Horoirangi Marine Reserve

Assessing the effects of protection on key subtidal reef species

2006 - 2025

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Cover: DOC diver reeling in combined slate/transect at Horoirangi Marine Reserve. *Photo: Stew Robertson*

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Abstract

The Horoirangi Marine Reserve is a 904 hectare no-take marine reserve in the Nelson-Glenduan area on the northern South Island. Established in 2005, it was created to address concerns about localised depletion of fished species and to protect unique boulder bank and rocky reef habitats and their associated communities. Stretching along 5 km of accessible coastline, the reserve is well placed to protect a range of commonly targeted inshore species.

We analysed a long-term dataset to test for spatial and temporal reserve effects on several key species of ecological, cultural, recreational, or commercial importance. Diver-based Underwater Visual Census surveys recorded the abundance and size of rock lobster, fish communities, kina, and pāua across 21 sites within the reserve and 19 nearby non-reserve sites between 2006 and 2025.

Overall, results indicated that after almost 20 years of protection the Horoirangi Marine Reserve has had positive effects on some key species compared to nearby non-reserve sites. Rock lobster abundance was consistently greater inside the reserve and increased substantially through time, while remaining low outside. Lobsters were also larger inside the reserve, although this size difference was not statistically significant.

Reef fish communities differed between reserve and non-reserve sites. Blue cod were significantly more abundant inside the reserve, while tarakihi, spotted wrasse, marblefish, and goatfish were more common outside. Blue cod and tarakihi were larger inside the reserve, and a greater proportion of blue cod inside the reserve exceeded the current harvest size.

There was no significant difference in kina abundance between reserve and non-reserve sites, although kina were only surveyed once in 2006. Pāua were surveyed three times but were not found to be more abundant or larger inside the reserve. All pāua measured inside and outside the reserve were below the minimum legal harvest size.

We determined that the coastal portion of the Horoirangi Marine Reserve has had positive effects on rock lobster, blue cod, and tarakihi compared to nearby non-reserve sites. There were no clear patterns for other target species such as pāua, blue moki, and snapper.

Some temporal trends were difficult to interpret due to limited sampling replication. Future monitoring would benefit from structured, regular sampling of fixed sites through time. The addition of methods such as baited underwater video and potting surveys would provide complementary data to help interpret reserve effects and address limitations associated with diver surveys.

This report provides information on the size and abundance of key species in the Horoirangi Marine Reserve and presents quantitative time series datasets. These results provide a baseline for future monitoring to continue assessing long-term marine reserve effects. The reserve provides an excellent opportunity for further investigation into species interactions in unfished areas (e.g. kina, kelp, and rock lobster interactions), recovery trajectories of highly exploited stock, and long-term effects from spatial protection.

Keywords

Marine reserve, monitoring, species, pāua, rock lobster, fish, kina, reserve effects, recovery.

Introduction

Marine reserves are specially designated areas in the territorial sea with the highest level of regulatory protection from human disturbances. Within Aotearoa New Zealand's marine reserves, the removal of any living or non-living material from the foreshore (mean high-tide mark) to the sea floor is prohibited, but non-extractive activities such as diving and boating are allowed. Under the Marine Reserves Act 1971[1], marine reserves are established to preserve marine biodiversity, allow ecosystems to return to a natural state, and to provide an unfished reference for scientific research.

The Horoirangi Marine Reserve was gazetted in 2005 to protect a unique marine ecosystem, promote biodiversity recovery, and serve as a scientific reference site for long-term ecological monitoring. The reserve, located in the Nelson/Glenduan area at the northern end of New Zealand's South Island, covers 904 hectares within Tasman Bay. It extends one nautical mile offshore to a maximum depth of approximately 68 metres and encompasses around five kilometres of coastline (Fig. 1). It includes part of the Nelson Boulder Bank, a unique geological formation of large boulders that provides complex habitat for a range of marine species such as rock lobster, pāua and fish. The reserve also contains intertidal and subtidal rocky reefs, as well as soft-sediment habitats, the latter of which account for approximately three-quarters of the seafloor (Grange 2006; Keeley 2006). Its proximity to Nelson (~11 km) allows relatively easy foot and boat access to the reserve. Prior to the reserve's establishment, the area was described as highly degraded, with surveys by the Department of Conservation and NIWA showing very low abundances of targeted fish species (e.g., blue cod) and rock lobster (Forest and Bird Protection Society 1999).

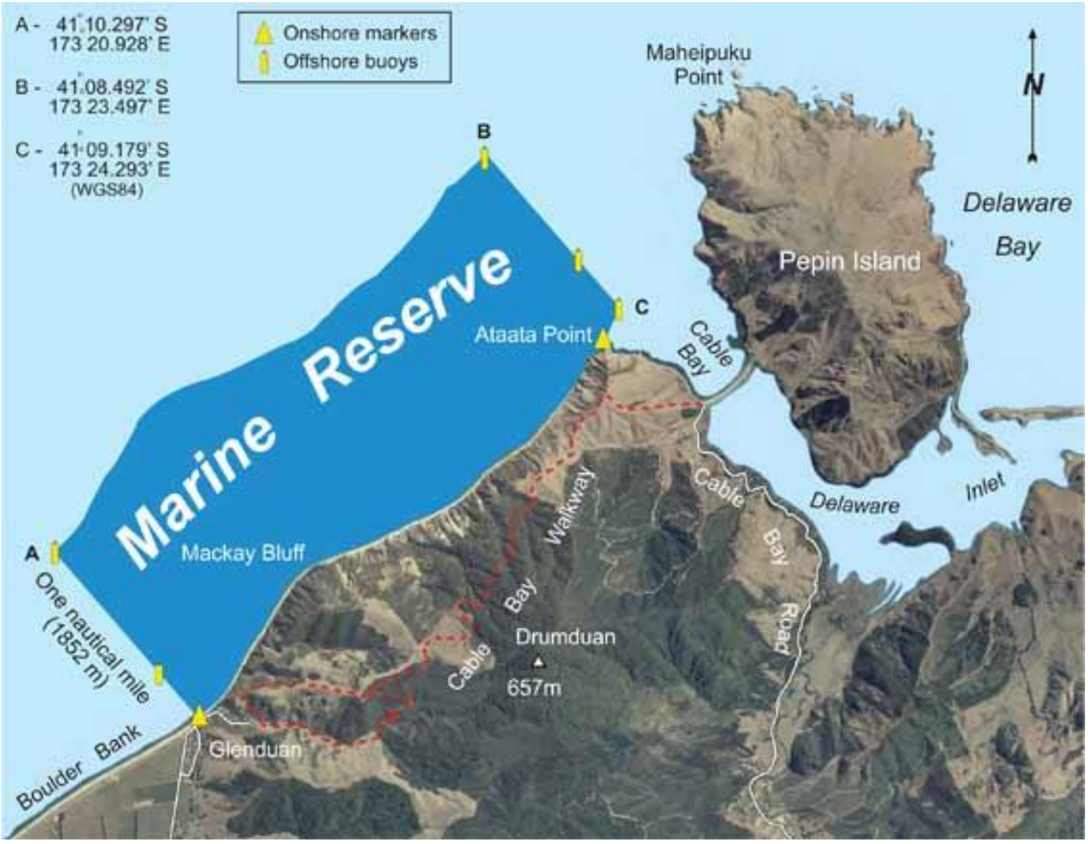

Figure 1. Map showing the spatial extent of Horoirangi Marine Reserve (blue polygon). Sourced from Department of Conservation.

[1] Marine Reserves Act 1971. https://www.legislation.govt.nz/act/public/1971/15/en/latest/#DLM397838

The marine communities within Horoirangi Marine Reserve and nearby sites have been surveyed intermittently from 2006-2025. A baseline survey by the Cawthron Institute of soft-sediment habitats within the reserve characterised the seafloor and described the infaunal and epibiotic communities as typical of the Tasman Bay (Keeley 2006). Department of Conservation research divers surveyed rocky subtidal reef habitats in and out of the reserve semi-regularly since 2006. A 2013 analysis of this data showed that after 7 years of protection there were significant positive reserve effects on the abundance and size of rock lobster and blue cod, but not on pāua and other commonly targeted species such as blue moki and tarakihi (Davidson et al. 2013).

This report provides an updated analysis and interpretation of DOC dive survey data on rocky subtidal reefs from 2006-2025 within the Horoirangi Marine Reserve and nearby non-reserve sites, and tests for reserve effects on key species over a 19-year period. Key species are defined in the Marine Monitoring and Reporting Framework (2022) as species that:

…"either have a disproportionately large role in maintaining the structure of an ecological community (termed ecological keystone species; Paine 1969) or shape the cultural identity of a people, as reflected in the fundamental roles they have in diet, materials, medicine, recreation, economies and/or spiritual practices (termed cultural keystone species; Garibaldi & Turner 2004)".

The two research questions are:

1. Is the number and size of key species significantly different inside the Horoirangi Marine Reserve compared to nearby non-reserve sites?

2. Does the number and size of key species change over time inside the Horoirangi Marine Reserve?

This report provides a concise assessment of the effects of Horoirangi Marine Reserve on key species between 2006 and 2025 and offers recommendations to optimise future monitoring efforts.

Methods

Survey overview

Between 2006 and 2025, rock lobster (kōura, *Jasus edwardsii*), pāua (*Haliotis iris*), kina (*Evechinus chloroticus*), and reef fish communities were sampled inside and outside of the Horoirangi Marine Reserve in repeated Underwater Visual Census surveys (UVC, Table 1).

Table 1. Underwater Visual Census (UVC) survey events (grey fill) per key species over the survey period from 2006-2025. Years when size measurements were taken that were analysed in this report are denoted with an "S". Some annual size data were not analysed in this report due to the incompatibility of data types (i.e. categorical vs. numerical).

Survey Type	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024	2025
UVC Lobster												S	S	S		S		S		S
UVC Fish															S		S		S	
UVC Kina																				
UVC Pāua																	S			

Rock lobster abundance and size were surveyed from 2006–2025 using UVC survey methods adapted from previous marine reserve studies (e.g., Freeman et al. 2012). Transects 25 m in length were haphazardly placed within suitable rocky-reef habitat, avoiding soft sediment where possible. Divers recorded the abundance, sex, and carapace length of all lobsters visible within a 4 m-wide band (total sample area 100 m^2). Because lobsters could not be handled, size was estimated *in situ* using a ruler placed along the carapace. Transects ranged from 6–14 m depth, with 10–25 transects surveyed per site at 6–8 reserve sites and 10–16 transects per site at 4–6 non-reserve sites depending on the year (Fig. 2). Surveys were conducted only when underwater visibility exceeded 2 m.

To convert carapace length to a more intuitive measurement for managers, we used the sex-specific relationships developed by Fry et al. (2014) to convert *J. edwardsii* carapace length to tail width. This is the measurement used for analysis throughout this report. Earlier size measurements (from 2006 – 2017) were removed from the dataset and not analysed because they were too coarse (only <10 or >10 cm) to provide meaningful insights.

Reef fish communities were surveyed by UVC along 60 m^2 transects established parallel to shore in boulder and rocky reef habitat at 5–10 m depth. Reef fish abundance (density) and size (maximum length) was sampled between 2006 and 2024 by UVC along twelve transects at 6-8 reserve and 6-8 non-reserve sites (Fig. 2). At each transect, divers deployed a weighted line and counted all non-cryptic fish within a tunnel-shaped area of 2 × 2 × 30 m (i.e., 120 m^3), identifying individuals to species. Counts were conducted at a consistent swim speed, only when visibility was ≥ 4 m. To reduce diver movements that could potentially attract or repel fish, divers continuously sampled along three consecutive transects without stopping. Common reef fish included marblefish (*Aplodactylus arctidens*), butterfly perch (*Caesioperca lepidoptera*), magpie perch (*Pseudogoniistius nigripes*), red moki (*Cheilodactylus spectabilis*), blue moki (*Latridopsis ciliaris*), leatherjacket (*Meuschenia scaber*), tarakihi (*Nemadactylus macropterus*), spotty (*Notolabrus celidotus*), banded wrasse (*Notolabrus fucicola*), butterfish (*Odax pullus*), blue cod (*Parapercis colias*), scarlet wrasse (*Pseudolabrus miles*), sweep (*Scorpis lineolata*), goatfish (*Upeneichthys lineatus*), and snapper (*Pagrus auratus*). Note that divers saw other non-focal fish species that were not recorded to ensure consistency and ecological relevance, including longfin boarfish (*Zanclistius elevatus*) and copper moki (*Latridopsis forsteri*).

Actual size (total length) of red moki, blue moki, tarakihi, butterfish, snapper and blue cod were measured during UVC surveys between 2020 and 2024 at 8 reserve and non-reserve sites. Fish length was visually estimated by trained divers to the nearest centimetre of fish body length (Davidson et al. 2013). Note size measurements of blue cod were collected in 2017, however they were removed from the dataset and not analysed because they were too coarse (only <30 or 30 cm) to provide meaningful insights.

Kina abundance (density) was measured in 2006 by UVC in 1 x 1 m ($1m^{-2}$) quadrats at 6 reserve and 6 non-reserve sites (n = 30 quadrats per site; Fig. 2). Quadrats were haphazardly placed on suitable rocky reef habitat ranging from 0.5-6 m depth.

Pāua abundance (density) was measured in 2006, 2013 and 2022 by UVC in 1 x 1 m (1 m^2) quadrats at 6 reserve and 6 non-reserve sites (n = 28-30 quadrats $site^{-1}$ in 2006 and 2013 and 60 in 2022; Fig. 2). Quadrats were placed haphazardly on rocky reef at 0-2 m depth. In 2022, maximum shell length (mm) of pāua was measured by divers at each site using callipers. A minimum of 50 pāua were measured at each site. At low density sites divers searched for pāua outside of quadrats as necessary to reach the minimum, and at high density sites only a subset of pāua were measured.

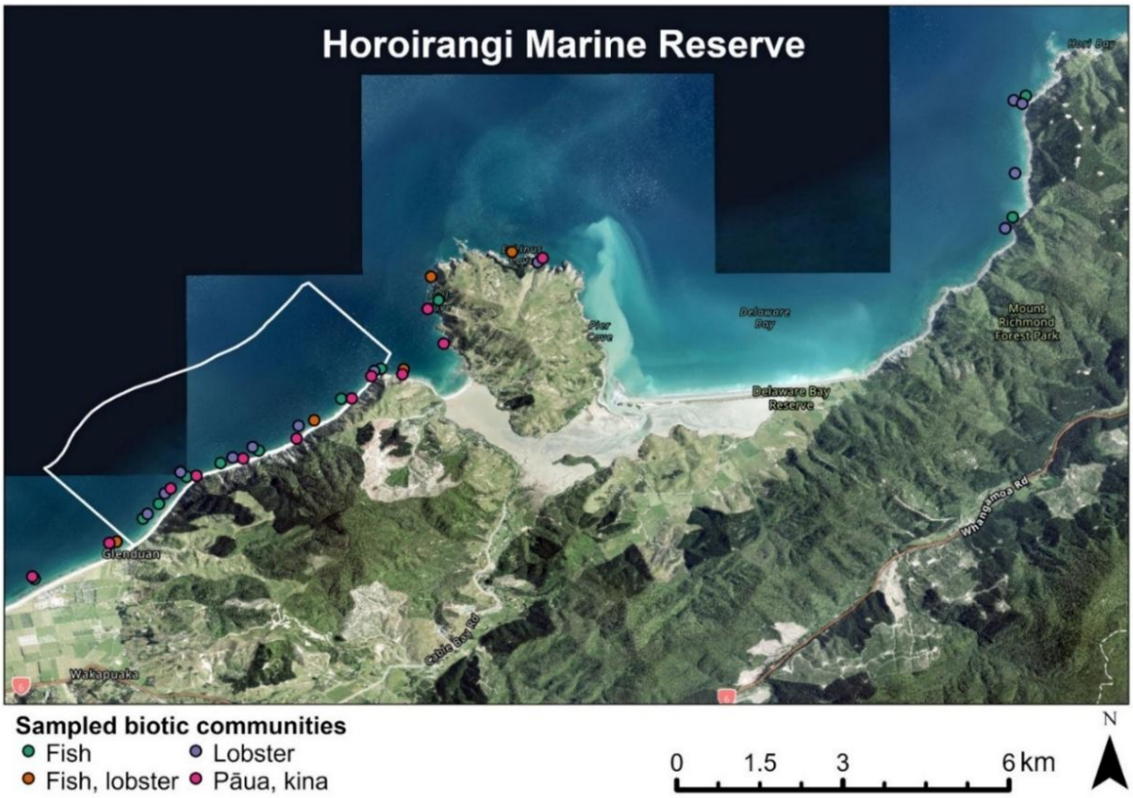

Figure 2. Survey sites inside and outside of the Horoirangi Marine Reserve. Reserve boundaries are shown with white lines. Each dot represents one survey site, and the colour of a dot represents the sampled biotic community at that location.

Statistical methods

Abundance data for rock lobster and pāua were analysed using PERMANOVA with Protection (reserve or non-reserve) and Year as fixed factors (and Sex, for lobsters only) and with Site nested within Protection as a random factor using PRIMER 6 with the PERMANOVA+ add-on (Anderson et al. 2008). These univariate data were analysed using PERMANOVA because they failed to meet the assumptions of ANOVA, even after transforming. Euclidian distance was used to calculate the resemblance matrix. Kina abundance data were analysed using the same model, but using ANOVA in Minitab 17 (Minitab 2014). Based on data visualisation of the rock lobster data, a segmented (i.e., breakpoint) regression using the Segmented package in R (Muggeo 2003) was run for rock lobster reserve and non-reserve data. The purpose of this analysis is to test if any increases in rock lobster abundance resulted in a significant change in the slope of the line. For the segmented regression average abundance (per site) was the response variable while Year was the continuous predictor.

Fish community data (fourth root transformed) were analysed using PERMANOVA following the same design as abundance, except Bray Curtis (with a dummy variable of 0.1) was used to create the resemblance matrix. Significant fixed effects were further investigated using pairwise comparisons and SIMPER analyses.

To analyse size structure of rock lobster, pāua, and fish, discrete size classes were created from shell length, tail width, or total length every 5 or 10 mm (or cm, for fish) between the minimum and maximum recorded sizes. The percentage of individuals in each size class was calculated as a proportion of the total population abundance. The cumulative percentage across each Site x Year combination was then calculated. The cumulative percentage of rock lobster, pāua, or fish in each size class was then analysed using a PERMANOVA with Year and Protection as fixed factors and Site, nested within Protection, as a random factor. The resemblance matrix was created using Euclidian distance. Significant fixed factors were further investigated using pairwise comparisons.

Results

Rock lobster

Rock lobster abundance (pooling over sex) was significantly higher inside than outside ($PseudoF_{1,12}$ = 7.75, p = 0.016) of the Horoirangi Marine Reserve (long-term average ± SE lobsters 100 m^{-2}: 0.5 ± 0.02 outside and 2.1 ± 0.06 inside; Fig. 3). Lobster abundance also increased significantly through time (i.e., year) within the marine reserve ($PseudoF_{14, 138}$ = 2.65, p = 0.002). This significant increase was identified in both the segmented regression (t = 2.11, p = 0.037) and PERMANOVA. At the marine reserve sites, the segmented regression identified two significant increases in rock lobster abundance, one in 2014 and another in 2024. No significant increases were identified for the non-reserve sites. The PERMANOVA similarly showed that rock lobster abundances were significantly lower within the marine reserve before 2010 compared to 2014 – 2025. There was no significant difference between the abundance of males and females ($PseudoF_{1,28}$ = 0.15, p = 0.709).

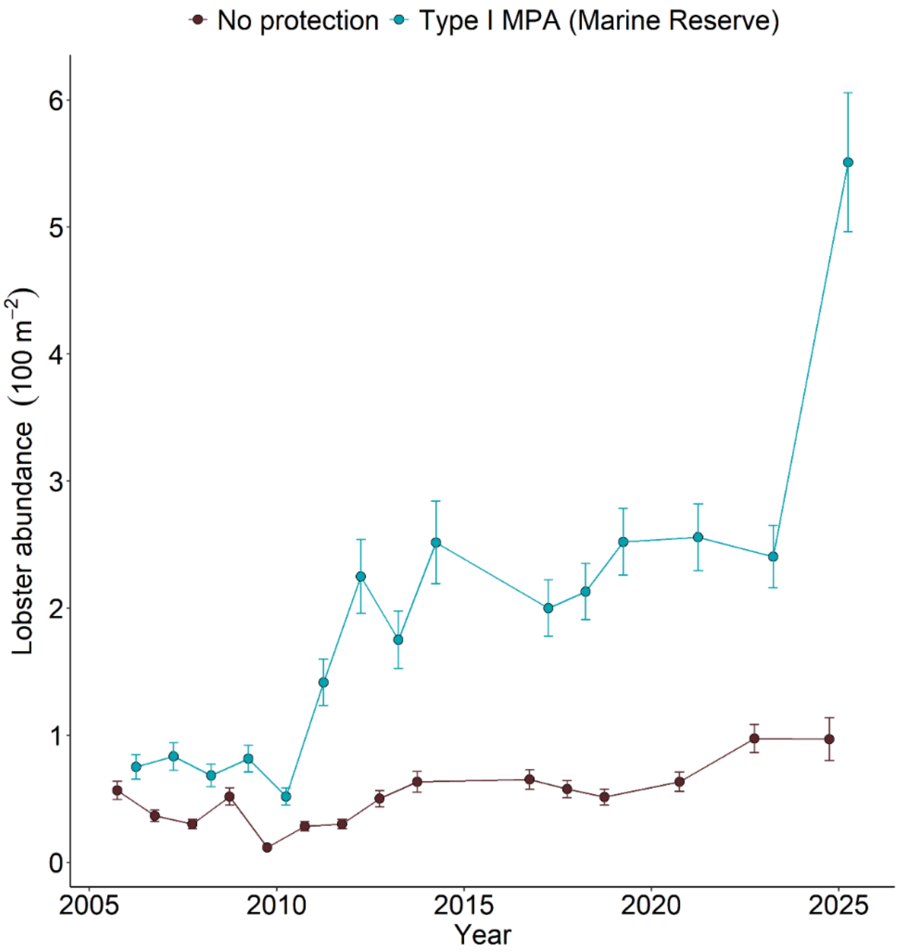

Figure 3. Abundance (mean ± SE 100m^{-2}) of rock lobster (*Jasus edwardsii*) inside and outside of the Horoirangi Marine Reserve from 2006 to 2025. N (the number of sites sampled per year) was 6–8 in the reserve and 4–6 outside of the reserve. Note that reserve and non-reserve sites were sampled at the same time each year but the Type I MPA datapoints have been offset right for visual ease.

On average, lobsters were larger inside (65.0 ± 0.37 mm tail width) than outside (58.0 ± 0.89 mm tail width) of the reserve, however this was not statistically significant (separate sexes: $PseudoF_{1,11}$= 0.75, p = 0.481; pooling over sex: $PseudoF_{1,11}$ = 1.20, p = 0.283; Fig. 4). This may have been due to the overall low number of individuals found in the non-reserve sites increasing the variation in the data.

Average size across all sites was significantly different between sexes ($PseudoF_{1,9}$ = 7.15, p = 0.003), with males being larger (70.5 ± 0.5 mm tail width) than females (65.2 ± 0.4 mm tail width).

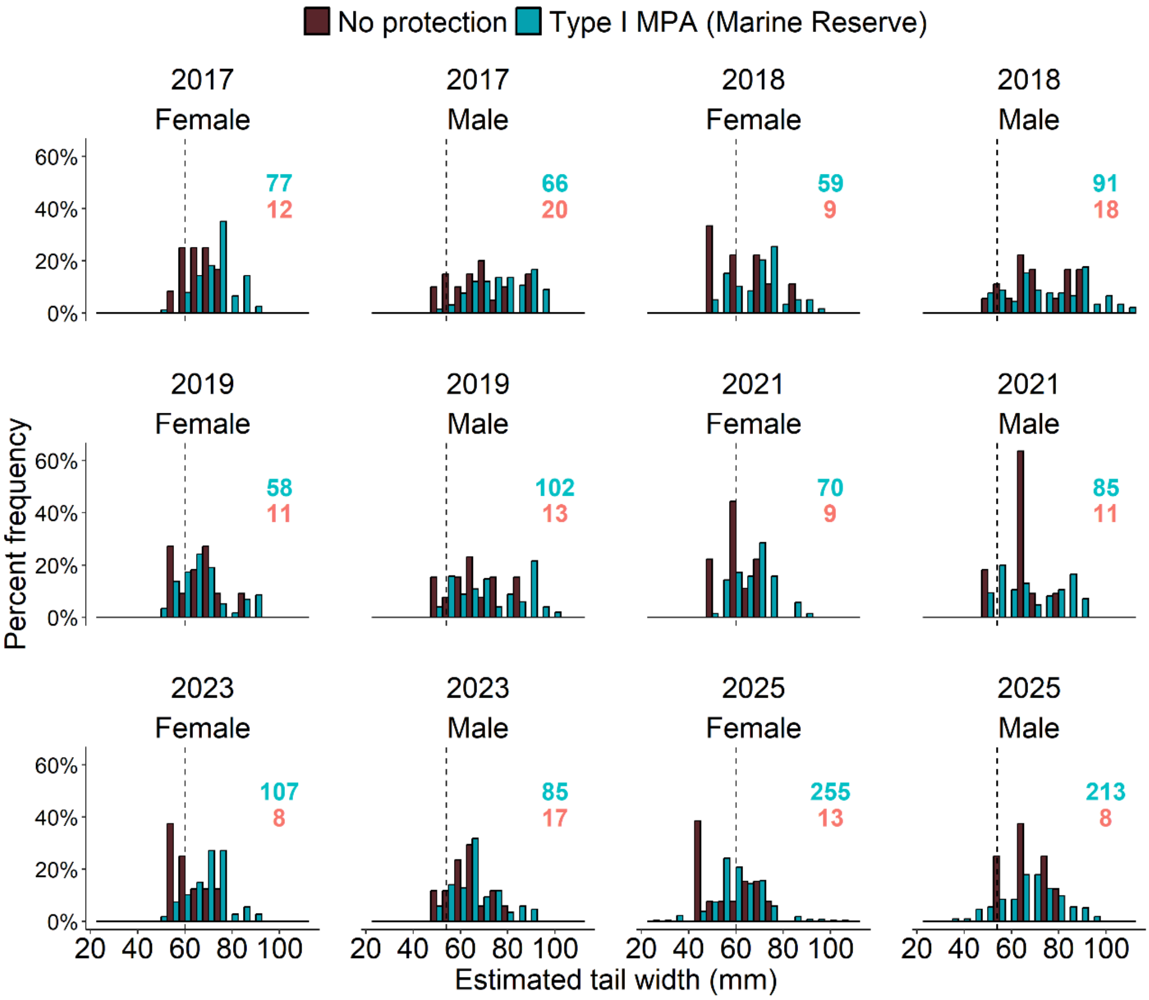

Figure 4. Percent length frequency histograms of male and female rock lobster (*Jasus edwardsii*) sampled inside and outside of the Horoirangi Marine Reserve from 2017 to 2025. The total number of measured individuals within and outside of the marine reserve are presented for each year and sex combination. Dashed lines represent the minimum tail width for legal retention in males and females. N (the number of sites sampled per year) was 6–8 in the reserve and 4–6 outside of the reserve.

Fish communities

Reef fish communities varied significantly between sites inside and outside of the marine reserve over time, as indicated by a significant interaction between Protection and Year in the PERMANOVA ($PseudoF_{14,\ 183}$ = 1.56, p = 0.01; Fig. 5 and 6). Further investigation of this significant interaction revealed that reef fish communities were significantly different between reserve and non-reserve sites in 2007, 2008, and 2014. Based on SIMPER comparisons for these years, differences were mainly due to i) higher abundances of blue cod inside the marine reserve than outside and ii) higher abundances of tarakihi, spotty, marblefish, and goatfish outside than inside of the marine reserve (Appendix 1). Based on the other SIMPER comparisons, in the years where there was no significant difference between reserve and non-reserve sites blue cod, spotty, tarakihi, goatfish, and blue moki were all major contributors to the difference in community composition between sites. Other species that were present but did not typically contribute much (<10%) to the overall dissimilarity between protection types, typically due to overall low abundances and high variability, were leatherjacket, butterfly perch, and banded wrasse.

Focusing on blue cod as an important key species, blue cod abundance increased significantly through time at both reserve and non-reserve sites (significant Year and Protection status interaction; $F_{12,168} = 2.04$, $p = 0.024$). Based on post-hoc comparisons, however, the increase was significantly greater in the reserve (from 0.06 in 2006 to 2.08 individuals 120 m^{-3} in 2022) than outside of the reserve (from 0.06 in 2006 to 1.36 individuals 120 m^{-3} in 2022). In the most recent survey (2024), blue cod abundance decreased significantly at both reserve and non-reserve sites (0.4-0.5 individuals 120 m^{-3}), marking their lowest levels since 2014/2016 (Fig. 7).

Size structure of measured reef fishes (blue cod, tarakihi, red moki) was not significantly different between reserve and non-reserve sites for red moki ($PseudoF_{1,10} = 0.86$, $p = 0.529$; total length: 29 ± 1.9 cm inside and outside of the reserve; Fig. 8). There was however a significant difference in size structure for blue cod and tarakihi between reserve and non-reserve sites, with both being consistently larger inside the marine reserve (blue cod: 29.0 ± 0.4 cm; tarakihi: 14.6 ± 1.0 cm) than outside (blue cod: 24.8 ± 0.4 cm; tarakihi 12.2 ± 0.5 cm) (blue cod: $PseudoF_{1,14} = 11.9$, $p = 0.003$; tarakihi: $PseudoF_{1,13} = 5.8$, $p = 0.013$). Of the blue cod inside the reserve, 42% were larger than the current minimum legal size (33 mm), compared to only 11% outside of the reserve (pooling over years). Tarakihi size was significantly smaller in 2020 than in 2022 or 2024.

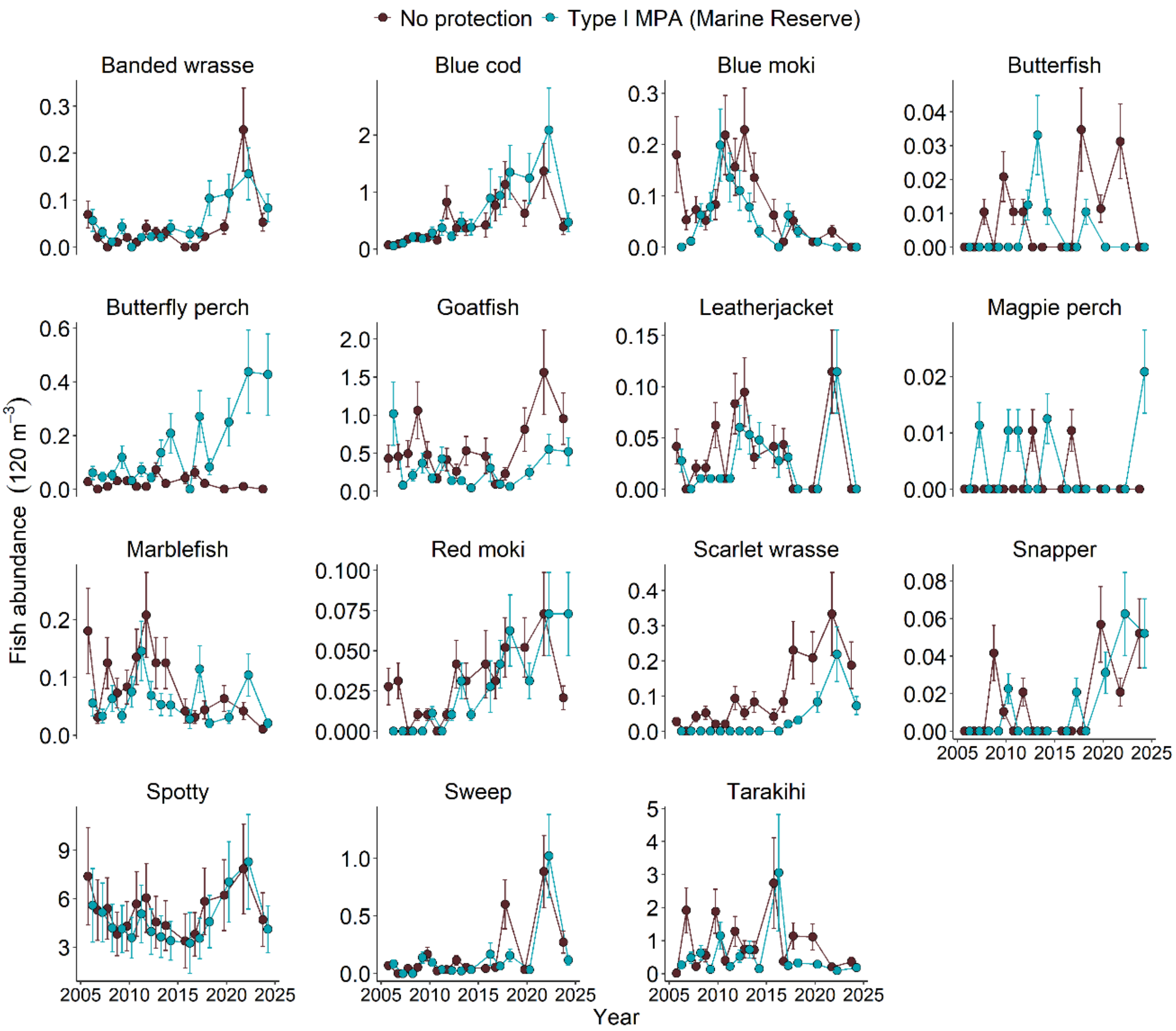

Figure 5. Reef fish abundance (mean ± SE 120m^{-3}) at sites inside and outside of the Horoirangi Marine Reserve from 2006 to 2024. N = 6-8 sites inside and outside of the reserve. Note that reserve and non-reserve sites were sampled at the same time each year, but the Type I MPA datapoints have been offset right for visual ease.

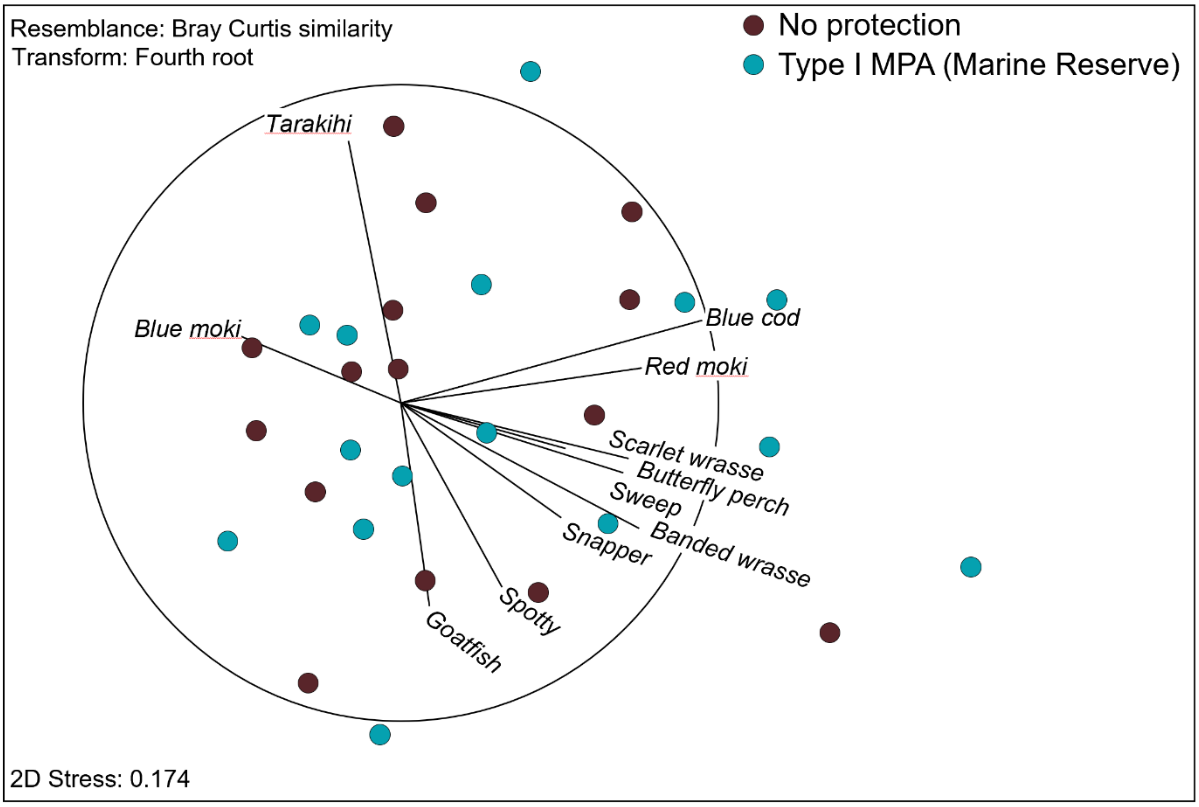

Figure 6. Non-metric Multidimensional Scaling plot (nMDS) for reef fish communities inside and outside of the Horoirangi Marine Reserve from 2006 to 2024. Each dot represents the average community composition within each protection type per year. Note that this averaging was done to decrease 2D stress below 0.2, thereby giving a good representation of multidimensional patterns in 2D space. The vector overlay represents Pearson correlations (r) between a species and nMDS axis. Each line shows the direction of increased density across the plot, while the length of the line relative to the circle represents the strength of that correlation. Only correlations with r > 0.4 are shown.

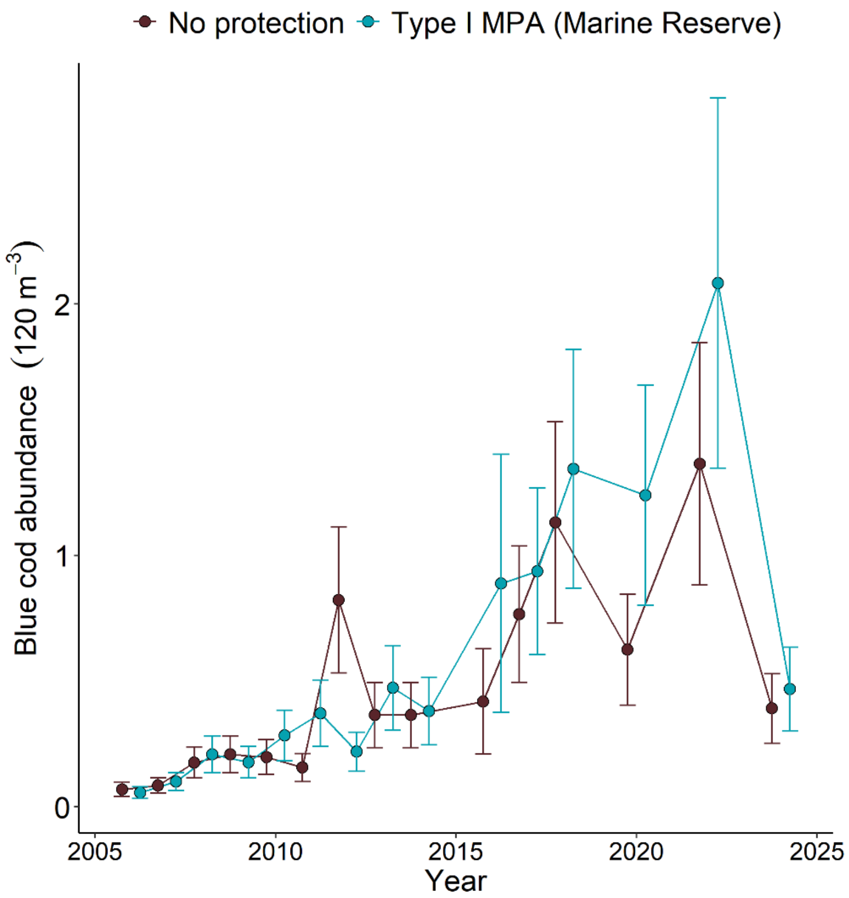

Figure 7. Blue cod abundance (mean ± SE 120m^{-3}) at sites inside and outside of the Horoirangi Marine Reserve from 2006 to 2024. N = 6-8 sites inside and outside of the reserve. Note that reserve and non-reserve sites were sampled at the same time each year but the Type I MPA datapoints have been offset right for visual ease.

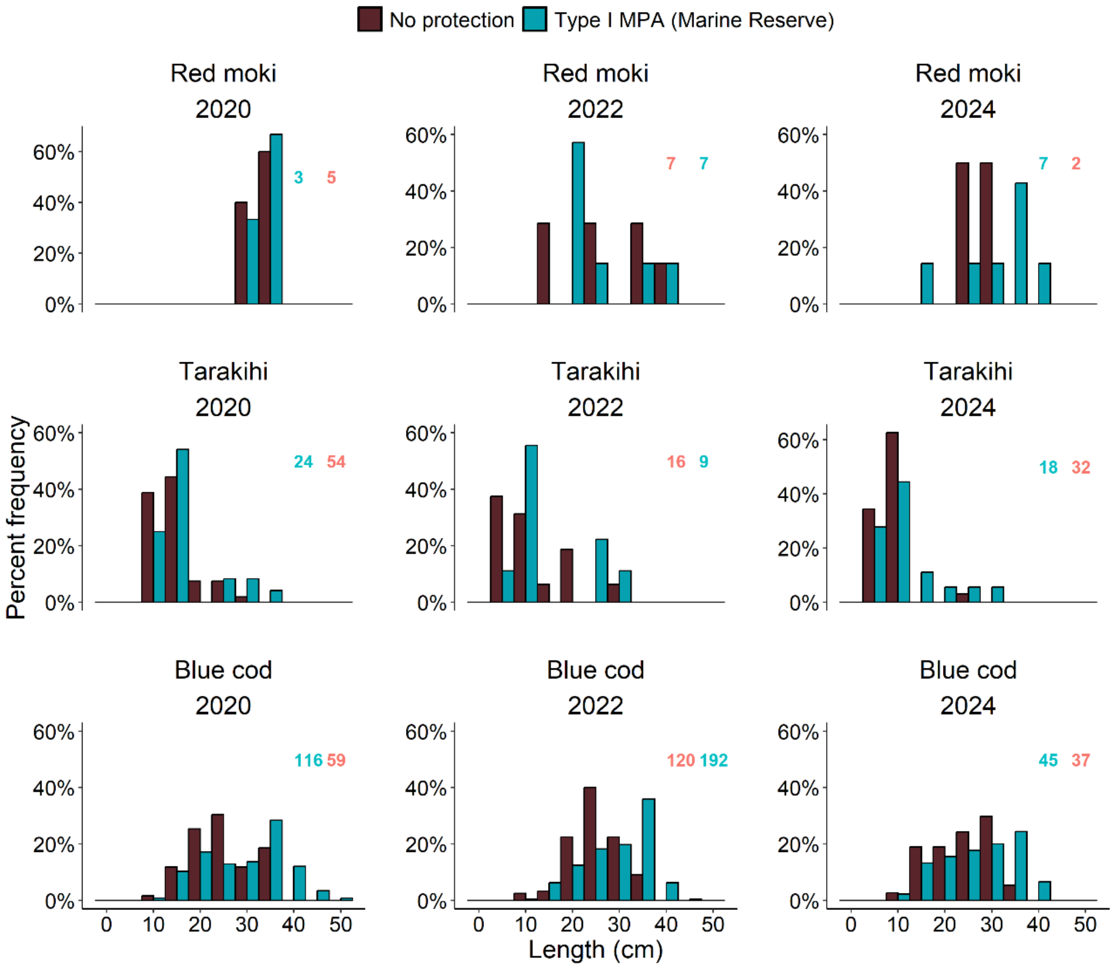

Figure 8. Percent length frequency histograms of reef fish (red moki, tarakihi, and blue cod) sampled inside and outside of the Horoirangi Marine Reserve from 2020 to 2024. The total number of measured individuals within and outside of the marine reserve are presented for each year. N = 8 sites inside and outside of the reserve.

Kina

There was no significant difference in kina abundance inside and outside of the marine reserve in 2006 ($F_{1,10}$ = 0.17, p = 0.688). Kina abundance inside the reserve was 2.9 ± 1.2 (mean ± SE kina m^{-2}) versus 3.10 ± 1.3 outside the reserve (Fig. 9).

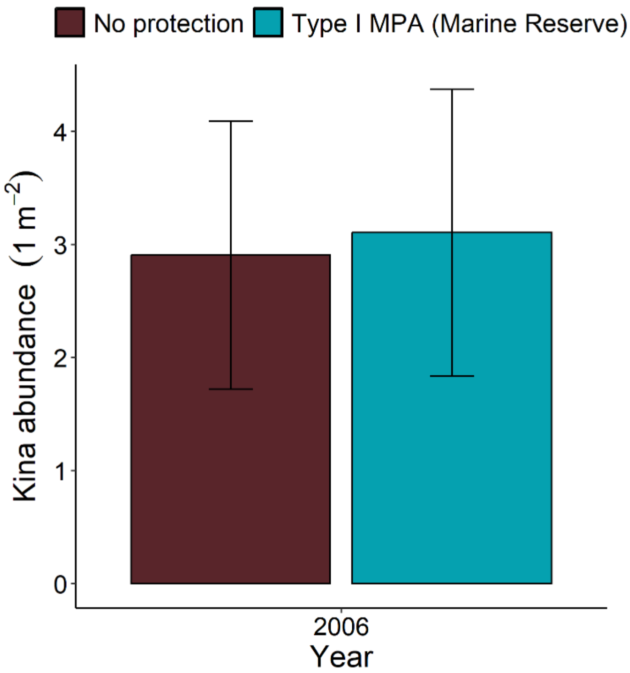

Figure 9. Abundance of kina (*Evechinus chloroticus*; mean ± SE m^{-2}) at sites inside and outside of the Horoirangi Marine Reserve in 2006. N = 6 reserve and 6 non-reserve sites.

Pāua

There was no significant difference in the abundance ($PseudoF_{1,10}$ = 1.8, p = 0.208) or size structure ($PseudoF_{1,9}$ = 2.8, p = 0.067) of pāua between reserve and non-reserve sites (Fig. 10 and 11). There was, however, a significant difference in abundance between years ($PseudoF_{2,20}$ = 12.0, p < 0.001), with 2013 having significantly higher abundance than 2022 and 2006, and 2022 having significantly higher abundance than 2006. Average pāua shell length, which was only measured in 2022, was slightly larger outside of the reserve (77.4 ± 1.0 mm) than inside (66.3 ± 0.7 mm; Fig. 10).

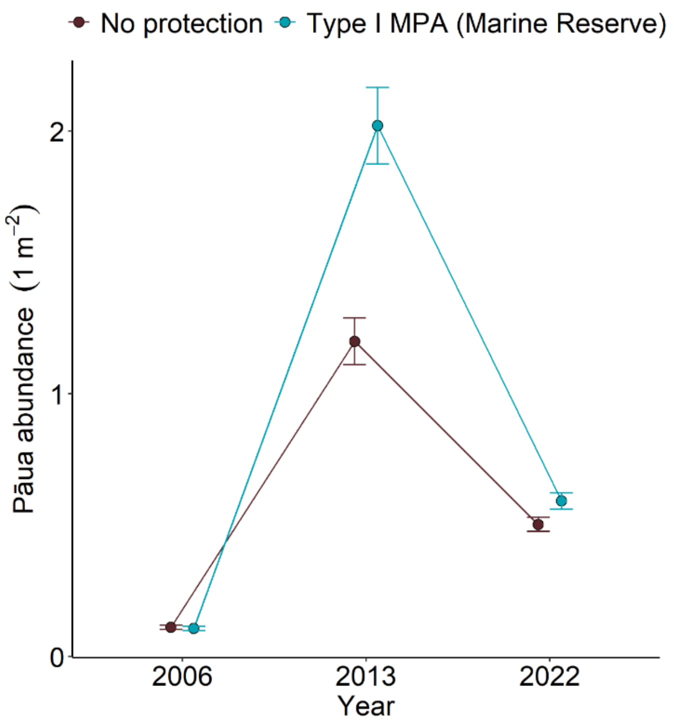

Figure 10. Abundance of pāua (*Haliotis iris*; mean ± SE m^{-2}) at sites inside and outside of the Horoirangi Marine Reserve from 2013 to 2022. N = 6 reserve and non-reserve sites. Note: reserve and non-reserve sites were sampled at the same time each year: Type I MPA datapoints are offset for visual ease.

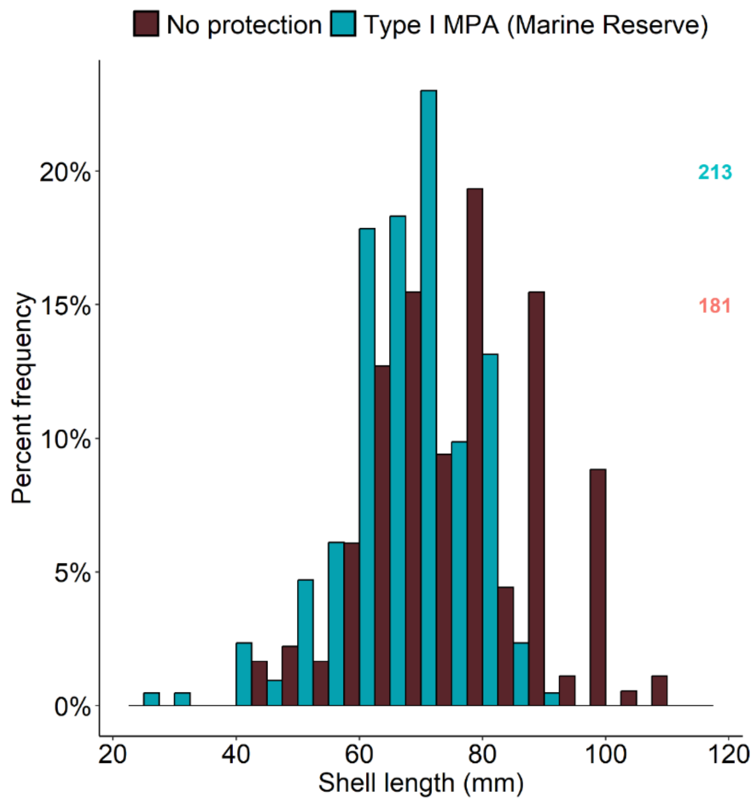

Figure 11. Percent length frequency histogram of pāua (*Haliotis iris*) size inside and outside of the Horoirangi Marine Reserve in 2022. N = 6 reserve and non-reserve sites. The total number of measured individuals within and outside of the marine reserve are presented in the graph. Bin width = 5 mm, i.e., 120-124.9 mm is immediately to the right of 120 mm on the x-axis and 115-19.9 is immediately to the left of 120 on the x axis.

Discussion

Rock lobster

Rock lobster are keystone species that play an important role in the structure and functioning of healthy rocky reef ecosystems, and support high-value and culturally important fisheries (Eddy et al. 2014, 2017). After 20 years of the Horoirangi Marine Reserve being implemented and monitored, the reserve has significantly increased lobster abundance (by 6x), compared to consistently low abundance at non-reserve sites. Lobster abundance within Horoirangi Marine Reserve (~5.5 lobster 100 m^{-2}) was higher than that recorded in nearby Tonga Island Marine Reserve in 2023 after 31 years of protection (~3.3 lobster 100 m^{-2}, Gerrity and Virgin 2026), but lower than Long Island – Kokomohua Marine Reserve in 2014 after 21 years of protection (~14 lobster 100 m^{-2}, Davidson et al. 2014). Other marine reserves such as Taputeranga to the southeast and Kapiti to the north have had similar positive effects on rock lobster abundance (Rojas-Nazar et al. 2019).

Monitoring of other marine reserves in the Nelson/Marlborough region, including Tonga Island and Long-Island Kokomohua, have consistently shown a lag of 5 to 10 years before significant benefits to rock lobster are observed (Davidson et al. 2007, 2009, 2013, Freeman et al. 2012). These increases in rock lobster abundance are likely due to a combination of settling larvae, migration of gregarious juveniles and adults, and reduced fishing pressure (Kelly et al. 1999, 2000, Davidson et al. 2013). Given the significant increase in lobster abundance within the Horoirangi Marine Reserve in 2024, the reserve could be experiencing another permanent surge in lobster abundance (as in 2013/14), potentially indicating that it could support even larger lobster populations. If this trend continues, it has important implications for rocky reefs throughout the Nelson/Tasman region and further afield because rock lobster larvae disperse hundreds to thousands of kilometres from where they are released (Booth 1994). Spawning aggregations within the Horoirangi Marine Reserve may therefore contribute to recruitment in non-reserve areas, both locally and at great distances from the reserve. Although larval export from marine reserves is not well quantified, Kelly et al. (2000) estimated that rock lobster egg production increases by 4.8–9.1% per year of protection in New Zealand marine reserves. Marine reserves can therefore act as source populations for rock lobsters, although the spatial scale and extent of such subsidies are not well understood.

Fish communities

Research divers recorded 15 focal fish species inside and outside the Horoirangi Marine Reserve over the period of monitoring, in addition to observing numerous other species that were not recorded. Overall fish abundances were low and highly variable, with no consistent spatial or temporal patterns for most species. Higher fish counts often had high variance, indicating that schools were encountered sporadically at only one or a few sites. For many species, observations were too few to support confident trend interpretation. Red moki, for example, seem to have increased in abundance and size within the reserve, but not significantly. Interestingly, butterfly perch showed a significant positive response to protection despite not being a commonly targeted species. Snapper abundance increased over time at both reserve and non-reserve sites. Snapper are a known beneficiary of marine reserves and are highly mobile, so the reserve may be having flow-on effects to non-reserve habitats (Cole et al. 1990, Denny et al. 2004).

Efforts were made to minimise sampling bias during fish counts, including the continual sampling of consecutive transects in order to reduce behavioural effects on diver-attracted fish. Even so, it is likely that UVC methods imparted at least some bias, particularly for fish that may be attracted to or aversive to divers (e.g. blue cod vs. snapper). Addition of complimentary methods such as baited remote underwater video may provide alternative abundance estimates that can be considered in tandem with UVC data.

Consistent with findings from other reserve studies, average blue cod size increased following protection (e.g., Pande et al. 2008, Davidson 2009, Gerrity and Virgin 2026). Throughout the monitoring, blue cod were consistently larger in Horoirangi, with about 40 percent of sampled fish exceeding the current legal size compared with only 11 percent at non-reserve sites. This pattern is expected given the known effects of fishing on blue cod population structure and the intensive fishing pressure in the Nelson region (Beentjes 2023). Increased size structure within marine reserves has been reported previously for blue cod in northeastern New Zealand and is generally interpreted as evidence of reduced fishing pressure and increased survivorship of older individuals (Pande et al. 2008). Blue cod are categorised as a low-productivity species that are vulnerable to fishing impacts on population structure and local depletion (Carbines 1999, Beentjes 2023), and they are therefore common beneficiaries of spatial closures.

Surprisingly, blue cod abundance in Horoirangi did not differ significantly between reserve and non-reserve sites. Abundance increased steadily at reserve and non-reserve sites for about 15 years, potentially reflecting more stringent regional fishing regulations and catch reductions, before an abrupt decline was detected in the 2024 survey when abundance reached a decadal low. It is unclear whether this decline represents an ecological trend, sampling error, or an aberration, but future surveys or alternative sampling methods should help clarify the pattern.

A potential source of variability in the fish data, particularly at non-reserve sites, is the changing of fishing regulations in the Tasman/Nelson region. These include major changes to blue cod bag limits and seasons, TACC reductions, and the 2020 ban on commercial and recreational set nets out to 4 nautical miles along the coastline (Ministry of Primary Industries 2020). Such regulatory shifts may account for some observed trends in fish abundance, such as the increase in red moki after 2020. These factors should be noted when interpreting reserve effects.

Kina and pāua

Kina were only surveyed in 2006, so we cannot interpret temporal trends or reserve effects. At that time kina were relatively abundant (~ 3 m^{-2}) compared to recent abundances at nearby marine reserves including Tonga Island (<1.5 m^{-2} in 2018) and Hikurangi (~0.5 m^{-2} in 2024) (Gerrity et al. 2025, Gerrity and Virgin 2026). An updated survey of kina populations scheduled for 2026 will allow for analysis of temporal changes in kina abundance and potential reserve effects.

Pāua populations were similar in size structure and abundance across reserve and non-reserve sites, although in 2013 the abundance was significantly higher in the reserve. Pāua were generally small with none of the 394 individuals measured reaching the minimum legal size for harvest. Pāua growth is known to be variable in this region with some populations exhibiting limited or stunted growth, restricting recreational and commercial pāua harvest (Neubauer 2023). The limited pāua growth inside the reserve may reflect drivers of pāua growth unassessed in this monitoring, e.g. environmental conditions (Naylor et al. 2007).

General discussion and recommendations

This investigation demonstrates that after almost 20 years of protection, the Horoirangi Marine Reserve has had positive effects for several key reef species, particularly rock lobster and blue cod. These findings are consistent with other New Zealand marine reserve studies, where rock lobster and blue cod are among the most consistent beneficiaries of protection (Cole et al. 1990, Davidson et al. 2009, Pande et al. 2008, Rojas-Nazar et al. 2019, Gerrity et al. 2025, Gerrity and Virgin 2026).

Reserve effects were not consistent across all species. Many fish species showed high spatial and temporal variability, making trends difficult to interpret. This variability likely reflects a combination of patchy species distributions, environmental variability, and limitations associated with UVC methods. Similar issues have been documented in other studies, where reserve effects were evident but difficult to detect statistically because of high variance and limited replication (Cole et al. 1990). For some species, low encounter rates and high variance reduced the ability to detect significant reserve effects despite apparent differences between reserve and non-reserve sites.

This report also highlights the importance of long-term monitoring. Positive responses for rock lobster only became clearly evident after approximately 6–10 years of protection, consistent with recovery trajectories observed in nearby marine reserves such as Tonga Island and Long Island–Kokomohua (Davidson et al. 2007, Davidson et al. 2009, Freeman et al. 2012). Recovery trajectories within marine reserves are influenced by a range of interacting factors including recruitment dynamics, larval supply, habitat quality, environmental conditions, and fishing pressure outside reserve boundaries (Pande et al. 2008).

Interpreting reserve effects was further complicated by broader fisheries management changes during the monitoring period, including changes to blue cod regulations, catch limits, and set-net restrictions in the Nelson/Tasman region (Ministry for Primary Industries 2020, Fisheries New Zealand 2022). It is uncertain whether wider management changes may have contributed to some of the observed trends in fish abundance both inside and outside the reserve.

This report was based entirely on UVC surveys, which are widely used for monitoring reef communities but have recognised limitations including observer bias, disturbance effects on mobile species, and reduced detectability of cryptic or nocturnal taxa (Brock 1982, Lincoln-Smith 1988, 1989, Cheal and Thompson 1997). While UVC methods are effective for conspicuous reef fish and benthic invertebrates such as pāua and kina, complementary survey methods may improve the ability to characterise reserve effects across a wider range of species.

Continued long-term monitoring at fixed sites is recommended, with surveys repeated every two to three years using consistent methods to strengthen the detection of temporal trends and reserve effects. Standardising sampling effort, survey area and replication among years would improve comparability and reduce variance, particularly for patchily distributed species. Incorporating complementary methods such as baited remote underwater videos and potting surveys would enhance sampling of mobile, nocturnal or diver-responsive species, including snapper, blue cod and rock lobster, and aligns with recent successful applications in other New Zealand marine reserves (Gerrity et al. 2025).

Monitoring could also be expanded to consider the use of rock lobster puerulus collectors to quantify larval abundance in the reserve. This would help distinguish between larval recruitment and adult migration as drivers of population change (Kelly et al. 2000, Freeman et al. 2012). Also, adding habitat-level measurements such as macroalgal composition, sediment cover and substrate type would provide essential ecological context for interpreting trends in fish and invertebrate communities and improve understanding of broader ecosystem-level reserve effects.

Conclusion

The Horoirangi Marine Reserve remains an important scientific reference site for understanding the long-term effects of spatial protection in northern South Island reef ecosystems. Continued monitoring using consistent methods, alongside the incorporation of complementary survey approaches and broader ecosystem indicators, will improve understanding of reserve performance and support future marine conservation and fisheries management decisions.

Author contributions

SG: Writing – first draft, review and editing. SV: Writing – first draft, review and editing; formal analysis.

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Data availability statement

Data may be available on request to DOC.

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Appendix

Appendix 1. SIMPER (similarity percentages) for reef fish communities at sites inside and outside of the Horirangi marine reserve from 2006 to 2024. Years with significant differences between reserve and non-reserve sites are indicated in bold.

Year	Common name	Scientific name	Abundance (4th root)		Average dissimilarity	Ratio (diss. / SD)	Percent contribution
			No protection	Type I MPA			
2006	Goatfish	*Upeneichthys lineatus*	0.31	0.56	12.05	1	36.16
	Spotty	*Notolabrus celidotus*	1.57	1.48	6.97	1.17	20.92
	Marblefish	*Aplodactylus arctidens*	0.13	0.04	3.28	0.42	9.84
	Blue cod	*Parapercis colias*	0.07	0.06	2.28	0.36	6.83
	Tarakihi	*Nemadactylus macropterus*	0.01	0.11	2.25	0.34	6.76
	Banded wrasse	*Notolabrus fucicola*	0.05	0.06	1.64	0.32	4.92
2007	Tarakihi	*Nemadactylus macropterus*	0.46	0.15	10.64	0.77	31.09
	Spotty	*Notolabrus celidotus*	1.42	1.42	8.83	0.96	25.78
	Goatfish	*Upeneichthys lineatus*	0.26	0.07	6.2	0.58	18.1
	Blue cod	*Parapercis colias*	0.07	0.09	2.93	0.39	8.56
2008	Tarakihi	*Nemadactylus macropterus*	0.21	0.47	10.32	0.89	28.64
	Goatfish	*Upeneichthys lineatus*	0.23	0.19	6.73	0.65	18.67
	Spotty	*Notolabrus celidotus*	1.48	1.37	6.33	0.76	17.56
	Blue cod	*Parapercis colias*	0.11	0.18	4.77	0.54	13.24
	Marblefish	*Aplodactylus arctidens*	0.11	0.05	2.67	0.41	7.41
	Blue moki	*Latridopsis ciliaris*	0.06	0.04	1.66	0.29	4.62
2009	Goatfish	*Upeneichthys lineatus*	0.51	0.28	11.17	0.93	30.13
	Spotty	*Notolabrus celidotus*	1.33	1.38	6.06	1.22	16.35
	Blue cod	*Parapercis colias*	0.18	0.16	5.41	0.59	14.59
	Tarakihi	*Nemadactylus macropterus*	0.21	0.11	5.21	0.54	14.05
	Marblefish	*Aplodactylus arctidens*	0.07	0.03	2.05	0.32	5.54
	Blue moki	*Latridopsis ciliaris*	0.05	0.04	1.53	0.29	4.14
2010	Tarakihi	*Nemadactylus macropterus*	0.36	0.14	7.46	0.61	18.99
	Spotty	*Notolabrus celidotus*	1.36	1.32	6.54	0.79	16.66
	Blue cod	*Parapercis colias*	0.16	0.21	6.22	0.58	15.83
	Goatfish	*Upeneichthys lineatus*	0.24	0.17	6.17	0.65	15.71
	Blue moki	*Latridopsis ciliaris*	0.07	0.12	3.29	0.43	8.38
	Marblefish	*Aplodactylus arctidens*	0.08	0.08	2.58	0.39	6.57
	Sweep	*Scorpis lineolata*	0.06	0.08	2.23	0.35	5.67
2011	Spotty	*Notolabrus celidotus*	1.41	1.43	7.67	0.8	19.29
	Goatfish	*Upeneichthys lineatus*	0.13	0.29	6.9	0.66	17.35
	Tarakihi	*Nemadactylus macropterus*	0.24	0.19	6.68	0.68	16.78
	Blue cod	*Parapercis colias*	0.11	0.27	6.6	0.62	16.58
	Marblefish	*Aplodactylus arctidens*	0.14	0.14	4.38	0.53	11

	Blue moki	*Latridopsis ciliaris*	0.13	0.1	3.7	0.46	9.3
2012	Tarakihi	*Nemadactylus macropterus*	0.61	0.39	11.27	1	27.13
	Blue cod	*Parapercis colias*	0.39	0.2	7.28	0.78	17.52
	Spotty	*Notolabrus celidotus*	1.46	1.35	6.35	0.9	15.3
	Goatfish	*Upeneichthys lineatus*	0.24	0.14	4.68	0.61	11.27
	Marblefish	*Aplodactylus arctidens*	0.18	0.06	3.2	0.5	7.7
	Blue moki	*Latridopsis ciliaris*	0.12	0.08	2.74	0.43	6.6
	Leatherjacket	*Meuschenia scaber*	0.07	0.03	1.73	0.3	4.17
2013	Tarakihi	*Nemadactylus macropterus*	0.43	0.33	9.73	0.87	22.6
	Blue cod	*Parapercis colias*	0.26	0.37	9.01	0.79	20.92
	Spotty	*Notolabrus celidotus*	1.4	1.29	6.9	0.84	16.03
	Goatfish	*Upeneichthys lineatus*	0.19	0.14	4.85	0.57	11.28
	Marblefish	*Aplodactylus arctidens*	0.13	0.05	2.57	0.43	5.97
	Leatherjacket	*Meuschenia scaber*	0.09	0.05	2.31	0.37	5.36
	Blue moki	*Latridopsis ciliaris*	0.12	0.04	2.18	0.36	5.07
2014	Blue cod	*Parapercis colias*	0.24	0.29	8.43	0.73	20.85
	Tarakihi	*Nemadactylus macropterus*	0.38	0.14	8.04	0.73	19.86
	Spotty	*Notolabrus celidotus*	1.35	1.3	7.48	0.85	18.5
	Goatfish	*Upeneichthys lineatus*	0.27	0.03	5.48	0.54	13.54
	Marblefish	*Aplodactylus arctidens*	0.12	0.05	2.72	0.41	6.72
	Blue moki	*Latridopsis ciliaris*	0.08	0.02	1.74	0.31	4.3
	Butterfly perch	*Caesioperca lepidoptera*	0.02	0.08	1.63	0.28	4.04
2016	Tarakihi	*Nemadactylus macropterus*	0.79	0.71	14.52	1.1	33.71
	Blue cod	*Parapercis colias*	0.34	0.52	10.48	0.92	24.34
	Spotty	*Notolabrus celidotus*	1.25	1.28	6.68	0.73	15.51
	Goatfish	*Upeneichthys lineatus*	0.25	0.1	4.84	0.57	11.24
2017	Blue cod	*Parapercis colias*	0.5	0.64	12.69	1.01	31.87
	Spotty	*Notolabrus celidotus*	1.32	1.31	6.78	0.79	17.02
	Tarakihi	*Nemadactylus macropterus*	0.2	0.19	5.53	0.61	13.88
	Goatfish	*Upeneichthys lineatus*	0.1	0.07	2.46	0.4	6.18
	Butterfly perch	*Caesioperca lepidoptera*	0.03	0.12	2.27	0.33	5.7
	Marblefish	*Aplodactylus arctidens*	0.03	0.09	1.86	0.33	4.68
	Scarlet wrasse	*Pseudolabrus miles*	0.09	0.02	1.63	0.33	4.1
2018	Blue cod	*Parapercis colias*	0.61	0.78	11.84	1.05	27.24
	Tarakihi	*Nemadactylus macropterus*	0.47	0.24	8.36	0.86	19.24
	Spotty	*Notolabrus celidotus*	1.4	1.37	7.65	0.86	17.6
	Sweep	*Scorpis lineolata*	0.19	0.07	3.51	0.44	8.08
	Goatfish	*Upeneichthys lineatus*	0.15	0.05	2.78	0.43	6.39
	Scarlet wrasse	*Pseudolabrus miles*	0.15	0.03	2.41	0.41	5.55
2020	Blue cod	*Parapercis colias*	0.44	0.73	10.64	1.08	25.39
	Tarakihi	*Nemadactylus macropterus*	0.44	0.22	7.61	0.84	18.18

	Goatfish	*Upeneichthys lineatus*	0.37	0.21	6.58	0.78	15.71
	Spotty	*Notolabrus celidotus*	1.5	1.58	5.24	0.91	12.51
	Scarlet wrasse	*Pseudolabrus miles*	0.14	0.08	2.91	0.46	6.95
	Banded wrasse	*Notolabrus fucicola*	0.04	0.11	2.33	0.4	5.56
2022	Blue cod	*Parapercis colias*	0.65	0.91	9.91	1.07	22.25
	Goatfish	*Upeneichthys lineatus*	0.56	0.31	7.74	0.94	17.38
	Spotty	*Notolabrus celidotus*	1.57	1.64	5.17	0.82	11.6
	Sweep	*Scorpis lineolata*	0.24	0.29	5.04	0.67	11.32
	Scarlet wrasse	*Pseudolabrus miles*	0.18	0.2	3.82	0.62	8.57
	Banded wrasse	*Notolabrus fucicola*	0.15	0.14	2.9	0.52	6.52
	Tarakihi	*Nemadactylus macropterus*	0.13	0.09	2.22	0.46	4.98
	Leatherjacket	*Meuschenia scaber*	0.06	0.11	1.78	0.38	4
2024	Goatfish	*Upeneichthys lineatus*	0.51	0.32	9.93	0.92	22.99
	Blue cod	*Parapercis colias*	0.26	0.37	8.21	0.81	19.02
	Spotty	*Notolabrus celidotus*	1.37	1.38	6.36	0.8	14.73
	Tarakihi	*Nemadactylus macropterus*	0.28	0.16	5.89	0.67	13.64
	Scarlet wrasse	*Pseudolabrus miles*	0.14	0.07	3.01	0.45	6.97
	Sweep	*Scorpis lineolata*	0.11	0.09	2.56	0.41	5.94
	Butterfly perch	*Caesioperca lepidoptera*	0	0.15	2.26	0.34	5.24