Biological monitoring of marine protected areas at Banks Peninsula using baited underwater video (BUV)

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Abstract

Biological monitoring of marine protected areas (MPAs) is important to assess their effects and to determine whether they are meeting their objectives. However, traditional monitoring has proven difficult at Banks Peninsula. In this study, we investigated whether baited underwater video (BUV) is a suitable method for monitoring fish assemblages here. BUV deployments (n = 134) were made in Pōhatu and Akaroa Marine Reserves marine reserves and in three control areas over seven consecutive days in February and March 2017. The relative abundance and size distribution of key species, and species richness were then compared, and species–habitat relationships investigated. We recorded 28 fish species, which included the commonly fished species blue cod (*Parapercis colias*), blue moki (*Latridopsis ciliaris*) and tarakihi (*Nemadactylus macropterus*). The relative abundance of legal-sized blue cod (>300 mm) was 3.6 and 2.1 times greater in Pōhatu and Akaroa Marine Reserves, respectively, than in the control areas, and was positively related to coarse sediment, cobble habitat and depth, and negatively related to the distance from reef structure. The relative abundance of legal-sized blue moki (>400 mm) was 10 and 8 times greater at Pōhatu and Akaroa Marine Reserves, respectively, than in the control areas, and was strongly associated with canopy-forming algae and depth. Our findings indicated that BUV is an effective tool for monitoring fish populations and investigating species-habitat relationships at Banks Peninsula. Regular surveys should be undertaken to assess the effects of these reserves on local fish assemblages.

Keywords: baited underwater video, marine protected areas, Banks Peninsula, blue cod, *Parapercis colias*, blue moki, *Latridopsis ciliaris*, species distribution modelling.

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1. Introduction

Marine protected areas (MPAs) have been established throughout the world's oceans to help protect ecosystems from increasing anthropogenic pressures (Murray et al. 1999; Edgar et al. 2014; Lubchenco et al. 2016). It has been demonstrated that MPAs can facilitate the recovery of previously exploited fish species both in terms of abundance and size class distribution (Lester et al. 2009; Edgar et al. 2014). Further, the recovery of top predators within MPAs has far-reaching ecological consequences, often resulting in the reinstatement of the ecological complexity that underpins healthy ecosystems (Shears & Babcock 2002; Byrnes et al. 2006). However, not all MPAs are effective due to a lack of compliance or poor design (Agardy et al. 2003; Edgar et al. 2014).

Biological monitoring can help to determine the effectiveness of a given MPA while also supporting wider management of the marine environment (Pomeroy et al. 2005; Day 2008). Monitoring not only provides information on how specific biological communities are responding to protection but can also give important insights into the pressures that are faced by surrounding marine ecosystems. This may be particularly valuable for fisheries managers where marine reserves serve as a spatial reference for the state of adjacent fish stocks (Willis 2013). Thus, managers are encouraged to institute a robust and regular monitoring regime when a new MPA is established.

A broad range of scientific sampling methods and study designs have been developed to monitor the effectiveness of MPAs. Commonly used methods for sampling fish populations include underwater visual census (UVC) by divers (Cole 1994; Willis et al. 2000; Edgar et al. 2004), extractive sampling using fishing techniques (Willis et al. 2000; Davidson 2001) and baited underwater video (BUV) (Willis & Babcock 2000; Cappo et al. 2004). Each of these methods has its own advantages and disadvantages (Willis & Babcock 2000), so the optimal method will depend on the objectives of the research, the habitat/environmental conditions at the sample site and the ecological characteristics of the fish communities.

New Zealand has a long history of MPA establishment and monitoring, with much of the first evidence for the effectiveness of MPAs having arisen from studies on New Zealand marine reserves (Cole et al. 1990; Walls 1998). However, nearly all of this early research was undertaken in marine reserves located in the north of the North Island (Cole et al. 1990; Willis et al. 2003), with few published examples being available on marine reserves in the South Island (see Davidson (2001) for an exception). This is partly due to a lack of MPAs in this area – until 2014, Pōhatu Marine Reserve at Banks Peninsula was the only marine reserve on the east coast of the South Island. However, remoteness and harsh conditions (e.g. low water visibility) have also limited routine biological monitoring programmes in those MPAs that have been established (Allum 2008).

Banks Peninsula is located on the east coast of the South Island of New Zealand and has a marine environment that is of high social, economic and ecological value. The peninsula is of volcanic origin and has a large, varied and rocky coastline that is highly productive – which contrasts with Canterbury's otherwise relatively featureless sand and gravel beaches. The mixing of water masses coupled with a shallow continental shelf has led to a diversity in physical habitats that has promoted extensive fisheries of high recreational, commercial and cultural value. However, ongoing fishing pressure has likely contributed to the decline of some fish stocks, particularly those of the once-abundant red cod (*Pseudophycis bachus*) and blue cod (*Parapercis colias*) (Beentjes & Carbines 2005; Källqvist et al. 2005). Therefore, the protection and restoration of such fisheries will require new management initiatives (e.g. MPAs and customary management areas) that are informed by ongoing scientific research and monitoring.

Two marine reserves have been established at Banks Peninsula. Pōhatu Marine Reserve, which was established in 1999 and covers 215 ha of the Flea Bay area and adjacent coastline, is one

of the smallest marine reserves in New Zealand. Akaroa Marine Reserve was established in 2014 and covers 475 ha on the eastern side of the outer Akaroa Harbour and a small section of the open coastline. A baseline survey on the relative abundance, size class distribution and species richness of fishes and invertebrates at Pōhatu Marine Reserve and nearby control areas was carried out by Davidson et al. (2001) using UVC. The survey recorded ten species of reef-associated fishes and did not find any significant differences in the abundance or size of targeted fish species (e.g. blue cod and blue moki (*Latridopsis ciliaris*)) between the reserve and control areas. This monitoring programme was then repeated in 2002 (Davidson & Abel 2003), at which time legal-sized blue cod (> 300 mm) were found to be more common at sites inside the marine reserve than in control areas and the mean size of blue cod was slightly larger at the marine reserve sites. No further monitoring surveys have been carried out at Pōhatu and no baseline surveys for fishes or invertebrates have been undertaken at Akaroa Marine Reserve.

The collection of baseline data and establishment of routine MPA monitoring at Banks Peninsula has proven difficult due to issues around coordinating the presence of dive teams to coincide with suitable water visibility (Allum 2008). Consequently, alternatives to UVC methods have been suggested to make monitoring more feasible (Allum 2008). One of these alternatives – BUV – has a well-documented history for monitoring fish species both internationally (Cappo et al. 2003; Westera et al. 2003) and in New Zealand (Willis & Babcock 2000; Willis et al. 2000; Denny & Babcock 2004). Like UVC, BUV methods can collect concurrent data on habitat types, allowing identification of the habitats that support higher abundances of key fish species (Moore et al. 2011). However, BUV also provides several advantages over traditional UVC methods, including logistical simplicity, less reliance on clear water conditions, no effect of observer bias on fish count and size estimates, and reduced bias associated with the response of fishes to divers (Edgar et al. 2004; Stobart et al. 2007). BUV methods also have their own drawbacks and assumptions though (Willis et al. 2000; Cappo et al. 2004; Stobart et al. 2007), making it important that their appropriateness as a sampling tool for a given MPA is investigated.

BUV has not previously been used for studying coastal fish populations in the South Island. Therefore, the aims of this study were to appraise the suitability of BUV for monitoring the response of fish assemblages to MPAs at Banks Peninsula and to determine whether the data obtained can be used to build species distribution models for certain 'indicator' species (i.e. particular fish that have high recreational or commercial fishing value and are heavily targeted in this area).

The specific aims were to:

1. Investigate variability in the relative abundance of indicator species for all individuals and those that are above the legal minimum size among survey areas and exposure strata with varying degrees of protection.
2. Compare the population size class distribution of blue cod among survey areas with varying degrees of protection.
3. Investigate whether the data obtained from BUV deployments can be used to build species distribution models to investigate the habitat preferences of blue cod and blue moki.
4. Evaluate three key questions related to the BUV methodology at Banks Peninsula:
   a) How long should BUV deployments last?
   b) Are deployments within 200 m of each other spatially autocorrelated?
   c) What is the optimal sampling design to maximise statistical power to detect trends?

2. Methods

2.1 Survey methods

Three purpose-built 'L-frame' type BUV units were used in this study (Fig. 1). These units consisted of two hinged aluminium bars, one of which had a measurement scale marked at 100-mm intervals. GoPro Hero4 Silver cameras that had standard housings and standard lenses with magenta filters were mounted 1135 mm above the scale bar, giving an underwater field of view of 2.4 m^2. The base bar of each frame was weighted with approximately 3 kg of lead and a small pressure float was attached to the camera bar to keep the camera orientated towards the sea floor. Each unit was baited with 300 g of pilchard (*Sardinops neopilchardus*), which was placed in a standard plastic bait holder attached to the base bar.

The study area spanned 13 km of the southern Banks Peninsula coastline and included five 'survey areas': two areas with Type 1 MPA (no-take marine reserve) protection (Akaroa Marine Reserve – AKAMR) and Pōhatu Marine Reserve – POHMR), two areas with Taiāpure protection (outer Akaroa Harbour control – OHCNTRL) and Damon's Bay control – DACNTRL), and one area with no protection (Otanerito Bay control – OTCNTRL) (Fig. 2). The Akaroa Harbour Taiāpure is a customary management area in which the total allowable daily catch for recreational fishers has been reduced from a mixed species limit of 30 per person to 10 per person, and where there are further restrictions on the number of key species that can be harvested, including blue cod (3 per person) and blue moki (3 per person).

Deployment sites were selected using a stratified random design, whereby a random point generator randomly chose sites at each study area within two pre-defined exposure strata. Intermediate strata were defined as all areas >300 m inshore of the headlands that bound the open coastline. Exposed strata stretched from the boundaries of intermediate strata to the open coastline (Fig. 2). All of the key fish species that are used for biological monitoring of protected areas in this region are associated with reef habitat (Davidson et al. 2001) and preliminary trial deployments indicated that few reef-associated species were present at distances of >30 m from the reef edge. At Banks Peninsula, reef structure is generally confined to a coastal fringe, with few examples existing beyond the narrow coastal margin (Fig. 2). The extent of the coastal reef fringe

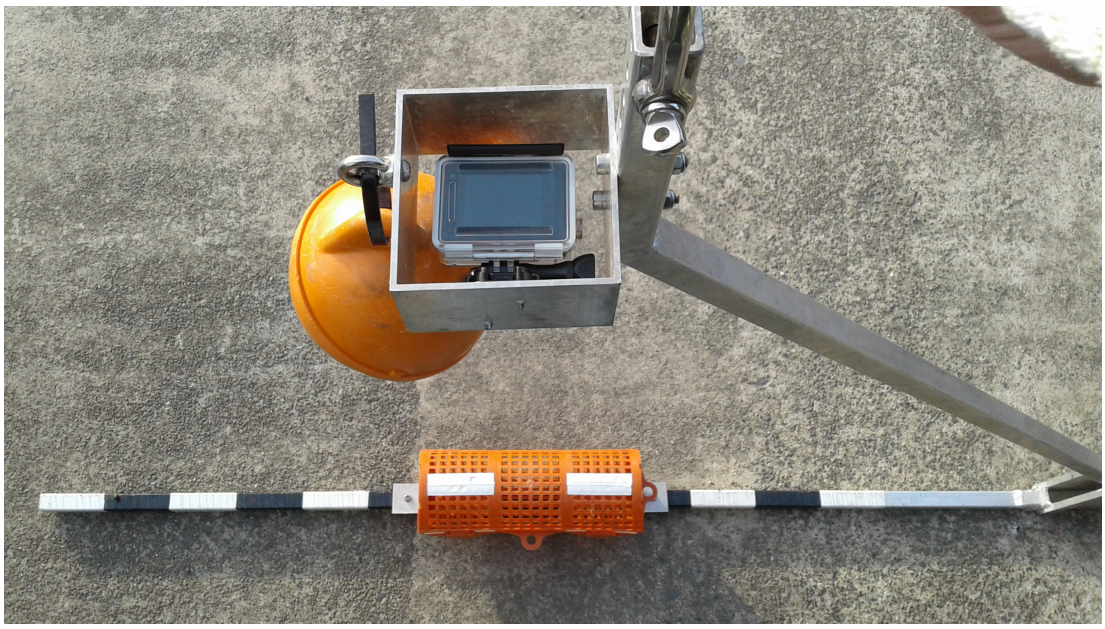

Figure 1. Example of the 'L-frame' baited underwater video (BUV) system used in this study.

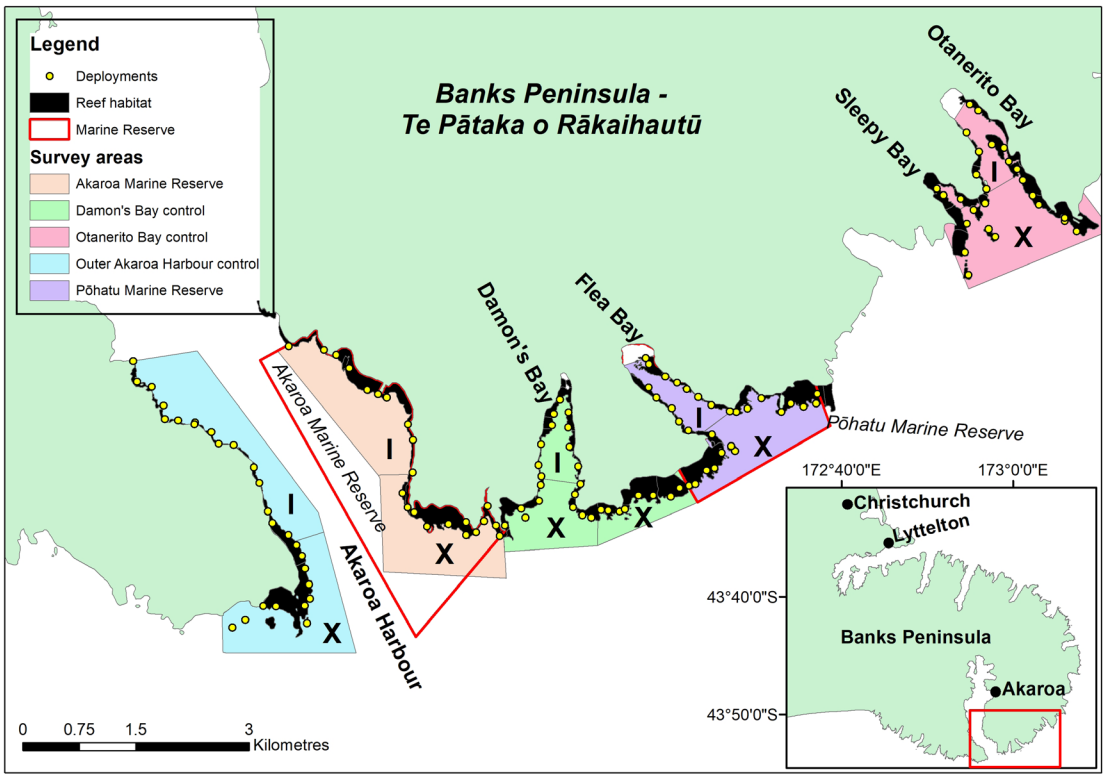

Figure 2. Diagram of the five study areas on the south side of Banks Peninsula. The division within each study areas shows the boundary between the two exposure strata. The location of all 134 baited underwater video (BUV) deployment sites are shown relative to the distribution of reef habitat obtained from side-scan sonar surveys. The Akaroa Harbour Taiāpure covers all of Akaroa Harbour and Damon's Bay. Thus, Otanerito and Sleepy Bays are the only locations with no protection. Exposed (X) and intermediate (I) exposure strata are given for each survey area.

has previously been mapped for the entire study area using side-scan sonar (Dunmore et al. 2015; Brough 2017). Therefore, deployment sites were established within 10 m of either side of the reef edge using a random point generation algorithm in ArcMap (v 10.4.1; ESRI 2017) that incorporated polygon features to delineate the extent of the coastal reef margin. 'Reef' habitat followed the draft New Zealand marine habitat classification system (Dohner 2013) and included bedrock, boulder, cobble and biogenic reefs.

The number of sites sampled per area is a trade-off between obtaining sufficient replication and the potential risk of spatial autocorrelation. Since no information was available on either of these factors for this area, we chose to use the maximum number of replicates possible within each area while maintaining a minimum separation distance of 150 m. This resulted in between 25 and 30 random deployment sites being sampled in each area depending on its size (Table 1). We then investigated the level of statistical power and spatial autocorrelation for this level of replication.

The three units were deployed at the sites in a step-wise manner, with each unit being deployed, one after the other at closely located sites along a particular portion of the coastline. Typically, deployments began at the edge of one study area and worked towards the opposite boundary. The closest site to a deployed unit was skipped in each step to prevent fish from following the bait plume between sites. A maximum of six deployments occurred at a given exposure stratum before sampling was switched to an adjacent stratum or survey area. A full deployment lasted at least 30 minutes. In any instance where a deployment failed (typically due to low water visibility, battery failure or poor unit placement), the deployment at that site was repeated on a later sampling date. Data relating to the date and time of deployment, site ID, count number, geographic coordinates, depth, and water visibility were recorded for each deployment in the field. All deployments occurred during daylight hours.

Table 1. Number of deployment sites for each survey area and exposure stratum. Survey areas were Akaroa Marine Reserve (AKAMR), outer Akaroa Harbour control (OHCNTRL), Damon's Bay control (DACNTRL), Otanerito Bay control (OTCNTRL) and Pōhatu Marine Reserve (POHMR).

AREA	EXPOSED SITES	INTERMEDIATE SITES	TOTAL
AKAMR	11	12	23
DACNTRL	11	16	27
OHCNTRL	15	13	28
OTCNTRL	13	14	27
POHMR	13	16	29
Total			**134**

2.2 Video processing

Videos were processed in accordance with the Department of Conservation's marine monitoring toolbox 'Baited underwater video surveys for fish' (Zintzen 2016). The maximum number of individuals of each species that was observed in any one frame of the BUV footage (Max*N*) was recorded for each 30-minute deployment to give an estimate of species relative abundance for a particular deployment (Willis et al. 2000; Cappo et al. 2003). A sampling window of 1 minute was used to generate Max*N* values, meaning that the Max*N* of each species was evaluated for each 1-minute interval in the BUV footage. The maximum value among the thirty 1-minute replicates was then used in the analysis. Any instances of predator occurrence, intra- and interspecific aggression, and BUV unit drift were also recorded. Data on physical and biological habitat types (Dohner 2013), distance to reef, weather, and additional notes were saved for each processed video.

Size data were obtained by measuring the total length of every individual of a given species at the time of Max*N* for each deployment. Measurement was undertaken on extracted frames with Image J software (imagej.nih.gov) using the 100-mm section of the scale on the BUV unit closest to each fish as a reference. Any instances where measurement error was likely to be significant (i.e. where fish were vertically displaced from the scale or at the edge of the field of view) were discarded. Any size-related bias (due to large fish displacing smaller individuals to the periphery) was reduced by selecting frames where the total fish to be measured were as centrally located in the field of view as possible. Since blue moki were rarely observed at a vertical position that was appropriate for measurement, they were simply estimated as being larger or smaller than the minimum legal size (400 mm). All size estimations and measurements were undertaken by an analyst who was experienced in estimating the size of the target species underwater.

2.3 Data analysis

The data for each deployment were summarised in terms of the number of species, Max*N* for each observed species and the size of individuals at the time of Max*N*.

Data were summarised for four indicator fish species in each deployment over the five survey areas and ten exposure strata. These species included blue cod, blue moki and tarakihi (*Nemadactylus macropterus*), which are popular targets for recreational and commercial fishers, and scarlet wrasse (*Pseudolabrus miles*), which is commonly caught as bycatch in both fisheries.

All analyses were undertaken in R (R Development Core Team 2018). Generalised linear models (GLMs) were used to investigate the variation in relative abundance of each indicator species among survey areas and exposure strata. Separate models were fitted to each indicator species, in which the Max*N* value for a particular deployment was included as the response variable

and either Survey Area or Exposure Stratum was included as a categorical predictor (fixed effects, five and ten levels, respectively). Data were not pooled into 'marine reserve' v. 'control' areas because of the likelihood that differences existed between the two reserves (due to their different ages) as well as among the controls (Taiāpure v. no protection). Models were fitted with a Poisson distribution and a log link function. All models were checked for violation of the model assumptions using appropriate diagnostic plots (e.g. Zuur et al. 2009). Pairwise Tukey's post-hoc tests were performed on each model to test for differences between different survey areas and exposure strata using the *multcomp* package in R (Hothorn et al. 2017).

Size data were analysed in two different ways using GLMs. First, the relative abundance of legal-sized blue cod and blue moki was assessed among survey areas and exposure strata by including the number of legal-sized fish in a particular deployment as the response variable and either Survey Area or Exposure Stratum as a categorical predictor. Second, differences in the absolute size of blue cod were modelled among the two spatial blocks (survey area, exposure) by including the size of the measured fish as the response variable and fitting the data with a Gaussian distribution.

The number of predatory/scavenger species among survey areas was investigated by generating values of species richness for each deployment. The spatial distribution of species richness across survey areas and exposure strata was assessed using a GLM that was fitted with a Gaussian distribution and Tukey's post-hoc tests. While BUV has known biases for assessing diversity, namely due to sampling carnivorous fishes only, information on the distribution of species richness is informative within the context of MPA monitoring due to the influence fishing pressure can have on carnivorous fish communities (see Discussion).

2.4 Validation of the BUV methodology

Two fundamental assumptions of the BUV methodology were checked using statistical procedures. First, the method assumes that Max*N* represents a valid estimate of the relative abundance of a particular species at the deployment site. The validity of this assumption can be assessed by generating a 'discovery curve' that plots the effect of an increasing number of counts (e.g. 1-minute count intervals) on the final Max*N* value for a given deployment and examining whether the curve plateaus before the last count value (the 30th in this case). Such a plateau may indicate that all of the individuals in the deployment area had visited the bait station and so a 30-minute deployment was adequate to measure a valid Max*N*. To examine this, we calculated the mean and standard error of count intervals across all deployments. Plotting these values provides an indication of a plateau in Max*N*. This process was undertaken for blue cod and blue moki.

A second assumption of the BUV methodology, and indeed all statistical analyses, is that the deployments are independent from each other. This assumption will be violated if spatial autocorrelation exists among deployment sites (i.e. from the same survey area). The 150-m site separation used in this study was smaller than has been used elsewhere (Zintzen 2016) and was set to maximise the number of deployments in each area. To appraise non-independence of closely located deployments, we generated spatial correlograms that calculated and plotted the extent of correlation between pairs of deployments as a function of increasing lag distance (Zuur et al. 2009: 481). We investigated spatial autocorrelation at the level of both the raw count data and the model residuals using a spline-based correlogram and bootstrapped 95% confidence intervals. This was undertaken for both blue cod and blue moki counts. Some repeated sites were also included in this analysis to assess correlation at distances of <150 m.

Since this study was one of the first to use BUV methodology for the biological monitoring of coastal habitat in the South Island, it was also important to assess the power of the sampling design to identify trends among spatial blocks. Therefore, we used the package *SimR* (Green & Macleod 2016) to simulate the power of the design to reject the null hypothesis of no difference

among survey areas. *SimR* is designed for both general linear and general linear mixed effects models with a range of response distributions (Green & Macleod 2016). The probability of the model to reject the null hypothesis was determined via 1000 Monte Carlo simulations, and the significance of the survey area parameter was determined by likelihood ratio tests for each simulation. The *powerCurve* function was used to investigate:

1. The power of this study to reject the null hypothesis given the size of the effects we observed. The effect sizes were calculated as 0.15 and 0.2 for the power analyses of survey area and exposure stratum, respectively, which were calculated using adjusted r^2 values (see Ford 2017).
2. To determine the minimum number of deployments required to achieve a reasonable amount of statistical power (80%) under a range of effects sizes.

2.5 Habitat study

Habitat data were obtained from the BUV deployments and the side-scan sonar habitat maps produced by Dunmore et al. (2015) and Brough (2017). BUV deployments provide a point sample of the habitat at the deployment site and were summarised into physical and biological habitat types following a draft New Zealand classification system (Dohner 2013) based on the primary habitat component identified in the field of view of the camera. Deployment depth was also recorded at the time of sampling using the research vessel's echo-sounder. Parameters derived from habitat maps included reef width (the width of the coastal reef fringe at the deployment site), seabed slope and distance to reef. These parameters were calculated in ArcMap version 10.4 for each deployment.

Species–habitat models were developed using generalised additive models (GAMs), which were fitted using the package *mgcv* in R. Models were constructed for blue cod and blue moki, using their respective Max*N* values as the response variables at a given deployment site. First, global models that contained every habitat descriptor were fitted to each species dataset. Continuous covariates (reef width, distance from reef, slope and depth) were fitted as either linear terms or smoothed functions using a cubic spline, with the final form being decided by selecting models with the lowest Akaike Information Criterion (AIC) values (Akaike 1973). The number of degrees of freedom for the smoothed terms were estimated at 'optimum' levels (Wood 2006). Second, a model selection table was produced that contained a model formulation for every possible combination of input variables. Each model was fitted and evaluated using an information-theoretic model selection process. Models that had the lowest AIC scores and highest model weights were selected as the 'top' models (Burnham & Anderson 1998).

The statistical significance, magnitude and nature of the effect of parameters that were retained in the top models were assessed by evaluating the *P* values for linear and smoothed terms. In addition, smoothed terms were plotted in order to visualise the nature and magnitude of their effects. The influence of the categorical habitat type variables that were retained in the top models was assessed by setting all other input variables to their mean values and using the models to predict the value of interest.

The validity of the model assumptions was checked for both the global and top models using a variety of diagnostic procedures to determine co-linearity (and concurvity), residual independence and homogeneity, and over- and under-smoothing.

3. Results

3.1 Sampling

A total of 187 deployments were made across the study area, 134 of which were retained for analysis following the removal of bad deployments and repeats of failed deployments. The sampling effort was similar between areas and exposure strata (Table 1). AKAMR had a lower number of completed deployments than the other areas because some sites in this area needed to be discarded due to repeated bad visibility.

A total of 28 fish species were recorded over the entire study area (Table 2). The most commonly observed species were blue cod, followed by scarlet wrasse, banded wrasse (*Notolabrus fucicola*) and blue moki. The least common species included southern bastard cod (*Pseudophycis barbata*),

Table 2. Frequency of occurrence of species observed during the baited underwater video (BUV) deployments. No. sites = the number of unique deployment sites where species were observed. Survey areas were Akaroa Marine Reserve (AKAMR), Damon's Bay control (DACNTRL), outer Akaroa Harbour control (OHCNTRL), Otanerito Bay control (OTCNTRL) and Pōhatu Marine Reserve (POHMR). The total number of deployments per area is given in parentheses.

SCIENTIFIC NAME	COMMON NAME	NO. SITES	AKAMR (23)	DACNTR (27)	OHCNTRL (28)	OTCNTRL (27)	POHMR (29)
Parapercis colias	Blue cod	127	23	25	25	25	29
Pseudolabrus miles	Scarlet wrasse	110	22	26	19	18	25
Notolabrus fucicola	Banded wrasse	61	11	12	14	7	17
Latridopsis ciliaris	Blue moki	56	13	5	8	13	17
Meuschenia scaber	Leatherjacket	47	6	13	10	8	10
Jasus edwardsii	Spiny rock lobster	34	7	7	5	7	8
Nemadactylus macropterus	Tarakihi	33	5	4	7	6	11
Hypoplectrodes huntii	Redbanded perch	28	4	7	7	5	5
Helicolenus percoides	Sea perch	21	5	3	6	6	1
Thyrsites atun	Barracouta	18	3	0	1	10	4
Congiopodus leucopaecilus	Southern pigfish	12	2	1	5	2	2
Latris lineata	Trumpeter	12	5	4	0	1	2
Lotella rhacina	Rock cod	12	1	5	2	1	3
Eptatretus cirrhatus	Hagfish	11	2	1	2	4	2
Cephaloscyllium isabellum	Carpet shark	9	2	0	6	1	0
Pseudophycis bachus	Red cod	9	0	3	1	3	2
Arripis trutta	Kahawai	8	1	1	1	2	3
Conger verreauxi	Conger eel	8	1	2	2	2	1
Squalus acanthias	Spiny dogfish	8	1	0	1	5	1
Zearaja nasuta	Rough skate	8	1	0	1	4	2
Aplodactylus arctidens	Marblefish	7	2	1	0	1	3
Notolabrus cinctus	Girdled wrasse	7	1	1	1	3	1
Dipturus innominatus	Smooth skate	4	2	1	1	0	0
Notorynchus cepedianus	Seven gill shark	4	0	0	0	0	4
Odax pullus	Butterfish	4	0	1	1	0	2
Macroctopus maorum	Octopus	3	1	1	0	0	1
Mustelus lenticulatus	Rig	2	1	0	0	0	1
Pseudophycis barbata	Southern bastard cod	2	1	0	1	0	0
Chelidonichthys kumu	Gurnard	1	1	0	0	0	0
Galeorhinus galeus	School shark	1	0	0	0	1	0

rig (*Mustelus lenticulatus*) and gurnard (*Chelidonichthys kumu*). Several fish species that are important for recreational and/or commercial fishing were recorded, including blue cod, blue moki, tarakihi, sea perch (*Helicolenus percoides*), trumpeter (*Latris lineata*), red cod and kahawai (*Arripis trutta*). However, only the first three of these were encountered frequently enough for statistical analysis. The observed fish species were typically demersal or benthic, although some epipelagic species (e.g. kahawai and barracouta (*Thyrsites atun*)) were also recorded. As expected, the majority of species seen in this survey were predators or scavengers. Exceptions to this included butterfish (*Odax pullus*) and marblefish (*Aplodactylus arctidens*), which are herbivorous and were observed at four and seven sites, respectively. Several large-bodied predators, including seven-gill shark (*Notorynchus cepedianus*) and school shark (*Galeorhinus galeus*), were recorded in low numbers at five sites. The spiny rock lobster (*Jasus edwardsii*) was also recorded in reasonably high numbers and was distributed throughout the study area (Table 2).

3.2 Relative abundance

3.2.1 Variation among survey areas

There were statistically significant differences in the relative abundances of blue cod, blue moki, terakihi and scarlet wrasse among the survey areas. Blue cod had the highest relative abundance at POHMR and the lowest relative abundance at OHCNTRL and DACNTRL (Fig. 3). However, this species also had a high relative abundance at OTCNTRL. The abundance of blue cod at AKAMR was significantly higher than at DACNTRL (Est. = −0.43, z = −3.49, df = 129, P = 0.004) and OHCNTRL (Est. = −0.52, z = −4.15, P < 0.001), but was similar to OTCNTRL (Fig. 3). The GLM and post-hoc tests confirmed that there was no statistically significant difference in blue cod relative abundance between sites at OTCNTRL and the two marine reserves (AKAMR: Est. = −0.07, z = −0.61, df = 129, P = 0.97; POHMR: Est. = 0.18, z = 1.82, df = 129, P = 0.36). However, significant differences were observed between both marine reserves and the other two control areas (Appendix 1).

Blue moki were most abundant at AKAMR and POHMR and least abundant at DACNTRL (Fig. 3). Statistical analysis confirmed that there was a significant difference in the relative abundance of moki among survey areas (Appendix 1), with both marine reserves having

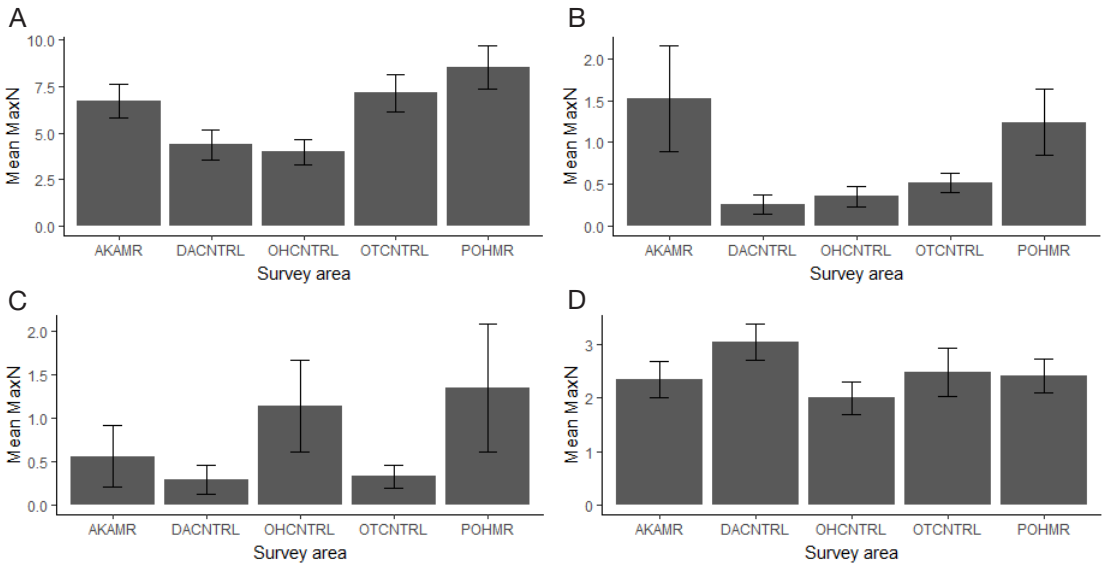

Figure 3. Mean relative abundances (Max*N* values) of A. blue cod (*Parapercis colias*), B. blue moki (*Latridopsis ciliaris*), C. terakihi (*Nemadactylus macropterus*) and D. scarlet wrasse (*Pseudolabrus miles*) for all deployments across the five survey areas. Error bars represent standard errors. AKAMR: Akaroa Marine Reserve; DACNTRL: Damon's Bay control; OHCNTRL: outer Akaroa Harbour control; OTCNTRL: Otanerito Bay control; POHMR: Pōhatu Marine Reserve.

significantly higher abundances than the three control areas (Est. = >0.8, $z = 2.5$, df = 129, $P < 0.05$ for all pairwise comparisons). There was no significant difference in the relative abundance of blue moki between AKAMR and POHMR (Est. = −0.2, $z = -0.86$, df = 129, $P = 0.91$; Appendix 1).

The relative abundance of tarakihi was highest at POHMR but was also high at OHCNTRL (Fig. 3). However, there was substantial variation in Max*N* at both of these locations. Tarakihi had a low abundance at the remaining three survey areas. The relative abundance of tarakihi was significantly higher at POHMR than at DACNTRL (Est. = 1.51, $z = 3.90$, df = 129, $P < 0.001$) and OTCNTRL (Est. = 1.39, $z = 3.77$, df = 129, $P = 0.001$). However, there was no significant difference in tarakihi abundance between POHMR and OHCNTRL (Est. = 0.16, $z = 0.68$, df = 129, $P = 0.96$; Appendix 1).

Scarlet wrasse were most abundant at DACNTRL but had a similar abundance across all five survey areas (Fig. 3), with no significant difference among areas ($P > 0.05$ for all pairwise comparisons).

3.2.2 Variation among exposure strata

There were also substantial differences in the relative abundances of the four indicator fish species among sites with different exposure levels (Fig. 4). The two survey areas where blue cod were most common (OTCNTRL and POHMR) had higher abundances at exposed sites. By contrast, there was no difference in blue cod abundance between exposed and intermediate sites at AKAMR, while blue cod were present at higher numbers at intermediate sites at DACNTRL and OHCNTRL (Fig. 4). The difference in abundance between exposed and intermediate sites was found to be significant for POHMR (Est. = 0.62, $z = 4.50$, df = 124, $P < 0.001$) and DACNTRL (Est. = −0.82, $z = -4.36$, df = 124, $P < 0.001$) (Appendix 2).

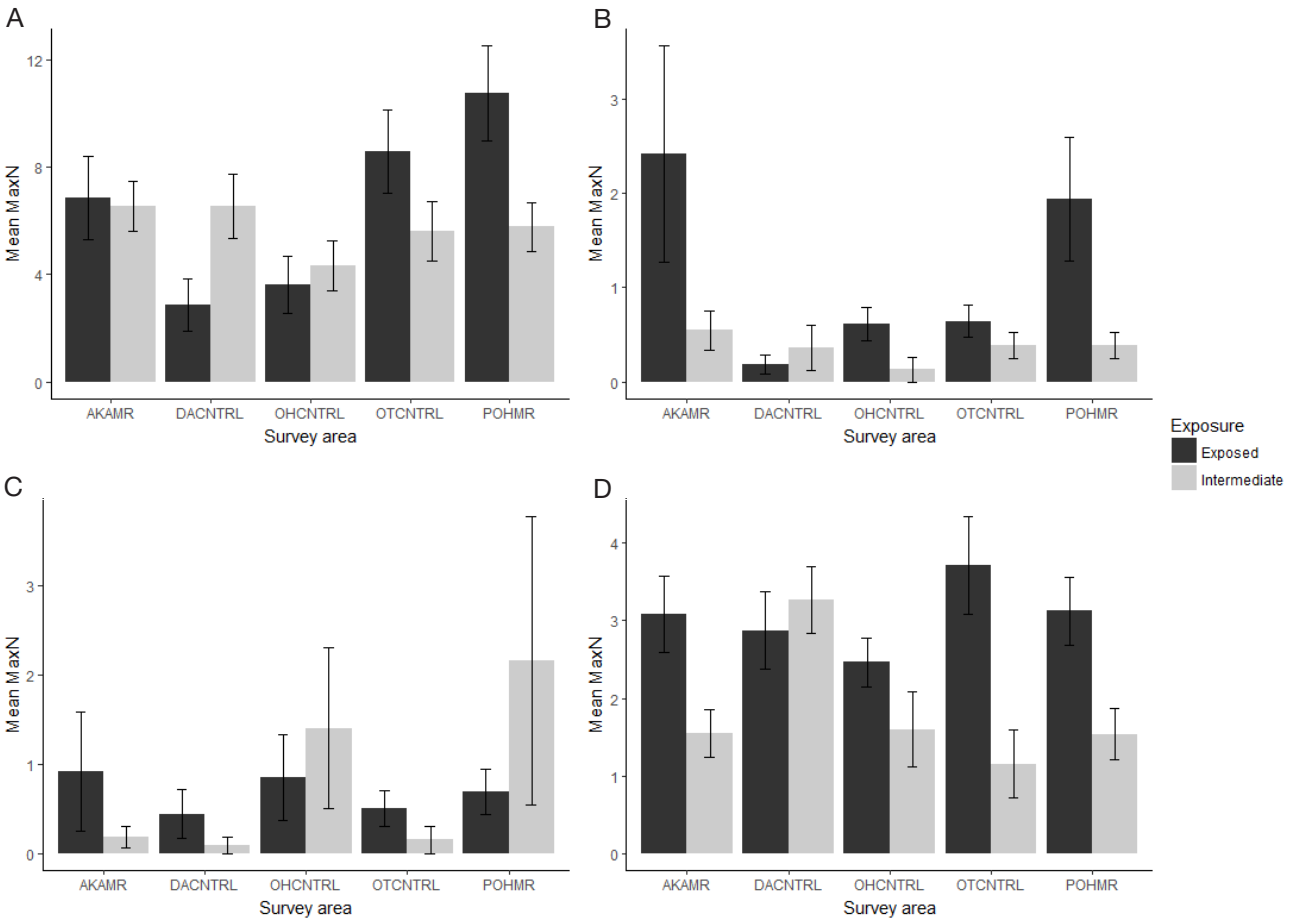

Figure 4. Mean relative abundances (Max*N* values) of A. blue cod (*Parapercis colias*), B. blue moki (*Latridopsis ciliaris*), C. terakihi (*Nemadactylus macropterus*) and D. scarlet wrasse (*Pseudolabrus miles*) for all deployments at exposed and intermediate exposure strata for each of the five survey areas. Error bars represent standard errors. AKAMR: Akaroa Marine Reserve; DACNTRL: Damon's Bay control; OHCNTRL: outer Akaroa Harbour control; OTCNTRL: Otanerito Bay control; POHMR: Pōhatu Marine Reserve.

Exposed sites in the two marine reserves had particularly high blue moki abundances, with mean Max*N*s that were up to three times higher than exposed sites in the control areas (Fig. 4). By contrast, intermediate sites had similar abundances of blue moki among all survey areas (Fig. 4). There was a significant difference in blue moki abundance between exposed and intermediate sites at AKAMR (Est. = 1.49, z = 3.32, df = 124, P = 0.01) and POHMR (Est. = 1.62, z = 3.36, df = 124, P = 0.01), and exposed sites in these marine reserves also had significantly higher abundances of blue moki than all three of the control sites (Appendix 2).

Exposure had a variable effect on the abundance of tarakihi, with no clear patterns being observed among survey areas.

Scarlet wrasse were more abundant at exposed than intermediate sites for all survey areas except DACNTRL (Fig. 4).

3.3 Fish size

A total of 716 individual blue cod were measured throughout the study area, but only 643 of these were retained after grading for measurement quality. These blue cod ranged from 72 to 521 mm in length with an overall mean of 239 mm, and 134 of these fish were above the minimum legal size of 300 mm (Fig. 5). On average, blue cod were largest at POHMR (261 ± 6.2 mm) and AKAMR (234 ± 6.9 mm), intermediate at OTCNTRL (231 ± 5.3 mm), and smallest at OHCNTRL (222 ± 6.05 mm) and DACNTRL (226 ± 7.3 mm). Pairwise comparisons of marine reserve and control areas showed that differences in the size of fish were small but statistically significant (Est. = > 0.03 < 0.16, z > 3.65 < 20.15, df = 129, P < 0.05), with the exception of the comparison between AKAMR and OTCNTRL (Appendix 3).

The relative abundance of blue cod above the minimum legal size was substantially different among survey areas (Fig. 6). POHMR had the greatest number of blue cod above the minimum size limit (2 ± 0.4 per deployment), followed by AKAMR (1.17 ± 0.39), whereas all three control areas had low numbers of legal-sized blue cod (pooled mean = 0.56). Differences in the number of large blue cod between each marine reserve and control site were statistically significant (Est. = > 0.89 < 1.56,

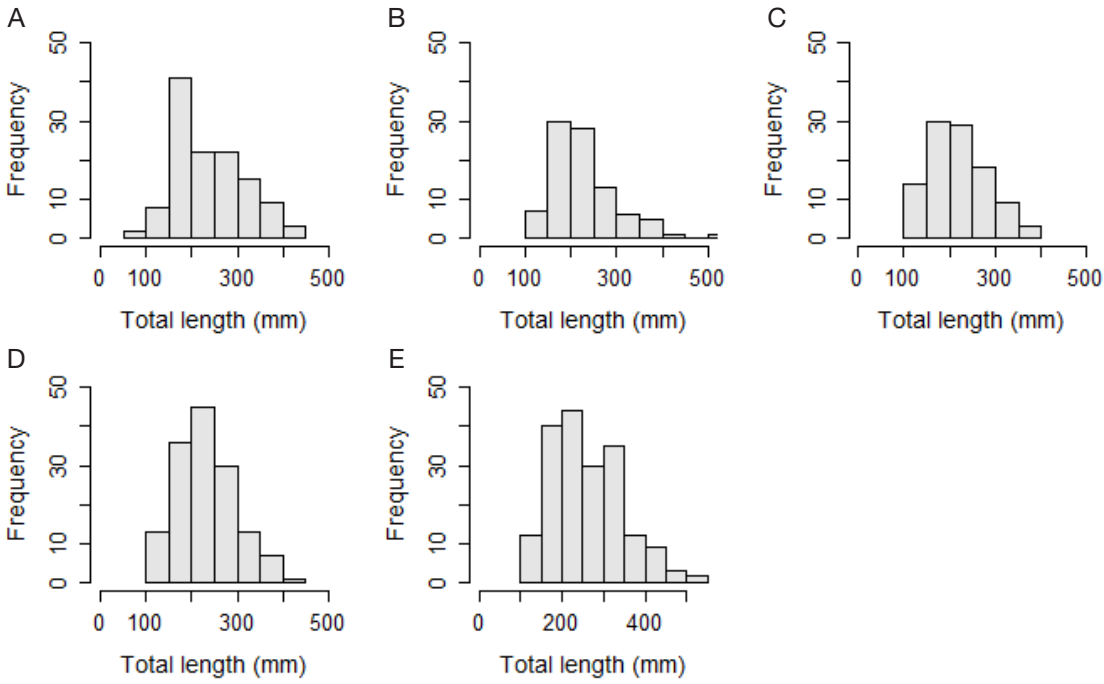

Figure 5. Population size class distribution of blue cod (*Parapercis colias*) at baited underwater video (BUV) deployment sites across the five survey areas: A. Akaroa Marine Reserve (AKAMR), B. Damon's Bay control (DACNTRL), C. outer Akaroa Harbour control (OHCNTRL), D. Otanerito Bay control (OTCNTROL) and E. Pōhatu Marine Reserve (POHMR). The minimum legal size for blue cod at Banks Peninsula is 300 mm.

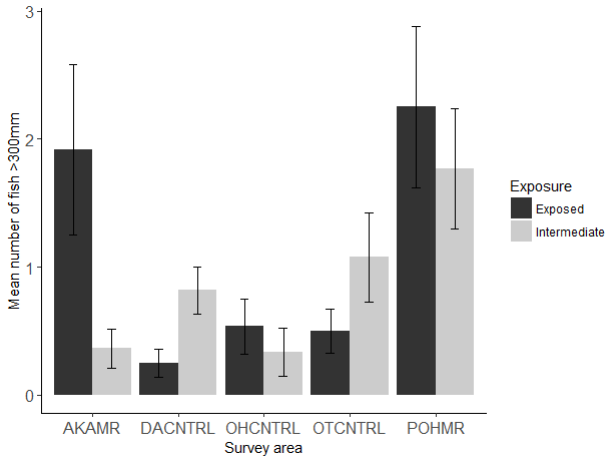

Figure 6. Mean relative abundance of legal-sized blue cod (*Parapercis colias*) in exposed and intermediate strata in each survey area. Error bars represent standard errors. AKAMR: Akaroa Marine Reserve; DACNTRL: Damon's Bay control; OHCNTRL: outer Akaroa Harbour control; OTCNTRL: Otanerito Bay control; POHMR: Pōhatu Marine Reserve.

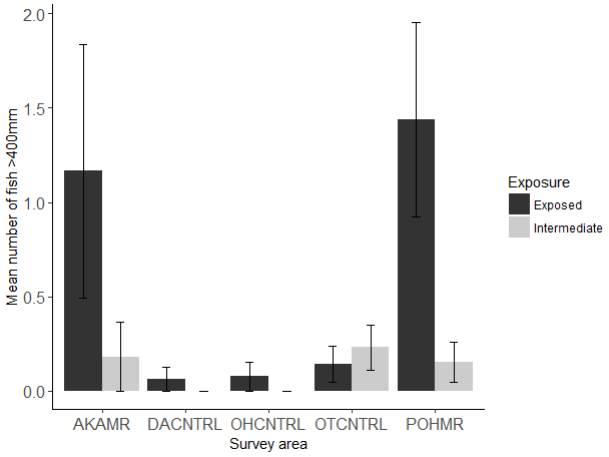

Figure 7. Mean relative abundance of legal-sized blue moki (*Latridopsis ciliaris*) in exposed and intermediate strata in each survey area. Error bars represent standard errors. AKAMR: Akaroa Marine Reserve; DACNTRL: Damon's Bay control; OHCNTRL: outer Akaroa Harbour control; OTCNTRL: Otanerito Bay control; POHMR: Pōhatu Marine Reserve.

$z > 2.64 < 4.92$, df = 129, $P < 0.05$), with the exception of the difference between AKAMR and OTCNTRL (Est. = −0.41, $z = -1.42$, df = 129, $P = 0.61$; Appendix 4). The number of legal-sized blue cod was greater at exposed sites than at intermediate sites in the marine reserve areas (Fig. 6). However, this difference was only statistically significant for AKAMR (Est. = 1.66, $z = 3.1$, df = 124, $P = 0.03$), as both intermediate and exposed sites at POHMR had high numbers of large blue cod.

A total of 90 blue moki were measured throughout the study area, 48 of which were above the minimum legal size of 400 mm. The distribution of large blue moki among survey areas was similar to that of blue cod, with fish above the minimum legal size being more common in the marine reserve survey areas (mean number per deployment = 0.70 at AKAMR, 0.86 at POHMR, 0.04 at OHCNTRL and DACNTRL, and 0.19 at OTCNTRL; Fig. 7). All pairwise comparisons of the relative abundance of legal-sized blue moki between marine reserve and control sites were statistically significant (Est. = > 1.54 < 3.18, $z > 2.85 < 3.14$, df = 129, $P < 0.05$) (Appendix 4), except for the difference between AKAMR and OTCNTR. This result appeared to have been driven by the high numbers of legal-sized blue moki at exposed sites in both marine reserves. No exposure effect was evident in the control survey areas (Fig. 7).

In total, 101 tarakihi were measured in this study, only nine of which were above the minimum legal size of 250 mm. The majority of these ($n = 7$) were found at sites within POHMR, while no legal-sized fish were recorded at OHCNTRL or DACNTRL. The distribution of legal-sized tarakihi was not analysed statistically due to the low sample size.

3.4 Species richness

There was no significant difference in species richness among survey areas (Fig. 8 and Appendix 5). Some differences in species richness were observed at particular exposure strata, however. In all survey areas, exposed sites had a higher richness than intermediate sites (Fig. 8). This effect was most notable at OHCNTRL, OTCNTRL and POHMR (Appendix 5).

3.5 BUV validation

Plots of the mean fish counts over 1-minute intervals indicated that Max*N* values were achieved well before the end of a deployment for the two most common key species (Fig. 9). For blue cod, the count values plateau at around 12 minutes into deployments, with further counts having no effect on the blue cod abundance. This suggests that a 30-minute deployment may be

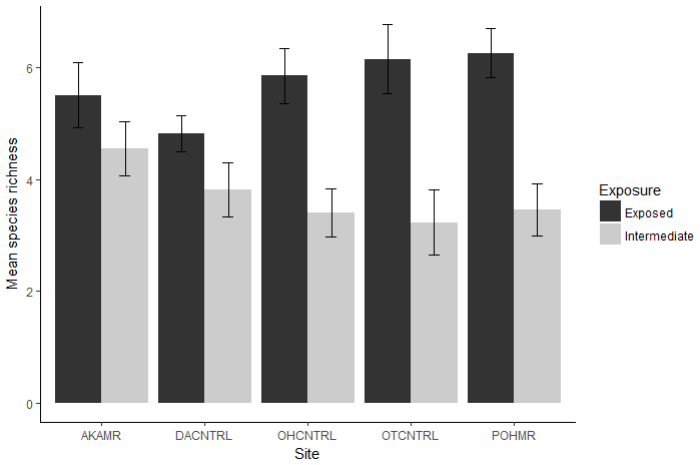

Figure 8. Patterns in the species richness of carnivorous fish species at two different exposure strata in the five survey areas. Error bars represent standard errors. AKAMR: Akaroa Marine Reserve; DACNTRL: Damon's Bay control; OHCNTRL: outer Akaroa Harbour control; OTCNTRL: Otanerito Bay control; POHMR: Pōhatu Marine Reserve.

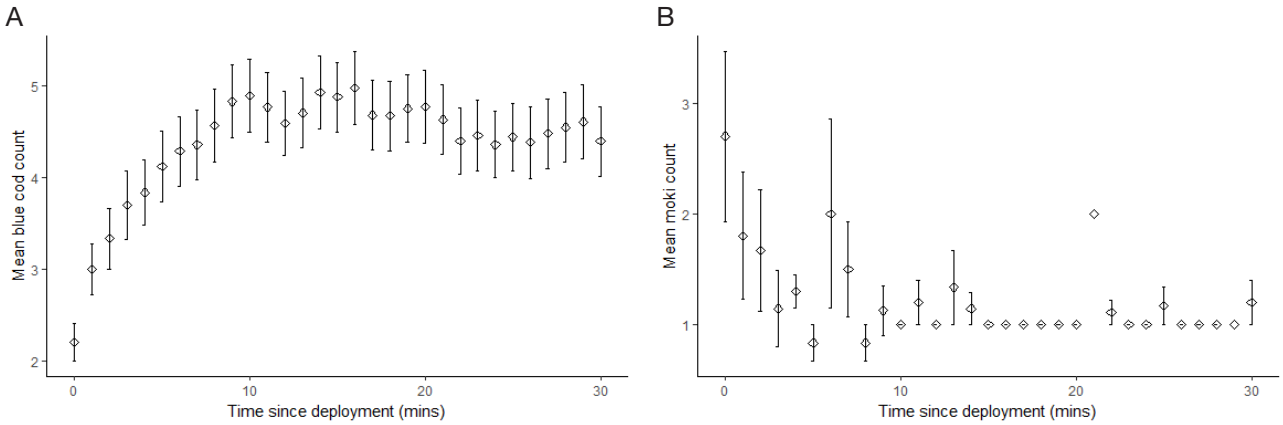

Figure 9. Plots showing the effect of deployment time on count values for A. blue cod (*Parapercis colias*) and B. blue moki (*Latridopsis ciliaris*) within each deployment. These 'discovery curves' show the influence of increasing numbers of 1-minute count intervals on the final Max*N* value in a deployment.

adequate for sampling the number of fish at a deployment site. However, deployments longer than 30 minutes are required to confirm this. For blue moki, an increasing deployment time had an inverse effect on the number of fish counted within each deployment but this effect also plateaued at around 15 minutes into a deployment (Fig. 9). This result is consistent with our field observations that schools of blue moki initially approached the bait but quickly lost interest. Since Max*N* values were reached well before the last count interval in a deployment, 30-minute deployments also appear to be adequate for assessing the relative abundance of blue moki.

Analysis of the spatial autocorrelation among deployment sites showed some evidence of a correlation at distances of <100 m for blue cod (Fig. 10). However, correlation values were close to zero at lag distances of >150 m, indicating that the minimum deployment spacing of 150 m that was used in this study provided independent response variables in the statistical analyses. The correlation of blue moki counts among sites was similarly low, although there was some evidence of a negative correlation at lag distances of <150 m. This was likely due to only a very small number of sites being included in the analysis at such close proximity. The correlation values were close to zero for blue moki at a distance of 150 m. This lack of correlation only serves as an indication that fish were not travelling between deployment sites, however, with data on the movement patterns of individual fish being required to confirm this.

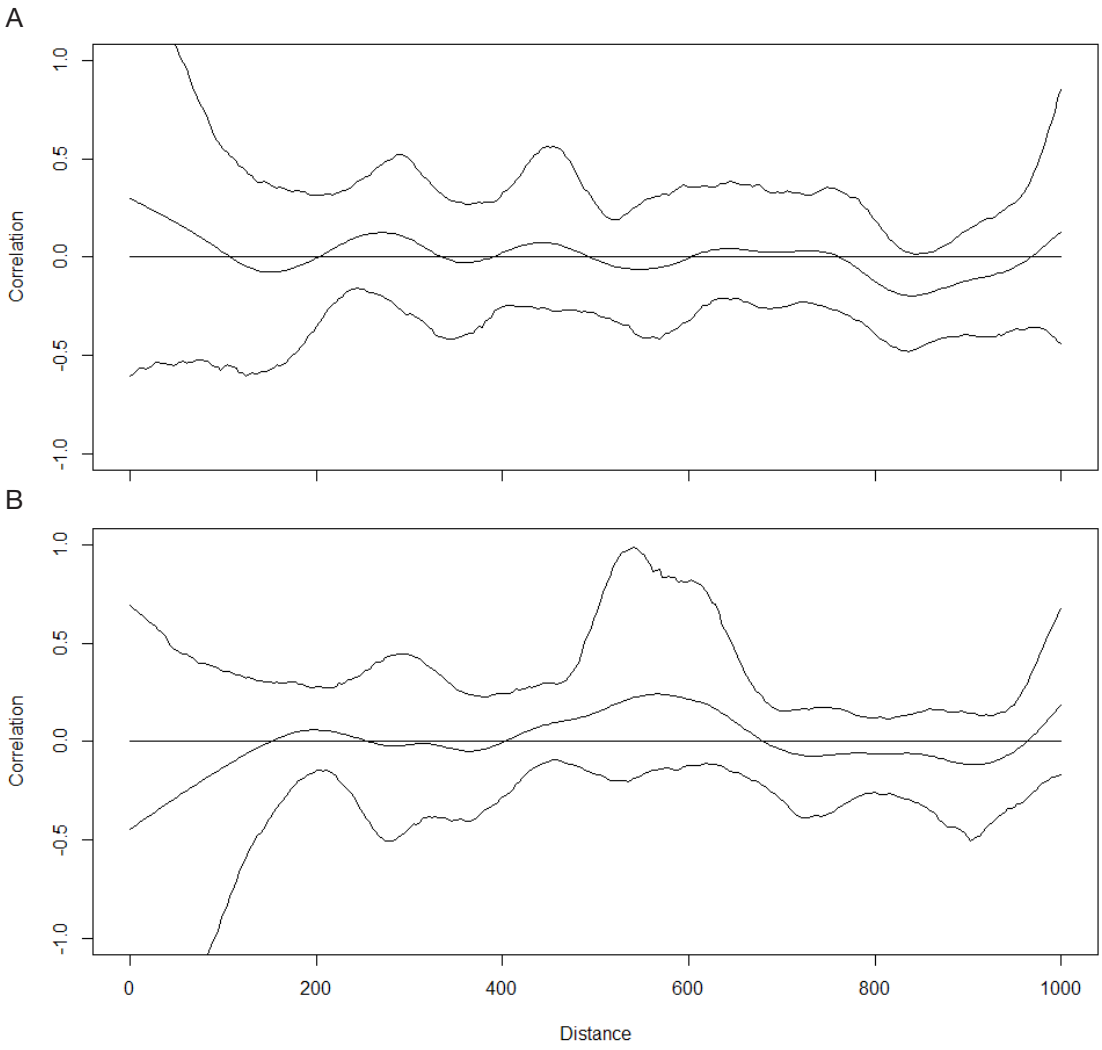

Figure 10. Spline-based spatial correlograms for the simulation of spatial autocorrelation among deployment sites with increasing lag distance (m). Correlation estimates are bound by bootstrapped 95% confidence intervals. The analysis was undertaken for the abundance of A. blue cod (*Parapercis colias*) and B. blue moki (*Latridopsis ciliaris*).

The power analysis showed that there was ample power to detect trends among survey areas given the number of deployments (power = 93%, 95%CI = 91.24 – 94.5). Likewise, the power to detect differences among exposure strata was also high (power = 98%, 95%CI = 96.34 – 100). Therefore, we could have detected differences with a reasonable level of power (80%) by using 116 deployments for differences among survey areas and 87 deployments for differences among exposure strata (Fig. 11). If differences among spatial blocks were small (e.g. effect size = 0.05), we would need to substantially increase the number of deployments in order to detect significant differences (Fig. 11).

3.6 Species–habitat relationships

The global model was found to be the best model for assessing the relationship between habitat characteristics and the relative abundance of blue cod (Appendix 6). This model contained the factors biological and physical habitat types, a linear term for reef width, and the smoothed terms of depth, distance to reef and slope, and explained 34% of the variation in cod abundance.

All of the variables that were included in the best model for blue cod relative abundance were statistically significant, with the exception of the linear term reef width. In terms of biological habitat, a significant positive relationship was observed between bare reef and blue cod abundance (Appendix 7), as indicated by the high predicted values for blue cod in this habitat type (Fig. 12A). However, high variation around the predicted values suggests that differences

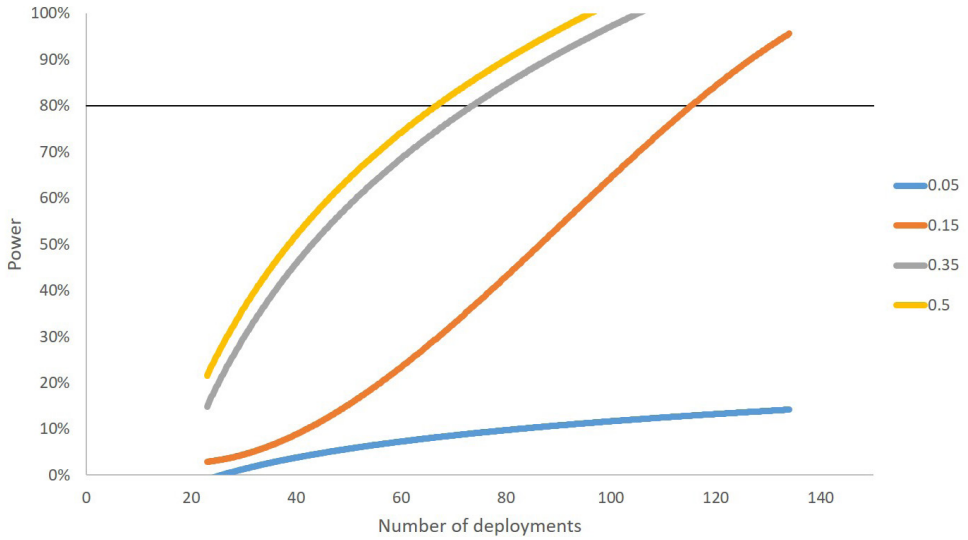

Figure 11. Plots of power curves generated from the *SimR* function to determine the number of deployments required to achieve reasonable statistical power (80%) to reject the null hypothesis of no differences among survey areas or exposure strata. Curves are given for four effect sizes that are indicative of small (0.05), medium (0.15) and large (>0.35) according to Ford (2017). In this study, effect sizes were calculated as 0.15 and 0.20 for survey areas and exposure strata respectively. All power analyses were undertaken with sigma = 0.05.

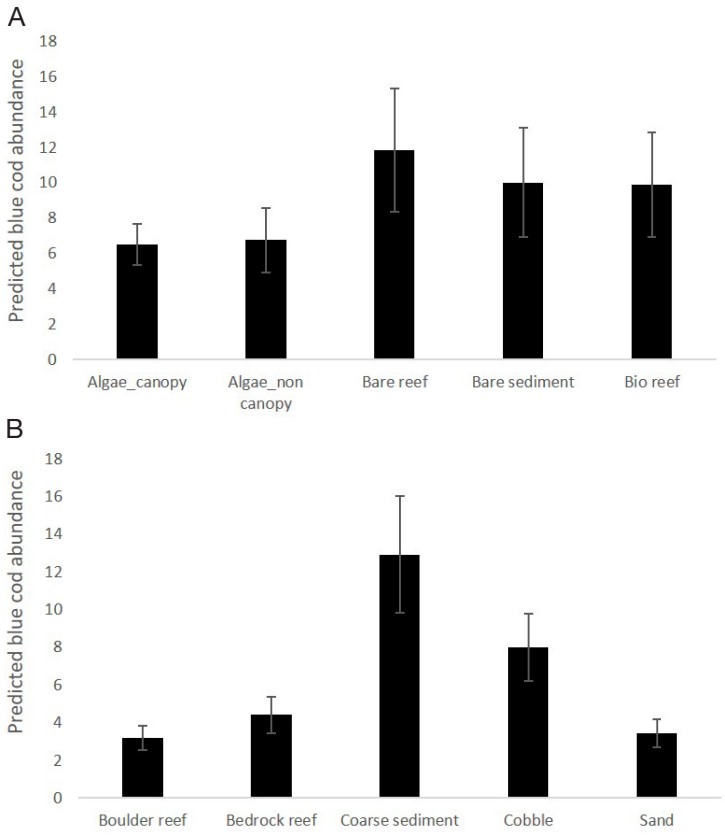

Figure 12. Predicted values of blue cod (*Parapercis colias*) abundance for each A. biological and B. physical habitat type. Predictions were based on the best model that was used to examine the relationship among blue cod counts and habitat characteristics, and were generated using mean values of the other input parameters so that each value represents the influence of one particular habitat type. Error bars represent standard errors of the predicted values.

among biological habitats were small, with contrasts between bare reef and algae-dominated habitats being the only comparisons that held much certainty. Physical habitat had a much stronger effect on blue cod abundance (Fig. 12B). In particular, coarse sediment and, to a lesser extent, cobble had a strong positive effects on blue cod abundance (Fig. 12B). By contrast, boulder reef, bedrock reef and sand had similar, low predicted values of blue cod abundance.

The smoothed effect of seabed slope had a variable effect on blue cod abundance (Fig. 13A). Low slope values had a small positive influence on blue cod abundance, while slopes greater than 12 degrees had a strong negative effect. Increasing depth had a positive effect on blue cod abundance, although the strength of this effect appeared to plateau at a depth of 15–20 m and this effect was negative at depths of <10 m (Fig. 13B). Distance to reef had a clear negative effect on blue cod abundance (Fig. 13C). However, since few deployments were made further than 15 m from the reef edge, there is some imprecision around the true effect at greater distances.

For blue moki, the model that best described the relationship between habitat characteristics and relative abundance included biological habitat type, a linear term for reef width, and the smoothed terms of depth and slope (Appendix 6). This model explained 48% of the deviance in moki abundance.

The parameter estimates and predicted values from the top model suggest that algal canopy habitat has a strong positive effect on blue moki abundance, yielding predicted values that were nearly four times greater than the next best habitat type (Algae_non canopy; Fig. 14). All other biological habitats had a significant negative effect on blue moki abundance compared with algal canopy, particularly bare reef, bare sediment and biogenic reef (Appendix 7). No results are presented for the influence of physical habitat as this variable was not included in the best model for blue moki and so is unlikely to be important.

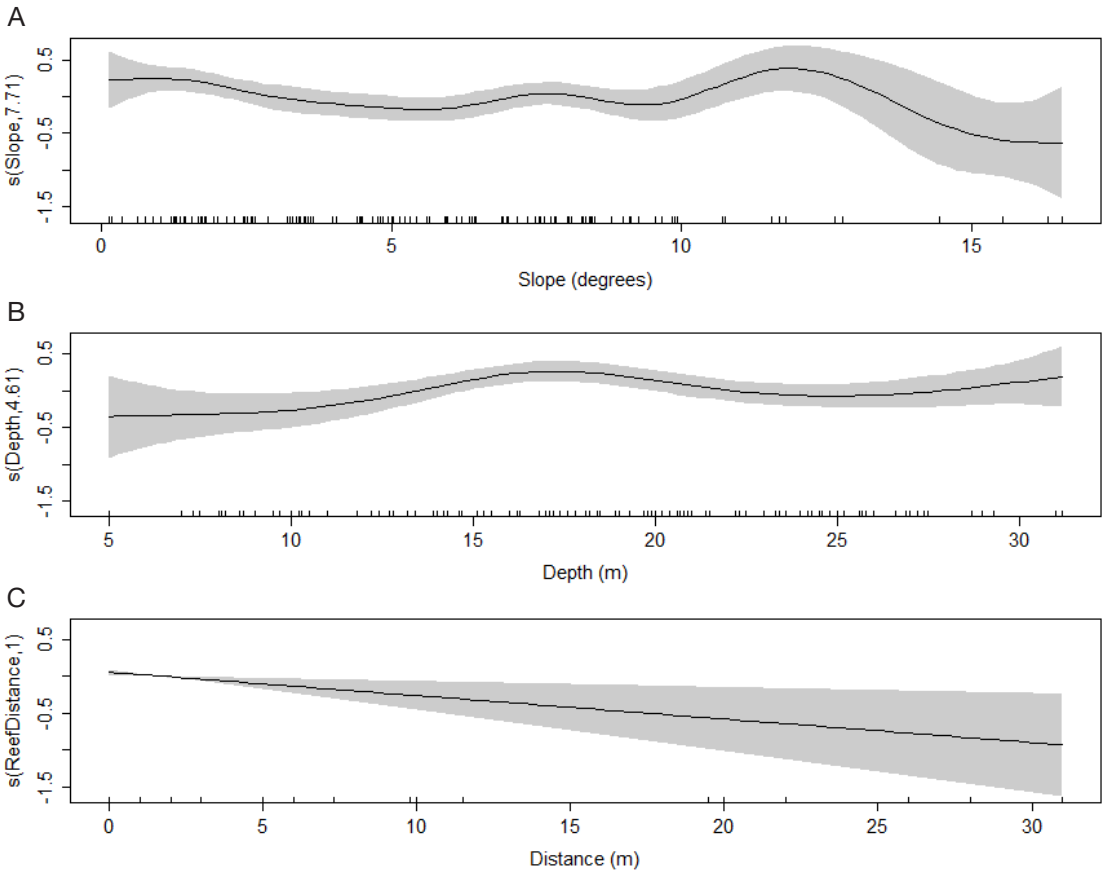

Figure 13. Smoothed term effects from generalised additive models for the relationships between blue cod (*Parapercis colias*) relative abundance and A. seabed slope, B. deployment depth and C. distance to reef. The shaded region on each chart show the 95% confidence interval for the given smoothed effect.

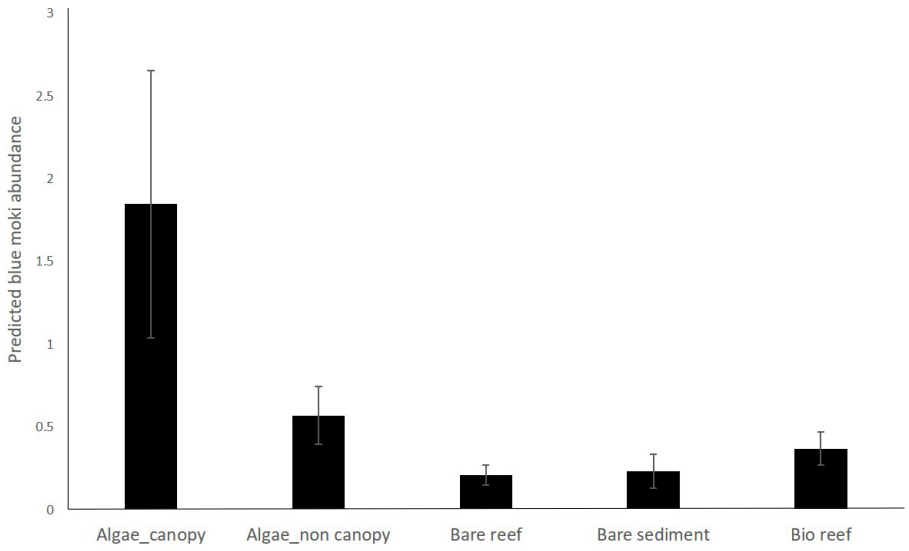

Figure 14. Predicted values of blue moki (*Latridopsis ciliaris*) abundance for each biological habitat type. Predictions were based on the best model that was used to examine the relationship among moki counts and habitat characteristics, and the predicted values were generated at mean values of the other input parameters so that each value represents the influence of one particular habitat type. Error bars represent standard errors of the predicted values.

The smoothed effect of sea bed slope showed that increased abundances of blue moki were generally associated with steeper slopes (Fig. 15A). However, this trend became less certain beyond a slope of 10 degrees. The low sample size beyond 10 degrees may have caused this negative 'dip' in the trend. The smoothed effect of depth showed that moki are more likely to inhabit areas that are deeper than 15 m, with depths of <10 m having a strong negative effect on their abundance (Fig. 15B).

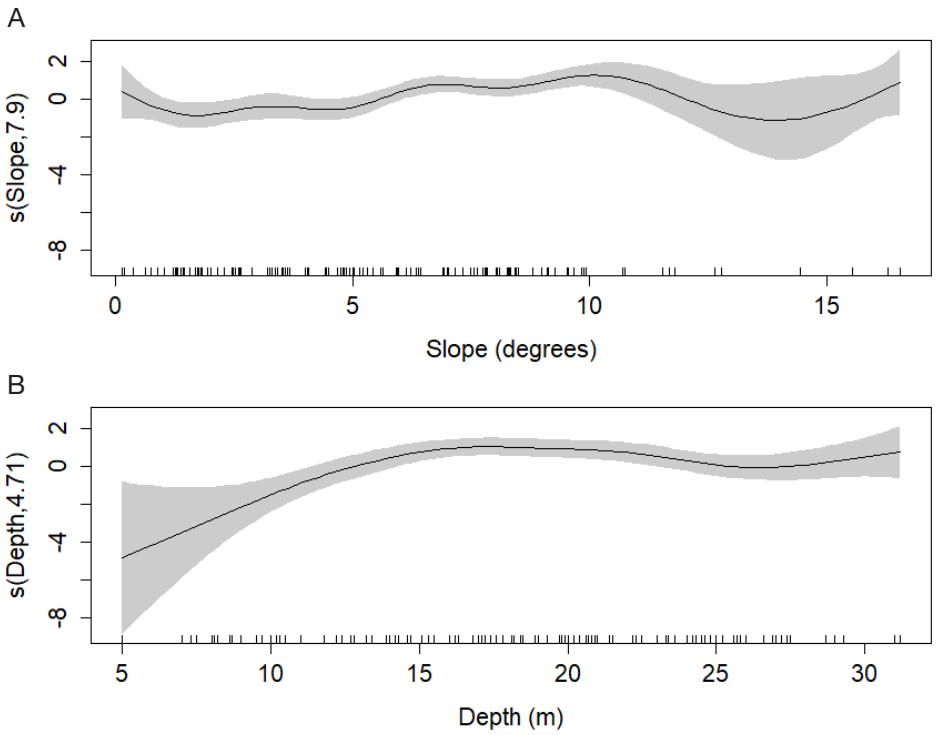

Figure 15. Smoothed term effects from generalised additive models for the relationships between blue moki (*Latridopsis ciliaris*) relative abundance and A. seabed slope and B. depth. The shaded region on each chart shows the 95% confidence interval for the given smoothed effect.

4. Discussion

4.1 Species inventory

A larger number of fish and macro-invertebrate species were observed in this study than have previously been observed in this region using UVC surveys. For example, Davidson et al. (2001) recorded only ten fish species in Pōhatu Marine Reserve compared with the 28 (including elasmobranchs) that were observed across the study region in the present study. The majority of these species were rarely observed on the video footage, with only 14 being encountered in more than ten deployments (Table 2). However, several species that were seen in this study were notably absent from previous work, including tarakihi, sea perch, trumpeter and red cod, all of which have high recreational, commercial and cultural value. Unfortunately, none of these additional species except tarakihi were seen in sufficient numbers to allow detailed analysis. However, the acquisition of data on their presence/absence and abundance is valuable, as further studies in this region may provide sufficient statistical power to detect trends if their abundances increase. The low numbers that were observed here likely indicate the rarity of these species in this area at present (Källqvist et al. 2015).

The difference in the number of species observed in this area using BUV and UVC methods is likely due to the more extensive study area and sampling that was conducted in the present study. The use of BUV allowed deeper, more exposed habitats to be sampled that would have been inaccessible during dive surveys (Davidson et al. 2001; Davidson & Abel 2003). Since fish abundance and richness appear to be associated with exposure, the ability to sample such habitat is a substantial advantage of the BUV method. Moreover, the response of some species to the two methodologies likely explains the higher number of large, more mobile fish species that were recorded with BUV (Colton & Swearer 2010).

The fact that BUV sampling obtained sufficient data on several key species for a formal analysis confirms the usefulness of this technique for biological monitoring in the Banks Peninsula region. Blue cod is the most popular species targeted by recreational fishers in the South Island and also supports an important coastal commercial fishery (Davey et al. 2008), but there is evidence of a decline in several blue cod stocks (Beentjes & Carbines 2005). Therefore, knowledge of how the species responds to protected areas is of major importance. Although previous studies have shown that blue cod responds well to BUV surveys in the North Island (Willis & Babcock 2000; Willis et al. 2000), few studies have been conducted in the South Island. Blue moki is also a popular species among recreational fishers and has a modest amount of commercial catch (Manning et al. 2010; Morrison et al. 2014). The diet of blue moki is generally thought to consist of crustaceans, shellfish and worms (Francis 2001), but these fish can also be caught with a baited hook, suggesting that they may be opportunistic predators/scavengers. This is substantiated by the prevalence of blue moki responding to the bait plume in this study (they were seen in 42% of deployments). This is the first study to show that blue moki respond well to BUV survey methods. Tarakihi is another species that is regularly caught recreationally using hook and line, indicating that BUV holds promise for monitoring this species in coastal settings. However, tarakihi have also not traditionally been surveyed using BUV methods. Here, we found that tarakihi were relatively rare in the study area, which probably reflects the high fishing pressure in the area or a more offshore distribution on the shallow shelf off Banks Peninsula.

4.2 Relative abundance and size of blue cod

Blue cod have been shown to respond well to marine reserve protection in several locations in the North Island (Willis & Babcock 2000; Willis et al. 2000) and in the Marlborough Sounds (Davidson 2001). For example, Willis & Babcock (2000) found that blue cod at Hahei, on the Coromandel Peninsula of the North Island, were 4.2 times more abundant at marine reserve sites than at control sites. The lack of statistically significant differences in blue cod relative abundance between the marine reserve and control survey areas in the present study appears to have been due to the high abundance of blue cod at OTCTRL. Otanerito has a greater area of light foul habitat (e.g. cobble and coarse sediment) than the other control sites, and this has been shown to be associated with high blue cod abundances (Fig. 12; Cole et al. 2012). Further, OTCNTL was the most remote survey area, which may result in there being less fishing pressure at this area.

Pōhatu Marine Reserve is very small, so the impacts of edge effects and the low availability of habitat may be limiting the effectiveness of the reserve for increasing blue cod abundance (Roberts et al. 2001; Halpern & Warner 2003). Large blue cod are known to be particularly territorial (Willis et al. 2000) and the size structure of the population at POHMR was clearly different from that in the control areas (see below). Therefore, if large fish are forcing juvenile recruits out of Pōhatu Marine Reserve, the abundance of fish may be limited. Also, fisher non-compliance will influence blue cod abundance within the reserve to some degree – seven boats were caught poaching in the reserves at Banks Peninsula over 2 months in summer 2017/18 (T. MacTavish, pers. obs.).

Blue cod potting surveys were conducted on strata within and around Pōhatu Marine Reserve by Carbines (2014) in 2008 and 2012, and by Beentjes et al. (2017) in 2017. These surveys had variable results due to variation in water clarity, which seems to have a strong effect on blue cod catchability, as well as variation in the sampling design used (fixed v. random site selection). The random design resulted in a large number of pots being placed in non-preferred habitat (i.e. soft sediment), resulting in lower estimates of relative abundance. The results from the 2012 survey provide the best comparison to the results of the present study and showed that blue cod relative abundance was significantly higher (approximately twice as high) at marine reserve sites compared with two adjacent control areas. By contrast, the surveys that were conducted in 2008 and 2017 did not detect any difference between these areas.

Some aspects of the BUV methodology may also have limited our ability to detect differences between survey areas when fish density was high (e.g. POHMR v. OTCNTRL). Willis et al. (2000) found that the observed differences in relative abundance between fished and unfished areas were smaller (and not statistically significant) using BUV methods rather than UVC methods. Clearly, there is an upper limit to the number of fish that will be visible within the camera's field of view at any one time (Willis et al. 2000; Cappo et al. 2006). In POHMR, the Max*N* values often exceeded 20 fish, which is higher than any previously reported value in the literature. However, it remains unknown whether this is close to an upper limit. Consequently, any comparison between reserve and control areas will probably be conservative, as the abundance will tend to be underestimated in areas where fish abundance is high. The aggressive tendencies of large blue cod may also cause relative abundance estimates to be lower when the number of large fish at the bait is high (Coghlan et al. 2017). Blue cod in the Marlborough Sounds are bolder and more aggressive at marine reserves than in control areas (Davidson 2001), so it is also feasible that aggressive behaviour may have influenced the results of our study.

Our findings suggest that site protection may have a significant positive effect on the size of blue cod, as fish were larger at POHMR and, to a lesser extent, AKAMR. Furthermore, the relative abundance of legal-sized blue cod was significantly higher at POHMR. Similarly, in a meta-analysis of blue cod size data from five marine reserves, Pande et al. (2008) found that blue cod were typically larger in protected areas. We found that blue cod at POHMR were, on average,

29 mm, 38 mm and 34 mm larger than those at OTCTRL, DACTRL and OHCTRL, respectively. However, these differences are small compared with those observed by Pande et al. (2008), especially given the age (18 years) of Pōhatu Marine Reserve at the time of sampling. Davidson & Abel (2003) found that the average size of blue cod at Pōhatu Marine Reserve was 273 mm, compared with 214 mm across pooled control areas, which is surprisingly similar to the results of our study. This suggests that either there has been little change in the average size of blue cod throughout the study area between 2002 and 2017, or there are differences in the way in which the UVC and BUV methods sample size structure.

The potting results obtained during the 2012 survey by Carbines (2014) were very similar to those of this study in terms of the size structure of the blue cod populations inside and outside Pōhatu Marine Reserve. The substantial decrease in the number of fish of >300 mm that was observed at all control sites but was not seen at POHMR or, to a lesser extent, at AKAMR, was particularly notable (Fig. 7). Similarly, Carbines (2014) found that the mean size of blue cod was 313 mm inside Pōhatu Marine Reserve and 265 mm outside it. BUV has been shown to be a more accurate method than UVC for measuring the size distribution of many marine fishes (Harvey et al. 2004; Cappo et al. 2006), including blue cod (Willis et al. 2000). However, although we removed any fish that were obviously vertically or strongly horizontally displaced from the scale bar from the length analysis, it is likely that some fish were retained that were affected by error associated with vertical displacement and lens distortion. This may explain the greater range in length measurements that was observed in this study compared with the potting results of Carbines (2014). Nonetheless, many of the sub-150-mm fish we observed may not have been caught by Carbines (2014), despite the fine mesh size that was used.

The significant difference in the relative abundance of legal-sized blue cod between the reserve and control areas in our study was also observed by Carbines (2014). In the 2012 potting survey, the relative abundance of fish >300 mm was 2.97 kg (per pot per hour) in Pōhatu Marine Reserve and 0.45 kg in control areas, suggesting that the biomass of legal-sized cod was 6.6 times greater inside the reserve than at control sites. We found that the mean Max*N* for legal-sized blue cod was 2 at POHMR and 0.56 across the three control areas, indicating a 3.6 times greater abundance in the reserve. Similarly, the abundance of blue cod was 2.1 times higher at AKAMR than in the pooled control areas (note: Carbines (2014) did not survey Akaroa Marine Reserve). Together, the findings of these two studies suggest that the protection afforded by marine reserves may have a positive effect on the size structure of blue cod at Banks Peninsula.

4.3 Relative abundance and size of blue moki

On average, blue moki were three times more abundant at POHMR and four times more abundant at AKAMR than in the three control areas. Significant variability was associated with the mean Max*N* for blue moki, particularly at AKAMR. This would be expected given that a small number of deployments had very high blue moki counts (Max*N* up to 13) while the majority of deployments did not record any blue moki. However, the results were still statistically significant, providing the first evidence that marine reserve protection may increase blue moki abundance. In a previous study at Kapiti Marine Reserve, Pande & Gardner (2012) did not detect any difference in the abundance of blue moki between reserve and control sites and, similarly, Davidson & Abel (2003) did not report any change in blue moki abundance during the second sampling of Pōhatu Marine Reserve in 2002.

There was a pronounced difference in the relative abundance of legal-sized blue moki between the reserve and control areas, with POHMR and AKAMR having ten and eight times higher relative abundances of legal-sized fish, respectively, than the pooled control areas. Similarly, Pande & Gardner (2012) also found a higher abundance of legal-sized blue moki within Kapiti Marine Reserve compared with control sites.

Much important information on the biology of blue moki is lacking. There is evidence that adult fish undergo a northwards spawning migration from the upper South Island to areas along the east coast of the North Island south of East Cape (Francis 1981) and the majority of the commercial take for this species is also in this area (Manning et al. 2010). Fish from as far south as Kaikoura are known to take part in this spawning migration (Francis 1981), but it is currently unknown whether blue moki from Banks Peninsula also migrate north. If they do, the small marine reserves at Banks Peninsula are unlikely to influence the abundance or size of this species due to its extensive range. There is currently no evidence for site fidelity in blue moki at fine spatial scales, as has been observed in blue cod (Cole et al. 2000) and snapper (*Pagrus auratus*) (Willis et al. 2001). However, the findings of the present study and of Pande & Gardner (2012) suggest that there must be some site fidelity during a period of the species' life history.

4.4 Relative abundance and size of tarakihi

The observed patterns in tarakihi abundance among survey areas were inconclusive, largely due to the high variability in the Max*N* estimates, which was a product of the relative rarity of tarakihi throughout the study area. Despite this, significant differences in the relative abundance of tarakihi were observed, with the highest levels occurring in POHMR. However, the relative abundance at POHMR was not significantly different from that at OHCTRL. To our knowledge, this is the first study to have assessed the relative abundance of tarakihi relative to protection status. This species is widely distributed on the New Zealand continental shelf and is most common at depths of <100 m (Morrison et al. 2014), but it is unknown whether individuals show any site fidelity at fine spatial scales.

The majority (91%) of tarakihi observed in this study were below the minimum legal size. While the sample size was too low for a formal analysis, most legal-sized fish ($n = 7$) were observed in POHMR, where they were recorded at five deployments sites. Further sampling may help to determine whether reserve status is truly having an effect on the size structure of tarakihi.

The relatively high abundance of small tarakihi in some areas may be an indication of nursery habitat. Most measured fish were between 120 and 200 mm, which represents age 0+ and 1+ fish (Morrison et al. 2014). Although there are some well-known spawning grounds for tarakihi in New Zealand waters, nothing is known about their nursery habitats. In the Canterbury Bight, small tarakihi are most frequently caught in shallow habitat at depths of 30–100 m and a large spawning ground exists in Pegasus Bay to the north of Banks Peninsula (Morrison et al. 2014). Therefore, it is possible that some of the comparatively shallow coastal habitat that was surveyed in this study represents tarakihi nursey habitat. However, more sampling, particularly across multiple seasons, is required to confirm this.

4.5 Species richness

There was no significant difference in species richness among the survey areas. This was not entirely unexpected given the relative proximity of each area. Although changes in fish diversity have been attributed to protection status in some areas (Jennings et al. 1995), any such changes were unlikely to have been detected in the present study due to the small size of the marine reserves on Banks Peninsula and the fact that the largest reserve (Akaroa) was only 3 years old at the time of survey. There are also questions around the suitability of BUV methods for monitoring diversity, as BUV can only provide information on species that respond to the bait plume. Measuring changes in the community of carnivorous fishes holds some merit for investigating the impact of fishing on exploited populations as these species are generally the target of fishers and so changes in their numbers across protection boundaries identifies whether certain species groups are being extirpated at local scales. However, any assessment of biodiversity at the community level should be undertaken using a multivariate approach (e.g. Cappo et al. 2006; Pande & Gardner 2012) with data from alternative sampling methods (e.g. UVC).

Interestingly, we found that more exposed locations had higher species richness. This pattern could be explained by the greater variability in physical habitat and the oceanographic conditions around headlands and along exposed coastlines (e.g. stronger current). These sites were characterised by the presence of some larger, more mobile fish species (e.g. trumpeter and kahawai) that were not present at sheltered sites which is similar to the findings of Fulton & Bellwood (2004).

4.6 Habitat preferences

The data we collected using BUV sampling provided an opportunity to investigate links between habitat characteristics and the abundances of blue cod and blue moki. The models performed well, explaining a reasonable amount of the variation in relative abundance for both species. Furthermore, although these were intended to be reasonably basic models to illustrate the suitability of BUV-derived data for species distribution modelling, some of our results appeared to be novel.

While some general knowledge on the habitat preferences of blue cod exists (Morrison et al. 2014), few studies have tested these relationships empirically, particularly at fine spatial scales. Jones (1988) suggested that blue cod are more abundant in areas with turfing algal communities, and Carbines et al. (2004) found that cod are associated with biogenic reefs in Foveaux Straight. In one of the only studies that has determined the distribution of blue cod among habitat types, Cole et al. (2012) found that blue cod were associated with coarse sediment, including gravel, shell rubble and sand. Further, the abundance of blue cod was greater at depths of >8 m and was negatively associated with algal canopy cover. All of these results agree with those observed in the present study, where we found that blue cod were more abundant on coarse sediment and cobble substrates with little algal cover at depths of >12 m and within 5 m of the reef edge.

Limited information is available on the distribution of blue moki relative to habitat characteristics and, to the best of our knowledge, this was the first study to test these relationships empirically. Morrison et al. (2014) stated that 'Adult moki occur over sand and mud bottoms, as well as reefs'. Our results suggest that blue moki are abundant in habitats with algal canopies, a pattern that is well known to fishers who target the species in coastal habitat.

Using BUV data on demersal fish assemblages, Moore et al. (2010) showed that habitat characteristics – depth, boulder reef and relief – explained 37.4% of the variation in fish distribution, and similar results were observed by Moore et al. (2011). Zintzen et al. (2012) related differences in the diversity and community composition of fishes to depth along a gradient on the New Zealand continental shelf. The results from these studies and the present study suggest that BUV methods hold much promise for unravelling species-habitat relationships. In turn, this information will help to inform the future creation of MPAs so that they better recognise the patchy distribution of the coastal fish species they are designed to protect.

4.7 Appraisal of the BUV method

Analysis of the count data over 1-minute intervals within a deployment indicated that 30-minute deployment times were sufficient to reach stable Max*N* values for both blue cod and blue moki. Willis & Babcock (2000) also demonstrated that 100% of the Max*N* for blue cod was obtained within 20 minutes of a deployment. No other study has assessed such trends for blue moki. Deployment times of 30 minutes are commonly used in a wide range of habitats (Cappo et al. 2003) and our results suggest that these are also appropriate at Banks Peninsula. However, analyses of longer deployments (e.g. 60 minutes) are required to ensure no further increases in Max*N* are apparent after 30 minutes.

The minimum site separation distance of 150 m that was used in the present study is much smaller than has been used in most BUV studies (Cappo et al. 2003), with 450 m typically

being recommended to ensure independent samples (Cappo et al. 2004). The independence of deployment sites is generally thought to be a product of bait plume dispersal and the swimming speed of the fish reacting to the plume (Cappo et al. 2003). While this is undoubtedly true, factors relating to the sampled habitat may also be important. It is conceivable that in homogenous habitat, or mosaicked habitat that provides corridors, a fish may take a direct route to the bait. However, when habitat is fragmented by the contours of the coastline, as occurs at Banks Peninsula, fish may not swim directly towards the bait. Therefore, the strong species-habitat relationships we observed may explain the lack of spatial autocorrelation between the closely located sites used in this study. A lack of movement between habitat types was clearly evident in trial deployments, with no reef fish being present at deployments further than 30 m from the reef edge.

The 450-m site separation suggested by Cappo et al. (2004, 2006) was calculated for 60-minute deployment times and for inter-reef sediment-dominated (i.e. homogenous) habitat in Australia. The applicability of this value for sampling coastal reef-associated species in New Zealand is unknown and previous coastal BUV surveys in New Zealand have not reported the distances between deployment sites (Willis & Babcock 2000; Willis et al. 2000; Denny & Babcock 2004). Our statistical analysis of spatial autocorrelation between pairs of deployment sites showed no evidence of correlation at 150 m. This indicates that 150 m is an appropriate separation distance for BUV monitoring at Banks Peninsula. Such spatial autocorrelation should, however, be examined after further surveys using more randomly selected sites to determine a cut-off point for the minimum deployment separation that is required in this habitat. Also, analyses to determine the bait plume response radius would provide more certain evidence as to an appropriate minimum site separation distance.

Our results suggest that BUV survey techniques can provide useful information on the relative abundance, population size distribution, species richness and habitat preferences of several key fish species at Banks Peninsula. However, it is worthwhile considering the following limitations of the method for MPA monitoring:

1. An upper limit to the Max*N* values for some species may cause differences in relative abundance between reserve and control areas to be underestimated when fish density is high.
2. Intra- and interspecific aggressive behaviours may influence estimates of relative abundance for some species. The prevalence of such behaviours may differ between reserve and control areas.
3. Size estimation based on the current DOC BUV system carries some measurement error, particularly when a BUV unit is deployed on highly uneven substrate (e.g. jagged reef) or where the field of view incorporates habitat below the scale bar. Issues associated with lens distortion may also cause measurement error at the edges of the image.
4. BUV tends to attract mainly scavenger and carnivorous species, meaning that biodiversity and community analyses are not possible with this method.
5. Similar to other sampling methods (e.g. UVC), BUV only provides a relative estimate of fish density, not an absolute estimate, because detailed information on the range and effectiveness of bait attraction is not readily available to the observer. If estimates of population abundance are required, alternative sampling methods should be considered (e.g. mark-recapture approaches).

Despite these limitations, the key advantages of BUV over UVC – logistical simplicity, the ability to sample more sites and a greater range of habitat, and less observer bias in abundance and size estimation – strongly favour its adoption for routine marine reserve monitoring at Banks Peninsula.

4.8 Caveats

When interpreting the results of this study, it is important to consider that they are the product of a single survey. Ecology is founded on the principle that ecosystems are complex and dynamic, and that the abundance, size and distribution of species will vary greatly through space and time. Thus, for example, it would be premature to definitively link a higher abundance of blue cod with the effect of marine reserve protection at Akaroa Marine Reserve, as no baseline data are available. Indeed, it is possible that the differences observed here were already present before protection. Therefore, in the absence of a traditional before-after-control-impact experiment (BACI), a consistent, ongoing monitoring regime will now be needed to establish the true effect of Akaroa Marine Reserve.

For Pōhatu Marine Reserve, baseline surveys were carried out but these used the UVC method, which means that comparisons with this study are not straightforward. While the trends that were observed in the respective surveys can be compared, the data cannot be pooled into a single dataset for a BACI analysis. Therefore, although the range of historical data collected in Pōhatu Marine Reserve (using UVC and blue cod potting) provides some confidence that the trends observed in this study are true effects of protection, repeated monitoring using the same method should be conducted before any definitive inference is drawn.

Two caveats of the species–habitat models deserve attention. Firstly, although point samples of habitat are common in BUV studies (Moore et al. 2010, 2011; Zintzen et al. 2012), this creates a mismatch in the spatial scale of sampling fish abundance (100s of metres) and habitat characteristics (c. 1 m^2). In this study, it was not possible to determine the effective bait plume radius, which would allow a calculation and subsequent sampling of habitat at the same scale as fish Max*N*. It was assumed that using a point sample for habitat characteristics would introduce less bias than selecting an arbitrary detection radius that may sample habitat that is meaningless to the reef-associated fishes included in this study. Due to the predominance of soft-sediment habitat in the study area and the perceived disinclination of reef fishes to move onto sediment in response to the bait plume (e.g. Fig. 13C), this assumption appears valid. Secondly, there may be some confounding of the species distribution models due to data being used across survey areas with different protection statuses. Species-habitat relationships will exist irrespective of fishing pressure unless species have been extirpated locally, which did not occur in this study. However, it is likely that the distribution models would be significantly improved by including data on protection status or direct fishing pressure.

5. Conclusions and future research

The establishment of MPAs, which include marine reserves, is now widely recognised as a tool that helps in the protection and recovery of marine ecosystems. The monitoring of key indicator species is an important means of assessing whether a specific MPA is meeting its objectives and can support the management of adjacent fisheries. However, the collection of baseline data and establishment of ongoing monitoring of key indicator species has proven challenging in Banks Peninsula's two marine reserves – Akaroa and Pōhatu – due to the poor water clarity and exposed coastline.

In this study, BUV was successfully trialled at Banks Peninsula as an alternative method to UVC techniques. The BUV method proved to be relatively easy to apply in the field and provided an initial snapshot of fish assemblages both inside and outside Akaroa and Pōhatu Marine Reserves. A total of 28 fish species were recorded, which is high in comparison to previous studies and suggests that the BUV method is capable of monitoring a range of Banks Peninsula's fish species. Of particular note was the detection of larger, more mobile species, including seven-gill shark, school shark and trumpeter. Furthermore, our findings suggest that data on fish abundance from BUV can provide novel insights into the fine-scale habitat preferences of some fish species, which may help to inform and refine the creation of future MPAs in the South Island.

Our study also detected significant spatial differences in the relative abundances and sizes of blue cod and blue moki, both of which are highly sought after by fishers. The relative abundance of blue cod was significantly greater in both marine reserves compared with OHCTRL and DACTRL, but the reserves had similar abundances to OTCTRL. In terms of fish size, blue cod were significantly larger at POHMR than in all three control areas and were larger at AKAMR than in two of the control areas. Given the absence of baseline data for Akaroa Marine Reserve, it would be premature to attribute such spatial differences to protection status. However, our findings support those of previous studies in suggesting that the abundance and size of blue cod have increased significantly within Pōhatu Marine Reserve, presumably due to protection from direct fishing pressure.

Similar spatial trends were observed for blue moki, which were also significantly larger and more abundant in the two marine reserves compared with the control areas. In the absence of a baseline or repeat monitoring data, these findings are perhaps most notable in demonstrating that BUV may represent a suitable method for studying the size and relative abundance of this species. Nevertheless, our study presents some of the first evidence that marine reserves may protect blue moki, although repeat monitoring and further studies are required.

Future research could reduce some of the limitations and test assumptions that were identified in our study and could include:

1. Using cheap stereo-camera systems (e.g. stereo GoPro systems) to more accurately measure fish size and, possibly, condition.
2. Modelling the dispersal area and effect of the bait plume on the Max*N* for particular species.
3. Investigating the upper limit of Max*N* values for blue cod.

Future research could also improve the simplistic species distribution models that were developed in this study by investigating the relationships between a broad range of habitat parameters, protection statuses, indices for fishing pressure and species abundances/diversites. Such models could provide novel information on the mechanisms that shape the fish assemblages of Banks Peninsula and, in turn, inform future management decisions in this area of high cultural, recreational and commercial value.

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Appendix 1

Effect of survey area on relative abundance

Results of Tukey's post-hoc tests for the generalised linear models (GLMs with Poisson error distributions) testing the effects of Survey Area (fixed effect, five levels) on the relative abundances of blue cod (*Parapercis colias*), blue moki (*Latridopsis ciliaris*) and tarakihi (*Nemadactylus macropterus*). Relative abundance is the Max*N* derived from baited underwater video deployments. The nature and magnitude of the difference between survey areas is given by the estimate and its statistical significance is reflected by its *P* value (* $P < 0.05$). Survey areas: Akaroa Marine Reserve (AKAMR), Pōhatu Marine Reserve (POHMR), Damon's Bay Taiāpure control (DACNTRL), outer Akaroa Harbour Taiāpure control (OHCNTRL) and Otanerito Bay control (OTCNTRL).

Blue cod abundance	**Estimate**	**Std. error**	***z* value**	***P* value**
AKAMR – OTCNTRL == 0	–0.07	0.11	–0.61	0.970
DACNTRL – OTCNTRL == 0	–0.49	0.12	–4.21	<0.001*
OHCNTRL – OTCNTRL == 0	–0.58	0.12	–4.89	<0.001*
POHMR – OTCNTRL == 0	0.18	0.10	1.82	0.360
DACNTRL – AKAMR == 0	–0.43	0.12	–3.49	0.004*
OHCNTRL – AKAMR == 0	–0.52	0.12	–4.15	<0.001*
POHMR – AKAMR == 0	0.24	0.10	2.34	0.130
OHCNTRL – DACNTRL == 0	–0.09	0.13	–0.67	0.960
POHMR – DACNTRL == 0	0.67	0.11	5.96	<0.001*
POHMR – OHCNTRL == 0	0.76	0.11	6.64	<0.001*

Blue moki abundance	**Estimate**	**Std. error**	***z* value**	***P* value**
AKAMR – OTCNTRL == 0	1.08	0.32	3.41	0.010*
DACNTRL – OTCNTRL == 0	–0.69	0.46	–1.50	0.550
OHCNTRL – OTCNTRL == 0	–0.37	0.41	-0.90	0.890
POHMR – OTCNTRL == 0	0.87	0.32	2.77	0.040*
DACNTRL – AKAMR == 0	–1.77	0.41	–4.27	<0.001*
OHCNTRL – AKAMR == 0	–1.45	0.36	–4.04	<0.001*
POHMR – AKAMR == 0	–0.20	0.24	–0.86	0.910
OHCNTRL – DACNTRL == 0	0.32	0.49	0.65	0.960
POHMR – DACNTRL == 0	1.57	0.41	3.79	<0.001*
POHMR – OHCNTRL == 0	1.25	0.36	3.49	0.000*

Tarakihi abundance	Estimate	Std. error	*z* value	*P* value
AKAMR – OTCNTRL == 0	0.53	0.43	1.22	0.730
DACNTRL – OTCNTRL == 0	–0.12	0.49	–0.24	1.000
OHCNTRL – OTCNTRL == 0	1.23	0.38	3.27	0.010*
POHMR – OTCNTRL == 0	1.39	0.37	3.77	0.001*
DACNTRL – AKAMR == 0	–0.65	0.45	–1.44	0.590
OHCNTRL – AKAMR == 0	0.70	0.33	2.14	0.190
POHMR – AKAMR == 0	0.87	0.32	2.71	0.050
OHCNTRL – DACNTRL == 0	1.35	0.40	3.42	0.010*
POHMR – DACNTRL == 0	1.51	0.39	3.90	<0.001*
POHMR – OHCNTRL == 0	0.16	0.24	0.68	0.960

Scarlett Wrasse abundance	Estimate	Std. error	*z* value	*P* value
AKAMR – OTCNTRL == 0	–0.06	0.18	–0.30	1.00
DACNTRL – OTCNTRL == 0	0.20	0.16	1.23	0.74
OHCNTRL – OTCNTRL == 0	–0.22	0.18	–1.19	0.76
POHMR – OTCNTRL == 0	–0.03	0.17	–0.16	1.00
DACNTRL – AKAMR == 0	0.26	0.18	1.47	0.58
OHCNTRL – AKAMR == 0	–0.16	0.19	–0.84	0.92
POHMR – AKAMR == 0	0.03	0.18	0.15	1.00
OHCNTRL – DACNTRL == 0	–0.42	0.17	–2.41	0.11
POHMR – DACNTRL == 0	–0.23	0.16	–1.41	0.62
POHMR – OHCNTRL == 0	0.19	0.18	1.05	0.83

Appendix 2

Effect of exposure stratum on relative abundance

Results of Tukey's post-hoc tests for the generalised linear models (GLMs with Poisson error distributions) testing the effects of Exposure Stratum (fixed effect, ten levels) on the relative abundances of blue cod (*Parapercis colias*) and blue moki (*Latridopsis ciliaris*). Relative abundance is the Max*N* derived from baited underwater video deployments. The nature and magnitude of the difference between exposure strata is given by the estimate and its statistical significance is reflected in its *P* value (* $P < 0.05$). Survey areas: Akaroa Marine Reserve (AKAMR), Pōhatu Marine Reserve (POHMR), Damon's Bay Taiāpure control (DACNTRL), outer Akaroa Harbour Taiāpure control (OHCNTRL) and Otanerito Bay control (OTCNTRL). Survey strata: exposed (exp) and intermediate (int).

Blue cod abundance	Estimate	Std. error	*z* value	*P* value
OHCNTRLexp - OHCNTRLint == 0	–0.18	0.19	–0.95	0.990
AKMRexp - AKMRint == 0	0.04	0.16	0.27	1.000
POHMRexp - POHMRint == 0	0.62	0.14	4.50	<0.001*
DACNTRLexp - DACNTRLint == 0	–0.82	0.19	–4.36	<0.001*
OTCNTRLexp - OTCNTRLint == 0	0.42	0.15	2.85	0.060
POHMRint - OHCNTRLint == 0	0.29	0.17	1.69	0.660
POHMRint - DACNTRLint == 0	–0.13	0.17	–0.77	1.000
POHMRint - OTCNTRLint == 0	0.03	0.16	0.16	1.000
POHMRexp - OHCNTRLexp == 0	1.09	0.16	6.62	<0.001*
POHMRexp - DACNTRLexp == 0	1.32	0.17	7.95	<0.001*
POHMRexp - OTCNTRLexp == 0	0.23	0.12	1.90	0.500
AKMRint - OHCNTRLint == 0	0.41	0.17	2.41	0.190
AKMRint - DACNTRLint == 0	0.00	0.17	0.00	1.000
AKMRint - OTCNTRLint == 0	0.15	0.17	0.92	0.990
AKMRexp - OHCNTRLexp == 0	0.64	0.18	3.48	<0.001*
AKMRexp - DACNTRLexp == 0	0.87	0.18	4.70	<0.001*
AKMRexp - OTCNTRLexp == 0	–0.23	0.14	–1.58	0.740

Blue moki abundance	Estimate	Std. error	*z* value	*P* value
OHCNTRLexp - OHCNTRLint == 0	1.53	0.79	1.94	0.468
AKMRexp - AKMRint == 0	1.49	0.45	3.32	0.013*
POHMRexp - POHMRint == 0	1.62	0.48	3.36	0.012*
DACNTRLexp - DACNTRLint == 0	–0.66	0.76	–0.87	0.991
OTCNTRLexp - OTCNTRLint == 0	0.51	0.56	0.92	0.987
POHMRint - OHCNTRLint == 0	1.06	0.84	1.27	0.906
POHMRint - DACNTRLint == 0	0.06	0.67	0.08	1.000
POHMRint - OTCNTRLint == 0	0.00	0.63	0.00	1.000
POHMRexp - OHCNTRLexp == 0	1.15	0.40	2.89	0.051
POHMRexp - DACNTRLexp == 0	2.34	0.60	3.86	0.002*
POHMRexp - OTCNTRLexp == 0	1.10	0.38	2.91	0.048*
AKMRint - OHCNTRLint == 0	1.41	0.82	1.73	0.625
AKMRint - DACNTRLint == 0	0.41	0.65	0.63	0.999
AKMRint - OTCNTRLint == 0	0.35	0.61	0.58	1.000
AKMRexp - OHCNTRLexp == 0	1.37	0.40	3.43	0.009*
AKMRexp - DACNTRLexp == 0	2.56	0.61	4.22	<0.001*
AKMRexp - OTCNTRLexp == 0	1.32	0.38	3.47	0.008*

Appendix 3

Differences in the absolute size of blue cod (*Parapercis colias*) among survey areas

Results of Tukey's post-hoc tests for the generalised linear models (GLMs with Gaussian error distributions) testing the effects of Survey Area (fixed effect, five levels) on the size of blue cod (*Parapercis colias*). Size data were obtained for blue cod measured at the Max*N* of each deployment. The nature and magnitude of the difference between survey areas is given by the estimate and its statistical significance is reflected in its *P* value (* $P < 0.05$). Survey areas: Akaroa Marine Reserve (AKAMR), Pōhatu Marine Reserve (POHMR), Damon's Bay Taiāpure control (DACNTRL), outer Akaroa Harbour Taiāpure control (OHCNTRL) and Otanerito Bay control (OTCNTRL).

Difference in blue cod size	Estimate	Std. error	*z* value	*P* value
DACNTRL - AKAMR == 0	–0.03	0.01	–3.65	<0.001*
OHCNTRL - AKAMR == 0	–0.05	0.01	–5.80	<0.001*
OTCNTRL - AKAMR == 0	–0.01	0.01	–1.28	0.700
POHMR - AKAMR == 0	0.11	0.01	14.76	<0.001*
OHCNTRL - DACNTRL == 0	–0.02	0.01	–1.89	0.320
OTCNTRL - DACNTRL == 0	0.02	0.01	2.60	0.070
POHMR - DACNTRL == 0	0.14	0.01	17.25	<0.001*
OTCNTRL - OHCNTRL == 0	0.04	0.01	4.80	<0.001*
POHMR - OHCNTRL == 0	0.16	0.01	20.15	<0.001*
POHMR - OTCNTRL == 0	0.12	0.01	16.97	<0.001*

Appendix 4

Effect of survey area on the relative abundance of legal-sized fish

Results of Tukey's post-hoc tests for the generalised linear models (GLMs with Poisson error distributions) testing the effects of Survey Area (fixed effect, five levels) on the relative abundances of legal-sized blue cod (*Parapercis colias*) and blue moki (*Latridopsis ciliaris*). Relative abundance is the Max*N* derived from baited underwater video deployments for fish above the minimum legal size limit. The nature and magnitude of the difference between survey areas is given by the estimate and its statistical significance is reflected in its *P* value (* $P < 0.05$). Survey areas: Akaroa Marine Reserve (AKAMR), Pōhatu Marine Reserve (POHMR), Damon's Bay Taiāpure control (DACNTRL), outer Akaroa Harbour Taiāpure control (OHCNTRL) and Otanerito Bay control (OTCNTRL).

Legal-size blue cod	Estimate	Std. error	*z* value	*P* value
DACNTRL – AKAMR == 0	–0.89	0.34	–2.64	0.040*
OHCNTRL – AKAMR == 0	–1.01	0.35	–2.90	0.020*
OTCNTRL – AKAMR == 0	–0.41	0.29	–1.42	0.610
POHMR – AKAMR == 0	0.55	0.23	2.37	0.120
OHCNTRL – DACNTRL == 0	–0.12	0.40	–0.29	1.000
OTCNTRL – DACNTRL == 0	0.48	0.35	1.36	0.640
POHMR – DACNTRL == 0	1.44	0.31	4.70	<0.001*
OTCNTRL – OHCNTRL == 0	0.60	0.36	1.65	0.460
POHMR – OHCNTRL == 0	1.56	0.32	4.92	<0.001*
POHMR – OTCNTRL == 0	0.96	0.25	3.78	<0.001*

Legal-size blue moki	Estimate	Std. error	*z* value	*P* value
DACNTRL – AKAMR == 0	–2.93	1.03	–2.85	0.030*
OHCNTRL – AKAMR == 0	–2.97	1.03	–2.88	0.030*
OTCNTRL – AKAMR == 0	–1.32	0.51	–2.58	0.060*
POHMR – AKAMR == 0	0.21	0.32	0.67	0.960
OHCNTRL – DACNTRL == 0	–0.04	1.41	–0.03	1.000
OTCNTRL – DACNTRL == 0	1.61	1.10	1.47	0.540
POHMR – DACNTRL == 0	3.15	1.02	3.09	0.010*
OTCNTRL – OHCNTRL == 0	1.65	1.10	1.50	0.520
POHMR – OHCNTRL == 0	3.18	1.02	3.12	0.010*
POHMR – OTCNTRL == 0	1.54	0.49	3.14	0.010*

Appendix 5

Analysis of fish species diversity among exposure strata using three diversity indices

Results of Tukey's post-hoc tests for the generalised linear models (GLMs with Gaussian error distributions) testing the effects of Exposure Stratum (fixed effect, ten levels) on species richness. The nature and magnitude of the difference between exposure strata is given by the estimate and its statistical significance is reflected in its *P* value (* $P < 0.05$). Survey areas: Akaroa Marine Reserve (AKAMR), Pōhatu Marine Reserve (POHMR), Damon's Bay Taiāpure control (DACNTRL), outer Akaroa Harbour Taiāpure control (OHCNTRL) and Otanerito Bay control (OTCNTRL). Exposure strata: exposed (exp) and intermediate (int).

Species richness	Estimate	Std. Error	*z* value	*P* value
OHCNTRLexp – OHCNTRLint == 0	0.54	0.18	2.99	0.04*
AKMRexp – AKMRint == 0	0.19	0.19	1.02	0.98
POHMRexp – POHMRint == 0	0.59	0.18	3.29	0.02*
DACNTRLexp – DACNTRLint == 0	0.23	0.19	1.21	0.93
OTCNTRLexp – OTCNTRLint == 0	0.64	0.19	3.41	0.01*
POHMRint – OHCNTRLint == 0	0.02	0.20	0.09	1.00
POHMRint – DACNTRLint == 0	–0.10	0.21	–0.46	1.00
POHMRint – OTCNTRLint == 0	0.07	0.21	0.32	1.00
POHMRexp – OHCNTRLexp == 0	0.07	0.15	0.44	1.00
POHMRexp – DACNTRLexp == 0	0.26	0.15	1.72	0.64
POHMRexp – OTCNTRLexp == 0	0.02	0.15	0.12	1.00
AKMRint – OHCNTRLint == 0	0.29	0.20	1.46	0.82
AKMRint – DACNTRLint == 0	0.17	0.21	0.83	0.99
AKMRint – OTCNTRLint == 0	0.34	0.21	1.63	0.71
AKMRexp – OHCNTRLexp == 0	–0.06	0.17	–0.36	1.00
AKMRexp – DACNTRLexp == 0	0.13	0.17	0.80	1.00
AKMRexp – OTCNTRLexp == 0	–0.11	0.16	–0.68	1.00

Appendix 6

Model selection tables for modelling species–habitat relationships

Model selection tables for the top generalised additive models used to assess the relationships between habitat variables and the relative abundances of blue cod (*Parapercis colias*) and blue moki (*Latridopsis ciliaris*). The top model for each species had the lowest Akaike Information Criterion (AIC) value and highest model weight. Parameters include biological habitat type (Bio_hab), physical habitat type (Phy_hab), reef width, depth, distance to reef (ReefD) and slope. 's' indicates that a variable was fitted using a cubic regression spline.

Formula ~ blue cod	df	logLik	AIC	Delta	Weight
Bio_hab + Phy_hab + Reef.width + s(Depth) + s(ReefD) + s(Slope)	23	–398.47	854	0.0	0.45
Bio_hab + Phy_hab + s(Depth) + s(ReefD) + s(Slope)	22	–400.09	855	0.8	0.31
Bio_hab + Phy_hab + Reef.width + s(Depth) + s(ReefD)	14	–412.26	858	4.1	0.06
Phy_hab + Reef.width + s(Depth) + s(ReefD) + s(Slope)	19	–406.03	858	4.4	0.05
Bio_hab + Phy_hab + Reef.width + s(Depth) + s(Slope)	22	–402.03	859	5.1	0.04
Bio_hab + Phy_hab + Reef.width + s(ReefD) + s(Slope)	16	–410.13	859	5.5	0.03

Formula ~ blue moki	df	logLik	AIC	Delta	Weight
Bio_hab + Phy_hab + Reef.width + s(Depth) + s(ReefD) + s(Slope)	18	–132.28	308	0.0	0.48
Bio_hab + Phy_hab + s(Depth) + s(ReefD) + s(Slope)	19	–131.66	309	0.7	0.35
Bio_hab + Phy_hab + Reef.width + s(Depth) + s(ReefD)	22	–129.19	312	4.2	0.06
Phy_hab + Reef.width + s(Depth) + s(ReefD) + s(Slope)	14	–140.42	314	5.5	0.03
Bio_hab + Phy_hab + Reef.width + s(Depth) + s(Slope)	23	–128.17	314	5.6	0.03
Bio_hab + Phy_hab + Reef.width + s(ReefD) + s(Slope)	15	–139.18	314	5.7	0.03

Appendix 7

Model results for the best models used to model species–habitat relationships

Statistical significance of the parameters that were retained in the top models explaining the relationships between habitat covariates and the abundances of blue cod (*Parapercis colias*) and blue moki (*Latridopsis ciliaris*). Parameters include biological habitat type (Bio_hab), physical habitat type (Phy_hab), reef width, depth, distance to reef (ReefD) and slope. 's' indicates that a variable was fitted using a cubic regression spline.

Parametric coefficients blue cod	**Estimate**	**Std. error**	***z* value**	***P* value**
(Intercept)	1.59	0.23	6.83	0.000
Bio_habAlgae_non canopy	0.04	0.20	0.19	0.853
Bio_habBare reef	0.60	0.22	2.73	0.006
Bio_habBare sediment	0.43	0.24	1.77	0.078
Bio_habBio reef	0.42	0.22	1.89	0.058
Phy_habBoulder reef	–0.33	0.10	–3.22	0.001
Phy_habCoarse sediment	1.07	0.18	5.96	0.000
Phy_habCobble	0.59	0.16	3.82	0.000
Phy_habSand	–0.26	0.17	–1.52	0.128
Reef.width	0.00	0.00	–1.92	0.055
Smooth terms blue cod	**edf**	**Ref.df**	**Chi.sq**	***P* value**
s(Slope)	7.71	8.57	21.28	0.009
S(Depth)	4.61	5.54	16.93	0.007
S(ReefDistance)	1.00	1.00	7.20	0.007

Parametric coefficients blue moki	**Estimate**	**Std. error**	***z* value**	***P* value**
(Intercept)	0.03	0.47	0.07	0.946
Bio_habAlgae_non canopy	–1.17	0.52	–2.27	0.023
Bio_habBare reef	–2.17	0.56	–3.88	0.000
Bio_habBare sediment	–2.07	0.66	–3.12	0.002
Bio_habBio reef	–1.61	0.54	–2.97	0.003
Reef.width	0.01	0.00	4.35	0.000
Smooth terms blue moki	**edf**	**Ref.df**	**Chi.sq**	***P* value**
s(Slope)	7.90	8.65	30.15	0.000
S(Depth)	4.71	5.72	26.37	0.000