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Assessing population recovery inside British Columbia's Rockfish Conservation Areas with a remotely operated vehicle

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ABSTRACT

Between 2004 and 2007, Fisheries and Oceans Canada undertook a management action to conserve overfished populations of Inshore Rockfishes by designating 164 Rockfish Conservation Areas (RCAs) closed to most recreational and commercial fishing. However, no research has yet assessed the effectiveness of the RCA network at promoting groundfish population recoveries. We surveyed the fish communities of 35 RCAs and adjacent unprotected areas in southern British Columbia using a remotely operated vehicle (ROV) between 2009 and 2011. We investigated the effect of protection and habitat on fish densities for six species or species groups (Quillback, Yelloweye, Greenstriped Rockfish, Kelp Greenling, Lingcod and all Inshore Rockfish combined) on transects inside and outside of RCAs. Habitat features such as percent rocky substrates and depth influenced fish density while reserve status did not. Next, we calculated habitat-based average densities and used the mean log response ratio (RR) of the density inside to outside of RCAs to determine if the amount of fishing outside the RCA, previous fishing history, the age, area or perimeter to area ratio influenced population recovery. Few positive reserve effects were apparent for any species/group. No clear patterns of RR with age were found for the RCAs, which ranged from 3 to 7 years old at the time of sampling (mean = 4.6). In addition, the intensity of fishing, size, and perimeter-to-area ratio failed to explain RR for most species. There were also no differences in size structure (length) of fish between RCAs and unprotected areas. The results give little indication that demersal fish populations have recovered inside the RCA system. Ongoing monitoring is essential to assess population recovery over time and evaluate the RCAs in terms of criteria such as habitat quality, habitat isolation and the level of compliance in order to enhance their effectiveness.

1. Introduction

Networks of Marine Protected Areas (MPAs), or reserves that exclude fisheries, are being implemented worldwide to conserve exploited species and sustain fisheries (Gaines et al., 2010a). The use of MPAs has been shown to be a successful strategy to increase the size, abundance and diversity of species protected within them (e.g., Allison et al., 1998; Alcala et al., 2005; Claudet et al., 2008; Edgar et al., 2014). Monitoring is critical to the implementation of MPA networks in fisheries management because ineffective reserves can give resource managers a false sense of security and prevent actions that might otherwise help to achieve the goals of MPAs (Allison et al., 1998; National Research Council, 2006; Gaines et al., 2010a,b; Hamilton et al., 2010). Ecosystem Based Management and Adaptive Management require an understanding of which MPAs contribute the most to recovery of over-exploited populations to inform future actions (Hamilton et al., 2010; White et al., 2011).

In response to conservation concerns associated with a sharp decline in catches of inshore rockfishes throughout the 1990s in the Northeast Pacific, Fisheries and Oceans Canada (DFO) implemented a system of 164 Rockfish Conservation Areas (RCAs) in British Columbia (BC), Canada, as part of a Rockfish Conservation Strategy. Rockfish Conservation Areas were established between 2004 and 2007 and prohibit commercial and recreational hook and line fisheries and bottom trawl fisheries; although fisheries for invertebrates by trap and hand, and seining, gill-netting and mid-water

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trawling are still permitted. The RCAs protect almost 30% of rockfish habitats between Vancouver Island and the mainland (inside waters) and approximately 15% of habitat on the rest of the coast (outside waters) (Yamanaka and Logan, 2010). Inshore rockfishes include six species of the genus *Sebastes* (Copper Rockfish *S. caurinus,* Quillback Rockfish *S. maliger,* Black Rockfish *S. melanops,* China Rockfish *S. nebulosus,* Tiger Rockfish *S. nigrocinctus,* and Yelloweye Rockfish *S. ruberrimus*) that are found on shallow (<200 m) rocky reefs. Numerous other fish species including Lingcod *Ophiodon elongatus*, and Kelp Greenling *Hexagrammos decagrammus* and the Greenstriped Rockfish *S. elongatus* are also protected in RCAs. Although the RCAs are often not considered to be MPAs because they are managed as fishery closures under the *Fisheries Act* as opposed to being permanently protected as MPAs by Canada's *Oceans Act* (Robb et al., 2011), they are spatially defined areas where fisheries that target or lead to substantial bycatch of rockfishes are prohibited.

Marine Protected Areas may be an effective tool to conserve Pacific rockfishes as they are long-lived (some >100 years), and have small home ranges (Matthews, 1990b; Yoklavich, 1998; Parker et al., 2000). Marine Protected Areas have been effective for conserving rockfishes in California. Two marine reserves in California had significantly larger rockfishes and greater biomass (and therefore greater reproductive output) than non-reserve sites, while a 1-yr old reserve showed no difference from open areas (Paddack and Estes, 2000). Five years after the Channel Islands marine reserve network was established, the biomass of targeted fish species, including five rockfish species, was approximately two times higher inside reserves than outside and targeted species biomass trajectories increased with time (Hamilton et al., 2010). Despite overall declining trends in US west coast groundfish CPUE in a fishery independent trawl survey, higher CPUE and larger fish were found for numerous rockfishes in American RCAs that were continuously closed to trawling (Keller et al., 2014). Spatial fisheries closures may therefore be effective for promoting rockfish population recovery; however, these studies used tools that cannot survey the entire habitat area encompassed by the RCAs (e.g., Keller et al., 2014 used bottom trawls not effective in rocky areas and Hamilton et al., 2010 used SCUBA methods that cannot survey in deeper depths).

The effectiveness of the RCA network in BC has yet to be evaluated in terms of population recovery. Two studies examined the performance of individual RCAs (Marliave and Challenger, 2009; Cloutier, 2011). Marliave and Challenger (2009) collected relative abundance and habitat data while SCUBA diving inside and outside of three RCAs in Howe Sound and concluded that the habitat model used to designate the RCAs had not included fine-scale habitat features with the highest rockfish abundances such as boulder piles. They did not find any evidence of reserve effect in relative abundance of Copper or Quillback Rockfishes in the RCAs compared to open areas, but acknowledged that the early timing of the surveys constitute a baseline condition for comparison with future population trends. Cloutier (2011) completed SCUBA surveys between 8 and 15 m of depth at 15 sites in southern BC and evaluated RCA performance using rockfish presence and density. He found that RCAs had 1.6 times the rockfish density (all rockfish species pooled) than non-RCA sites while accounting for differences in habitat quality. SCUBA surveys are limited in that they can only assess rockfish species or lifestages within shallow depth ranges (<20 m) that typically include Copper Rockfish and juvenile Quillback Rockfish (Richards 1987; Love et al., 2002). Quillback Rockfish, a threatened species (COSEWIC, 2009), have been observed to 182 m in BC, but are most abundant between 20 and 60 m (Richards, 1986). Yelloweye Rockfish, a species of Special Concern, are caught between 20 and 250 m of depth (COSEWIC, 2008) but highest densities have been observed between 40 and 100 m (Richards, 1986). Quillback and Yelloweye Rockfishes are the two inshore rockfish species most heavily targeted by fisheries (Yamanaka and Logan, 2010). It is therefore important to use methods that can access deeper waters than SCUBA can to assess rockfish abundance in RCAs.

Rockfish species segregate niches by habitat type as well as depth (Richards, 1986; Matthews, 1990a; Anderson and Yoklavich, 2007; Laidig et al., 2009). The highest abundances of Quillback Rockfish in the Strait of Georgia were found in complex habitats above 60 m, while Yelloweye Rockfish were found in similar habitats but in deeper water (Richards, 1986). Yelloweye and Greenstriped Rockfishes were found in similar depth ranges, but in different habitat types with Greenstriped Rockfish using fine-sediment habitats as opposed to rocky substrates (Richards, 1986). The other inshore rockfishes are also associated with complex rocky habitats (Richards, 1987; Matthews, 1990b; Love et al., 2002). Complex living habitats such as sponge reefs are important rockfish habitats for both juvenile and adult fishes in BC (Cook et al., 2008; Marliave et al., 2009). Habitat is an important source of variability for fish communities; therefore, habitat structure must be accounted for in both the design and assessment of MPAs (Parnell et al., 2006; Claudet and Guidetti, 2010; Miller and Russ, 2014). Habitat variables should be collected as covariates with fish observations and included in models of reserve effectiveness to determine if a positive response is related to protection or to intrinsic structural features of the protected area (Pelletier et al., 2008; Claudet and Guidetti, 2010).

Remotely Operated Vehicles (ROVs) are one of the most effective non-destructive monitoring tools to sample populations in protected areas (Field et al., 2006; Stoner et al., 2008). Size, operability, and cost of ROVs have all decreased in recent years. In addition to fish abundance and size data, visual surveys also have the ability to collect information on habitat use, behaviour and associations with other species (Yoklavich et al., 2002; Laidig et al., 2009; Love et al., 2009; O'Farrell et al., 2009). Stoner et al. (2008) contend that there is no better way to monitor fishes in structurally complex habitats. ROV surveys can better encompass the entire area of the RCAs than methods used in previous studies (e.g., Marliave and Challenger, 2009; Hamilton et al., 2010; Cloutier, 2011; Keller et al., 2014). ROV surveys targeting deeper-dwelling species were also undertaken in the Channel Islands within the first five years after establishment; the authors compared the density of fishes on hard-bottom substrates between 20 and 100 m in depth among three sites pairs inside and outside of reserves (two other site pairs were dropped) (Karpov et al., 2012). Results varied among sites and years; however, effects of protection on some rockfishes were apparent. The authors expected that reserve effects would become more pronounced with additional time for growth and recruitment (Karpov et al., 2012).

In this study, we assess the effectiveness of the RCA network in promoting rockfish recovery in BC using ROV surveys. We focus on Quillback Rockfish, Yelloweye Rockfish, Greenstriped Rockfish, Lingcod and Kelp Greenling, as well as the inshore rockfish species in a combined group. We aim to address the three questions.

1 Is there any evidence of an increased density or body size of targeted or non-targeted groundfish in RCAs in BC? Species that are targeted by fisheries have been found to show stronger reserve effects than non-targeted species (Cote et al., 2001; Hamilton et al., 2010). If reserves significantly reduce mortality, species targeted by fisheries such as Lingcod, Yelloweye Rockfish and Quillback Rockfish should have higher densities inside reserves while non-targeted species such as Greenstriped Rockfish and Kelp Greenling may not show a difference.
2 Does the age of the reserve influence the reserve effect? The effectiveness of MPAs is likely to vary spatially among species with different life histories and with the age of the reserve (Molloy et al., 2009; Babcock et al., 2010). The life-history characteristics

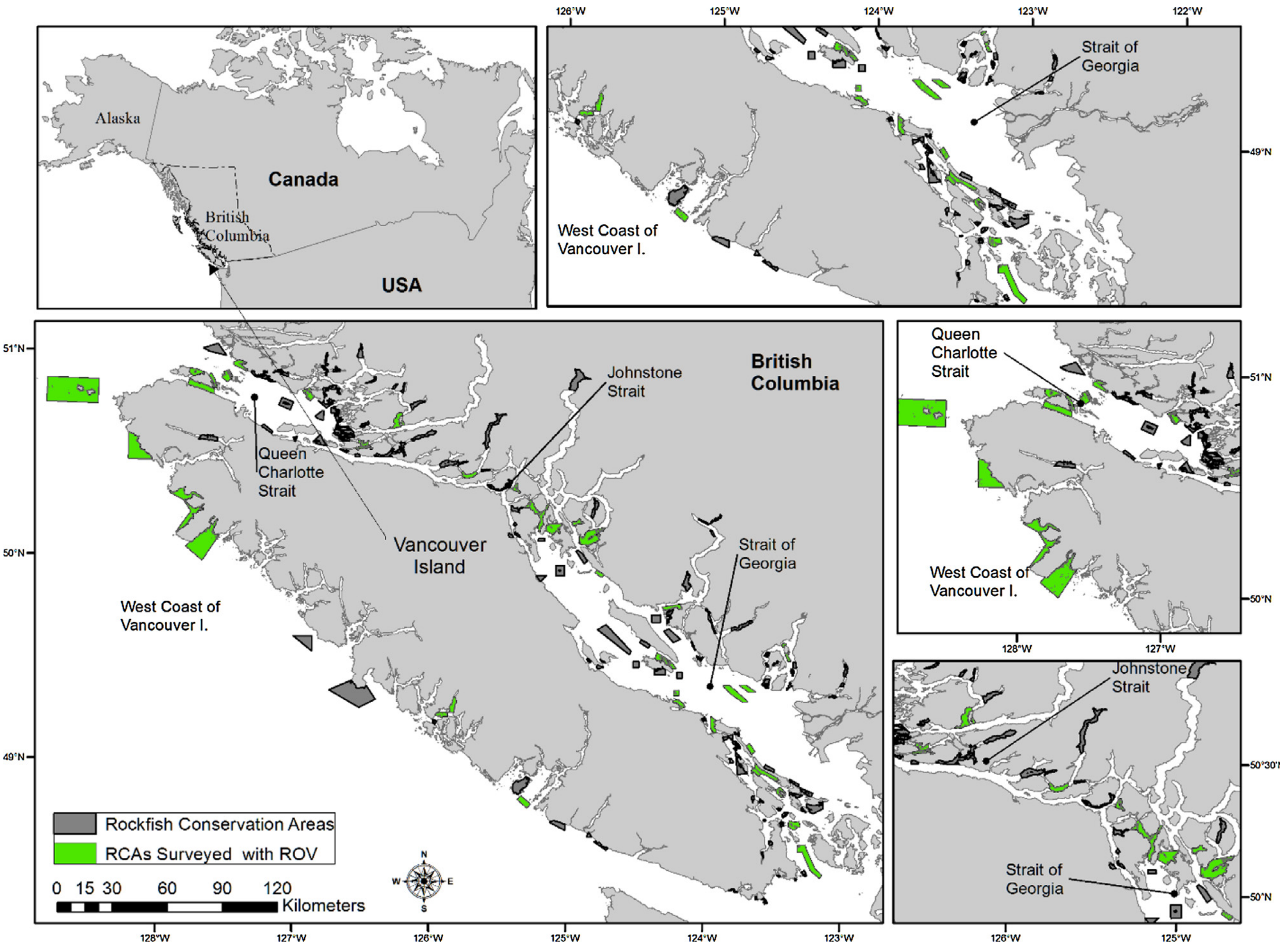

Fig. 1. Rockfish Conservation Areas in southern BC that were sampled using an ROV between 2009 and 2011 are shown in green.

that make rockfishes susceptible to overfishing, including slow growth, large size and old age at maturity, and episodic recruitment, also indicate that their recovery will be slow (Hutchings, 2001; Dulvy et al., 2003, 2004; Hutchings and Reynolds, 2004). The earliest we are likely to be able to detect any reserve effects for rockfishes is predicted to be between 5–10 years (Yoklavich, 1998). Reserve effects may be more apparent earlier for fast growing species like Lingcod that are mature at 3–5 years (Cass et al., 1990). We expect that RCAs that have been in place for more than five years at time of sampling to show stronger reserve effects than younger RCAs.

3 Does adjacent fishing pressure influence the reserve effect? The intensity of fishing occurring outside of an MPA may also determine the strength of a reserve effect (Mosqueira et al., 2000; Claudet and Guidetti, 2010). Reserve effects may not be apparent if effort also decreases outside of reserves, or if effort was initially low in adjacent waters before reserve designation. In addition to establishing RCAs, Fisheries and Oceans Canada also decreased the fishing mortality in commercial and recreational fisheries by decreasing the total allowable catch and daily limits of the two fishing sectors, respectively. Commercial and recreational catch estimates can be used to measure the fishing effect size adjacent to conservation areas (Claudet and Guidetti, 2010). We expect RCAs in areas of the coast with strong fishing pressure to show greater reserve effects than those with low adjacent fishing pressure.

2. Materials and methods

2.1. Study area

We surveyed RCAs on BC's south coast. Three bodies of water are between Vancouver Island and the mainland: the Strait of Georgia, Johnstone Strait-Discovery Passage and Queen Charlotte Strait (Fig. 1). These are collectively called "Inside Waters." The Strait of Georgia has shallow depths, large fresh water inputs from the Fraser River, and numerous islands with high tidal flows between them. Johnstone Strait is characterized by rapid tidal streams and vigorous mixing, steep rocky walls and depths to 500 m. Numerous deep coastal fjords with steep sides are found on the mainland side of Johnstone Strait. The Queen Charlotte Strait is a shallow, island-strewn basin with less intense tidal currents than Johnstone Strait except in passages between islands. The West Coast of Vancouver Island (WCVI) is part of "Outside Waters" along with BC's north coast. The outer coast is characterized by a rocky shoreline and much greater wave exposure. The WCVI has five main Sounds with numerous islands that lead into long coastal inlets (Thomson, 1981). DFO manages the "Inside" and "Outside" inshore rockfish fishery separately (Yamanaka and Logan, 2010).

2.2. Data collection

We surveyed 35 RCAs in four regions of BC (Strait of Georgia, Johnstone Strait, Queen Charlotte Strait, and the West Coast of Vancouver Island) with an ROV on seven research cruises between

Table 1
The number of transects, mean and Standard Deviation (SD) of Fish Densities (#/100m^2) inside and outside of RCAs observed on ROV surveys by region. SG = Strait of Georgia, JS = Johnstone Strait, QCST = Queen Charlotte Strait, WCVI = West Coast of Vancouver Island. The number of RCAs sampled per region is shown in parentheses.

Region	RCA	N	QB		YE		LC		IRF		GS		KG	
			$\bar{x}$	SD	$\bar{x}$	SD	$\bar{x}$	SD	$\bar{x}$	SD	$\bar{x}$	SD	$\bar{x}$	SD
SG	In	122	0.57	0.68	0.10	0.18	0.12	0.19	0.79	0.84	0.26	0.40	0.17	0.27
(13)	Out	81	0.70	0.71	0.12	0.21	0.13	0.20	0.89	0.82	0.28	0.38	0.16	0.22
JS	In	13	1.01	1.34	0.05	0.10	0.08	0.15	1.06	1.33	0.19	0.19	0.02	0.05
(5)	Out	15	1.27	1.14	0.13	0.16	0.06	0.15	1.40	1.09	0.29	0.44	0.04	0.10
QCST	In	18	0.86	0.91	0.26	0.36	0.04	0.08	1.31	1.36	0.05	0.10	0.34	0.60
(5)	Out	16	0.82	1.20	0.12	0.13	0.10	0.14	1.40	1.84	0.05	0.08	0.34	0.60
WCVI	In	46	0.31	0.40	0.20	0.25	0.17	0.21	0.98	0.94	0.01	0.02	0.36	0.52
(7)	Out	54	0.37	0.42	0.20	0.26	0.21	0.25	0.99	0.74	0.02	0.07	0.24	0.25

February 2009 and July 2011 (Fig. 1). Because there are no comparable survey data from before the RCAs were established, this study employs the Control-Impact design (Underwood, 1992) whereby data from inside the RCAs are compared to near-by sites that are open to fishing in order to infer reserve effects (Glasby, 1997; Pelletier et al., 2008; Claudet and Guidetti, 2010). We identified paired transects 300–900 m long using GIS targeting rockfish habitat inside and outside of RCAs based on depth contours using nautical charts or multibeam bathymetry. All data were collected during daylight hours. Most transect lines were perpendicular to the shore and the ROV traveled from deep to shallow, although transects in Johnstone Strait with very steep walls had to be run parallel to shore. The Saltspring Island N. RCA is adjacent to the Trincomali Passage RCA, so these were pooled for analysis. Similarly, the Dommett Point and Pam Rock, Halibut Bank and McCall Bank, Clio Channel and Viscount Island, and Octopus Island and Read Island RCAs are all close to each other and transects outside of the RCAs could be applied to either RCA. Therefore, we pooled data from these RCAs for analysis, resulting in 30 comparisons of RCAs and unprotected sites in this analysis (Fig. 1, Table 1).

ROV surveys were conducted on the Canadian Coast Guard vessels the "Vector" and the "Neocaligus" using a Deep Ocean Engineering Phantom HD2 + 2 ROV with a 300 m umbilical. We deployed the ROV from the ship and temporarily fastened the umbilical to a wire with a 225 kg clump-weight to allow for greater control and operability of the ROV and to keep the ROV below the ship to improve ROV tracking. The position of the ROV was determined using "TrackPoint 3" ultra-short baseline acoustic tracking system (Ore International) and ROV and ship navigation data were recorded and mapped using Hypack 2009. The typical speed of the ROV was about 0.4–0.8 kt, or 1/4 to 1/2 meter per second. The ROV was equipped with two 120 W ROS Q-LED lights positioned below the camera and one 150 W Super SeaArc HID light above the camera for a total of 18,000 lumens of light. Lights were used during all transects. Video was captured using a Sony EVI 300 Zoom video camera mounted on the ROV, and data were recorded digitally onto hard drives and onto redundant Mini DV tapes. Two green lasers spaced ten cm apart mounted on the ROV were used to estimate the length of fishes as well as the width of the field of view which is taken to be the transect's width.

Following the surveys, we reviewed the videos and all fish, habitat, and field of view observations were recorded in a program specially designed for rockfish surveys (AVLog). Viewers counted all fish observed and identified them to the lowest taxonomic level possible. When the entire length of the fish was visible, viewers used a ruler to measure body length when it was in plane with the lasers, as well as the space between the lasers. We found the length of the fish using the ratio of the actual laser width (10 cm) to the laser width measured on screen. Fish length was assigned to categories in 5 cm bins to account for the limited accuracy in this method. We determined the width of the field of view using the same method. Every 30 s, the reviewer recorded the distance between the lasers and the width of viewing window on screen and used the laser ratio to find the field of view.

Habitat was recorded as a continuous variable. Each time the habitat changed, viewers paused the video to record a new habitat. Although the change to a new habitat is sometimes incremental and difficult to pinpoint, the observers recorded the change as best as they could when the lasers crossed on to what they considered the new substrate. Primary substrate was categorized in nine classes that were subsequently lumped into three classes for analysis: rock, mixed coarse and fine. Similarly, 19 classes of biocover were recorded in the original dataset but these were grouped into three classes: bare, encrusting organisms (e.g., barnacles, tube worms, hydroids) and emergent organisms (e.g., *Metridium* anemones, sponges, sea pens). Habitat complexity and relief were also recorded in four classes which were re-classified into high and low.

The ROV transects were mapped in ArcGIS from track points downloaded from the acoustic tracking system. ROV tracking is subject to various errors including bottom characteristics such as steep rock walls that scatter or reflect acoustic energy and affect positional accuracy. Therefore, points that were clear outliers were eliminated or edited. Smoothed transect lines were then created from the edited points and calibrated with the date-time. The actual distance the ROV traveled over the ocean floor was then calculated using the z-field (depth) in the ArcGIS tool 3D Analyst to determine the surface length. Area-swept polygons were created using the field of view measurements and the buffer tool with half of the field of view measurement as the radius of the buffer. Habitat variables were assigned to each polygon. The date-time field associated with each track point links the field of view measurements, habitat and fish observations and allows them each to be mapped onto the transects. Fish observations from the video analysis were then mapped as points along each transect line. Data are all managed in ArcGIS geodatabases as well as in a master data base maintained at the Pacific Biological Station (PACGFVideo).

2.3. Analysis

2.3.1. Fish density

We estimated the densities of the following groundfish species: Quillback Rockfish, Yelloweye Rockfish, Greenstriped Rockfish, Lingcod and Kelp Greenling. Because observations of inshore rockfish species other than Quillback and Yelloweye were less common, we pooled data from all inshore rockfish species, including Quillback and Yelloweye, and analyzed them as a species group. To calculate fish density we summed the number of fish observed, divided by the area surveyed on each transect and expressed it as the number per 100 m^2. Prior to analysis, we ln(x + 0.1) transformed the densities to normalize variance with the presence of zeros in the data (Zar, 1996). We plotted the log density by region and level of protection and used Analysis of Variance (ANOVA) to determine if fish densities differed by region.

2.3.2. Effects of protection and habitat by transect

We constructed a linear mixed effects model (LME) using the package lmertest (Kuznetsova et al., 2014) in R (R Development Core Team, 2008) with a nested design to test if fish density was dependent on protection status, habitat and depth. For this analysis, we included only data from the last sampling date for transects that we sampled more than once to ensure independence in our dataset. We modeled the density of each species/species group separately and fit the final models using the restricted maximum likelihood (REML) (Zurr et al., 2009). Hamilton et al. (2011) found that biogeography was important to include in assessments of networks of protected areas because different regions studied have different physical and ecological characteristics. The regions we studied (the Strait of Georgia, Johnstone Strait, Queen Charlotte Strait, and the West Coast of Vancouver Island), have different physical and ecological characteristics, as well as separate exploitation histories and management. Therefore, we used the identity of the RCAs nested within Region as random variables. First we tested the effect of habitat and depth on fish density. To describe the habitat encountered on the transects, we combined three levels of substrate (rock, mixed coarse and mixed fine) and three levels of biocover (bare, encrusting and emergent organisms) resulting in nine biocover-substrate classes (Bare-Rock, Encrusted-Rock, Emergent-Rock, Bare-Mixed Coarse etc.). We calculated the percent of each transect covered by the nine habitat classes, as well as the percent high complexity and high relief. Prior to analysis, we used the logit transformation on the percentages (Fox and Weisberg, 2011). We included all transformed habitat variables and the mean transect depth in a habitat model for each species. We used the package MuMIn (Barton, 2013) and maximum likelihood estimation (Zurr et al., 2009) to find the optimum model structure using the second-order Akaike Information Criterion (AICc) for small sample sizes to rank the subsequent models. We found a subset of the models that explain 95% of the model weight and averaged them together. Habitat variables that were significant ($p < 0.1$), were retained and used in the next modeling step. Next, we included the level of protection (RCA or Open) in the model along with habitat variables that were found to influence fish density. Once again, we used MuMIn (Barton, 2013) to find the optimum model structure and to perform model averaging on the models that explained 95% of the variance using the full model-averaged coefficients (i.e. with shrinkage). We calculated the relative variable importance (RVI) for each model.

In order to determine if similar habitats were sampled inside and outside of RCAs, we plotted the mean and standard error of the percent habitat type and depth observed on transects inside and outside of RCAs both by Region and RCA to see if the Standard Errors overlapped.

We examined the depth ranges of the fish species by calculating the total area sampled over all of our transects by depth class and summed the number of fish of each species observed in each depth class. We used a Chi-squared analysis to determine if each species had a preference for certain depth classes by comparing the observed number per class to an expected number assuming no preference. We calculated the 95% confidence interval to determine which classes each species preferred and compared the preferred depth range of each species to the depths sampled.

2.3.3. Effects of catch, age, area and shape on RCA effectiveness

We analyzed data from RCAs to determine what factors influence effectiveness at promoting population recovery using the log response ratio (RR) (Eq. (1)) as a measure of reserve effectiveness (Hedges et al., 1999; Russ et al., 2005; Claudet et al., 2010; Hamilton et al., 2010; Edgar et al., 2014).

$$RR = \ln\left(\frac{\bar{X}_{in} + 1}{\bar{X}_{out} + 1}\right) \quad (1)$$

If the RR is greater than 0, the species is more abundant inside the reserve; while RRs less than zero indicate the species in more abundant outside the reserve.

To control for the effect of habitat variability on species' densities, we used information on habitat-use by each species/group from the LME models to calculate a habitat-specific density. In ArcGIS, we selected the habitat features used by each species/group and then selected the fish observations that intersected with or were within 2 m of the preferred habitat. We calculated the area of the transect that included preferred habitat for each species and excluded other habitats and fish observations from the habitat-based density calculations. Next we calculated the mean fish density on transects inside and outside of each RCA for each sampling trip and calculated a new RR. We plotted the mean and standard error (SE) of the Quillback, Yelloweye and Lingcod RRs by RCA and the RRs for each species/group over all RCAs. We also plotted the log RR of each species/group versus the age of the RCA.

To quantify the fishing history and current exploitation levels, we used the commercial catch data from the BC trawl and hook and line fisheries (data courtesy of Fisheries and Oceans Canada, N. Olson, Groundfish Data Unit). We did not consider recreational catch in this analysis because data were not available at a comparable spatial resolution. We used ArcGIS 10.2 to calculate the total commercial harvest weight (kg) for each of our groundfish species/groups in a buffer area 5 km-wide around each RCA and divided it by the area of the buffer for a total kg/km^2. We used trawl data from 1996 to 2006 and hook and line data from 2002 to 2006 for the period before the RCAs were established in the RCA buffer as well as in the RCA because these were the years available with the required spatial and species resolution. Data from the period after establishment are from 2007 to 2011 for both fisheries. We calculated the area (km^2), and the area to open-water perimeter ratio of each RCA (i.e., perimeter of the RCA that abuts land was not included in the perimeter-area ratio) in ArcGIS 10.2. The age of the RCA at the time it was sampled was the number of years since establishment. We used a Linear Mixed Effects model of the RRs with Region as the random variable and the fixed effects: log catch + 1 in the RCA before establishment, log catch + 1 outside of the RCA before establishment, log catch + 1 outside of the RCA after establishment, RCA age, log RCA area, RCA perimeter-to-area ratio. We fit the model using REML (Zurr et al., 2009).

2.3.4. Length of fish inside to outside of RCAs

We compared the length frequency of Quillback, Yelloweye, Greenstriped, and Copper Rockfishes as well as Lingcod and Kelp Greenling found inside and outside of all RCAs pooled together using the Kolmogorov-Smirnov test and length frequency histograms.

3. Results

3.1. Fish density

We surveyed 199 transects inside 35 RCAs, and 166 outside (Table 1). Three RCAs were sampled twice, Brethour was sampled on three occasions and Northumberland was sampled four times. Thirteen of the surveyed RCAs were in the Strait of Georgia, five in Johnstone Strait, five in Queen Charlotte Strait and seven in the WCVI. Rockfish Conservation Areas had been designated for between 3 and 7 years at the time of sampling (Table A.1 in

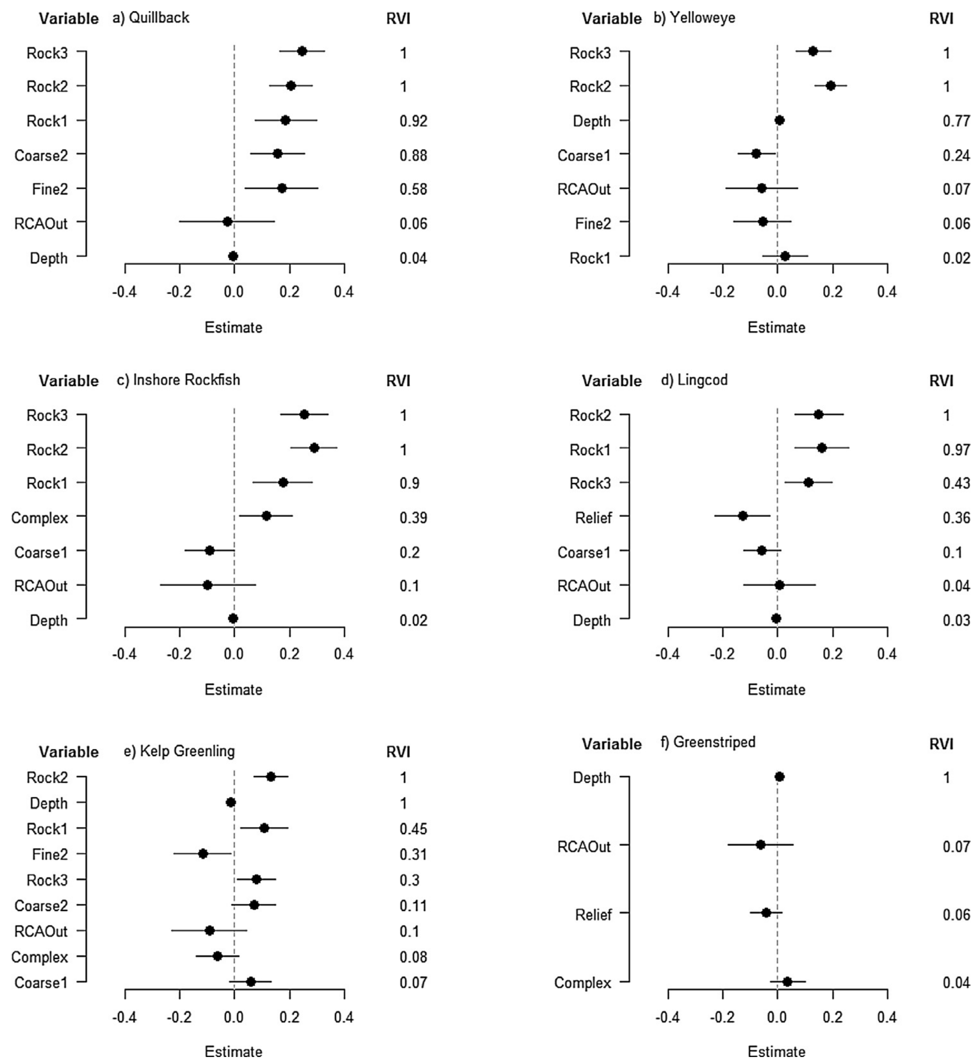

Fig. 2. The% Relative Variable Importance (RVI), estimate and 95% confidence intervals of habitat and protection variables retained in the linear mixed effects averaged model.1 = Bare, 2 = Encrusted, 3 = Emergent Biocover.

Supplementary material). Because the RCAs in the same region were established in the same year, and RCAs close to each other were sampled on the same cruise, RCA age is somewhat confounded with the regions.

Quillback Rockfish were the most abundant species of interest observed on transects (Table 1). The density of all of the fish species we studied except for the group of inshore rockfishes differed among regions (Table A.2 in Supplementary material). The pattern of abundance varied among species (Table A.1 in Supplementary material). For instance, Quillback Rockfish were most abundant in Johnstone Strait while Lingcod were most abundant in WCVI (Table 1).

3.2. Effects of protection and habitat

The LME model for each species revealed habitat features that had positive and negative effects on fish densities (Fig. 2). The averaged LME model for each fish species/group explained significant variation in densities except for the Kelp Greenling model (Table 2). Protection status did not have a significant effect on transect density for any species/group. The level of Protection (RCA/Open) had one of the lowest variable importance scores for all models. Of the habitat variables, rocky substrates with encrusting and emergent organisms and bare rock often had the highest densities of the species/group studied. Quillback Rockfish density was greatest on rocky substrates as well as mixed coarse and mixed fine substrates with encrusting organisms. Yelloweye Rockfish density was lower on mixed fine and mixed coarse substrates and not sensitive to bare rock. Although all rock categories, bare and encrusted mixed coarse and fine encrusted substrates were retained in the habitat for Kelp Greenling, only encrusted rock was significant in the averaged model (Table 2). No substrate variables were retained in the Greenstriped Rockfish habitat model (Table 2, Fig. 2).

The Chi-square analysis showed that all fish species had a preferred depth range (Table 3). Our survey targeted the preferred depth range of most of our fish species of interest. Ninety-two percent of the area sampled was between 25 and 125 m deep. Quillback, Tiger, Yelloweye and Greenstriped Rockfishes, and Lingcod were found within this depth range, although Yelloweye and Greenstriped Rockfishes were also distributed below this depth (Table 3). Kelp Greenling, Copper and China Rockfishes have shallower distributions and only 3% of the area surveyed was above 25 m. The Chi-square analysis results were corroborated by the LME model. Mean transect depth was retained in all of the LME models (Table 2, Fig. 2). Depth was the most important variable in the Kelp Greenling and Greenstriped models and was also important in the Yelloweye model. Depth was the variable of lowest

Table 2
Linear Mixed Effects Model results of averaged model (with shrinkage) of habitat and RCA status on fish density (fish/100 m^2) on ROV transects (n = 298). RVI = relative Variable Importance. Bold values denote significance.

Term	Estimate	SE	z-value	p	RVI
Quillback					
Intercept	1.45	0.60	2.42	**0.02**	
Depth	−0.0002	0.001	1.19	0.85	0.04
Rock-Bare	0.17	0.07	2.3	**0.02**	**0.92**
Rock-Encrusted	0.20	0.04	5.25	**<0.0001**	1
Rock-Emergent	0.25	0.04	5.87	**<0.0001**	1
Coarse-Encrusted	0.14	0.07	1.97	**0.05**	0.88
Fine-Encrusted	0.10	0.1	1.01	0.31	0.58
RCA	−0.002	0.02	0.07	0.94	0.06
Yelloweye					
Intercept	−1.49	0.28	5.24	**<0.0001**	
Depth	0.005	0.003	1.62	0.11	0.77
Rock-Bare	0.0006	0.007	0.08	0.94	0.02
Rock-Encrusted	0.19	0.03	6.61	**<0.0001**	1
Rock-Emergent	0.12	0.03	4.01	**0.0002**	1
Coarse-Bare	−0.02	0.04	0.5	0.62	0.24
Fine-Encrusted	−0.003	0.02	0.18	0.85	0.06
RCA	−0.004	0.02	0.17	0.86	0.07
Inshore Rockfish					
Intercept	1.31	0.33	3.97	**<0.0001**	
Depth	−0.00001	0.0001	0.116	0.91	0.02
Rock-Bare	0.016	0.007	2.17	**0.03**	0.90
Rock-Encrusted	0.29	0.04	6.86	**<0.0001**	1
Rock-Emergent	0.25	0.04	5.81	**<0.0001**	1
Coarse-Bare	−0.02	0.04	0.44	0.66	0.20
Complexity	0.05	0.06	0.71	0.48	0.39
RCA	−0.01	0.07	0.25	0.81	0.10
Lingcod					
Intercept	−1.07	0.28	3.82	**0.0001**	
Depth	−0.0001	0.001	0.001	0.17	0.03
Rock-Bare	0.16	0.06	2.75	**0.006**	0.97
Rock-Encrusted	0.15	0.05	3.32	**0.001**	1
Rock-Emergent	0.05	0.06	0.77	0.44	0.43
Coarse-Bare	−0.01	0.02	0.28	0.78	0.10
Relief	−0.05	0.07	0.67	0.5	0.36
RCA	0.0002	0.01	0.02	0.98	0.04
Kelp Greenling					
Intercept	−0.25	0.41	0.60	0.55	
Depth	−0.01	0.002	8.86	**<0.0001**	1
Rock-Bare	0.05	0.06	0.8	0.42	0.45
Rock-Encrusted	0.13	0.03	4.30	**<0.0001**	1
Rock-Emergent	0.02	0.04	0.58	0.56	0.30
Coarse-Bare	0.004	0.02	0.23	0.82	0.07
Coarse-Encrusted	0.01	0.02	0.23	0.76	0.11
Fine-Encrusted	−0.04	0.06	0.60	0.55	0.31
Complexity	−0.01	0.02	0.26	0.80	0.08
RCA	−0.01	0.04	0.27	0.79	0.10
Greenstriped					
Intercept	−2.26	0.22	10.0	**<0.0001**	
Depth	0.01	0.001	4.46	**<0.0001**	1
Complexity	0.002	0.01	0.16	0.88	0.04
Relief	−0.002	0.01	0.16	0.84	0.06
RCA	−0.004	0.02	0.20	0.85	0.07

importance in the Quillback Rockfish, Lingcod and inshore rockfish models (Table 2, Fig. 2).

We successfully targeted similar habitats on transects inside and outside of most RCAs. The average habitat sampled and mean depths were very similar (Fig. 3). Although habitat sampled was similar inside and outside most RCAs, some such as D'Arcy Island (where we sampled more rocky substrates in the RCA), had habitat differences inside vs. out (Fig. A.1 in Supplementary material). Some RCAs such as Halibut Banks, Ballenas I., Saltspring-Trincomali, and Thurston I. had relatively low percentage of rocky substrates. We sampled more rocky substrates outside of the Sarnac I. and Thurston I. RCAs (Fig. A.1 in Supplementary material).

In order to eliminate the confounding effect of habitat differences in our analysis of the RCAs, we calculated habitat-based fish densities. For inshore rockfishes, Yelloweye Rockfish and Lingcod, we used rock substrates that were bare or had encrusting or emergent biocover as the preferred habitat type (Table 2). Quillback Rockfish habitat consisted of all of the rocky substrates as well as mixed coarse and mixed fine with encrusting biocover, which likely indicates the presence of boulders (Table 2). Kelp Greenling also used all rocky substrates and bare and encrusting mixed-coarse substrate. Greenstriped Rockfish are limited by depth but no substrates were significant in the habitat LME (Table 2). We used all substrates along the transects but only depths greater than 40 m because no Greenstriped Rockfish were observed above 40 m.

Analyzing fish density only within the preferred habitat of each species changed the ranking of RR among the RCAs (Fig. 4a). For instance, D'Arcy I. had the highest RR using the total area of the

Table 3
The proportion of depth classes sampled on 420 ROV transects and the expected and observed proportion of fish species and 95% Confidence Intervals by depth class. Numbers in bold indicate the preferred depth zones of each species. A Chi-squared analysis indicated that species are not distributed evenly across depth classes. Bold values denote significance.

Depth Class (m)	0–25	26–50	51–75	76–100	101–125	126–150	151–200
Habitat Area (m^2)	16,517	173,933	160,382	87,062	36,388	13,280	8260
Proportion by Area	0.03	0.35	0.32	0.18	0.07	0.03	0.02
Copper	N = 349, χ^2 = 312, df = 6, p = **<0.0001**						
Expected	**0.03**	**0.35**	**0.32**	0.18	0.07	0.03	0.02
Upper CI	**0.18**	**0.68**	**0.23**	0.04	0.01	0.01	0.00
Observed	**0.15**	**0.63**	**0.19**	0.03	0.00	0.00	0.00
Lower CI	**0.11**	**0.58**	**0.15**	0.01	0.00	0.00	0.00
China	N = 530, χ^2 = 347, df = 6, p = **<0.0001**						
Expected	**0.03**	**0.35**	**0.32**	0.18	0.07	0.03	0.02
Upper CI	**0.11**	**0.70**	**0.29**	0.01	0.00	0.00	0.00
Observed	**0.09**	**0.65**	**0.25**	0.00	0.00	0.00	0.00
Lower CI	**0.06**	**0.61**	**0.22**	0.00	0.00	0.00	0.00
Kelp Greenling	N = 1151, χ^2 = 809, df = 6, p = **<0.0001**						
Expected	**0.03**	**0.35**	**0.32**	0.18	0.07	0.03	0.02
Upper CI	**0.12**	**0.69**	**0.22**	0.04	0.00	0.00	0.00
Observed	**0.10**	**0.67**	**0.20**	0.03	0.00	0.00	0.00
Lower CI	**0.08**	**0.64**	**0.18**	0.02	0.00	0.00	0.00
Quillback	N = 2995, χ^2 = 220, p = **<0.0001**						
Expected	**0.03**	**0.35**	**0.32**	**0.18**	**0.07**	0.03	0.02
Upper CI	**0.04**	**0.47**	**0.34**	**0.16**	**0.04**	0.01	0.003
Observed	**0.03**	**0.45**	**0.32**	**0.15**	**0.04**	0.01	0.002
Lower CI	**0.03**	**0.43**	**0.31**	**0.13**	**0.03**	0.01	0.000
Lingcod	N = 869, χ^2 = 66, df = 6, p = **<0.0001**						
Expected	**0.03**	**0.35**	**0.32**	**0.18**	**0.07**	0.03	0.02
Upper CI	**0.037**	**0.47**	**0.36**	**0.184**	**0.06**	0.003	0.006
Observed	**0.026**	**0.44**	**0.33**	**0.159**	**0.04**	0.001	0.002
Lower CI	**0.016**	**0.41**	**0.30**	**0.134**	**0.03**	−0.001	−0.001
Tiger	N = 117, χ^2 = 26, p = **0.0002**						
Expected	**0.03**	**0.35**	**0.32**	**0.18**	**0.07**	0.03	0.02
Upper CI	**0.04**	**0.33**	**0.50**	**0.32**	**0.07**	0.02	0.00
Observed	**0.02**	**0.26**	**0.42**	**0.25**	**0.04**	0.01	0.00
Lower CI	**0.00**	**0.19**	**0.35**	**0.19**	**0.01**	−0.01	0.00
Yelloweye	N = 793, χ^2 = 46, df = 6, p = **<0.0001**						
Expected	0.03	**0.35**	**0.32**	**0.18**	**0.07**	**0.03**	**0.02**
Upper CI	0.02	**0.30**	**0.44**	**0.24**	**0.09**	**0.03**	**0.02**
Observed	0.01	**0.27**	**0.40**	**0.21**	**0.07**	**0.02**	**0.01**
Lower CI	0.01	**0.24**	**0.37**	**0.18**	**0.05**	**0.01**	**0.01**
Greenstriped	N = 700, χ^2 = 767, df = 6, p = **<0.0001**						
Expected	0.03	0.35	**0.32**	**0.18**	**0.07**	**0.03**	**0.02**
Upper CI	0.00	0.01	**0.30**	**0.41**	**0.26**	**0.10**	**0.05**
Observed	0.00	0.00	**0.27**	**0.37**	**0.23**	**0.08**	**0.04**
Lower CI	0.00	0.00	**0.24**	**0.34**	**0.20**	**0.06**	**0.02**

transect, and Thurston I. had the lowest RR (Fig. 4b). When we adjusted fish density to account for preferred habitat, the RRs of these two RCAs were not different from zero (Fig. 4a). The mean habitat-adjusted RR of most (19) RCAs was not different from zero (the standard error bars overlap zero, Fig. 4a) indicating fish densities were not different inside to outside of the RCA. Eight RCAs had mean RRs less than zero, indicating more fish were observed outside and nine had RR's greater than zero, indicating more fish were observed inside (Fig. 4a).

No fish species/groups showed any difference in RR over all the RCAs (Fig. 5). Kelp Greenling and Greenstriped Rockfish are not targeted by commercial fisheries and had the narrowest range of RRs (Fig. 5).

3.3. Effects of catch, age, area and shape on RCA effectiveness

The age of the RCA did not affect the RR; the highest RR values were found in RCAs that had been in place for 5 years when they were sampled (Fig. 6). The age of the RCA is, however, confounded with Region (Fig. A.2 in Supplementary material), with the oldest RCAs found in the WCVI which we sampled last.

None of the factors tested in the LME explained any of the variation in RR among RCAs. Commercial catch outside of the RCA or inside the RCA before establishment, catch outside after RCA establishment, RCA age, RCA area, or perimeter-to-area ratio all failed to explain significant variability in RR (Table A.2, Fig. A.3 in Supplementary material). Many RCAs, particularly in the Strait of Georgia, had no or very low levels of commercial catch of groundfish species outside of the RCA after RCA implementation (Fig. A.4 in Supplementary material).

None of the fish species showed any difference in length frequencies by protection status (Fig. 7). Most of the Yelloweye Rockfish observed were juveniles (less than 50 cm, Fig. 7) that have not yet reached 50% maturity.

4. Discussion

Our survey provides little indication that recovery is underway for demersal fish populations inside BC's Rockfish Conservation Areas three to seven years after their establishment. No difference in fish numbers and sizes was detected in visual survey of protected and reference sites. Different species showed clear habitat

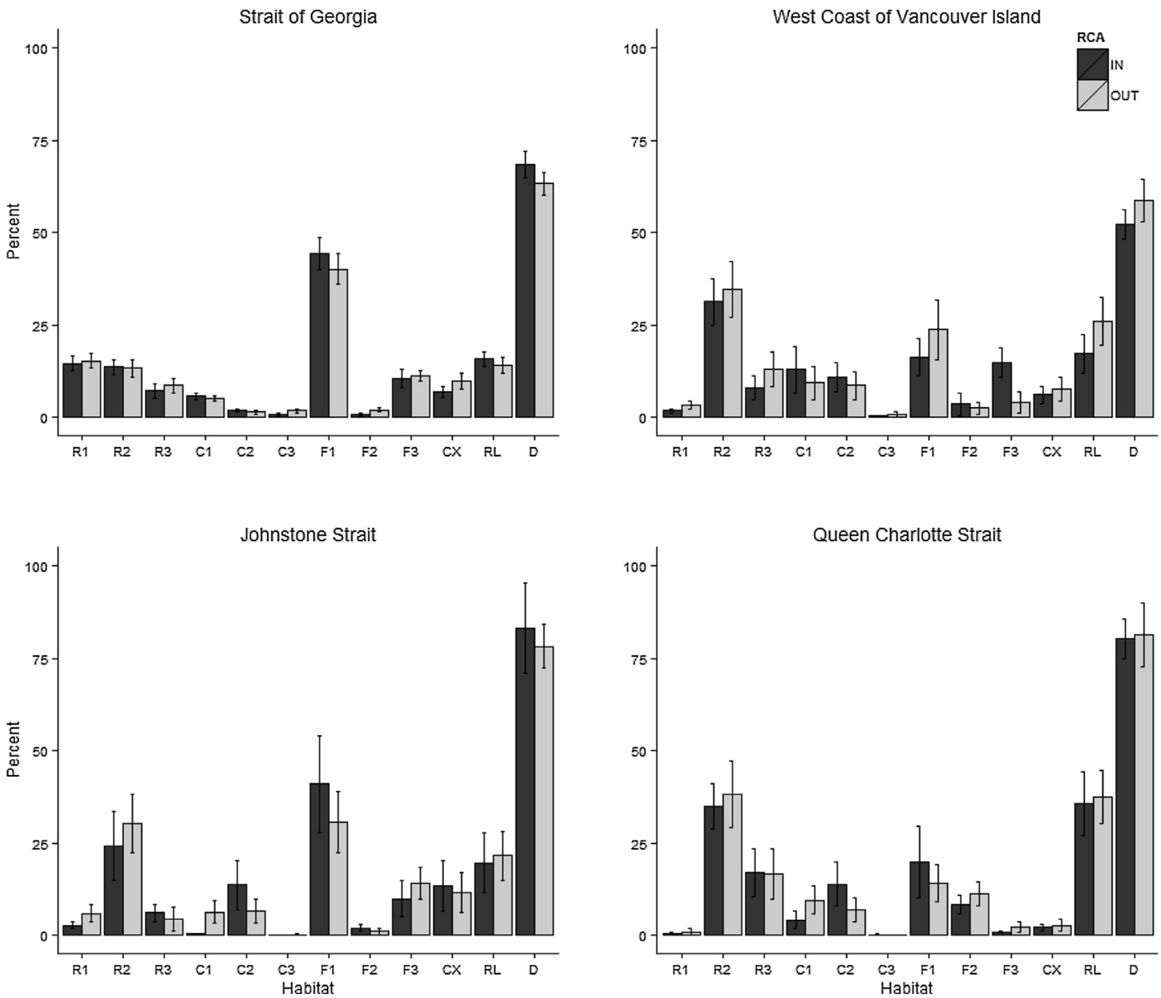

Fig. 3. The mean and SE of the percent of habitat type by region on transects inside and outside of the RCAs. R1 = Bare Rock, R2 = Rock with encrusting biota, R3 = Rock with emergent biota; C1 = Bare Coarse substrates, C2 = Coarse with encrusting biota, C3 = Coarse with emergent biota; F1 = Bare Fine substrate, F2 = Fine with encrusting biota, F3 = Fine with emergent biota; CX = High Complexity, RL = High Relief, D = Depth (in m).

affinities, and inclusion of protection status added no explanatory power in a model to predict fish densities based on depth and bottom cover types. These results have a number of implications for spatial fisheries closures as a management tool for promoting recovery of rockfishes and other groundfish. First, the level of protection afforded by the RCA network may be inadequate to allow populations to recover if illegal fishing or fishing that is allowed within the boundaries of RCAs continues to impose high mortality. Alternatively, sufficient time may not have elapsed for growth and recruitment to replenish depleted stocks inside RCAs. Ongoing visual population assessments are necessary to distinguish between these two possibilities, and our data provide a baseline for future comparisons. ROV surveys are an effective way to monitor inshore rockfishes in RCAs. Unlike longline or other fishing surveys, they concurrently collect habitat and fish data and are non-extractive. However, great technical expertise is required to run successful ROV surveys, record data from videos, and to process tracking data after the surveys.

Depth and habitat were the only variables that explained fish densities in our models. Other studies have shown that rockfishes partition habitats by depth and by substrate type at different life history stages (Richards, 1986; Richards, 1987; Matthews, 1990a,b; Yoklavich et al., 2000; Love et al., 2002; Ingram and Shurin, 2009). Depth was an important variable in the averaged LME model for all species/groups, but the estimate was often close to zero. Fish species often do not show linear relationships with depth, but instead show a humped or skewed distribution centered on their preferred depth range (Richards, 1986). Depth was not a very important variable for predicting densities of Quillback Rockfish, Lingcod, or the combined inshore rockfishes group, as they were distributed more evenly across the depth range sampled than species where depth was important. Depth was an important predictor for Kelp Greenling, which have a shallower depth distribution and Yelloweye and Greenstriped Rockfishes that have deeper distributions (Richards, 1986; Love et al., 2009). Depth is, therefore, an important covariate in the analysis of RCAs.

Habitat features such as substrate type also influenced fish densities. Unlike many habitat characterization schemes that record the two most common substrate types observed (e.g., Love et al., 2009), we only recorded the primary substrate type. However, we also recorded the primary biological cover. Rocky substrates with encrusted or emergent organisms were the most used habitat types by our species of interest. Although this might point to important associations between fish and sessile invertebrates, the presence of certain types of organisms may also indicate the presence of boulders or more complex habitats that many rockfishes prefer (Love et al., 2002). Yelloweye Rockfish density was not associated with bare rock and the biological cover may have indicated more complex boulder habitats that they often prefer (Richards, 1986; O'Connell and Carlile, 1993). Alternatively, the relationship with

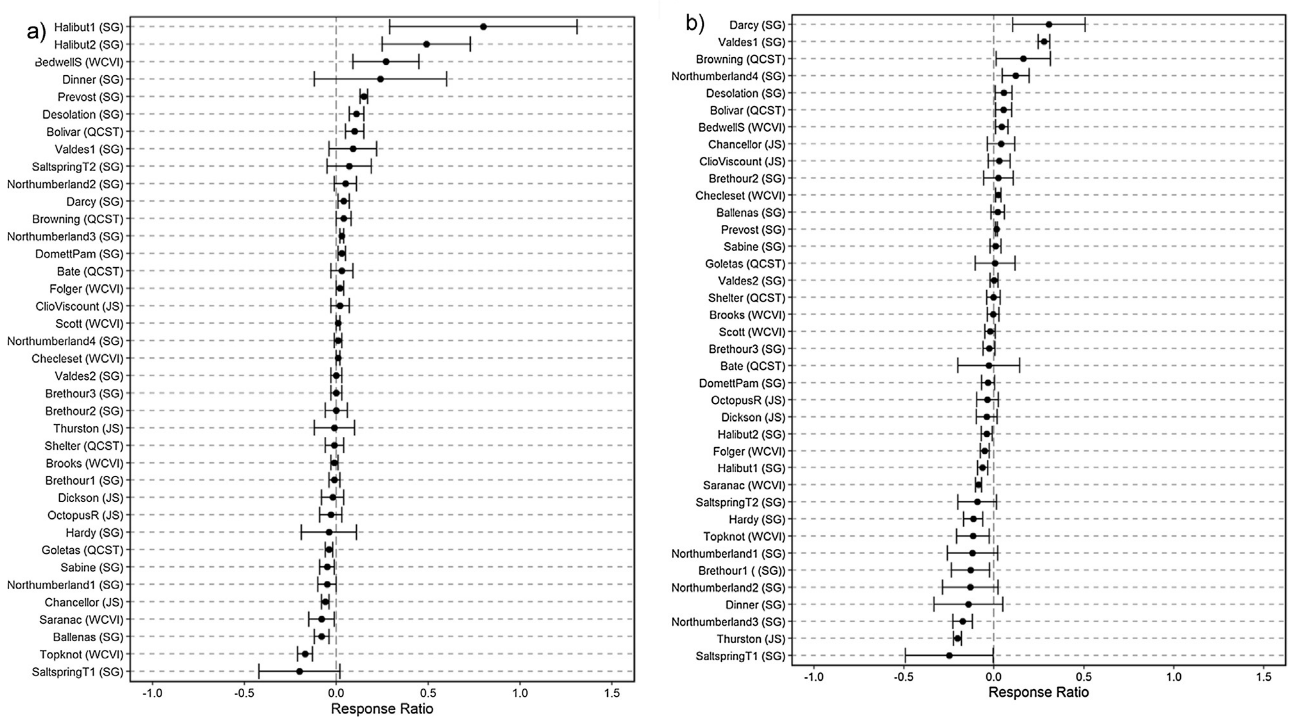

Fig. 4. The mean log Response Ratio (RR) of Quillback, Yelloweye and Lingcod density inside to outside of all RCAs sampled calculated over (a) preferred habitat and (b) total transect area. Error bars are standard errors. Ratios greater than zero indicate greater densities inside the RCA.

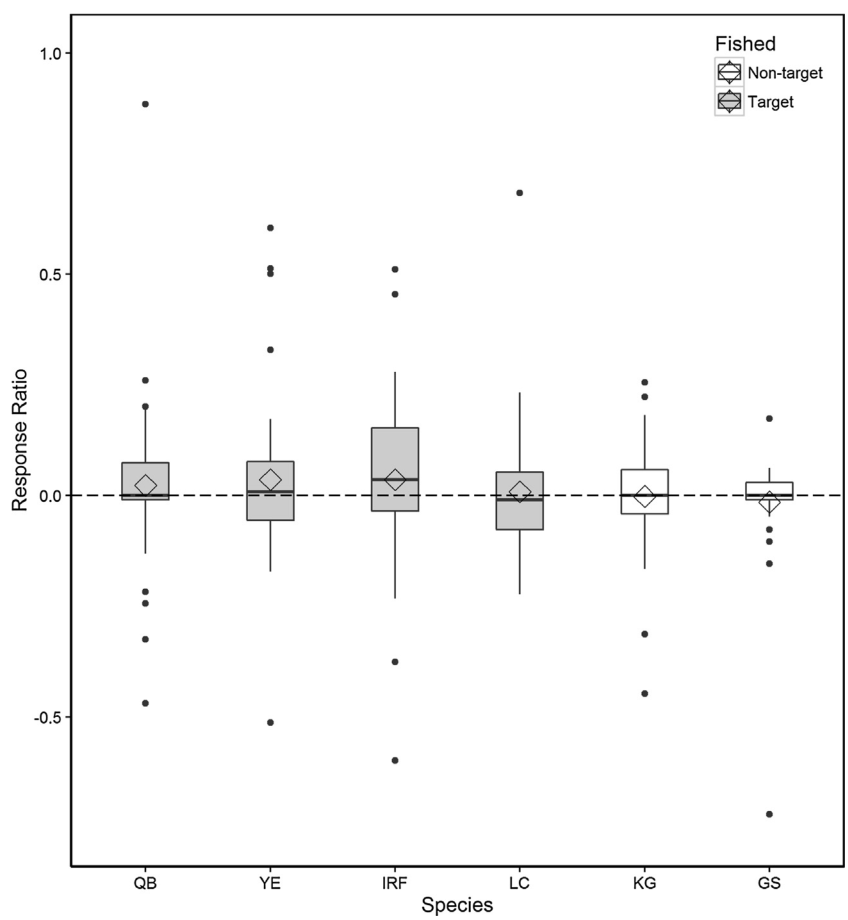

Fig. 5. Log Response Ratio (RR) for targeted and non-targeted fish species in RCAs. QB = Quillback, YE = Yelloweye, IRF = Inshore rockfishes, LC = Lingcod, KG = Kelp Greenling, GS = Greenstriped Rockfish.

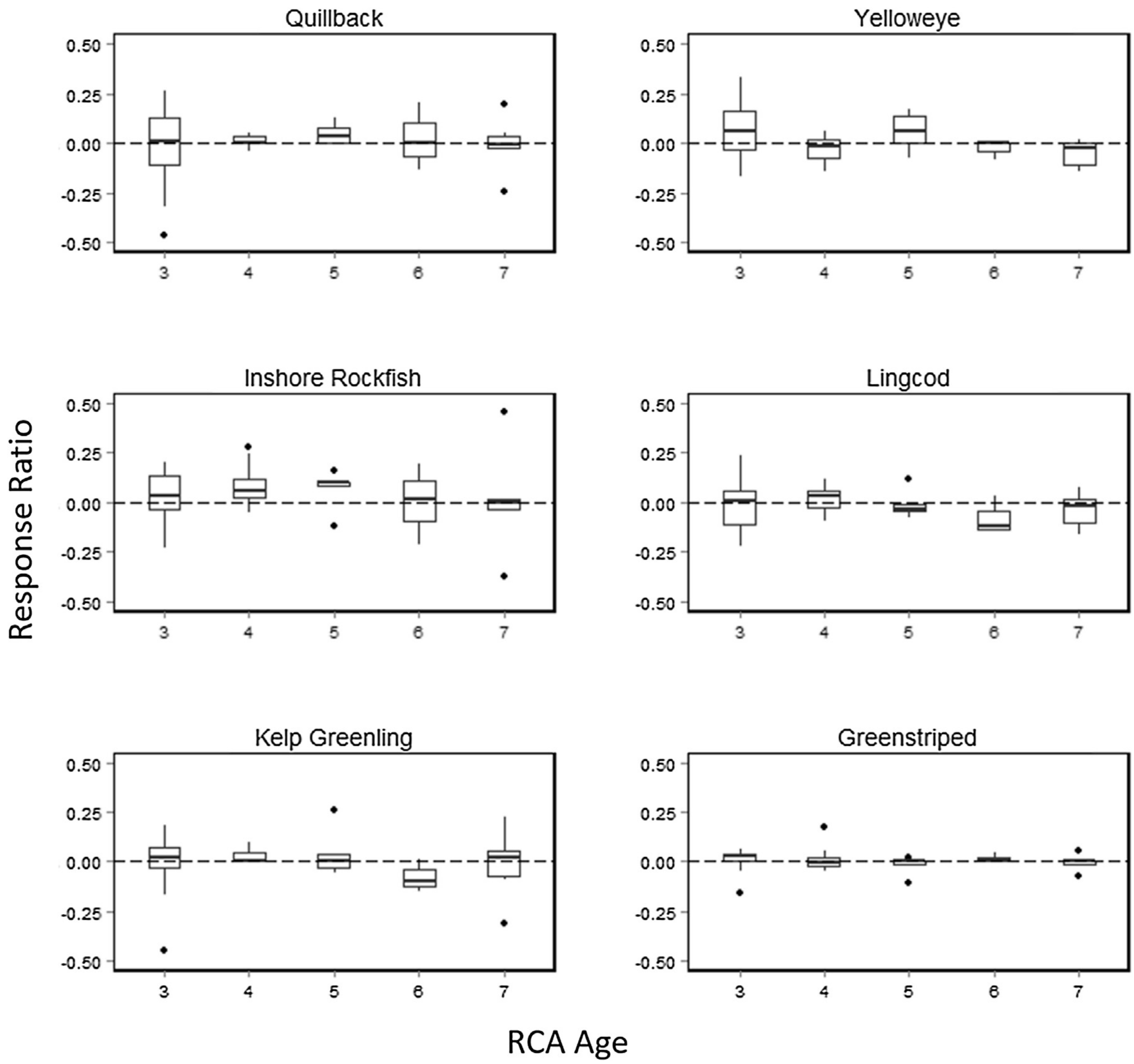

Fig. 6. Boxplots of the log Response Ratio (RR) by Species/Species Group and by the years of protection at the time of sampling.

biocover types and Yelloweye Rockfish might be a result of differences in the amount of bare rock sampled in the different regions. We sampled more bare rock in the Strait of Georgia, which also had lower Yelloweye Rockfish abundance, than in other regions. Most rocky substrates we sampled on the WCVI and the Queen Charlotte Strait, where Yelloweye Rockfish densities were greatest, were covered by invertebrates. As such, we included bare rock as a habitat type used by Yelloweye Rockfish because they were associated with bare rock in the Strait of Georgia. Encrusting biocover on mixed-coarse and fine substrates probably indicated the presence of boulders as a secondary substrate type; which could explain the importance of these habitats to Quillback Rockfish. We recommend that the secondary substrate be recorded on future surveys. We averaged habitat characteristics over the entire transect; therefore, we likely missed finer-scale habitat associations of all species. Our ROV data could also be analyzed for finer-scale use of habitat patches (e.g., Anderson and Yoklavich, 2007) including associations with particular types of biocover.

Our study highlights the importance of collecting fish habitat descriptors concurrently with abundance data in order to assess protected areas using the control-impact method to compare data from within protected areas with open areas (Claudet and Guidetti, 2010). Miller and Russ (2014) cautioned that the evaluation of MPAs must include an evaluation of habitat effects, particularly when data from before reserve implementation are not available. Despite our efforts to sample similar habitats inside and outside of reserves, our RRs changed when we accounted for fish habitat encountered along the transects. For instance, we could have concluded a positive reserve effect for the D'Arcy Island RCA; however, differences in fish abundance inside to outside the reserve were a result of better habitat sampled inside the RCA. Other studies have found that controlling for habitat altered the assessment of MPAs on fish densities (Chapman and Kramer, 1999; Miller and Russ, 2014). We sampled poorer habitat inside some RCAs. When transects that did not sample any appropriate habitat were excluded from analysis, the area of habitat sampled in some RCAs, such as Halibut Bank and Bedwell Sound, was very small. If the habitat we sampled with the ROV is representative of the entire RCA, then some RCAs contain very little high quality rockfish habitat. Although we did not exhaustively sample each RCA or randomly stratify our ROV surveys to assess habitat throughout the RCA, we did target rocky reef habitat both inside and outside of the RCA. It was easier to locate rocky habitat in some RCAs than others. Ensuring appropriate habitat for the target species is represented as one of the most important criteria for MPA effectiveness (Parnell et al., 2006). Habitat in each RCA should be thoroughly assessed. In addition to the amount and quality of habitat, habitat distribution with respect to the boundary of the RCAs should be assessed because spillover from seasonal and

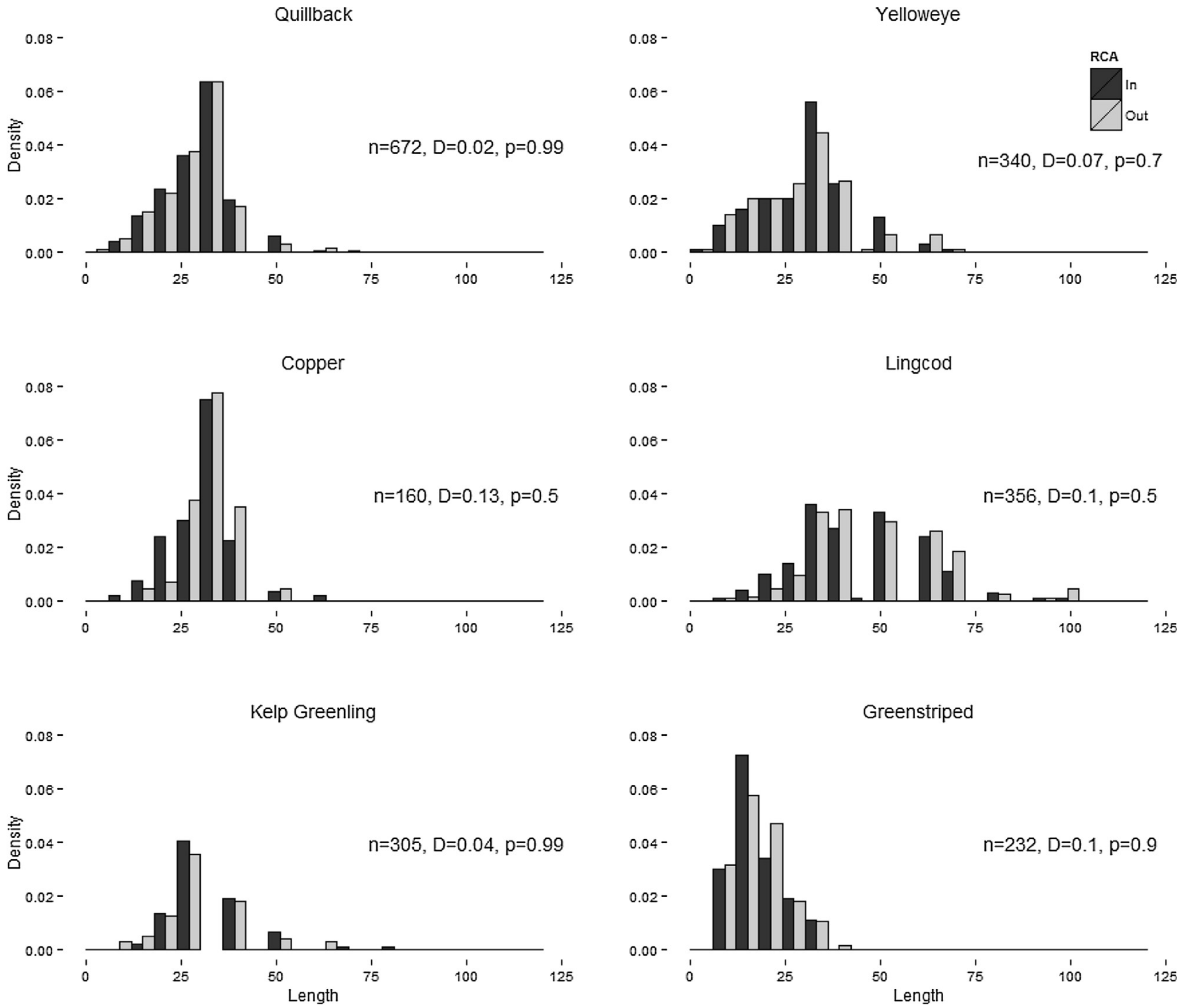

Fig. 7. Histograms of the length (cm) of fish inside and outside of the RCAs and results of the Kolmogorov-Smirnoff test. Note that the bars for the level of protection are plotted beside rather than on top of each other. There is no difference in the length frequency between fish observed inside and outside of RCAs.

day-to-day fish movements across reserve boundaries also affects reserve success (Kramer and Chapman, 1999; Babcock et al., 2010; Edgar et al., 2014).

Reserve status did not influence the density of any of the groundfish species or the inshore rockfish group collectively. Although the mean response ratio (RR) of the groundfish species we studied for some RCAs was greater than zero, on average RRs were not different from zero, and more fish were observed outside of some RCAs. Furthermore, none of the features we analyzed explained variation in RR among the RCAs or among species, although we may not have had the power to detect significant effects with a sample size of 38. Several factors may explain the lack of reserve effects (Hamilton et al., 2010; Keller et al., 2014). First, the RCAs were still relatively young when they were surveyed (three to seven years). In a global meta-analysis of MPA surveys, Babcock et al. (2010) determined time to first detection of a reserve effect on target species was 5.13 ± 1.9 years, a surprisingly short time given that many target species shared life history characteristics of longevity and slow growth. Although reserve effects were also found after only five years for some species of rockfishes in California (Hamilton et al., 2010), the rockfish species that responded quickly are relatively short lived (30–44 years) and mature early (4–5 years) (Love et al., 2002). In contrast, Quillback and Yelloweye Rockfishes live to be 95 and 115 years old and 50% maturity is not reached until 11 and 15–20 years of age, respectively (COSEWIC, 2008, 2009). Few Yelloweye Rockfish observed in this study had reached 50% maturity, indicating that recovery for this species will be particularly slow. The much longer generation time for these species as well as the sporadic recruitment success of rockfish, implies that it will take longer than 3–7 years for reserve effects to develop as a result of increased growth and reproductive output of these species in RCAs. Similarly, Starr et al. (2015) found strong reserve effects in an MPA that had been closed since 1973 but not in newer reserves in California that were sampled within the first 7 years of protection. They projected that MPAs in Central California may take 20 years or more to show significant changes in response variables (Starr et al., 2015). Most rockfishes distributed along the continental slope did not show a greater proportion of large fish in closed areas of a large RCA in the US, unlike other species with shorter lifespans (Keller et al., 2014). An ROV study of deeper waters of the Channel Islands marine reserve also failed to find significant reserve effects for many species within the first five years of protection (Karpov et al., 2012). Although the time since protection of the RCAs varied between 3 and 7 years, we did not find a trend with our RRs and the age of the RCA. The oldest reserves were found in the WCVI region, therefore the age effect was confounded with region.

Indirect effects through trophic interactions are also possible in MPAs, although they typically take longer to be detected (Babcock et al., 2010). Lingcod populations in RCAs may increase faster than rockfish populations would as a result of Lingcod's shorter lifespan (15–20 years) and earlier age at maturity (2–5 years) (Cass et al., 1990). Higher catch rates and larger Lingcod were found in reserves in the San Juan Islands that had been in place for around 20 years (Beaudreau, 2009). It has been hypothesized that increased Lingcod abundance in conservation areas would prevent rockfish from recovering as a result of increased Lingcod predation (Beaudreau and Essington, 2007; Tinus, 2012). We have no evidence that Lingcod are more abundant or larger in the RCAs. Furthermore, two studies of Lingcod diet preference have shown that rockfishes make up a small proportion of Lingcod's diet and among rockfishes, the small-bodied Puget Sound Rockfish (*S. emphaeus*) is the primary prey item (Beaudreau and Essington, 2007; Tinus, 2012). It is therefore unlikely that predation by Lingcod is hindering the recovery of rockfishes in RCAs.

Species that are directly targeted by fisheries typically respond to protection more quickly than species that are not directly targeted (Molloy et al., 2009; Babcock et al., 2010; Claudet et al., 2010; Hamilton et al., 2010). Quillback and Yelloweye Rockfishes, Lingcod and the combined inshore rockfish group are all targeted by commercial and recreational fisheries in BC. Greenstriped Rockfish and Kelp Greenling are also taken as bycatch in both commercial and recreational fisheries; however, they are typically not targeted and the catch of these species is much lower than the other groundfish species we surveyed (Fig. A.4 in Supplementary material). Although none of the species/groups mean RR differed from zero, the mean RR for Greenstriped Rockfish and Kelp Greenling fell right on zero, suggesting that the response to the RCAs was neutral, whereas the targeted species had non-zero (but very close to zero) RRs.

The intensity of commercial fishing outside of the RCAs did not influence the RR. In order to improve the assessment of marine reserves, Claudet and Guidetti (2010) recommend that the actual fishing pressure outside of an MPA be quantified, rather than assumed, when effectiveness is assessed relative to external controls. Other management measures coincident with the creation of conservation areas complicates the evaluation of RCAs (Keller et al., 2014; Starr et al., 2015). In addition to establishing RCAs, DFO also greatly reduced the directed catch (Total Allowable Catch, TAC) for inshore rockfishes. Commercial TAC was reduced by 50% for outside populations and 75% for inside populations, and recreational limits decreased from five to one fish per day on the inside and from five to three fish on the WCVI (Yamanaka and Logan, 2010). In addition to the changes associated with the Rockfish Conservation Strategy, all of the commercial groundfish fisheries in BC were integrated under the Commercial Groundfish Initiative. A guiding principle for this initiative was to account for all rockfish catch (targeted and bycatch); which was achieved by instating individual transferable quotas (ITQs) and 100% at-sea observer coverage or electronic monitoring (Davis, 2008). As a result, we measured the catch for our species of interest around each RCA and hypothesized it would greatly influence the RR. Our analysis did show that areas open to fishing adjacent to many RCAs, particularly in the Strait of Georgia, are currently not fished. However, the RR varies greatly even among RCAs with adjacent fishing pressure and the level of catch outside RCAs did not influence the RR for any of our species. Catch inside or outside the RCA from before they were established was also unrelated to RR; however, the time-period of the historical data we used (1996–2006) likely did not adequately represent the exploitation history because landings had already declined considerably by 1990 (Yamanaka and Logan, 2010). Compliance with regulations is a key feature that influences the effectiveness of MPAs (Edgar et al., 2014). Although commercial compliance is known to be high within RCAs, recreational compliance has been found to be lacking (Lancaster et al., 2015, Haggarty et al., in press). We did not consider recreational catches in this study; however, recreational non-compliance may impede population recovery inside the RCAs (Marliave and Challenger, 2009; Haggarty et al., in press).

The area and shape of MPAs are other design criteria that may affect reserve effectiveness (Halpern 2003; Gaines et al., 2010a); however empirical evidence is mixed (White et al., 2011). We found no relationship between RR and RCA size or perimeter-to-area ratio. Rockfish Conservation Areas effectiveness may be determined by a number of potentially interacting factors that may be difficult to detect by regression analyses. Identifying the factors that influence reserve success is, however, critical in order to inform adaptive management to improve the effectiveness of future actions (White et al., 2011). A thorough analysis of the habitat quality, habitat isolation, compliance and other key features of the entire RCA network are badly needed. Continued monitoring of the RCAs is essential given the demographic time-lag associated with long-lived rockfishes. Our study shows that ROV surveys are an effective way to monitor RCAs and our data can be used to represent the initial conditions of the RCAs and be compared to in the future. At present, we see little indication of population recovery inside RCAs.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.fishres.2016.06.001.

References

Alcala, A.C., Russ, G.R., Maypa, A.P., Calumpong, H.P., 2005. A long-term: spatially replicated experimental test of the effect of marine reserves on local fish yields. Can. J. Fish. Aquat. Sci. 62, 98–108.

Allison, G.W., Lubchenco, J., Carr, M.H., 1998. Marine reserves are necessary but not sufficient for marine conservation. Ecol. Appl. 8, S79–S92.

Anderson, T.J., Yoklavich, M.M., 2007. Multiscale habitat associations of deepwater dernersal fishes off central California. Fish. Bull. 105, 168–179.

Babcock, R.C., Shears, N.T., Alcala, A.C., Barrett, N.S., Edgar, G.J., Lafferty, K.D., McClanahan, T.R., Russ, G.R., 2010. Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. Proc. Natl. Acad. Sci. U. S. A. 107, 18256–18261.

Barton, K., 2013. MuMIn: Multi-model Inference. R Package Version 1.9.11. http://CRAN.R-project.org/package=MuMIn.

Beaudreau, A.H., Essington, T.E., 2007. Spatial, temporal, and ontogenetic patterns of predation on rockfishes by lingcod. Trans. Am. Fish. Soc. 136, 1438–1452.

Beaudreau, A.H., 2009. The Predatory Role of Lingcod (*Ophiodon elongatus*) in the San Juan Archipelago, Ph.D. Dissertation. School of Aquatic and Fishery Sciences, University of Washington, Seattle.

COSEWIC, 2008. COSEWIC assessment and status report on the Yelloweye Rockfish *Sebastes ruberrimus*, Pacific Ocean inside waters population and Pacific Ocean outside waters population, in Canada. Ottawa, Committee on the Status of Endangered Wildlife in Canada.

COSEWIC, 2009. COSEWIC assessment and status report on the Quillback Rockfish *Sebastes maliger* in Canada. Ottawa, Committee on the Status of Endangered Wildlife in Canada.

Cass, A.J., Beamish, R.J., McFarlane, G.A., 1990. Lingcod (*Ophiodon elongatus*). Can. Spec. Publ. Fish. Aquat. Sci., 109.

Chapman, M.R., Kramer, D.L., 1999. Gradients in coral reef fish density and size across the Barbados Marine Reserve boundary: effects of reserve protection and habitat characteristics. Mar. Ecol. Prog. Ser 181, 81–96.

Claudet, J., Guidetti, P., 2010. Improving assessments of marine protected areas. Aquat. Conserv. Mar. Freshw. Ecosyst. 20, 239–242.

Claudet, J., Osenberg, C.W., Benedetti-Cecchi, L., Domenici, P., Garcia-Charton, J.A., Perez-Ruzafa, A., Badalamenti, F., Bayle-Sempere, J., Brito, A., Bulleri, F., Culioli, J.M., Dimech, M., Falcon, J.M., Guala, I., Milazzo, M., Sanchez-Meca, J., Somerfield, P.J., Stobart, B., Vandeperre, F., Valle, C., Planes, S., 2008. Marine reserves: size and age do matter. Ecol. Lett. 11, 481–489.

Claudet, J., Osenberg, C.W., Domenici, P., Badalamenti, F., Milazzo, M., Falcon, J.M., Bertocci, I., Benedetti-Cecchi, L., Garcia-Charton, J.A., Goni, R., Borg, J.A., Forcada, A., de Lucia, G.A., Perez-Ruzafa, A., Afonso, P., Brito, A., Guala, I., Le Direach, L., Sanchez-Jerez, P., Somerfield, P.J., Planes, S., 2010. Marine reserves: fish life history and ecological traits matter. Ecol. Appl. 20, 830–839.

Cloutier, R.N., 2011. Direct and Indirect Effects of Marine Protection: Rockfish Conservation Areas as a Case Study. Department of Biological Sciences, Simon Fraser University.

Cook, S.E., Conway, K.W., Burd, B., 2008. Status of the glass sponge reefs in the Georgia Basin. Mar. Environ. Res. 66, S80–S86.

Cote, I.M., Mosqueira, I., Reynolds, J.D., 2001. Effects of marine reserve characteristics on the protection of fish populations: a meta-analysis. J. Fish Biol. 59, 178–189.

Davis, N.A., 2008. Evaluating collaborative fisheries management planning: a Canadian case study. Mar. Policy 32, 867–876.

Dulvy, N.K., Sadovy, Y., Reynolds, J.D., 2003. Extinction vulnerability in marine populations. Fish Fish. 4, 25–64.

Dulvy, N.K., Ellis, J.R., Goodwin, N.B., Grant, A., Reynolds, J.D., Jennings, S., 2004. Methods of assessing extinction risk in marine fishes. Fish Fish. 5, 255–276.

Edgar, G.J., Stuart-Smith, R.D., Willis, T.J., Kininmonth, S., Baker, S.C., Banks, S., Barrett, N.S., Becerro, M.A., Bernard, A.T.F., Berkhout, J., Buxton, C.D., Campbell, S.J., Cooper, A.T., Davey, M., Edgar, S.C., Forsterra, G., Galvan, D.E., Irigoyen, A.J., Kushner, D.J., Moura, R., Parnell, P.E., Shears, N.T., Soler, G., Strain, E.M.A., Thomson, R.J., 2014. Global conservation outcomes depend on marine protected areas with five key features. Nature 506, 216–220.

Field, J.C., Punt, A.E., Methot, R.D., Thomson, C.J., 2006. Does MPA mean 'major problem for assessments'? Considering the consequences of place-based management systems. Fish Fish. 7, 284–302.

Fox, J., Weisberg, S., 2011. An {R} Companion to Applied Regression, second edition. Sage, Thousand Oaks, CA.

Gaines, S., White, C., Carr, M., Palumbi, S.R., 2010a. Designing marine reserve networks for both conservation and fisheries management. Proc. Natl. Acad. Sci. U. S. A. 107, 18286–18293.

Gaines, S.D., Lester, S.E., Grorud-Colvert, K., Costello, C., Pollnac, R., 2010b. Evolving science of marine reserves: new developments and emerging research frontiers. Proc. Natl. Acad. Sci. U. S. A. 107, 18251–18255.

Glasby, T.M., 1997. Analysing data hom post-impact studies using asymmetrical analyses of variance: a case study of epibiota on marinas. Aust. J. Ecol. 22, 448–459.

Haggarty, D.R., Martell, S.J.D., Shurin, J.B., 2016. Lack of recreational fishing compliance may compromise effectiveness of Rockfish Conservation Areas in British Columbia. Can. J. Fish. Aquat. Sci., http://dx.doi.org/10.1139/cjfas-2015-0205.

Halpern, B.S., 2003. The impact of marine reserves: do reserves work and does reserve size matter? Ecol. Appl. 13, 117–137.

Hamilton, S.L., Caselle, J.E., Malone, D.P., Carr, M.H., 2010. Incorporating biogeography into evaluations of the Channel Islands marine reserve network. Proc. Natl. Acad. Sci. U. S. A. 107, 18272–18277.

Hamilton, S.L., Caselle, J.E., Lantz, C.A., Egloff, T.L., Kondo, E., Newsome, S.D., Loke-Smith, K., Pondella, D.J., Young, K.A., Lowe, C.G., 2011. Extensive geographic and ontogenetic variation characterizes the trophic ecology of a temperate reef fish on southern California (USA) rocky reefs. Mar. Ecol. Prog. Ser. 429, 227–244.

Hedges, L.V., Gurevitch, J., Curtis, P.S., 1999. The meta-analysis of response ratios in experimental ecology. Ecology 80, 1150–1156.

Hutchings, J.A., Reynolds, J.D., 2004. Marine fish population collapses: consequences for recovery and extinction risk. Bioscience 54, 297–309.

Hutchings, J.A., 2001. Conservation biology of marine fishes: perceptions and caveats regarding assignment of extinction risk. Can. J. Fish. Aquat. Sci. 58, 108–121.

Ingram, T., Shurin, J.B., 2009. Trait-based assembly and phylogenetic structure in Northeast Pacific rockfish assemblages. Ecology 90, 2444–2453.

Karpov, K., Bergen, M., Geibel, J., 2012. Monitoring fish in California Channel Islands marine protected areas with a remotely operated vehicle: the first five years. Mar. Ecol. Prog. Ser. 453, 159–172.

Keller, A., Wakefield, W., Whitmire, C., Horness, B., Bellman, M., Bosley, K., 2014. Distribution of demersal fishes along the US west coast (Canada to Mexico) in relation to spatial fishing closures (2003–2011). Mar. Ecol. Prog. Ser. 501, 169–190.

Kramer, D.L., Chapman, M.R., 1999. Implications of fish home range size and relocation for marine reserve function. Environ. Biol. Fishes 55, 65–79.

Kuznetsova, A., Bruun Brockhoff, P., Haubo Bojesen Christensen, R., 2014. lmerTest: Tests in Linear Mixed Effects Models R package version 2. 0–20. https://cran.r-project.org/web/packages/lmerTest/index.html.

Laidig, T.E., Watters, D.L., Yoklavich, M.M., 2009. Demersal fish and habitat associations from visual surveys on the central California shelf. Estuar. Coast. Shelf Sci. 83, 629–637.

Lancaster, D., Dearden, P., Ban, N.C., 2015. Drivers of recreational fisher compliance in temperate marine conservation areas: a study of RCAs in British Columbia, Canada. Global Ecol. Conserv., 645–657.

Love, M., Yoklavich, M., Thorsteinson, L., 2002. The Rockfishes of the Northeast Pacific. University of California Press, Los Angeles.

Love, M., Yoklavich, M., Schroeder, D., 2009. Demersal fish assemblages in the Southern California Bight based on visual surveys in deep water. Environ. Biol. Fishes 84, 55–68.

Marliave, J., Challenger, W., 2009. Monitoring and evaluating rockfish conservation areas in British Columbia. Can. J. Fish. Aquat. Sci. 66, 995–1006.

Marliave, J.B., Conway, K.W., Gibbs, D.M., Lamb, A., Gibbs, C., 2009. Biodiversity and rockfish recruitment in sponge gardens and bioherms of southern British Columbia, Canada. Mar. Biol. 156, 2247–2254.

Matthews, K.R., 1990a. A comparative-study of habitat use by young-of-the-year subadult, and adult rockfishes on 4 habitat types in Central Puget Sound. Fish. Bull. 88, 223–239.

Matthews, K.R., 1990b. An experimental study of the habitat preferences and movement patterns for copper, quillback, and brown rockfishes (*Sebastes* spp.). Environ. Biol. Fishes 29, 161–178.

Miller, K.I., Russ, G.R., 2014. Studies of no-take marine reserves: methods for differentiating reserve and habitat effects. Ocean Coast. Manage. 96, 51–60.

Molloy, P.P., McLean, I.B., Cote, I.M., 2009. Effects of marine reserve age on fish populations: a global meta-analysis. J. Appl. Ecol. 46, 743–751.

Mosqueira, I., Cote, I.M., Jennings, S., Reynolds, J.D., 2000. Conservation benefits of marine reserves for fish populations. Anim. Conserv. 3, 321–332.

National Research Council, 2006. Dynamic Changes in Marine Ecosystems: Fishing, Food Webs, and Future Options. National Academies Press, Washington (DC).

O'Connell, V.M., Carlile, D.W., 1993. Habitat-specific density of adult yelloweye rockfish *Sebastes-ruberrimus* in the Eastern Gulf of Alaska. Fish. Bull. 91, 304–309.

O'Farrell, M.R., Yoklavich, M.M., Love, M.S., 2009. Assessment of habitat and predator effects on dwarf rockfishes (*Sebastes* spp.) using multi model inference. Environ. Biol. Fishes 85, 239–250.

Paddack, M.J., Estes, J.A., 2000. Kelp forest fish populations in marine reserves and adjacent exploited areas of central California. Ecol. Appl. 10, 855–870.

Parker, S.J., Berkeley, S.A., Golden, J.T., Gunderson, D.R., Heifetz, J., Hixon, M.A., Larson, R., Leaman, B.M., Love, M.S., Musick, J.A., O'Connell, V.M., Ralston, S., Weeks, H.J., Yoklavich, M.M., 2000. Management of pacific rockfish. Fisheries 25, 22–30.

Parnell, P.E., Dayton, P.K., Lennert-Cody, C.E., Rasmussen, L.L., Leichter, J.J., 2006. Marine reserve design optimal size, habitats, species affinities, diversity, and ocean microclimate. Ecol. Appl. 16, 945–962.

Pelletier, D., Claudet, J., Ferraris, J., Benedetti-Cecchi, L., Garcia-Charton, J.A., 2008. Models and indicators for assessing conservation and fisheries-related effects of marine protected areas. Can. J. Fish. Aquat. Sci. 65, 765–779.

R Development Core Team, 2008. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Richards, L.J., 1986. Depth and habitat distributions of 3 species of rockfish (*Sebastes*) in British-Columbia—observations from the submersible Pisces-iv. Environ. Biol. Fishes 17, 13–21.

Richards, L.J., 1987. Copper rockfish (*Sebastes-caurinus*) and quillback rockfish (*Sebastes-maliger*) habitat in the Strait of Georgia, British-Columbia. Can. J. Zool. 65, 3188–3191.

Robb, C.K., Bodtker, K.M., Wright, K., Lash, J., 2011. Commercial fisheries closures in marine protected areas on Canada's Pacific coast The exception, not the rule. Mar. Policy 35, 309–316.

Russ, G.R., Stockwell, B., Alcala, A.C., 2005. Inferring versus measuring rates of recovery in no-take marine reserves. Mar. Ecol. Prog. Ser. 292, 1–12.

Starr, R.M., Wendt, D.E., Barnes, C.L., Marks, C.I., Malone, D., Waltz, G., Schmidt, K.T., Chiu, J., Launer, A.L., Hall, N.C., Yochum, N., 2015. Variation in responses of fishes across multiple reserves within a network of marine protected areas in temperate waters. PLoS One 10, e0118502.

Stoner, A.W., Ryer, C.H., Parker, S.J., Auster, P.J., Wakefield, W.W., 2008. Evaluating the role of fish behavior in surveys conducted with underwater vehicles. Can. J. Fish. Aquat. Sci. 65, 1230–1243.

Thomson, R.E., 1981. Oceanography of the British Columbia coast. Can. Spec. Publ. Fish. Aquat. Sci., 56.

Tinus, C.A., 2012. Prey preference of lingcod (Ophiodon elongatus): a top marine predator: implications for ecosystem-based fisheries management. Fish. Bull. 110, 193–204.

Underwood, A.J., 1992. Beyond BACI—the detection of environmental impacts on populations in the real, but variable, world. J. Exp. Mar. Biol. Ecol. 161, 145–178.

White, J.W., Botsford, L.W., Baskett, M.L., Barnett, L.A.K., Barr, R.J., Hastings, A., 2011. Linking models with monitoring data for assessing performance of no-take marine reserves. Front. Ecol. Environ. 9, 390–399.

Yamanaka, K.L., Logan, G., 2010. Developing British Columbia's inshore rockfish conservation strategy. Mar. Coast. Fish. 2, 28–46.

Yoklavich, M.M., Greene, H.G., Cailliet, G.M., Sullivan, D.E., Lea, R.N., Love, M.S., 2000. Habitat associations of deep-water rockfishes in a submarine canyon: an example of a natural refuge. Fish. Bull. 98, 625–641.

Yoklavich, M., Cailliet, G., Lea, R.N., Greene, H.G., Starr, R., De Marignac, J., Field, J., 2002. Deepwater habitat and fish resources associated with the Big Creek Marine Ecological Reserve. Calif. Coop. Ocean. Fish. 43, 120–140.
Yoklavich, M., 1998. In: Yoklavich, M. (Ed.), Marine Harvest Refugia for West Coast Rockfish: A Workshop. NOAA Technical Memorandum NMFS, Pacific Grove, California.
Zar, J.H., 1996. Biostatistical Analysis. Prentice Hall, Upper Saddle River, New Jersey.
Zurr, A.F., Leno, E.N., Walker, N., Savaliev, A.A., Smith, G.M., 2009. Mixed Effects Models and Extensions in Ecology with R. Springer, New York.