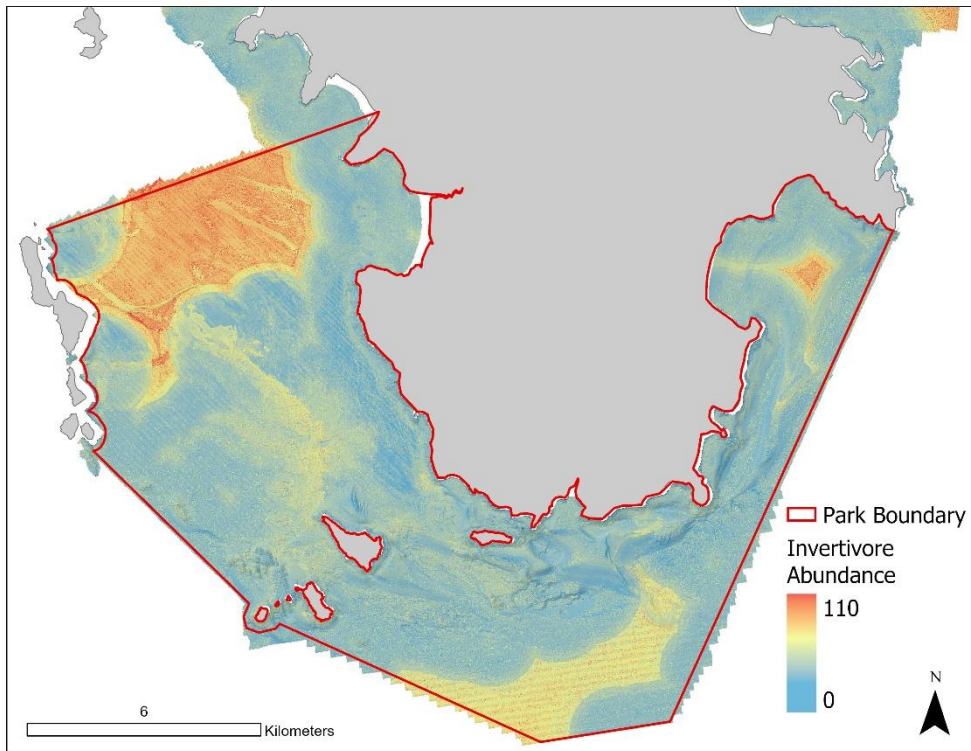


Figure 3.29: Predicted invertivore abundance in 2021 from BRUVS sampling across the whole study site. The random forest model explained 9.0% of the variance and had a pseudo- R^2 value of 0.81

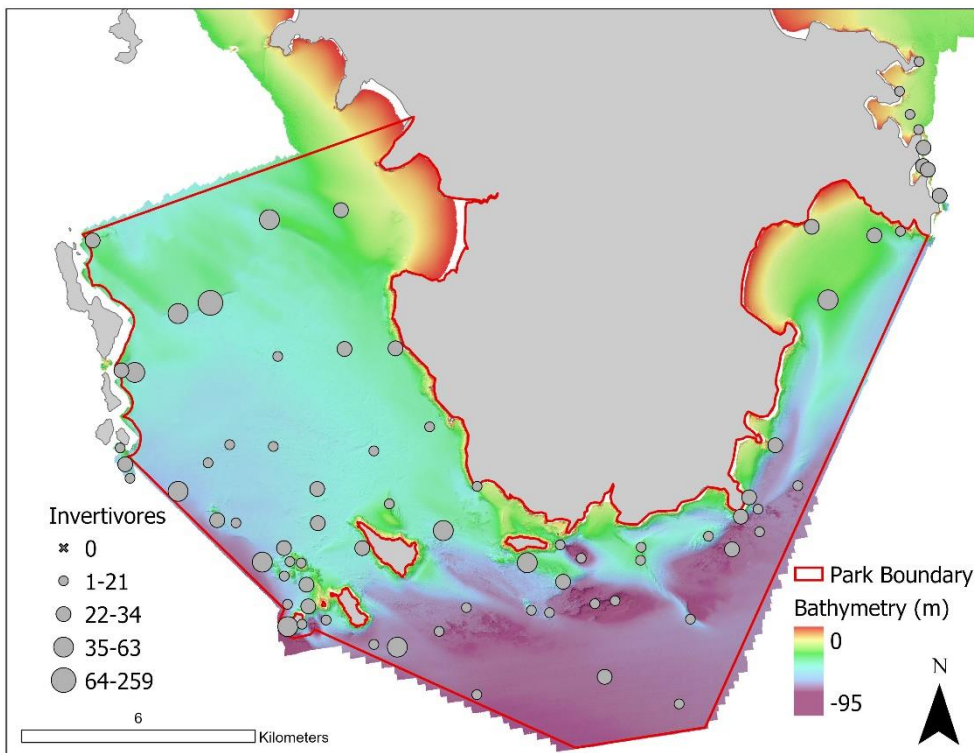


Figure 3.30: Invertivore abundance of fish for all BRUVS deployments in 2021. The size of each site marker corresponds to the invertivore abundance of fish observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

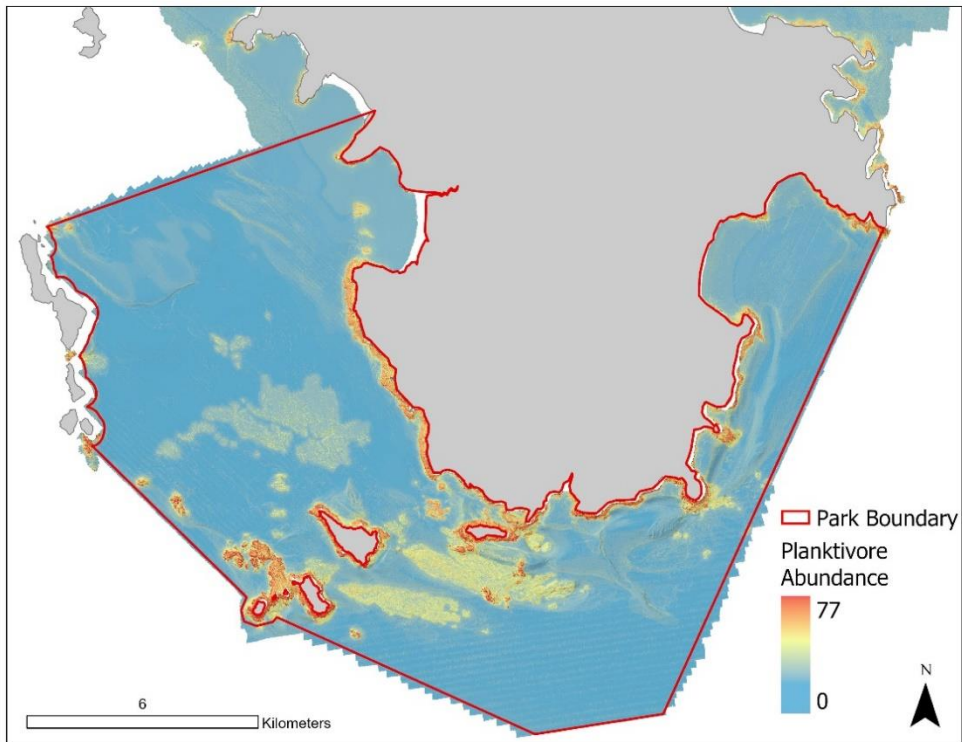


Figure 3.31: Predicted planktivore abundance in 2021 from BRUVS sampling across the whole study site. The random forest model explained 36.8% of the variance and had a pseudo- R^2 value of 0.88

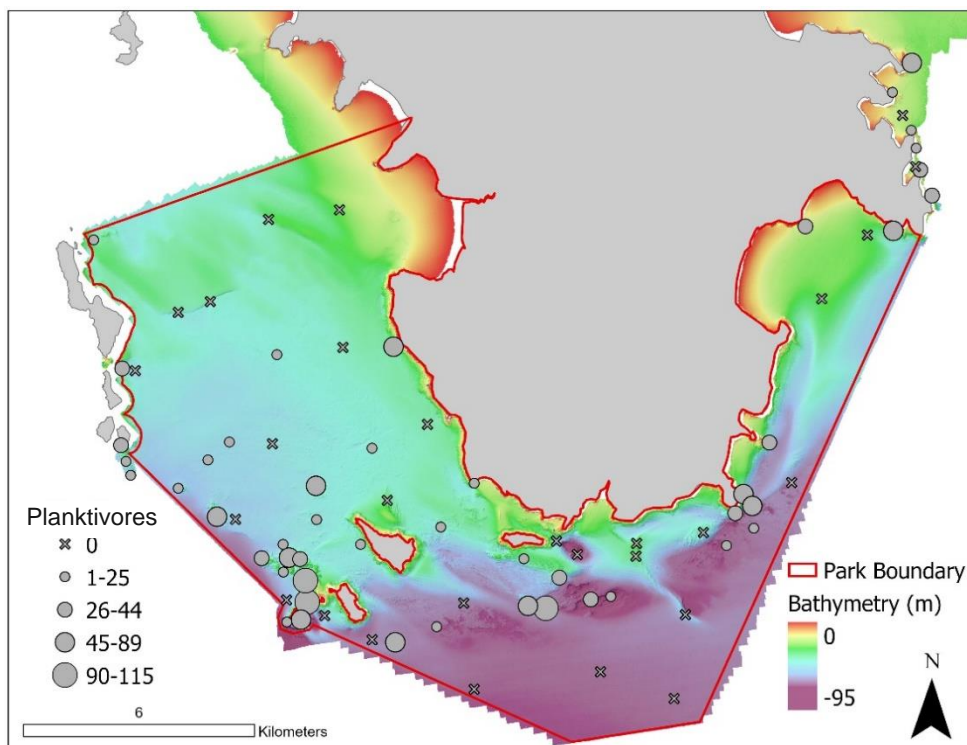


Figure 3.32: Planktivore abundance of fish for all BRUVS deployments in 2021. The size of each site marker corresponds to the planktivore abundance of fish observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

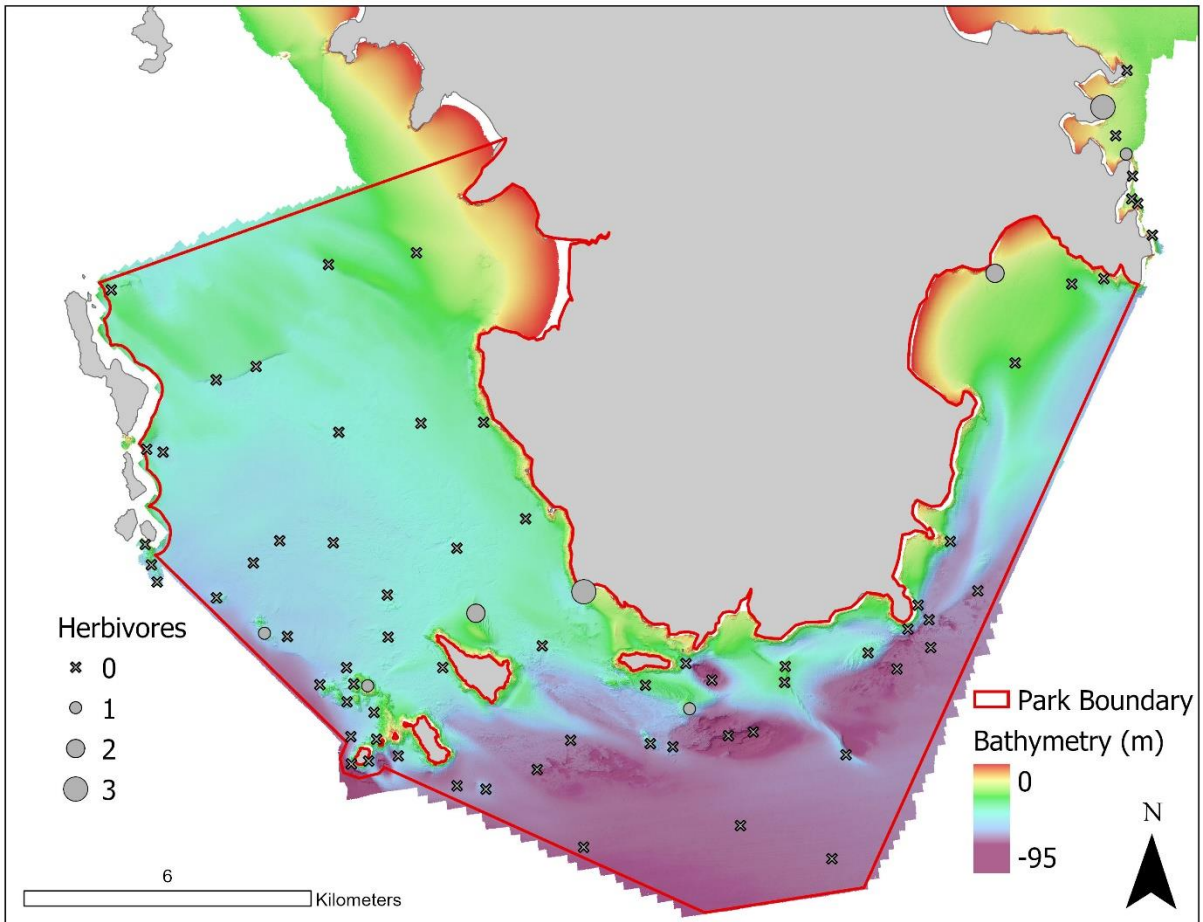


Figure 3.33: Herbivore abundance for all BRUVS deployments in 2021. The size of each site marker corresponds to the herbivore abundance of fish observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

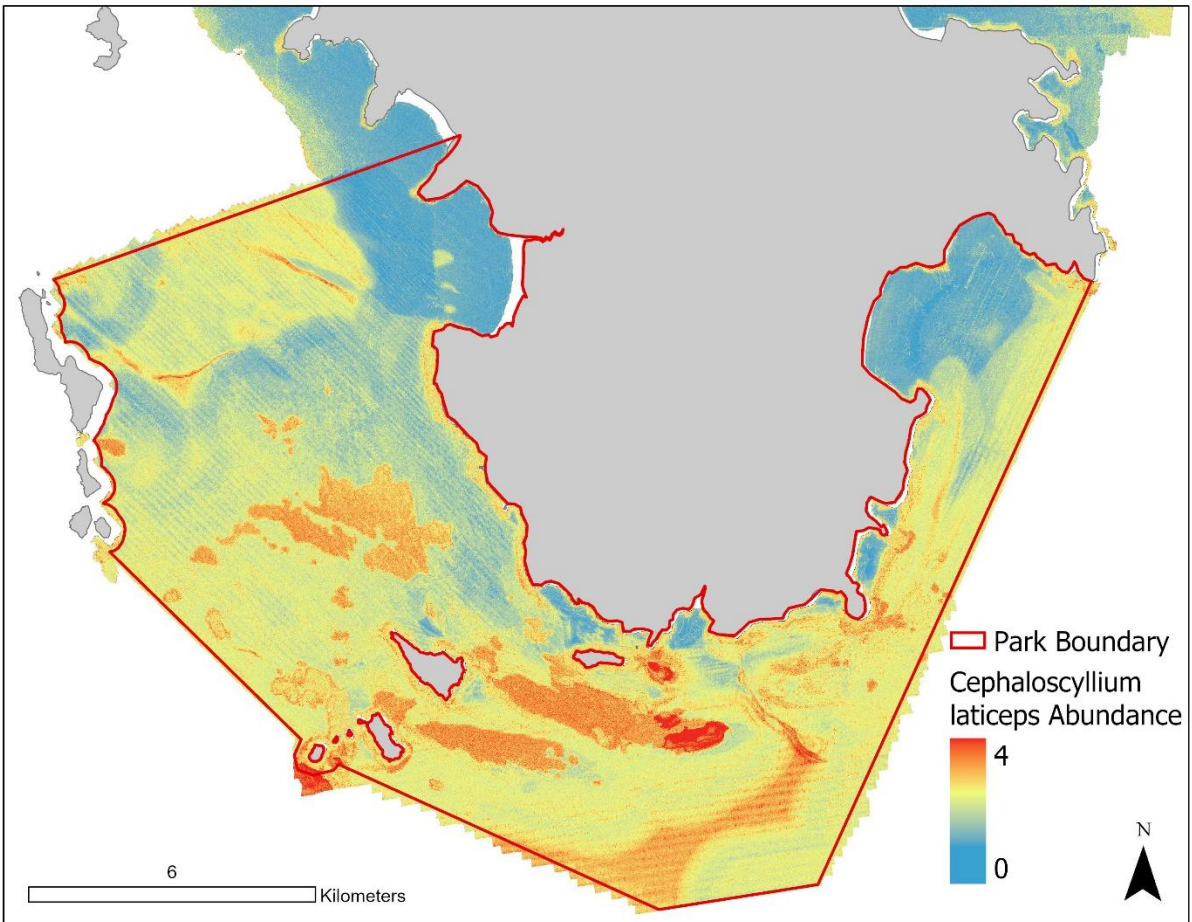


Figure 3.34: Predicted abundance of *Cephaloscyllium laticeps* in 2021 from BRUVS sampling across the whole study site. The random forest model explained 15.3% of the variance and had a pseudo- R^2 value of 0.84

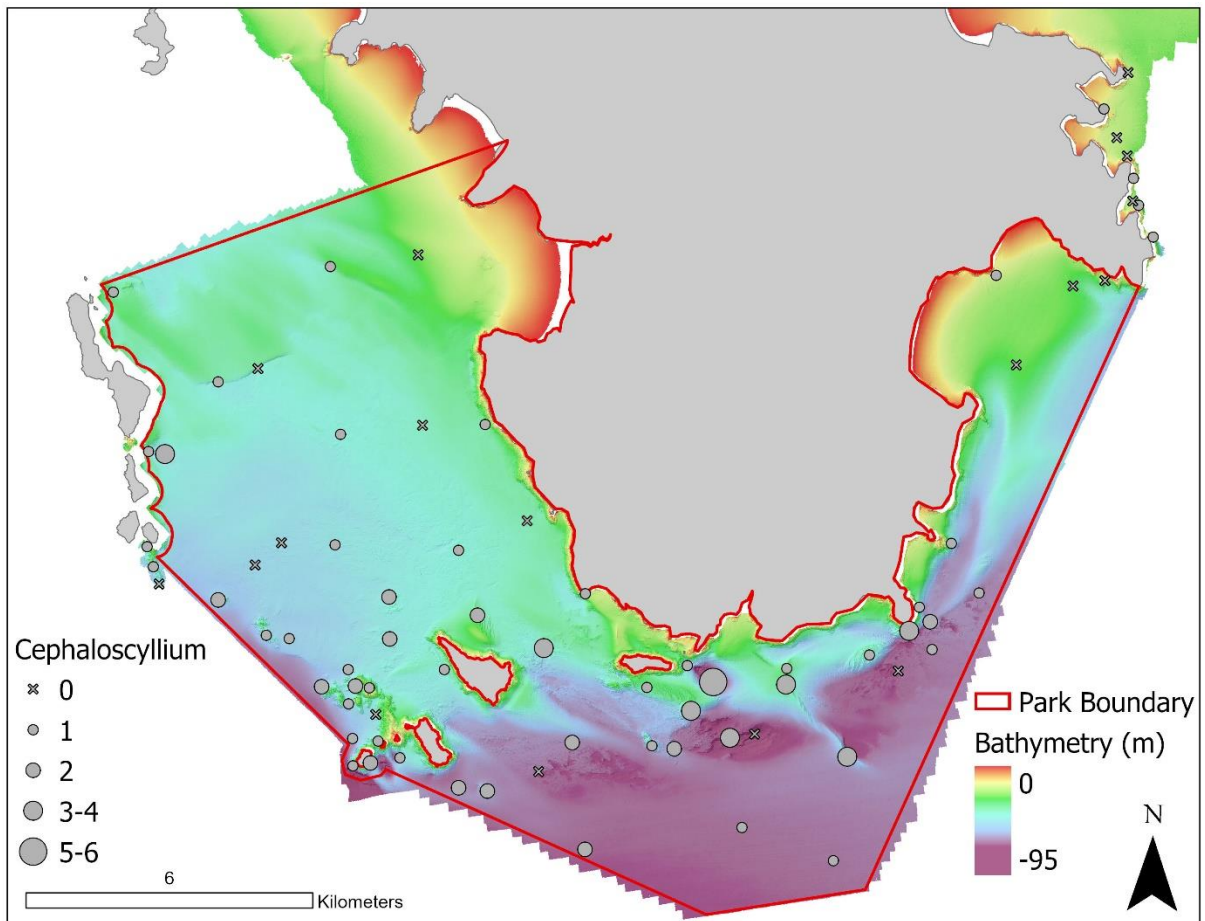


Figure 3.35: Abundance of *Cephaloscyllium laticeps* for all BRUVS deployments in 2021. The size of each site marker corresponds to the abundance of *C. laticeps* observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

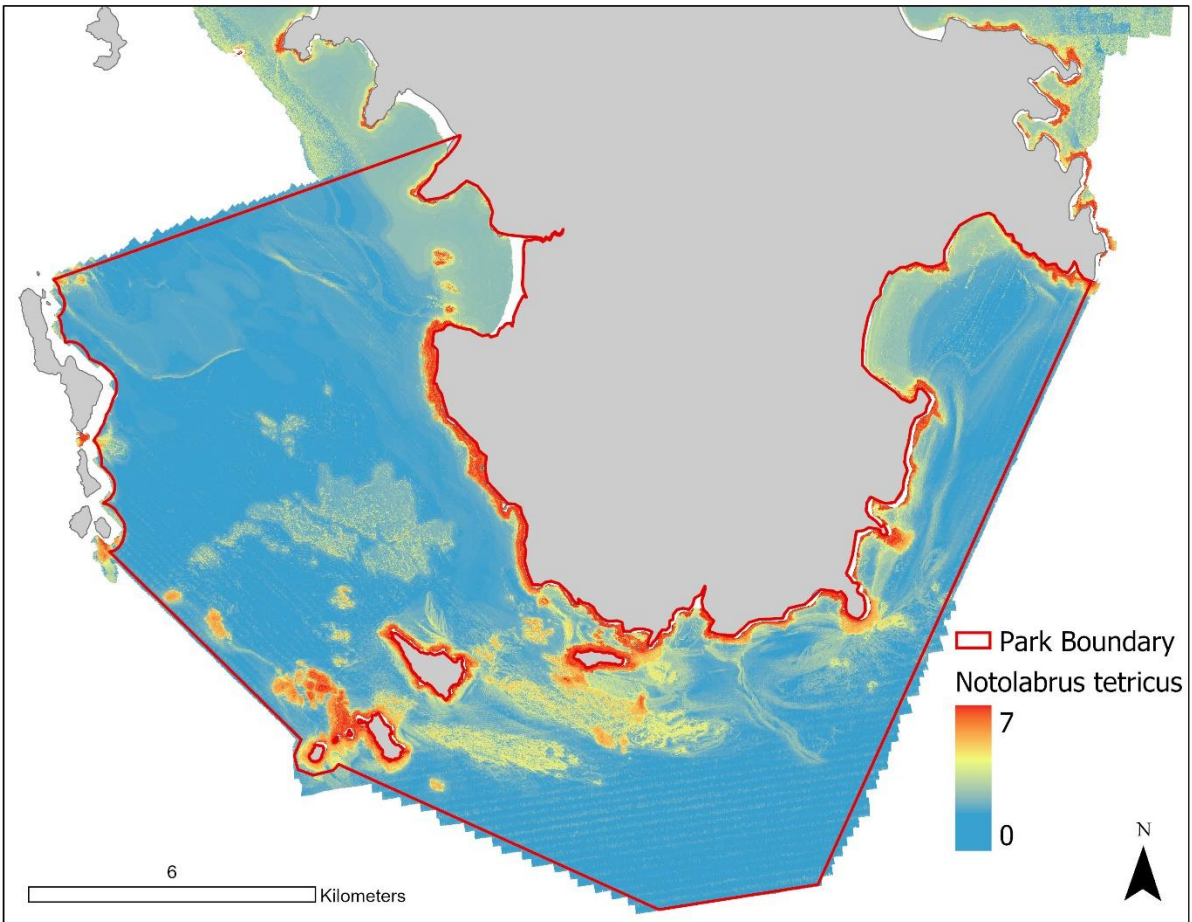


Figure 3.36: Predicted abundance of *Notolabrus tetricus* in 2021 from BRUVS sampling across the whole study site. The random forest model explained 44.5% of the variance and had a pseudo- R^2 value of 0.90

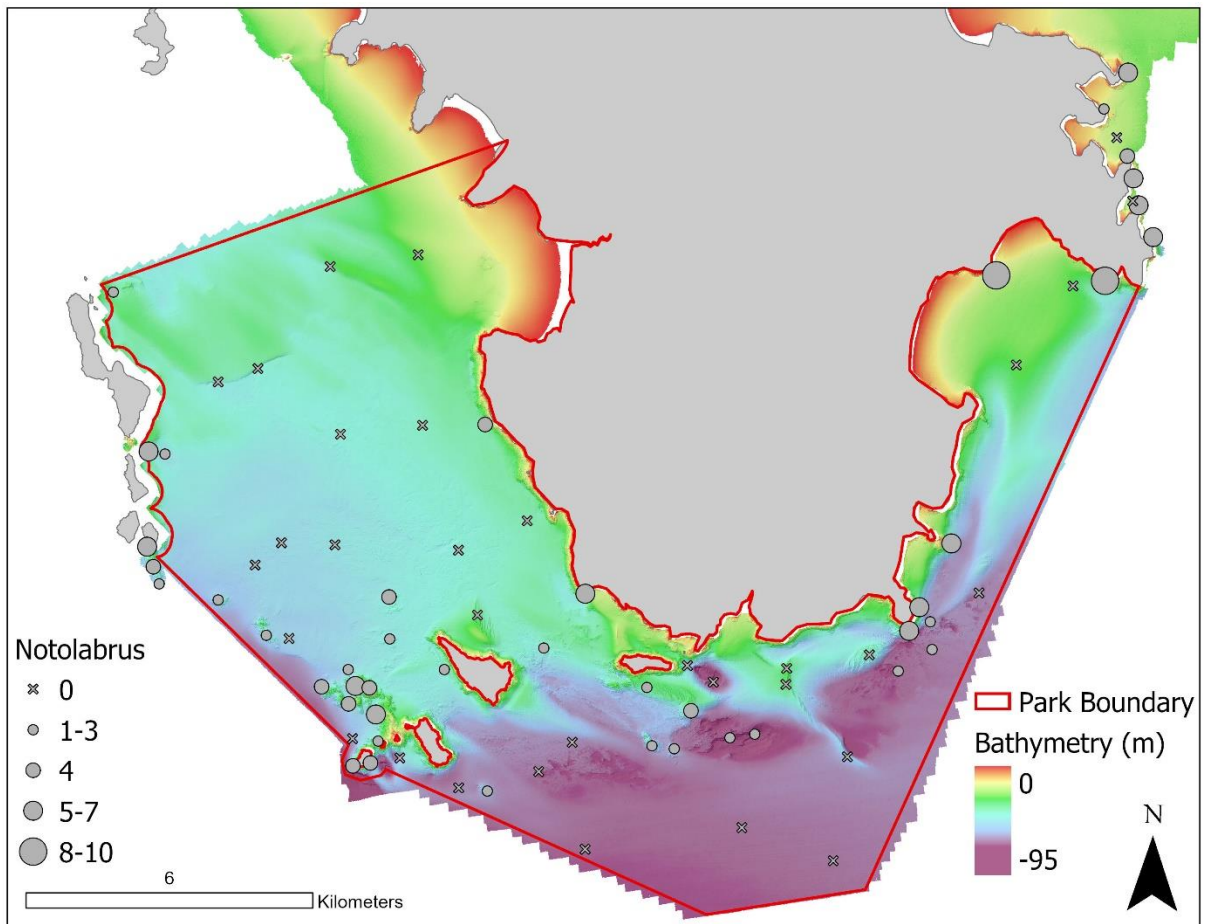


Figure 3.37: Abundance of *Notolabrus tetricus* for all BRUVS deployments in 2021. The size of each site marker corresponds to the abundance of *N. tetricus* observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

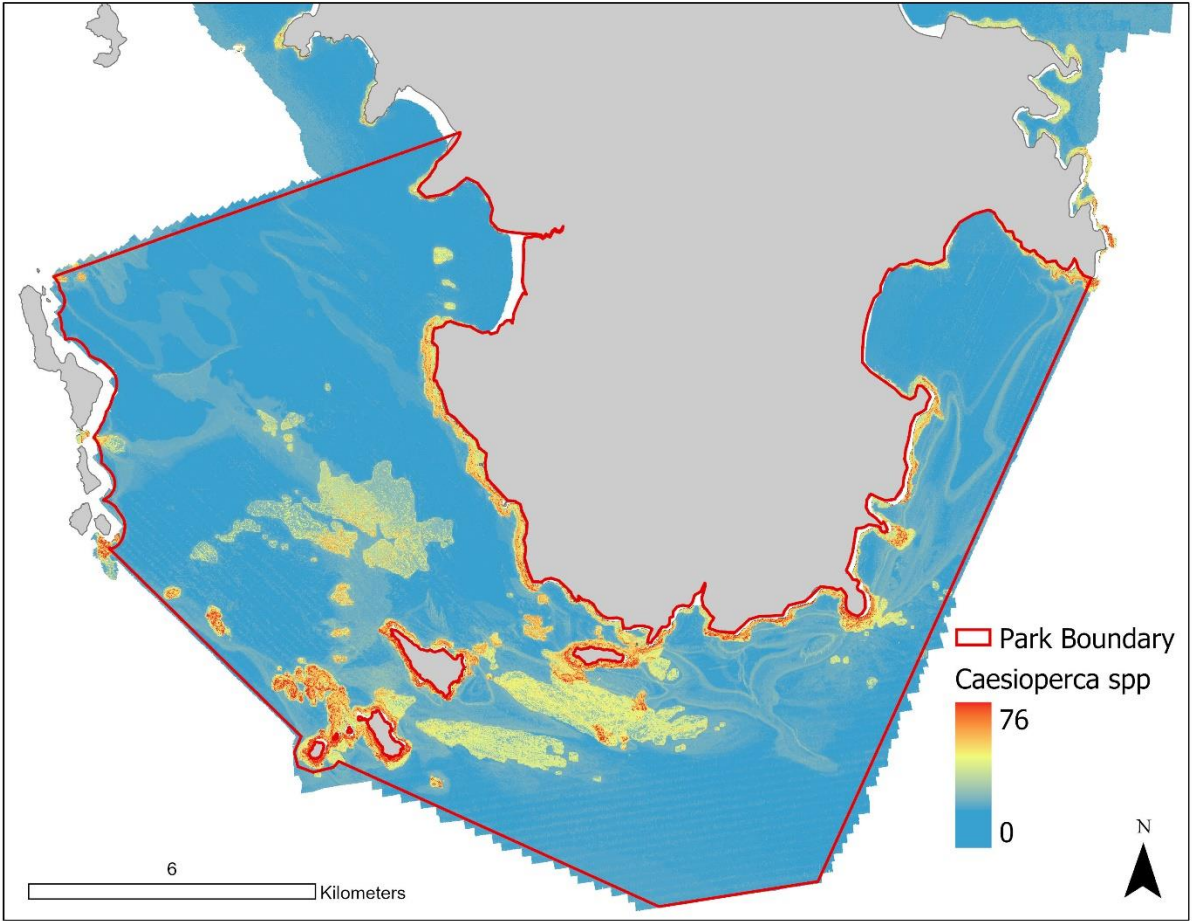


Figure 3.38: Predicted abundance of *Caesioperca* spp. in 2021 from BRUVS sampling across the whole study site. The random forest model explained 21.9% of the variance and had a pseudo- R^2 value of 0.84

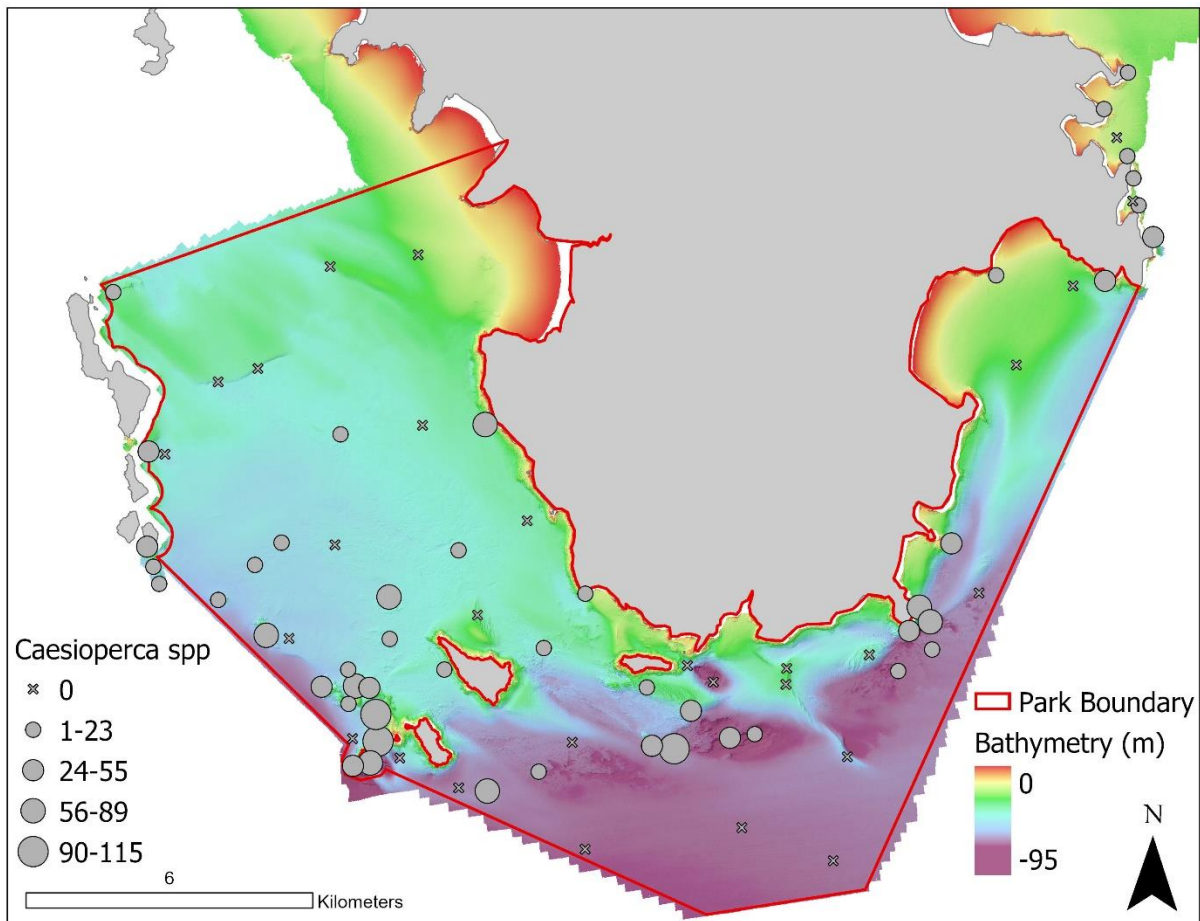


Figure 3.39: Abundance of *Caesioperca* spp. for all BRUVS deployments in 2021. The size of each site marker corresponds to the abundance of *Caesioperca* spp. observed at that site. These sites are overlaid on hillshaded bathymetry of the study area, coloured by depth

3.3.6 BRUVS surveys vs. Reef Life Survey

The fish assemblages observed via BRUVS differed from the assemblages observed via RLS. For fish species alone (i.e. teleosts and chondrichthyans), 50 species were common to both methods, 40 species were unique to BRUVS and only 19 species were unique to RLS (Figure 3.40). The species unique to BRUVS included large, more mobile species, such as sharks and rays (e.g. Melbourne Skate (*Spiniraja whitleyi*), Southern Sawshark (*Pristiophorus nudipinnis*) and Gummy Shark (*M. antarcticus*)) and species that are more common in deep waters or sandy areas, such as Red Gurnard (*Chelidonichthys kumu*), flathead (*Platycephalus* spp.) and Red Gurnard Perch (*Helicolenus percoides*). The species unique to RLS included those which are small, cryptic and associated with habitat, such as Little Rock Whiting (*Neoodax balteatus*), Weedy Seadragon (*Phyllopteryx taeniolatus*) and Common Threefin (*Trinorfolkia clarkei*).

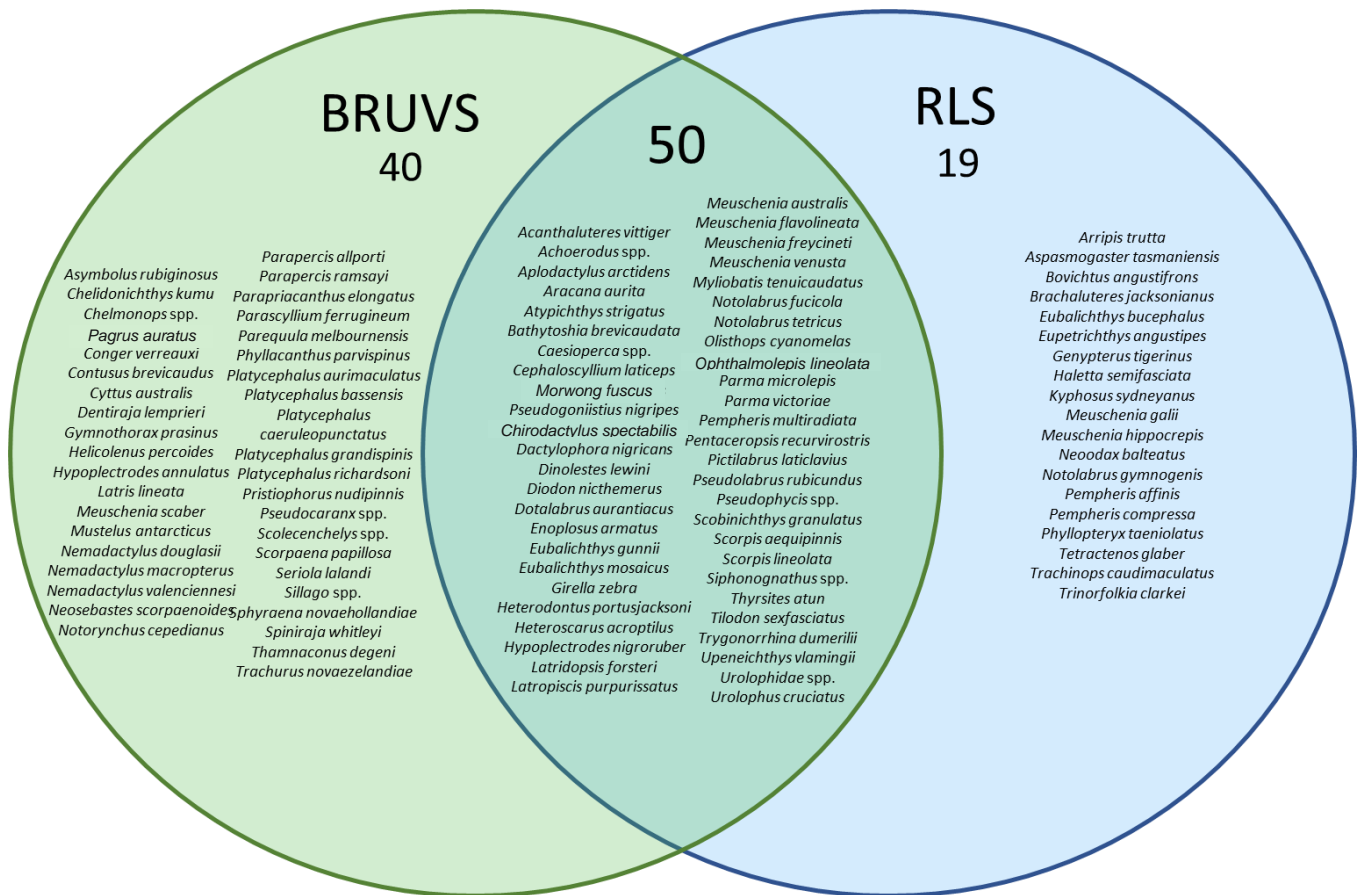


Figure 3.40: Overlapping and unique species observed in BRUVS and RLS/SRMP surveys for the fish-only assemblage observed across all years sampled combined

4. Discussion

4.1 Patterns in sea-surface temperatures

Although this region of Australia is a climate change hotspot where temperatures are increasing at rates up to 4 times the global average (Ridgway, 2007), Wilsons Promontory MNP seems to be buffered from these temperature increases. Summer SST is slightly increasing while annual SST has seen a decrease in recent years and winter temperatures have no discernible pattern. The strong currents coming from the east and wrapping around the point may be having some impact on buffering the impacts from warming waters to the east. Additionally, the water movement and mixing resulting from increased wave energy due to the high winds the coast is subject to may have a mellowing effect on increasing temperatures in this area. A better understanding of the oceanographic patterns of this region is needed to determine why SST is not increasing in Wilsons Promontory MNP as much as in other parts of the state.

Despite not increasing across all seasons and annually, the increase in summer SST can still have effects on the species present in the MPA. Observed decreases in some species of macroalgae, invertebrates and fish could all be directly or indirectly related to the increase in summer SST. Macroalgae are particularly vulnerable to increasing temperatures, and a loss in macroalgae could have indirect effects on other species that depend on it for food and habitat (Filbee-Dexter, 2018; Krumhansl, 2016; Staehr, 2009). Additionally, increased temperatures in summer, when many invertebrates spawn, has the potential to alter patterns in spawning and recruitment. Further studies are needed to better understand how the direct and indirect effects of temperature are affecting species within Wilsons Promontory MNP.

4.2 Subtidal reefs (including shallow and deep reefs)

4.2.1 Benthic communities

Overall, control charts showed that macroalgal indicators are in moderate to poor condition, with few observations of the brown alga *Seirococcus axillaris* and 2 species of red algae, *Phacelocarpus peperocarpos* and *Plocamium angustum*, since 2014 within and outside the MPA (Figure 3.9). It is unclear what is causing these declines, but they could be related to increasing summer temperatures or increased wave energy on the western side of the park. A reduction in carnivorous fish and invertebrates, resulting in an increase in herbivores, could also cause such a decline, but we did not find strong evidence of this in the key species we monitored. On the other hand, the canopy-forming species Crayweed (*Phyllospora comosa*) and Golden Kelp (*Ecklonia radiata*) have increased in cover during this time. Although some bare stipes of *P. comosa* were present in 2014 outside the park, there have been some increases in *P. comosa* since then with percentage cover within the MPA well above the lower limit of acceptable change. SRMP surveys previously found stable coverage of *P. comosa* from 2001 to 2011 and some evidence of increased cover in 2017 and 2020. Declines in *S. axillaris*, *P. peperocarpos* and *P. angustum* could potentially result from being outcompeted by the larger kelps *E. radiata* and *P. comosa*. More studies are needed to understand the interaction between the canopy-forming, understory and turfing algae species to determine their opposing patterns in recovery and declines. Notably, overabundant urchin populations, which have been a cause of major decline in macroalgae cover along the eastern coast of Victoria and in Port Phillip

Bay, were not observed in Wilsons Promontory MNP and are not linked to the observed declines in macroalgae there.

Reef habitat was assessed using video from BRUVS surveys. Two BRUVS surveys have been conducted at Wilsons Promontory MNP, allowing for habitat change to be observed for the matching sites across both years. A shift in habitat type from pebble cover to coarse sand was observed in the deep depth stratum (>60 m); however, overall patterns in habitat type were similar between the years sampled. The shift in substratum type over time could be an indication of the sediment dynamics present in this park, which experiences high currents as water moves around the point.

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

4.2.2 Key mobile fish (including sharks and rays)

The RLS control charts showed that 4 of the 6 key mobile fish species had healthy populations inside Wilsons Promontory MNP in 2021, falling within the zone of good condition. *Caesioperca rasor* was below the lower limit of acceptable change inside the park, while *C. lepidoptera* was not observed within the MPA or at reference sites outside the park and so is considered to be in poor condition.

Apart from *C. lepidoptera* and *C. rasor*, the fish species analysed from the RLS dataset (*Notolabrus fucicola*, *Notolabrus tetricus*, *Enoplosus armatus* and *Scorpius aequipinnis*) had higher abundances within the marine park than outside the park and were either stable or increasing in abundance. In contrast, the 2021 BRUVS dataset found higher average abundances of these species outside the park and a decline in abundance of *N. tetricus* from 153 individuals in 2016 to 82 in 2021. However, fewer than 20 individuals in total were observed for each of *E. armatus*, *N. fucicola* and *S. aequipinnis*, so we should interpret the trends with caution. Overall, conditions are generally favourable for these species within the marine park. The decline in *N. tetricus* abundance could be due to the natural variation in species abundances; however, a small decline is also observed in the reference sites in the RLS dataset, so the trend is likely to be widespread.

Both *Caesioperca* species showed a declining trend in abundance and were shown to be in poor (*C. lepidoptera*) or low (*C. rasor*) condition based on the RLS dataset. These 2 species could not be reliably differentiated based on the BRUVS footage so were grouped into one taxon: *Caesioperca* spp. In line with the RLS data, this taxon showed a decline in abundance from 1,464 individuals in 2016 to 1,171 in 2021. However, the BRUVS data contrast with the RLS data in finding a higher average abundance of *Caesioperca* spp. within the marine park compared with outside the park in 2021. The reason for the decline in *Caesioperca* spp. is not well understood. *Caesioperca* is a planktivorous genus that typically occurs in high abundance across reefs of south-eastern Australia. The only other planktivorous species observed in the BRUVS dataset were sweep (*Scorpius aequipinnis* and *Scorpius lineolata*). However, these species were observed too infrequently to accurately assess trends over time or within or outside the park and thus cannot be used to further

interpret the trends in *Caesioperca* spp. The decline in *Caesioperca* spp. could be due to a range of factors, including changes in food availability, unfavourable habitat conditions within the area, changes in predator abundance or natural fluctuations in abundance. This species should be a focus of ongoing longer-term monitoring at Wilsons Promontory to determine whether the decline observed in this study is persistent or a fluctuation the population will recover from.

The BRUVS dataset did not find that the marine park had higher total abundance, species richness or total biomass compared to fished areas, but the assemblage composition did differ between the areas sampled. Five species influencing the difference between the marine park and fished areas had higher abundances within the park, including Draughtboard Shark (*Cephaloscyllium laticeps*), perch (*Caesioperca* spp.), Velvet Leatherjacket (*Meuschenia scaber*), Sand Flathead (*Platycephalus bassensis*) and Common Gurnard Perch (*Neosebastes scorpaenoides*). Four of these species, particularly *P. bassensis* and *N. scorpaenoides*, could be considered fisheries targets and are likely benefiting from protection within the park. Overall, we would expect higher species richness, abundance and biomass within the park (Lester et al., 2009). The reason no differences were observed could be park age, size, lack of enforcement or lack of connectivity (Edgar et al., 2014). Wilsons Promontory MNP was established in 2002, and BRUVS sampling occurred 14 and 19 years after protection commenced. This timeframe should have been long enough for the influence of protection on fish assemblages to be observed (Babcock et al., 2010; Harasti et al., 2018). Ideally a baseline dataset would have been established prior to park establishment to allow direct changes in assemblages to be observed. There was no observed increase in abundance or richness from 2016 to 2021 and only minor changes to individual species, indicating that assemblages are stable. Wilsons Promontory MNP is the largest marine park in the state at 156 square kilometres (km²) and would be considered a large (>100 km²) marine park by global standards (Edgar et al., 2014). Thus, park size is unlikely to be a limiting factor in observing effects from protection. In contrast, the remoteness and difficulty accessing Wilsons Promontory MNP likely reduces fishing pressure in the region as a whole. The lack of difference between inside and outside the park boundaries may reflect generally low impacts of fishing in the regions adjacent to the marine national park. Small-scale habitat differences between the fished areas and the park may also be inhibiting the interpretation of the results, although care was taken to sample areas as comparable as possible.

Distinct fish assemblages were recorded for each habitat type (soft sediment, infralittoral reef and circalittoral reef) within the BRUVS dataset. Low-complexity soft-sediment areas were characterised by Sand Flathead (*Platycephalus bassensis*), Common Gurnard Perch (*Neosebastes scorpaenoides*), Degen's Leatherjacket (*Thamnaconus degeni*), Yellowtail Scad (*Trachurus novaezelandiae*) and Gummy Shark (*Mustelus antarcticus*). Soft-sediment habitats had the lowest species richness of any habitat type. Circalittoral reef and infralittoral reef had higher species richness than did soft sediment. Similarities between the assemblages in the 2 reef areas included high abundances of *Caesioperca* spp., Rosy Wrasse (*Pseudolabrus rubicundus*), Six-Spine Leatherjacket (*Meuschenia freycineti*) and Blue-Throat Wrasse (*Notolabrus tetricus*) in both habitats. The habitat types differed in that circalittoral reef was characterised by high abundances of Jackass Morwong (*Nemadactylus macropterus*) and Longfin Pike (*Dinolestes lewini*) while infralittoral reef was characterised by Maori Wrasse (*Ophthalmolepis lineolata*) and Silver Sweep (*Scorpiis lineolata*).

Including BRUVS in monitoring also allowed for the creation of spatially explicit models for various diversity metrics and abundance of key groups and species throughout the park, aided by the distribution of sample locations across environmental gradients observed. Reaching the same

density of field locations using underwater visual census 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 227 km², including across the entire extent of Wilsons Promontory MNP. Our findings identified strong relationships with environmental drivers for some species and community groups assessed, but not for all groups. For example, the models for planktivores and carnivores explained a significant amount of the variance in the relative abundance of these species and predicted their distribution with relatively good accuracy. On the other hand, some models, such as those for invertivore abundance and total abundance, did not explain much of the variation and could not make predictions with high confidence. These differences in results could be related 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 future sampling 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 that have not previously been visited.

The habitat modelling showed the importance of depth, sand cover, macroalgae cover and invertebrate cover for fish assemblages, with a relatively high percentage of the variation explained. Some species were positively correlated with an increase in depth (e.g. *Neosebastes scorpaenoides* and *Mustelus antarcticus*), while others were positively correlated with increases in habitat cover (e.g. *Platycephalus bassensis* with sand cover and *Meuschenia freycineti* with invertebrate cover). The highly variable nature of mobile fish assemblages and the complexities of multivariate modelling mean that field-based settings rarely produce high-percentage explanations. Model fit could be improved by including additional variables and, potentially, by increasing sample number.

4.2.3 Key macroinvertebrates

Wilsons Promontory MNP provides refuge for a number of key macroinvertebrates, such as the commercially important Blacklip Abalone (*Haliotis rubra*) (Woods et al., 2014). Control charts developed in this study found the *H. rubra* population within the MPA to be in good condition, while outside the MPA the population abundance is just below lower limit of acceptable change. Previous studies have shown that *H. rubra* predominantly feeds on Crayweed (*Phyllospora comosa*) (Holland et al., 2021). Cover of *P. comosa* has increased inside and outside the MPA in recent surveys, with greater increases inside. There may be a direct link between the increases in these 2 species, as we showed in Holland et al. (2021) across the state waters of Victoria.

Overall, Gunns Six-Armed Star (*Meridiastra gunnii*), Large-Plated Seastar (*Nectria macrobrachia*) and Purple Sea Urchin (*Heliocidaris erythrogramma*) have been declining in condition since 1999, with all 3 species of macroinvertebrates in moderate to poor condition. Given that 3 of the 5 characteristic

macroinvertebrates have shown substantial decline, it will be important to understand whether any management action could aid in the recovery of these species and whether there is a link between the decline in these species and the decline of certain macroalgae species. Future studies will need to analyse the interaction between these species and the biotopes they are associated with. Shifts in biotopes through time likely have a direct effect on the species that use specific biotopes as habitat. For example, *H. erythrogramma* is a herbivore, and the decline in its abundance could be related to the decline in cover of common macroalgal species *Seirococcus axillaris*, *Phacelocarpus peperocarpus* and *Plocamium angustum*. Understanding the links between these invertebrates and their habitats can help to focus future management actions, especially if those habitats are disappearing from the MPAs.

The declines in the carnivorous *N. macrobrachia* and *M. gunnii* could also be associated with declines in macroalgae, but indirectly: their food sources may depend on the abundance of macroalgae. Thus, the declines in these species may be an indication of overall declines in macroalgae within this region. More information is needed on how the changing communities are impacting individual species and what is causing those changes.

5. Conclusion

Many key species and habitat types in Wilsons Promontory MNP were found to be in good condition. Most key mobile fish species had healthy abundances, as did ecologically important macroinvertebrates like Blacklip Abalone (*Haliotis rubra*) and Orange Feather Star (*Cenolia trichoptera*), while major canopy-forming macroalgae had fair to good cover. However, there are signs of decline for several key species of fish, macroinvertebrates and, significantly, large macroalgal species. The underlying drivers of this decline are unclear.

The only major change in threats identified in this monitoring program was an increase in summer sea-surface temperatures. The impacts of increased temperatures on species in Wilsons Promontory MNP has not been studied in detail and warrants further investigation. More generally, ongoing long-term monitoring and further investigations of species in potential decline are needed to understand the drivers and guide management responses. Both underwater visual census and BRUVS surveys found that 10 species were more abundant inside the MPA than outside, which included some species commonly the target of fisheries. This suggests these species benefit from the no-take marine national park. However, species richness, total abundance and total biomass did not differ inside and outside the marine national park, which may also reflect a low impact of fishing pressure in areas surrounding the park.

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8. 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	Year	Total observed	Relative abundance (mean ±SE)	Total lengths	Mean length (cm) (±SE)	Total biomass observed (g)	Relative total biomass (g) (mean ±SE)
Aplodactylidae	<i>Aplodactylus arctidens</i>	Marblefish	2016	4	0.06 ± 0.03	1	38.87 ± 0	2,635.29	42.5 ± 20.72
			2021	1	0.01 ± 0.01	NA	NA	NA	NA
Aracanidae	<i>Aracana aurita</i>	Shaw's Cowfish	2016	1	0.02 ± 0.02	NA	NA	NA	NA
Aracanidae	<i>Aracana</i> spp.	Cowfish	2021	3	0.03 ± 0.02	2	15.79 ± 2.57	255.2	2.87 ± 1.76
Asteriidae	Asteriidae spp.	Seastar	2021	1	0.01 ± 0.01	NA	NA	NA	NA
Aulopidae	<i>Latropiscis purpurissatus</i>	Sergeant Baker	2016	15	0.24 ± 0.05	8	28.79 ± 2.79	2,733.56	44.09 ± 12.95
			2021	15	0.17 ± 0.04	9	30.04 ± 3	3,212.21	36.09 ± 11.38
Carangidae	<i>Pseudocaranx</i> spp.	Trevally	2016	1	0.02 ± 0.02	NA	NA	NA	NA
			2021	74	0.83 ± 0.37	23	14.53 ± 0.42	3,702.47	41.6 ± 18.08

	<i>Seriola lalandi</i>	Yellowtail Kingfish	2016	10	0.16 ± 0.12	1	118.86 ± 0	218,753.5	3,528.28 ± 2,524.96
			2021	10	0.11 ± 0.04	7	100.21 ± 3.91	135,072.8	1,517.67 ± 605.33
	<i>Trachurus novaezelandiae</i>	Yellowtail Scad	2016	276	4.45 ± 2.26	79	18.82 ± 0.71	48,283.83	778.77 ± 378.18
			2021	405	4.55 ± 1.41	40	15.38 ± 0.77	38,469.86	432.25 ± 130.83
Chaetodontidae	<i>Chelmonops</i> spp.	Talma	2021	1	0.01 ± 0.01	NA	NA	NA	NA
Cheilodactylidae	<i>Chirodactylus spectabilis</i>	Banded Morwong	2021	6	0.07 ± 0.03	3	51.28 ± 2.05	31,988.52	359.42 ± 143.5
	<i>Dactylophora nigricans</i>	Dusky Morwong	2021	1	0.01 ± 0.01	1	58.49 ± 0	1,268.29	14.25 ± 14.25
	<i>Morwong fuscus</i>	Red Morwong	2021	1	0.01 ± 0.01	NA	NA	NA	NA
	<i>Nemadactylus douglasii</i>	Grey Morwong	2016	3	0.05 ± 0.03	3	34.79 ± 3.46	1916.99	30.92 ± 19.25
			2021	6	0.07 ± 0.03	5	28.74 ± 3.85	2,397.25	26.94 ± 12.92
	<i>Nemadactylus macropterus</i>	Jackass Morwong	2016	120	1.94 ± 0.4	111	27.32 ± 0.6	40,120.44	647.1 ± 119.04
			2021	214	2.4 ± 0.56	77	27.54 ± 0.54	65,868.21	740.09 ± 160.57
	<i>Nemadactylus valenciennesi</i>	Queen Snapper	2016	20	0.32 ± 0.07	13	38.37 ± 2.09	15,685.99	253 ± 59.84
			2021	8	0.09 ± 0.03	5	35.7 ± 3.86	5,160.36	57.98 ± 24.79
	<i>Pseudogoniistius nigripes</i>	Magpie Perch	2016	30	0.48 ± 0.1	17	31.28 ± 1.36	12,438.57	200.62 ± 42.84

			2021	33	0.37 ± 0.06	22	34.56 ± 0.52	17,748.2	199.42 ± 35.58
Cidaridae	<i>Phyllacanthus parvispinus</i>	Eastern Slate-Pencil Urchin	2021	6	0.07 ± 0.05	NA	NA	NA	NA
Congridae	<i>Conger verreauxi</i>	Southern Conger	2021	1	0.01 ± 0.01	NA	NA	NA	NA
Cyttidae	<i>Cyttus australis</i>	Silver Dory	2016	4	0.06 ± 0.03	2	23.27 ± 6.63	1,127.29	18.18 ± 10.26
			2021	14	0.16 ± 0.04	8	22.59 ± 1.95	3,521.96	39.57 ± 11.99
Dasyatidae	<i>Bathytoshia brevicaudata</i>	Smooth Stingray	2016	1	0.02 ± 0.02	1	103.91 ± 0	14,143.63	228.12 ± 228.12
			2021	12	0.13 ± 0.04	7	140.84 ± 18.45	522,374.8	5,869.38 ± 2,458.97
Dinolestidae	<i>Dinolestes lewini</i>	Longfin Pike	2016	177	2.85 ± 1.26	76	25.61 ± 0.58	17,962.95	289.73 ± 123.83
			2021	91	1.02 ± 0.48	17	31.15 ± 1.14	16,386.82	184.12 ± 84.35
Diodontidae	<i>Diodon nictemerus</i>	Globe Fish	2016	1	0.02 ± 0.02	1	22.71 ± 0	256.79	4.14 ± 4.14
Diogenidae	Diogenidae spp.	Hermit crab	2021	16	0.18 ± 0.07	NA	NA	NA	NA
Enoplosidae	<i>Enoplosus armatus</i>	Old Wife	2016	13	0.21 ± 0.07	11	21.3 ± 0.71	728.41	11.75 ± 4.12
			2021	17	0.19 ± 0.06	6	23.04 ± 0.93	1,189.69	13.37 ± 4.14
Gempylidae	<i>Thyrsites atun</i>	Barracouta	2016	3	0.05 ± 0.03	NA	NA	NA	NA
			2021	5	0.06 ± 0.03	2	46.15 ± 4.89	3,098.86	34.82 ± 17.6
Gerreidae	<i>Parequula melbournensis</i>	Silverbelly	2016	142	2.29 ± 0.5	108	13.64 ± 0.25	5,961.42	96.15 ± 19.33
			2021	225	2.53 ± 0.79	59	13.74 ± 0.57	9,549.98	107.3 ± 28.04

Heterodontidae	<i>Heterodontus portusjacksoni</i>	Port Jackson Shark	2016	31	0.5 ± 0.17	25	70.9 ± 4.06	87,024.75	1,403.63 ± 611.76
			2021	31	0.35 ± 0.07	22	68.64 ± 2.92	70,544.58	792.64 ± 153.98
Hexanchidae	<i>Notorynchus cepedianus</i>	Broadnose Sevengill Shark	2021	6	0.07 ± 0.03	4	180.55 ± 8.62	239,200.2	2,687.64 ± 1,091.92
Kyphosidae	<i>Atypichthys strigatus</i>	Mado Sweep	2016	1	0.02 ± 0.02	1	13.13 ± 0	50.59	0.82 ± 0.82
			<i>Girella zebra</i>	Zebra Fish	2016	3	0.05 ± 0.03	1	37 ± 0
			2021	2	0.02 ± 0.02	2	30.6 ± 0.4	1,006.77	11.31 ± 7.96
	<i>Scorpis aequipinnis</i>	Sea Sweep	2016	2	0.03 ± 0.03	1	31.58 ± 0	931.62	15.03 ± 15.03
			2021	4	0.04 ± 0.02	4	29.38 ± 2.53	1,597.51	17.95 ± 9.48
	<i>Scorpis lineolata</i>	Silver Sweep	2016	19	0.31 ± 0.23	15	22.33 ± 1.12	2,721.55	43.9 ± 35.51
			2021	149	1.67 ± 0.76	35	16.14 ± 0.62	7,267.08	81.65 ± 34.34
	<i>Tilodon sexfasciatus</i>	Moonlighter	2016	2	0.03 ± 0.02	1	18.35 ± 0	274.84	4.43 ± 3.11
			2021	4	0.04 ± 0.03	2	23.75 ± 1.58	1,197.07	13.45 ± 8.48
Labridae	<i>Achoerodus viridis</i>	Eastern Blue Groper	2016	2	0.03 ± 0.02	2	48.64 ± 3.24	4,473.98	72.16 ± 51.61
			2021	2	0.02 ± 0.02	NA	NA	NA	NA
	<i>Dotalabrus aurantiacus</i>	Castelnau's Wrasse	2016	2	0.03 ± 0.02	1	18.65 ± 0	145.81	2.35 ± 1.65
			2021	1	0.01 ± 0.01	NA	NA	NA	NA
	<i>Notolabrus fucicola</i>	Purple Wrasse	2016	5	0.08 ± 0.05	3	27.29 ± 3.44	1,438.83	23.21 ± 13.74
			2021	5	0.06 ± 0.03	3	28.08 ± 4.92	1,678.78	18.86 ± 12.54
	<i>Notolabrus tetricus</i>	Blue-Throat Wrasse	2016	188	3.03 ± 0.47	162	28.53 ± 0.57	15,608.66	251.75 ± 37

			2021	205	2.3 ± 0.25	105	30.8 ± 0.65	19,323.85	217.12 ± 22.93
	<i>Ophthalmolepis lineolata</i>	Maori Wrasse	2016	16	0.26 ± 0.09	16	23.64 ± 1.83	1905.7	30.74 ± 12.87
			2021	64	0.72 ± 0.12	33	26.1 ± 0.91	8,572.87	96.32 ± 16.85
	<i>Pictilabrus laticlavus</i>	Senator Wrasse	2016	17	0.27 ± 0.07	13	20.34 ± 0.98	1,724.06	27.81 ± 8.33
			2021	27	0.3 ± 0.07	14	23.33 ± 1.11	4,131.3	46.42 ± 10.94
	<i>Pseudolabrus rubicundus</i>	Rosy Wrasse	2016	225	3.63 ± 0.68	187	16.39 ± 0.26	12,486.4	201.39 ± 40.15
			2021	195	2.19 ± 0.42	94	17.71 ± 0.34	12,840.09	144.27 ± 22.97
Latridae	<i>Latridopsis forsteri</i>	Bastard Trumpeter	2016	1	0.02 ± 0.02	1	26.43 ± 0	221.38	3.57 ± 3.57
			2021	1	0.01 ± 0.01	1	44.98 ± 0	1,103.07	12.39 ± 12.39
	<i>Latris lineata</i>	Striped Trumpeter	2016	1	0.02 ± 0.02	1	66.72 ± 0	4,224.02	68.13 ± 68.13
Loliginidae	<i>Sepioteuthis australis</i>	Southern Calamary	2016	5	0.08 ± 0.04	NA	NA	NA	NA
			2021	5	0.06 ± 0.02	NA	NA	NA	NA
Monacanthidae	<i>Acanthaluteres vittiger</i>	Toothbrush Leatherjacket	2016	10	0.16 ± 0.13	3	24.26 ± 3.76	2,443.28	39.41 ± 28.64
			2021	13	0.15 ± 0.05	7	20.95 ± 2.26	2,167.49	24.35 ± 9.97
	<i>Eubalichthys gunnii</i>	Gunn's Leatherjacket	2016	14	0.23 ± 0.07	10	24.13 ± 1.85	2,507.63	40.45 ± 13.96
			2021	23	0.26 ± 0.06	14	27.36 ± 1.29	5,673.59	63.75 ± 14.6
	<i>Eubalichthys mosaicus</i>	Mosaic Leatherjacket	2016	6	0.1 ± 0.04	4	23.33 ± 2.48	1,337.99	21.58 ± 9.46

	<i>Meuschenia australis</i>	Brown-Striped Leatherjacket	2016	1	0.02 ± 0.02	1	31.87 ± 0	518.62	8.36 ± 8.36
	<i>Meuschenia flavolineata</i>	Yellow-Stripe Leatherjacket	2021	5	0.06 ± 0.03	4	25.34 ± 1.27	1,353.02	15.2 ± 9.45
	<i>Meuschenia freycineti</i>	Six-Spine Leatherjacket	2016	66	1.06 ± 0.17	54	27.51 ± 1.05	26,600.87	429.05 ± 85.1
			2021	103	1.16 ± 0.14	64	29.85 ± 0.88	48,494.64	544.88 ± 76.31
	<i>Meuschenia scaber</i>	Velvet Leatherjacket	2016	75	1.21 ± 0.18	62	18.94 ± 0.59	22,189.89	357.9 ± 64.42
			2021	97	1.09 ± 0.16	57	19.92 ± 0.39	30,420.61	341.8 ± 50.44
	<i>Meuschenia venusta</i>	Stars And Stripes Leatherjacket	2016	10	0.16 ± 0.05	7	17.41 ± 1.21	942.99	15.21 ± 4.83
			2021	8	0.09 ± 0.03	5	17.71 ± 0.57	756.17	8.5 ± 2.91
	<i>Scobinichthys granulatus</i>	Rough Leatherjacket	2021	2	0.02 ± 0.02	2	22.12 ± 0.57	359.33	4.04 ± 2.85
	<i>Thamnaconus degeni</i>	Degen's Leatherjacket	2016	192	3.1 ± 0.99	189	15.4 ± 0.27	14,172.2	228.58 ± 60.68
			2021	433	4.87 ± 2.91	63	17.09 ± 0.61	37,700.08	423.6 ± 241.6
Moridae	<i>Pseudophycis</i> spp.	Cod	2016	96	1.55 ± 0.37	72	31.77 ± 0.62	28,880.19	465.81 ± 112.78
			2021	94	1.06 ± 0.17	36	36.82 ± 0.93	43,243.7	485.88 ± 78.41
Mullidae	<i>Upeneichthys vlamingii</i>	Southern Goatfish	2016	45	0.73 ± 0.32	35	17.15 ± 0.53	3,249.13	52.41 ± 22.27
			2021	44	0.49 ± 0.1	23	19.67 ± 0.89	4,821.76	54.18 ± 11.13

Muraenidae	<i>Gymnothorax prasinus</i>	Green Moray	2016	2	0.03 ± 0.02	NA	NA	NA	NA
			2021	2	0.02 ± 0.02	NA	NA	NA	NA
Myliobatidae	<i>Myliobatis tenuicaudatus</i>	Eagle Ray	2016	2	0.03 ± 0.02	2	74.15 ± 2.31	36,056.58	581.56 ± 409.67
Neosebastidae	<i>Neosebastes scorpaenoides</i>	Common Gurnard Perch	2016	46	0.74 ± 0.11	33	28.06 ± 0.81	22,195.01	357.98 ± 54.44
			2021	86	0.97 ± 0.15	43	29.02 ± 0.54	44,318.98	497.97 ± 73.3
Octopodidae	<i>Macroctopus maorum</i>	Maori Octopus	2016	1	0.02 ± 0.02	NA	NA	NA	NA
Octopodidae	<i>Octopus</i> spp.	Octopus	2016	3	0.05 ± 0.04	NA	NA	NA	NA
			2021	4	0.04 ± 0.02	NA	NA	NA	NA
Odacidae	<i>Heteroscarus acroptilus</i>	Rainbow Cale	2016	1	0.02 ± 0.02	NA	NA	NA	NA
			2021	3	0.03 ± 0.02	3	25.88 ± 0.75	469.24	5.27 ± 3.03
	<i>Olisthops cyanomelas</i>	Herring Cale	2016	2	0.03 ± 0.02	NA	NA	NA	NA
			2021	6	0.07 ± 0.03	3	37.71 ± 0.64	2,979.78	33.48 ± 13.29
	<i>Siphonognathus</i> spp.	Weed Whiting	2016	1	0.02 ± 0.02	NA	NA	NA	NA
			2021	3	0.03 ± 0.03	1	11.17 ± 0	21.73	0.24 ± 0.18
Ommastrephidae	<i>Nototodarus gouldi</i>	Gould's Squid	2016	16	0.26 ± 0.08	NA	NA	NA	NA
			2021	2	0.02 ± 0.02	NA	NA	NA	NA
Ophichthidae	<i>Scolecenchelys</i> spp.	Worm eel	2021	1	0.01 ± 0.01	NA	NA	NA	NA

Otariidae	<i>Arctocephalus pusillus</i>	Australian Fur Seal	2021	5	0.06 ± 0.02	NA	NA	NA	NA
Palinuridae	<i>Jasus edwardsii</i>	Southern Rock Lobster	2016	5	0.08 ± 0.03	NA	NA	NA	NA
			2021	3	0.03 ± 0.02	NA	NA	NA	NA
Parascylliidae	<i>Parascyllium ferrugineum</i>	Rusty Catshark	2016	1	0.02 ± 0.02	1	64.18 ± 0	1,694.14	27.32 ± 27.32
			2021	6	0.07 ± 0.03	4	61.82 ± 8.53	10,145.52	113.99 ± 51.55
Pempheridae	<i>Parapriacanthus elongatus</i>	Slender Bullseye	2021	2	0.02 ± 0.02	NA	NA	NA	NA
	<i>Pempheris multiradiata</i>	Common Bullseye	2016	2	0.03 ± 0.02	NA	NA	NA	NA
			2021	6	0.07 ± 0.05	3	16.2 ± 0.65	404.64	4.55 ± 3.33
Pentacerotidae	<i>Pentaceropsis recurvirostris</i>	Long-Snouted Boarfish	2016	3	0.05 ± 0.03	3	31.15 ± 2.57	1946.61	31.4 ± 18.74
			2021	7	0.08 ± 0.03	4	29.28 ± 1.74	3,691.92	41.48 ± 16.69
Pentanchidae	<i>Asymbolus rubiginosus</i>	Orange Spotted Catshark	2021	1	0.01 ± 0.01	1	41.29 ± 0	407.27	4.58 ± 4.58
Pinguipedidae	<i>Parapercis allporti</i>	Barred Grubfish	2021	1	0.01 ± 0.01	1	10.64 ± 0	9.86	0.11 ± 0.11
	<i>Parapercis ramsayi</i>	Spotted Grubfish	2016	2	0.03 ± 0.03	1	9.93 ± 0	15.91	0.26 ± 0.26
			2021	9	0.1 ± 0.04	4	11.08 ± 1.17	105.62	1.19 ± 0.51
Platycephalidae	<i>Platycephalus aurimaculatus</i>	Toothy Flathead	2021	1	0.01 ± 0.01	1	41.37 ± 0	822.01	9.24 ± 9.24
	<i>Platycephalus bassensis</i>	Sand Flathead	2016	87	1.4 ± 0.36	78	29.27 ± 0.79	15,296.3	246.71 ± 62.18

			2021	134	1.51 ± 0.33	58	34.06 ± 0.93	34,387.08	386.37 ± 81.18
	<i>Platycephalus caeruleopunctatus</i>	Bluespotted Flathead	2021	1	0.01 ± 0.01	1	34.45 ± 0	268.36	3.02 ± 3.02
	<i>Platycephalus grandispinis</i>	Longspine Flathead	2021	3	0.03 ± 0.03	1	21.02 ± 0	170.34	1.91 ± 1.91
	<i>Platycephalus richardsoni</i>	Tiger Flathead	2016	1	0.02 ± 0.02	NA	NA	NA	NA
Pomacentridae	<i>Parma microlepis</i>	White-Ear	2016	8	0.13 ± 0.04	7	16.81 ± 0.48	837.26	13.5 ± 4.58
			2021	3	0.03 ± 0.02	2	17.24 ± 1.6	339.94	3.82 ± 2.24
Pomacentridae	<i>Parma victoriae</i>	Victorian Scalyfin	2016	4	0.06 ± 0.03	3	17.2 ± 0.25	443.49	7.15 ± 3.49
Pristiophoridae	<i>Pristiophorus nudipinnis</i>	Southern Sawshark	2016	2	0.03 ± 0.02	NA	NA	NA	NA
			2021	4	0.04 ± 0.02	1	86.09 ± 0	16,945.31	190.4 ± 93.56
Rajidae	<i>Dentiraja lemprieri</i>	Thornback Skate	2016	2	0.03 ± 0.02	NA	NA	NA	NA
			2021	4	0.04 ± 0.02	1	50.77 ± 0	5,016.57	56.37 ± 27.7
	<i>Spiniraja whitleyi</i>	Melbourne Skate	2016	2	0.03 ± 0.02	1	110.07 ± 0	21,104.97	340.4 ± 238.72
			2021	4	0.04 ± 0.02	3	98.27 ± 28.48	40,996.3	460.63 ± 275.21
Rhinobatidae	<i>Trygonorrhina dumerilii</i>	Southern Fiddler Ray	2016	19	0.31 ± 0.07	12	78.31 ± 4.34	67,518.68	1,089.01 ± 325.73
			2021	22	0.25 ± 0.05	14	74.73 ± 3.83	67,659.22	760.22 ± 177.4
Scorpaenidae	<i>Scorpaena papillosa</i>	Southern Red Scorpionfish	2016	2	0.03 ± 0.02	NA	NA	NA	NA

			2021	8	0.09 ± 0.04	2	14.51 ± 2.36	332	3.73 ± 1.75
Scyliorhinidae	<i>Cephaloscyllium laticeps</i>	Draughtboard Shark	2016	124	2 ± 0.19	105	74.42 ± 1.1	282,047.9	4,549.16 ± 486.33
			2021	102	1.15 ± 0.12	56	76.06 ± 1.17	239,375.6	2,689.61 ± 259.24
Sebastidae	<i>Helicolenus percoides</i>	Red Gurnard Perch	2016	15	0.24 ± 0.07	12	22.25 ± 1.68	4,701.31	75.83 ± 26.9
			2021	7	0.08 ± 0.04	3	26.79 ± 2.01	3,410.26	38.32 ± 18.63
Serranidae	<i>Caesioperca</i> spp.	Perch	2016	1,669	26.92 ± 6.05	1,482	15.01 ± 0.09	56,948.38	918.52 ± 198.31
			2021	1968	22.11 ± 3.05	299	16.2 ± 0.15	74,373.65	835.66 ± 115.36
	<i>Hypoplectrodes annulatus</i>	Blackbanded Seaperch	2016	2	0.03 ± 0.03	2	19.75 ± 3.24	141.7	2.29 ± 2.29
			2021	2	0.02 ± 0.02	1	19.3 ± 0	122.68	1.38 ± 0.97
	<i>Hypoplectrodes nigroruber</i>	Banded Seaperch	2016	10	0.16 ± 0.05	9	17.76 ± 1.25	539.45	8.7 ± 2.84
			2021	1	0.01 ± 0.01	NA	NA	NA	NA
Sillaginidae	<i>Sillago</i> spp.	Whiting	2021	17	0.19 ± 0.17	1	22.98 ± 0	1,262.2	14.18 ± 12.55
Sparidae	<i>Pagrus auratus</i>	Snapper	2016	14	0.23 ± 0.17	14	28.49 ± 1.38	7,163.28	115.54 ± 95.22
			2021	106	1.19 ± 0.46	38	25.95 ± 0.72	39,016.98	438.39 ± 164.06
Sphyraenidae	<i>Sphyraena novaehollandiae</i>	Snook	2021	1	0.01 ± 0.01	NA	NA	NA	NA

Tetraodontidae	<i>Contusus brevicaudus</i>	Prickly Toadfish	2016	3	0.05 ± 0.03	2	18.11 ± 1.27	342.1	5.52 ± 3.18
			2021	3	0.03 ± 0.02	3	16.54 ± 0.86	264.76	2.97 ± 1.73
Triakidae	<i>Mustelus antarcticus</i>	Gummy Shark	2016	46	0.74 ± 0.13	43	94.33 ± 3.04	281,362.5	4,538.11 ± 861.27
			2021	45	0.51 ± 0.09	32	102.32 ± 4.2	348,617.3	3,917.05 ± 709.5
Triglidae	<i>Chelidonichthys kumu</i>	Red Gurnard	2016	3	0.05 ± 0.03	3	34.67 ± 1.87	1,266.74	20.43 ± 11.94
Urolophidae	Urolophidae spp.	Stingaree	2016	4	0.06 ± 0.03	1	71.44 ± 0	17,296.74	278.98 ± 136.02
			2021	14	0.16 ± 0.04	9	32.11 ± 3.42	6,504.65	73.09 ± 28.76
	<i>Urolophus cruciatus</i>	Banded Stingaree	2021	15	0.17 ± 0.04	6	29.02 ± 0.97	4,226.13	47.48 ± 12.12

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

Taxon	2014	2015	2017	2020
<i>Acanthaluteres vittiger</i>	10 \pm 7.07	0 \pm 0.00	2,355 \pm 86.32	516 \pm 27.73
<i>Achoerodus gouldii</i>	0 \pm 0.00	0 \pm 0.00	0 \pm 0.00	28 \pm 2.04
<i>Aplodactylus arctidens</i>	10 \pm 7.07	0 \pm 0.00	89 \pm 4.22	72 \pm 3.34
<i>Aracana aurita</i>	0 \pm 0.00	0 \pm 0.00	4 \pm 0.60	8 \pm 1.26
<i>Arctcephalus pusillus</i>	0 \pm 0.00	0 \pm 0.00	180 \pm 25.62	64 \pm 6.33
<i>Arripis trutta</i>	0 \pm 0.00	0 \pm 0.00	0 \pm 0.00	280 \pm 44.27
<i>Aspasmogaster tasmaniensis</i>	0 \pm 0.00	0 \pm 0.00	5 \pm 0.75	0 \pm 0.00
<i>Atypichthys strigatus</i>	0 \pm 0.00	0 \pm 0.00	917 \pm 45.61	426 \pm 26.99
<i>Bathytoshia brevicaudata</i>	0 \pm 0.00	0 \pm 0.00	0 \pm 0.00	5 \pm 0.79
<i>Bovichtus angustifrons</i>	0 \pm 0.00	0 \pm 0.00	0 \pm 0.00	4 \pm 0.63
<i>Brachaluteres jacksonianus</i>	0 \pm 0.00	0 \pm 0.00	5 \pm 0.75	0 \pm 0.00
<i>Caesioperca lepidoptera</i>	15 \pm 10.61	1 \pm 0.71	660 \pm 42.87	0 \pm 0.00
<i>Caesioperca rasor</i>	205 \pm 88.39	596 \pm 292.74	11,361 \pm 537.87	3,888 \pm 139.16
<i>Cephaloscyllium laticeps</i>	0 \pm 0.00	0 \pm 0.00	0 \pm 0.00	10 \pm 1.10
<i>Chirodactylus spectabilis</i>	0 \pm 0.00	0 \pm 0.00	30 \pm 2.11	77 \pm 4.89
<i>Dactylophora nigricans</i>	5 \pm 3.54	0 \pm 0.00	41 \pm 2.74	0 \pm 0.00
<i>Dinolestes lewini</i>	5 \pm 3.54	755 \pm 533.87	578 \pm 27.57	528 \pm 29.85
<i>Diodon nictemerus</i>	5 \pm 3.54	0 \pm 0.00	0 \pm 0.00	4 \pm 0.63
<i>Dotalabrus aurantiacus</i>	5 \pm 3.54	0 \pm 0.00	94 \pm 6.19	10 \pm 1.10
<i>Enoplosus armatus</i>	10 \pm 7.07	12 \pm 5.66	691 \pm 17.11	512 \pm 19.99
<i>Eubalichthys bucephalus</i>	0 \pm 0.00	0 \pm 0.00	0 \pm 0.00	15 \pm 1.75
<i>Eubalichthys gunnii</i>	0 \pm 0.00	0 \pm 0.00	19 \pm 1.39	4 \pm 0.63
<i>Eubalichthys mosaicus</i>	0 \pm 0.00	0 \pm 0.00	5 \pm 0.75	0 \pm 0.00
<i>Eupetrichthys angustipes</i>	0 \pm 0.00	5 \pm 3.54	20 \pm 1.81	4 \pm 0.63
<i>Genypterus tigerinus</i>	0 \pm 0.00	1 \pm 0.71	0 \pm 0.00	0 \pm 0.00
<i>Girella zebra</i>	20 \pm 0.00	10 \pm 7.07	251 \pm 9.14	234 \pm 18.85
<i>Haletta semifasciata</i>	0 \pm 0.00	0 \pm 0.00	17 \pm 1.51	45 \pm 3.16
<i>Heterodontus portusjacksoni</i>	0 \pm 0.00	0 \pm 0.00	63 \pm 6.10	8 \pm 1.26
<i>Heteroscarus acroptilus</i>	5 \pm 3.54	6 \pm 2.83	94 \pm 3.76	29 \pm 1.60
<i>Hypoplectrodes nigroruber</i>	5 \pm 3.54	0 \pm 0.00	20 \pm 1.77	19 \pm 1.84
<i>Kyphosus sydneyanus</i>	0 \pm 0.00	0 \pm 0.00	15 \pm 2.26	125 \pm 18.33
Labrid spp.	0 \pm 0.00	0 \pm 0.00	5 \pm 0.75	0 \pm 0.00

<i>Latridopsis forsteri</i>	0 ± 0.00	0 ± 0.00	4 ± 0.60	8 ± 0.88
<i>Latropiscis purpurissatus</i>	0 ± 0.00	0 ± 0.00	4 ± 0.60	4 ± 0.63
<i>Meuschenia australis</i>	0 ± 0.00	1 ± 0.71	0 ± 0.00	0 ± 0.00
<i>Meuschenia flavolineata</i>	0 ± 0.00	5 ± 3.54	259 ± 9.02	108 ± 5.29
<i>Meuschenia freycineti</i>	0 ± 0.00	1 ± 0.71	287 ± 17.14	108 ± 4.71
<i>Meuschenia galii</i>	0 ± 0.00	0 ± 0.00	10 ± 1.05	0 ± 0.00
<i>Meuschenia hippocrepis</i>	0 ± 0.00	0 ± 0.00	61 ± 3.38	47 ± 4.01
<i>Meuschenia venusta</i>	0 ± 0.00	0 ± 0.00	4 ± 0.60	0 ± 0.00
Monacanthid spp.	0 ± 0.00	0 ± 0.00	0 ± 0.00	8 ± 0.88
<i>Morwong fuscus</i>	0 ± 0.00	0 ± 0.00	5 ± 0.75	0 ± 0.00
<i>Myliobatis australis</i>	0 ± 0.00	0 ± 0.00	4 ± 0.60	0 ± 0.00
<i>Neodax balteatus</i>	0 ± 0.00	0 ± 0.00	14 ± 1.20	0 ± 0.00
<i>Notolabrus fucicola</i>	90 ± 7.07	39 ± 21.92	382 ± 18.29	739 ± 28.83
<i>Notolabrus gymnogenis</i>	0 ± 0.00	0 ± 0.00	24 ± 2.21	12 ± 1.07
<i>Notolabrus tetricus</i>	305 ± 3.54	88 ± 50.91	4,842 ± 70.96	3,603 ± 88.10
<i>Olisthops cyanomelas</i>	55 ± 38.89	7 ± 4.95	946 ± 20.87	924 ± 20.24
<i>Ophthalmolepis lineolata</i>	0 ± 0.00	0 ± 0.00	110 ± 4.53	41 ± 3.00
<i>Parma microlepis</i>	0 ± 0.00	0 ± 0.00	128 ± 8.09	57 ± 3.92
<i>Parma victoriae</i>	35 ± 10.61	46 ± 31.11	296 ± 8.71	133 ± 4.93
<i>Pempheris affinis</i>	0 ± 0.00	0 ± 0.00	15 ± 2.26	0 ± 0.00
<i>Pempheris compressa</i>	0 ± 0.00	0 ± 0.00	5 ± 0.75	0 ± 0.00
<i>Pempheris multiradiata</i>	0 ± 0.00	10 ± 7.07	148 ± 8.73	209 ± 12.89
<i>Pentaceropsis recurvirostris</i>	0 ± 0.00	0 ± 0.00	95 ± 6.35	37 ± 2.47
<i>Phyllopteryx taeniolatus</i>	0 ± 0.00	0 ± 0.00	5 ± 0.75	0 ± 0.00
<i>Pictilabrus laticlavus</i>	5 ± 3.54	10 ± 7.07	258 ± 7.84	169 ± 5.25
<i>Pseudogoniistius nigripes</i>	50 ± 7.07	34 ± 18.38	928 ± 15.88	610 ± 11.23
<i>Pseudolabrus rubicundus</i>	10 ± 7.07	5 ± 3.54	127 ± 6.38	0 ± 0.00
<i>Pseudophycis barbata</i>	0 ± 0.00	0 ± 0.00	4 ± 0.60	0 ± 0.00
<i>Scobinichthys granulatus</i>	0 ± 0.00	4 ± 2.83	9 ± 0.95	0 ± 0.00
<i>Scorpius aequipinnis</i>	150 ± 106.07	157 ± 108.19	2,058 ± 75.01	3,665 ± 112.76
<i>Scorpius lineolata</i>	95 ± 67.18	224 ± 46.67	1,388 ± 63.92	1,415 ± 62.89
<i>Sepia apama</i>	0 ± 0.00	0 ± 0.00	5 ± 0.75	0 ± 0.00
<i>Sepioteuthis australis</i>	0 ± 0.00	0 ± 0.00	200 ± 30.15	0 ± 0.00
<i>Siphonognathus beddomei</i>	45 ± 31.82	6 ± 2.83	185 ± 7.78	1,041 ± 77.16
<i>Tetractenos glaber</i>	5 ± 3.54	0 ± 0.00	25 ± 1.69	4 ± 0.63
<i>Thyrsites atun</i>	0 ± 0.00	0 ± 0.00	0 ± 0.00	5 ± 0.79

<i>Tilodon sexfasciatus</i>	10 ± 7.07	5 ± 3.54	195 ± 7.88	40 ± 2.73
<i>Trachinops caudimaculatus</i>	0 ± 0.00	0 ± 0.00	784 ± 64.93	294 ± 28.31
<i>Trinorfolkia clarkei</i>	5 ± 3.54	0 ± 0.00	0 ± 0.00	0 ± 0.00
<i>Trygonoptera mucosa</i>	5 ± 3.54	0 ± 0.00	0 ± 0.00	0 ± 0.00
<i>Trygonorrhina dumerilii</i>	0 ± 0.00	0 ± 0.00	0 ± 0.00	10 ± 1.58
<i>Upeneichthys vlamingii</i>	0 ± 0.00	1 ± 0.71	192 ± 8.61	54 ± 3.36
<i>Urolophus cruciatus</i>	0 ± 0.00	0 ± 0.00	4 ± 0.60	0 ± 0.00
<i>Urolophus paucimaculatus</i>	0 ± 0.00	0 ± 0.00	4 ± 0.60	0 ± 0.00
<i>Urolophus spp.</i>	0 ± 0.00	0 ± 0.00	5 ± 0.75	0 ± 0.00

Appendix 1.3. Relative abundances (mean ± SE) of all macroinvertebrate and cryptic fish species observed in Reef Life Surveys, for each year sampled (2017 and 2020)

Taxon	2017	2020
<i>Aetapcus maculatus</i>	0 ± 0.00	4 ± 0.85
<i>Aspasmogaster tasmaniensis</i>	5 ± 0.94	0 ± 0.00
<i>Bovichtus angustifrons</i>	13 ± 1.37	0 ± 0.00
<i>Conger wilsoni</i>	0 ± 0.00	5 ± 1.07
<i>Diodon nichthemerus</i>	22 ± 1.73	0 ± 0.00
<i>Genypterus tigerinus</i>	0 ± 0.00	4 ± 0.85
<i>Heteroclinus johnstoni</i>	0 ± 0.00	9 ± 1.33
<i>Heteroclinus spp.</i>	13 ± 1.37	5 ± 1.07
<i>Heteroclinus tristis</i>	28 ± 3.07	0 ± 0.00
<i>Hypoplectrodes maccullochi</i>	5 ± 0.94	0 ± 0.00
<i>Hypoplectrodes nigroruber</i>	30 ± 2.19	21 ± 1.81
<i>Latropiscis purpurissatus</i>	4 ± 0.76	0 ± 0.00
<i>Lotella rhacina</i>	13 ± 1.37	4 ± 0.85
<i>Optivus agastos</i>	4 ± 0.76	0 ± 0.00
<i>Paraplesiops meleagris</i>	0 ± 0.00	5 ± 1.07
<i>Parapriacanthus elongatus</i>	4 ± 0.76	0 ± 0.00
<i>Pempheris multiradiata</i>	199 ± 14.52	364 ± 34.72
<i>Scorpaena papillosa</i>	5 ± 0.94	0 ± 0.00
<i>Trachichthys australis</i>	8 ± 1.05	9 ± 1.33
<i>Trinorfolkia clarkei</i>	13 ± 1.37	4 ± 0.85
<i>Vincentia conspersa</i>	0 ± 0.00	17 ± 2.72

