Tonga Island Marine Reserve

Assessing the effects of protection on key subtidal reef species

1993 - 2024

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Cover: Department of Conservation research vessel in the Tonga Island Marine Reserve. *Photo: Leon Everett*

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Abstract

The Tonga Island Marine Reserve is an 1835-hectare no-take marine reserve located in the Nelson/Tasman region of the northern South Island. It was established in 1993 to protect a diverse array of marine habitats and biodiversity. It is now one of the most visited marine reserves in New Zealand due to its location alongside Abel Tasman National Park.

Underwater Visual Census surveys by research divers sampled abundances and sizes of select key species across 13 coastal sites within the reserve and 15 nearby non-reserve sites semi-regularly between 1993 and 2024. Key species included rock lobster, fish communities, kina, pāua, horse mussels, and scallops. Data were compiled and analysed to test for spatial and temporal differences between reserve and non-reserve sites.

After ~31 years of protection, the reserve has had moderate effects on some key species relative to non-reserve sites. Rock lobster abundance was variable but consistently higher inside the reserve over time, with an 8x increase during the first decade of protection. However, abundance declined in recent years and was not significantly different to non-reserve sites in 2023. Rock lobsters were significantly larger in the reserve.

Reef fish communities varied significantly through time and between reserve and non-reserve sites. Differences were driven by higher abundances of blue cod, scarlet wrasse, tarakihi and butterfly perch inside the reserve, and higher abundances of spotted wrasse outside. Blue cod were consistently larger inside the reserve.

Kina were found in low abundances (< 1.5 per m^2) at reserve and non-reserve sites, and abundance was consistent through time. Pāua were all sub-legal size (< 125 mm) and were more abundant outside the reserve, although not significantly so. Pāua abundance at non-reserve sites decreased by ~69% over time across non-reserve sites, but was stable at reserve sites. There were no clear reserve effects on horse mussels. Scallops were more abundant at non-reserve sites, although these analyses were inconclusive due to low statistical power.

Overall, the Tonga Island Marine Reserve has had positive effects on the size of rock lobster and blue cod, but additional reserve effects are unclear. The reserve may have protected pāua, which appear to have been depleted at non-reserve sites. The lack of reserve effects for scallops is unsurprising given the closure of the regional scallop fishery in 2017.

Data showed high site-to-site variability and uneven survey replication through time, particularly in early years, making some conclusions difficult. Future monitoring would benefit from regular sampling of fixed sites and the addition of alternative survey methods such as baited underwater video and potting surveys for mobile and nocturnally active species. Underwater Visual Census surveys remain necessary for sedentary species such as kina, pāua, horse mussels, and scallops.

This report provides information on the size and abundance of key species in the Tonga Island Marine Reserve, and presents quantitative time series datasets for ongoing monitoring of long-term marine reserve effects.

Keywords

Marine reserve, reserve effects, key species, pāua, rock lobster, reef fish, kina, scallops, horse mussels.

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 Tonga Island Marine Reserve was gazetted in 1993 to protect a unique marine ecosystem, promote biodiversity recovery, and serve as a scientific reference site for long-term ecological monitoring. The reserve covers 1,835 hectares of seabed and foreshore along the northern coast of New Zealand's South Island and extends one nautical mile offshore to a maximum depth of ~30 m (Fig. 1). It is situated along the coastal portion of the Abel Tasman National Park, which has an estimated 300,000 annual visitors, and is one of the most visited marine reserves in New Zealand.

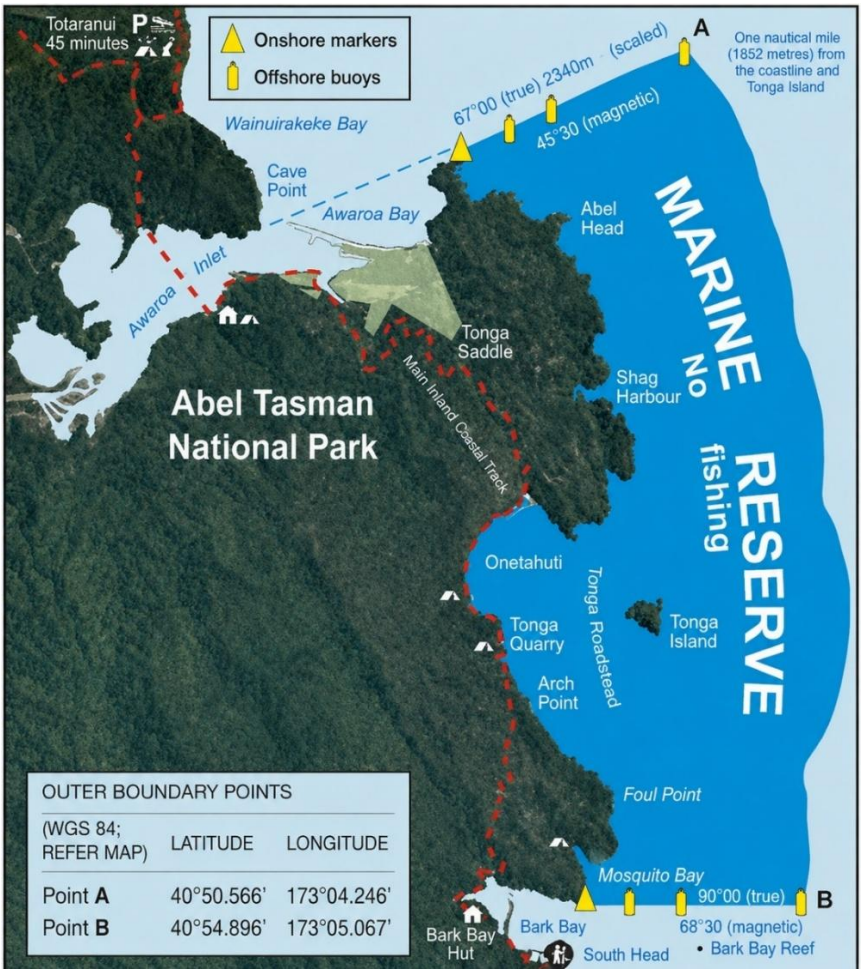

Figure 1. Map of Tonga Island Marine Reserve boundary (blue polygon). Sourced from the Department of Conservation.

The substrate of the Tonga Island Marine Reserve is composed of rocky granitic reef extending from the shallow subtidal zone to depths of roughly 4-14 m, where it transitions to gently sloping soft-sediment seafloors (Davidson and Richards 2013). The shallow soft-sediment zone is composed

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

mainly of broken shell and coarse sand, grading with depth into finer sands, silts, and clays (Davidson and Richards 2013). The reserve is protected from large oceanic swells but is subject to significant wind wave action. The sea temperature within the reserve ranges from 13˚C in winter to 19˚C in summer (NIWA 2026).

Monitoring of subtidal rocky reef and soft-sediment habitats within the Tonga Island Marine Reserve and adjacent coastal areas has been carried out by research divers semi-regularly since 1993. Surveys assessed the abundance and size structure of several key marine species, including rock lobster, reef fish, kina, pāua, horse mussels and scallops. A 2013 analysis of the data by Davidson and Richards showed that after 20 years of protection there were significant reserve effects on several key species (Davidson and Richards 2013). Rock lobster abundance and size were greater in the reserve, as were the sizes of blue cod and blue moki (Davidson and Richards 2013). Pāua abundance was low in the reserve compared to non-reserve sites, and pāua were generally of sublegal size (< 125 mm, Davidson and Richards 2013). Kina abundance declined in the reserve and increased at non-reserve controls (Davidson and Richards 2013). Interestingly, the scallop and horse mussel populations inexplicably declined in the reserve around 2008 (Davidson and Richards 2013). Since the 2013 assessment, monitoring has continued at fixed sites, producing a ~31-year time series that describes long-term trends in key species' abundance and size from which reserve effects can be evaluated.

The Marine Monitoring and Reporting Framework (2022) provides DOC's approach to monitoring marine reserves, outlining 10 themes as main monitoring objectives. Theme 4 addresses the importance of monitoring key species, which are defined in the framework 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)".*

This report provides an updated analysis and interpretation of the key species monitoring dataset from 1993 – 2024 within the Tonga Island Marine Reserve and nearby non-reserve sites, and tests for reserve effects on key species over this 31-year period. The two research questions are:

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

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

This report provides a concise assessment of the Tonga Island Marine Reserve relative to key species and offers recommendations to optimise future monitoring efforts.

Methods

Survey overview

Between 1993 and 2024, rock lobster (kōura, *Jasus edwardsii*), reef fish communities, kina (*Evechinus chloroticus*), pāua (*Haliotis iris*), scallops (*Pecten novaezelandiae*) and horse mussels (*Atrina zelandica*) were sampled inside and outside of the Tonga Island Marine Reserve using Underwater Visual Census (UVC, Table 1). UVC surveys are diver-based methods used to estimate the abundance, size structure, and composition of fish and invertebrate communities. Divers swim along fixed transects and count and/or measure target organisms within a defined area. UVC is widely used in New Zealand marine monitoring because it provides repeatable, non-destructive data on subtidal reef communities (Cole et al. 1990).

Table 1. The Horoirangi Marine Reserve monitoring schedule including Underwater Visual Census (UVC) survey events (grey fill) per key species monitored from 1993-2024. 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	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024
UVC Rock Lobster						S		S																	S		S		S		S	
UVC Fish																												S		S		S
UVC Kina	S																															
UVC Pāua																														S		
UVC Scallops																											S					
UVC Horse Mussel																																

Rock lobster abundance (density) and size (carapace length) were surveyed between 1994 and 2023 by UVC methods adapted from previous marine-reserve studies (e.g., Freeman et al. 2012, Davidson et al. 2007). Transects 25 m in length were haphazardly placed within suitable rocky-reef habitat, avoiding soft sediment where possible, and divers recorded the abundance, sex, and carapace length of all lobsters visible within a 4 m-wide band (total sample area 100 m²). Because lobsters could not be handled, size was estimated *in situ* using a ruler placed along the carapace. Transects ranged from 4-11 m depth, with 5-13 transects surveyed per site at 5-7 reserve sites and 5-12 transects per site at 4-7 non-reserve sites depending on the year (Fig. 2). Surveys were conducted only when underwater visibility exceeded 2.5 m (Davidson et al. 2007). 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.

Reef fish abundance (density) was surveyed between 1993 and 2024 by UVC along twelve 30 m long transects at 2–7 reserve and 2–6 non-reserve sites (Fig. 2). Transects were haphazardly established parallel to shore on rocky reef habitat and ranged from 5 to 12 m in depth (Davidson et al. 2007). At each transect, divers deployed a weighted line and counted all non-cryptic fish within a 2 × 2 × 30 m tunnel (sample area of 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*), and goatfish (*Upeneichthys lineatus*). Note that divers saw other non-focal fish species that were not recorded to ensure consistency and ecological relevance.

Actual size (total length) of red moki, tarakihi, and blue cod were estimated to the nearest centimetre by divers during UVC surveys between 2020 and 2024 at 7 reserve and 6 non-reserve sites. Fish length was visually estimated by trained divers to the nearest centimetre of fish body length (Davidson et. al 2013). Earlier size measurements (from 1994 – 2017) were removed from the dataset and not analysed because they were too coarse to provide meaningful insights. For example, lengths of 10-30 cm, >30 cm, and <30 cm, were not analysed here. A minimum visibility of 4.5 m was required for these surveys to be undertaken.

Kina abundance (density) was measured between 2002 and 2018 by UVC in 1 x 1 m (1 m^2) quadrats at 4 reserve and 3 or 4 non-reserve sites (n = 60 quadrats per site; Fig. 2). Quadrats were haphazardly placed on suitable rocky reef habitat. Kina size (test width) was measured by divers using callipers once in 1993 at 11 reserve and 5 non-reserve sites.

Pāua abundance (density) was measured in 2013 and 2022 by UVC in 1 x 1 m (1 m^2) quadrats at 5 reserve and 4 non-reserve sites (n = 30 quadrats per site in 2013 and 60 quadrats per site in 2022; Fig. 2). Quadrats were haphazardly placed on suitable rocky reef habitat. Pāua size (shell length) was measured once in 2022 by divers using callipers.

Scallop and horse mussel abundances were surveyed 8 times between 1994 and 2019 by UVC along ten 1 x 50 m (50 m^2) transects at 2-3 reserve and 2 non-reserve sites (Fig. 2). Divers deployed each transect along a different compass bearing and at least 10 m from the previous replicate. All scallops and horse mussels were counted within each quadrat, and in 2019 scallop maximum width was measured using callipers.

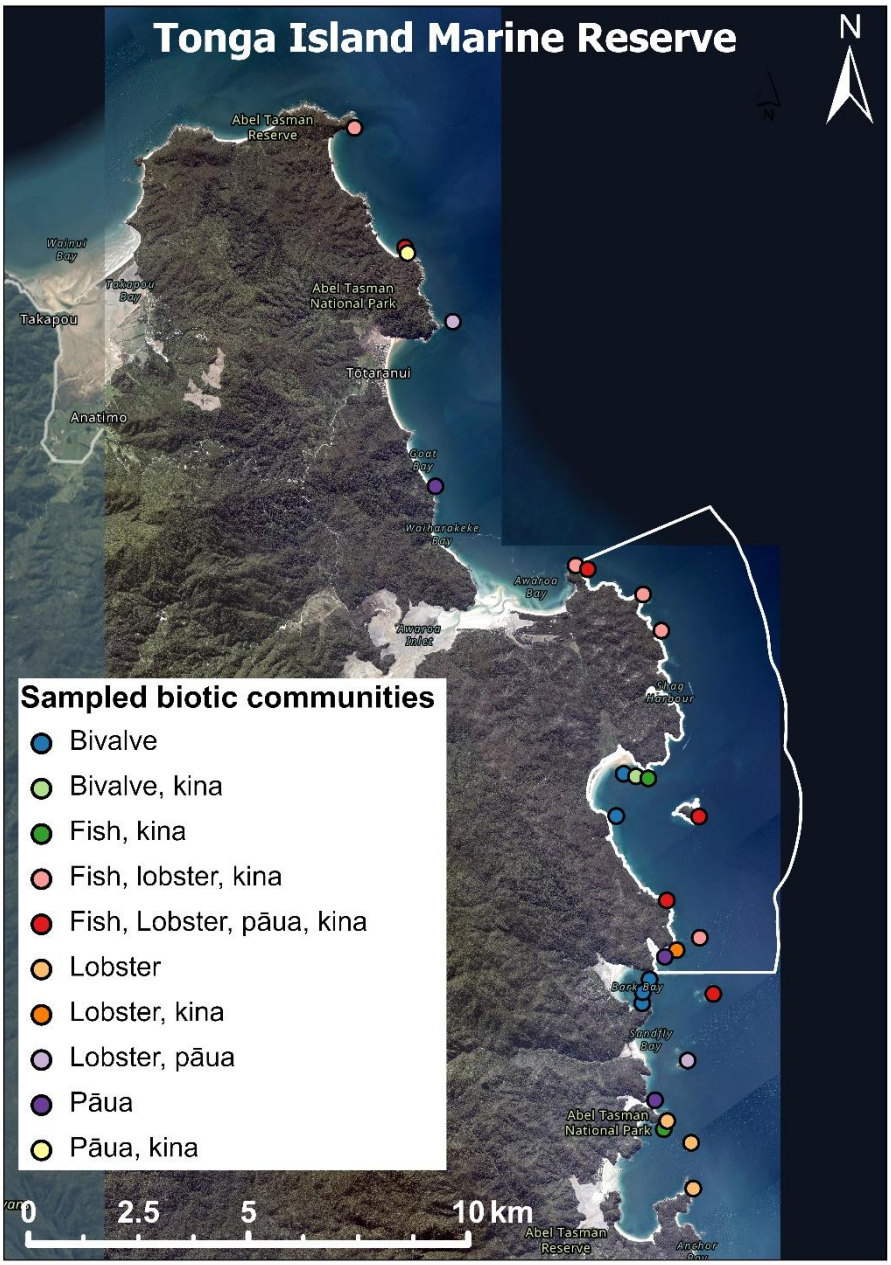

Figure 2. Survey sites inside (n=13) and outside (n=15) of the Tonga Island Marine Reserve. Reserve boundaries are shown with white lines. Each dot represents one survey site, and the colour of a dot represents the sampled biotic community at that location.

Statistical methods

Abundance data for rock lobster, kina, pāua, scallops, and horse mussels were analysed using ANOVA (Minitab version 17; Minitab 2014) or PERMANOVA (PRIMER V6 with the PERMANOVA+ add-on; Anderson et al. 2008) with Protection (reserve or non-reserve) and Year (and Sex, for lobsters only), as fixed factors and with Site nested within Protection as a random factor. Univariate data for lobster and pāua were analysed using PERMANOVA because they failed to meet the assumptions of ANOVA, even after transforming. For these datasets, Euclidian distance was used to calculate the resemblance matrix.

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, fish, kina, pāua, and scallops, discrete size classes were created from shell length, tail width, or test width every 5 or 10 mm 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, kina, scallops, 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.

Results

Rock lobster

Average rock lobster abundance (pooling over sex) was variable but consistently higher inside than outside of the marine reserve throughout the monitoring period (long-term average ± SE lobsters 100 m^{-2}: 4.0 ± 0.12 inside compared to 1.2 ± 0.04 outside; Fig. 3). This variability is reflected in the significant Year and Protection interaction ($PseudoF_{18, 143}$= 2.97, p <0.001) in the PERMANOVA. Further investigation using pairwise comparisons found that while lobster abundance in reserve and non-reserve sites did not significantly differ in most years, lobster abundances were significantly higher inside the reserve from 2006 – 2017 and in 2021. Pairwise comparisons of the interaction between Year and Protection showed that rock lobster abundance within the reserve decreased significantly over time, with a peak of 8.8 ± 1.6 lobsters 100 m^{-2} in 1999 decreasing to 3.3 ± 0.32 lobsters 100 m^2 when last sampled in 2023. There was no significant difference between the abundance of males and females ($PseudoF_{1,9}$ = 1.78, p = 0.5).

Size structure of rock lobster populations did not significantly differ between males (male long-term average ± SE tail width: 74.5 ± 0.48 mm inside and 65.36 ± 1.08 mm outside) and females (female long-term average ± SE: 71.48 ± 0.45 mm inside and 67.27 ± 1.37 mm outside) inside or outside of the marine reserve or through time (Protection x Sex interaction $PseudoF_{1,9}$ = 0.54, p= 0.589 and Year x Sex interaction $PseudoF_{5,9}$ = 1.82, p= 0.06; Fig. 4). Pooling over all sexes (i.e., males, females, juveniles, and unknowns) there was a significant difference in size structure between reserve and non-reserve sites ($PseudoF_{1,10}$=5.34, p= 0.009), with the reserve sites having size structures significantly skewed towards larger individuals (average ± SE tail width through time: 70.3 ± 0.35 mm inside the marine reserve and 57.6 ± 0.77 mm outside).

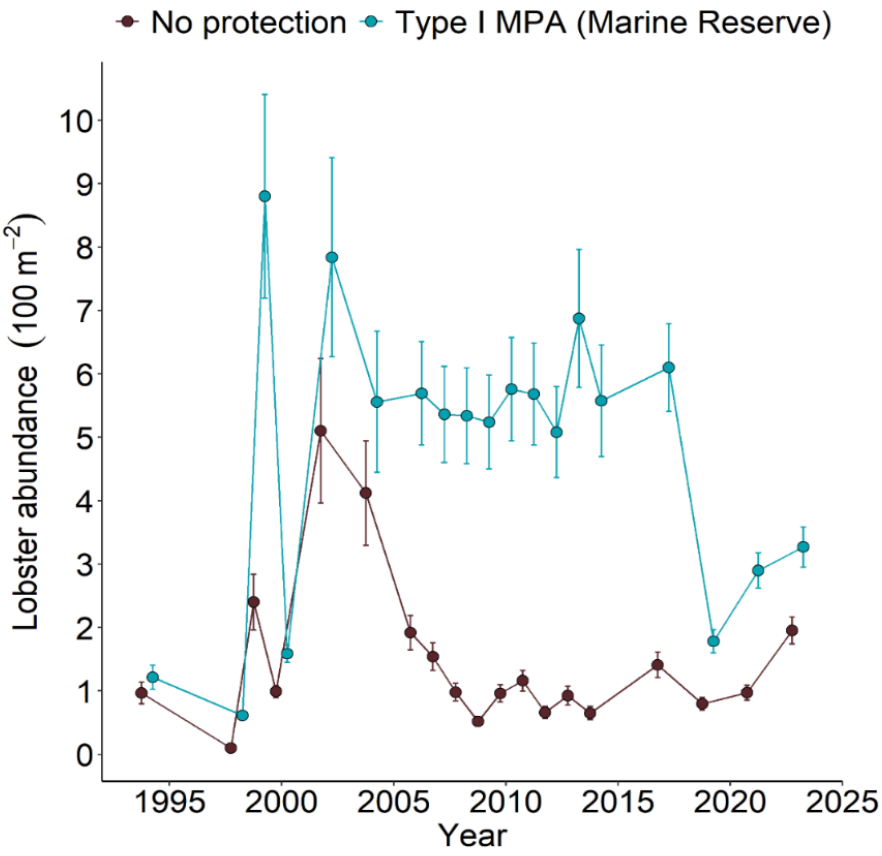

Figure 3. Abundance (mean ± SE 100m^{-2}) of rock lobster (*Jasus edwardsii*) inside and outside of the Tonga Island Marine Reserve from 1993 to 2023. N (the number of sites sampled per year) was 5–7 in the reserve and 4–7 outside of the reserve. Note: reserve and non-reserve sites were sampled at the same time each year: Type I MPA datapoints are offset for visual ease.

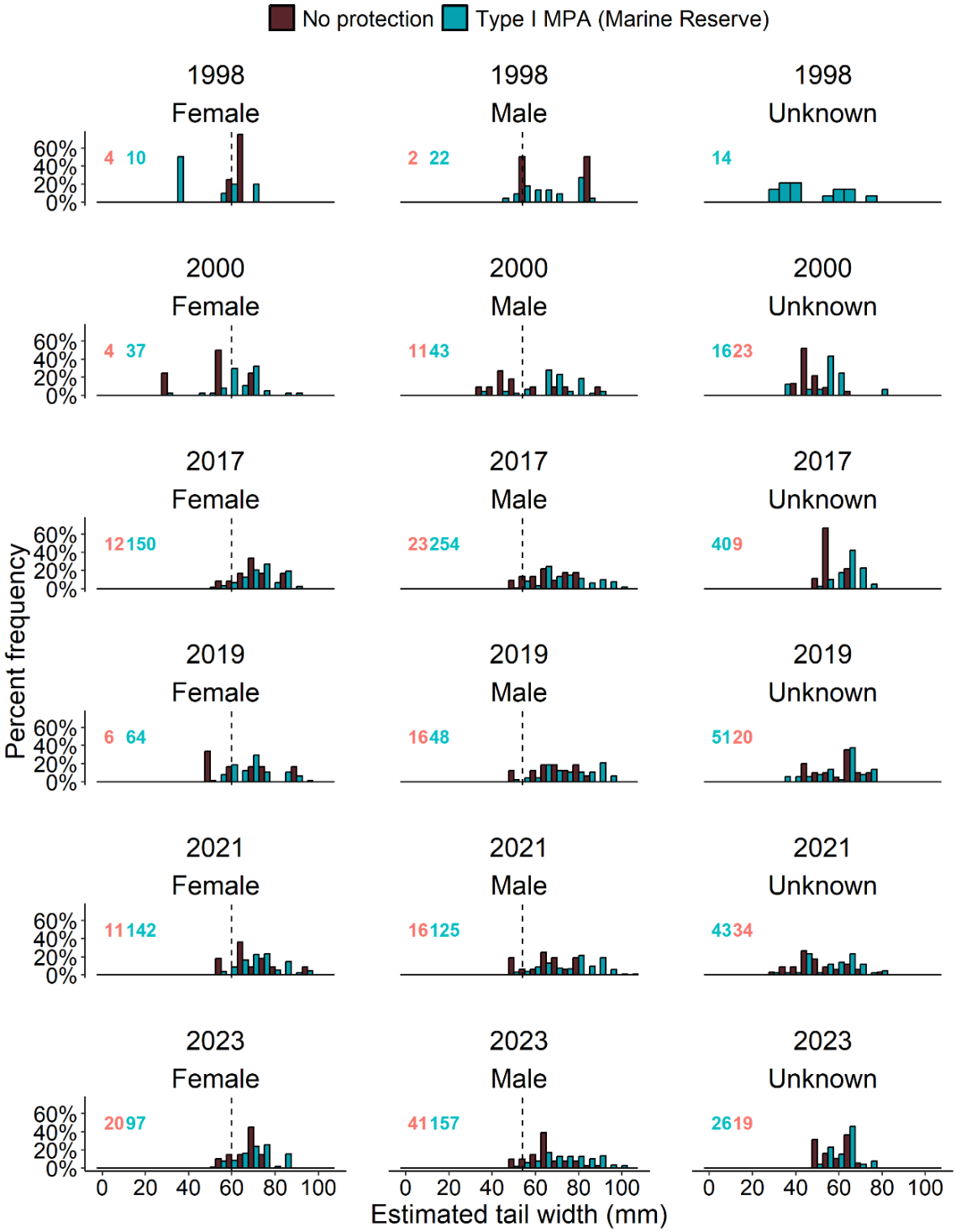

Figure 4. Percent length frequency histograms of male, female or unsexed (unknown) rock lobster (*Jasus edwardsii*) sampled inside and outside of the Tonga Island Marine Reserve from 1998 to 2023. The total number of measured individuals within and outside of the marine reserve are presented per sex and per year. Dashed lines represent the minimum tail width for legal retention in males and females. N (the number of sites sampled per year) was 5–7 in the reserve and 4–7 outside of the reserve. Bin width = 5 mm, i.e., 60-64.9 mm is immediately to the right of 60 = on the x-axis and 55-59.9 is immediately to the left of 60 on the x axis.

Fish communities

Reef fish communities varied significantly through time and inside versus outside of the marine reserve, as indicated by a significant fixed factor interaction between Protection and Year in the PERMANOVA ($PseudoF_{20, 203}$ = 1.50, p = 0.005; Fig. 5 and 6). This analysis revealed that reef fish communities were significantly different between reserve and non-reserve sites in 2017 and 2024. SIMPER comparisons for these years showed that differences were driven by; i) higher abundances of blue cod, scarlet wrasse, tarakihi, and butterfly perch inside the marine reserve than outside, and; ii) higher abundances of spotty outside than inside of the marine reserve (Appendix 1). Based on SIMPER comparisons in the years where the PERMANOVA identified no significant difference in community composition between reserve and non-reserve sites blue cod, spotty, scarlet wrasse, tarakihi, sweep, goatfish, and butterfly perch 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 were blue moki, red moki, marblefish, and banded wrasse. Snapper were not commonly observed within or outside of the marine reserve, with a maximum average density of only 0.8 ± 0.3 individuals 60 m^{-3} within the marine reserve in 2022; most other years had 0 or <0.1 individuals 60 m^{-3}.

For size structure of measured reef fishes (red moki, tarakihi, blue cod), red moki and tarakihi did not significantly differ between reserve and non-reserve sites, whereas blue cod were consistently and significantly larger inside (29.4 ± 0.4 cm) than outside (24.7 ± 0.5 cm) of the marine reserve ($PseudoF_{1,11}$ = 16.0, p <0.001; Fig. 7). There were significant differences between years for all species, with red moki being larger in 2020 than 2022 or 2024, tarakihi having no consistent pattern, and blue cod being larger in 2024 than 2020 and 2022.

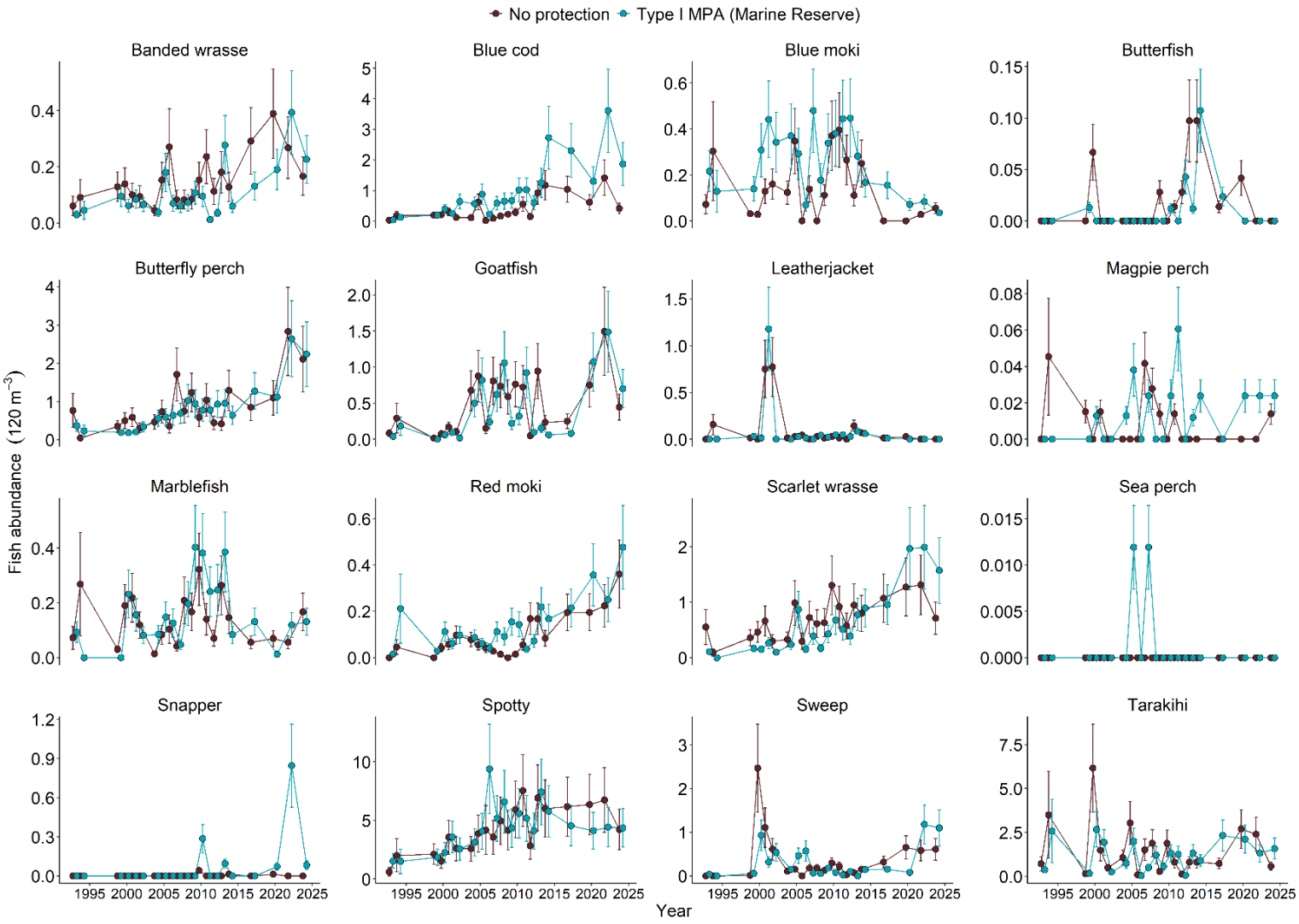

Figure 5. Reef fish abundance (mean ± SE 120 m^{-3}) at sites inside and outside of the Tonga Island Marine Reserve from 1993 to 2024. N = 2-7 sites inside the reserve and 2-6 sites outside the reserve. Note: reserve and non-reserve sites were sampled at the same time each year, but Type I MPA datapoints are offset for visual ease.

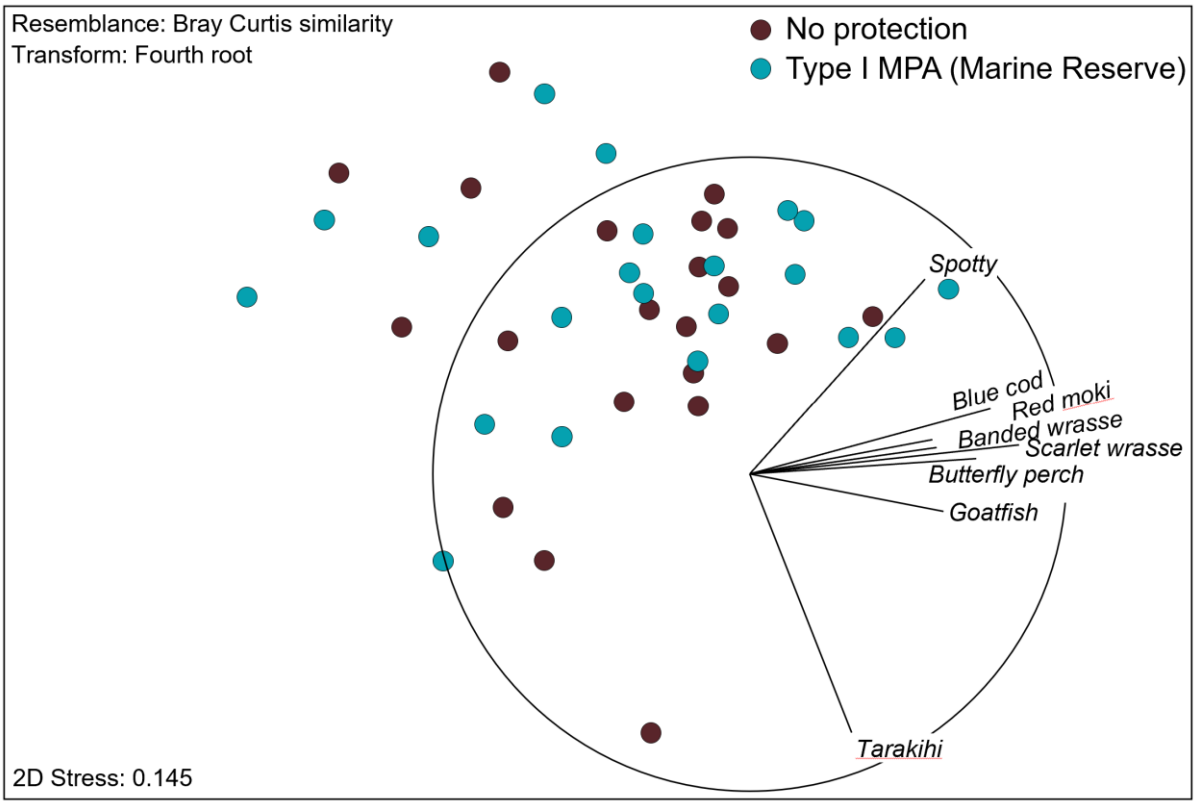

Figure 6. Non-metric Multidimensional Scaling plot (nMDS) for reef fish communities inside and outside of the Tonga Island Marine Reserve from 1993 to 2024. Each dot represents the average community composition within each protection type per year. Note that this averaging was done to decrease 2D stress 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.

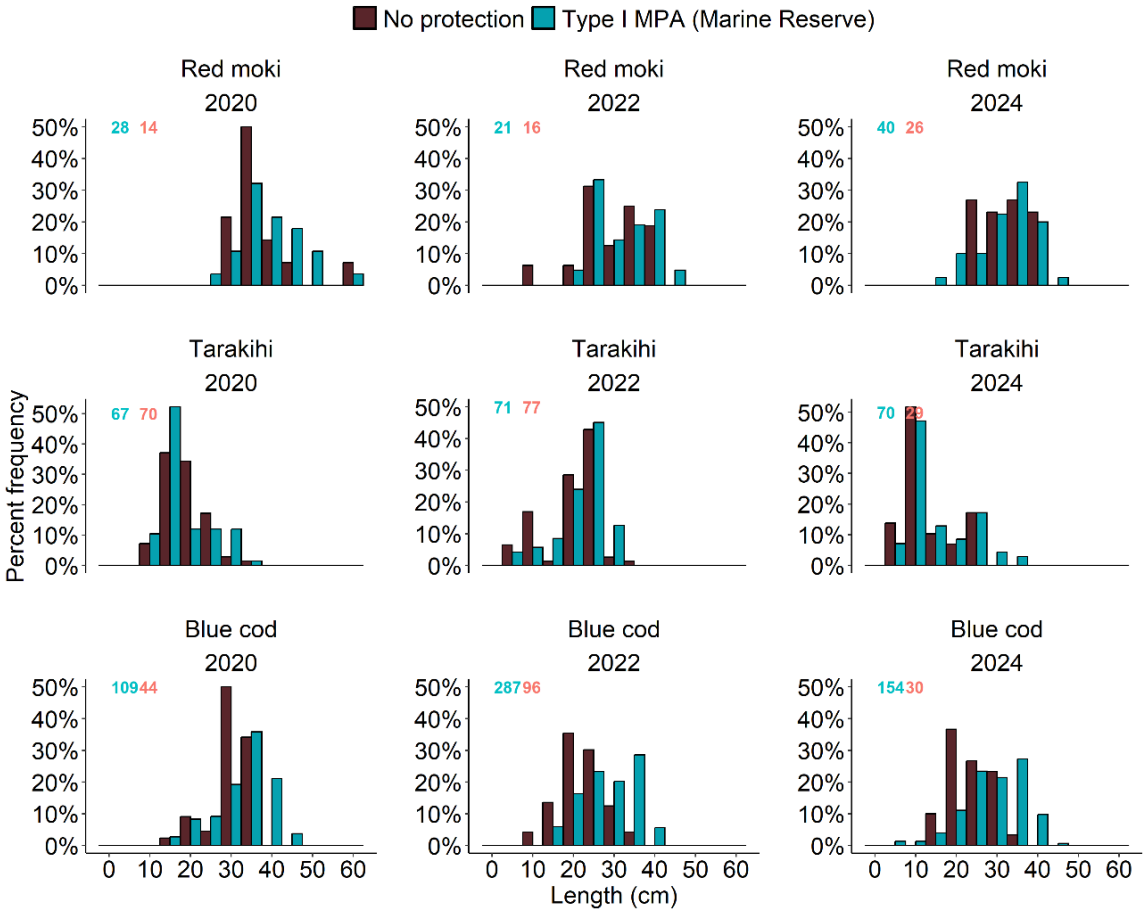

Figure 7. Percent length frequency histograms of reef fish (red moki, tarakihi, and blue cod) sampled inside and outside of the Tonga Island Marine Reserve from 2020 to 2024. The total number of measured individuals within and outside of the marine reserve are presented for each year. N = 7 reserve and 6 non-reserve sites. Bin width = 5mm, i.e., 20-24.9mm is immediately to the right of 20mm on the x-axis and 15-19.9 is immediately to the left of 20 on the x axis.

Kina

There was no significant difference in kina abundance ($F_{2,10} = 0.15$, $p = 0.866$) or size structure ($PseudoF_{1,14} = 0.23$, $p = 0.758$) between reserve and non-reserve sites (Figs. 8 and 9). Within the reserve the long-term average of kina abundance (mean ± SE) was 1.22 ± 0.35 m^{-2} compared to 1.42 ± 0.45 m^{-2} outside of the reserve. Average kina sizes, which were only measured in 1993, were 56.9 ± 0.24 mm inside the reserve and 58.1 ± 0.36 mm outside the reserve.

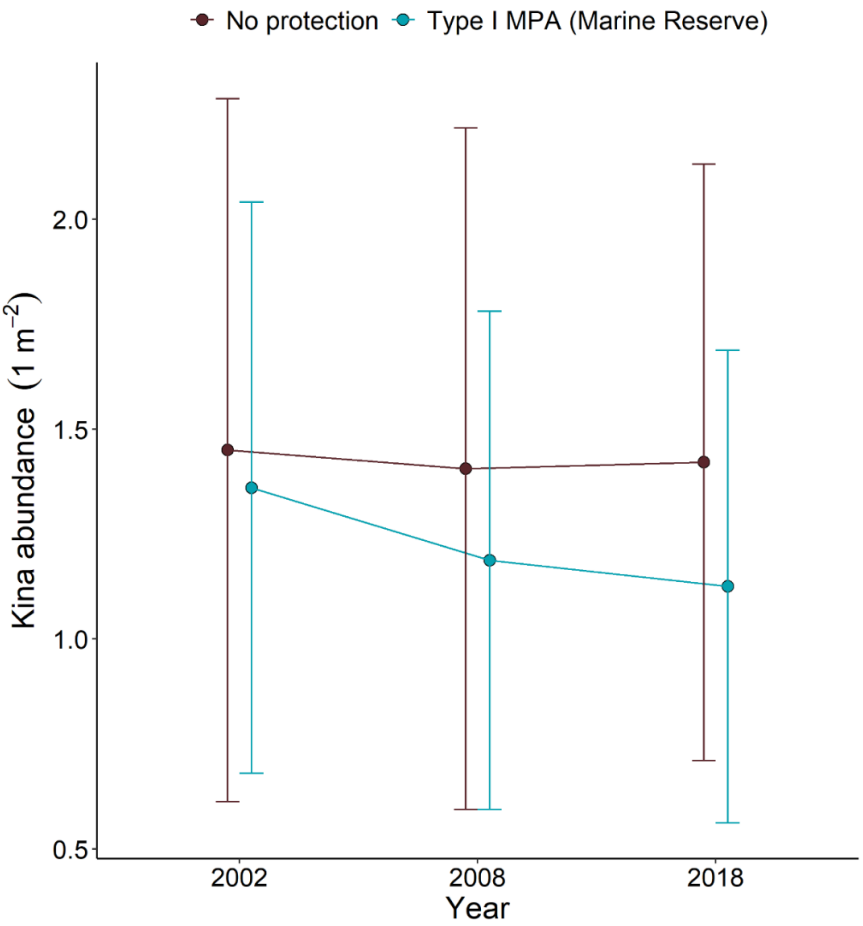

Figure 8. Abundance of kina (*Evechinus chloroticus*; mean ± SE m^{-2}) at sites inside and outside of the Tonga Island Marine Reserve in 2002, 2008 and 2018. N = 4 reserve sites and 3-4 non-reserve sites. Note: reserve and non-reserve sites were sampled at the same time each year: Type I MPA datapoints are offset for visual ease.

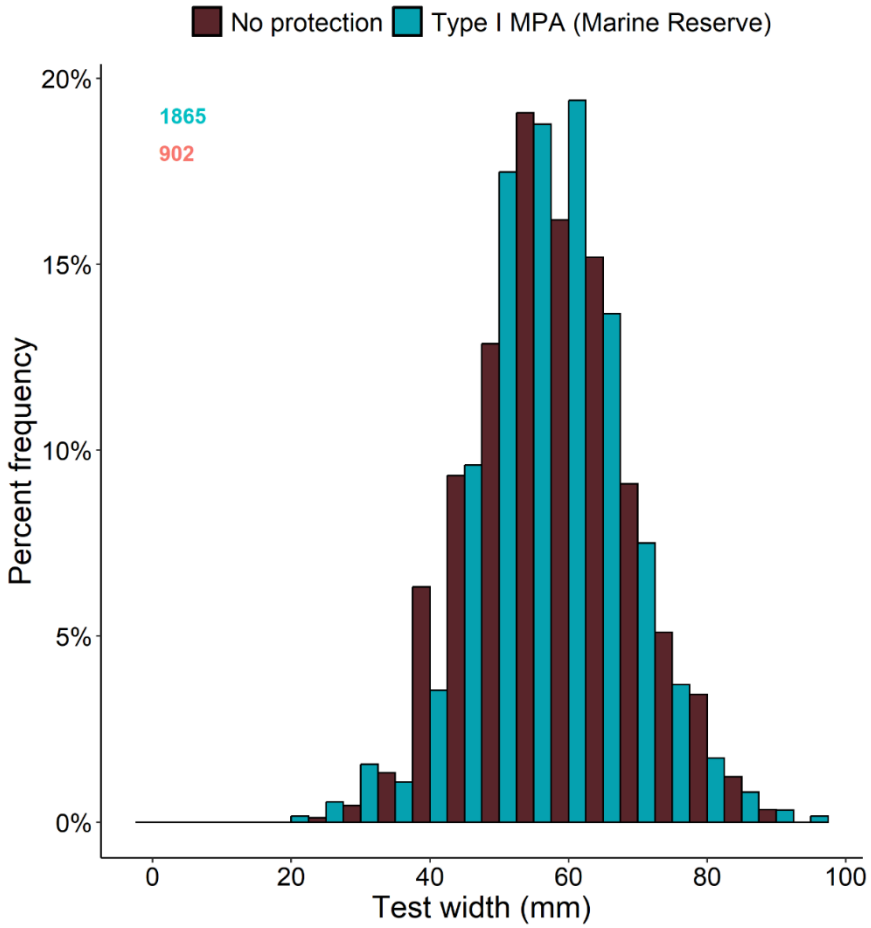

Figure 9. Percent length frequency histogram of kina (*Evechinus chloroticus*) size (mm) inside and outside of the Tonga Island Marine Reserve in 1993. N = 11 reserve sites and 5 non-reserve sites. The total number of measured individuals within and outside of the marine reserve are presented in the graph. Bin width = 5mm, i.e., 40-44.9mm is immediately to the right of 40mm on the x-axis and 35-39.9 is immediately to the left of 40 on the x axis.

Pāua

There was no significant difference in the abundance ($PseudoF_{1,12}$ = 2.1, p = 0.153) or size structure ($PseudoF_{1,7}$ = 0.22, p = 0.935) of pāua between reserve and non-reserve sites (Fig. 10 and 11). Although pāua were more abundant on average outside of the reserve (1.25 ± 0.06 mean ± SE pāua m^{-2}) than inside of the reserve (0.69 ± 0.03 mean ± SE pāua m^{-2}), this was not statistically significant. This may be due to the variability across the 9 sampled sites, within which pāua densities ranged from 0.02 – 4.9 m^{-2} outside of the reserve and 0 – 3.4 m^{-2} inside the reserve. Notably, average pāua abundance declined from 2.3 to 0.72 pāua m^{-2} at non-reserve sites between 2013 and 2022. Average pāua shell length, which was only measured in 2022, was 91.7 ± 1.1 mm outside of the reserve and 90.0 ± 1.4 mm inside of the reserve. The largest pāua measured was 121 mm.

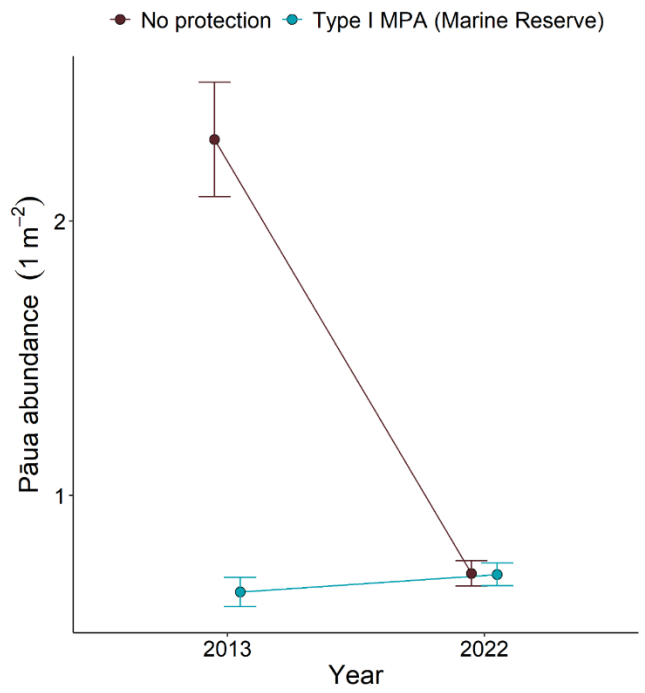

Figure 10. Abundance of pāua (*Haliotis iris*; mean ± SE m^{-2}) at sites inside and outside of the Tonga Island Marine Reserve in 2013 and 2022. N = 5 reserve sites and 4 non-reserve sites. Note: reserve and non-reserve sites were sampled at the same time each year: Type I MPA datapoints are offset for visual ease.

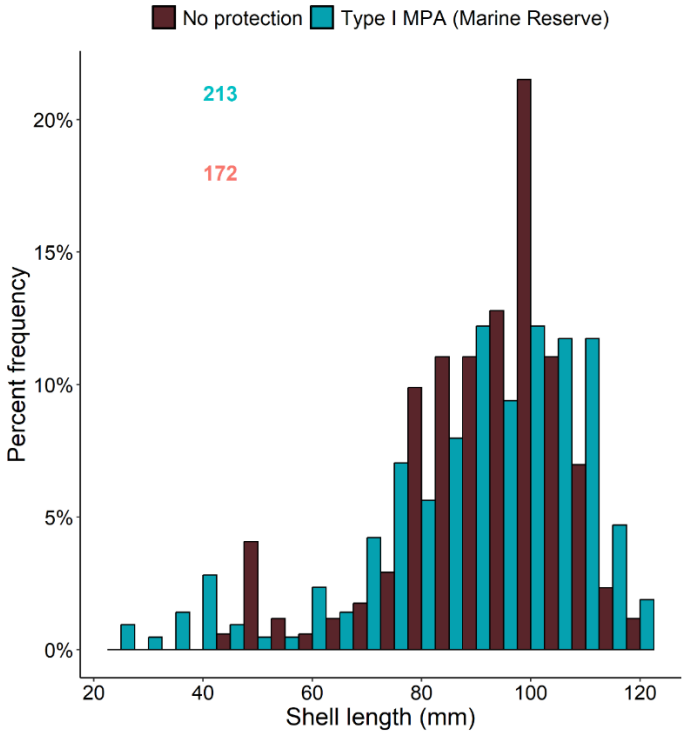

Figure 11. Percent length frequency histogram of pāua (*Haliotis iris*) size inside and outside of the Tonga Island Marine Reserve in 2022. N = 5 reserve sites and 4 non-reserve sites. The total number of measured individuals within and outside of the marine reserve are presented in the graph. Bin width = 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.

Scallops and horse mussels

The abundances of scallops and horse mussels between reserve and non-reserve sites were highly variable through time. There was a significant interaction between Year and Protection for both scallops ($F_{7,277}$ = 18.46, p <0.001) and horse mussels ($F_{6,275}$ = 9.69, p <0.001). This interaction reflects the lack of a clear pattern in abundance per site, as any differences in abundance inside and outside of the reserve fluctuated significantly depending on the year (Fig. 12). In contrast to abundance, scallop size structure in 2019 was not significantly different between reserve and non-reserve sites (*PseudoF*$_{1,2}$ = 3.24, p = 0.112), although scallops were bigger outside (83.8 ± 0.7 mm) than inside (74.0 ± 1.9 mm) of the marine reserve (Fig. 13). It is worth noting that the overall power of this size analysis was low due to the small number of sampling sites within and outside of the marine reserve (2-3 each) where lengths were measured.

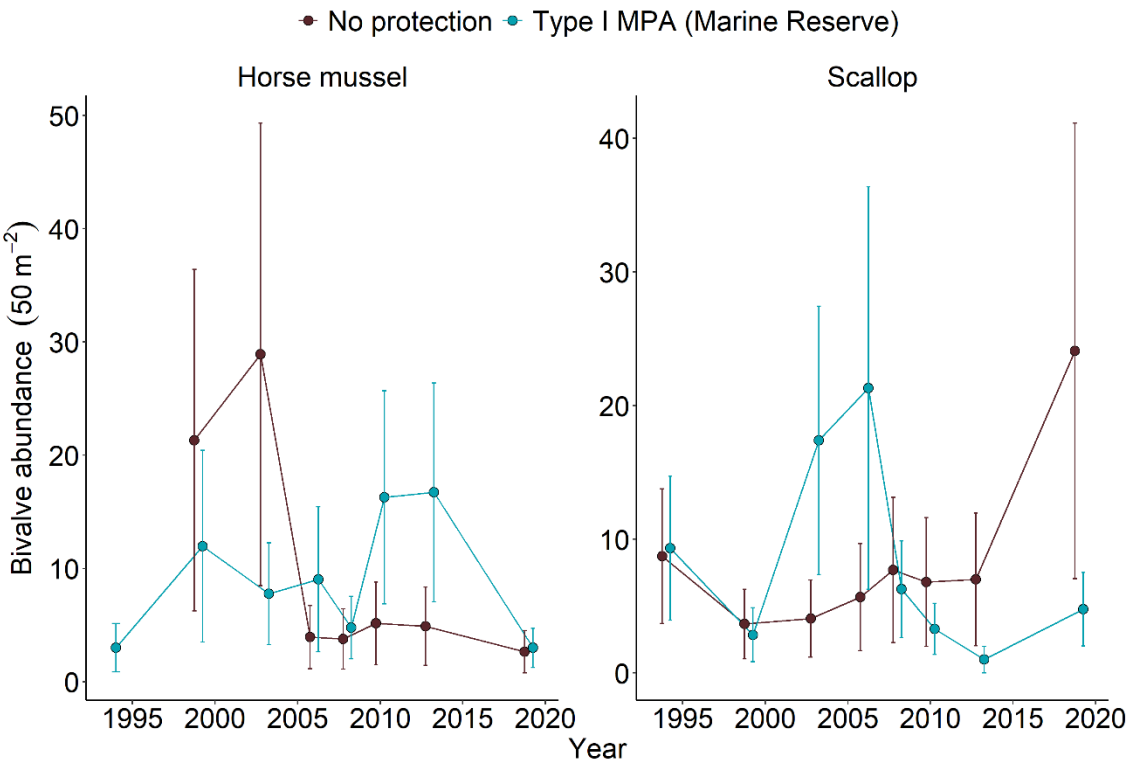

Figure 12. Abundance (mean ± SE 50 m^{-2}) of horse mussels (*Atrina zelandica*) and scallops (*Pecten novaezelandiae*) at sites inside and outside of the Tonga Island Marine Reserve from 1994 to 2019. N = 2-3 reserve sites and 2 non-reserve sites. Note: reserve and non-reserve sites were sampled at the same time each year: Type I MPA datapoints are offset for visual ease.

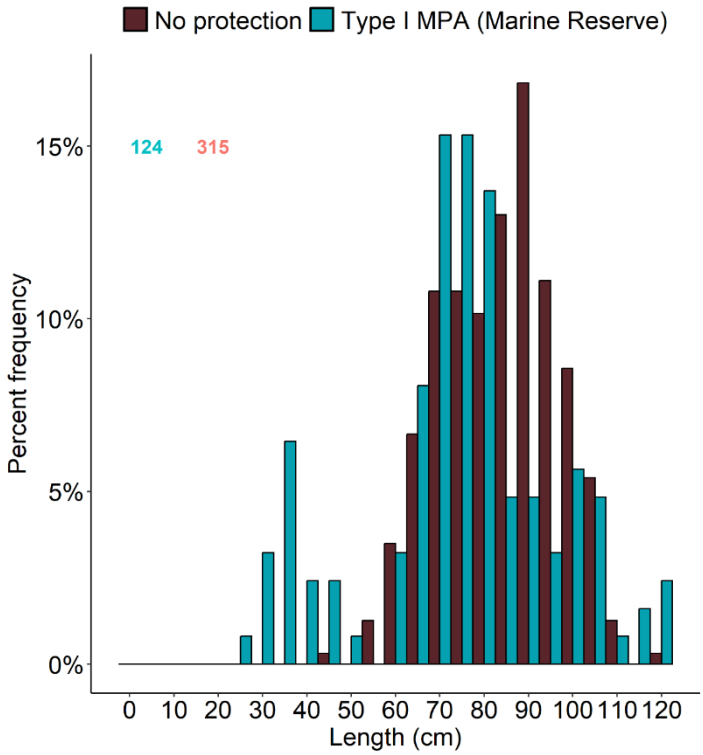

Figure 13. Percent length frequency histogram of scallop size inside and outside of the Tonga Island Marine Reserve in 2019. N = 3 reserve sites and 2 non-reserve sites. The total number of measured individuals within and outside of the marine reserve are presented in the graph. Bin width = 5mm, i.e., 60-64.9mm is immediately to the right of 60mm on the x-axis and 55-59.9 is immediately to the left of 60 on the x axis.

Discussion

Rock lobster

The Tonga Island Marine Reserve has had clear beneficial effects on rock lobster abundance and size structure. Rock lobster densities increased more than 8x within the reserve during the first decade of protection and remained significantly higher than at non-reserve sites throughout most of the monitoring period. Lobsters within the reserve were consistently larger, indicating that protection has enhanced both abundance and demographic structure.

These findings are consistent with reports from marine reserves in New Zealand and internationally, where exploited invertebrates often show strong responses to protection (Edgar and Barrett 1999, Pande et al. 2008). Positive reserve effects on rock lobster are common in similar New Zealand reserves, including Horoirangi, Hikurangi, Long Island, Taputeranga and Kapiti Marine Reserves, suggesting that *Jasus edwardsii* responds positively to long-term protection despite being a relatively mobile species (Pande et al. 2008, Davidson et al. 2014, Rojas-Nazar et al. 2019, Gerrity et al. 2025, Gerrity and Virgin 2026). Tagging studies elsewhere in New Zealand have demonstrated that many adult lobsters show high site fidelity and can remain associated with reef habitats over extended periods, allowing populations within reserves to accumulate larger and older individuals over time (MacDiarmid and Breen 1992). Reserves are also subject to larval influx and inward migration of adults, which are gregarious and will join established populations (Kelly et al. 1999, 2000, Davidson et al. 2013).

The increase in lobster size structure within the Tonga Island reserve is ecologically important because larger females typically contribute disproportionately to reproductive output through greater fecundity and larval production (Annala and Bycroft 1987). Consequently, marine reserves may function not only as refuges from fishing mortality, but also as potential spawning banks that support surrounding populations through larval export. Larval export from marine reserves is not well quantified, but Kelly et al. (2000) estimated that within New Zealand reserves rock lobster egg production increases by 4.8–9.1% per year of protection, suggesting an increasing contribution to wider recruitment over time.

Despite consistently higher abundances within the reserve, lobster abundance declined by about 50% in recent surveys and were similar to non-reserve sites in 2023. Similar declines have been observed in the Hikurangi Marine Reserve (Gerrity et al. 2025), suggesting that broader factors are influencing protected lobster populations. Potential explanations of the noted decline include natural recruitment variability, depletion of source populations in non-reserve locations, illegal harvest within the reserve, lobster migrating out of reserves, habitat changes, and environmental disturbances such as marine heat waves. Although further investigation into such causative factors is outside the scope of this report, we note that marine reserve rangers regularly detect illegal fishing within the reserve. Also, it is well established that marine heat waves have affected a range of marine species in New Zealand coastal ecosystems in recent years (Thomsen et al. 2021, Bell et al. 2023). More frequent disturbances under climate change may influence the trajectory of reserve populations irrespective of protection status. These examples highlight that marine reserves can reduce fishing mortality but cannot fully buffer populations against non-compl wider ecosystem effects and environmental change.

Fish communities

Reef fish assemblages differed between reserve and non-reserve sites, although reserve effects were species-specific and variable through time. The most consistent response was observed for blue cod, which were significantly larger inside the reserve than outside. Blue cod are one of the most heavily targeted fish species in the region and tend to respond strongly to spatial protection because of their relatively limited home ranges, high site fidelity, and susceptibility to fishing mortality (Carbines 1999, Beentjes 2021). Increased size structure within marine reserves has been reported previously for blue cod in New Zealand reserves and is generally interpreted as evidence of reduced fishing pressure and increased survivorship of older individuals (Pande et al. 2008). Larger blue cod within the reserve may have important ecological and fisheries implications. Larger individuals typically have greater reproductive output and may contribute disproportionately to egg production and population replenishment (Beer et al. 2013). In addition, changes in predator abundance and body size can alter trophic interactions within reef ecosystems, although such effects were not assessed in this monitoring (Kolodzey et al. 2023).

Fish community differences between reserve and non-reserve sites were also influenced by higher abundances of scarlet wrasse, butterfly perch, tarakihi, and blue cod within the reserve, whereas spotty were more abundant outside the reserve. However, responses among species were inconsistent and likely reflect differences in mobility, habitat associations, life history, and fishing pressure. For example, species such as tarakihi and blue moki are relatively mobile and may move across reserve boundaries where they are vulnerable to fishing mortality, reducing the likelihood of strong reserve effects.

Snapper abundance remained low throughout the monitoring despite snapper being a species commonly associated with positive reserve responses elsewhere in northern New Zealand (Babcock et al. 1999, Denny et al. 2004). This may partly reflect the biogeographic position of the reserve near

the southern extent of substantial snapper populations, but may also reflect limitations of diver-based survey methods. Snapper avoid divers in fished areas and are often underrepresented in UVC surveys (Cole 1994). Baited remote underwater video monitoring is currently underway, so investigation of this data within future reports would potentially improve detectability of mobile and diver-shy species.

Overall, the reef fish results demonstrate that reserve effects are not uniform across species and that long-term responses depend strongly on species life histories and local environmental conditions. While exploited species with strong site fidelity such as blue cod showed clear benefits from protection, responses among other species were weaker or more variable.

Kina

After approximately 25 years of protection, there was no significant reserve effect on kina abundance or size. Densities remained low both inside and outside the reserve and were at the lower end of values reported for pristine habitats (McShane and Naylor 1991). Although previous studies have shown that increases in large rock lobster within reserves can reduce kina abundance through predation (Shears and Babcock 2003), the decline observed here was non-significant. Kina densities at non-reserve sites remained similarly low despite a scarcity of large lobsters. This pattern suggests that factors other than predation, such as environmental conditions, habitat characteristics, larval supply, or recruitment processes, are likely regulating kina populations here. Given the substantial variability in the data and the absence of monitoring since 2018, it is unknown whether kina populations have remained stable or undergone changes in recent years.

Pāua

Pāua abundance remained stable within reserve sites but declined by 66% at non-reserve sites between 2013 and 2022. Although high site-to-site variability meant this decline was not statistically significant, the contrasting trajectories inside and outside the reserve suggest that protection may have contributed to greater population stability within the reserve, although further monitoring is required to test this.

Interpretation of these results should be cautious because pāua populations are naturally patchy and closely associated with specific habitat features, particularly reef structure, wave exposure, and macroalgal availability. Habitat variability among sites may therefore obscure broader reserve effects, especially given the limited number of survey years (two), so full scale habitat assessment and/or habitat classification during surveys would aid in understanding reserve effects on pāua. Nonetheless, the apparent decline outside the reserve is notable and may reflect a combination of fishing pressure, environmental variability, or recruitment fluctuations.

All pāua measured during this monitoring were below the current minimum legal size for retention, and individuals were much smaller than those commonly present along colder and more exposed coastlines elsewhere in New Zealand (Naylor et al. 2007; Gerrity et al. 2025). Previous studies have shown that pāua growth and maximum size are strongly influenced by environmental conditions, with cooler temperatures, high water movement, and productive macroalgal habitats generally supporting faster growth and larger individuals (Naylor et al. 2007). The relatively sheltered and warm conditions within the Tonga Island Marine Reserve may therefore naturally constrain growth potential (Albuquerque et al. 2021; Stevens et al. 2021). Thus, the predominant drivers of pāua population change may be more related to natural and environmental factors such as habitat quality, food availability, and recruitment.

Scallops and horse mussels

No clear reserve effects were detected for scallops or horse mussels, although interpretation of these results was limited by high spatial variability. Both taxa displayed highly patchy distributions through time, with substantial variation among sites and survey years.

The absence of clear reserve effects is not unexpected given the ecology of these species and the broader history of Tasman and Golden Bays. Both scallops and horse mussels are strongly influenced by factors independent from fishing, including sediment characteristics, water quality and temperature, recruitment variability, and intensive benthic disturbance (Williams et al 2024). Previous studies have suggested that historical scallop dredging and other seabed disturbances may have altered benthic habitats throughout the region, potentially limiting recovery of habitat-forming and sediment-associated species even after fishing pressure is reduced (Newcombe et al. 2015). Other studies have characterized historical sediment deposition into Tasman Bay resulting from coastal land-use practices and have suggested this may be a source of habitat degradation and ecological decline, particularly for benthic soft-sediment dwelling taxa (Davidson et al. 2018, Handley 2019).

Horse mussels are considered important ecosystem engineers that enhance habitat complexity and biodiversity within soft-sediment ecosystems (Lohrer et al. 2013, MacDiarmid et al. 2013). Declines in horse mussel abundance may therefore have broader ecological implications from reductions in biogenic habitat.

Scallops once comprised a major commercial and recreational fishery in the Tasman Bay region but was closed in 2017 due to severe depletion[2]. Scallop abundance appeared greater at non-reserve sites in the most recent surveys, although this pattern was highly variable and difficult to interpret confidently. Notably, scallop enhancement efforts (spat seeding) have occurred outside of the reserve, although our scallop monitoring data analysis was unable to account for where, when, and to what extent this was done. The closure of the regional scallop fishery in 2017 further complicates interpretation of reserve effects because fishing mortality has been effectively eliminated across both reserve and non-reserve areas. Recovery of these species in the Tasman Bay region may therefore depend less on protection status alone and more on broader habitat recovery and environmental conditions.

General discussion and recommendations

After ~31 years of protection the Tonga Island Marine Reserve has had measurable benefits for some, but not all key species. Revisiting the research questions, the reserve has had positive effects on the size and abundance of some key species, including positive impacts observed over time. Results corroborate other studies indicating that exploited species with high site-fidelity typically benefit the most from spatial protection if their habitats are viable. However, reserve effects were inconsistent among taxa, and no effects were detected for many key species.

This monitoring highlights that the effects of marine reserve protection are not independent of broader ecological processes, anthropogenic stressors, and environmental change. Population trajectories within and outside of the reserve are shaped by a variety of ecological and environmental factors. For several key species, particularly scallops, horse mussels, and pāua, habitat quality and

[2] Ministry for Primary Industries 2017. Temporary Closure of the Southern Scallop (SCA 7) Fishery. Decision Paper 2017/24. Ministry for Primary Industries, Wellington.

environmental drivers may be just as important as fishing pressure in determining long-term population status.

In some cases, the non-response of key species to protection may reflect prior damage to critical habitats or legacy effects from historic overfishing. Additional disturbances, such as marine heatwaves and sediment deposition, may also have long-term consequences for key species and their habitats. Previous studies have suggested that the absence of strong reserve effects on key species within the Tonga Island Marine Reserve may result from several factors, including species-specific life histories, legacy effects of historical overfishing, intense fishing pressure adjacent to the reserve, the isolated nature of reef habitats within the reserve, and the generally low productivity of the ecosystem (Davidson and Richards 2013).

This report demonstrates the importance of long-term ecological monitoring. Population trajectories of marine species can fluctuate substantially through time, and short-term studies often fail to distinguish reserve effects from natural variability. The approximately 31-year time series presented here therefore represents a valuable dataset for understanding ecological responses to marine protection in temperate New Zealand ecosystems.

Continued monitoring of key species will further clarify the long-term ecological effects of protection at Tonga Island Marine Reserve. UVC surveys should continue as the core monitoring method as they provide a consistent time series, particularly for sedentary species. Incorporation of complementary methods such as baited remote underwater video monitoring may improve detection of mobile, cryptic, or diver responsive species that UVC are known to mischaracterise (e.g. Brock 1982, Thresher and Gunn 1986, McCormick and Choat 1987; Cheal and Thompson 1997). All surveys should be carried out consistently, ideally with the same dive team, and with standardised sampling effort and spatial coverage. Increasing replication of sites should be prioritised over replication within sites, particularly for surveys with few sites such as horse mussel and scallop assessments.

Recruitment and habitat assessments should also be incorporated into future monitoring and research if possible. Deploying rock lobster puerulus collectors to monitor larval abundance would indicate whether declines in adult abundance are linked to larval supply. Habitat assessments in soft sediment areas are needed to evaluate whether habitat quality or juvenile recruitment limitations are constraining recovery of horse mussel and scallop populations. Deployment of temperature loggers on the sea floor would permit monitoring of the thermal conditions and potential thermal stressors that may affect shellfish populations. Lastly, quantifying macroalgal abundance and species composition would provide an important indicator of ecosystem condition, as long term changes in macroalgal communities can reflect temperature driven events or trophic shifts.

Conclusion

As a whole the Tonga Island Marine Reserve has provided long-term ecological benefits for at least some key species. The 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 Tonga Island Marine Reserve from 2000 to 2024. Years with significant differences between reserve and non-reserve sites are indicated in bold.

Year	Scientific name	Common name	Abundance (fourth root)		Average dissimilarity	Ratio (Diss / SD)	Percent contribution
			No protection	Type I MPA			
2000	*Nemadactylus macropterus*	Tarakihi	0.8	0.6	14.88	1.03	23.37
	Notolabrus celidotus	Spotty	0.85	1	11.02	0.89	17.31
	Scorpis lineolata	Sweep	0.41	0.34	7.98	0.8	12.53
	Parapercis colias	Blue cod	0.15	0.29	6.73	0.62	10.57
	Caesioperca lepidoptera	Butterfly perch	0.28	0.12	5.32	0.59	8.36
	Pseudolabrus miles	Scarlet wrasse	0.28	0.11	4.52	0.61	7.09
	Aplodactylus arctidens	Marblefish	0.15	0.18	3.83	0.55	6.02
	Latridopsis ciliaris	Blue moki	0.03	0.2	3.04	0.46	4.78
2001	*Nemadactylus macropterus*	Tarakihi	0.66	0.84	10.93	1	20.37
	Notolabrus celidotus	Spotty	1.24	1.22	6.61	0.83	12.32
	Scorpis lineolata	Sweep	0.44	0.16	5.66	0.76	10.56
	Pseudolabrus miles	Scarlet wrasse	0.39	0.18	5.47	0.76	10.21
	Parapercis colias	Blue cod	0.2	0.2	4.73	0.6	8.83
	Aplodactylus arctidens	Marblefish	0.18	0.15	3.87	0.56	7.21
	Caesioperca lepidoptera	Butterfly perch	0.27	0.11	3.85	0.57	7.18
	Latridopsis ciliaris	Blue moki	0.12	0.26	3.78	0.6	7.05
	Meuschenia scaber	Leatherjacket	0.14	0.12	3.39	0.39	6.31
2002	*Notolabrus celidotus*	Spotty	0.98	1.1	12.78	0.91	20.97
	Scorpis lineolata	Sweep	0.36	0.21	9.03	0.73	14.82
	Parapercis colias	Blue cod	0.1	0.36	8.17	0.69	13.41
	Nemadactylus macropterus	Tarakihi	0.3	0.19	7.61	0.69	12.5
	Pseudolabrus miles	Scarlet wrasse	0.23	0.1	5.68	0.56	9.32
	Latridopsis ciliaris	Blue moki	0.13	0.19	4.99	0.54	8.19
	Caesioperca lepidoptera	Butterfly perch	0.16	0.08	4.01	0.43	6.58
	Cheilodactylus spectabilis	Red moki	0.07	0.08	2.52	0.38	4.13
	Aplodactylus arctidens	Marblefish	0.07	0.08	2.42	0.38	3.97
2004	*Notolabrus celidotus*	Spotty	1.03	1.03	13.47	0.94	20.97

	Nemadactylus macropterus	Tarakihi	0.43	0.41	10.4	0.87	16.19
	Upeneichthys lineatus	Goatfish	0.33	0.27	7.71	0.72	12.01
	Parapercis colias	Blue cod	0.08	0.39	7.56	0.72	11.78
	Pseudolabrus miles	Scarlet wrasse	0.26	0.21	6.95	0.65	10.83
	Latridopsis ciliaris	Blue moki	0.1	0.28	6.18	0.58	9.62
	Caesioperca lepidoptera	Butterfly perch	0.14	0.19	4.47	0.5	6.95
	Scorpis lineolata	Sweep	0.08	0.09	2.61	0.36	4.06
2005	*Nemadactylus macropterus*	Tarakihi	0.76	0.71	10.24	1.04	17.77
	Notolabrus celidotus	Spotty	1.18	1.16	8.38	0.92	14.54
	Pseudolabrus miles	Scarlet wrasse	0.52	0.47	7.79	0.95	13.54
	Parapercis colias	Blue cod	0.36	0.47	7.24	0.89	12.57
	Upeneichthys lineatus	Goatfish	0.45	0.33	6.77	0.86	11.75
	Caesioperca lepidoptera	Butterfly perch	0.23	0.2	3.78	0.59	6.57
	Latridopsis ciliaris	Blue moki	0.16	0.23	3.64	0.61	6.32
	Aplodactylus arctidens	Marblefish	0.07	0.14	2.71	0.42	4.71
	Notolabrus fucicola	Banded wrasse	0.13	0.14	2.57	0.5	4.47
2007	*Nemadactylus macropterus*	Tarakihi	0.54	0.38	8.8	0.92	15.44
	Notolabrus celidotus	Spotty	1.22	1.29	8.75	0.84	15.34
	Upeneichthys lineatus	Goatfish	0.44	0.31	7.86	0.82	13.79
	Caesioperca lepidoptera	Butterfly perch	0.44	0.27	7.86	0.77	13.78
	Pseudolabrus miles	Scarlet wrasse	0.43	0.26	7.05	0.83	12.37
	Parapercis colias	Blue cod	0.05	0.37	5.5	0.66	9.65
	Latridopsis ciliaris	Blue moki	0.11	0.28	4.35	0.62	7.62
	Scorpis lineolata	Sweep	0.11	0.05	1.97	0.38	3.45
2008	*Notolabrus celidotus*	Spotty	1.37	1.28	9.72	0.89	17.01
	Nemadactylus macropterus	Tarakihi	0.43	0.43	8.5	0.88	14.88
	Upeneichthys lineatus	Goatfish	0.41	0.41	7.93	0.88	13.89
	Caesioperca lepidoptera	Butterfly perch	0.27	0.36	7.17	0.7	12.55
	Parapercis colias	Blue cod	0.16	0.4	6.1	0.74	10.68
	Pseudolabrus miles	Scarlet wrasse	0.36	0.16	5.55	0.73	9.71
	Aplodactylus arctidens	Marblefish	0.16	0.18	4.26	0.56	7.46
	Latridopsis ciliaris	Blue moki	0	0.16	2.08	0.41	3.64
	Scorpis lineolata	Sweep	0.11	0.03	1.76	0.35	3.09

2009	*Caesioperca lepidoptera*	Butterfly perch	0.3	0.34	7.04	0.72	13.25
	Pseudolabrus miles	Scarlet wrasse	0.36	0.31	7.04	0.81	13.25
	Parapercis colias	Blue cod	0.16	0.4	6.94	0.75	13.07
	Notolabrus celidotus	Spotty	1.33	1.33	6.28	0.93	11.82
	Nemadactylus macropterus	Tarakihi	0.22	0.27	5.43	0.68	10.22
	Upeneichthys lineatus	Goatfish	0.29	0.17	5.23	0.63	9.85
	Aplodactylus arctidens	Marblefish	0.13	0.28	5.14	0.64	9.68
	Latridopsis ciliaris	Blue moki	0.09	0.21	3.41	0.54	6.41
	Cheilodactylus spectabilis	Red moki	0	0.12	1.97	0.33	3.7
2010	*Nemadactylus macropterus*	Tarakihi	0.49	0.57	8.36	0.97	14.51
	Notolabrus celidotus	Spotty	1.37	1.32	7.79	0.84	13.53
	Pseudolabrus miles	Scarlet wrasse	0.53	0.44	7.54	0.93	13.09
	Parapercis colias	Blue cod	0.17	0.51	6.74	0.82	11.7
	Aplodactylus arctidens	Marblefish	0.27	0.28	5.39	0.71	9.36
	Latridopsis ciliaris	Blue moki	0.25	0.3	5.24	0.73	9.09
	Upeneichthys lineatus	Goatfish	0.34	0.22	5.12	0.74	8.88
	Caesioperca lepidoptera	Butterfly perch	0.24	0.28	4.71	0.67	8.17
	Notolabrus fucicola	Banded wrasse	0.15	0.08	1.96	0.47	3.4
2011	*Parapercis colias*	Blue cod	0.35	0.57	7.85	0.92	13.95
	Notolabrus celidotus	Spotty	1.44	1.37	7.45	0.87	13.24
	Pseudolabrus miles	Scarlet wrasse	0.51	0.32	7	0.89	12.44
	Upeneichthys lineatus	Goatfish	0.32	0.43	6.74	0.85	11.99
	Nemadactylus macropterus	Tarakihi	0.34	0.38	6.57	0.78	11.68
	Caesioperca lepidoptera	Butterfly perch	0.28	0.32	5.58	0.69	9.93
	Latridopsis ciliaris	Blue moki	0.32	0.27	5.33	0.76	9.48
	Aplodactylus arctidens	Marblefish	0.14	0.19	3.65	0.56	6.49
	Notolabrus fucicola	Banded wrasse	0.18	0.01	1.9	0.44	3.38
2012	*Notolabrus celidotus*	Spotty	1.11	1.26	11.23	0.88	20.05
	Pseudolabrus miles	Scarlet wrasse	0.33	0.34	8.42	0.81	15.03
	Parapercis colias	Blue cod	0.15	0.39	7.56	0.76	13.5
	Latridopsis ciliaris	Blue moki	0.14	0.32	6.83	0.67	12.18
	Caesioperca lepidoptera	Butterfly perch	0.18	0.3	6.74	0.61	12.03
	Aplodactylus arctidens	Marblefish	0.07	0.2	4.38	0.51	7.82

	Cheilodactylus spectabilis	Red moki	0.14	0.07	2.91	0.42	5.2
	Nemadactylus macropterus	Tarakihi	0.08	0.05	2.35	0.34	4.2
2013	*Parapercis colias*	Blue cod	0.46	0.68	8.04	1.02	15.14
	Nemadactylus macropterus	Tarakihi	0.46	0.54	7.19	0.97	13.55
	Pseudolabrus miles	Scarlet wrasse	0.47	0.5	7.01	0.95	13.21
	Notolabrus celidotus	Spotty	1.49	1.48	5.99	0.84	11.29
	Aplodactylus arctidens	Marblefish	0.2	0.29	4.5	0.7	8.49
	Caesioperca lepidoptera	Butterfly perch	0.15	0.35	4.38	0.67	8.25
	Upeneichthys lineatus	Goatfish	0.3	0.12	3.56	0.6	6.72
	Cheilodactylus spectabilis	Red moki	0.16	0.19	3.17	0.57	5.98
	Latridopsis ciliaris	Blue moki	0.1	0.21	3.01	0.55	5.66
	Notolabrus fucicola	Banded wrasse	0.16	0.17	2.94	0.56	5.54
2014	*Parapercis colias*	Blue cod	0.6	0.96	10.67	1.06	20.86
	Pseudolabrus miles	Scarlet wrasse	0.48	0.57	7.87	0.99	15.38
	Notolabrus celidotus	Spotty	1.43	1.33	7.05	0.94	13.79
	Nemadactylus macropterus	Tarakihi	0.28	0.38	5.9	0.78	11.54
	Caesioperca lepidoptera	Butterfly perch	0.29	0.31	5.47	0.73	10.69
	Latridopsis ciliaris	Blue moki	0.13	0.14	2.65	0.5	5.17
	Aplodactylus arctidens	Marblefish	0.12	0.07	2.26	0.42	4.42
	Upeneichthys lineatus	Goatfish	0.16	0.05	2.03	0.44	3.96
	Cheilodactylus spectabilis	Red moki	0.06	0.15	2.02	0.46	3.94
	Notolabrus fucicola	Banded wrasse	0.11	0.06	1.98	0.39	3.87
2017	*Parapercis colias*	Blue cod	0.43	0.97	12.28	1.07	22.03
	Pseudolabrus miles	Scarlet wrasse	0.5	0.58	8.73	0.97	15.66
	Notolabrus celidotus	Spotty	1.42	1.21	8.53	0.76	15.3
	Caesioperca lepidoptera	Butterfly perch	0.29	0.38	6.39	0.73	11.45
	Nemadactylus macropterus	Tarakihi	0.27	0.36	6.09	0.69	10.92
	Cheilodactylus spectabilis	Red moki	0.18	0.18	3.78	0.58	6.78
	Notolabrus fucicola	Banded wrasse	0.21	0.12	3.15	0.56	5.65
	Aplodactylus arctidens	Marblefish	0.06	0.13	1.94	0.43	3.47

2020	*Pseudolabrus miles*	Scarlet wrasse	0.58	0.93	8.71	1.1	15.86
	Nemadactylus macropterus	Tarakihi	0.74	0.57	8.5	1.08	15.47
	Parapercis colias	Blue cod	0.39	0.7	7.61	1.03	13.85
	Notolabrus celidotus	Spotty	1.43	1.19	7.26	0.89	13.21
	Upeneichthys lineatus	Goatfish	0.36	0.36	5.77	0.81	10.51
	Caesioperca lepidoptera	Butterfly perch	0.32	0.34	5.54	0.73	10.09
	Cheilodactylus spectabilis	Red moki	0.17	0.28	3.93	0.69	7.16
	Notolabrus fucicola	Banded wrasse	0.19	0.14	3.31	0.48	6.02
2022	*Parapercis colias*	Blue cod	0.66	1.2	8.24	1.05	15.45
	Pseudolabrus miles	Scarlet wrasse	0.6	0.94	7.23	1.06	13.56
	Caesioperca lepidoptera	Butterfly perch	0.6	0.59	6.76	0.98	12.68
	Nemadactylus macropterus	Tarakihi	0.61	0.49	6.53	0.94	12.24
	Upeneichthys lineatus	Goatfish	0.61	0.44	6.34	0.97	11.89
	Notolabrus celidotus	Spotty	1.47	1.24	5.87	0.84	11
	Notolabrus fucicola	Banded wrasse	0.22	0.29	3.61	0.72	6.76
	Scorpis lineolata	Sweep	0.19	0.22	2.95	0.55	5.54
	Cheilodactylus spectabilis	Red moki	0.19	0.22	2.94	0.63	5.52
2024	*Parapercis colias*	Blue cod	0.28	0.98	9.86	1.31	17.21
	Pseudolabrus miles	Scarlet wrasse	0.46	0.65	7.37	1.01	12.86
	Nemadactylus macropterus	Tarakihi	0.3	0.62	6.89	0.95	12.04
	Caesioperca lepidoptera	Butterfly perch	0.46	0.53	6.86	0.91	11.98
	Notolabrus celidotus	Spotty	1.34	1.26	5.36	0.87	9.36
	Cheilodactylus spectabilis	Red moki	0.3	0.39	5.23	0.84	9.13
	Upeneichthys lineatus	Goatfish	0.25	0.38	4.88	0.78	8.53
	Notolabrus fucicola	Banded wrasse	0.16	0.21	3.4	0.59	5.94
	Scorpis lineolata	Sweep	0.16	0.26	3.27	0.56	5.71