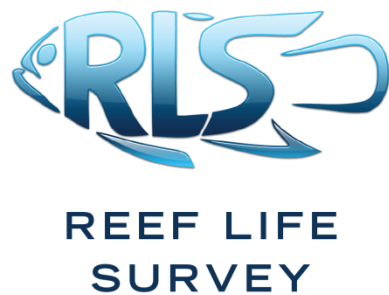# Reef Life Survey Assessment of Biodiversity in Northern New Zealand Marine Reserves and Associated Coastlines

Report for New Zealand Department of Conservation

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PUBLICATION SUMMARY

Cover images (top left to bottom right): RLS survey at Curtis Island in the Kermadecs; *Evechinus chloroticus* at Northland; Dougall Rock, Raoul Island at Kermadec MR; *Arctides antipodarum* at Northland; *Jasus edwardsii* at Bay of Islands, *Sagmariasus verreauxi* at Bay of Islands, *Lessonia variegata* at Northland.

Executive Summary

This report summarises results of subtidal surveys of marine animals and macroalgae undertaken between September 2012 and January 2013 within northern New Zealand marine reserves and fished coastlines. Surveys were undertaken through the Reef Life Survey program (RLS), an initiative that utilises the skills and enthusiasm of volunteer divers trained to a scientific level in underwater visual census techniques.

RLS surveys of 123 rocky reef sites distributed across northern New Zealand and the Kermadec Islands confirmed the global uniqueness of New Zealand's marine natural values. In particular, over one-third of the fish species recorded on rocky reefs naturally occur only in New Zealand waters. The most localised of these were the bluefinned butterfish (*Odax cyanoallix*), found only at the Three Kings Islands, and numerous species only known from the Kermadec Islands. Sixteen species of triplefin (Tripterygiidae) were also recorded, a consequence of New Zealand's position as the global diversity hotspot for members of this family. None of the New Zealand triplefin species are naturally occurring outside the country, although three species have been introduced to Australia over the past century.

The Kermadec Island fish fauna was highly distinctive in a global context, with closest similarity to the fauna at Norfolk Island, rather than to northern New Zealand or Minerva Reefs, the closest islands at 750 km distance. Very little (<5%) coral cover was encountered. A dispersal barrier between the Kermadec Islands and Minerva Reefs apparently blocks movement of tropical fish and invertebrate larvae to the region, presumably due to an absence of southward flowing currents.

The fish and mobile invertebrate communities, as well as cover of macroalgae and sessile invertebrates, were assessed within eight marine reserves: Cape Rodney – Okakari Point, Whanganui A Hei, Kermadec Islands, Poor Knights Islands, Tawharanui, Te Matuku, Te Paepae o Aotea (Volkner Rocks), Tuhua/Mayor Island. In general, sites within reserves were characterised by a greater biomass of large fishes than would be expected based on observations from sites in fished areas outside reserves. Fish biomass was modestly elevated in most marine reserves but substantially greater than expected in the Kermadec Islands and Poor Knights reserves. In a global analysis of 87 marine protected areas (MPAs) surveyed by RLS divers, the Kermadec Islands Marine Reserve was identified as one of the four most effective MPAs because it is no-take, enforced, old, large and isolated. Higher total fish biomass and more sharks were observed in this reserve than in any other region investigated across the Pacific.

Four large fish species – the snapper *Pagrus auratus,* stingray *Dasyatis brevicaudata,* porae *Nemadactylus douglasii* and blue cod *Parapercis colias* – showed consistently higher biomass within marine reserves than fished areas. By far the biggest protection effect was shown by snapper, with mean biomass of 2.74 kg per 500 m^2 within marine reserves and 0.070 kg at fished sites across all sites studied, a forty-fold difference.

In line with conceptual model predictions, significant reserve-related effects were evident across the North Island marine reserve network that not only reflected direct interactions between fishers and targeted species (increased large fish biomass; higher snapper and lobster abundance) but also second order interactions (decreased urchin abundance), third order interactions (increased *Ecklonia radiata* cover) and fourth order interactions (decreased 'other foliose' and turf algal cover). Strong underlying ecological drivers associated with fishing were presumably affecting food webs throughout the region.

Four outcomes of the study of reserve effects were unexpected: (i) the consistency (~20%) of the increase in *Ecklonia* cover across North Island reserves relative to nearby fished sites regardless of lobster and urchin density, (ii) an inconsistent response of crustose coralline algae to urchin density, (iii) low cover of other foliose algae and turf algae in marine reserves where levels of urchin herbivory were low, and (iv) fish and benthic invertebrate communities show greater variability between sites within marine reserves than fished locations whereas algal communities show a contrasting trend with reduced spatial variability within reserves. If confirmed at greater scales, the high site to site variability for fishes and invertebrates within reserves indicates that the small proportion of the northern New Zealand coastal zone that is located within the marine reserve network (approximately 89km^2 or 0.2% of the northeastern New Zealand biogeographic region) plays a disproportionately large role in encompassing the full range of national marine biodiversity.

The benefits and limitations of different methods for monitoring ecological indicators in marine reserve networks are discussed, and value of a long-term RLS program for monitoring New Zealand marine reserves outlined. Recommendations for establishing a New Zealand RLS program include:

- Identification and engagement of 'champion' scientists, or well-trained and capable recreational divers, who can act as RLS organisers in terms of directing activities, and helping with IDs and data entry on organised survey weekends,

- An annual timetable for survey weekends developed in consultation with local divers, considering the needs for reasonable dive conditions (activities need to remain enjoyable to be sustainable), and to keep divers actively surveying throughout the year, with modest committed funding to support these activities,

- Identification of appropriate priority sites associated with each reserve, which balance the need for appropriate statistical design, but also consider safety and enjoyment of participants,

- Maintaining regular direct feedback from DOC to divers to ensure participants are reminded of the value of their work for management,

- Close collaboration with the Australian RLS program, as that program can assist with training support and data management in early stages of New Zealand RLS development. The Australian RLS program can also provide advice on lessons learned and feedback in terms of the value of their work for broader understanding of issues relating to biodiversity conservation. Further benefits include a more active collaboration between marine scientists and managers in Australia and New Zealand, increased scientific knowledge of species and ecosystems that span the Tasman, increased power of scientific analyses by combining data on, for example, marine protected areas from both regions, and opportunities for RLS divers on both sides of the Tasman to broaden their skills and experience by participating in activities in the other country (as well as more broadly in international RLS activities).

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INTRODUCTION

New Zealand was arguably the first country worldwide to recognise a need for the establishment of a network of no-take marine protected areas ('marine reserves'), a process facilitated by the *Marine Reserves Act* 1971 (Ballantine 1991, Babcock 2013). New Zealand was also amongst the first countries worldwide to establish a marine reserve, with the Cape Rodney to Okakari Point Marine Reserve declared in 1977. Under the Marine Reserves Act, New Zealand's marine reserves have a primary purpose of providing for scientific research, but are also recognised for the important role they play in protecting the range of marine biodiversity in New Zealand's waters (Department of Conservation and Ministry of Environment 2000, Department of Conservation and Ministry of Fisheries 2005).

Public benefits generated by marine reserve networks are now universally recognised. They include: (i) safeguarding representative examples of local marine biodiversity for future generations, (ii) providing reference sites for scientific research that are relatively free from human impacts, (iii) augmenting opportunities for non-extractive recreational and educational activities, and (iv) providing insurance against fishery stock collapse during an era of changing climate when traditional fisheries management models are approaching known environmental bounds (Ballantine and Langlois 2008).

While still far from complete and small in total area (~7% of coastal waters when large reserves at Kermadec and the subantarctic islands are included), the New Zealand marine reserve system currently encompasses 34 marine reserves and is amongst the largest no-take marine protected area (MPA) networks worldwide. A range of other types of tools are also considered to be MPAs in New Zealand, including some fisheries management tools and cable protection zones (Ministry of Fisheries and Department of Conservation 2008, Department of Conservation 2013). The MPA model applied in New Zealand is, however, unusual in a global context, with a focus on small no-fishing marine reserves rather than large multi-zoned marine parks, as are commonly applied, for example, across Australia. Within the Australian system, small no-fishing areas are typically interspersed within larger 'habitat protection zones' utilised by recreational and in some cases commercial fishers (e.g., the Great Barrier Reef and Lord Howe Island Marine Parks). Regulations within habitat protection zones within marine parks vary greatly between jurisdictions, ranging from open access zones that allow all conventional forms of fishing to line angling from beaches only.

For the purposes of marine protected area planning, New Zealand's territorial sea has been divided into 14 biogeographic regions, based on broad-scale spatial patterns in biodiversity (Ministry of Fisheries and Department of Conservation 2008). The intention of New Zealand's marine protected areas policy (Department of Conservation and Ministry of Fisheries 2005) is that the full range of habitats within each biogeographic region will be represented in at least one marine reserve. Large gaps exist in the existing marine protected area network, however, with some habitats and biogeographic regions remaining unrepresented in the network (Department of Conservation and Ministry of Fisheries 2011).

Improved understanding of the ecological benefits of different marine reserve and marine park management models requires observations at a range of locations where protected zones have been established. Marine species interact with each other and their environment in such complex ways that, using current knowledge, any predictions of ecological changes following establishment of a new MPA will likely prove inaccurate (Langlois and Ballantine 2005).

Removal of fishing pressure, and the consequent recovery of large predatory species that were formerly targeted by fishers, causes changes to the food web at various trophic levels. Thus, fishing impacts ripple synergistically and antagonistically across ecosystems, resulting in biological communities and habitats that can differ markedly from unfished areas (Shears and Babcock 2003, Edgar et al. 2009).

Four types of fishing impact on reef communities have been described: (i) direct removal of targeted species, (ii) increase in numbers of the prey of targeted species once predators are removed, (iii) decrease in abundance of macroalgae and small animals consumed by the prey of targeted species, and (iv) changed abundance of organisms affected by decreased plant cover. Seminal studies demonstrating such effects have largely been undertaken within New Zealand marine reserves, where prohibitions on fishing have resulted in: (i) increased populations of exploited species such as snapper and rock lobsters, (ii) decreased urchin numbers following increased predation pressure from rock lobsters, (iii) transformation of urchin barrens to kelp forests following reduction in urchin grazing pressure, and (iv) increased numbers of amphipods and other organisms associated with kelp. Such trends are, however, far from universal across the New Zealand marine reserve system (Langlois and Ballantine 2005).

The present study was initiated with the primary aim of assessing ecological differences in New Zealand marine reserves compared to fished coastlines, to allow more informed predictive models when assessing benefits and costs associated with expanding reserve networks. In order to address the questions of most importance to managers and the public when considering new marine reserves, such as how large should reserves be and how far apart should they be spaced, data are needed from numerous reserves that encompass the full range of potential options. The present study extends studies undertaken in Australia (Edgar and Stuart-Smith 2009), where comparative data using similar Reef Life Survey methodology have now been obtained from 24 MPAs (Edgar et al. in prep). Clearly, data obtained from MPA studies conducted in similar habitat types on both sides of the Tasman usefully inform management decisions in both countries through increased generality in models and accuracy in predictions.

Because of consistency in survey methodology across numerous New Zealand marine reserves, data described in the present study also provide an integrated snapshot of the distribution of shallow reef biodiversity across northern New Zealand, and ecological similarities between the different reserves. Our investigation was primarily focused on regional- rather than local-scale patterns. An important caveat of the study is that patterns associated with individual reserves could be biased by idiosyncratic location of survey sites given that relatively few sites were assessed in most reserves. Also, the Poor Knights Islands and Tuhua/Mayor Island Marine Reserves have distinctive biotas that differ in some respects from associated reference sites (the Mokohinau Group and eastern Mayor Island, respectively).

Reef Life Survey

Reef Life Survey was initially established in December 2007 in Australia to explore the potential for a cost-effective, nationally standardised marine biodiversity monitoring program that engages the best members of the recreational SCUBA diving community. With an extensive roll-out of new MPAs around Australia, and the limited resources within the responsible management agencies to monitor these, RLS aimed to provide a standardised approach to biodiversity monitoring that was highly cost-effective. Methods were designed to complement existing scientific monitoring programs. The program also aimed to engage members of the recreational diving community, who as a group

observe more subtle changes in the subtidal realm than other stakeholder groups and managers, but require an appropriate framework to make observations useful for scientific application.

A three-year initial pilot project funded by the Australian Government's 'Commonwealth Environment Research Facilities' initiative developed and tested appropriate data collection methodology, training protocols, and data entry and management procedures. Following this pilot project, which was regarded by all participants and stakeholders as highly successful, the network of RLS divers has expanded. Divers join the program only after substantial one-on-one training by experienced trainers, and then provide data that are entered into a database with in-built checking mechanisms.

Two current data collection mechanisms are used to generate field data. All trained RLS divers undertake regular surveys of their local dive sites and visit other areas on an ad-hoc basis, providing opportunistic monitoring data and contributing to a broader synoptic picture of reef biodiversity. Such activities are not managed, but remote support is provided for divers with identification or data entry queries, and data are all sent in to the University of Tasmania team responsible for RLS data management, where they are checked and managed within the global RLS dataset.

The second data collection mechanism, which is more structured and provides the greatest opportunities for targeted field surveys in any location (particularly remote or difficult to ordinarily dive locations), is based around organised survey expeditions. In Australia, these typically involve a group of previously-trained RLS divers gathering at the target location for survey dives from Friday through till Monday. The same priority sites are resurveyed on an annual basis during these targeted surveys, to identify long-term changes in reef condition. A scientist or experienced survey diver leads the team and ensures that sites are prioritised according to local management needs, and provides on-ground support with logistics, identifications and data entry. Such survey expeditions in Australia typically involve RLS divers covering their own transport costs to get to the location, bringing their own dive gear, and covering the majority of meal costs, but diving (boat charter and tank fills) and accommodation costs are covered by RLS grant funding whenever possible.

A major component of the pilot project was assessment of the quality of data produced by trained divers. An analysis of the similarity of data collected by trained volunteers at the same sites (but different transect lines) as experienced scientists showed that the variation attributable to diver experience was negligible in comparison to the differences between sites and regions (Edgar and Stuart-Smith 2009). Further to this, volunteers who generated the best quality data (as assessed by similarity to data produced by experienced scientists) tended to be the most productive divers, in terms of greatest ongoing involvement and number of transects surveyed (Edgar and Stuart-Smith 2009). This pattern has continued, with most of the best original divers still actively contributing to RLS over five years later. Thus, there is a degree of self-regulation to RLS in terms of quality and commitment, where the keenest volunteers tend to also collect the best data and stay involved for the longest time. A critical step in the RLS training process is the selection and focussing of resources on divers who are most enthusiastic and likely to contribute high quality data through the long-term.

Thanks to active RLS volunteer teams in Australia and Spain, and support from numerous overseas collaborators and institutions, the RLS database now includes data from over 5,700 transects in 75 of the world's ecoregions (Spalding et al. 2007), 40 countries and spanning 133° latitude, in all ocean basins. Some Australian locations (e.g. Rottnest Island) have been monitored annually for up to six consecutive years, with time-series monitoring data accumulating for key MPAs and urban regions around Australia. Such widespread consistency in data collection adds enormous value to any further surveys using these methods. For example, application of RLS methods for New Zealand reef

surveys allows data from that region to be assessed in a global context. Furthermore, New Zealand data can then also contribute to broad-scale analyses elsewhere.

Report objectives

This report, commissioned by the New Zealand Department of Conservation, is based on data collected through voluntary efforts of divers participating in the RLS program, and has two key objectives:

(i)   To provide a summary of results from recent surveys of New Zealand marine biodiversity, including assessment of protection effects,

(ii)   To assess the suitability of the Reef Life Survey methodology to assessment of New Zealand marine biodiversity, and provide recommendations for establishing long-term monitoring of New Zealand marine biodiversity through Reef Life Survey

METHODS

One hundred and fifteen rocky reef sites were surveyed by Reef Life Survey (RLS) dive teams along the North Island of New Zealand and at the Kermadec Islands from the 30th September 2012 to the 1st January 2013 (Figure 1, Appendix 1, 2). Data from 12 sites that were surveyed previously (mostly in 2009) from the Poor Knights Islands and Leigh have also been included in analyses for this report (with four of these sites resurveyed in 2012). Sites were chosen at approximately equal intervals apart within marine reserves, ideally with similar numbers at nearby locations outside reserves as reference sites, and with more sites at the larger more important reserves; however, logistic constraints associated with time and seastate almost inevitably meant that compromises from the ideal design were needed, particularly at locations where wave action was too excessive for diving at most potential sites. Of 123 sites analysed, 66 were in or near eight marine reserves (Appendix 1), while 51 were interspersed along the coast and islands, and a further six were at the Three Kings Islands (Appendix 2). Sites were spread across three of the shallow marine ecoregions of the world (Spalding et al. 2007): Kermadec Islands, Three Kings – North Cape, and North-eastern New Zealand, which correspond to three of the biogeographic regions recognised in New Zealand for the purposes of marine protected area planning (Ministry of Fisheries and Department of Conservation 2008).

All surveys were undertaken using the standardised underwater visual census methods applied globally by Reef Life Survey. A summary of these methods is provided here, but further details can be downloaded at: http://reeflifesurvey.com/files/2008/09/NEW-Methods-Manual_15042013.pdf. Divers and boat crew who participated in surveys all did so in a volunteer capacity, including many who worked with scientific institutions.

Three survey components were completed along the same 50 m transect lines, each laid as close as possible to the depth contour of the starting position on predominantly rocky reef habitat. Depth contours surveyed at a site were selected to encompass a wide depth range, but recognising constraints associated with diving bottom time (<22 m depth, other than HMNZS Canterbury at 27 m), excessive swell at shallow depths at some sites, and the reef/sand edge, which was typically shallow at sheltered sites. Multiple transects were usually surveyed at each site, generally parallel at different depths when the reef was sufficiently wide. Underwater visibility and depth were recorded at the time of each survey, with visibility measured as the furthest distance at which large objects could be seen along the transect line, and depth as the average depth (m) at which the transect line was laid.

Plate 1 Sailing vessel *Reef Dragon* with divers undertaking RLS surveys at Raoul Island, Kermadec Islands Marine Reserve.
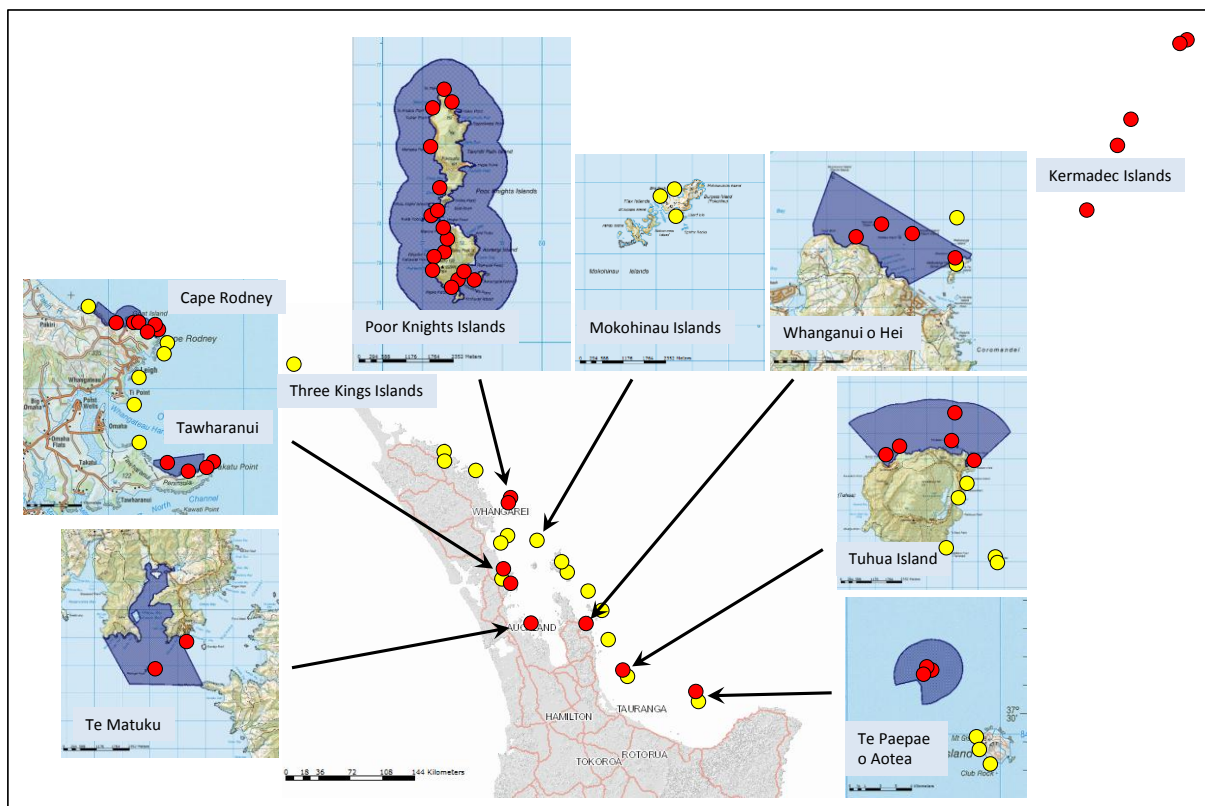
Figure 1 Map of New Zealand sites surveyed by Reef Life Survey divers. Note that overlapping sites are hidden (N=123). The Mokohinau Islands were used as reference sites for the Poor Knights Islands. The map of sites can be zoomed and explored on the RLS website (http://reeflifesurvey.com/sites/) and the Australian Ocean Data Network (http://portal.aodn.org.au/aodn/). Site coordinates are provided in Appendix 1.

Fish Surveys (Method 1)

All fish species sighted within 5 m blocks either side of the transect line were recorded on waterproof paper as divers swam slowly along the line. The number and estimated size-category of each species was also recorded. Size categories used were 25, 50, 75, 100, 125, 150, 200, 250, 300, 350, 400, 500, 625 mm, and above, which represent total fish length (from snout to tip of tail). All species sighted within the blocks were recorded, even for those with unknown identity. Digital photographs were often used to later confirm identities with appropriate taxonomic experts. In occasional circumstances when no photograph was available, taxa were recorded to the highest taxonomic resolution for which there was confidence (e.g. genus or family). Other large pelagic animals such as mammals, reptiles and cephalopods were also recorded during the Method 1 fish survey, but were excluded for analyses focussing on fishes. Species observed outside the boundaries of the survey blocks or after the fish survey had been completed were recorded as 'Method 0'. Such records are a presence record for the time and location but were not used in quantitative analyses at the site level. 'Method 0' sightings can also be made of invertebrates or any other taxonomic groups.

Macroinvertebrate and cryptic fish surveys (Method 2)

Large macro-invertebrates (molluscs, echinoderms and crustaceans > 2.5 cm) and cryptic fishes were surveyed along the same transect lines set for fish surveys. Divers swam along the bottom, up each side of the transect line, recording all mobile macroinvertebrates and cryptic fishes on the reef surface within 1 m of the line. This required brushing aside the kelp canopy when present and searching along crevices and undercuts, but without moving rocks. Cryptic fishes include those from particular, pre-defined families that are inconspicuous and closely associated with the seabed (and are thus likely to be overlooked during general fish surveys). The global list of families defined as cryptic for the purpose of RLS surveys can be found in the online methods manual. As data from Method 2 were collected in blocks of a different width to that used for Method 1 and were analysed separately from those data, individuals of cryptic fishes known to already be recorded on Method 1 were still recorded as part of Method 2. Sizes were estimated for cryptic fishes using the same size classes as for Method 1. Standard RLS methods have recently been updated to include size estimates for rock lobster and abalone (*Haliotis*) species, but these were not included in data reported here.

Photo-quadrats of benthic cover (Method 3)

Information on the percentage cover of sessile animals and macroalgae along the transect lines set for fish and invertebrate surveys were recorded using photo-quadrats taken every 2.5 m along the 50 m transect. Digital photo-quadrats were taken vertically-downward from a height sufficient to encompass an area of approximately 0.3 m x 0.3 m. Photo-quadrats were lacking at some sites due to poor image quality or camera failure. In total, images were available for 107 of the 123 sites investigated for fishes and benthic invertebrates.

The percentage cover of different macroalgal, coral, sponge and other attached invertebrate species was obtained from photo-quadrats by recording the functional group observed under each of five points overlaid on each image, such that 100-110 points were counted for each transect. To provide a percentage cover estimate for that transect, the number of points counted for each functional group was divided by total points less undefined shadow and tape areas.

Functional groups used were the standard 40 groups used for broad-scale analysis of RLS data; however, as listed in Appendix 3, only 30 of these groups were present in New Zealand photo-

quadrats (no acroporid or pocilloporid corals, for example). The label *Ecklonia radiata* applies to a category that encompasses all laminarian kelps, and includes a few records of *Lessonia variegata*; however, those records comprise a very small proportion of the total, so *Ecklonia* is used for brevity. With greater time investment by a specialist operator than was achievable for this report, higher taxonomic resolution analyses are possible using the photo-quadrat set. Images have been archived and are available for processing at any resolution at a future date.

Plate 2 Diver undertaking RLS survey at Curtis Island in the Kermadecs.

Statistical analyses

Univariate analyses

A range of univariate metrics were calculated from survey data: total fish biomass, biomass of large fishes (> 25cm), total abundance of fishes, fish species richness, total abundance of sea urchins (all species), abundance of rock lobsters (Palinuridae and Scyllaridae), and percent cover of *Ecklonia*, fucoid kelps (e.g. *Carpophyllum* spp., *Xiphophora chondrophylla*), other foliose algae (reds, greens and browns), turf algae (<10 cm height), and sand. All metrics represent mean values per 500 m^2 transect block for Method 1 fishes, and per 50 m^2 transect block for mobile invertebrate and cryptic fishes.

Collection of detailed data on fishes, including species-level identities, length classes and abundance information, allow the calculation of species-specific biomass estimates. The RLS database includes coefficients for length–weight relationships obtained for each species (in some cases genus and family) from Fishbase (www.fishbase.org). When length–weight relationships were described in Fishbase in terms of standard length or fork length rather than total length, additional length-length relationships provided in Fishbase allowed conversion to total length, as estimated by divers. For improved accuracy in biomass estimates, the bias in divers' perception of fish size underwater was additionally corrected using the mean relationship provided in Edgar et al. (2004), where a consistent bias was found amongst divers that led to underestimation of small fish sizes and overestimation of large fish sizes. Note that estimates of fish abundance made by divers can be greatly affected by fish behaviour for many species (Edgar et al. 2004); consequently, biomass determinations, like abundance estimates, can reliably be compared only in a relative sense (i.e. for comparisons with data collected using the same methods) rather than providing an accurate absolute estimate of fish biomass for a patch of reef.

The significance of marine reserve effects was assessed using univariate PERMANOVA (Anderson et al. 2008). For regional analyses, a nested mixed-model design was applied with the categorical fixed factor '*protection*' (two levels: marine reserve and fished coast) crossed with another fixed factor '*location*' (two levels: offshore island, coastal), with the random factor '*reserve*' nested hierarchically below (six levels). *Reserve* levels comprised the six marine reserves where surveys were completed both within the reserves and at adjacent reference sites, with three coastal locations (Cape Rodney – Okakari Point, Whanganui A Hei, and Tawharanui) and three offshore locations (Te Paepae o Aotea, Poor Knights Islands, and Tuhua/Mayor Island). Mean data were calculated from all transects at each site, and these values considered separate replicates in PERMANOVA runs. Residuals were permutated under a reduced Type III (partial) model (Anderson et al. 2008). One-tailed tests were applied as tests related to one-directional hypotheses (e.g. fish biomass is greater in marine reserves than fished coasts).

PERMANOVA calculations utilised a similarity matrix based on Euclidean Distance, with log (x+1) transformation applied except for percent cover data, which were not transformed. Analytical outputs (sum of squares, mean squares, *F*-values) were thus identical to those calculated using mixed-model analysis of variance (ANOVA) other than *P*-values, which were calculated using permutation procedures rather than with Gaussian distribution assumptions (Anderson et al. 2008). The important reserve effect test involved an F-value calculated with *protection* as numerator and *location* as denominator. The power of this test of consistency in response between reserves was much lower than if calculated on a region-wide basis using residual error, but the test was more general.

Multidimensional scaling (*MDS*)

Non-metric multidimensional scaling was used to visualise relationships between fish faunas of the three New Zealand ecoregions surveyed and other locations surveyed by RLS in the Southern Pacific, from Easter Island to the eastern Australian coastline. Additional MDS plots were created to assess relationships in community structure among New Zealand marine reserves. Mean biomasses of different species were firstly calculated for each site, then the mean of site means used for similarity matrices associated with ecoregional and marine reserve plots. Bray-Curtis dissimilarity and log (x+1) transformed data were applied for multivariate analyses involving Method 1 fish biomass, and cryptic fish and invertebrate abundance, and Euclidian distance and non-transformed data applied for percent cover digitised from photo-quadrats.

Random forest models

Inside/outside marine reserve comparisons are potentially confounded by idiosyncratic variability between sites surveyed, such that means associated with a group of sites within marine reserves can differ from means associated with reference locations because of natural pre-existing spatial variability that is unrelated to a reserve effect. In a related analysis of five Australian marine protected areas (MPAs) (Edgar and Barrett 2012), effects associated with change through time within MPAs relative to outside (the protection effect) were subtle (4% of total variation) and required considerable power to distinguish, while pre-existing differences between sets of sites inside and outside the MPAs were much more easily observed (8% of total variation).

Such spatial confounding has been reduced in this study by interspersion of reference sites across a system of multiple reserves, as it is unlikely that similar trends in ecological difference between reserve and reference locations occur systematically across marine reserves located tens to hundreds of kilometres apart. We additionally reduced the likelihood of spatial confounding using models that took into account variability in natural features to predict characteristics of marine reserves if that area was open to fishing. Thus, in addition to using the mean of data values from nearby fished sites as a reference for assessing change associated with reserves, we also calculated a reference mean using all fished sites surveyed and their associations with latitude, longitude, depth and environmental factors.

Models were developed using random forests (Breiman 2001), a machine learning protocol that predicted the distribution of total fish biomass and other community metrics for fished locations around northern New Zealand. Each random forest consisted of numerous (2000 in this case) regression trees, where each tree was fitted to a bootstrap sample of the biological data using a recursive partitioning procedure. Random forest analyses also contain cross validation routines based on random subsets of survey sites and covariate predictors that are excluded during development of each tree (the 'out-of-bag' data). Cross validation using out-of-bag data allow estimation of prediction performance (R^2).

Random forests models based on survey data obtained from 66 fished sites studied (146 transects) described relationships between the distribution of 10 environmental and geospatial covariates (Table 1) and mean biomass of all fishes observed per 500 m^2 transect. Six other community metrics (large (>25 cm length) fish biomass, large fish abundance, fish species richness, *Ecklonia* cover, fucoid algal cover, and other foliose algae cover) were similarly investigated. These models were then used to predict the seven community metrics at different transect depths at the 57 marine reserve sites investigated (112 transects). Protection effects for fish data were then calculated for each transect as the difference between values observed and values predicted if the site was fished,

using the log ratio of observed/predicted value (e.g., log(Bm/Bp), where Bm is measured fish biomass and Bp is biomass predicted if the site was fished). Effect size was calculated as the difference between observed and predicted values for photo-quadrat cover data. Because of numerous zero values, random forest models could not be developed for rock lobster density, urchin density, crustose coralline algal cover, or turf algal cover.

Table 1 Covariate data used as predictor variables in global random forest models. The index of population pressure was calculated by fitting a smoothly tapered surface to each settlement point on a year 2000 world population density grid (CIESIN and CIAT 2005) using the quadratic kernel function described by Silverman (Silverman 1986). Populations were screened for a density greater than 1000 people per 0.04 degree cell, and the search radius was set at 3.959 degrees. GDP data were referenced from http://en.wikipedia.org/wiki/List_of_countries_by_GDP_%28nominal%29_per_capita.

Var abbrev	Variable	Units	Scale	Reference (if applicable)
POP_index	Index of population pressure	index	2.46 arcmin (4.6 km)	
BIO_phosphate	mean phosphate	umol/l	5 arcmin	(Tyberghein et al. 2012)
BIO_silicate	mean silicate	umol/l	5 arcmin	(Tyberghein et al. 2012)
BIO_parmean	Photosynthetically available radiation	Einstein/m^3/day	5 arcmin	(Tyberghein et al. 2012)
BIO_SST_mean	mean sea surface temperature	^{o}C	5 arcmin	(Tyberghein et al. 2012)
BIO_SST_range	range of sea surface temperature	^{o}C	5 arcmin	(Tyberghein et al. 2012)
Isolation	Isolated offshore island or nearshore coastal location	yes/no		
Depth	Transect depth	m		
Visibility	Underwater visibility	m		
SiteLat	Site latitude	decimal degrees	0.0001^o	
SiteLong	Site longitude	decimal degrees	0.0001^o	

RESULTS

Ecoregional relationships of New Zealand reef fish fauna

A total of 226,310 individual fishes were sighted during 8,734 separate counts of 116 fish species during Method 1 surveys of New Zealand reefs. During Method 2 surveys, a total of 20,465 cryptic fishes were counted during 1869 transect counts of 43 species, and 41,770 invertebrates from 1831 counts of 24 species of echinoderm, 60 molluscs and 10 crustaceans.

The fish fauna surveyed in New Zealand is unique in the context of the broader South Pacific. The two North Island ecoregions surveyed (Three Kings-North Cape and North-Eastern New Zealand) were most similar to Tasmania and the south-eastern corner of the Australian mainland in terms of fish community structure, but this was no more similar than Tasmania is to northern NSW (Figure 2). The Kermadec Islands is even more dissimilar to other regions surveyed by RLS, with a fish fauna

almost as unique as that recorded at Easter Island in the south-eastern Pacific. Despite lying in slightly closer proximity to the isolated Minerva Reefs (650 km) than to New Zealand, the reef fish community at the Kermadecs displays very little similarity with Minerva. This resulted from the presence of a large number of range-restricted species and near absence of species with tropical affinities at the Kermadecs, whereas Minerva Reefs are coral atolls, characterised by coral reef fish fauna highly similar to the majority of South Pacific coral reef areas.

Fishes that are endemic to the Kermadecs include *Enneapterygius kermadecensis*, *Girella fimbriata*, *Parma kermadecensis* and *Chironemus microlepis*. These species are generally restricted to the Kermadec Islands, although occasional vagrants of the latter two species have been recorded in the New Zealand North Island, Norfolk and/or Lord Howe Islands. Another species of biogeographic interest is *Chrysiptera rapanui*, which was recorded on 72% of transects at the Kermadec Islands. This species is only known from populations at the Kermadec Islands, Austral Islands (southern French Polynesia), and Easter Island. The Kermadec Islands form of *C. rapanui* has very different coloration compared to the blue Easter Island form and is more elongate, so further investigation may indicate that it warrants separate species status (Plate 2).
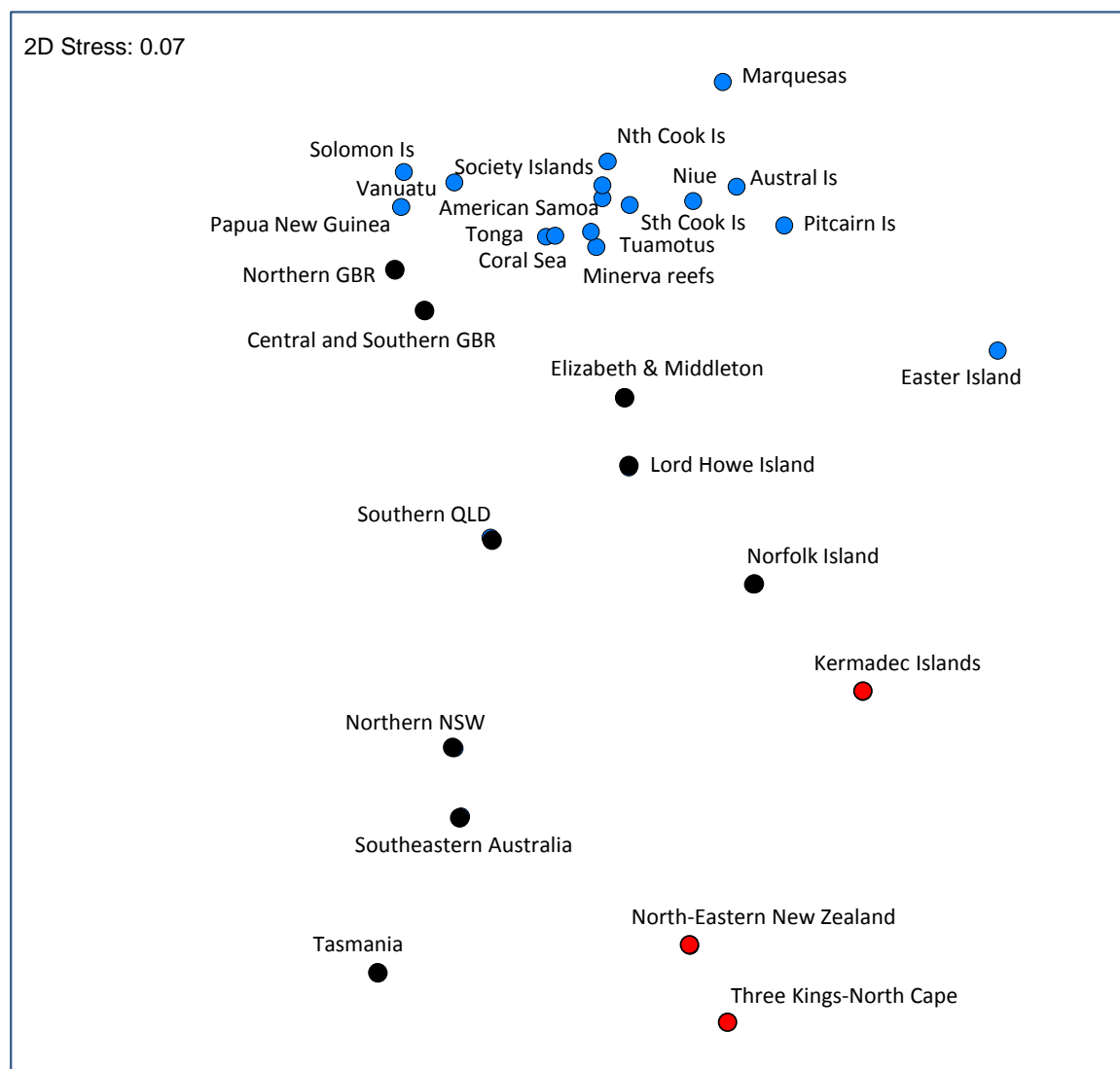
Figure 2 Non-metric MDS plot of fish community structure for locations surveyed by Reef Life Survey divers in the southern Pacific Ocean. Based on Bray-Curtis dissimilarity matrix of log(x+1) transformed fish biomass data. New Zealand locations with red symbols, Australian with black, and other Pacific locations with blue.

Numerous fish species recorded in RLS fish surveys in New Zealand waters are range-restricted or endemic species (29 of the 116). The greatest representation of endemics was in the triplefins (Tripterygiidae), with all 14 species recorded in fish surveys endemic to New Zealand waters (*Forsterygion varium, Grahamina gymnota* and *Forsterygion lapillum* have been included in this number even though introduced to Australia). A further two endemic triplefin species (*Bellapiscis lesleyae* and *B. medius*) were recorded only during the cryptic fish surveys (Method 2). Likewise, the bluefinned butterfish *Odax cyanoallix* was recorded at the Three Kings, which is the only known location for this species. Images of some of the range-restricted species are shown in Plate 2.

When fish community structure of New Zealand sites is considered independently of the broader Southern Pacific region (Figure 3), the majority of sites in the two North Island ecoregions form one large group, with sub-groupings evident for different reserve locations. Offshore islands tend to cluster in the top left of this large group, with greater similarity to each other than with inshore sites despite large distances between them in some cases (e.g. Three Kings and Tuhua/Mayor Island, >500 km apart). Inshore sites lie towards the middle and lower part of the group. Note that sites just outside reserves (considered reference sites) have also been given the same symbols as sites inside the reserves, and thus this plot is not intended for comparison of protection effects, but for assessment of regional patterns in fish community structure.

Apart from the ecological uniqueness of the Kermadecs in relation to the remainder of the region, the Poor Knights Islands and Te Matuku lie at opposite ends of the main group, as the most distinctive among the North Island sites. This pattern reflects a gradient in underwater visibility from turbid inshore sites to locations bathed by oceanic water. Extremely poor visibility (1-2 m) at Te Matuku, which lies in close proximity to Auckland, precluded unbiased fish assessments to the extent that data from that location are only useful in indicating presence of the fishes recorded. Thus, all fish survey results for this reserve need to be carefully interpreted in light of the low effectiveness of the fish survey in poor visibility. Visibility was also relatively poor at Tawharanui (mean 4 m) and Cape Rodney to Okakari Point (CROP; mean 5 m) marine reserves at the time of surveys.

Consistencies among New Zealand sites included eight fish species recorded in all three ecoregions surveyed: *Chromis dispilus*, *Scorpis violacea*, *Pseudocaranx georgianus*, *Parablennius laticlavius*, *Atypichthys latus*, *Seriola lalandi*, *Centroberyx affinis*, and an undescribed species of *Hypoplectrodes*. Of these, *Chromis dispilus* was also amongst the most frequently recorded species, occurring on 78% of all New Zealand transects surveyed (Table 2). The majority of fish species (63%) were only recorded in one of the three ecoregions, while a further 30% were recorded in two ecoregions. Table 2 lists the most frequently recorded species, considering only the ecoregions in which they were observed. A full list of species recorded during fish surveys is provided in Appendix 4.

Plate 3 Range-restricted species recorded during RLS fish surveys in New Zealand. *Chrysiptera rapanui* **(top left, Kermadec form; top right, Easter Island form for comparison),** *Enneapterygius kermadecensis* **(mid left),** *Parma kermadecensis* **(mid right),** *Girella fimbriata* **(bottom left) and** *Odax cyanoallix* **(bottom right).**
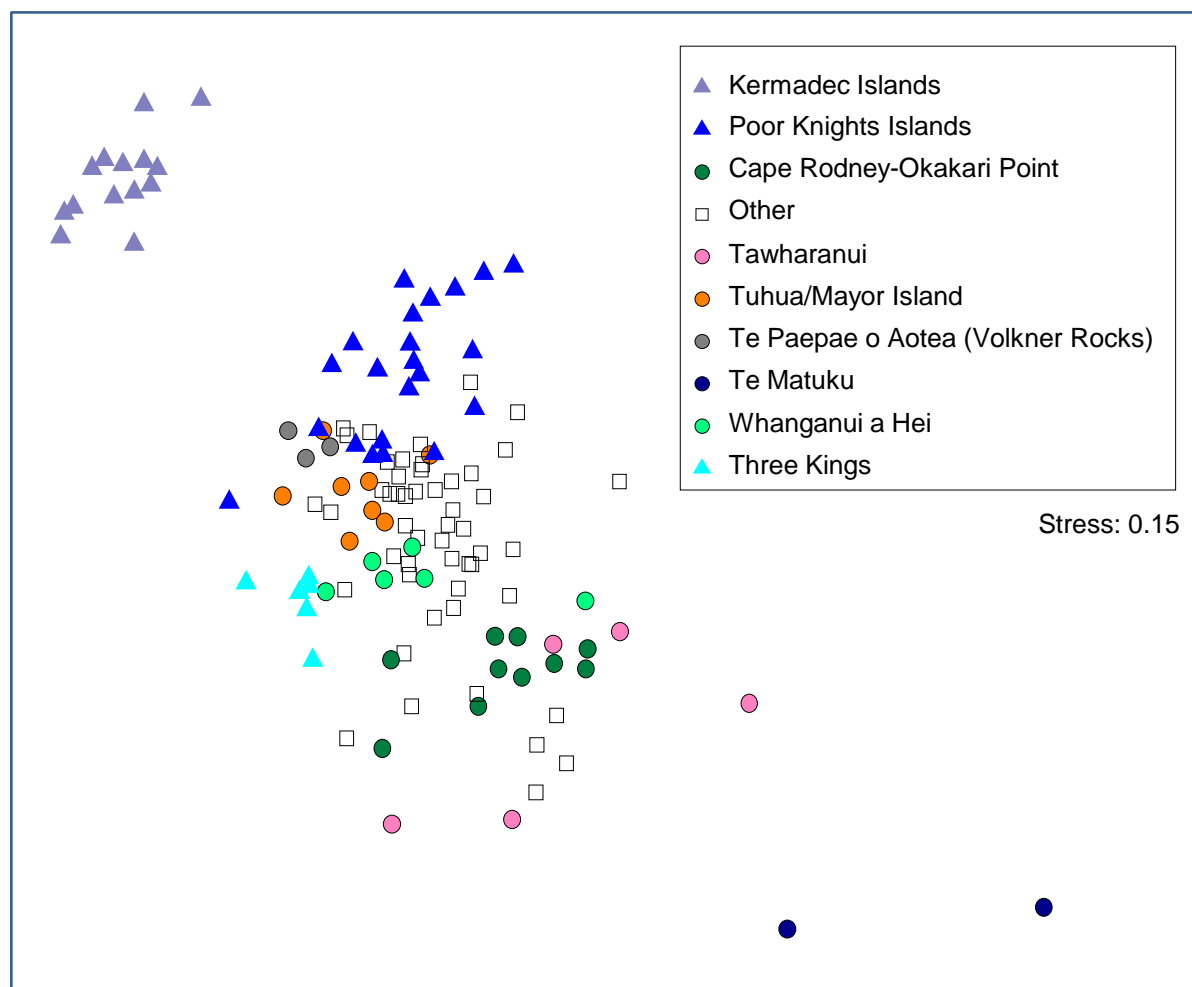
Figure 3 Non-metric MDS plot of fish community structure for 123 sites surveyed by Reef Life Survey divers in 10 locations within three New Zealand ecoregions. Based on Bray-Curtis dissimilarity matrix of log(x+1) transformed fish biomass data. Note that sites nearby, but outside reserves (reference sites) have the same symbols as sites inside the reserves.

Table 2 Most frequently recorded fishes. Ecoregions are Kermadec Islands (Ke), North-Eastern New Zealand (NE) and Three-Kings – North Cape (3K), *F*% is the percentage of transects within these ecoregions, *N* and B are the mean density and biomass per 500 m^2 when present, respectively.

Species	Family	Ecoregions	*F* %	*N*	B (kg)
Kyphosus pacificus	Kyphosidae	Ke	89.7	65.3	46.4
Meuschenia scaber	Monacanthidae	NE, 3K	80.8	5.0	1.9
Chromis dispilus	Pomacentridae	Ke,NE,3K	78.3	426.8	16.7
Chrysiptera rapanui	Pomacentridae	Ke	72.4	93.8	0.3
Parma kermadecensis	Pomacentridae	Ke	72.4	9.6	3.0
Cheilodactylus spectabilis	Cheilodactylidae	NE, 3K	70.5	3.7	2.9
Forsterygion maryannae	Tripterygiidae	NE, 3K	63.2	527.8	0.6
Notolabrus fucicola	Labridae	NE, 3K	62.4	4.0	1.1
Carcharhinus galapagensis	Carcharhinidae	Ke	62.1	6.0	99.0
Stegastes fasciolatus	Pomacentridae	Ke	62.1	10.3	0.7

A particularly notable characteristic of the fish fauna of the Kermadec Islands was a predominance of Galapagos sharks. Total shark densities were much higher than other areas of the Pacific surveyed by RLS divers, with the exception of the outer Pitcairn Island group where a large aggregation was observed at one site (Figure 4). Only a single shark was sighted during surveys of northern New

Zealand coasts and offshore islands, a whaler shark *Carcharhinus brachyurus* observed at Opoupoto Bay, Mayor I.
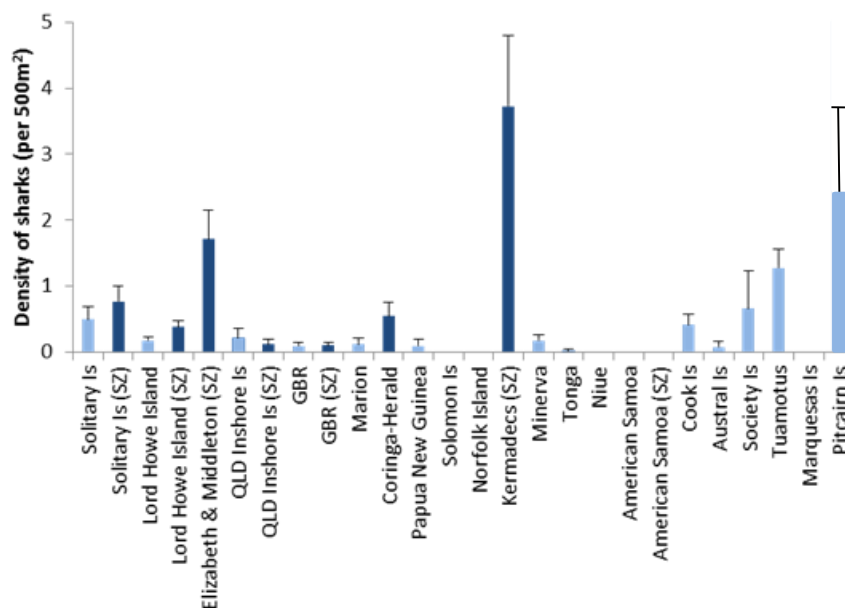
Figure 4 Mean (± SE) density of sharks sighted per transect at different locations surveyed by RLS divers across the Pacific. Marine reserve (MPA no-take sanctuary zone) locations are abbreviated as SZ and shown in dark blue. Standard error bars are based on variability between sites.

The frequent presence of large sharks in the Kermadec Islands Marine Reserve contributed to anomalously high total fish biomass for this region, with an estimated mean value of 212 kg per 500 m^2 transect, a value that exceeded observed values in all other archipelagos investigated across the Pacific (Figure 5). Levels of total fish biomass in other New Zealand marine reserves are discussed below in the section 'Comparison among marine reserves', where fish community metrics from sites are summarised for each reserve individually (Figure 10).
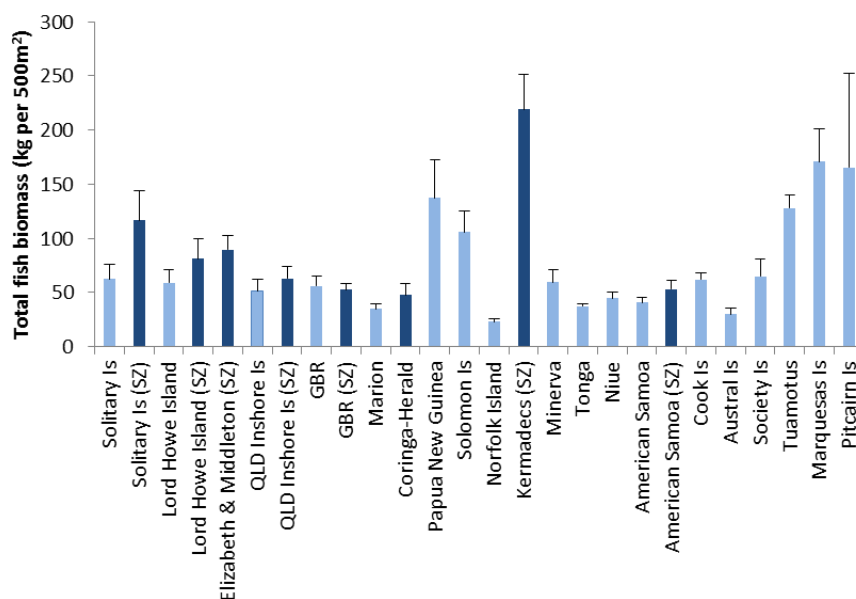
Figure 5 Total fish biomass at different locations surveyed by RLS divers across the Pacific. MPA (sanctuary zones) locations are abbreviated as SZ. Standard error bars reflecting variability between sites are shown.

Cryptic fishes

The cryptic fishes surveyed with the mobile invertebrates in Method 2 include many of the same species as recorded in the Method 1 fish surveys reported above. While such species represent important components of Method 1 fish surveys in terms of overall fish species composition, the intensity of search effort undertaken over a smaller scale in Method 2 means that richness and density are more accurately characterised through this survey component for this subset of small species. Cryptic fishes represent an important component of New Zealand rocky reef fauna due to their relatively high diversity and uniqueness compared to other temperate regions globally. RLS cryptic fish searches recorded 50 species overall, with an average of 4 cryptic fish species and 44 individuals per 50 m^2 transect in northern New Zealand, but these figures were much lower for the Kermadecs (only 2 and 5, respectively) and Three Kings-North Cape (1 and 1, respectively) ecoregions.

Members of eighteen families of cryptic fishes were recorded overall, with the triplefins (Tripterygiidae; 16 species), moray eels (Muraenidae; 6 species) and sea basses/groupers (Serranidae; 5 species) making up the majority. Only a single species was recorded in all three ecoregions (*Parablennius laticlavius* Plate 3), while 76% of species were only recorded in a single ecoregion. The most frequently recorded species are listed in Table 3 (considering only the ecoregions in which they were observed).

Table 3 Most frequently recorded cryptic fishes. Ecoregions are Kermadec Islands (Ke), North-Eastern New Zealand (NE) and Three-Kings – North Cape (3K), *F*% is the percentage of transects within these ecoregions, and *N* is the mean density per 100 m^2 when present.

Species	Family	Ecoregions	*F* %	*N*
Notoclinops segmentatus	Tripterygiidae	NE, 3K	71.7	12.3
Forsterygion varium	Tripterygiidae	NE, 3K	58.8	12.6
Ruanoho whero	Tripterygiidae	NE, 3K	53.2	3.0
Forsterygion malcolmi	Tripterygiidae	NE	50.0	8.8
Forsterygion lapillum	Tripterygiidae	NE, 3K	46.8	10.7
Notocirrhitus splendens	Cirrhitidae	Ke	37.9	1.9
Forsterygion maryannae	Tripterygiidae	NE, 3K	31.3	51.3
Cirripectes alboapicalis	Blenniidae	Ke	31.0	3.7
Parablennius laticlavius	Blenniidae	Ke, NE, 3K	30.2	5.1
Pempheris adspersa	Pempherididae	NE	28.2	15.2

Plate 4 Images of common cryptic fishes. *Notoclinops segmentatus* **(top left),** *Forsterygion varium* **(top right),** *Ruanoho whero* **(bottom left),** *Parablennius laticlavius* **(bottom right).**

Mobile invertebrates

Overall, 101 taxa of mobile invertebrates (Echinodermata, Mollusca and Crustacea) were recorded from across the three ecoregions (Appendix 5). Gastropod molluscs dominated the invertebrate fauna in terms of number of species, with 61 species recorded from 25 families. No molluscs were found in all three regions, and the most commonly recorded species were *Tectus royanus* (Kermadecs) and *Cookia sulcata* (North Island; see Table 4). Echinoderms dominated in terms of density and frequency of occurrence, with *Evechinus chloroticus* recorded on 66% of transects across the two North Island ecoregions at an average density of 135 per 50 m^2 when present. *Centrostephanus rodgersii* was the only invertebrate recorded in all three ecoregions.

Table 4 Most frequently recorded mobile invertebrates. Ecoregions are Kermadec Islands (Ke), North-Eastern New Zealand (NE) and Three-Kings – North Cape (3K), *F%* **is the percentage of transects within these ecoregions, and** *N* **is the mean density per 100 m^2 when present.**

Species	Family	Ecoregions	*F %*	*N*
Evechinus chloroticus	Echinometridae	NE, 3K	65.7	135.1
Tectus royanus	Trochidae	Ke	65.5	15.8
Centrostephanus rodgersii	Diadematidae	Ke, NE, 3K	58.4	35.7
Cookia sulcata	Turbinidae	NE, 3K	54.1	23.4
Tropiometra afra	Tropiometridae	Ke	51.7	58.3
Astrostole rodolphi	Asteriidae	Ke	48.3	5.6
Phyllacanthus parvispinus	Cidaridae	Ke	48.3	5.7
Ophidiaster kermadecensis	Ophidiasteridae	Ke	41.4	4.8
Ceratosoma amoena	Chromodorididae	NE	37.7	2.5
Dicathais orbita	Muricidae	NE	29.5	3.5
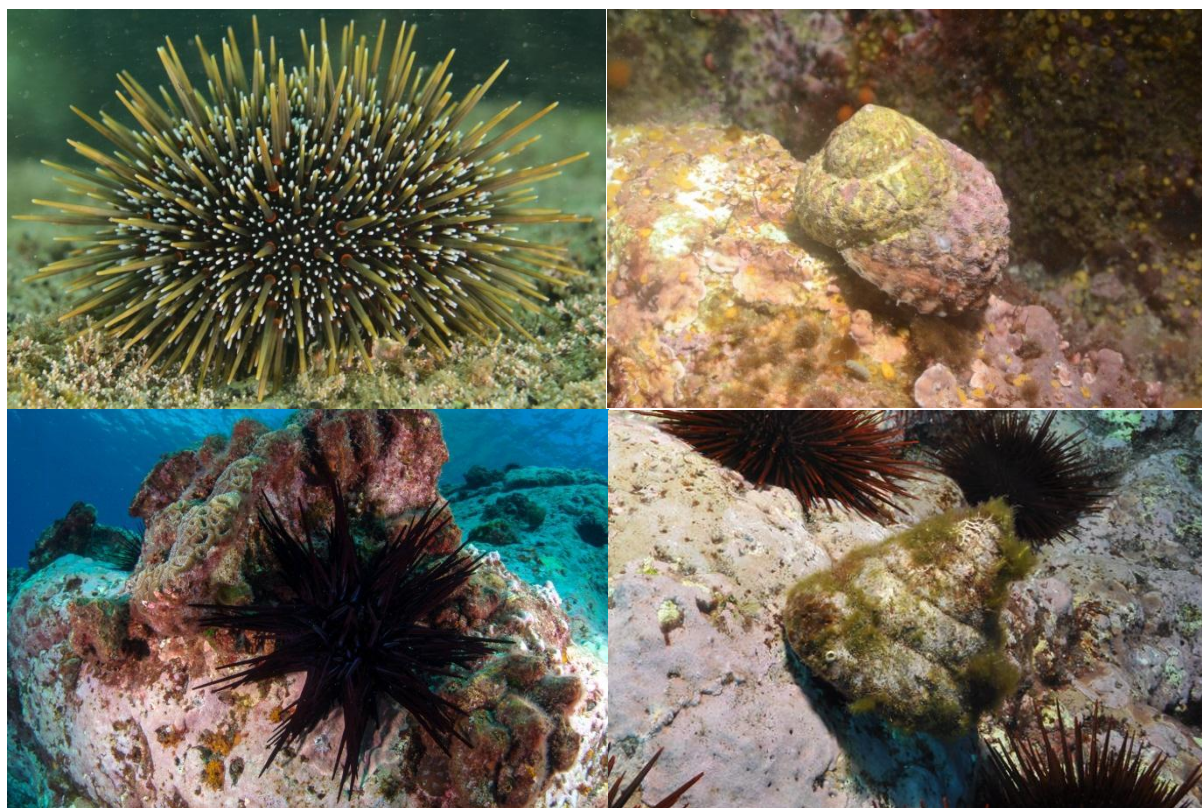
Plate 5 Images of some common mobile invertebrates recorded including *Evechinus chloroticus* (top left), *Cookia sulcata* (top right), *Centrostephanus rodgersii* (bottom left), and *Tectus royanus* (bottom right).

Benthic cover

Analysis of benthic photo-quadrats revealed the dominant substratum cover to differ between the three ecoregions (Table 5). The laminarian kelp *Ecklonia radiata* dominated reefs around much of the North Island, but was absent at the Kermadecs and relatively rare at the Three King Islands. North Island reefs were covered with a patchy mix of algal species, including fucoid kelps, other foliose algae and turf algae. Reefs at the Three Kings Islands were dominated by foliose algae, including a large component of fucoid kelps. Crustose coralline algal barrens predominated on Kermadec reefs, with leathery encrusting algae (e.g., *Peysonnelia* spp.) also common. Despite its subtropical location, <5% cover of hard and soft corals was observed on transects in the Kermadecs, while virtually no corals were recorded in the other two ecoregions.

Table 5 Percent cover (± SE) of important groups of sessile flora and fauna.

Ecoregion	Stony coral	Soft coral	Sponge	*Ecklonia radiata*	Fucoid algae	Foliose algae	Turf algae	Leathery algae	Crustose corallines
Kermadecs	2.1±1	1.4±1	0.9±0.3	0	0.1±0.1	19.6±9.4	17.3±3.7	10±2.3	37.4±6.8
North Island	0.3±0.1	0	3.1±0.5	38.9±2.2	4.9±0.8	14.9±2.9	11±0.9	2.3±0.3	16.5±1.1
Three Kings	0	0	2.6±1.4	10.7±3.6	23.7±5.3	48.3±7.6	0.6±0.3	3±1.9	7.3±2.7

Comparison among marine reserves

Faunal and floral community patterns

Assemblages of fishes observed along 5 m wide transects varied greatly between different marine reserve locations, with extreme outlier values at the Kermadec Islands and Te Matuku (Figure 6). However, fish data from Te Matuku were badly affected by poor underwater visibility (<2 m), so are not comparable to data obtained from other sites. Faunas within reserves were generally in close proximity to associated reference sites within the MDS plot; nevertheless, fish faunas outside marine reserves tended to group closely together overall, while faunas within marine reserves tended to be slightly outlying (Figure 6). The offshore marine reserves (Poor Knights Islands, Kermadec Islands, Tuhua and Te Paepae o Aotea) showed a consistent faunal shift to the bottom right of the plot when moving from nearby fished coast to reserve, while inshore reserves (Cape Rodney – Okakari Point, Tawharanui, Whanganui o Hei) trended to the top and right.
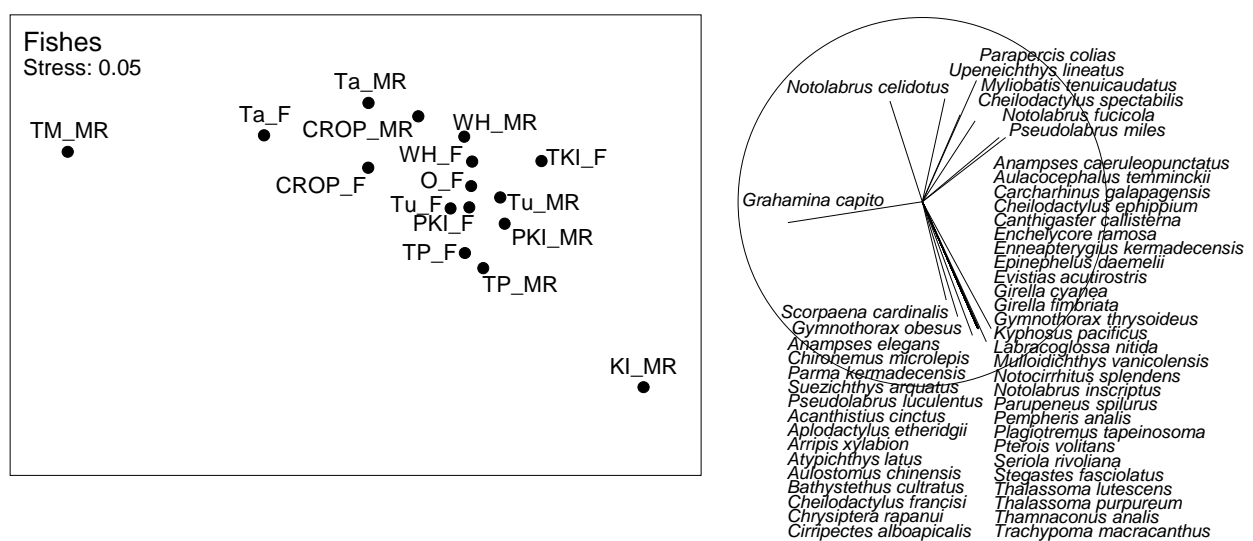
Figure 6 MDS plot of faunal relationships based on mean biomass of fish species at different sites in marine reserves (_MR), fished reference sites adjacent to marine reserves (_F), and fished sites at the Three Kings Islands (TKI_F) and around Northland (O_F). Marine reserves investigated are Cape Rodney-Okakari Point (CROP), Tawharanui (Ta), Whanganui o Hei (Wh), Te Matuku (TM), Poor Knights Islands (PKI), Kermadec Islands (KI) and Te Paepae o Aotea (TP). Vector plots are shown for fish species with high (>0.5) correlations with axes.

Vector plots indicated a major separation between species associated with mainland northern New Zealand (7 species) and species associated with the Kermadec Islands plus offshore New Zealand reefs (43 species), while the threefin *Grahamina capito* was associated with Te Matuku (Figure 6).

Cryptic fish assemblages observed on 1 m wide transects showed similar patterns to fishes on 5 m wide transects, but with greater distinctness of the Three Kings Islands fauna (Figure 7). Protection effects tended to be slightly less pronounced than for the larger fishes, although Poor Knights Islands and Te Paepae o Aotea marine reserves both showed a shift to the right of the plot when associated fished areas are compared to reserves. Vector plots indicated a separation between species largely restricted to the Kermadecs and threefins present in the turbid inshore environment at Te Matuku marine reserve.
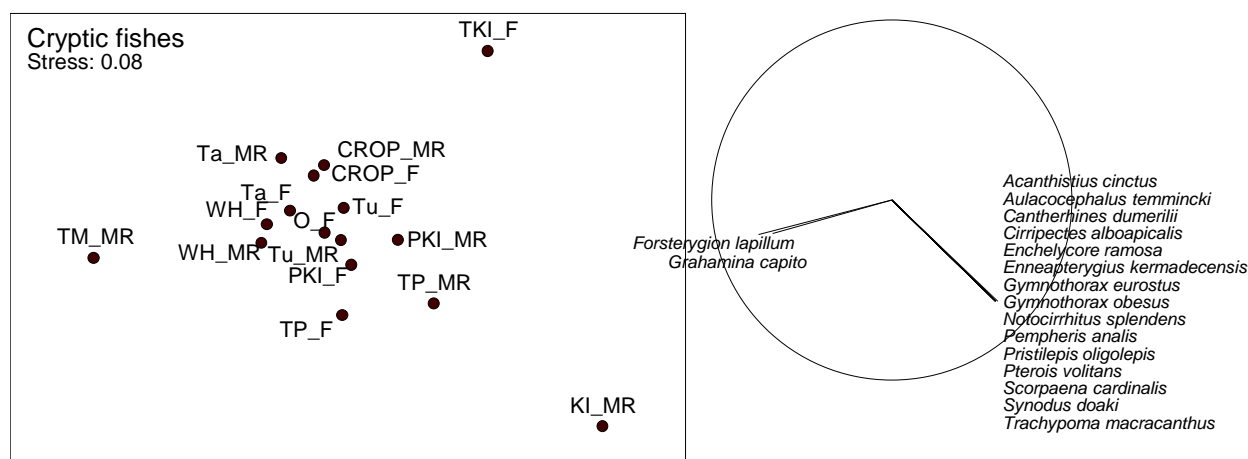
Figure 7 MDS plot of faunal relationships based on mean density of cryptic fish species at different sites in marine reserves (_MR), fished reference sites adjacent to marine reserves (_F), and fished sites at the Three Kings Islands (TKI_F) and around Northland (O_F). Marine reserves investigated are Cape Rodney-Okakari Point (CROP), Tawharanui (Ta), Whanganui o Hei (Wh), Te Matuku (TM), Poor Knights Islands (PKI), Kermadec Islands (KI) and Te Paepae o Aotea (TP). Vector plots are shown for fish species with high (>0.5) correlations with axes.

As with fishes, invertebrate assemblages at Te Matuku and Kermadec Islands marine reserves were highly distinctive; coastal marine reserves tended to be above and to the right of associated fished coasts in plots, and offshore marine reserves tended to be below and to the right of fished coasts (Figure 8). As was the case with fishes on 5 m transects but not cryptic fishes, the invertebrate assemblage at the Three Kings islands differed little from North Island assemblages. Four distinctive assemblages were evident in the invertebrate vector plot: (i) a coastal North Island assemblage that includes the rock lobster *Jasus edwardsii* and topshell *Cookia sulcata*, (ii) an offshore North Island assemblage that includes the seastar *Astrostole scabra*, (iii) a turbid inshore assemblage that includes the seastar *Patiriella regularis*, and (iv) a large Kermadec Island assemblage that includes the predatory seastar *Acanthaster planci* and urchin *Centrostephanus rodgersii*.
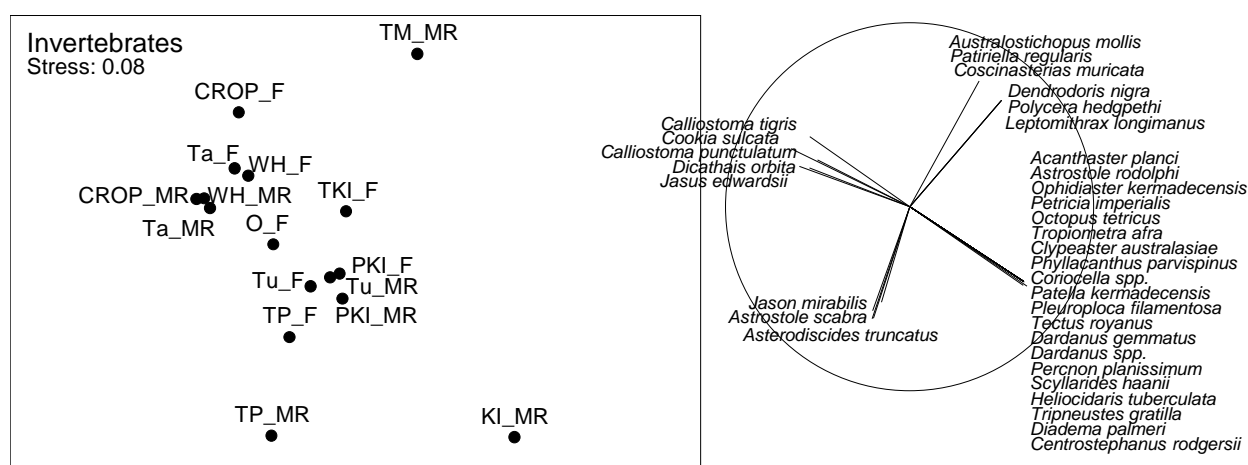
Figure 8 MDS plot of faunal relationships based on mean density of benthic invertebrate species at marine reserves (_MR), fished reference sites adjacent to marine reserves (_F), and fished sites at the Three Kings Islands (TKI_F) and around Northland (O_F). Marine reserves investigated are Cape Rodney-Okakari Point (CROP), Tawharanui (Ta), Whanganui o Hei (Wh), Te Matuku (TM), Poor Knights Islands (PKI), Kermadec Islands (KI) and Te Paepae o Aotea (TP). Vector plots are shown for invertebrate species with high (>0.5) correlations with axes.

On the basis of photo-quadrat images, both the Kermadec and Three Kings Islands were found to possess habitat types distinctly different from North Island sites (Figure 9). Reefs at Three King Islands tended to be dominated by red foliose algae and fucoid kelps, while substratum categories that were disproportionately represented at Kermadec Islands included crustose coralline algae, encrusting leathery algae (e.g. *Peysonnelia*), bare rock, soft corals and encrusting stony corals.
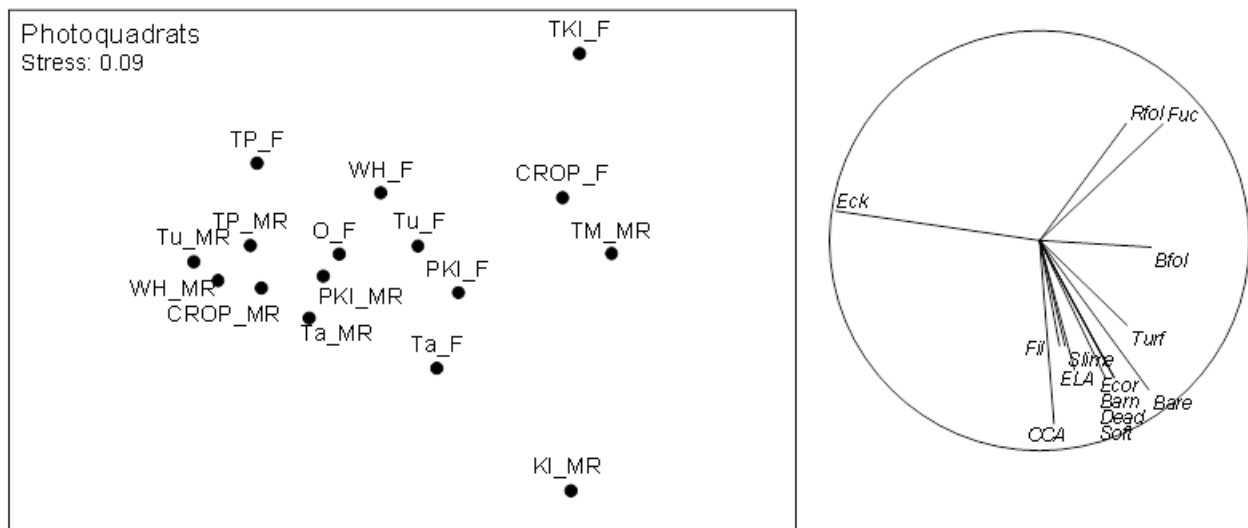
Figure 9 MDS plot of biotic relationships based on percent cover of different habitat types at marine reserves (_MR), fished reference sites adjacent to marine reserves (_F), and fished sites at the Three Kings Islands (TKI_F) and around Northland (O_F). Marine reserves investigated are Cape Rodney – Okakari Point (CROP), Tawharanui (Ta), Whanganui o Hei (Wh), Te Matuku (TM), Poor Knights Islands (PKI), Kermadec Islands (KI) and Te Paepae o Aotea (TP). Vector plots are shown for taxa with high (>0.5) correlations with axes. Taxa abbreviations are explained in Appendix 3.

In contrast to patterns evident for fishes and invertebrates, where North island marine reserve locations tended to be more tightly clumped than fished locations, habitat types showed a greater range of variability at fished locations than reserve locations. North Island marine reserve locations other than Te Matuku were tightly clustered in the MDS plot (Figure 9), in part because of a close association of *Ecklonia* with these reserve sites. When assessed quantitatively using a multivariate index of dispersion (MvDisp in PRIMER, Carr 1996), habitat types at sites within the six North Island marine reserves (excluding Te Matuku) showed greater homogeneity (0.98) than associated fished reference sites (1.14); whereas for Method 1 fishes, cryptic fishes and benthic invertebrates the opposite patterns prevailed, with higher values in marine reserves (1.02, 1.03 and 1.02, respectively) than associated fished sites (0.88, 0.75 and 0.81, respectively).

Differences between marine reserves and fished coasts

Rocky reef communities protected inside marine reserves differed considerably among the reserves surveyed. The Kermadec Islands and Te Paepae o Aotea Marine Reserves had the highest fish biomass recorded, averaging ~220 and 330 kg fish biomass per 500 m^2, respectively (Figure 10). In the Kermadecs, this biomass resulted from high densities of Galapagos sharks (*Carcharhinus galapagensis*), large schools of drummer (*Kyphosus pacificus*), bluefish (*Girella cyanea*) and blue maomao (*Scorpis violacea*). Large kingfish (*Seriola lalandi*) were also common. In Te Paepae o Aotea, vast schools of large blue maomao, two-spot demoiselles (*Chromis dispilus*) and pink maomao (*Caprodon longimanus*) were present at all three sites, along with numerous large blue moki (*Latridopsis ciliaris*) at one site (Small Volkner Rock). Te Matuku had the lowest average fish biomass,

but surveys at that site were compromised by extremely poor visibility, and fish data are not comparable to other surveys as a result.

Similar but more exaggerated patterns are evident when abundance and biomass of large (>25 cm length) fish are considered (Figure 10). For all six marine reserves with associated fished reference sites, large fish density was higher inside the reserve compared to outside, in most cases markedly so.
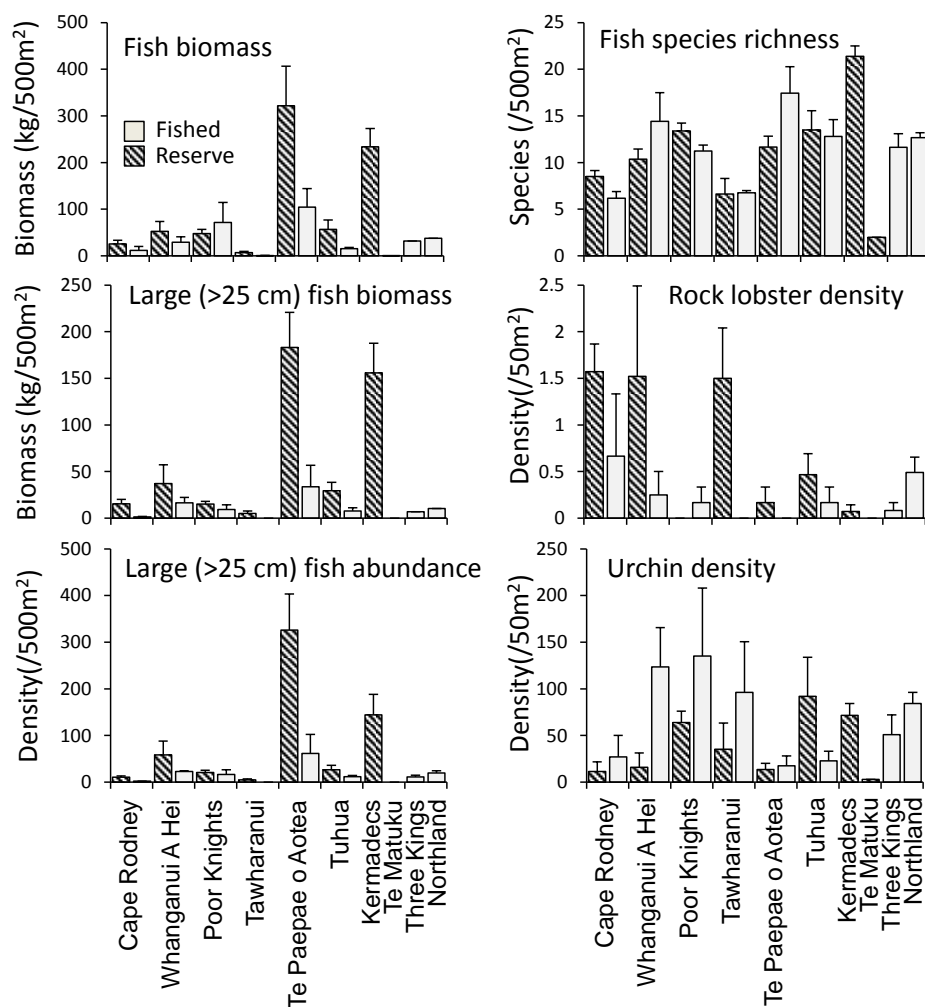
Figure 10 Means (± SE) of six fish community metrics in marine reserves, fished reference sites adjacent to marine reserves, and fished sites at the Three Kings Islands and around Northland. Note: scale of y-axis varies between panels.

Given considerable variability in responses at different marine reserves, no consistent protection effect was evident across the network of six marine reserves (df = 1/8, F =2.19, P = 0.100) when total fish biomass (log (x+1)) was analysed using PERMANOVA. A mixed model design was applied with *protection* (two levels: marine reserve and fished coast) and *location* (two levels: offshore island, coastal) as fixed factors, *reserve* as a nested random factor, and mean transect data from each site entered as replicates. Nor was a significant protection effect detected for density of fishes >25 cm length (df = 1/8, F = 2.23, P = 0.095); however, when biomass of fishes > 25 cm length was examined, a significant protection effect was evident (df = 1/8, F = 4.10, P = 0.018). Thus, an increase in biomass of large fishes occurred consistently across the full marine reserve network.

Four fish species showed significant ($P < 0.05$) increases that were consistent at all marine reserves when mean values for the six reserves with associated fished sites were assessed using t-test– the snapper *Pagrus auratus,* stingray *Dasyatis brevicaudata,* porae *Nemadactylus douglasii* and blue cod *Parapercis colias.* These species were all large species. By far the biggest protection effect was shown by snapper, with mean biomass of 2.74 kg per 500 m^2 within marine reserves and 0.070 kg at fished sites across all sites studied, a forty-fold difference (Figure 11).
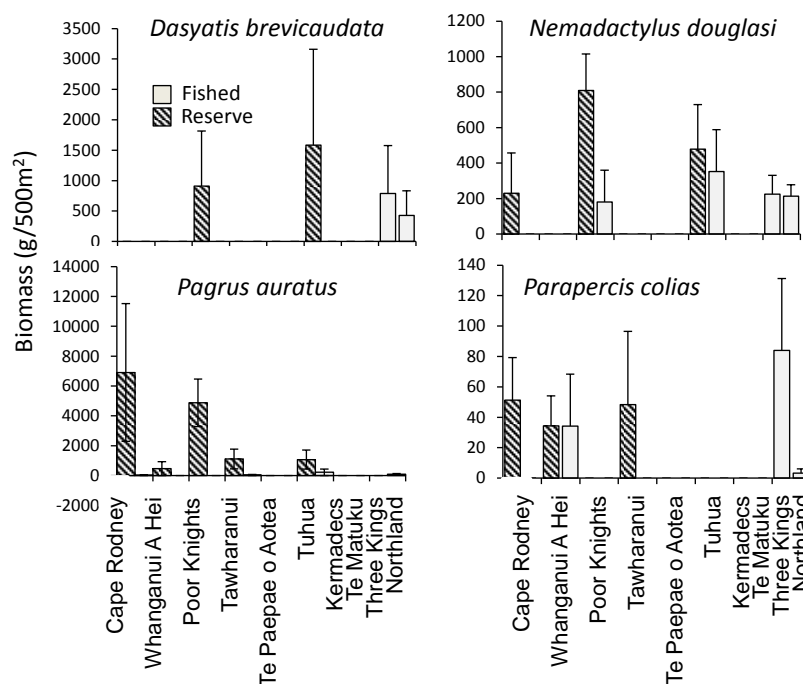
Figure 11 Mean biomass (± SE) of four species in marine reserves surveyed, fished reference sites adjacent to marine reserves, and fished sites at the Three Kings Islands and around Northland. Note: scale of y-axis varies between panels.

Fish species richness was greatest at the Kermadec Islands, which includes sub-tropical species. Species richness was also noticeably higher in reserves at offshore islands around the North Island than in coastal reserves (Figure 10). Some reserves showed higher mean fish species richness in the reserve compared to associated fished sites, while others showed the opposite trend. No consistent protection effect was detected using PERMANOVA (df = 1/8, *F* =0.17, *P* = 0.357).

Rock lobster density was significantly higher inside marine reserves compared to outside (df = 1/8, *F* =13.085, *P* = 0.004). Mean rock lobster density was consistently observed to be ~ 1.5 animals per 50 m^2 transect in the Cape Rodney-Okakari Point, Tawharanui and Whanganui a Hei reserves, while numbers elsewhere were ~0.3 per transect (Figure 10).

Sea urchin density also showed a consistent protection effect across reserves (df = 1/8, *F* =3.87, *P* = 0.045), with lower densities inside reserves than outside, an outcome consistent with the hypothesis that increased lobster numbers in reserves reduce urchin densities. Within marine reserves, sea urchin densities were highest at Tuhua/Mayor Island and the Poor Knights Islands. Densities at these two reserves were similar to fished coastlines at ~80 per 50 m^2 transect.

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The relative importance of different covariates to random forest models developed for different community metrics are shown in Figure 12. Mean photosynthetically active radiation (BIO_parmean) was the most important covariate for predicting total fish biomass, total fish abundance, and fish species richness. However, the most important covariate for predicting large fish biomass was human population density, with lowest large fish biomass values at sites near population centres. Annual sea surface temperature range (BIO_SST_range) was another important predictor for all fish metrics. Visibility was included as a predictor in random forest analyses, but, somewhat surprisingly given its known influence on fish counts along transects, was amongst the four most important predictors only for fish species richness and abundance of large fishes.
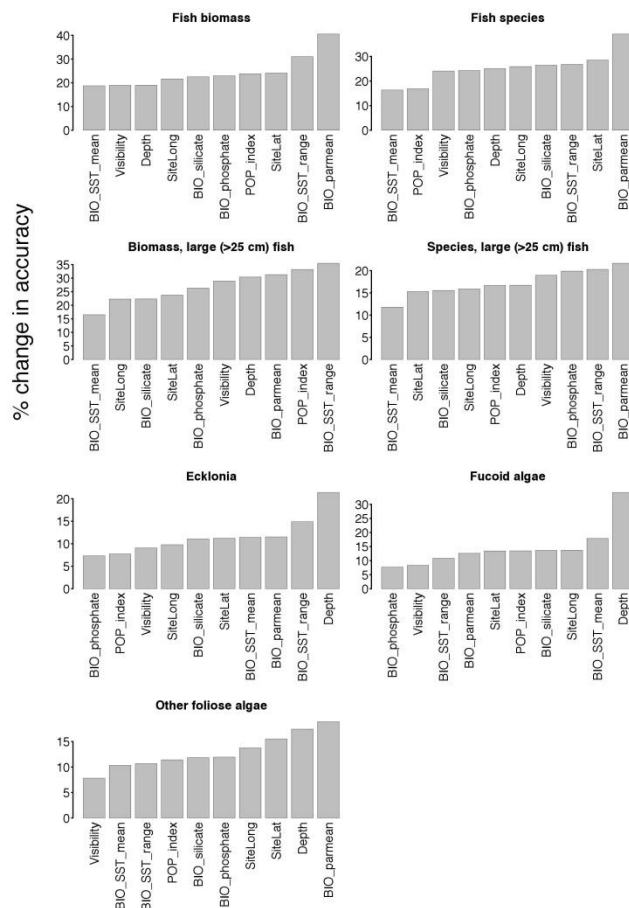
Figure 12 Relative importance of the 10 covariates used in prediction models developed with random forests. Note: scale of y-axis varies between panels.

Comparison of observed values with random forest predictions produced outcomes that agreed well with comparisons based on nearby reference sites. All six marine reserves investigated using random forests had significantly higher biomass and abundance of large fishes (> 25 cm) than expected from data from fished coasts (Figure 13). The mean log ratio for large fish biomass across the six reserves was 1.58, which translates to a 388% increase. Fish species richness showed no consistent trend, while total fish biomass was disproportionately high in all reserves other than Tawharanui, the reserve with the largest protection effect for large fish biomass.
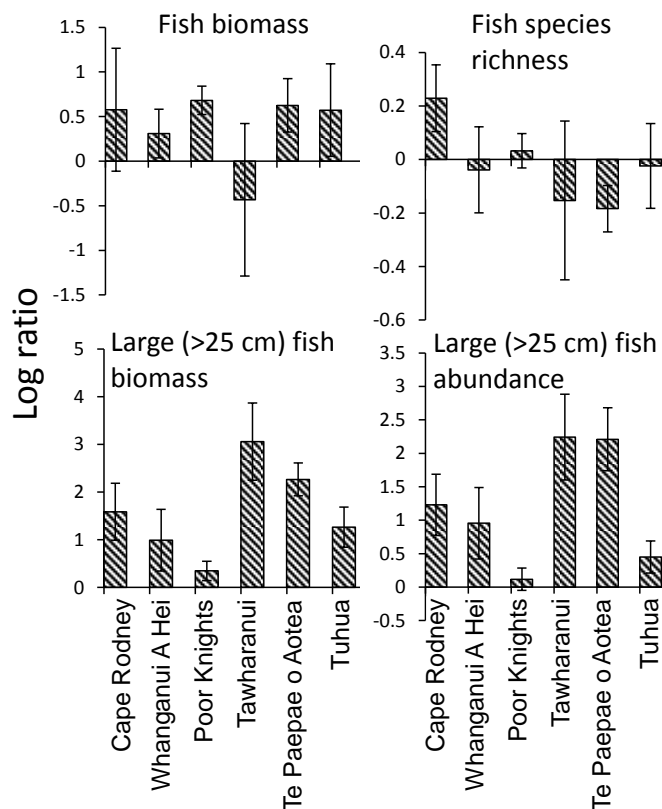
Figure 13 Effect size (± SE) for four fish community metrics at six marine reserves. Effect size was calculated using the log ratio (ln (observed) – ln (predicted)) where predictions were based on random forest relationships with 10 environmental covariates. Note: scale of y-axis varies between panels.

Digital photo-quadrat images revealed strong consistencies in algal components across the marine reserve network (Figure 14). All reserves possessed higher mean cover of *Ecklonia* than associated fished locations, with double the cover at Cape Rodney-Okakari Point, Tawharanui and Tuhua. This increase was found to be highly significant in PERMANOVA analysis across the six marine reserves with fished reference sites (df = 1/8, *F* =7.59, *P* = 0.004).

The total cover of foliose algae other than fucoid and laminarian kelps also showed a highly significant difference between reserves and fished sites, but in this case a decrease was evident (Figure 14; df = 1/8, *F* =18.67, *P* = 0.001). Turf algae additionally showed consistently lower cover in marine reserves than nearby fished sites (df = 1/8, *F* =7.59, *P* = 0.004), while fucoid algae (df = 1/8, *F* =1.54, *P* = 0.142), crustose coralline algae (df = 1/8, *F* =1.16, *P* = 0.152), and sand (df = 1/8, *F* =0.68, *P* = 0.198) showed inconsistent patterns between marine reserves and fished sites, and no significant protection effect overall.
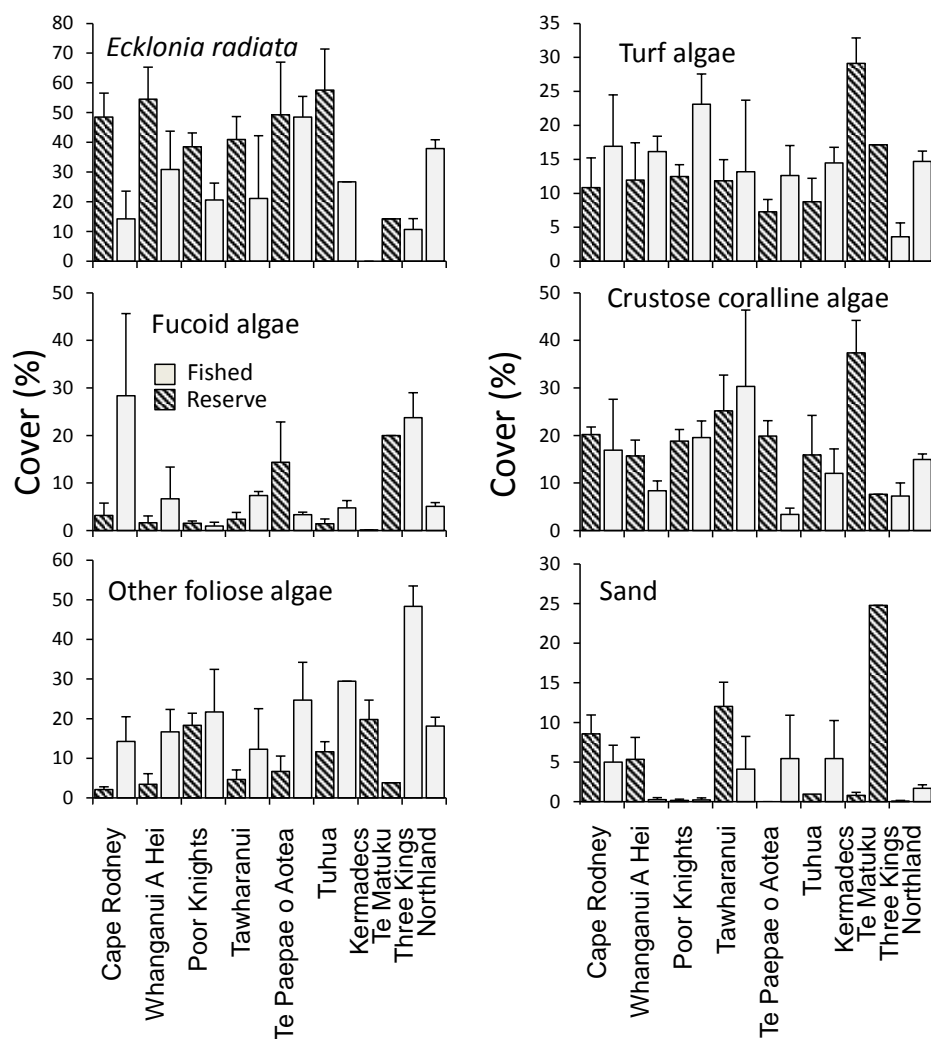
Figure 14 Mean cover (± SE) of different habitat types in marine reserves, fished reference sites adjacent to marine reserves, and fished sites at the Three Kings Islands and around Northland. Note: scale of y-axis varies between panels.

The most important predictor of cover in random forest models for both *Ecklonia* and total fucoid algae was depth, while mean photosynthetically-active radiation was marginally more important than depth for the third photo-quadrat metric examined – 'other foliose algae' (Figure 11). *Ecklonia* cover was ~20% higher at all marine reserves than predicted from models based on data on fished coasts (Figure 15). Fucoid algae showed a variable response, with very high levels at Te Paepae o Aotea marine reserve but little difference from predictions at other reserve locations. Other foliose algae tended to be lower than predictions, particularly at Cape Rodney-Okakari Point and Tuhua.
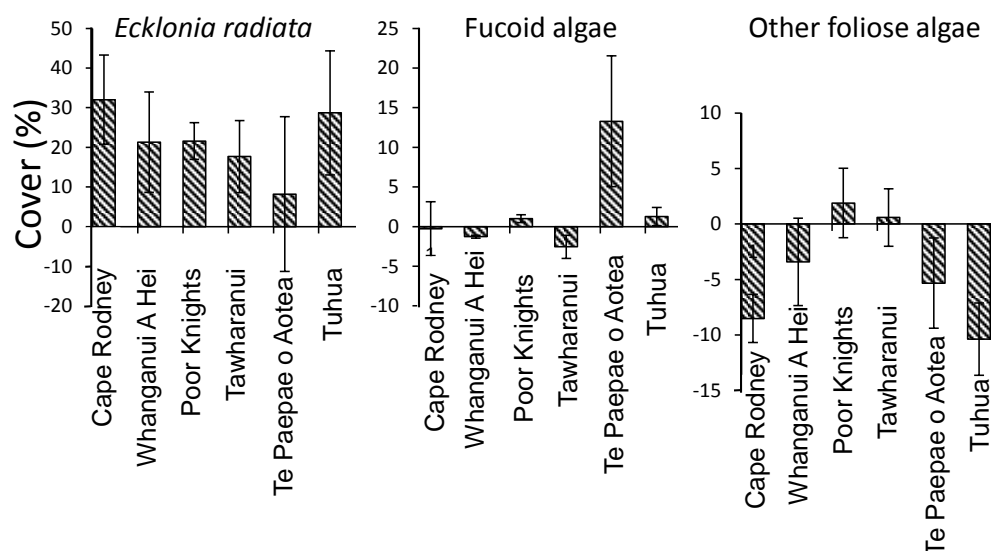
Figure 15 Effect size (± SE), as calculated using difference between observed and predicted values, for three algal cover metrics at six marine reserves. Predictions were based on random forest relationships with 10 environmental covariates. Note: scale of y-axis varies between panels.

Threatened species

The majority of fish (92 of 116) and invertebrate (97 of 101) species recorded on RLS surveys have not been assessed for the IUCN Red List, and thus their global conservation status is unknown. Of the 24 fish species which have been assessed, three are listed as 'Near Threatened' (*Carcharhinus brachyurus*, *Carcharhinus galapagensis* and *Epinephelus daemelii*), one as 'Data Deficient' (*Pagrus auratus*) and 20 as 'Least Concern'. All four assessed invertebrates are listed as 'Least Concern' (*Arctides antipodarum*, *Jasus edwardsii*, *Sagmariasus verreauxi* and *Scyllarides haanii*).

Under the New Zealand Threat Classification System list (NZTCS (Molloy et al. 2002), only 12 and 4 of the recorded fish and invertebrate species have been assessed, respectively. The fishes *Carcharhinus galapagensis*, *Chrysiptera rapanui*, *Enneapterygius kermadecensis*, *Girella fimbriata*, *Odax cyanoallix*, and *Parma kermadecensis* are categorised as 'Range Restricted', while *Stigmatopora macropterygia* is categorised as 'Data Deficient', and *Carcharhinus brachyurus*, *Dasyatis brevicaudata*, *Myliobatis tenuicaudatus*, *Zanclistius elevatus*, and *Epinephelus daemelii* as 'Not Threatened'. Most are additionally qualified as 'Secure Overseas', although *Epinephelus daemelii* is qualified as 'Conservation Dependent' and 'Threatened Overseas'. Three recorded invertebrates, *Cominella quoyana*, *Scutellastra (Patella) kermadecensis* and *Tectus royanus* are listed as 'At Risk' under the sub-category 'Naturally Uncommon' due to being range-restricted. The former was recorded in Rangiwhakaea Bay in North-Eastern New Zealand, while the latter two were common at the Kermadec Islands. *Astraea heliotropium* has also been assessed, but is classified as 'Not Threatened'.

DISCUSSION

The primary objective of this report was to summarise results of surveys of New Zealand marine biodiversity undertaken by divers participating in the Reef Life Survey program, focussing on patterns in marine reserves relative to fished sites. In addition, the report provides an assessment of the suitability of the RLS methodology for application in New Zealand, and recommendations for establishing a long-term marine reserve monitoring program.

Biogeographic patterns

Results summarised here support previous findings of the global uniqueness of New Zealand's marine biodiversity including the Kermadec Islands (e.g., Schiel et al. 1986, Cole et al. 1992, Francis 1996, Brook 1998). Numerous range-restricted species were recorded, and a very high biomass of species uncommon or absent from other southern Pacific locations was observed at the Kermadec Islands and Te Paepae o Aotea.

Few surveys of Kermadec Islands marine life have been undertaken (Schiel et al. 1986, Cole et al. 1992, Gardner et al. 2006), and none covering as wide a range of taxa and islands surveyed using quantitative methods as that undertaken by the RLS team. Despite sharing a distinct faunal component with Norfolk and Lord Howe Islands, the Kermadecs fauna includes fewer species with tropical affinities and a lower overall richness of fish species than other isolated subtropical Pacific locations. The Kermadec reef fauna included a number of range-restricted species, some of which are known to be threatened elsewhere (e.g. black cod and Galapagos sharks), and which contribute to a high average transect biomass in a global context. In a recent global analysis of MPA effectiveness using RLS data from 87 MPAs around the world, Kermadec Islands surveys ranked in the top four in terms of the total fish biomass and shark biomass observed (Edgar et al. in prep). Moreover, the Kermadecs Marine Reserve was recognised in that study as one of the most effective MPAs globally for biodiversity conservation. It was one of only five MPAs investigated worldwide with five important conservation features: no-take, well enforced, old, large, and isolated.

Kermadec Islands reef communities differed from those on the New Zealand North Island for all four ecosystem components investigated (large fishes, cryptic fishes, benthic invertebrates, sessile flora and fauna), with much greater similarity to Norfolk Island communities. The lack of tropical elements in the Kermadec Islands biota despite its subtropical location was particularly notable. Few corals were recorded (no acroporids, pocilloporids or faviids) and virtually no coral reef associated species. For example, butterflyfishes in the genus *Chaetodon* are well represented at Lord Howe Island (21 species), Norfolk Island (14 species), and Minerva Reefs (11 species), but none were sighted in our Kermadec surveys and only one species (*Chaetodon auriga*) has been recorded from the area (Francis 1993). A prior study indicates that coral can occur at up to 15% cover at some sites off Raoul Island (Gardner et al. 2006); however, such sites are exceptional and were not encountered during our surveys.

The almost complete lack of overlap between the fish faunas of the Kermadec Islands and Minerva Reefs, only 700 km distant, is remarkable given the propensity for coral reef fishes to disperse long distances as larvae. Clearly, a major oceanographic barrier exists between these regions, with inference that no southward flowing currents could occur.

Biotic relationships between the Three King Islands and Northland coasts were complex, with a distinctive cryptic fish fauna and algal flora at the Three Kings, but with a high level of overlap in benthic invertebrates and large fishes (the endemic bluefinned butterfish *Odax cyanoallix* comprising an exception) (Figs 6-9).

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Further evidence in RLS surveys of the uniqueness of the New Zealand fish fauna is that 16 of the 26 endemic triplefin species known from New Zealand waters (Hickey et al. 2009) were recorded on transects. While triplefins (Tripterygiidae) are speciose and widely distributed, occurring in a range of habitats in tropical, temperate and sub-polar oceans, New Zealand is the global hotspot for triplefin diversity (Hickey et al. 2009). More than twice the number of triplefin species occur in the North-Eastern New Zealand ecoregion than in any of the other 74 ecoregions covered in the RLS global dataset, including more species than recorded around the entire Australian continent (on >3,000 transects).

The differences in rocky reef community structure among the Kermadec Islands, Three Kings Islands and northeastern New Zealand are consistent with the biogeographic classification utilised for marine protected area planning in New Zealand (Ministry of Fisheries and Department of Conservation 2008). These surveys have confirmed that the Kermadec Islands and Three Kings Islands differ significantly from mainland New Zealand in terms of their marine biodiversity. While the entire territorial sea within the Kermadec Islands Biogeographic Region is encompassed within a marine reserve, there are currently no marine protected areas within the Three Kings Biogeographic Region (Department of Conservation and Ministry of Fisheries 2011).

Threatened species

Two conclusions can be reached from our assessment of threatened species recorded on New Zealand RLS surveys. The first is that very little is known about the level of threat for the majority of New Zealand's sub-tidal reef fauna. 79% and 96% of the fishes and invertebrates recorded, respectively, have not been assessed under either national or global criteria. The second conclusion is that of those assessed, most are known from only a restricted range, and this is particularly the case at the Kermadec islands.

The Galapagos shark (*Carcharhinus galapagensis*) is of particular note, listed in the NZTCS on the basis of its restricted NZ range, despite being widespread globally. Its listing in the NZTCS reflects its range in New Zealand waters being limited to the Kermadecs, while globally it is known from numerous, distinct populations in the Pacific, Atlantic and Indian Oceans. Despite this, it is listed by the IUCN as 'Near Threatened' because many of these isolated populations are subjected to high levels of fishing pressure and the potential for local declines is high. The Kermadecs clearly represent a global stronghold for the species. It was recorded on 62% of transects at an average of 6 individuals and 99 kg per 500 m^2 across the four main Islands. Further sightings of numerous individuals were made in the vicinity of the boat when near inshore reefs during the survey expedition. An aggregation of considerably more than 50 individuals was observed off transect at L'Esperance Rock, with 45 individuals present in one video frame grab.

Marine reserve effects

In line with conceptual model predictions, significant protection effects were evident across the North Island marine reserve network that not only reflected direct interactions between fishers and targeted species (increased large fish biomass and lobster abundance) but also second order (decreased urchin abundance), third order (increased *Ecklonia* cover) and fourth order (decreased 'other foliose' and turf algal cover) interactions. While consistent with the conceptual model, the strength and regularity of observed differences between fished and unfished areas was surprising in that previous studies had indicated locally-idiosyncratic responses in different reserves (Langlois and Ballantine 2005). Given that coincidental responses at six marine reserves distributed across a range of oceanographic conditions seems extremely unlikely, strong underlying ecological drivers associated with fishing were presumably operating.

Four outcomes of our study were unexpected: (i) the consistency (~20%) of the increase in *Ecklonia* cover across North Island reserves relative to nearby fished sites regardless of lobster and urchin density, (ii) an inconsistent response of crustose coralline algae to urchin density, (iii) low cover of other foliose algae and turf algae in marine reserves where levels of urchin herbivory were low, and (iv) decoupling between responses of algal habitat to protection and responses of fishes and invertebrates to protection, as indicated by analysis of community heterogeneity across different sites. These results indicate a need for some refinement in the reef conceptual model.

Fishes as well as rock lobsters probably contribute substantially to control of herbivores in marine reserves, given that relatively few rock lobsters were observed in the Poor Knights Islands, Tuhua and Te Paepae o Aotea reserves where high cover of *Ecklonia* was present. Large fishes were more than twice as abundant in all three of these reserves than random forest predictions; consequently, fish species such as snapper, which partly feed on juvenile urchins when above 450 mm length (Thompson 1991) and are an order of magnitude more abundant inside marine reserves, possibly contribute to control of urchins at reserve locations with few rock lobsters. Alternatively, significantly greater biomass of *Ecklonia* within the Poor Knights Islands Marine Reserve compared to fished reference sites in the Mokohinau group, in particular, may relate to the unique combination of local factors at the Poor Knights confounding our analysis. In this case, greater abundance of *Ecklonia* in the reserve area may not be due to a protection affect but rather natural spatial variation in community structure.

A habitat-engineering role of urchins through grazing of macroalgae was supported by consistently low numbers of urchins at reserve sites with elevated cover of *Ecklonia*. At Te Paepae o Aotea, the location where urchin density was lowest outside as well as inside the marine reserve, *Ecklonia* cover was anomalously high at fished as well as protected sites. However, urchin density was high within the Tuhua marine reserve but *Ecklonia* density was also high.

Urchin density was not, however, closely related to the extent of barrens. Crustose coralline algae, the main photoquadrat indicator of barrens formation, was generally consistent across the different North Island marine reserves, with no clear pattern inside versus outside reserves (Fig. 14). Establishment of crustose coralline algal barrens thus appears to be a more complex phenomenon than simply reflecting high urchin density, otherwise barrens would be largely absent from marine reserves with low urchin densities. As is the case with the extensive barrens in the Kermadec Islands (Cole et al. 1992), factors additional to urchins appear to limit cover of macroalgae in some situations, perhaps including excessive wave action.

Moreover, depressed densities of urchins inside reserves was associated with low densities of 'other foliose algae' and turf algae. This result is contrary to the classical 'lobster eats urchins which graze macroalgae model', in that decreased urchin grazing within reserves is expected to lead to increased densities of all macro-algal types, not just *Ecklonia*. Fourth order trophic interactions involving competition between algal species presumably occurred within reserves, such that when grazing pressure was low then *Ecklonia* ultimately outcompeted compact algae by developing a canopy that blocked light passing to smaller algae below. Experimental manipulative studies indicate that *Ecklonia* can inhibit development of turf algae and mid-height foliose algae in this way (Kennelly and Underwood 1993, Melville and Connell 2001).

The deviations from conceptual model predictions noted above at individual marine reserves may be a consequence of low transect replication at those particular reserves, and consequent high error in estimates of effect sizes. If, for example, more sites had been surveyed in Tuhua Marine Reserve then a lower estimate of mean urchin density may have resulted, an outcome more in line with

predictions. Nevertheless, the consistent elevation of *Ecklonia* cover within all reserves regardless of lobster and urchin density indicates that other important pathways linking exploited fishes to macroalgae are probably operating.

Herbivorous fishes such as butterfish (*Odax pullus*) that fall prey to large predatory fishes potentially provide an important additional functional node in food webs. Amphipods and other small grazing invertebrates ('mesograzers') probably play an even greater functional role, as indicated by their high overall productivity and algal consumption rates (Taylor 1998, Poore et al. 2012). For example, amphipods have been implicated as an agent facilitating kelp dieback (Cole and Syms 1999), and the amphipod *Orchomenella aahu* has been observed to destroy beds of *Ecklonia* stressed by bleaching or storm events (Haggitt and Babcock 2003). Investigation of control of mesograzer outbreaks by snapper and other large invertebrate-feeding fishes should prove a fruitful avenue for further research.

Perhaps the most intriguing outcome of the RLS surveys was the observation that the set of six marine reserves tended to possess a more consistent algal habitat structure than the associated set of fished sites, whereas for Method 1 fishes, cryptic fishes and benthic invertebrates the set of fished sites showed greater homogeneity in community structure than the set of marine reserve sites. Thus, as protected habitats within reserves became more homogeneous by converging on an *Ecklonia*-dominated pattern, fishes and invertebrates did not seem to track this habitat shift but instead became more dissimilar from each other in the different reserves, regardless that the same sites were sampled for flora and fauna, and that fishes and invertebrates are thought to respond closely to variability in algal habitat (Jones 1992, Kennelly and Underwood 1993, Anderson and Millar 2004).

The increased spatial heterogeneity in fish and benthic invertebrate communities within marine reserves compared to fished sites has important conservation implications that should be investigated further. If confirmed at broader scales, then it follows that the small proportion of the northern New Zealand coastal zone (~2%) that is located within marine reserves plays a disproportionately large role in the national conservation of marine biodiversity, in that the reserve network potentially includes a greater range of fish and invertebrate communities than is present along the 98% of the coast that is fished.

Suitability of the RLS model for monitoring inshore habitats of New Zealand

A variety of different methodologies have been used for monitoring ecological change in inshore environments, including grabs, dredges, nets, poisons, cores, photoquadrats, unbaited underwater video, baited remote underwater video (BRUV), remotely-operated vessel (ROV), autonomous underwater vessel (AUV), and underwater visual census techniques involving divers. All of these methods can be further subdivided into methodological subsets that differ between investigators, with variation in such factors as area sampled, size of survey footprint, time deployed at sea, distance covered, and bait type. The various methodologies have differing strengths and weaknesses, the relative importance of which depends on the particular study aims.

With respect to ecological monitoring of marine reserves, important considerations include: (i) relevance to management objectives, (ii) non-destructive, (iii) cost effective, and, ideally, (iv) inclusive of a wide range of monitoring metrics. Fishes and exploited invertebrates (particularly rock lobsters and paua) tend to be the most important targets of marine reserve monitoring programs, given that these species are most directly affected by management actions (i.e. a prohibition on fishing). An important consideration for studies of fishes in marine reserves is that the methodology

should be non-destructive where possible, consequently methods such as netting, dredging, and poisoning may not be suitable.

No single survey method can cover all fish species. As suggested by Taylor et al. (2005), a variety of techniques need to be employed to optimise monitoring of different groups of fishes in marine reserves. Methods widely used for marine reserve monitoring can be categorised as: (i) underwater visual transect census by diver (UVT), (ii) roaming video, as operated by diver, AUV or ROV, and (iii) stationary BRUVs. The major negative aspects of UVT methods are high logistic costs associated with deploying scientific dive teams, dive safety risk, biases associated with movement of fishes towards or away from divers, survey time restrictions associated with decompression, and depth limitation to a maximum of 30 m, unless highly technical training and equipment are utilised. UVT (and all other common methods) is also limited to locations with reasonable underwater visibility (>1 m, ideally >5 m). The major benefits of UVT relate to its cost effectiveness, in that a very wide range of taxa can be quantitatively surveyed in a short field period, and little post-processing of data is needed.

Roaming video methods are limited to a relatively small field of view that generally precludes survey of small and crevice-dwelling fishes, cost and reliability issues associated with the deployment of imaging equipment at sea, and considerable post-processing time. BRUVs have the same set of limitations as roaming video plus a bias associated with attraction of a limited subset of fishes to baits. However, this bias can prove a benefit in marine reserve situations when the major monitoring target is the set of fishes targeted by anglers.

An additional issue with video methods is that relative abundance estimates of fishes quantified from images do not scale linearly with absolute fish density, when calculated as the maximum number of individuals of a species present at any time in a time series of images (Nmax). For example, comparisons of two reserves where snapper densities are 5 and 500 individuals per 1000 m^2 are likely to show a much smaller relative difference of say 1 to 8 using Nmax, because the image becomes saturated with fishes at high density. Video methods are superior to UVT methods when paired stereo video units are utilised and an important monitoring output is accurate size estimates of fishes (Harvey et al. 2002). Stereo units are, however, prone to mechanical failure and involve considerable post-processing time and labour cost.

In summary, BRUVs tend to be most useful in surveys at depths > 25 m or when the target group of fishes comprises species caught by fishers, roaming video methods are most useful in hazardous diving situations and depths > 25 m where a wide range of fish species are of interest, and UVT methods are most useful in broad scale biodiversity surveys conducted in shallow habitats. Although UVT methods possess a number of recognised biases (Edgar et al. 2004, Taylor et al. 2005), these methods are now most heavily relied upon by marine ecologists to characterise fish and invertebrate community structure in marine reserves. With respect to different UVT methods, we advocate use of the RLS protocol, which includes coverage of all the major macroscopic ecosystem components associated with reef habitats (including cryptic fishes, invertebrates and floral elements).

Surveys at marine reserves should ideally cover a wide range of ecosystem-level components rather than be focused solely on exploited species. A lack of coverage of such groups as cryptic fishes, benthic invertebrates and macro-algae means that flow-on food web effects associated with recovery of fished species will not be documented. Thus, trophic cascades (sensu Shears and Babcock 2003, Mumby et al. 2007) will be overlooked, as will long-term changes to ecosystems associated with threats unrelated to fishing such as climate change, including the opportunity to understand interactions between climate change and fishing using the marine reserve network as a

manipulative experiment. Surveys of large fishes alone also preclude coverage of non-target species that may either be ecologically important, such as sea urchins, or represent important biodiversity heritage in their own right, such as the triplefins. For example, only a single species of triplefin was recorded in one of the more detailed reserve assessments of the Poor Knights Islands using both visual census and video methods (Denny et al. 2003). By contrast, RLS surveys of the Poor Knights Islands recorded 11 species from this family of cryptic fishes, all New Zealand endemics.

The RLS methodology has been refined over the past two decades to produce a cost-effective quantitative overview of marine biodiversity. RLS methods are now widely used for monitoring marine reserves in Australia, New Zealand and internationally (Edgar and Barrett 1997, Barrett et al. 2002, Barrett et al. 2007, Barrett et al. 2009, Edgar et al. 2011). Consequently, regardless of strengths and weakness of particular methodologies, application of RLS techniques uniquely allows survey results to be interpreted in a global context. A standardised approach can therefore be maintained across large scales, enabling direct comparisons between individual reserves, ecoregions and countries, such as the analysis of ecological relationships between the Kermadec Islands and other Pacific biotas, as outlined in this report.

Overall, we consider that application of the Reef Life Survey model represents the most cost-effective means to maintain large spatial and taxonomic coverage for marine reserve biodiversity monitoring. While a small number of species (e.g. those that avoid divers) will not be as effectively covered as with remote video methods, this negative is outweighed by the inclusion of numerous other taxonomic groups not otherwise covered (e.g. cryptic fishes) and the potential to cover a greater number of reserves due to low cost. Lower costs also mean that additional complementary methods, such as those suggested by Taylor et al. (2005), may still be supported when needed.

A key component of the RLS model is the utilisation of skills, enthusiasm and expertise of volunteer divers. All divers participating in the New Zealand surveys were volunteers, as were the boat crew (generally the same set of people). As long as scientific rigor and oversight is provided to the data collection activities of skilled volunteers, who are happy to donate their time to help protect the marine environment, high quality data can be obtained across geographic and temporal scales impossible for scientific teams to cover because of prohibitive cost. The quality of data provided by RLS volunteers has been assessed and found indistinguishable from data produced by professional marine ecology researchers (Edgar and Stuart-Smith 2009). In addition, involvement of the community in marine reserve monitoring in New Zealand would be consistent with the Department of Conservation's strategic direction in terms of engaging with others to help protect biodiversity and allowing the community to be involved in addressing conservation challenges (Department of Conservation 2013).

A cost-effectiveness analysis of the RLS process was undertaken for the Australian Government during the pilot project from 2007-2009. This evaluation included costs associated with organised survey weekends (where costs of accommodation, diving (including boat charter), and travel for a scientific organiser were covered by RLS) compared with costs of similar surveys undertaken by scientific teams. The cost per transect surveyed was less than one quarter for RLS compared with scientific teams (ca. AUD$150 cf. AUD$660).

Since this initial evaluation, RLS activity has shown a 'snowball effect', with considerable additional *ad hoc* data collected by RLS volunteers outside of organised survey weekends. While such additional data are less structured in terms of traditional project design, they contribute valuably to synoptic assessments of the state of the marine environment, alongside the more formally-established marine reserve monitoring data. Thus, cost effectiveness can be considered high for the

structured RLS weekend approach, which is preferable for monitoring marine reserves at remote locations. Cost effectiveness is extremely high for the overall quantity of data collected, which contributes to national state of the environment reporting and broad-scale scientific analysis.

Perhaps the best argument for the cost-effectiveness of RLS surveys is that the total cost of the three month New Zealand survey campaign described in this report, including surveys at the Kermadec Islands, was less than NZ$40,000 (plus donation of use of boat). Costs of a similar research program with scientific divers and chartered boats would likely cost an order of magnitude more.

Development of an RLS program in New Zealand would lead to further outcomes of real benefit in their own right. These include a more active collaboration between marine scientists and managers in Australia and New Zealand, and opportunities for RLS divers on both sides of the Tasman to broaden their skills and experience by participating in activities in the other country (as well as more broadly in international RLS activities).

A key benefit to a New Zealand program is that data quality control and checking procedures and infrastructure are already established. RLS data are managed in a state-of-the-art database at the University of Tasmania, alleviating the need for numerous, isolated sources and formats for raw data. Covariate data are also available on the RLS database for analyses of, for example, species traits or relationships with global environmental conditions (e.g. mean seawater temperature, seastate, annual variability in chlorophyll concentration).

Limitations to application of the RLS approach for the New Zealand marine environment include seasonality in recreational SCUBA activity, particularly in cooler southern regions, and often poor visibility in inshore and near metropolitan regions. The latter limitation applies regardless of which methods are used to survey fishes, but is generally best avoided by having local dive teams which can time survey dives to capitalise on periods of best visibility. The established protocol for recording underwater visibility during surveys also enables this variable to be accounted for in analyses, as was done in the random forest assessment of reserve effects for this report.

Seasonality in volunteer diver participation is a more serious consideration, but is most easily dealt with by scheduling an annual survey period for each marine reserve that encompasses the time most suitable for diving in that region. In Australia, annual survey weekends at each key MPA for which data are needed by management partners are scheduled for times that maximise volunteer participation and a need to sustain surveys at the same time each year over the long term. This often means long weekends during the summer period, particularly for cooler regions. One other aspect of seasonality is the need for volunteer divers to maintain at least semi-frequent survey activity to remain 'fresh' with identification skills. This mostly applies to less experienced surveyors, so is most critical during the early years of establishment, but should be kept in mind at all times. Arranging and encouraging involvement of divers from cooler regions in surveys of warmer regions during the winter helps to reduce this by maintaining activity year round. Similarly, encouraging cross-fertilisation with Australian activities may not greatly assist maintenance of ID skills for New Zealand species other than during surveys in Tasmania, but will still assist with maintenance of enthusiasm and building of survey experience.

Primary recommendations for establishing Reef Life Survey capacity in New Zealand include:

- Identification and engagement of 'champion' scientists, or well-trained and capable recreational divers, who can act as RLS organisers in terms of directing activities, and helping with IDs and data entry on organised survey weekends,

- An annual timetable for survey weekends developed in consultation with local divers, considering the needs for reasonable dive conditions (activities need to remain enjoyable to be sustainable), and to keep divers actively surveying throughout the year, with modest funding that is committed through the long-term to support these activities,

- Identification of appropriate priority sites associated with each reserve, which balance the need for appropriate statistical design, but also consider safety and enjoyment of participants,

- Maintaining regular direct feedback from management partners to divers to ensure participants are reminded of the value of their work for management,

- Close collaboration with the Australian RLS program, as that program can assist with training support and data management in early stages of New Zealand RLS development. The Australian RLS program can also provide advice on lessons learned and feedback in terms of the value of their work for broader understanding of issues relating to biodiversity conservation. Further benefits include a more active collaboration between marine scientists and managers in Australia and New Zealand, increased scientific knowledge of species and ecosystems that span the Tasman, increased power of scientific analyses by combining data on, for example, marine protected areas from both regions, and opportunities for RLS divers on both sides of the Tasman to broaden their skills and experience by participating in activities in the other country (as well as more broadly in international RLS activities).

Acknowledgments

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Appendix 1 Marine reserve sites surveyed by Reef Life Survey divers 2012-2013. GPS coordinates in WGS84, Vis = estimated underwater visibility (average across transects at site). *Surveyed previous to 2012.

RLS Code	Site Name	Latitude	Longitude	Vis	Zone	Depths
Kermadec Islands						
KER1	L'Esperance Rock N	-31.35397	-178.82567	40	No Take	9, 11
KER2	L'Esperance Rock S	-31.35606	-178.82681	35	No Take	15
KER3	Curtis Is	-30.54484	-178.55803	35	No Take	11, 19
KER4	Cheeseman Is	-30.53823	-178.569	30	No Take	9, 16
KER5	Macaulay South	-30.24063	-178.43498	25	No Take	8, 10
KER6	Macauley Island, SW	-30.23757	-178.43808	30	No Take	7, 14
KER7	Meyer Is	-29.248138	-177.87839	25	No Take	7, 14
KER8	Nash Pt	-29.283432	-177.89233	40	No Take	9, 15
KER9	Te Konui Pt	-29.28075	-177.89578	30	No Take	6, 12, 13
KER10	Egeria Rock	-29.24917	-177.89496	22	No Take	9, 12
KER11	Milne Rocks	-29.281771	-177.90259	35	No Take	8, 12
KER12	Milne Rocks W	-29.280891	-177.90442	30	No Take	9, 12
KER13	Dougall Rock	-29.296268	-177.91518	40	No Take	8, 14
KER14	Smith Bluff	-29.29233	-177.96092	30	No Take	7, 7.5, 10
Poor Knights Island Marine Reserve						
NZ1	Poor Knights Middle Arch*	-35.45753	174.73282	8	No Take	15
NZ2	NZ Poor Knights South Harbour	-35.48828	174.73844	20	No Take	7, 12
NZ3	NZ Poor Knights South Harbour	-35.48813	174.74038	8	No Take	7, 12, 13
NZ4	The Gardens	-35.4794	174.7376	20	No Take	6, 8, 9
NZ5	Crystal Cave*	-35.484883	174.73365	8	No Take	10
NZ6	Dutchman s Cove*	-35.47262	174.734	8	No Take	11
NZ7	Riko Riko Mouth West wall	-35.481717	174.735483	8	No Take	7, 10
NZ8	Northern Arch South Bay*	-35.44901	174.73225	8	No Take	10
NZ9	North of Trev s Rock*	-35.47748	174.73709	8	No Take	9
NZ10	The Canyons*	-35.46929	174.73308	8	No Take	8
NZ11	Landing Bay Pinnacle*	-35.46574	174.73483	8	No Take	14
PAC75	Poor Knights NE bay	-35.44121	174.73471	12	No Take	8, 22
PAC98	Labrid Passage North	-35.48555	174.74181	18	No Take	7, 10
PAC99	Aorangaia Is SW Corner PKI	-35.48771	174.74544	20	No Take	10
PAC114	Bay north of Northern Arch	-35.44755	174.73459	30	No Take	7, 16
PAC115	Northern Arch	-35.47567	174.73784	30	No Take	9
PAC118	Serpent Rock	-35.47318	174.73123	20	No Take	21
PAC119	Rikoriko Point	-35.48164	174.73383	18	No Take	9

RLS code	Site Name	Latitude	Longitude	Vis	Zone		Depths
Cape Rodney – Okakari Point Marine Reserve							
NZ12	Leigh*	-36.26817	174.79579	4	No Take		2, 3
PAC94	Goat Island West	-36.26514	174.79451	7	No Take		7, 8
PAC95	Goat Island East	-36.26608	174.79994	7	No Take		6, 13
PAC96	Goat Island Table Top	-36.26928	174.80469	6	No Take		8, 9
PAC123	Cape Rodney south	-36.28157	174.82162	4	Reference		6, 12
PAC124	West of Kemp's Bay	-36.25787	174.7504	4	Reference		4, 6
PAC125	One spot reef	-36.27372	174.8172	6	No Take		7.5, 14, 15
PAC126	Martin's Rock	-36.26729	174.78468	4	No Take		7.5, 11, 12
PAC127	Alphabet Bay	-36.26722	174.79429	4	No Take		3.5, 4, 5
PAC169	Panetiki Point	-36.29041	174.81665	6	Reference		5, 6
Tawharanui Marine Reserve							
PAC110	Comet Rocks	-36.36057	174.82628	5	No Take		7
PAC111	East Anchor Bay	-36.36647	174.84309	4	No Take		5, 6
PAC112	North East Bay	-36.36258	174.85864	4	No Take		7, 11
PAC171	Tawharanui East	-36.36535	174.8567	4	No Take		5, 7
PAC172	Tawharanui outside W	-36.34748	174.80226	2	Reference		8
PAC170	Ti Point	-36.3217	174.7994	5	Reference		4, 6
Whanganui A Hei Marine Reserve							
PAC159	Tower Rock, Whanganui A Hei	-36.81414	175.78822	6	No Take		11, 14, 18
PAC160	Hahei I	-36.82129	175.7822	8	No Take		5, 6, 9
PAC161	Mahurangi Islet	-36.82609	175.81898	9	No Take		6, 9
PAC162	Mahurangi I outside reserve	-36.82862	175.81818	8	Reference		3, 6
PAC163	Kingfish Rock	-36.82312	175.81224	9	No Take		10, 13
PAC164	South Sunk Rock	-36.81777	175.81898	8	Reference		8, 10, 14
Te Matuku Marine Reserve							
PAC157	Passage Rock	-36.85658	175.12931	2	No Take		4, 5
PAC158	Te Matuku East Pt	-36.84864	175.14354	1	No Take		2
Tuhua/Mayor Island marine reserve							
PAC139	Crater Bay	-37.28588	176.27328	16	Reference		3, 8
PAC140	Crater Bay North Point	-37.28242	176.27655	19	Reference		6.2, 14
PAC141	Archway Island	-37.27693	176.27815	25	Reference		5, 7, 9, 12, 15
PAC142	Turanganui Bay	-37.27255	176.27847	12	No Take		5, 12
PAC165	Two Fathom Reef	-37.25845	176.26906	11	No Take		9, 12, 14
PAC166	Mayor I, North Rock	-37.26556	176.27007	11	No Take		10, 14
PAC167	Motuneke, Mayor I	-37.26878	176.24698	12	No Take		6, 12
PAC168	Opoupoto Bay, Mayor I	-37.27266	176.2412	9	No Take		9, 12
PAC120	Pokohinu North Face	-35.8997	175.1174	20	Reference		21
PAC121	The Canyon	-35.9032	175.1134	15	Reference		10
PAC122	Lighthouse Reef	-35.909	175.1174	16	Reference		5, 6, 7, 10
Te Paepae o Aotea (Volkner Rocks) Marine Reserve							
PAC146	Big Volkner Rock	-37.47694	177.13533	13	No Take		6, 12, 18
PAC147	Diadema Rock	-37.47833	177.12904	12	No Take		9, 18
PAC148	Small Volkner Rock	-37.48079	177.12724	18	No Take		6, 9, 12
PAC149	Black Rock	-37.5193	177.1672	16	Reference		4, 9, 18
PAC150	Homestead Reef	-37.5259	177.175	15	Reference		5
PAC151	Club Rocks	-37.5372	177.1827	14	Reference		6, 12, 16

Appendix 2 Additional fished sites surveyed by Reef Life Survey 2012

RLS code	Site Name	Latitude	Longitude	Vis	Depths
PAC70	Moturahurahu Island NE	-35.2216	174.271	9	6, 14
PAC71	Moturahurahu Island SW	-35.2223	174.2686	8	2.5, 3
PAC72	Moturahurahu Island N islet	-35.2186	174.1607	9	7, 14
PAC73	Whangamumu heads south	-35.2495	174.3203	10	7, 13
PAC74	Whangamumu heads north	-35.2401	174.3108	6	4, 6
PAC77	Waiwiri I	-35.1938	174.3427	9	9, 16
PAC78	Motukokoko I	-35.165	174.3413	11	8, 14
PAC79	Motuhuia I	-35.038	173.9639	10	8, 15
PAC80	Motukawaiti I	-35.0313	173.9645	12	8, 11
PAC81	Motuiwi I	-35.055	173.9394	11	7, 12
PAC82	Waiana Bay	-35.0599	173.9333	11	5, 8
PAC83	Rainbow Warrior wreck	-34.975	173.9351	14	19
PAC84	Horonui Island	-34.9706	173.9341	10	7, 10
PAC85	HMNZS Canterbury wreck	-35.1939	174.2939	7	27
PAC86	Deepwater Cove North	-35.192	174.2947	7	8, 9
PAC87	Deep Water Cove NW Islet	-35.1917	174.293	7	6, 10
PAC88	Three Kings I North Bay	-34.1516	172.1377	25	11, 14
PAC89	Three Kings I SE Bay east	-34.1528	172.1521	25	11, 15
PAC90	S E Bay	-34.1539	172.1448	20	6, 10
PAC91	SE Bay 2	-34.1534	172.1453	20	9, 11
PAC92	Three Kings I south bay	-34.1631	172.1443	25	10, 12
PAC93	Three Kings Island ESE	-34.1621	172.1459	20	10, 10.1, 11
PAC100	Pitokuku Island	-36.2554	175.4917	12	5, 6
PAC101	Pitokoku headland	-36.2554	175.4918	10	8
PAC102	Waikaro Point	-36.1104	175.4403	8	8, 8.1
PAC103	Waikaro Point2	-36.1084	175.4379	8	8, 8.1
PAC104	East of Waikaro Pt	-36.1058	175.428	10	11
PAC105	East of Waikaro Pt 2	-36.1056	175.4257	10	7, 9
PAC106	Rangiwhakaea Bay south	-36.0887	175.4153	11	5, 5.1, 10
PAC107	Taranga Pinnacles	-35.9544	174.7028	14	7, 12
PAC108	Lady Alice Island	-35.8853	174.7252	15	8, 10
PAC109	Taronga 8 m Pinnacle	-35.9628	174.6989	12	7, 8
PAC113	Bostaquet Bay west headland	-36.4454	174.857	5	7, 8
PAC128	Mathesen Island	-36.3057	174.8001	3	6, 9.5, 10
PAC131	Cuvier west point	-36.4347	175.7599	15	8, 9.9, 10, 11
PAC132	Cuvier North point	-36.4262	175.7672	18	8.5, 11, 14
PAC133	Kawhiti Island (east bay)	-36.6396	175.889	18	5, 11
PAC134	Kawhiti Rock Pinnacles	-36.6336	175.8936	20	10, 15, 17
PAC135	Kawhiti SE cove	-36.6422	175.8916	12	9, 12
PAC136	Von Luckner's Cove	-36.6203	175.9412	12	9.5, 10
PAC137	Red Mercury SE point	-36.6306	175.948	16	5, 8, 10, 12
PAC138	Lunch Bay	-36.6305	175.9336	18	5, 7, 9, 12
PAC143	Tuhua Reef	-37.3053	176.288	20	9
PAC144	Tuhua Reef East	-37.306	176.2907	23	15, 20

PAC145	Ruakikeno Point	-37.3048	176.268	18	9, 12
PAC152	Anchor Bay	-36.9775	177.0782	15	10, 12
PAC153	Anchor Bay East Rocks	-36.9786	176.0812	15	6, 10
PAC154	Nudibranch Wall	-36.9577	176.058	9	5, 9, 15
PAC155	Ruamahuaiti Island NE	-36.9708	176.0841	11	6, 9, 12, 17
PAC156	Ugly Point	-36.9506	176.0787	11	9, 12, 15

Appendix 3 Substrate categories used for RLS benthic photo-quadrat processing and observed in New Zealand.

Category code	Category name
Aherm	Ahermatypic corals
Anem	Anemones & Zoanthids
Asc	Ascidians
Bare	Bare Rock
Barn	Barnacles
Bfol	Small to medium foliose brown algae
Bry	Bryozoan
Caul	*Caulerpa*
CCA	Crustose coralline algae
Dead	Dead Coral
Eck	Laminarian kelp, generally *Ecklonia radiata*
Ecor	Encrusting corals
ELA	Encrusting leathery algae
Fil	Filamentous epiphytic algae
FRA	Filamentous rock-attached algae
Fuc	Large brown fucoid kelps
GCA	Geniculate coralline algae
Gcal	Green calcified algae
Gfol	Other foliose green algae
Hyd	Hydroids
Peb	Pebbles/unconsolidated rocky bottom/coral rubble
Pol	Polychaete
Rfol	Foliose red algae
Sand	Sand
Slime	Diatom/algal/cyanobacterial fuzz/slime on bare rock
Soft	Soft corals and gorgonians
SpE	Sponges (encrusting)
Spo	Sponges (erect)
Stony	Large-polyp stony corals (e.g. fungiids)
Turf	Turfing algae (<2 cm high algal/sediment mat on rock)

Appendix 4 Species recorded during fish surveys in New Zealand by Reef Life Survey.

Family	Species	Ecoregions	F %	N	B (kg)
Aplodactylidae	*Aplodactylus arctidens*	2	25.6	2.5	1.3
Aplodactylidae	*Aplodactylus etheridgii*	2	10.4	4.6	1.0
Arripidae	*Arripis trutta*	1	0.9	24.0	6.9
Arripidae	*Arripis xylabion*	1	13.8	19.0	15.9
Aulostomidae	*Aulostomus chinensis*	1	27.6	1.5	0.7
Berycidae	*Centroberyx affinis*	3	1.5	1.0	0.7
Blenniidae	*Cirripectes alboapicalis*	1	55.2	4.0	0.0
Blenniidae	*Parablennius laticlavius*	3	14.8	5.4	0.0
Blenniidae	*Plagiotremus tapeinosoma*	2	6.4	5.4	0.0
Carangidae	*Decapterus koheru*	2	19.7	282.3	23.9
Carangidae	*Pseudocaranx georgianus*	3	19.8	38.3	4.7
Carangidae	*Seriola lalandi*	3	6.5	2.4	15.2
Carangidae	*Seriola rivoliana*	1	6.9	1.0	0.3
Carangidae	*Trachurus novaezelandiae*	1	10.0	178.4	42.9
Carcharhinidae	*Carcharhinus brachyurus*	1	0.5	1.0	20.7
Carcharhinidae	*Carcharhinus galapagensis*	1	62.1	6.0	99.0
Chaetodontidae	*Amphichaetodon howensis*	2	3.6	2.2	0.3
Cheilodactylidae	*Cheilodactylus ephippium*	2	9.6	4.4	1.6
Cheilodactylidae	*Cheilodactylus francisi*	1	3.4	1.0	0.3
Cheilodactylidae	*Cheilodactylus spectabilis*	2	70.5	3.7	2.9
Cheilodactylidae	*Nemadactylus douglasii*	2	19.7	1.8	1.3
Cheilodactylidae	*Nemadactylus macropterus*	1	2.3	8.2	1.3
Chironemidae	*Chironemus marmoratus*	1	11.8	2.0	0.4
Chironemidae	*Chironemus microlepis*	1	6.9	7.5	0.2
Cirrhitidae	*Notocirrhitus splendens*	1	51.7	1.8	0.2
Clinidae	*Cristiceps aurantiacus*	1	0.9	1.0	0.0
Dasyatidae	*Dasyatis brevicaudata*	2	2.1	1.0	16.6
Diodontidae	*Tragulichthys jaculiferus*	1	0.9	1.0	4.9
Gobiesocidae	*Dellichthys morelandi*	1	0.5	10.0	0.0
Kyphosidae	*Girella cyanea*	2	9.6	88.7	38.1
Kyphosidae	*Girella fimbriata*	1	20.7	9.3	3.4
Kyphosidae	*Girella tricuspidata*	1	9.5	24.1	14.9
Kyphosidae	*Kyphosus pacificus*	1	89.7	65.3	46.4
Kyphosidae	*Kyphosus* sp. [Kermadecs]	1	3.4	2.0	3.1
Kyphosidae	*Kyphosus sydneyanus*	1	8.6	4.8	8.3
Kyphosidae	*Labracoglossa nitida*	1	58.6	65.2	3.2
Labracoglossidae	*Bathystethus cultratus*	1	3.4	80.0	3.2
Labridae	*Anampses caeruleopunctatus*	1	48.3	3.7	0.2
Labridae	*Anampses elegans*	1	44.8	3.2	0.2
Labridae	*Bodianus unimaculatus*	1	37.3	2.7	1.3
Labridae	*Coris picta*	1	0.5	1.0	0.0
Labridae	*Coris sandeyeri*	2	49.4	6.8	1.4
Labridae	*Notolabrus celidotus*	1	57.3	5.0	0.3
Labridae	*Notolabrus fucicola*	2	62.4	4.0	1.1

Family	Species	Ecoregions	F %	N	B (kg)
Labridae	*Notolabrus inscriptus*	2	16.1	2.5	0.7
Labridae	*Pseudolabrus luculentus*	2	24.9	57.1	0.6
Labridae	*Pseudolabrus miles*	2	24.8	3.0	0.4
Labridae	*Suezichthys arquatus*	1	41.4	3.3	0.0
Labridae	*Suezichthys aylingi*	2	11.1	5.9	0.1
Labridae	*Thalassoma lutescens*	1	10.3	3.7	0.1
Labridae	*Thalassoma purpureum*	1	6.9	1.0	0.3
Latridae	*Latridopsis ciliaris*	1	7.7	3.4	10.0
Latridae	*Latridopsis forsteri*	1	0.9	1.0	0.5
Monacanthidae	*Meuschenia scaber*	2	80.8	5.0	1.9
Monacanthidae	*Thamnaconus analis*	1	34.5	1.8	0.1
Moridae	*Lotella rhacina*	2	2.6	1.3	0.2
Mullidae	*Mulloidichthys vanicolensis*	1	10.3	2.3	0.4
Mullidae	*Parupeneus spilurus*	1	34.5	3.0	0.8
Mullidae	*Upeneichthys lineatus*	2	30.5	3.7	0.5
Muraenidae	*Enchelycore ramosa*	1	3.4	1.0	2.2
Muraenidae	*Gymnothorax nubilus*	1	1.4	1.7	1.1
Muraenidae	*Gymnothorax obesus*	2	1.2	1.0	0.1
Muraenidae	*Gymnothorax prasinus*	1	6.8	1.1	0.7
Muraenidae	*Gymnothorax* spp.	1	3.4	1.0	0.2
Muraenidae	*Gymnothorax thrysoideus*	1	3.4	1.0	0.0
Myliobatidae	*Myliobatis tenuicaudatus*	2	3.0	1.4	5.2
Odacidae	*Odax cyanoallix*	1	28.6	1.8	0.0
Odacidae	*Odax pullus*	1	29.5	2.3	0.4
Pempherididae	*Pempheris adspersa*	2	24.8	33.6	0.6
Pempherididae	*Pempheris affinis*	1	0.5	6.0	0.1
Pempherididae	*Pempheris analis*	1	27.6	43.0	1.4
Pentacerotidae	*Evistias acutirostris*	1	3.4	17.0	2.7
Pentacerotidae	*Paristiopterus labiosus*	1	0.5	2.0	0.0
Pentacerotidae	*Zanclistius elevatus*	1	0.5	1.0	0.2
Pinguipedidae	*Parapercis colias*	2	5.6	1.5	0.3
Pleuronectidae	*Pleuronectid* spp.	1	0.5	1.0	0.0
Pomacentridae	*Chromis dispilus*	3	78.3	426.8	16.7
Pomacentridae	*Chromis hypsilepis*	1	0.9	3.0	0.5
Pomacentridae	*Chrysiptera rapanui*	1	72.4	93.8	0.3
Pomacentridae	*Parma alboscapularis*	2	48.6	7.0	4.2
Pomacentridae	*Parma kermadecensis*	1	72.4	9.6	3.0
Pomacentridae	*Stegastes fasciolatus*	1	62.1	10.3	0.7
Scorpaenidae	*Pterois volitans*	1	31.0	1.8	0.5
Scorpaenidae	*Scorpaena cardinalis*	2	13.7	1.5	2.1
Scorpaenidae	*Scorpaena papillosa*	1	3.6	1.1	0.0
Scorpididae	*Atypichthys latus*	3	9.5	43.9	2.0
Scorpididae	*Scorpis lineolata*	2	61.1	35.5	1.9
Scorpididae	*Scorpis violacea*	3	40.7	155.5	42.8
Serranidae	*Acanthistius cinctus*	1	58.6	2.3	0.9
Serranidae	*Aulacocephalus temminckii*	1	44.8	2.2	0.2
Serranidae	*Caesioperca lepidoptera*	2	14.1	14.5	3.7

Family	Species	Ecoregions	F %	N	B (kg)
Serranidae	*Caprodon longimanus*	2	5.1	112.5	40.6
Serranidae	*Epinephelus daemelii*	1	24.1	1.3	12.1
Serranidae	*Hypoplectrodes huntii*	2	2.1	2.8	0.1
Serranidae	*Hypoplectrodes* sp. (undescribed)	3	6.8	1.6	0.1
Serranidae	*Trachypoma macracanthus*	1	41.4	1.7	0.1
Sparidae	*Pagrus auratus*	1	21.4	7.6	6.3
Syngnathidae	*Stigmatopora macropterygia*	1	0.5	2.0	0.1
Tetraodontidae	*Canthigaster callisterna*	2	2.4	1.0	0.1
Trachichthyidae	*Optivus agastos*	2	20.9	23.4	0.3
Tripterygiidae	*Cryptichthys jojettae*	1	1.8	3.0	0.0
Tripterygiidae	*Enneapterygius kermadecensis*	1	31.0	8.0	0.0
Tripterygiidae	*Forsterygion flavonigrum*	1	8.2	3.9	0.0
Tripterygiidae	*Forsterygion lapillum*	1	24.5	20.9	0.0
Tripterygiidae	*Forsterygion malcolmi*	1	17.3	2.4	0.0
Tripterygiidae	*Forsterygion maryannae*	2	63.2	527.8	0.6
Tripterygiidae	*Forsterygion varium*	2	33.8	10.0	0.0
Tripterygiidae	*Grahamina capito*	1	1.4	23.0	0.0
Tripterygiidae	*Karalepis stewarti*	1	3.2	1.1	0.0
Tripterygiidae	*Notoclinops caerulepunctus*	1	1.8	1.8	0.0
Tripterygiidae	*Notoclinops segmentatus*	1	32.7	7.0	0.0
Tripterygiidae	*Notoclinops yaldwyni*	1	5.0	2.3	0.0
Tripterygiidae	*Notoclinus fenestratus*	1	0.5	1.0	0.0
Tripterygiidae	*Ruanoho whero*	2	14.1	2.4	0.0
Zeidae	*Zeus faber*	1	1.4	1.0	0.4