Can fishing gear protect non-target fish? Design and evaluation of bycatch reduction technology for commercial fisheries

by

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B.Sc., Simon Fraser University, 2008

Thesis Submitted In Partial Fulfillment of the
Requirements for the Degree of
Doctor of Philosophy

in the
Department of Biological Sciences
Faculty of Science

SIMON FRASER UNIVERSITY

Summer 2013

Approval

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Abstract

The impacts of commercial fishing extend far beyond direct effects on targeted species. As much as 40% of global marine catch is attributable to bycatch, or the capture of non-target organisms which occurs during fishing. The amount of bycatch in a fishery is determined in part by the selectivity of the industry's fishing gear, and bycatch mitigation often focuses on improving the selectivity of these gears. This thesis explores bycatch mitigation through the design and evaluation of bycatch reduction devices (BRDs), or fishing gear modifications aimed specifically at reducing non-target catch while maintaining the catch of target species. I examine BRDs using a three-pronged assessment, which tests a modified gear's effects on non-target catch, on target catch, and on practicality for use in commercial fisheries (all relative to unmodified gear). I first perform a global-scale meta-analysis on technologies designed to protect elasmobranchs (sharks and rays) from longline fisheries. I show that most technologies are broadly ineffective at reducing elasmobranch bycatch, and that many studies fail to adequately assess novel BRDs across all three dimensions of gear performance. The remainder of my thesis focuses on the research and development of BRDs for a British Columbia fishery which employs trapping gear to capture spot prawns (*Pandalus platyceros*). Using data from fishery-independent surveys, I show that these traps catch rockfish (*Sebastes* spp.) as bycatch, a multi-species genus which is depleted due to overfishing and which suffers high discard mortality due to barotrauma incurred during the fishing process. I demonstrate that a novel underwater camera system can be used to study prawn traps *in situ,* and use insights from this analysis to inform the design of BRDs for prawn traps. I conclude my thesis with an assessment of BRDs of my own design, using both catch data and *in situ* observations conducted using my underwater camera apparatus. Overall, this thesis demonstrates the challenges in designing effective BRDs, and provides a framework for assessment that can be used as a template in future studies of fishing gear design.

Keywords: Bycatch reduction device; fisheries; gear assessment; rockfish; spot prawn; elasmobranch

Acknowledgements

This dissertation is the most complex, challenging, and rewarding project that I have ever tackled. It would not have been possible without the contributions of a great network of supporters. I want to begin by thanking my senior supervisor Isabelle Côté for teaching me how to be an academic scientist. Her guidance, instruction, and mentoring have been invaluable in shaping my growth over the last several years. I also want to thank Stefanie Duff for playing a critical role in bringing together the people necessary to make this project a success, and for making unique opportunities available along the way. Finally I would like to thank John Reynolds for serving as a committee member and being consistently supportive throughout the project. I also thank Nicholas Dulvy, Arne Mooers, Anne Salomon, Andrew Cooper, and Wendy Palen for being open and available for useful discussions during my time as a PhD student.

A great many volunteers and assistants played critical roles in bringing this project together. Vin Fogarty contributed greatly in the field, and his reliability and professionalism was critical to this project's success. The Prawn Volunteers (J Chan, R Charish, M Cornall, T Dancau, A Fortune, C Hemsworth, L Hurteau, A Lam, B Li, E Lo, L McArthur, C McCune, K Pailis, K Randhawa, M Ray, A Shariff, A Thomas, B Vallis, and R Van Der Merwe) scored much of my underwater video, and I thank them for it. Jonathan Martin kept *CJ Walters* running smoothly, and I thank him for that as well.

Dennis Rutherford, Mike Kattilakoski, and Laurie Convey at Fisheries and Oceans Canada were extremely supportive of this work, and I greatly appreciate their input and contributions of data. Chris Sporer of the Pacific Prawn Fishers Association provided crucial input and encouragement, and it was a privilege to work with this industry. Paul Bevandick, Stewart MacDonald, and Steve Johansen welcomed me onto their fishing vessels to collect data and learn how to safely fish for prawns. I also thank the many other fishers who asked not to be named, but provided similar support.

I could not have made it through this process without the support and encouragement of friends and lab-mates. Stephanie Green, Doug Braun, Brendan Connors, Joel Harding, Jennifer Linton, Emily Darling, Michael Beakes, Corey Phillis, Sean Anderson, Evan Henderson, Justin Suraci, Matt Huntley, and Jordan Thompson were tremendously helpful and supportive throughout. I thank all members of the Côté lab, past and present, as well as the Earth to Ocean Research Group for providing a fun and stimulating environment.

I thank the Pacific Prawn Fishers Association, Mitacs, NSERC, The PADI Foundation, the BC Ocean and Marine Fisheries Branch, the Garfield Weston Graduate Fellowship, and the J. Abbott/M. Fretwell Fellowship for funding me and my research. I also thank the Australia Endeavour program and Hugh Possingham for supporting my travel to the University of Queensland, which contributed greatly to my intellectual growth.

I also thank my family and particularly my parents, Jim and Karen Favaro, for directly contributing to the project through use of their vehicles, storage space, and assistance in the field. They encouraged me to pursue graduate education, and supported me in countless ways.

Finally I want to express my fullest gratitude and appreciation to Corinna Lichota, who has spanned all the categories I have listed above. Corinna contributed academically, as a deckhand, video analyst, and co-author. Her love and support has been a constant source of inspiration, and I could not have done it without her.

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1. General Introduction

The extraction of biomass from the ocean through fishing is one of the major threats to marine ecosystems across the globe (Myers and Worm 2005). Between capture fisheries and aquaculture, approximately 148 million tonnes of seafood were produced in 2010 (FAO 2012). The overwhelming majority (87%) of fish stocks are fully exploited, overexploited, or depleted, and this proportion grows larger with each global assessment (FAO 2010, 2012). The status of marine ecosystems appears even more precarious if one compares current abundance to historical levels, rather than restricting the comparison to high-quality data recorded in the context of modern commercial fishing (Pauly 1995, Baum and Myers 2004, Pinnegar and Engelhard 2008). As human populations continue to increase this trend is unlikely to reverse, creating an ongoing challenge for managers and conservationists to maintain marine ecosystems that can sustain biodiversity and the provision of valuable goods and services.

The impact of commercial fisheries is linked to the technology used (Walsh et al. 2002a, Chuenpagdee et al. 2003). In addition to increasing catch per unit effort (Garcia and Newton 1995, Pauly et al. 2002), ongoing technological developments have extended the reach of fisheries, enabling fishers to access depths and spatial ranges previously unavailable to exploitation (Morato et al. 2006, Swartz et al. 2010). These innovations shared a common goal; they were all designed to increase the volume of organisms extracted from the oceans.

However, as capture efficiency has increased, so too has the amount of non-target species incidentally caught during the fishing process. This unintentional catch of non-target species which occurs during fishing is called “bycatch.” Bycatch can be defined as catch which is unused or unmanaged (Davies et al. 2009), and is composed of all non-target organisms caught in fishing gear. Bycatch can be retained or thrown back to the ocean – a behaviour known as discarding (Kelleher 2005). Bycatch is most likely to cause concern when target and non-target species possess different life histories, such that the non-target species is unable to sustain the fishing pressure exerted by the fishery for the target species (Hall et al. 2000). Globally, bycatch has been estimated to comprise as much as 40% of all catch (Davies et al. 2009).

Bycatch negatively impacts marine ecosystems and fisheries in several ways. Firstly, it can serve as a cause (sometimes the primary cause) of depletions of marine species (Weimerskirch et al. 1987, Hillestad et al. 1995, Hall 1998, Barbraud et al. 2012). Second, it can create a barrier to exploitation of robust stocks, as regulations require the cessation of fishing when quotas of bycatch species are reached (Squires et al. 1998, Abbott and Wilen 2009). In the US alone, the cost of these early closures have been estimated to reach over $6 billion annually (Patrick and Benaka 2013). Thirdly, it can reduce the profitability of fishing by increasing sorting times (Gjertsen et al. 2010), damaging gear (Mandelman et al. 2008), and by dissuading the public from purchasing the targeted product (Brown 2005). The ecological costs are substantial; bycatch has been implicated as a major threat to populations of sea birds (Lewison et al. 2005, Dillingham and Fletcher 2008, Watkins et al. 2008), sharks and rays (Ward et al. 2008), cetaceans (Read 2008), turtles (Wallace et al. 2008), and non-target fish stocks (Brown 2005).

There is no shortage of potential solutions to bycatch problems. Improvements in a fisheries' bycatch profile can be accomplished by fishing less, by managing and making use of non-target species caught in fishing gear, or by improving the selectivity of fishing gear (Kelleher 2005, Hall et al. 2007). Bycatch quotas, area closures, partial closures based on gear type, and reductions in fishing effort have all been applied to manage or mitigate non-target catch (Witherell and Pautzke 1997, Clark and Hare 1998, Roberts et al. 2005, Sanchirico et al. 2006). While traditional fisheries management has been based on single-species quotas, shifts towards transferrable quotas and multispecies management have reduced incentives to discard non-target species, thereby refocusing management objectives towards managing and making use of all catch rather than reducing the capture of non-target species (Branch et al. 2006).

However, when charismatic and/or threatened species are taken as bycatch, and when their ranges overlap with those of target species, regulatory reform alone may not suffice. In these cases, new fishing techniques must be invented to prevent the capture or promote the escapement of non-target species from fishing gear. Modifications to fishing gear which are primarily designed to reduce the capture of non-target species are called Bycatch Reduction Devices (BRDs) (FAO 2002). These devices are highly varied, and have been produced for many types of commercial fishing gear. BRDs can act on any part of the capture process, repelling organisms from fishing gear (Kaimmer and Stoner 2008, Stephenson and Wells 2008), preventing entry of non-target species into the fishing gear (Fonseca et al. 2005, Catchpole and Revill 2008, Stephenson et al. 2008), or promoting escapement of caught individuals (Ward et al. 2008, Johnson 2010). Global analyses suggest that if fisheries around the world employed BRD technology, and if the results were comparable to results achieved in experimental tests of the devices, then bycatch rates could be reduced by 25-64% worldwide (Hall and Mainprize 2005).

The potential for BRDs to solve global bycatch problems is controversial. While fishers and managers are often enthusiastic about these technologies (Campbell and Cornwell 2008, Molina and Cooke 2012), others decry them as akin to “treating a serious illness with aspirin” (Damanaki 2011), ignoring larger systemic problems with overfishing and with policies that inadvertently encourage discards. Nevertheless, fishers and managers often favour the development of BRD technology as it is one of the few potential conservation measures that does not require a reduction in fishing effort (Cox et al. 2007).

Effective BRDs must meet three main criteria. First, the device must achieve a goal for bycatch reduction. This goal could be to reduce bycatch by weight, reduce the capture of one or more particular species, or any other management goal which focuses on improving the specificity of gear. Second, the device must achieve a goal for target catch. In some cases, any reduction in target catch may be unacceptable. In other cases, a reduction in target catch may be tolerable to fishers if the motivation to reduce bycatch is strong enough. In addition, the quality, condition, and composition of the target catch must be considered to ensure that the product captured by the new gear is comparable to standard unmodified gear. Third, the device must be practical for use in the fishery, meaning that the procurement cost must be reasonable, the gear should be safe and easy to use, and it should be durable enough to be used in a commercial fishery. If a gear fails on any of these three criteria, it will be unlikely to succeed as a tool for protecting non-target species from fishing activities.

In this thesis, I assess novel BRDs under consideration for use in commercial fisheries using the three criteria above. First, I examine gear designed by other inventors and review the performance of these technologies in terms of bycatch reduction, target catch maintenance, and real-world applicability. Second, I embark on my own process of research and development of bycatch reduction devices for a local commercial fishery, and employ a rigorous analysis to assess my devices' ability to protect non-target species from the activities of that fishery.

In Chapter 2, I explore the technologies employed to mitigate the global problem of bycatch of elasmobranchs (sharks and rays) on longline gear (Molina and Cooke 2012). Longline gear is among the most popular technologies for catching large-bodied pelagic fish species, and elasmobranchs caught on longline hooks suffer high mortality rates when released (Beerkircher et al. 2002, Diaz and Serafy 2005). A wide variety of bycatch reduction technologies have been invented and tested with the goal of reducing this source of mortality. While these devices often generate substantial media attention for their apparent ability to reduce shark interactions with fishing gear (e.g. Shapiro 2012), until now there has been no comprehensive, quantitative assessment comparing the relative effectiveness of different families of BRD at reducing elasmobranch bycatch. In this chapter, I conduct a meta-analysis of existing elasmobranch BRD literature to assess the performance of these devices. I evaluate the various gears on three criteria: their effect on non-target catch, their effect on target catch, and their practical applicability for use in the fishery.

The remainder of my thesis focuses on the BC spot prawn trap fishery, a large-scale commercial fishery which occurs across the Pacific coast of Canada. Prawn trapping has been endorsed by conservation organizations due to its apparent sustainability and low bycatch by weight, especially relative to trawl-caught crustacean fisheries (Roberts 2005). Nevertheless, in-season monitoring within the prawn fishery showed that juvenile rockfish (*Sebastes* spp.) are occurring as bycatch in the fishery (DFO 2009). This is cause for concern, as rockfish have experienced decades of overfishing and suffer high discard mortality due to barotrauma which occurs during the gear retrieval process (Hannah and Matteson 2007, Yamanaka and Logan 2010). In addition, prawn trapping is a permitted activity within Rockfish Conservation Areas (RCAs), partial closure areas which prevent activities that harm rockfish (Fisheries and Oceans Canada 2007). Chapter 3 represents the first step in assessing and solving the rockfish bycatch problem in the BC prawn trap fishery. In this chapter, I gather available data on bycatch in prawn traps, and synthesize them to better understand the composition of catch and the scope of the bycatch problem. I use a previously unstudied database provided by Fisheries and Oceans Canada (DFO) to analyze the composition of catch in research traps deployed in a fishery-independent survey from 1999-2008. This chapter confirms that rockfish bycatch occurs in prawn traps, and identifies quillback rockfish as the most commonly caught fish species. This species is listed as threatened by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC 2009). These findings establish a motivation to develop bycatch mitigation options for the BC prawn fishery.

The prawn fishery is managed, but not by means of quotas or other regulatory tools that could be used for bycatch management (Boutillier and Bond 2000). Furthermore, while the majority of the fleet receives at least one visit per season by an at-sea observer, a small proportion of the catch is actually inspected by observers (Fisheries and Oceans Canada 2011). Rockfish bycatch is therefore primarily reported by uninspected logbooks, which can underestimate the actual amount of bycatch taken (Walsh et al. 2002b, Babcock et al. 2003, Burns and Kerr 2008). Hence, I elected to develop BRDs whose use would not require major changes to the management regime of the fishery or increased observer coverage to confirm quotas. However, little was known about the *in situ* interactions between rockfish, prawns and traps occurring at depth, which could inform the development of bycatch reduction technology. Testing of trap designs in the laboratory was unlikely to replicate the conditions occurring at the conventional prawn trapping depths of 100 m or more. Chapter 4 describes my solution to this problem – the development of a deep-water camera apparatus constructed to allow the observation of prawn trap deployments *in situ*. This chapter explores the technology I used and compares it against other tools used by researchers to date to study deep-water systems.

Using the insights I gained from the results in Chapter 4, and from interacting with fishers in the field, I designed a series of BRDs which were built to be retrofitted onto existing traps. My field-test of these gears is reported in Chapter 5. I designed a suite of prototype BRDs and deployed them in the field, under real-world conditions. I found that the devices substantially reduced bycatch, but also reduced target catch. I also found that the gears were robust and practical for use in the fishery, notwithstanding their performance at retaining target catch. Analysis of *in situ* videos gave us insights about the mechanism underpinning the catch differences between conventional and BRD-equipped traps.

In Chapter 6, I review my findings in a broad global context, and consider how the results of this thesis can inform the research, development, and application of bycatch reduction technology in commercial fisheries.

2. Do bycatch reduction devices in longline fisheries reduce capture of sharks and rays? A global meta-analysis[1]

2.1. Abstract

Bycatch in marine fisheries, particularly those using pelagic and demersal longlines, is a major driver of declines in abundance of sharks and rays around the world. A wide variety of bycatch reduction devices (BRDs), i.e. modified gears designed to reduce incidental captures of a variety of marine species while maintaining target catch rates, have been proposed, but the extent to which BRDs actually reduce the risk of catching sharks and rays remains unclear. We performed a meta-analysis of 27 publications that reported the capture of sharks and rays and, in some cases, of targeted teleosts in longline gear deployed with and without BRDs. The risk of shark and ray capture differed between types of BRDs, but only one BRD type, longlines raised off the bottom, reduced bycatch significantly. Circle hooks did not reduce the risk of capturing sharks and rays but might improve discard survival and are inexpensive, which might make them effective in reducing the detrimental effects of longlining on these species. In addition to being generally ineffective, some devices, such as electropositive and magnetic repellants, are expensive and have inherent construction drawbacks that are likely to make them unsuitable for commercial use. Overall, most BRDs did not affect the likelihood of catching targeted teleosts, but a substantial number of studies did not adequately assess target catch. We identified two poorly studied classes of BRD gear (i.e., raised demersal longlines, and monofilament nylon leaders) which represent promising directions for future research.

[1] A version of this chapter is published as Favaro, B. and Côté, I.M. *In Press* Do bycatch reduction devices in longline fisheries reduce capture of sharks and rays? A global meta-analysis. Fish and Fisheries.

2.2. Introduction

Bycatch, or the unintentional catch of non-target species occurring in fisheries (Kennelly, 2007), is a major source of mortality for shark and ray populations around the world (Gilman et al., 2008, Petersen et al., 2009). Nearly a third of currently assessed sharks, skates, and rays have been designated as threatened or near-threatened by the International Union for Conservation of Nature (IUCN, IUCN, 2012), usually mainly due to exploitation, either targeted or incidental, by industrial fisheries (Stevens et al., 2000, Molina and Cooke, 2012). Most sharks and rays are slow-growing, achieve sexual maturity at a late age, and produce few offspring, making their populations especially sensitive to fishing pressure (Dulvy et al., 2008, García et al., 2008, Hutchings et al., 2012).

Of all industrial fishing methods, longlining presents one of the highest risks to sharks and rays (Watson and Kerstetter, 2006, Gilman et al., 2008, Molina and Cooke, 2012). Interactions between sharks or rays and longline fishing gear can also decrease fishing profitability. Sharks often consume targeted fishes caught on hooks (Gilman et al., 2007), and hooked sharks and rays can damage gear, block the hooks from catching more valuable species, and increase the handling time of gear upon retrieval (Gilman et al., 2008). The costs imposed by these interactions are likely to be substantial, prompting calls for methods to repel sharks and rays from deployed fishing gear (Molina and Cooke, 2012).

A potential solution to the problem of bycatch is the use of bycatch reduction devices (BRDs), or technologies which prevent the capture, or facilitate escape, of non-target species from fishing gear (FAO, 2002). BRDs represent physical alterations to fishing gear, and are distinct from changes in fishing technique (e.g., changes to soak duration or timing), which use existing gear in novel ways to influence capture rates of target and non-target species (Ward and Hindmarsh, 2007). There is an ongoing global effort to develop and implement BRDs to reduce bycatch across all fisheries (Cox et al., 2007). BRDs are an attractive solution because, unlike area closures and other restrictive management measures (e.g. Grantham et al., 2008), they offer fishers the opportunity to maintain most of their fishing activities with little cost, other than that of purchasing the gear modification. The design and promotion of BRD technology are therefore widespread, including through a high-profile international competition to encourage the development of novel BRDs (World Wildlife Fund, 2011).

Devices that reduce shark and ray bycatch in longlines vary widely in design, ranging from electric and magnetic repellants to modified hooks. While qualitative reviews of the costs and benefits of the commonest types of BRDs have been conducted (Swimmer et al., 2008, Molina and Cooke, 2012), a quantitative assessment comparing the effectiveness of all existing BRD approaches is currently lacking. Here, we conducted a meta-analysis of peer-reviewed and grey literature to assess the effectiveness of existing technology at reducing the risk of catching sharks and rays on longlines. We first asked whether such BRD technology works in general. We combined all studies, irrespective of BRD type or species, to generate the first estimate of the overall magnitude of change in shark and ray catches caused by BRD technology compared to conventional longline gear. We then examined the effectiveness of different types of BRDs. We generally did not focus on species-specific effects because longlining gear typically capture a range of shark and ray species, thus BRD effectiveness should arguably be measured across all species captured. However, we did consider separately the results of studies focusing on a single shark or ray species and asked whether BRD effectiveness varied in relation to the level of endangerment of these species. Where possible, we also compared the effect of BRD on shark and ray capture with its effect on the capture of targeted teleost fishes, to highlight gears that manage to reduce bycatch while maintaining the capture of target species. We conclude by identifying promising avenues for future research and development.

2.3. Methods

We conducted a meta-analysis to generate a quantitative measure of the overall effectiveness of bycatch reduction technology applied to longline fishing gear (Harrison, 2011). We identified publications that reported the numbers of sharks and rays caught in fishing gears equipped with BRDs and without BRDs and deployed in the field. We searched three main databases: Bycatch.org, Web of Science, and the Aquatic Sciences and Fisheries Abstracts Database Guide (ASFA). We also considered papers cited in major review documents, and those which we encountered opportunistically (i.e., new papers not yet indexed in the above databases). We followed PRISMA best-practice protocols for conducting this review (Moher et al., 2009, File A.2).

Our criteria for inclusion were threefold. First, the paper had to compare experimentally the catch composition of two or more gear types (unmodified “control” gear versus some type of BRD). Second, the BRD had to be applied to a hook-based fishing gear (i.e., pelagic longline, demersal longline, or hook-and-line), which was similar to that used in commercial fisheries. Third, the experiment had to have been conducted in the field, and not in a laboratory environment. The BRDs did not have to be designed specifically to exclude sharks and rays – they could have been built primarily to protect other species; however, we included them if their effect on shark and ray catch was adequately measured and reported. We included all peer-reviewed literature and government publications that met these three criteria (Figure 2.1). Our search terms (see Appendix A) were intentionally broad to ensure the identification of as many publications as possible.

We used relative risk (RR) as our measure of effect size (Zhang and Yu, 1998). To do so, we recorded from each study: the number of control hooks and the number of BRD-equipped hooks deployed, the number of hooks of each type which caught a shark or ray, and the number of hooks of each type which caught a targeted teleost fish. Relative risk to sharks and rays was calculated as RR = (a/n1) / (b/n2), where a and b were the numbers of hooks equipped with a BRD or not, respectively, that caught a shark or ray, and n1 and n2 were the total numbers of hooks employed in the study with and without a BRD, respectively. We also calculated relative risk to targeted fishes, where a and b were the numbers of hooks equipped with a BRD or not, respectively, that caught a teleost fish. Relative risk is superior to the odds ratio as a measure of effect size for data where occurrence rates are above 10%, which was the case for our data (Zhang and Yu, 1998, Koricheva et al., 2013). Relative risk effect sizes are statistically significant if the 95% C.I. of the model coefficient does not span one. The log of the relative risk was used in the analyses because it normalizes the results and makes the coefficients symmetrical around zero (Viechtbauer, 2010).

We identified nine broad types of BRDs, which we described briefly below. (1) Circle hooks have a rounded shape and are designed to become lodged in the jaw of fish, rather than in the internal organs as can happen with traditional J-shaped hooks (Godin et al., 2012). Circle hooks were designed primarily to reduce capture of sea turtles and to reduce gut-hooking of fishes, which promotes post-release survival, but these hooks have also been assessed for their ability to reduce shark and ray bycatch (Godin et al., 2012). (2) Appendage hooks are circle hooks which contain an extension, or "appendage", which is designed to increase the hook's width and make it more difficult for undersized animals to ingest (Swimmer et al., 2011). (3) Electropositive repellants are created from alloys that oxidize in salt water, generating an electric field (Kaimmer and Stoner, 2008). Electropositive BRDs can take the form of a small ingot placed above the hook (e.g. Kaimmer and Stoner, 2008), or can be woven into the hooks themselves (e.g. O'Connell et al., 2012). (4) Magnetic repellants are similar to electropositive BRDs, but are built using permanent magnets and produce a magnetic rather than an electrical field. Both electric and magnetic fields are detectable by ampullae of Lorenzini, which are electroreceptors that trigger avoidance in sharks and rays when overstimulated (Murray, 1960). (5) Combined repellants, known as SMART™ hooks, include both electropositive and magnetic repellants integrated into the same hook (O'Connell et al., 2012). (6) Changes to bait colour involve dyeing bait blue to reduce visibility to seabirds, and this method has been assessed for its ability to reduce bycatch of sharks and rays as well (Yokota et al., 2006). (7) Breakable monofilament nylon leaders replace wire leaders (i.e., the lines which connect the hooks to the main longline) with less durable nylon, which sharks and rays can bite off, thus preventing their capture (Ward et al., 2008). (8) Tarred multifilament nylon leaders are thicker and darker than monofilament nylon leaders, and therefore may be easier for fish of all species to see and avoid (Stone and Dixon, 2001). (9) Raised lines involve the use of floats to raise the demersal longline off the bottom to avoid capture of bottom-dwelling shark and ray species (Afonso et al., 2011).

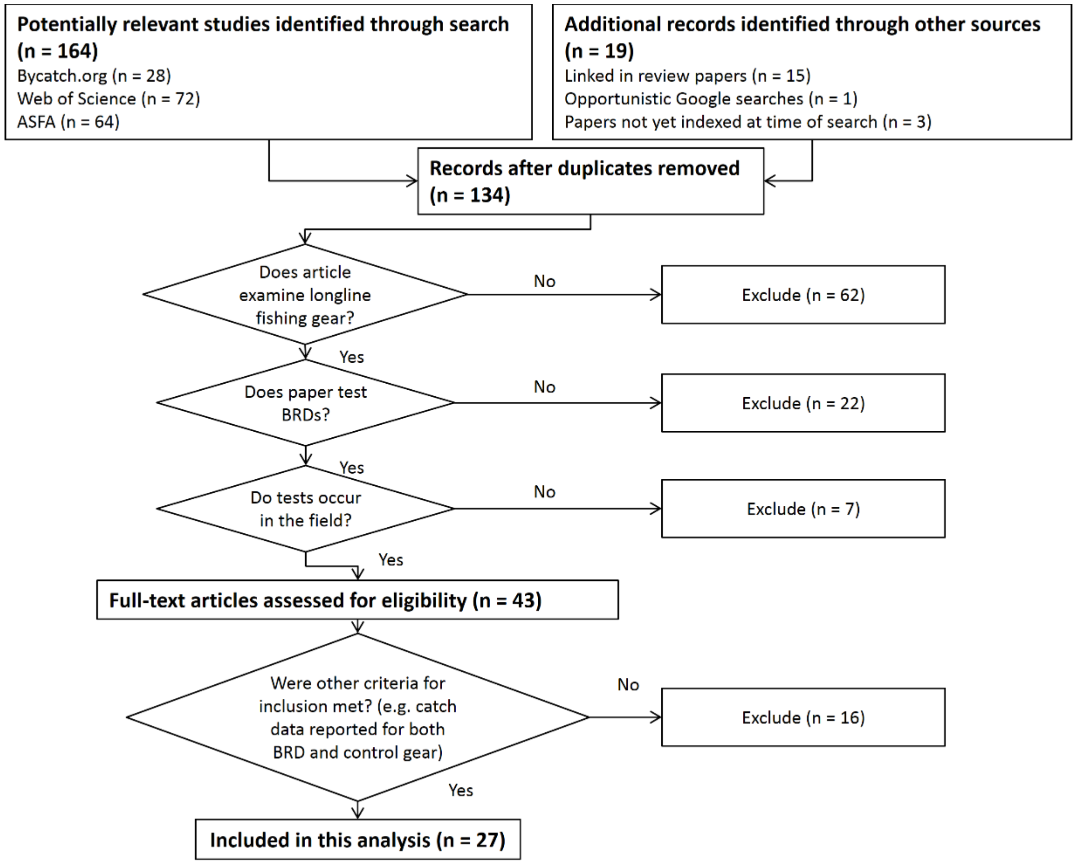

Figure 2.1: Flowchart outlining criteria for inclusion of papers into our meta-analysis.

2.3.1. Statistical analysis

Bycatch affects a wide range of shark and ray species, so an ideal BRD should be effective at reducing bycatch across species. We therefore performed our meta-analysis on shark and ray catch data pooled across all reported species, as the outcome of interest was the overall effect of BRDs rather than their species-specific effects. When publications reported the results of multiple independent field studies, we calculated a separate effect size for each study. In addition, when a study focused on a single species of shark or ray, we recorded the IUCN Red List status of that species (IUCN, 2012). For most types of BRD, the comparison (control versus modified) was clear and unambiguous. However, for magnetic and electropositive BRDs, a "procedural control" was usually employed, where a comparison was made between hooks with an inert metal attached (procedural control), and hooks with a BRD attached (e.g. Kaimmer and Stoner, 2008, Brill et al., 2009, O'Connell et al., 2011, Hutchinson et al., 2012, Godin et al., 2013). In the two magnet and electropositive BRD studies where procedural controls were not used, we calculated the effect size by comparing the BRD-equipped catch with standard controls (Tallack and Mandelman, 2009, Robbins et al., 2011).

We conducted two analyses, each applied to the risk of capture of sharks and rays and then of targeted teleosts. First, we used a random-effects model to generate a grand overall effect size across all types of BRDs on shark and ray bycatch (Viechtbauer, 2010). We employed random-effects modeling because we anticipated substantial heterogeneity among studies, owing to differences among species, locations, and unrecorded BRD details (e.g. different sized hooks, bait types, etc.). In addition, we sought to make an inference about the overall effect of BRDs that was not limited to the studies included in the analysis (Worm and Myers, 2003). Second, we constructed a mixed-effects model, which incorporated BRD type as a moderator, to compare the effectiveness of each class of BRD at reducing catch of sharks and rays (Viechtbauer, 2010). We measured heterogeneity for both sets of models using a restricted maximum-likelihood estimator (τ2) (Viechtbauer, 2005), instead of I2, which is another common measure of the proportion of variability due to heterogeneity. However, since the sample sizes in longline studies are extremely large, I2 would likely overestimate heterogeneity (Rücker et al., 2008). We tested the significance of τ2 using Cochran's Q-test (Hedges and Olkin, 1985). For each mixed-effects model, we conducted an omnibus test to test whether model coefficients were equivalent across BRD types (Hedges and Pigott, 2004, Viechtbauer, 2010). We conducted all analyses using the Metafor package in R (Viechtbauer, 2010, R Development Core Team, 2011).

2.4. Results

Our literature search yielded 183 publications, and after duplicates and non-relevant publications were removed, 27 remained for incorporation into the present meta-analysis (see online supplement). These 27 publications yielded 44 separate studies that met our criteria; 28 studies also reported teleost catch data.

Overall, the use of BRDs did not significantly lower the risk of capturing sharks and rays ($\exp(\widehat{\mathrm{RR}})$ = 0.881, 95% CI = 0.761 to 1.020; Figure 2.2) or targeted teleosts ($\exp(\widehat{\mathrm{RR}})$ = 0.934, 95% CI = 0.836 to 1.043; Figure 2.2). There was substantial heterogeneity among studies in risk to sharks and rays ($\tau^2 \pm 1$ S.E. = 0.221 ± 0.053, Q = 1840.5, df = 43, $p < 0.0001$). When BRD type was incorporated as a moderator, there remained significant residual heterogeneity among studies ($\tau^2 \pm 1$ S.E. = 0.185 ± 0.050, Q_E = 1366.1, df = 35, $p < 0.0001$). For teleosts, there was also significant heterogeneity in the overall model ($\tau^2 \pm 1$ S.E. = 0.068 ± 0.023, Q = 574.8, df = 27, $p < 0.0001$), and residual heterogeneity was significant but smaller with BRD type included as a moderator ($\tau^2 \pm 1$ S.E. = 0.045 ± 0.018, Q = 480.5, df = 20, $p < 0.0001$). The omnibus tests showed that BRD type had a significant effect on RR of both shark and ray capture (Q_M = 18.3, df = 9, p = 0.032) and teleost capture (Q_M = 19.4, df = 8, p = 0.013). The raised line device was the only BRD type that significantly reduced shark and ray catch (i.e., the 95% CI does not encompass 1, $\exp(\beta_9)$ = 0.329, 95% CI = 0.128 to 0.850). Furthermore, only multifilament nylon leaders reduced the RR of teleost catch ($\exp(\beta_8)$ = 0.443, 95% CI = 0.281 to 0.698), while other BRD types had no significant effect on risk of capture of either sharks and rays or teleosts. A forest plot of each study did not reveal any outliers (Figure A.3).

Twelve studies reported a single species of shark or ray caught as bycatch (Figure A.4). Effect sizes were highly variable, and there was no clear trend in capture risk across IUCN threat status categories. However, one study found that electropositive gears reduced capture of endangered juvenile scalloped hammerhead (*Sphyrna lewini*) by 57%. Another study found that circle hooks reduced the catch of pelagic stingray (*Pteroplatytrygon violacea*), a least concern species, by almost 75%.

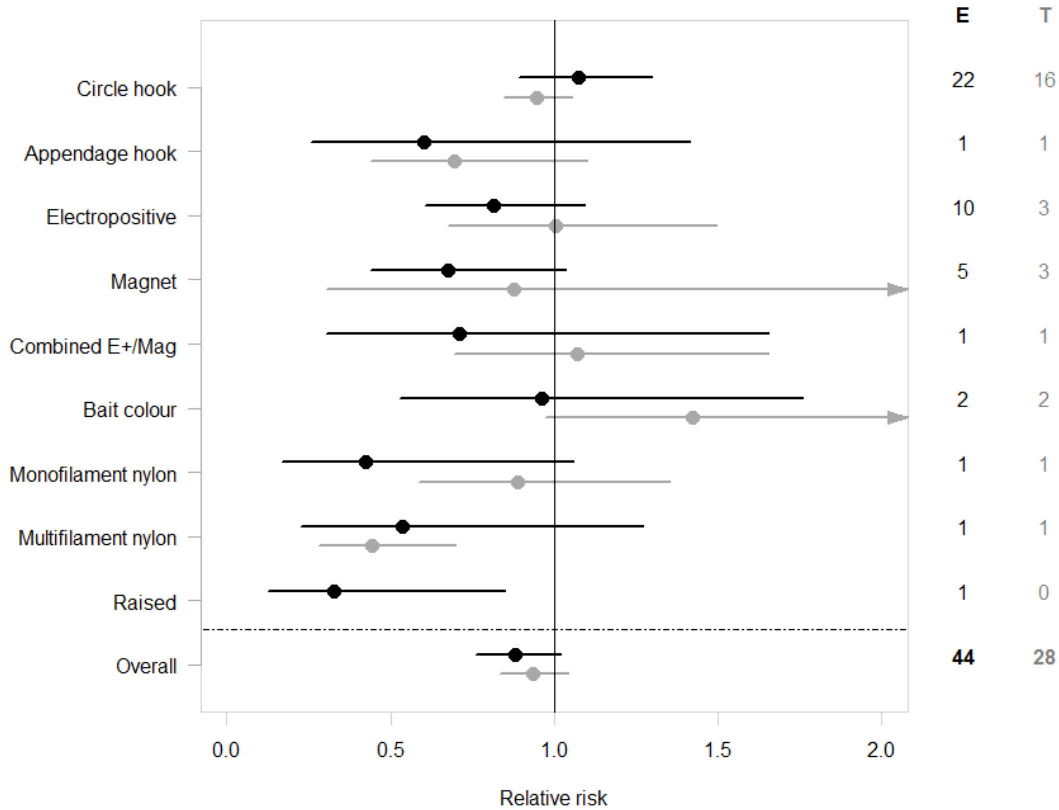

Figure 2.2: Effect of various types of bycatch reduction devices (BRD) on the risk of capturing non-targeted elasmobranchs (black) and targeted teleost fishes (grey). Points indicate modeled effect size as determined by random-effects modeling (for overall effect sizes) and by mixed-effects models for each BRD class. Bars represent 95% confidence intervals. Relative risk <1 indicates a lower risk of capture on BRD-equipped hooks relative to control gear, while >1 represents increased risk of capture. Numbers on the right represent the number of studies with elasmobranch data (E) and the numbers which also contained teleost data (T).

Studies included in each class of BRD:

Circle hook: (Bolten and Bjorndal 2005, Ingram et al. 2005, Watson et al. 2005, Kim et al. 2006, Yokota et al. 2006, Kim et al. 2007, Promjinda et al. 2008, Carruthers et al. 2009, Ward et al. 2009, Piovano et al. 2010, Sales et al. 2010, Afonso et al. 2011, Curran and Bigelow 2011, Pacheco et al. 2011, Domingo et al. 2012).
Appendage hook: (Swimmer et al. 2011).
Electropositive: (Kaimmer and Stoner 2008, Brill et al. 2009, Tallack and Mandelman 2009, Robbins et al. 2011, Hutchinson et al. 2012, Godin et al. 2013).
Magnet: (O'Connell et al. 2011, Robbins et al. 2011).
Combined: (O'Connell et al. In Press).
Bait colour: (Yokota et al. 2009).
Monofilament nylon: (Ward et al. 2008).
Multifilament nylon: (Stone and Dixon 2001).
Raised: (Afonso et al. 2011).

2.5. Discussion

Two of the main benefits of meta-analyses are that they bear a greater external validity than the results of any individual experiment (Shadish et al., 2002) and they allow a quantitative assessment of the overall influence of a predictor on an outcome measure. Our meta-analysis of the effectiveness of longline BRDs suggests that, overall, these devices reduce the risk of shark and ray capture by only 12% compared to standard hooks, and that this difference is marginally non-significant. By comparison, turtle excluder devices, a common type of BRD designed to reduce sea turtle bycatch in trawl nets, produce 99% reductions in turtle catch relative to standard gear (Brewer et al., 2006), and are broadly effective enough to be mandatory for usage in US trawl fisheries (OECD, 2005).

Our conclusion is likely to be conservative because publication bias – a common concern in meta-analysis – tends to favour significant studies, leading to a propensity for meta-analyses to report exaggerated overall effects (Rothstein et al., 2006). The great diversity of BRD types accounted for ~16% of overall variation in capture risk, with one device, raised lines, appearing to be effective at decreasing bycatch of sharks and rays. A great deal of variation in capture risk remains unexplained, which may be attributed to variation in ecosystems, fishing gears, and/or species. We also note that our meta-analysis only incorporated mitigation technology reported in published literature, and the possibility remains that other, more effective gears exist but have not yet been experimentally tested. In addition, other approaches that we did not classify as a BRD, including night-setting, deep-setting, and bait swapping, could be effective but were beyond the scope of our analysis.

Circle hooks are the best-studied type of longline BRDs, both for their effect on shark and ray bycatch as well as their functionality for catching teleosts. Circle hooks slightly increased the risk of capturing sharks and rays caught on longlines relative to control gear (7.6% non-significant increase; see also Godin et al. 2012). However, the propensity of circle hooks to promote jaw-hooking rather than gut-hooking improves the prospects of post-release survival for sharks and rays and potentially other hooked species (Read, 2007, Godin et al., 2012; but see (Ward et al., 2009). Circle hooks also appeared to be effective at reducing the bycatch of one species, the pelagic stingray (Figure A.4). For these reasons, and because they do not reduce target catches (Figure 2.2) and they cost the same as traditional J-hooks (USD ~$0.40 per hook, Pacific Net and Twine Ltd. Richmond, British Columbia, Canada), they represent a potentially viable option for reducing harm caused by longline fishing.

Electropositive, magnetic, and combined BRDs have received a disproportionate amount of coverage in popular media. Similar devices have been marketed as repellants to protect swimmers from shark attacks (Huveneers et al., 2012) and several patents have been granted or are currently pending for electropositive and magnetic-based shark repelling technology (e.g. Stowell, 1980, Wynne, 2006, Stroud, 2007). The enthusiasm for these BRDs stems largely from the fact that these BRDs target a sensory system which is specific to cartilaginous fishes and from the substantial behavioural effects observed on captive sharks in controlled laboratory conditions (e.g. Stoner and Kaimmer, 2008, Rigg et al., 2009). However, the media-hyped suggestions that such BRDs could reduce shark bycatch by as much as 70% (e.g., Shapiro, 2012) were not supported by our meta-analysis of rigorous field studies. Electropositive, magnetic and combined BRDs all failed overall to reduce shark and ray bycatch significantly (non-significant reductions of 18%, 32%, and 29% respectively, relative to control gear). One study found a reduction in catch of scalloped hammerhead, but that result was for juveniles and was inconsistent with the effects on adult sharks (Hutchinson et al., 2012). Furthermore, it seems unlikely that this technology would be adopted commercially even if effective, due to the high cost of electropositive and magnetic alloys, their hazardous manufacturing process (Stoner and Kaimmer, 2008), their poor durability as they dissolve quickly in seawater (Kaimmer and Stoner, 2008) and issues with large-scale deployment (e.g., if magnets stick together, Rigg et al., 2009). It is unlikely that these issues can be resolved with future improvements, as the problems are innate to the gear itself (i.e., electropositive alloys must dissolve to create the electric field). Electric fields in water can also be generated by a powered system which emits electrical pulses. However, the only experiment using such a device (which could not be included in our meta-analysis because the test did not employ fishing gear) demonstrated an effect on shark behaviour, but no effect on the propensity of sharks to take bait attached to a pulsating device (Huveneers et al., 2012).

Two types of BRDs may represent promising avenues for future research. Monofilament nylon leaders have been widely recommended as an effective tool to reduce bycatch reduction and improve target catch rates, and they are attractive because of their low cost.

A single study has so far directly tested the difference in catch between wire and nylon leaders in the field (Ward et al., 2008). Monofilament nylon leaders were 58% less likely to catch sharks and rays than wire leaders but the reduction was not statistically significant owing to the large confidence interval predicted by our model. However, the effect size of the single study that tested monofilament nylon leaders is significant when calculated on its own (i.e., not as part of a meta-analysis, Figure A). Additional research is needed to confirm whether this gear is effective across species. However, the potential population-level impacts of hooks attached to released sharks should also be evaluated, as ingested hooks can promote disease and cause delayed mortality in affected sharks (Bansemer and Bennett, 2010). In addition, longline fisheries that target sharks have paradoxically reported increases in catch when using monofilament leaders (Berkeley and Campos, 1988, Branstetter and Musick, 1993). Tarred multifilament nylon leaders did not reduce the risk of shark and ray capture but instead, reduced the risk of teleost catch by 66%, suggesting that this would be an unsuitable modification for bycatch reduction.

The success observed when raising demersal longlines off the ocean bottom using floats is similarly promising. Elevating the gear in the water column places it in a position where it is less likely to be encountered by demersal sharks and rays (Afonso et al., 2011), and reduced the risk of capture by 66% relative to non-raised gear. This approach of physically separating gear from non-target species is analogous to the weighting of pelagic longlines to sink hooks quickly beyond the reach of diving seabirds, which has been shown to reduce seabird bycatch significantly (Dietrich et al., 2008). It also highlights the potential importance of gear deployment depth in affecting bycatch rates (Ward and Myers, 2005). Further research into how to effectively place gear away from non-target species is therefore also warranted.

The incidental capture of sharks and rays in longline fisheries occurs around the world and affects a wide range of shark and ray species (Gilman et al., 2008). In terms of devising effective conservation strategies to tackle this source of shark and ray mortality, our results have three main implications. First, although a few individual studies have demonstrated that specific BRDs are effective, the weight of evidence across all studies suggests limited success so far. The effectiveness of a given BRD appears to be very context-dependent. Thus, perhaps not surprisingly, a single technological solution that reduces shark and ray bycatch across fisheries is yet to be found. Second, very few of the wide range of BRDs tested appear to affect the catches of targeted teleosts. This is an important finding because maintaining valuable catches is essential for the acceptance of BRDs, and other conservation measures, by the fishing industry. However, we also note that many studies did not assess teleost catch by number, and none assessed BRD-induced shifts in body size and price differentials among species, oversights that must change in future work. Finally, there are understudied classes of BRDs that could represent promising avenues of future work. In particular, raised demersal longlines and monofilament nylon leaders could emerge as potentially cost-effective tools for mitigating shark and ray mortality on longline gear but their impacts on bycatch, target catch, and their practicality for use in fisheries need to be rigorously assessed.

3. Bycatch of rockfish and other species in British Columbia spot prawn traps: Preliminary assessment using research traps[2]

3.1. Abstract

The spot prawn (*Pandalus platyceros*) trap fishery in British Columbia is endorsed by conservation organizations owing to the assumption of minimal bycatch. However, reported capture of juvenile rockfish (*Sebastes* spp.) in prawn traps has raised concern due to declines in abundance of many rockfish stocks. We document the bycatch observed in a 10-year (1999-2008) fishery-independent research survey that employed traps that are similar to the traps used in the commercial spot prawn fishery. Research traps produced 0.16-0.20 kg of non-target catch per kg of spot prawn catch, with bycatch consisting mainly of a variety of molluscs, non-target crustaceans, echinoderms, and fish. The overall rate of rockfish catch was low – 0.015 rockfish per trap. The annual rate of rockfish bycatch has increased since 2004, to 0.039 rockfish per trap in 2008, while catch rates of other species have remained relatively constant. Our results confirm that spot prawn traps produce a low amount of bycatch by weight. However, they also suggest that rockfish mortality due to prawn trapping should be quantified in the commercial prawn fishery to determine how this source of mortality may affect rockfish stocks. Furthermore, research into bycatch reduction technology to improve trap selectivity, and thus reduce rockfish bycatch, would be desirable.

[2] A version of this chapter is published as Favaro, B., Rutherford, D.T., Duff, S.D., and Côté, I.M. 2010 Bycatch of rockfish and other species in British Columbia spot prawn traps: Preliminary assessment using research traps. Fisheries Research **102**, 199-206.

3.2. Introduction

The negative impact of bycatch, i.e. the catch of non-target species that often occurs during fishing, on populations of fish, birds, turtles, and marine mammals is an issue of global importance (Kelleher 2005). It is therefore important to quantify and regulate bycatch in order to devise effective management regimes. The amount of bycatch produced by a given fishery depends on the type of gear used, and trawl fisheries generally produce the greatest overall amount of bycatch by weight (Kelleher 2005, Eayrs 2007). As a result, the bycatch produced by trawl fisheries has been highly scrutinized, and there have been concerted global efforts to improve technology and implement management that reduce trawling bycatch (Hall et al. 2000, Kelleher 2005, Macher et al. 2008). By contrast, trap-based fisheries tend to produce less bycatch than trawl fisheries (Alverson et al. 1994), and as a result the study of trapping bycatch has been less intensive. However, the assumption that traps uniformly produce low bycatch has rarely been tested.

In British Columbia (BC), on the west coast of Canada, spot prawns (*Pandalus platyceros*) are fished commercially with traps. The fishing season is short and intensive (~8 weeks in May and June), and spans the entire BC coast. In contrast to shrimp trawl fisheries elsewhere in the world, this trap-based fishery has been lauded as an example of a well-managed, sustainable fishery (Roberts 2005). It is assumed to produce a negligible amount of bycatch, comprised mostly of species that are resilient to exploitation, not currently threatened, and for which release mortality is assumed to be low. However, the prawn fisheries' in-season bycatch monitoring program has reported that juvenile rockfish (*Sebastes* spp.) are caught in the spot prawn traps (DFO 2009). The abundance of a number of rockfish species has declined sharply in recent years (Love et al. 1998, PFMC 2002, Yamanaka et al. 2004), which makes this source of rockfish mortality of particular interest. All rockfish species are vulnerable to overfishing due to their low population growth rates resulting from late age at maturity and variable recruitment (Love et al. 2002). Being caught as bycatch in the trap fishery could impose additional mortality because discarded rockfish are unlikely to survive swim bladder rupture and other physiological and physical damage caused by rapid ascent of retrieved traps (Hannah and Matteson 2007, Brill et al. 2008). All sources of rockfish mortality, including bycatch, should be evaluated.

The first step in bycatch reduction for any gear type or fishery is to quantify current rates of incidental capture (Broadhurst 2007). However, direct measures of bycatch in the commercial prawn fishery are currently unavailable. In order to quantify bycatch in spot prawn traps, we examined the catch composition of a fishery-independent prawn trap research survey. The purpose of the research survey was to determine population parameters for the stock-recruit model upon which the management of the prawn fishery is based (Boutillier and Bond 1999, DFO 2009). When conducting this survey, all catch (target and non-target) was quantified.

In this study, we used data gathered in the research survey to provide a first, preliminary assessment of the bycatch of rockfish and other species in spot prawn traps, and evaluate how bycatch has changed since observations began in 1999. We then estimated the average length of rockfish caught in an effort to assess the demographics of the rockfish bycatch. By evaluating the magnitude and composition of non-target catch, we tested the assertion that prawn traps produce low overall bycatch, and made a preliminary estimate as to whether these traps could be a source of rockfish mortality.

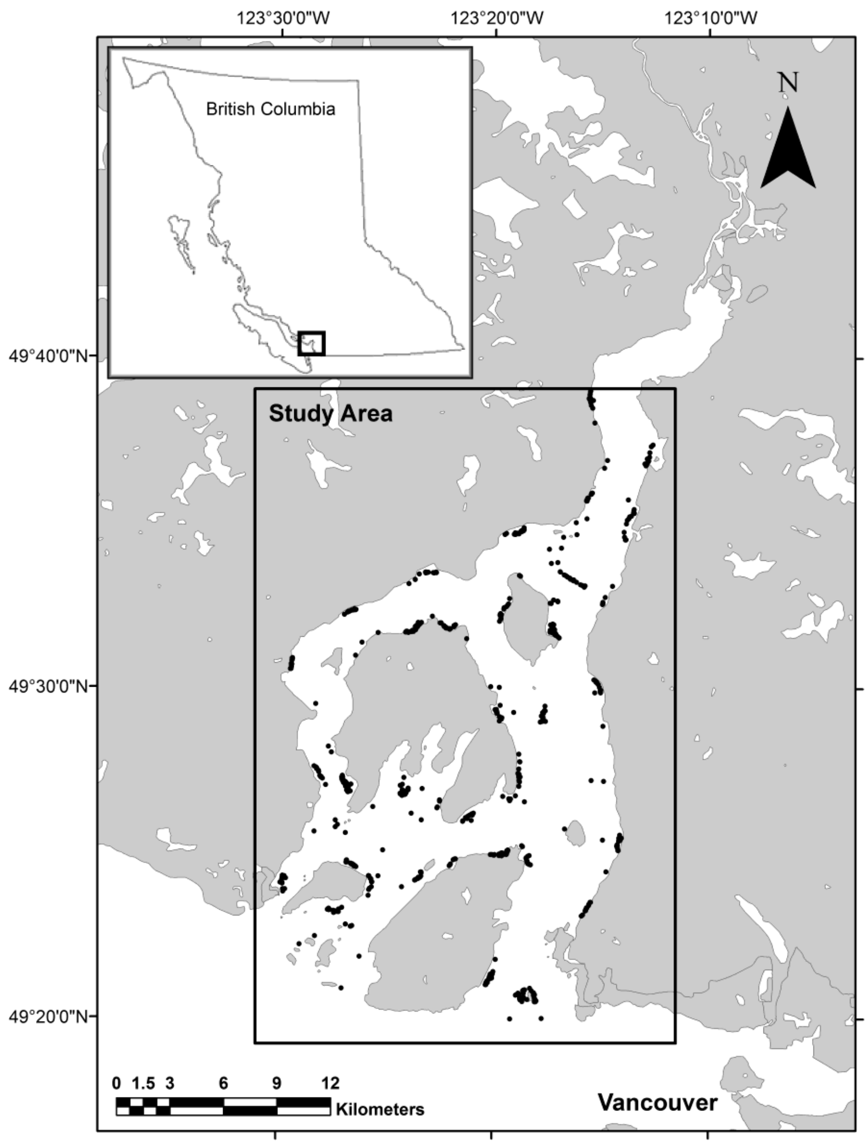

Figure 3.1. Map of the study area in Howe Sound, near Vancouver, British Columbia. Black dots indicate locations of gear deployment during the spot prawn research trap survey.

3.3. Materials and Methods

The research survey was first carried out in fall 1999, and then twice annually (fall and spring) from spring 2000 to fall 2008 (i.e., a total of 19 sampling periods) by Fisheries and Oceans Canada (DFO). During each sampling period, 39 – 65 strings of 20 prawn traps were deployed throughout Howe Sound, near Vancouver, British Columbia, Canada

(49°25’30” N, 123°20’00” W), at locations that included sites used by commercial fishers as well as sites that were not fished intensively (mean number of strings per sampling period ± 1 SD: 45 ± 7; for sampling locations, see Figure 3.1). A given site was usually sampled a single time within a sampling period, and similar (though not necessarily identical) locations were sampled across periods. Traps were deployed on the ocean floor with a target soak time of 24 h (mean depth ± 1 SD: 74.3 m ± 9.4 m). A total of 856 strings of traps (= 17,210 traps) were deployed during the study.

Traps were a standard truncated cone design supported by three stainless steel rings (Rutherford et al. 2004b), of a size and shape similar to commercial traps. Traps were approximately 75 cm in diameter at the base and 30 cm in height, and were covered in 1.3 cm soft mesh (commercial traps have a larger minimum mesh size of 3.8 cm). Each trap had three circular openings, each 7.6 cm in diameter. Three bait type combinations were used. The traps in 774 strings were baited exclusively with tuna cat food, which was the primary bait type used in the commercial fishery until the early 1990s, after which specially formulated fish pellets became the predominant bait type (Rutherford et al. 2004b). The remaining 82 strings had multiple bait types to enable comparison between the numbers of prawns caught using cat food and pellets. Of these, 70 had half the traps baited with cat food and the other half baited with fish pellets, and 12 had half the traps baited with fish pellets dipped in fish oil (a baiting method used by many commercial fishers), and the other half with standard unoiled pellets. The strings with multiple bait types were deployed intermittently from 2003 to 2008.

At the end of each soak, strings were retrieved using a hydraulic winch, and the contents of each trap were recorded. Data from each set of 20 traps were aggregated into a single value representing catch from the string. The total weight and number of spot prawns per string were recorded. Non-target species were identified to the lowest possible taxonomic level. The aggregate weight (but not the number of individuals) of each non-target species caught was recorded for each string. For rockfish, the number of individuals was also recorded, and the rockfish count was aggregated to the string level. When multiple bait types were used in a string, only the number of prawns caught in traps with each bait type was recorded; bycatch in these strings was recorded across bait types.

To determine whether rockfish caught as bycatch were pre-reproductive (juvenile) individuals, we had to estimate rockfish length. This was possible whenever a single rockfish was caught in a string since the (string-level) weight recorded for rockfish was for that individual. For each singleton, we calculated species-specific length using the power equation Weight = a*Length^b (Schneider et al. 2000), where a and b for each rockfish species were obtained from Love et al. (2002). If more than one set of parameters was given, the most recently published parameters were used. We report the mean length of singletons of each species based on these calculations.

We examined temporal variation in target and non-target catch using Generalized Linear Models in R. Using a manual forward stepwise approach, we considered year (including a squared year term to allow for non-linear variation across years) and season as potential predictors of numbers (prawn and rockfish) and weight (prawn, rockfish, crustaceans, molluscs and echinoderms, separately) caught per string. Rockfish numbers were modelled using a Poisson error distribution with log-link function; all other models had normal error distributions. The significance of each predictor added was assessed with analyses of deviance.

3.4. Results

Spot prawns made up the bulk of the catch during the 10-year research survey (Table 1). A total of 424,420 spot prawns were caught, with a combined mass of 13,020.40 kg (83.2% of total catch by weight). On average, 496 (± 264 SD) prawns were caught per string, or ~24.8 prawns per trap. The mean weight of prawns per string was 15.21 kg (± 7.58 SD), or 0.76 kg per trap (Table 3.1).

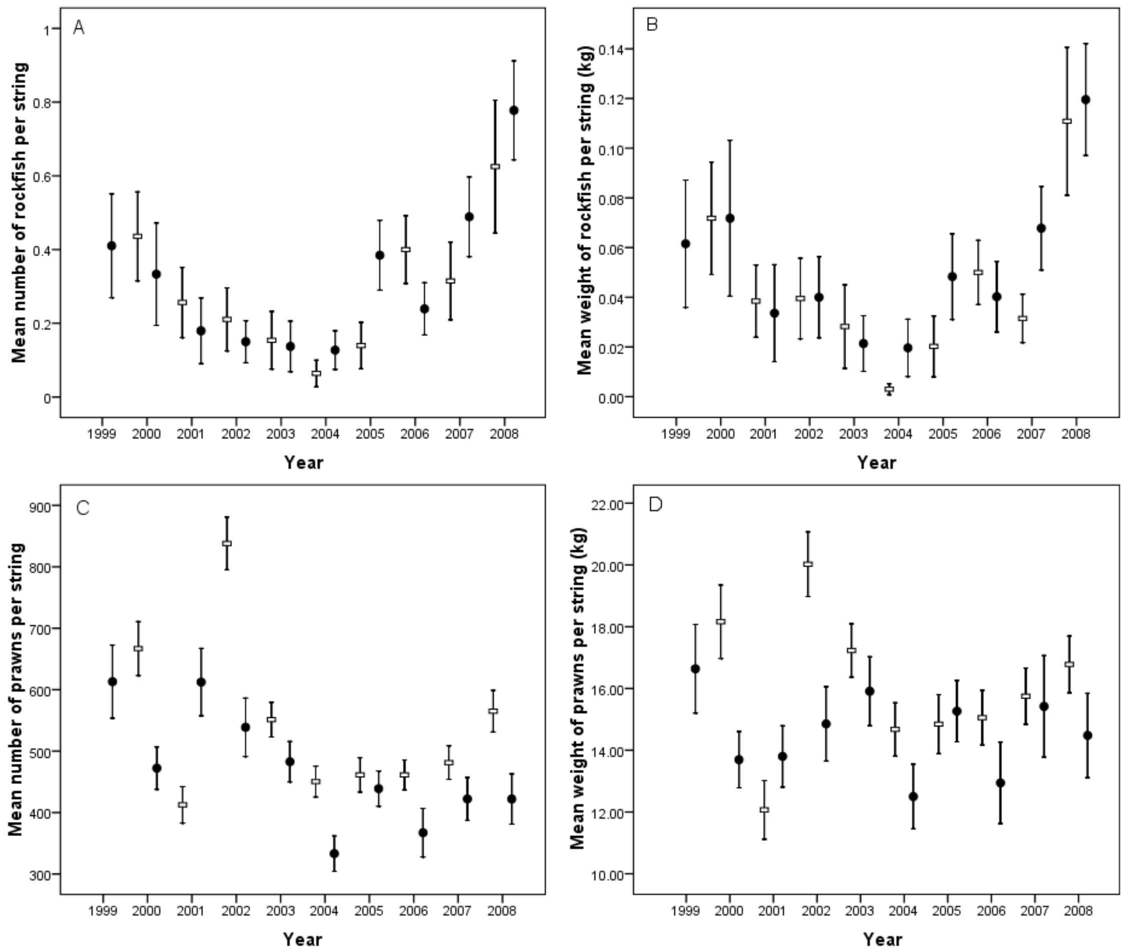

Figure 3.2. Catch of rockfish (A: by count; B: by weight) and spot prawns (C: by count; D: by weight) in a spot prawn trap research survey in Howe Sound, British Columbia. Means are shown (± 1 SE) per 20-trap string. The number of strings deployed per sampling period is given in Table 2. Black circles represent fall sampling, and white rectangles represent spring sampling.

The spot prawn research survey captured a wide diversity of non-target species. A total of 99 species, genera, or families were recorded as bycatch over the 10-year period (see Appendix B). The total catch of all non-target species was 2,630.51 kg (mean per string ± 1 SD: 3.07 ± 3.99 kg), which represented 16.8% of the total catch in the study, or 0.20 kg non-target catch per 1 kg of prawns. Squat lobsters (6.5% of total catch) and giant Pacific octopus (2.6%) were the second and third most abundant taxa by weight, after spot prawns (Table 3.1). Quillback rockfish (*Sebastes maliger*) were the most frequently caught fish, and the 12th most abundant non-target species by weight (Table 3.1).

Table 3.1. Top 12 species caught in the spot prawn research trap survey in Howe Sound, British Columbia, Canada. Total catch is summed over 10 years (1999-2008), with two sampling periods per year.

Common name	Scientific name	Total catch (kg)	Mean catch per 20-trap string (kg) (± 1 SD)	Mean catch per trap (kg)
Spot prawn	*Pandalus platyceros*	13020.4	15.21 (7.58)	0.761
Squat lobster	*Munida quadrispina*	1022.38	1.19 (2.71)	0.06
Giant Pacific octopus	*Enteroctopus dofleini*	410.54	0.48 (2.40)	0.024
Sunflower seastar	*Pycnopodia helianthoides*	316.32	0.37 (1.04)	0.018
Humpback shrimp	*Pandalus hypsinotus*	122.79	0.14 (0.52)	0.007
Pink shrimp (smooth)	*Pandalus jordani*	119.76	0.14 (0.40)	0.007
Dungeness crab	*Cancer magister*	110.48	0.13 (1.07)	0.006
Pink shrimp	*Pandalus eous*	92.1	0.11 (0.41)	0.005
Red rock crab	*Cancer productus*	88.85	0.10 (0.24)	0.005
Fish-eating seastar	*Stylasterias forreri*	47.88	0.06 (0.22)	0.003
Pacific red octopus	*Octopus rubescens*	43.28	0.05 (0.44)	0.003
Quillback rockfish	*Sebastes maliger*	35.61	0.04 (0.12)	0.002

The rate of capture of rockfish was low. A total of 264 rockfish were caught over 10 years. The species composition was as follows: 225 quillback rockfish, 19 copper rockfish (*S. caurinus*), 14 greenstriped rockfish (*S. elongatus*), 2 yelloweye rockfish (*S. ruberrimus*), 2 sharpchin rockfish (*S. zacentrus*), 1 darkblotched rockfish (*S. crameri*) and 1 splitnose rockfish (*S. diploproa*). With 856 strings hauled and 20 traps per string, the overall rockfish catch rate was 0.015 rockfish per trap. The majority of strings (674, or 78.7% of strings) caught no rockfish.

Table 3.2. Fishing effort and rockfish bycatch by the spot prawn research trap survey in Howe Sound, British Columbia, over 10 years. The number of sets of gear (strings of 20 traps) deployed per season and year, number of rockfish caught, and rockfish catch rate per trap are given for each sampling period.

Year	Season	No. gear deployments	Total no. rockfish caught	Percentage of strings with multiple baits	Rockfish catch rate per trap
1999	Fall	39	16	0	0.021
2000	Spring	39	17	0	0.022
	Fall	39	13	0	0.017
2001	Spring	39	10	0	0.013
	Fall	39	7	0	0.009
2002	Spring	38	8	0	0.011
	Fall	40	6	0	0.008
2003	Spring	39	6	0	0.008
	Fall	51	7	15.1	0.007
2004	Spring	47	3	14.9	0.003
	Fall	55	7	14.5	0.006
2005	Spring	43	6	0	0.007
	Fall	65	25	24.6	0.019
2006	Spring	45	18	13.3	0.02
	Fall	46	11	8.7	0.012
2007	Spring	54	17	20.3	0.016
	Fall	45	22	11.1	0.024
2008	Spring	48	30	18.6	0.031
	Fall	45	35	13.3	0.039
All periods		856	264	9.6	0.015

Rockfish catch rate showed a clear, non-linear trend over time (Figure 3.2 A and B). Both rockfish numbers and weight declined between 1999 and 2004, and then increased to a peak in fall 2008 (rockfish number: year: β = -242.0, $p < 0.0001$; year^2: β = -0.06, $p < 0.0001$; season: NS; rockfish weight: year: β = -14.1, $p < 0.0001$; year^2: β = -0.004, $p < 0.0001$; season: NS). The number of prawns caught per string also varied non-linearly over time (year: β = -16600.0, $p = 0.001$; year^2: β = 4.13, $p = 0.001$), with significant variation between seasons (season: β = -88.1, $p < 0.0001$; Figure 3.2 C). The total weight of prawn caught per string did not vary seasonally or among years (all coefficients NS; Figure 3.2 D). Rockfish and prawn catches were not correlated, when measured by count (Spearman's correlation; $r = 0.004$, $n = 856$, $p = 0.92$) or by weight (Spearman's correlation; $r = 0.04$, $n = 856$, $p = 0.27$). There were no significant seasonal or inter-annual differences in weight of other bycatch species, when aggregated at high taxonomic levels to generate suitable sample sizes (Figure 3.3).

A total of 137 rockfish were caught as singletons (i.e., a single individual of a given rockfish species in a string, Table 3.3), allowing the estimation of species-specific mean length at capture. Except for greenstriped rockfish, the mean length for each species was below the length at which 50% of individuals are mature, indicating that the majority of rockfish caught as bycatch were likely to be pre-reproductive individuals.

Table 3.3. Mean weight and estimated length of rockfish caught as bycatch in spot prawn traps, derived from singleton catches, i.e. strings of traps in which a single rockfish of a given species was caught (see Materials and methods). Parameters for weight-length calculation as well as values for length at 50% maturity are from Love et al. (2002).

Species	Number of singletons	Average weight (kg) (±1 S.D.)	Estimated length (cm) (±1 S.D.)	Length at 50% maturity females/males (cm)
Quillback rockfish	106	0.15 (± 0.09)	18.65 (± 1.74)	29/29
Copper rockfish	15	0.15 (± 0.10)	19.48 (± 4.34)	34/32
Darkblotched rockfish	1	0.05	13.91	39/37
Splitnose rockfish	1	0.01	11.1	27/27
Greenstriped rockfish	10	0.15 (± 0.07)	22.95 (± 3.24)	22/24
Yelloweye rockfish	2	0.14 (± 0.08)	20.61 (± 3.89)	46/54
Sharpchin rockfish	2	0.09 (± 0.02)	18.37 (± 1.41)	25/24

The bait type used in prawn traps affected prawn catches. In the 70 strings with cat food and fish pellet baits, the 10-trap batches baited with fish pellets caught significantly more prawns than those baited with cat food (mean ± 1 SD, fish pellet bait: 275 ± 176 prawns per 10 traps, cat food bait: 210 ± 95 prawns per 10 traps; two-tailed t-test for unequal variances, $t = 2.67$, $df = 106$, $p < 0.01$). In the 12 strings using both oiled and unoiled fish pellets, there was no difference in prawn catch between both bait types (mean ± 1 SD, oiled fish pellet bait: 265 ± 122 prawns per 10 traps, standard fish pellet bait: 285 ± 115 prawns per 10 traps; two-tailed t-test for equal variances, $t = 0.41$, $df = 22$, $p = 0.69$).

Because weights and numbers of individuals for all non-target species were collected at the level of the string, we could not directly test whether there was a difference in the catch rate of non-target species between bait types. Instead, we examined whether the presence of multiple bait types in a string affected catch rates of non-target organisms. There was no difference in the mean number of rockfish caught between strings baited with only cat food, strings with both cat food and fish pellets, and strings with both unoiled and oiled fish pellets (Kruskal Wallis test: $\chi^2 = 3.24$, $df = 2$, $p = 0.20$). In addition, the rate of rockfish bycatch was not affected by the proportion of traps baited with multiple bait types within a sampling period (Table 3.2). There was no difference in the mean weight of echinoderms, molluscs, or fish other than rockfish between strings using different mixtures of baits (Kruskal Wallis tests, molluscs: $\chi^2 = 3.52$, $df = 2$, $p = 0.17$, echinoderms: $\chi^2 = 0.96$, $df = 2$, $p = 0.62$, non-rockfish fish: $\chi^2 = 1.70$, $df = 2$, $p = 0.43$). Crustacean bycatch did vary between strings with different bait combinations (Kruskal Wallis test: $\chi^2 = 37.14$, $df = 2$, $p < 0.001$) and was highest in traps baited with only cat food.

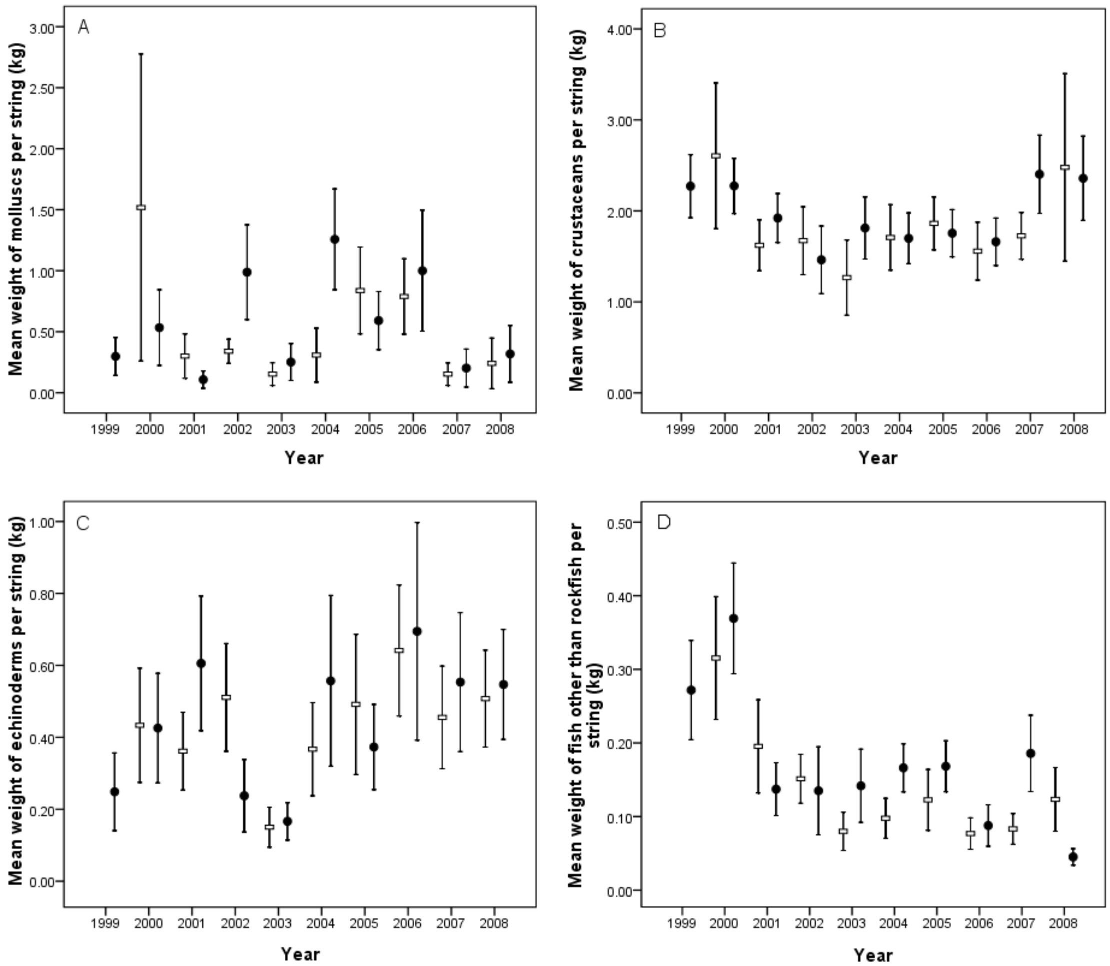

Figure 3.3. Weight of molluscs (A), crustaceans (B), echinoderms (C), and fish species other than rockfish (D) caught as bycatch in a spot prawn trap research survey in Howe Sound, British Columbia. Means are shown (± 1 SE) per 20-trap string. The number of strings deployed per sampling period is given in Table 3.2. Black circles represent fall sampling, and white rectangles represent spring sampling. A full list of species is provided in Appendix B.

3.5. Discussion

This study represents the first description of the catch composition of spot prawn traps. Overall, the non-target catch rate in the 10-year research trap-survey was low relative to that of the target species, but a wide diversity of species was caught as bycatch. The 11 most abundant species by weight were all invertebrates which, as they lack swim bladders, are assumed to have low discard mortality. However, it is currently unknown whether there are sublethal effects of handling on animals discarded from prawn traps. Except for rockfish, the non-target catch did not comprise any species of current concern.

It is useful to place the proportion of non-target catch observed in spot prawn traps into context by comparing it to the bycatch rates observed with other trap fisheries. In our research trap surveys, which used gear that was similar to the local commercial fishery, 16.8% of the total catch by weight was bycatch, or 0.20 kg of non-target species per kg of target species landed. If octopuses are considered part of the target catch, since they can be kept and sold in BC, then 13.9% of the total catch is non-target, or 0.16 kg of non-target species per kg of target species landed. However, the smaller mesh size in the research traps complicates comparison with the expected bycatch in the commercial fishery. For example, *Pandalus jordani* and *P. eous* were the 6th and 8th most abundant species in the research traps, but they would not be captured in the commercial traps as they would escape through the larger mesh. To our knowledge, there are no other large-scale shrimp or prawn fisheries that use traps to catch prawns, so a direct comparison of bycatch rates with other prawn trapping gear is not possible. Nevertheless, the bycatch rate in this study is roughly comparable to that of the Atlantic American lobster trap fishery, at 0.22 kg discards per kg landed (Alverson et al. 1994). The trap fishery for the lobster *Nephrops norvegicus* is another large-scale trap fishery, and although there are no estimates to date of the ratio of non-target to target catch, preliminary investigation revealed that *Nephrops* traps captured far fewer taxa than trawls for the same species (Morello et al. 2009). Based on the species composition of the non-target catch, and on the amount of non-target catch by weight observed in research traps, we conclude that prawn traps produce a low overall amount of bycatch relative to target species.

Quillback rockfish contributed the largest proportion of non-target catch, by weight, of any vertebrate in the research prawn survey. This species is widely distributed along the BC coast but declined substantially in abundance between 1986 and 2005 (Yamanaka et al. 2006). Quillback rockfish and spot prawns share the same preferred depth range and complex rocky bottom habitat (Canada 1999, Love et al. 2002), increasing the likelihood that quillback rockfish will encounter prawn traps. All species of rockfish brought up from depth are afflicted with barotrauma and often display severe physical symptoms including ruptured swim bladder, eye protrusion, and prolapsed stomach and cloacae (Rogers 2008). These injuries often prevent discarded fish from descending from the surface

(Hannah et al. 2008a), making them easy targets for sea birds and other predators. It is therefore likely that the mortality rate of all rockfish caught as bycatch in prawn traps is high.

The rate of rockfish catch in the surveys appears to have increased markedly since 2004. A similar trend is evident for spot prawn numbers, but not for catches of other major groups caught as bycatch. We suggest two non-mutually exclusive explanations for increased rockfish captures. First, the abundance of juvenile rockfish in the Howe Sound area may have increased, and this increase is reflected in bycatch rates. Increases in local abundance of rockfish may be driven by the implementation of conservation measures, or by environmental changes that have increased recent recruitment success. Rockfish Conservation Areas (RCAs) were implemented by the Canadian government in 2001 to promote population recovery of rockfish by providing refuges from directed fishing, and RCAs in Howe Sound were established in 2004 and 2007. It is possible that these reserves have been quickly effective at protecting rockfish (Roberts 2001), although their life histories would lead to the expectation of much slower numerical responses to protection (Russ 1998, Roberts 2001, Blyth-Skyrme 2006). Comparisons of rockfish abundance in and out of RCAs would be necessary to confirm this mechanism. However, it is not clear how rockfish protection would enhance prawn densities since prawn trapping is permitted in RCAs. It is also possible that environmental conditions have been recently favourable for rockfish (and prawn) recruitment. Rockfish recruitment success is closely related to oceanographic factors, such as upwelling intensity and water temperatures (Wilson 2008), and it would be informative to examine changes in potential environmental correlates of rockfish and prawn recruitment over the past decade in Howe Sound. Second, the catchability of rockfish and prawns may have increased in recent years due to a non-density-related phenomenon. However, the prawn trapping gear used in this study has not changed over time, and since rockfish were not targeted in this research survey there has been no effect over time of an improvement in expertise at catching rockfish using prawn gear. Changes in foraging behaviour and geographic distribution in response to environmental factors can change catchability of organisms from year to year (Tremblay 2006), potentially complicating the interpretation of temporal trends in trap catches.

The potential impacts of the relatively low rates of rockfish bycatch in prawn traps on rockfish populations are not clear. Due to the small size of the entrances, large mature rockfish are unable to enter prawn traps. This is fortunate because the loss of reproductively valuable adults has been suggested as one of the causes of declines in rockfish abundance (Berkeley et al. 2004). The loss of juveniles (i.e. < 23 cm total length, depending on the species) instead of adults may attenuate the effects of bycatch on rockfish populations because fecundity and larval growth and survival increase with maternal age in rockfish (Berkeley et al. 2004, Sogard 2008). Nevertheless, the presence of juvenile rockfish in prawn traps and the frequency at which they were caught in the present study suggests that a detailed evaluation of rockfish bycatch in the commercial prawn fishing fleet is warranted, and that managers should include this source of mortality in their management of inshore rockfish stocks.

In conclusion, the proportion of non-target catch by weight was low in spot prawn traps. The presence of rockfish in prawn traps and the observed rate of rockfish bycatch raise concerns about the magnitude of rockfish bycatch that may occur in the commercial prawn fishery, although it is important to note that our research survey and the commercial fishery differed in spatial extent, timing of fishing, trap mesh size and bait types. Nevertheless, due to the depressed state of rockfish populations, every effort should be made to reduce rockfish bycatch, regardless of gear type. Bycatch reduction technology that excludes rockfish while maintaining prawn catch rates would therefore be desirable. In addition, rockfish mortality due to commercial prawn trapping should be quantified to establish if this is a source of mortality that needs to be accounted for in the management of rockfish stocks.

4. TrapCam: an inexpensive system for studying deep-water animals[3]

4.1. Abstract

Behavioural research in deep water (>40 m depth) has traditionally been expensive and logistically challenging, particularly because the light and sound produced by underwater vehicles make them unsuitably disruptive. Yet, understanding the behaviour of deep-water animals, especially those targeted by exploitation, is important for conservation. For example, understanding interactions between animals and deep-water fishing gear could inform the design of devices that minimize bycatch.

We describe the 'TrapCam,' a self-contained, high-definition video system that requires neither the support of a vessel once deployed, nor special equipment to deploy or retrieve. This system can record 13-h videos at 1080p resolution and is deployable on any substrata at depths of up to 100 m. The system is inexpensive (<$3000 USD), versatile, and suited to the study of animal behaviour at depths inaccessible to scuba divers.

We evaluate the performance and cost-effectiveness of TrapCam, and analyze videos retrieved from pilot deployments to observe spot prawn (*Pandalus platyceros*) traps at 100 m depth. Preliminary analyses of animal-prawn trap interactions yield novel insights. We provide future directions for researchers to use this type of camera system to study deep water-dwelling species around the world.

[3] A version of this chapter is published as Favaro, B., Lichota, C., Côté, I.M., and Duff, S.D. 2012 TrapCam: an inexpensive camera system for studying deep-water animals. Methods in Ecology and Evolution **3**, 39-46.

4.2. Introduction

It is difficult and costly to study animals that live in deep water. Much of our knowledge of species that live below depths accessible to scuba divers (i.e. deeper than 40 m) comes from destructive sampling as organisms are brought to the surface in fishing or sampling gear. While these methods of collection can provide information about the animals' distribution, physiology, and diet, they are inappropriate for the study of behaviour. As a result, we know relatively little about the *in situ* behaviour of deep-dwelling organisms compared to shallow-living species.

Underwater cameras are a tool of choice for *in situ* observations of the behaviour of deep-water species. Dataloggers, pop-up satellite tags, and other such equipment, can provide information about selected behavioural aspects, such as habitat use and movement, of these animals but actual behaviour must often be inferred from tagging data and many tags are unsuitable for small animals. In contrast, cameras have the unique ability to provide direct visual information of organisms which often cannot be obtained in any other way. The use of camera technology can yield important information for conservation. In terrestrial systems, the use of (near-)continuous videography has been effectively used to gain insights about organisms that are difficult to access or phenomena that occur infrequently, such as predation in bird nests (Pietz and Granfors 2000, Kross and Nelson 2011), roosting behaviour in bat harems (Hoxeng et al. 2007), and even nocturnal foraging of spiders (Taylor and Bradley 2009). In the marine realm, however, most existing options for deep-water camera research are prohibitively expensive, and require access to large vessels, and expensive support technology and specially trained crew to operate. This is the case for submersibles, Remotely Operated Vehicles (ROVs), and permanent cabled bottom-mounted observatories, which have all been used to study species that live below scuba-diving depths (e.g. (Milliken and DeAlteris 2004, Piasente et al. 2004, Mills et al. 2005, Woodroffe and Round 2008). In addition, the light and sound produced by mobile, camera-bearing platforms can alter animal behaviour significantly (Popper 2003, Ryer et al. 2009) and it has been recently demonstrated that even crustacean behaviour can be greatly affected by sound transmission (Simpson et al. 2011). Surface-powered "drop cameras" can be attached to boats or surface floats (e.g. Mills et al., 2005) but these systems require either a stable vessel or a large electronics package on a surface float to be connected to the camera system for the duration of filming, which is logistically challenging for deep-water work. Baited Remote Underwater Video (BRUV) stations overcome some of these problems (Cappo et al. 2004), but their use of full spectrum illumination, reliance on bait as an attractant, and often time-lapse photography essentially limits them to assessments of relative abundance and species distribution (Cappo et al. 2006). Cameras which are attached directly to animals, or "Crittercams," are capable of revealing behaviour on deep dives, but are only suitable for animals that come to the surface, and that are large enough to support a mounted camera (Herman and Bakhtiari 2007).

A more versatile tool to study deep-water species would ideally meet five requirements, each of which has implications for design. First, the system should be able to record for long periods of time (e.g. 12 or more hours) at great depths (e.g. >40 m), which suggests the need for autonomy from the surface and the use of adequate pressure casings. Second, to glean maximum information about behavioural interactions – particularly those that are rapid and infrequent, the system should be capable of recording continuously, rather than simply taking photos or short video clips at intervals (e.g. Jury et al. 2001, Barber and Cobb 2009), thus requiring large storage capacity and power. Motion-activated video (e.g., Kross and Nelson 2011) is also not a viable option in underwater environments due to the near-constant movement of particles that would activate the video. Third, full-spectrum lighting, which can have a profound effect on animal behaviour in deep water environments (Olla et al. 2000, Widder et al. 2005, Ryer and Barnett 2006, Ryer et al. 2009), should be avoided. Fourth, the system should be deployable on uneven substratum types without the assistance of divers, thus mandating a righting system or the ability to record in any position. Finally, the system should be inexpensive to build and deployable from a small vessel with no special equipment, to appeal in behavioural and ecological research.

In this paper we describe a novel, deep-water, *in situ* recording system that meets the above requirements. The development of this system was prompted by a call for the design of improved traps that selectively catch spot prawns (*Pandalus platyceros*) while excluding juvenile rockfish (*Sebastes* spp.), a group of species of conservation concern in British Columbia, Canada, thus the custom-designed camera system was dubbed 'TrapCam'. We review the capabilities of the system, and provide results from a pilot study that suggest that the TrapCam can provide significant insights into interactions between deep-water animals and passive fishing gear beyond what catches can reveal. We then explain how modified versions of this basic design could be adapted to study deep-water animals, at depths far exceeding those described in the present study.

4.3. Materials and Methods

4.3.1. Design of camera apparatus

The camera and electronics for TrapCam were assembled with readily available parts from components marketed to semi-professional filmmakers (Table 4.1). Video was recorded with a Sony HDRXR550V Handycam, powered by an NPFV100 Sony battery pack. An additional external battery pack (PowerStream PST-MP3500-I with PST-MP3460 pack and Sony connectors) was connected to the camera's DC-in slot. This camcorder's lens was a 37 mm "G Lens™," and video was recorded in 16:9 widescreen, giving a wide field of view that we further increased by attaching a screw-on Opteka 0.43x wide-angle lens to the camcorder. This system was capable of recording at 1080p resolution (1920 x 1080) for 13 hours. Firmware limitation on the camcorder prevented recording videos longer than 13 hours, despite the fact that the camcorder has a 240 GB internal hard drive, which provides sufficient capacity for more than 24 hours of continuous recording. The electronics were enclosed in a cylindrical anodized aluminum pressure case (15 cm diameter x 25 cm inside depth) with a 2 cm-thick scratch resistant polycarbonate viewport at one end (custom manufactured by A.G.O Environmental Electronics, Victoria, British Columbia, Canada, Figure C.1). This case was certified by the manufacturer to sustain pressures of up to 11 atm (or 100 m depth). The camera and battery pack were held in place simply by contact with the inner sides of the case, and by packing the empty parts of the case with "Pick-N-Pluck™" foam (Figure C.1).

The pressure case was mounted atop a frame constructed of 7.62 cm diameter ABS pipe, which, for our purposes, was connected to a standard commercial truncated cone prawn trap (as described in Rutherford et al. 2004b; Figure 4.1, File C.2), but could be attached to any other structure or frame. The ABS pieces were secured using ABS cement, and the pipes were attached to the prawn trap using plastic cable ties. The case was locked into place by a steel bar placed through the top of the frame. The camera was oriented facing downward about one metre above the trap, giving a top-down view of the trap with a field of view of 110 cm by 80 cm. This view enabled us to see the contents of the trap as well as the area immediately surrounding the trap. In order to maintain an upright orientation when deployed, a cross-shaped ABS pipe frame was attached to the bottom of the trap. The trap was weighted internally with three standard red bricks (approximate total weight: 2.5 kg), to add mass close to the centre of gravity of the apparatus and thus minimize tipping once deployed.

Table 4.1: TrapCam components and estimated costs (USD in 2011). The list does not include the gear or frame on which the TrapCam is mounted.

Item	Cost
Pressure case (Anodized aluminum, 100m rating)	$1,250
Sony HDR-XR550V camcorder	$1,000
Sony NPFV-100 battery pack	$130
PowerStream PST-MP3500-I with PST-MP3460 battery pack	$270
Four Princeton Tec Torrent LED Scuba lights	$240
Four high output red LEDs	$20
ABS pipes and cement	$60
Total	**$2,970**

4.3.2. Lighting

Little ambient light is present at 100 m depths, and as a result an external lighting system was necessary. The retinal pigments in deep-sea fishes and many crustaceans tend to be insensitive to red light (Goldsmith and Fernandez 1968, Meyer-Rochow and Tiang 1984, Douglas et al. 1995), making such light a good tool for observations of natural behaviour. However, our initial attempts to use infrared light systems were unsuccessful as they did not provide sufficient illumination, owing to the high attenuation of infrared and red light in water (Pegau et al. 1997). Instead, we attached four Princeton Tec-Torrent LED scuba lights modified to use red LEDs in lieu of the full-spectrum illuminators that the lights are usually equipped with. We attached each light (powered by eight AA rechargeable batteries each) to the ABS frame to illuminate the field of view (Figure 4.1).

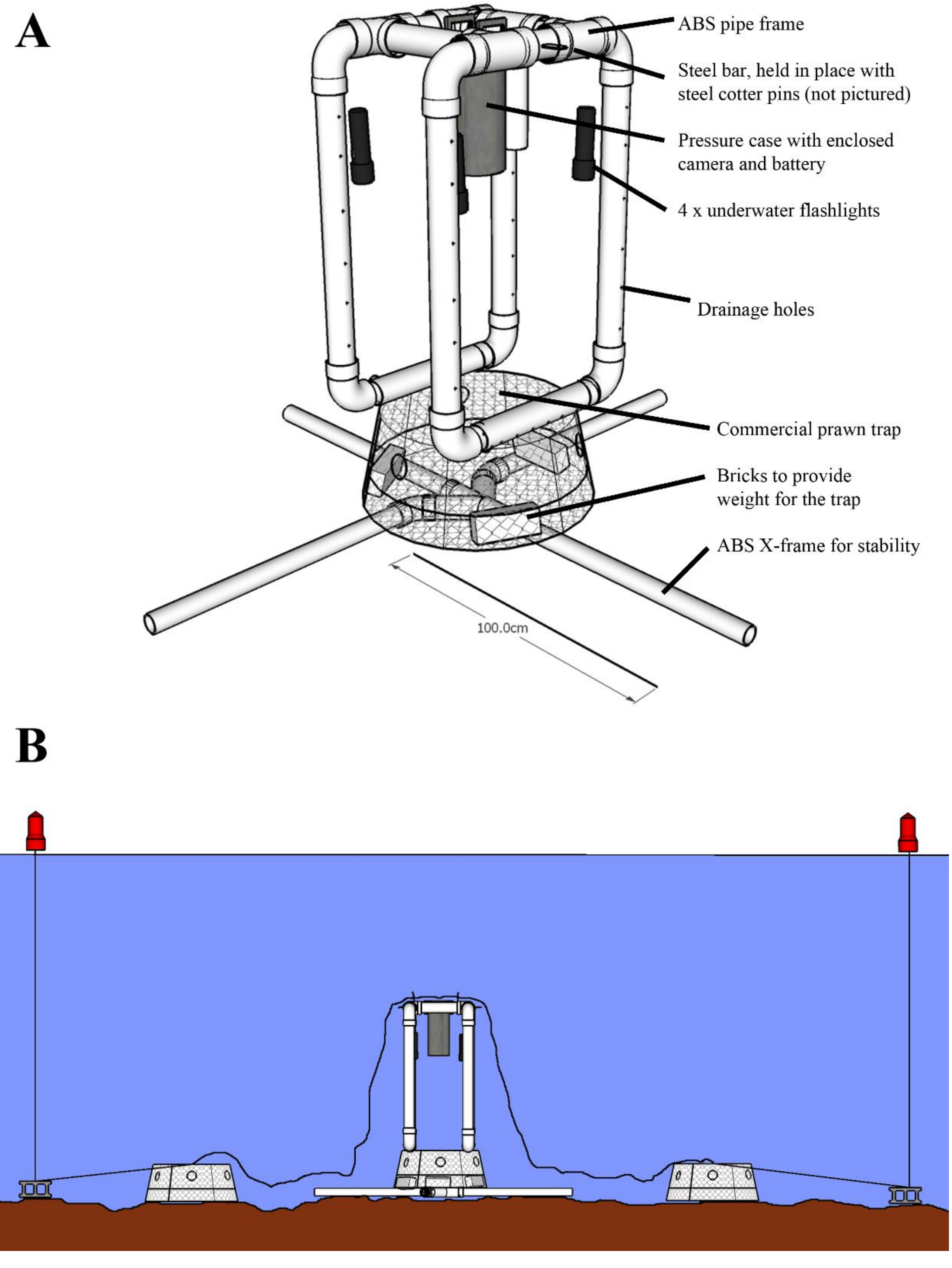

Figure 4.1. Labelled diagrams (not to scale) of (A) the TrapCam apparatus and (B) a schematic of how the apparatus is deployed at the centre of a 3-trap string, which is anchored to the bottom by a cinder block at either end. Each block anchors a surface buoy.

4.3.3. Gear deployment

The camera system was deployed for a pilot study in Howe Sound, British Columbia (49°25'30" N, 123°20'00"W), in July and August 2010, using the Simon Fraser University research vessel *CJ Walters* (length: 9.8 m; beam: 3.7 m). Based on published locations of regular prawn surveys by Canada's federal Department of Fisheries and Oceans, and on experience (by B. Favaro) on commercial vessels in the area (Figure C.3), we selected deployment sites that would maximize prawn catch and/or bycatch, thus maximizing potential interactions between prawns, rockfish, and traps. Gear was deployed in strings (i.e. multiple traps connected to a single line weighted with one cinder block at either end) as in the commercial prawn fishery (Figure 4.1B, File C.2, Video C.4). In our case, the string consisted of the camera-equipped trap paired with two unmodified traps, one of either side of the camera, to evaluate any difference in catch between traps with and without cameras. We deployed the 3-trap string on 16 occasions, and the deployments always commenced during daylight hours, recording into the night. We recorded the GPS coordinates as well as the depth of deployment (mean ± 1 SD = 88 ± 10 m, range = 60 to 100 m), taken from the vessel's depth sounder, for each deployment.

We left the strings to soak for 17 to 45 h (mean = 24 h). All but three strings were recovered the day after they were set. We retrieved the strings using an electric anchor puller. We counted the number of individuals of each species caught in the traps, identified to the lowest possible taxonomic level. To compare the number of prawns caught in traps with and without cameras, we performed a nested analysis of variance (ANOVA), with string nested within camera treatment to test for variability in catch between strings.

4.3.4. Video analysis

We recorded videos at a resolution of 1920 x 1080 pixels using the "night mode" and "low lux" settings on the camcorder. Each 13-h video was 100 GB in size, and the files were stored on external USB hard drives. We watched videos using Elecard AVC HD Player software on 24-inch 16:9 (widescreen) flat screen monitors.

For the purposes of the present analysis, we scored three of the videos (39 hours). We noted in detail the actions of all animals caught on film. An approach was recorded whenever an animal entered the field of view of the camera. We recorded data in 30-

second bins across the entire video to facilitate the scoring of rapid events while preserving the chronological order of such events. Since most organisms were not individually identifiable, individuals were undoubtedly counted multiple times as they left and re-entered the field of view. We recorded entries into the trap, as well as how the animal entered the trap (i.e. through a tunnel or through the mesh). Prawns, other crustaceans, rockfish, fish other than rockfish, and other animals were all recorded separately. When possible, we identified the organism in the field of view to species.

4.4. Results

4.4.1. Camera performance

We recorded a total of 208 h of video across the 16 camera deployments. We inferred from the videos that the apparatus was deployed upright for all but one of the 16 trials. The technique of attaching a camera-bearing trap to a string as with normal prawn traps was effective, and it was easy to both set and recover the gear. Image quality was excellent (Figure 4.2), and the lens was never affected by fog inside the case. Many species could be clearly identified (Video C.5). The lighting was effective, but dim at the periphery of the viewable area. In addition, images at the periphery appeared slightly distorted due to the use of the wide-angle lens, which made species identification more difficult. Illumination decreased slightly over the course of the 13 h videos, but did not inhibit observations. However, at the end of each 24-hour deployment, light output would have been insufficient to illuminate the trap. An improved battery system for the lights would therefore be required to record videos longer than 13 hours.

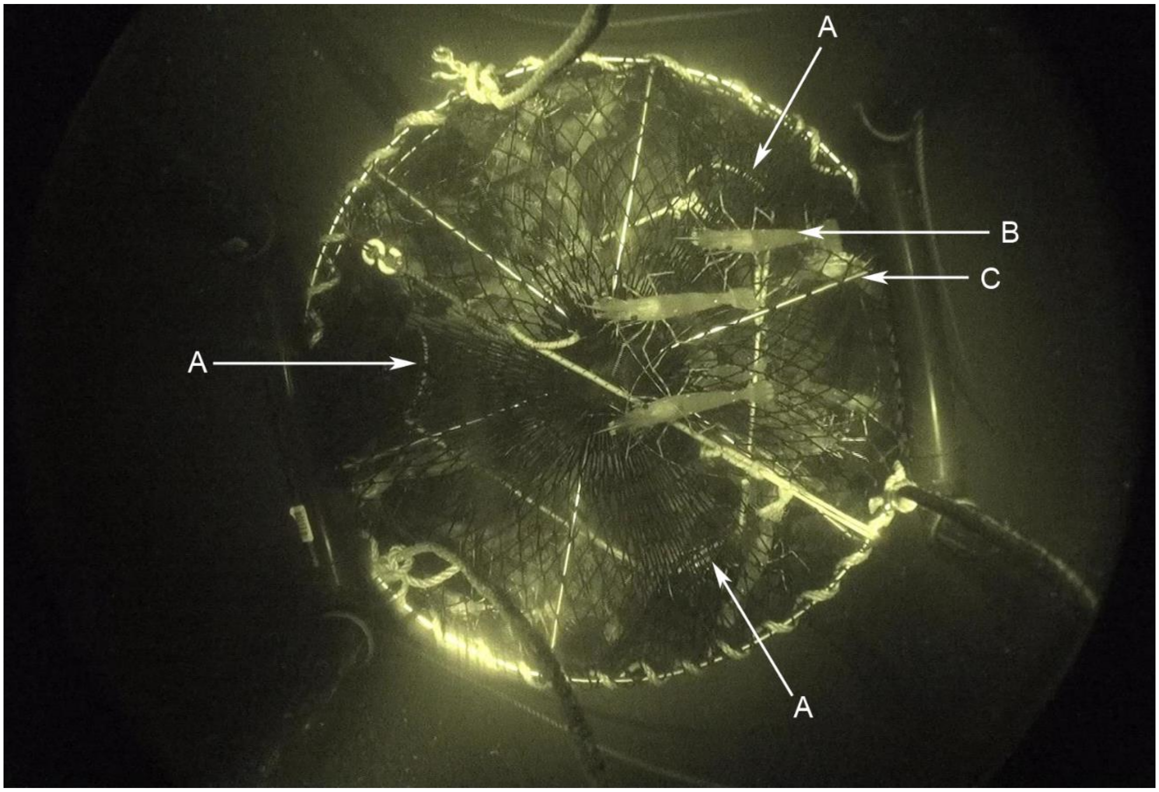

Figure 4.2. Sample TrapCam image. In this image, a top-down view of the prawn trap and its immediate surroundings are visible. The three openings are marked by (A). Prawns on top of the trap (B) and inside the trap (C) are visible in the image. This image was recorded 7 h into a deployment.

4.4.2. Effect of camera on catch

Spot prawns (*Pandalus platyceros*) were present in 94% of traps (mean ± 1 SD = 22.5 ± 21.4 prawns trap^{-1}, n = 48). The trap equipped with a camera caught similar numbers of prawns as traps without cameras (mean ± 1 SD; camera = 29.7 ± 27.9 prawns trap^{-1}, n = 16 deployments, no camera = 18.9 ± 16.6 prawns trap^{-1}, n = 32 deployments; nested ANOVA, camera effect: $F_{1,44}$= 2.75, p = 0.11). In addition, there was no difference in prawn catch between strings within each trap type ($F_{2,44}$ = 0.23, p = 0.79), suggesting that all strings caught comparable numbers of prawns across our study.

Table 4.2: Species recorded in three 13-h videos taken during deployments of a camera attached to a prawn trap at 100 m depth. Taxa are listed in order of decreasing frequency of approaches. Note that the final number caught when traps were retrieved can exceed the number of entries because only the first 13 hours of each soak were recorded.

Common name	Scientific name	Source	Number of approaches	Number of observed trap entries	Final number caught
Spot prawn	*Pandalus platyceros*	Brandt, 1851	1426	180	198
Quillback rockfish	*Sebastes maliger*	(Jordan & Gilbert, 1880)	16	1	0
Fish-eating star	*Stylasterias forreri*	(deLoriol, 1887)	10	0	0
Cancrid crabs	Family Cancridae		9	0	0
Flatfish	Order Pleuronectiformes		8	0	0
Dungeness crab	*Metacarcinus magister*	(Dana, 1852)	7	0	0
Red rock crab	*Cancer productus*	Randall, 1839	6	0	0
Tanner crab	Genus *Chionoecetes*		4	0	0
Pink shrimp	*Pandalus eous*	Makarov, 1935	3	0	0
Rockfish spp.	Genus *Sebastes*		2	0	0

4.4.3. Highlights of in situ observations of animal-trap interactions

Spot prawns comprised 95.8% of all approaches to the trap (Table 4.2). Across the three camera-trap deployments, an average of only 12.6 ± 12.9% of prawns that approached the trap actually entered it. Although only spot prawns were caught in the three deployments analysed (Table 4.2), we witnessed 65 interactions (1.7 ± 14.4 approaches per hour) between non-target species and prawn traps and an average of 4.6 ± 3.5 identifiable taxa (range = 1 to 8) in or on the camera-bearing trap during three deployments (Table 4.2).

We captured on film a quillback rockfish that entered the trap approximately 6 hours after deployment, and remained in the trap until the end of the 13-hour video. The fish successfully exited before the gear was retrieved. While in the trap, this rockfish attempted to escape by swimming upward against the top of the trap (Video C.6).

4.5. Discussion

We assembled and field-tested a new, inexpensive camera system, suitable for *in situ* behavioural studies of aquatic organisms. Various camera technologies have been previously used for similar or related purposes. A brief review of the most recent technologies (Table 4.3) highlights the unique niche of TrapCam in terms of cost, working depth and ease of deployment. Pilot deployments of TrapCam, performed as a proof of concept, yielded significant insights about interactions between deep-water animals and fishing gear.

4.5.1. Cost effectiveness and system flexibility

TrapCam has proven to be cost-efficient and more capable in many ways than other camera systems (Table 4.3). In terms of procurement cost, TrapCam was substantially less expensive than a mid-sized ROV, notwithstanding the fact that ROVs would have been much more invasive and not permitted the bottom time necessary for behavioural research on traps. A mid-sized ROV can cost between US$50,000 and US$100,000 to procure, and requires a large, crewed vessel with a crane to operate. Usage fees for a mid-sized ROV can approximate US$3,000 per day inclusive of vessel and crew (J. Martin, Simon Fraser University research vessel/ROV operations manager, pers. comm. March 2010). In contrast, each TrapCam unit cost approximately US$3,000 to procure and build (Table 4.1). The cost of each deployment is then limited to the cost of operating a small vessel. The only equipment required on the vessel is a winch or electric puller to assist in hauling the gear, although manual retrieval of the gear is possible (though physically taxing). The size of vessel required to deploy and retrieve the system would depend on the environmental conditions at the deployment site, but in calm seas we believe that our system could be safely deployed from a vessel as small as 6 m in length.

Table 4.3. Comparison of selected camera technologies that can be used to study animal behaviour underwater

Camera type	Source	Deployment depth (m)	Deployment method	Recording type	Video duration, resolution, and lighting	Platform type	Requires surface support?	Primary use	Relative cost
TrapCam	This study	100	Self-righting system that descends uncontrolled	Continuous (30 fps)	13 hours, 1920x1080, red LED (modified SCUBA lights)	Trap-mounted – Adaptable to any weighted frames	No	Behaviour – prawn traps at various depths, or other purposes requiring top-down view	$
Lobster TV	(Jury et al., 2001)	6	Controlled descent from boat	Time-lapse VHS (2 fps)	12 hours, 480x300, red LED	Attached directly to lobster trap	No	Behaviour – lobster traps in shallow water	$
Crab video system	(Barber and Cobb, 2009)	5	Controlled descent from boat	2 s record per 20 s interval	8-11 hours, 720x480, unlit	Attached directly to crab trap	No	Behaviour – crab traps in shallow water	$
Multi-camera system	(Mills et al., 2005)	30	Deployable anywhere where three anchors can be set, away from boat traffic	Continuous recording with up to eight cameras, variable FPS	24 hours at 1 frame per second, 480x300, red LED or infrared	Floating pontoon with electronics package, secured by three anchors, with camera as deep as 30 m	Yes – Tethered to floating platform – camera signals transmitted via microwave to separate station up to 10 km away	Behaviour – traps on reefs and other substrata	$$
University of Rhode Island underwater camera	(Milliken and DeAlteris, 2004)	Up to 1000m	Attached to trawl net, pulled with trawling boat	Continuous	2 hours, 320x240, no external lighting	Trawl-mounted	Yes – cable connects camera to vessel	Behaviour - trawling	$$
VENUS (CMAP Systems Cyclops Digital Camera)	(Woodroffe and Round, 2008)	100 - 300 m	Permanent observatory, camera mounted onto structure using ROV	Continuous (30 fps)	Unlimited duration, 720x480, 100 W incandescent lights	Permanent stationary platform, orientation controllable remotely	Yes – Cable connection to on-ground observatory	Community composition around observatory	$$$

Baited Remote Underwater Video station	Reviewed in (Cappo et al., 2006)	Up to 3420 m	Uncontrolled descent	Continuous or time-lapse (30 fps to fewer than 1 frame per minute)	Depends on camera used - minutes to hours, various resolutions, generally full-spectrum light	Camera mounted to frame, giving horizontal look-outward view	No	Abundance estimation and community composition	$ to $$
Crittercam	(Herman and Bakhtiari, 2007)	298 m	Mounted on large animals	Continuous (30 fps)	180 min, 720x480, no external lighting	Animal-mounted	No – After time, device detaches from animal, recovered by boat	Behaviour	$$

Improved versions of the systems described may exist but have not yet been used in published studies. Relative cost is indicated by the number of dollar signs ($ = USD 0–5000; $$ = USD 5000–20 000; $$$ = >USD 20 000)

A low-cost, self-contained camera system like the one implemented here could be used to study any trap fishery. Crab, sablefish, *Nephrops norvegicus*, and other trap fisheries would benefit from the behavioural insights gained, both in terms of designing more effective fishing gear and in studying the behavioural dynamics of species interacting with traps and each other at depth. In addition, an apparatus like TrapCam could be effective for visualizing the impacts of trap gear on habitat (i.e., by observing habitat destruction when the gear impacts the bottom). We could also envisage deploying TrapCams on short, scaffold-like frames that could be baited, in a deep-water, long-lasting version of BRUVs, to obtain estimates of relative species abundance and information on foraging behaviour and/or competitive interactions among species. The principle of a self-contained camera system could theoretically be applied to any depth, with deeper systems simply requiring a more robust pressure case.

4.5.2. Successful practical application of TrapCam

TrapCam successfully permitted long-duration, *in situ* observations of animals in and around prawns traps deployed in deep water. Importantly, the camera apparatus did not appear to affect the behaviour of prawns and other organisms observed. Indeed, there was no evidence of attraction or avoidance of the trap or camera from the behaviour of animals captured on video. In addition, the final number of prawns caught was similar between camera-bearing and control prawn traps.

Analysis of videos taken from only three deployments of our camera-bearing trap has shed light on three aspects of interactions between target and non-target species and passive fishing gear. First, a larger diversity of organisms is present in and around traps at depth than is observed in trap catches. Although up to eight identifiable taxa were present around prawn traps, only prawns were caught when the gear was retrieved, suggesting that prawn traps are highly effective at excluding or facilitating the exit of non-target species, and that high catch specificity was not simply a result of low species diversity at the deployment sites. Second, few of the prawns attracted to traps actually enter them, suggesting that prawn catchability may be quite low, potentially informing our assessment of the relationship between fishing effort and fishing mortality in this system. Finally, the entry into the trap and attempted escape by the rockfish provide important information which could help us redesign prawn traps to minimise rockfish bycatch. The rockfish in the trap attempted to escape by repeatedly swimming upward against the mesh on the top of the trap (Video C.6). If this behaviour is characteristic of trapped rockfish, then adding exit panels to the top of traps might facilitate fish exit.

4.5.3. Conclusion

The main goal of our study, i.e. developing a low-cost tool to perform minimally invasive observations of animal behaviour in deep water, was achieved. We were able to gather many hours of useful data, and preliminary analysis of the videos has already yielded results that could not have been obtained through examination of catch data alone. Though TrapCam was designed for the purpose of devising bycatch reduction modifications, this basic design could be employed by any researcher interested in the behavioural dynamics of animals in deep water, and is of particular use for studies of fish and shellfish traps. The use of TrapCam can enable researchers to perform minimally invasive observations on animals that were previously effectively inaccessible for behavioural study.

5. A trap with a twist: evaluating a bycatch reduction device to prevent rockfish capture in crustacean traps[4]

5.1. Abstract

Bycatch, or the incidental capture of non-target species, occurs in fisheries around the world, with often detrimental ecological consequences. Bycatch reduction devices (BRDs) that increase catch specificity have been used successfully in some fisheries, and the development of such devices remains an important component of the global effort to reduce bycatch rates. We tested novel devices designed to exclude juvenile rockfish (*Sebastes* spp.) from traps used to catch spot prawns (*Pandalus platyceros*), a commercially important species in British Columbia, Canada. The devices included reductions in trap opening sizes and novel bent-tunnel openings. Reducing trap opening size did not affect bycatch rates of rockfish or other non-target fish species. By contrast, bent-tunnel BRDs eliminated rockfish bycatch, and two of the bent-tunnel variants also excluded other fish species. However, prawn catch rates were reduced in all modified gear, and large prawns were often excluded more than small prawns. Videos recorded *in situ* revealed that prawn attempts to enter traps took longer and were more likely to fail in BRD-equipped than in unmodified traps. We conclude that bent-tunnel BRDs have the potential to be useful, but improvements are needed to increase prawn catch to levels similar to those of unmodified traps.

[4] A version of this chapter is published as Favaro, B., Duff, S.D., and Côté, I.M. 2013 A trap with a twist: evaluating a bycatch reduction device to prevent rockfish capture in crustacean traps. ICES Journal of Marine Science **70(1)**: 114-122.

5.2. Introduction

Bycatch – or the unintentional catch of non-target species during fishing – represents an ongoing challenge to fisheries managers. Globally, between eight and 40 percent of fishing mortality is attributed to the capture of non-target species during the fishing process (Kelleher 2005, Davies et al. 2009), and bycatch has been implicated in population declines of cetaceans (Read 2008), various species of seabirds (Lewison et al. 2005, Dillingham and Fletcher 2008, Watkins et al. 2008), turtles (Wallace et al. 2008), and sharks (Ward et al. 2008).

The magnitude of bycatch in a fishery depends in large part on the selectivity of the gear used (Chuenpagdee et al. 2003). Some gears are selective by nature, while others, such as benthic trawls, are notoriously unselective (Alverson et al. 1994). It is increasingly common to develop modifications, or Bycatch Reduction Devices (BRDs), which improve the selectivity of existing gears (FAO 2002). Some BRDs are highly successful (Isaksen et al. 1992, Shiode and Tadash 2004), but as a whole the development and testing of BRDs lag behind the number of identified bycatch issues in fisheries around the world (Kennelly and Broadhurst 2002).

Though some gear types are well known for their high bycatch rates, conservation problems can also arise from the use of traditionally selective gears. Trapping is a common fishing practice which is often assumed to be sustainable due to commonly low bycatch rates and minimal habitat destruction. However, as with any fishing gear, trap-based fisheries do capture some non-target species (Carlile et al. 1997). For example, in British Columbia, on the west coast of Canada, a large-scale trap fishery exists to catch spot prawns (*Pandalus platyceros*). Between 2002-2008, an average of 3.4 million traps were deployed each year during the eight-week season (Rutherford et al. 2010). Prawns traps are highly selective, but bycatch of juvenile rockfish (*Sebastes* spp.) has been observed, albeit at a low rate per trap (Favaro et al. 2010, Rutherford et al. 2010). Due to the large number of trap-days, even a low per-trap bycatch rate has the potential to produce significant numbers of fish lost in absolute terms. This bycatch is an issue because the majority of the 37 rockfish species occurring in BC waters are vulnerable to overfishing due to their late age at maturity and variable recruitment success (Leaman 1991, Love et al. 2002), leading to the decline of rockfish populations over the past decades (Love et al. 2002, Yamanaka et al. 2004, Yamanaka et al. 2006). In addition, rockfish caught in traps and discarded do not survive due in part to the barotrauma-induced rupture of their swim bladders when rapidly brought to the surface in fishing gear (Hannah et al. 2008b). The quillback rockfish (*Sebastes maliger*) is the most common rockfish species caught in prawn traps (Favaro et al. 2010, Rutherford et al. 2010), and has been listed as “threatened” by Committee on the Status of Endangered Wildlife in Canada (COSEWIC, COSEWIC 2009) – the independent scientific body which recommends species for listing under the Species at Risk Act in Canada.

The motivation to reduce rockfish bycatch in prawn traps is twofold. First, due to the depressed state of rockfish populations, any development that reduces mortality could assist the recovery of these species. Second, prawn trapping is one of the activities permitted within Rockfish Conservation Areas (RCAs), areas where most commercial fishing activities are banned to facilitate rockfish recovery (Fisheries and Oceans Canada 2007). The bycatch of rockfish by prawn traps in RCAs, even at low rates, could inhibit the recovery of rockfish within these areas, jeopardizing the mandate of RCAs to protect rockfish from “all mortality associated with recreational and commercial fisheries” (Fisheries and Oceans Canada 2007) and potentially leading to stricter fishing regulations.

The greatest challenge in tackling problematic bycatch is that BRDs must not only reduce or eliminate bycatch, but must also maintain catch rates of target to have as small an impact as possible on fisher livelihoods. It is therefore important when designing new devices to assess the catch rates of both target and non-target species as well as the selectivity of the conventional and modified fishing gears across body sizes of organisms (Holst and Revill 2009).

In this paper, we examine the effectiveness of various BRDs at eliminating rockfish bycatch while maintaining prawn catch at levels close to those of unmodified commercial traps. We test two broad families of BRD designs – simple reductions of the size of the trap openings, and novel opening attachments designed to facilitate prawn entry while excluding rockfish and other fishes from the traps. Using both catch data from a large field study as well as data collected *in situ* using an underwater camera apparatus purpose-built to record activity around deployed prawn traps (Favaro et al. 2012), we assess the performance of BRDs by comparing the catch of modified traps to the catch of unmodified commercial gear, and examine bycatch rates and size selectivity of each gear design. Our primary goal was therefore to assess these novel BRDs, test how they perform at excluding non-target species while retaining target species, and – based on observations from the video data – determine potential ways to improve the performance of the gear for future use in the commercial fishery.

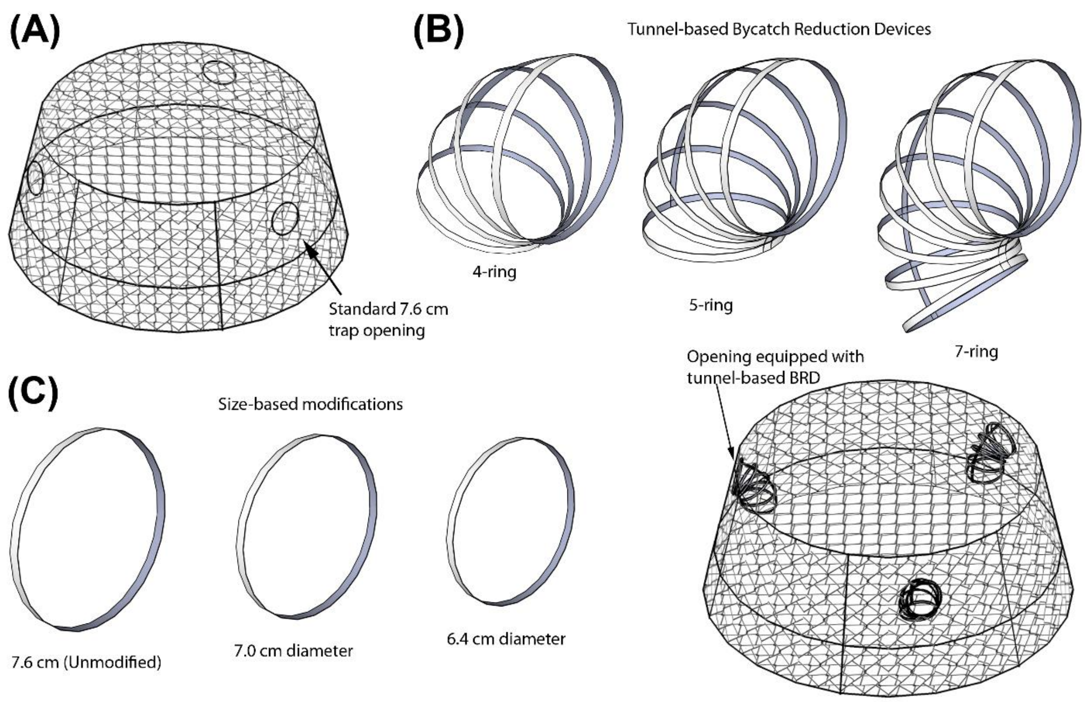

Figure 5.1. Bycatch reduction devices (BRD) used on spot prawn (Pandalus platyceros) traps. (A) Unmodified commercial prawn traps with three 7.6-cm diameter entrances. (B) The three tunnel-based BRDs used in this study, and (bottom right) shown when clipped onto the existing 7.6-cm entrances. (C) Two ring-based BRDs, which were reduced versions of the standard entrance diameter.

5.3. Materials and Methods

5.3.1. Bycatch reduction device designs

Commercial prawn traps have a truncated cone design made up of three stainless steel circular rings, covered by a 3.8 cm soft mesh (Figure 5.1A). There are three circular entrances, held open by 7.6 cm-diameter stainless steel rings, on the sides of each trap. There is no one-way door or other device which prevents prawns or other organisms from escaping the trap, but based on *in situ* video observation, escapement rates from traps are low (B. Favaro, unpublished data). The selectivity of these traps is determined largely by the diameter of the entrances (preventing large organisms from entering), as well as the size of the mesh on the trap (preventing escapement). However, since mature spot prawns and juvenile rockfish are similar in size, it is unlikely that a simple size adjustment of the mesh or entrances will exclude rockfish while maintaining prawn catch rates. We nevertheless tested two entrances of smaller diameters (6.4 cm and 7.0 cm) than the standard entrance ring (7.6 cm).

We also designed BRDs based on extensive observations of prawns and rockfish interacting with traps. Observations were made of animals in experimental aquaria, in the wild by scuba divers, and through analysis of video collected by a deep-water camera apparatus attached to a prawn trap (Favaro et al. 2012). Prawns move in three ways: by walking on their pereiopods (legs), swimming using their abdominal pleopods, and by eliciting a retrograde escape response where they flick their abdominal tail to escape predators (Bauer 2004). When prawns approach and enter prawn traps, they predominantly do so by walking along the mesh, up the side of the trap, and through the openings (BF, personal observation, Video D.1). By contrast, rockfish swim using a combination of labriform and subcarangiform swimming modalities (Sfakiotakis et al. 1999), in which they use their pectoral fins for slow, precise movement, and their tail fins for fast travel. We therefore designed a series of bent-tunnel BRDs which were built to restrict the ability of rockfish to move within the opening (i.e., by requiring an unnatural bend of the fish's body), while providing a ladder-like structure for prawns to crawl over. These devices attached to the trap entrance rings, and comprised a series of rings that formed a curved tunnel (Figure 5.1B). We used rings rather than a solid bent tunnel (such as with a PVC elbow) because prawns appeared to have difficulty crawling over smooth plastic surfaces (BF, personal observation). We tested three bent-tunnel BRDs of increasing length, i.e. with four, five, or seven rings (Figure 5.1B). These BRDs were hand-built by cutting a 7.6 cm stainless steel pipe into small rings, which were spot-welded in place. We tested PVC versions of the openings in a pilot study, but they were not durable and did not retain their shape during normal fishing use.

5.3.2. Field test

Between June and August 2010, we field tested five BRDs (i.e., two entrance-ring and three bent-tunnel variants) as well as unmodified traps (control) to identify the BRD design that offers the best trade-off between minimizing bycatch while maintaining prawn catch. From a 9.8 m-long research vessel, we deployed gear in “strings” which contained 10 traps connected to a single line weighted with one cinder block at each end. We deployed a total of 154 strings (i.e., 1540 traps). The most common configuration of traps in each string was: two control traps (7.6 cm entrances), one trap with 7.0 cm entrances, one trap with 6.4 cm entrances, and two of each BRD variant (4-ring, 5-ring, and 7-ring), with the order of traps being randomized within each string. Early in the study, we included PVC variants of the BRDs (so that each string had one steel and one PVC variant of each BRD type) but all PVC variants were eventually discarded because they were not durable (total of 155 PVC-BRD traps excluded). In addition, we included the 6.4 cm variant one week into the study, when we became curious about a more extreme reduction in trap opening size. One string of gear was lost during the study, while another was carried several kilometers from its original deployment site, and so its data were discarded. Three traps also became detached from one string line and were lost. Data from 1362 traps were therefore included in the present analysis (322 control traps (i.e., 7.6 cm entrances), 256 traps with 7.0 cm entrances, 145 traps with 6.4 cm entrances, 214 traps with 4-ring tunnels, 214 traps with 5-ring tunnels, and 211 traps with 7-ring tunnels).

We deployed gear in two regions of southern British Columbia (Figure D.2): Howe Sound, near Vancouver (49°25’30”N, 123°20’00”W), and the southern Gulf Islands, near Sidney (48°39’00”N, 123°23’00”W). We selected deployment sites based on personal experience of prawn fishing, input from commercial fishers, and local knowledge. We baited all traps with standard commercial prawn bait, which is made of fishmeal pellets (Rutherford et al. 2004a). Deployment depths ranged from 50 to 120 m (mean ± 1 SD: 82 ± 17 m, determined by depth sounder), and strings were deployed for an average of 26.5 ± 11.6 h (range: 12.8 to 98 h) before retrieval, thus matching commercial fishing conditions (~ 24 h, (Fisheries and Oceans Canada 2011).

Traps were retrieved with an electric Anchormax capstain winch, which pulled strings at a steady rate of approximately 0.2 meters per second. We recorded the number of individuals of each species caught, as well as the total weight of each species caught per trap. For fishes we recorded individual fish weight as well as total length, body width, and body depth at the deepest point. In addition, we recorded the carapace length (i.e., the distance from the posterior orbital rim to the median dorsal carapace edge, (Butler 1980) of each captured prawn.

5.3.3. Statistical analysis

To compare rates of fish bycatch and prawn catches across gear designs, we used generalized linear mixed-effects models (GLMMs) and linear mixed-effects models (summary of models; Table D.3; Bolker et al. 2009). GLMMs are a powerful tool for data analysis in ecology, and their use has become widespread because they can handle data that violate many assumptions necessary for simple linear models (Zuur et al. 2009). In addition, the nested nature of our experimental design (i.e. catch data nested within strings) can be incorporated in the models as random effects. We displayed most of our data using beanplots, a boxplot-like method of data presentation that shows all values recorded in a given category, while plotting an estimated distribution around the data (Kampstra 2008). In beanplots, there is no arbitrary exclusion of outliers – rather, all data are displayed along with a mean for easy comparison between groups (Kampstra 2008).

The first suite of models examined the bycatch of all fishes across fishing gears. There were too few captures of rockfish (see Results) to examine this group separately from other fish families. First, we examined the rate of fish bycatch per trap by testing the fixed effects of trap variant (control, 7.0 cm opening, 6.4 cm opening, 4-ring, 5-ring, and 7-ring tunnels) and fishing region (Howe Sound and Gulf Islands) while incorporating the random effect of string identity. Differences in overall catch rates between regions could make the interpretation of differences among trap variants within region difficult. Therefore, when catch rates varied significantly between regions, we repeated the analysis separately for each region, testing variants against unmodified traps. We assumed a negative binomial distribution of fish catch rates (verified with the Curvefit function in the VCD package in R - Likelihood Ratio: x^2 = 1.62, df = 2, p = 0.45, Meyer et al. 2011), and conducted the analyses using the glmmADMB package in R (Skaug et al. 2011). We also examined the body depth and body mass of fishes caught across trap variants and regions (both fixed effects, with string as a random effect) to determine potential underlying reasons of any exclusion attributed to the BRDs. These two variables were distributed normally, enabling us to construct linear mixed-effects models using the simpler NLME package (Pinheiro et al. 2011). We log-transformed fish body weight to improve the model fit.

While fish bycatch rates are reported mostly by count in the commercial fishery, prawn catch is reported by weight. Therefore, in our analysis of prawn catch, we examined the weight of prawns caught per trap rather than prawn number. We used a linear mixed-effects model to test the effects of trap variant and fishing region on weight of prawns caught per trap, while incorporating string identity as a random effect. Since there was a large difference in prawn catch rates between fishing regions (see Results), we conducted a separate analysis for each region. Finally, we tested the effect of trap variant and fishing region on the body sizes (i.e. carapace length) of prawns caught.

We then performed a catch comparison analysis, following the procedure outlined in Holst and Revill (2009), to test whether body size affected the likelihood of being caught in BRD gear versus control gear. In this procedure, GLMMs are used to plot the relationship between proportion of catch in traps of each BRD type versus control traps, and the body size of organisms caught in the gear (Holst and Revill 2009). For prawns, our unit of body size was carapace length, while for fishes it was body depth, as we expected the ability of fish to enter traps to be limited by the length of their dorsoventral axis. This framework is designed to highlight the differences in catch between unmodified and modified traps, and it tests the proportion of catch across the spectrum of observed body sizes which occurred in each trap variant versus control traps. Variability associated with sampling over multiple deployments of our gear is incorporated in the model as a random effect. We began by fitting polynomial regressions followed by reductions until all terms were significant. We used the glmmPQL function from the MASS package (Venables and Ripley 2002) in R to conduct this analysis separately for fishes and spot prawns.

5.3.4. Video deployment and analysis

While catch data can provide information on the effectiveness of each BRD, they do not reveal the mechanism of action, i.e why BRDs may be increasing or decreasing bycatch (Sala et al. 2011). We therefore deployed traps equipped with a specially designed underwater camera apparatus (Favaro et al. 2012) to compare the performance of control and BRD-equipped traps. We analyzed video collected from five deployments of our underwater camera apparatus at a location in Howe Sound (Figure D.2). Three deployments were conducted with unmodified, 7.6 cm opening traps, and two were equipped with 5-ring BRDs. Video duration ranged from 12.1 to 13 h per deployment (Table D.4).

We recorded data from our videos by counting the number of prawns which entered the field of view (termed "approaches"), and the number which attempted to enter ("attempts"). An attempt was recorded every time a prawn climbed onto the mesh immediately surrounding a trap opening. The time between prawn contacting the mesh and entering the trap through an opening ring was recorded as the "time to enter." Alternatively, if the prawn did not enter, and instead crawled or swam away from the trap opening after starting its approach, the attempt was recorded as a "failure to enter". Using these data, we calculated the average proportion of successful entry attempts, as well as the mean time to enter, across video deployments of control and 5-ring BRD-equipped traps, and compared them using t-tests of unequal variances (Ruxton 2006). In addition, we took qualitative notes about the prawns' entry process, focusing on identifying design issues that could be affecting prawn entry into the traps.

5.4. Results

5.4.1. Rockfish bycatch

We caught a total of only six rockfish across all traps. Three were caught in unmodified (control) traps (one greenstriped rockfish, *Sebastes elongatus*, two quillback rockfish, *S. maliger*), two in traps with 7.0-cm entrances (one quillback rockfish, one vermillion rockfish, *S. miniatus*), and one (Puget Sound rockfish, *S. emphaeus*) in traps with 6.4-cm entrances. We caught no rockfish in 639 deployments of tunnel-equipped traps.

5.4.2. Overall fish bycatch

We caught a total of 118 individual fish across the entire study, which comprised the aforementioned four species of rockfish as well as 17 other species or families of fish (Table D.5). Fish body weight ranged from <50 g to 900 g (mean ± 1 S.D. = 187 ± 184 g), and body depth ranged from 0.8 cm to 7.8 cm (mean ± 1 S.D. = 4.3 ± 1.4 cm).

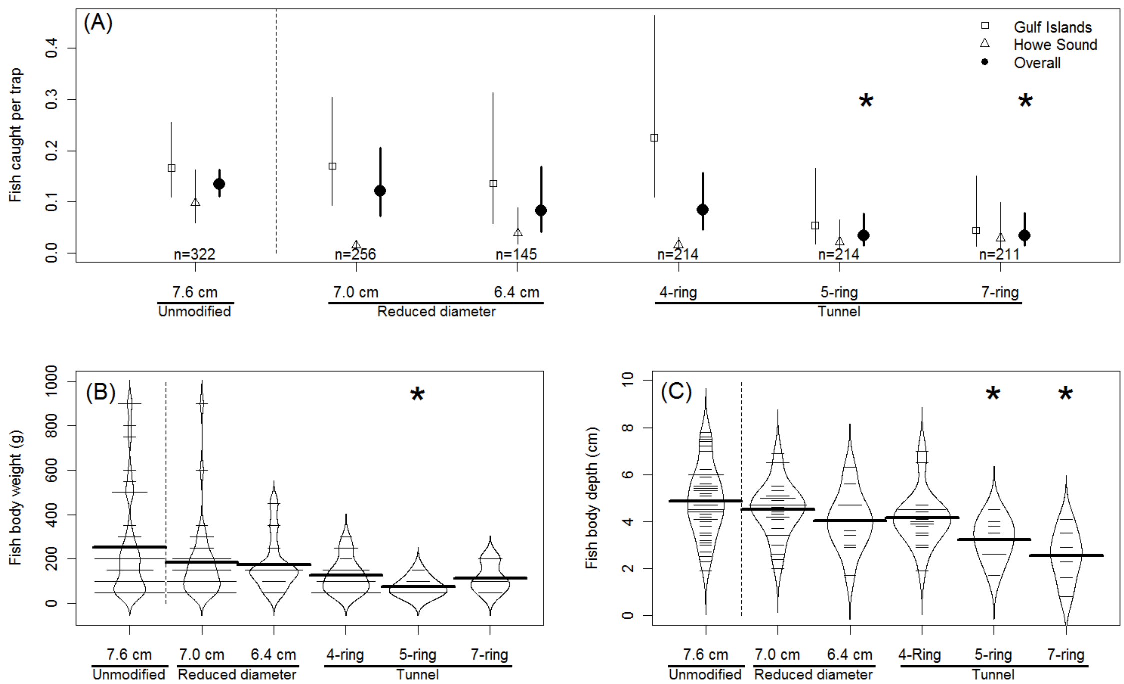

Figure 5.2: Characteristics of fish catch in unmodified (control) spot prawn traps and traps equipped with either reduced-diameter (7.0 cm or 6.4 cm entrances) or tunnel (4, 5 or 7 ring) bycatch reduction devices. (A) Fish catch rate (all fish combined; numbers caught per trap); means are shown with 95% C.I. (assuming a negative binomial error distribution) for each study region separately and combined. Asterisks indicate statistically significant differences (p < 0.05) from unmodified traps. (B) Fish body weight (g) and (C) fish body depth (cm), shown as bean plots of distributions. Each thin black line indicates a data point at a given Y value, with line width increasing with the number of observations per value. The thick black lines indicate group means. Asterisks indicate statistically significant differences (p < 0.05) from unmodified traps, shown to the left of the vertical dashed line.

Overall fish bycatch rates were 69% and 68% lower in the traps equipped with 5- and 7-ring BRDs, respectively, than in control traps (GLMM: 5-ring, $\beta = -1.178$, S.E. = 0.417, $z = -2.82$, $p = 0.005$; 7-ring, $\beta = -1.137$, S.E. = 0.419, $z = -2.72$, $p = 0.006$; Figure 5.2A). Fish catch rates in traps with other BRDs (i.e. both entrance diameter reductions, and 4-ring tunnel) did not differ significantly from those in the control traps. Fish capture rate in Howe Sound region was only 32% of that of the Gulf Island region ($\beta = -1.150$, S.E. = 0.292, $z = -3.94$, $p < 0.001$). Patterns of fish catch were therefore examined separately in each region. Within the Gulf Islands, the 5-ring and 7-ring designs reduced fish capture by 66 and 72%, respectively, relative to unmodified traps (Table D.6). In comparison, in Howe Sound, where only 28 fish were caught, all designs except the 6.4 cm-opening trap produced significantly lower fish bycatch rates than unmodified traps (reductions in fish catch: 7.0 cm; 87%, 4-ring; 86%, 5-ring; 78%, 7-ring; 71%; Table D.6).

Trap design also had significant influence on both the average body weight (Figure 5.2B) and body depth (Figure 5.2C) of trapped fishes. Fishes were, on average, 38% lighter in traps with 5-ring BRDs (LME: $\beta = -0.962$, S.E = 0.290, $t = -3.317$, $p = 0.002$), and had significantly shallower average body depth in traps with 5-ring and 7-ring entrances than in the control traps (LME: 5-ring, $\beta = -1.656$, S.E. = 0.495, $t = -3.347$, $p = 0.002$; 7-ring, $\beta = -2.316$, S.E = 0.521, $t = -4.450$, $p < 0.001$). Fish body weight and body depth did not vary across the other trap variants or between regions.

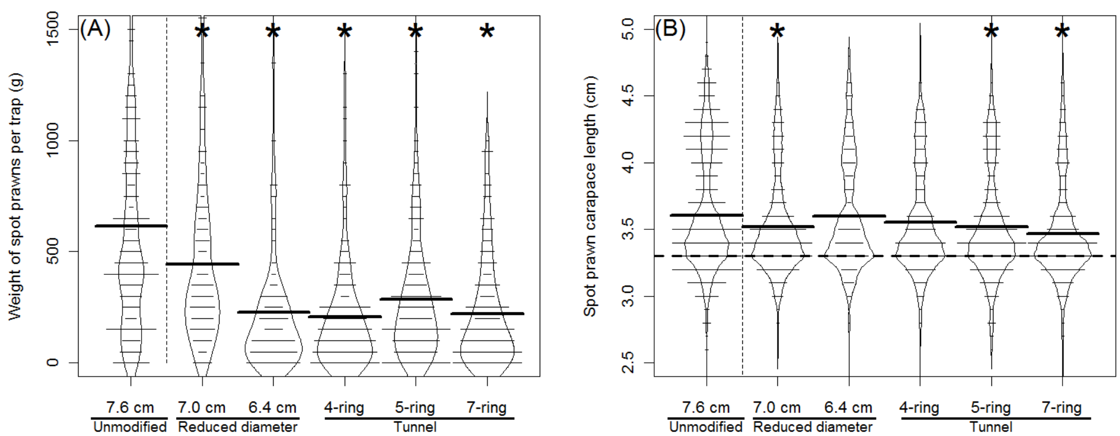

Figure 5.3. Beanplots comparing the distributions of (A) total catch weight and (B) individual carapace length of prawns caught per trap in Howe Sound, British Columbia, in unmodified (control) spot prawn traps and traps equipped with either reduced-diameter (7.0 cm or 6.4 cm entrances) or tunnel (4, 5 or 7-ring) bycatch reduction devices. The thick black lines indicate group means. The horizontal dashed line in (B) indicates the legal minimum size in the commercial fishery. Asterisks indicate significant differences (p < 0.05) relative to unmodified traps, shown to the left of the vertical dashed line. The results for the Gulf Islands region are not shown because prawn catches were generally low.

5.4.3. Prawn catch

Every trap variant caught fewer prawns overall than unmodified traps (Table D.6). The catch of prawns, in terms of total weight per trap, was greater in Howe Sound than in the Gulf Islands (mean ± 1 S.D: Howe Sound = 337 ± 380 g $trap^{-1}$, Gulf Islands = 140 ± 250 g $trap^{-1}$; LME: $\beta = 241$, S.E. = 36, $t = 6.6$, $p < 0.001$), leading to separate further analyses for each region. In the Gulf Islands, the low overall catch masked any effect of BRDs because all traps experienced low catches, although the 5- and 7-ring BRDs caught fewer prawns than control traps (Table D.6). By contrast, in Howe Sound where overall prawn catch was greater, all BRDs reduced prawn catch significantly compared to the control traps (Figure 5.3A, Table D.6). Overall, prawns caught in Howe Sound were significantly smaller than those caught in the Gulf Islands region (LME: $\beta = -0.117$, S.E. = 0.031, $t = -3.751$, $p < 0.001$). In Howe Sound, captured prawns were significantly larger in unmodified traps than in the traps equipped with 7.0 cm entrances (LME: $\beta = -0.072$, S.E. = 0.030, $t = -2.385$, $p = 0.017$), 5-ring (LME: $\beta = -0.103$, S.E. = 0.027, $t = -3.802$, $p < 0.001$), and 7-ring tunnels (LME: $\beta = -0.138$, S.E. = 0.029, $t = -4.793$, $p < 0.001$, Figure 5.3B). In the Gulf Islands, only the 7-ring tunnels caught significantly smaller prawns than unmodified traps (LME: $\beta = -0.167$, S.E. = 0.051, $t = -3.296$, $p = 0.001$).

5.4.4. Body size selectivity of trap variants

In the traps equipped with 5-ring BRDs, the proportion of total fish catch (expressed by numbers) dropped markedly as fish body depth increased (Figure 4, Table D.7). These BRDs were therefore disproportionately selective against increasingly deeper-bodied fishes. For traps equipped with 4-ring and 7-ring tunnels, and 7.0 cm, and 6.4 cm entrances, there was no relationship between fish catch proportion and body depth (Figure 5.4). For prawns, all BRD designs caught less than 50% of the total catch (by weight), and size-selectivity was evident for all trap designs except for the traps with 6.4 cm entrances (Figure 5.4). For traps with 5- and 7-ring tunnels, which were best described by a linear GLMM, the relative proportion of observed catch (by weight) in BRD-equipped traps decreased as carapace length of prawns increased (Figure 5.4, Table D.7). The relationships between catch proportion and carapace size for traps with 7.0 cm entrances and 4-ring tunnels were best described by quadratic models, suggesting that they were selective against both small and large prawns, but allowed the entry of mid-sized prawns, though still to a lesser extent than unmodified traps (Figure 5.4, Table D.7).

5.4.5. Mechanisms of action of BRDs: Video evidence

In five deployments of the camera apparatus, we recorded 38 hours of video of control traps, and 25 hours of video of traps equipped with 5-ring BRDs. Across all videos, we observed a total of 2,380 spot prawns approach the trap (Table D.4). Of those, 777 attempted to enter, and 243 did so successfully (180 in control traps, 63 in 5-ring deployments). The average proportion of successful entries was higher in unmodified traps (mean ± 1 S.D. = 46% ± 12%) than in traps with 5-ring tunnels (18% ± 12%; t = 3.273, df = 2.9, p = 0.049). Furthermore, it took longer for prawns to complete entries in the modified traps (mean ± 1 S.D. = 106 ± 9 s) than in control traps (57 ± 19 s; t = --3.871, df = 2.9, p = 0.033).

In control traps, prawns could easily crawl through the openings uninhibited. Prawns attempting to enter modified traps crawled easily over the rings of the BRD, but often caught their rostrums between the two bottom rings of the tunnel entrance (Video D.8). It often took multiple attempts for prawns in modified traps to successfully negotiate the opening.

We observed a single quillback rockfish enter an unmodified trap on video. The rockfish spent 26 min ~20-30 cm above the trap before attempting (and failing) to consume a Dungeness crab (*Metacarcinus magister)*, which had climbed on top of the trap (Video D.8). After this interaction, the rockfish remained above the trap for six more minutes before entering the trap through one of the openings. While inside the trap, the rockfish attempted to consume trapped prawns twice, but both attempts failed. The rockfish was present in the trap for the remainder of the video, but it escaped in the 8 hours between termination of recording and gear retrieval. We observed no attempt by rockfish to enter the 5-ring-equipped traps.

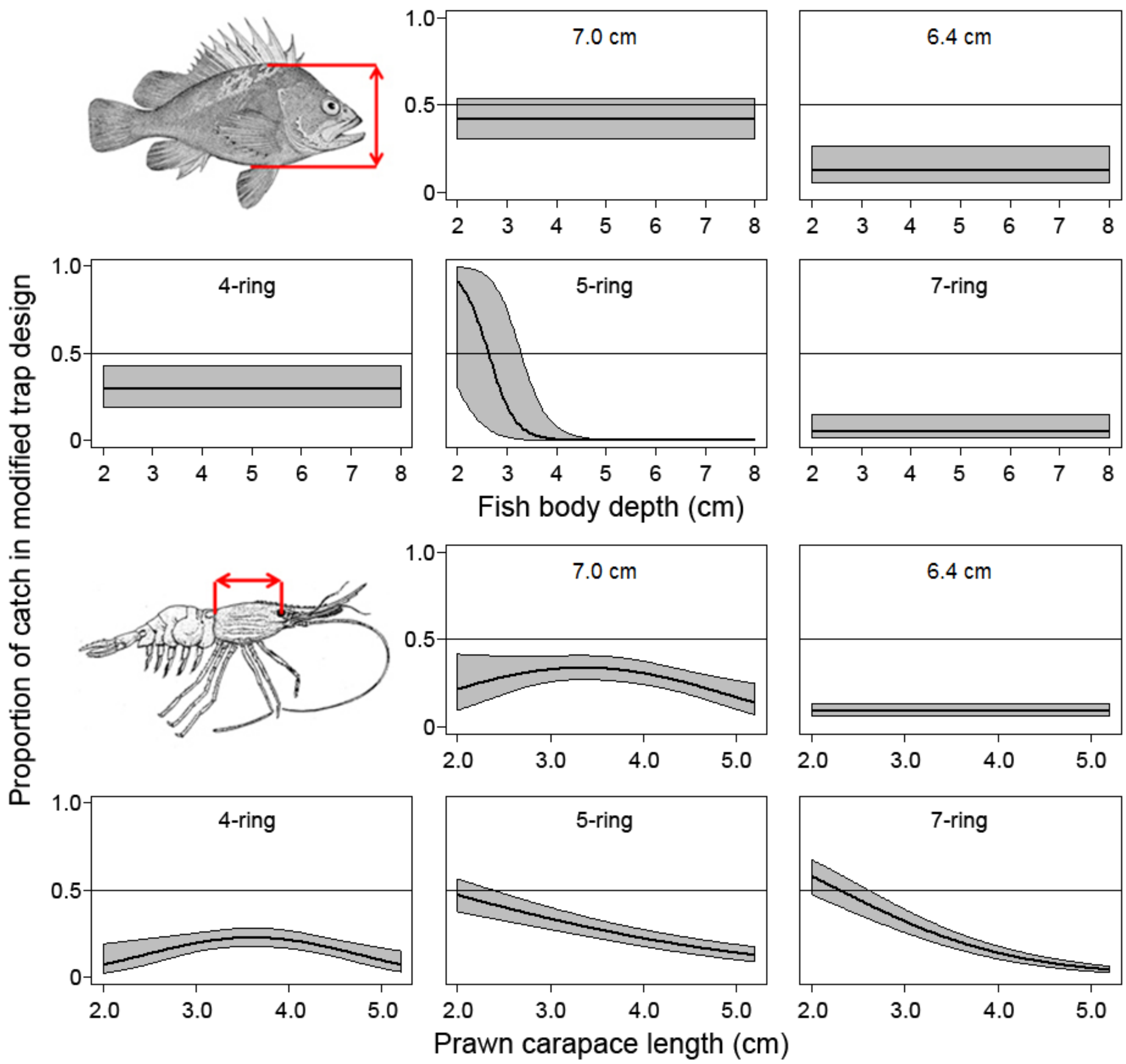

Figure 5.4: GLMM-modeled proportions of the number of fish and prawns caught in each gear type in relation to fish (top; body depth, the largest distance measured on the dorsoventral axis of the fish) or prawn (bottom; carapace length, the distance from the posterior orbital rim to the median dorsal carapace edge) body size. The horizontal line at y = 0.5 indicates that catch was similar between the modified and unmodified traps (i.e., 50% of the catch at any given body size occurred in each trap type). Lower values indicate that fewer organisms at a given length were caught in the BRD-equipped traps than in the unmodified traps. The thick black lines show the modeled means, while the grey shading indicates the 95% C.I. Line drawings of fish and prawn from (Fisheries and Oceans Canada 2011) and (Whitney 1911), respectively.

5.5. Discussion

In this study, we tested potential technical solutions to a bycatch problem in a trap fishery. To be adopted in a fishery, a BRD should satisfy three main conditions: it should achieve reduction in bycatch, it should maintain a target catch, and it should be practical for use in the fishery (e.g., durable, minimal alteration to fishing process). If these three criteria are met, the BRD will achieve the goal of maintaining a fishery's profitability while achieving conservation outcomes. Our tunnel BRDs appear to meet the first and third criteria: we found that novel bent-tunnel devices, which can be easily attached to standard commercial prawn traps, were effective at excluding rockfish and other species of fish, and that BRDs with 5- and 7-rings were more selective than 4-ring devices. However, none of the BRDs tested here met the second criterion. We found that any modification, including a simple reduction in the size of the trap openings, reduced significantly the capture of prawns. Our *in situ* observations of prawns interacting with traps give us insights into modification needed to develop an optimal BRD that can prevents fish entry into traps while maintaining prawn catches.

Our bent-tunnel entrances, particularly the 5- and 7-ring BRDs, were highly effective at excluding fishes from prawn traps, compared to unmodified traps. They were also better at excluding fishes than simple reductions in opening size, in terms of both the numbers and sizes of fish caught. The 5-ring tunnel BRD was especially selective against larger fishes of the sizes that correspond to the juvenile rockfish most commonly caught in commercial traps (BF, unpublished data). The complete exclusion of rockfish by the bent-tunnel BRDs is especially important because of the precarious state of many rockfish populations (Yamanaka and Lacko 2001, Love et al. 2002). The catch rates of rockfish in our unmodified traps was low but comparable to rates observed in the commercial fishery: the average rockfish encounter rate in the commercial fishery from 2002-2008 was 0.004 rockfish per trap in Howe Sound, and 0.002 and 0.008 for regions within the Southern Gulf Islands (Rutherford et al. 2010). The lower rate of capture in tunnel-equipped traps also extended to other fish species of commercial interest, such as Pacific cod (*Gadus macrocephalus*), which like rockfish are sensitive to barotrauma-induced mortality (Nichol and Chilton 2006). Those fish species which were not excluded by BRDs, such as small sculpins and eelpouts, are likely of less concern owing to their lack of a swim bladder and apparent ability to survive the capture and discard process (Berghahn et al. 1992).

Table 5.1: Adjusted fish bycatch rates for each trap design, accounting for the increased fishing effort which would be required to maintain existing catch rates, assuming full use of each BRD.

Design	Mean number of fish $trap^{-1}$	Average catch of prawns $trap^{-1}$ (g)	Proportion of prawn catch $trap^{-1}$ relative to unmodified design	Proportion of fishing effort required to maintain existing prawn catch	Fish bycatch relative to unmodified traps (%)
Unmodified	0.134	370	1.000	1.000	100.0
7.0 cm entrance	0.121	246	0.665	1.504	135.8
6.4 cm entrance	0.083	184	0.497	2.011	124.6
4-ring tunnel	0.084	192	0.519	1.927	120.8
5-ring tunnel	0.033	229	0.619	1.616	39.8
7-ring tunnel	0.033	174	0.470	2.126	52.4

From a product design perspective, the tunnel BRDs appeared to be practical to use in a commercial fishery. First, they are easy to attach to existing traps, hence fishers would not have to fully replace their usual complement of 300-500 traps. Second, the devices do not require alteration in fishing behaviour, so there is little risk that improper use will reduce the effectiveness of these devices. Third, they are extremely durable, and we experienced no damage to our devices which were used daily across two months of field study. Finally, since these devices attach to the inside of the traps only, they present no risk of snags or entanglements during the deployment and retrieval processes.

The reduction in fish bycatch in traps equipped with BRDs came at the expense of reduced prawn catches. The difficulty in maintaining prawn catch with BRDs was highlighted by the results from the BRDs with smaller entrances. Even the traps equipped with 7.0 cm entrances, which is a mere 0.6 cm reduction in opening diameter, yielded a 28% reduction in prawn catch in Howe Sound. The large effect on prawn catch of such a minor trap modification suggests that most physical devices that hamper slightly prawn entry into traps are likely to negatively affect prawn catch to some degree. Our *in situ* video data gave us insights into how such difficulties arise, at least with bent-tunnel BRDs. A high proportion of prawns that attempted to enter traps with tunnel entrances failed to negotiate the bend because their rostrum got stuck between the rings of the BRDs. As a result, it took substantially longer for prawns to complete a successful entry into modified than unmodified traps. It appears that to facilitate prawn entry, we should retain the lattice-like structure which allows prawns to crawl into the trap, but we need to develop a method to prevent rostrum entanglement. One possible solution might be to use a tunnel which is solid and smooth on top (on the outer bend) but ringed on the bottom (on the inner bend).

There may be alternative (non-design) ways for the fishing industry to cope with the use of BRDs that reduce the catch of target organisms. One possibility for the prawn fishery might be to extend the short fishing season to allow fishers to accumulate catches similar to those obtained without BRD-equipped traps. However, increasing overall fishing effort would also increase the absolute amount of bycatch produced by the fishery (Hall and Mainprize 2005). Simple back-of-the-envelope calculations (Appendix D.9) suggest that, given the BRD-specific observed reductions in prawn catch and the corresponding BRD-specific increases in fishing effort required to compensate for these reductions, the 5-ring and 7-ring BRDs would still achieve reduced fish bycatch relative to unmodified traps (Table 5.1). In fact, the 5-ring BRD appears to be the best option, extending the fishing season by ~60% but still producing only 39% as much fish bycatch as the current fishery (Table 5.1). By contrast, traps with reduced opening sizes would produce substantially more fish bycatch than unmodified traps during extended fishing seasons (Table 5.1).

Another possibility to reduce the negative effect of BRDs on prawn catch may be to adopt a flexible, site-dependent use of the devices. This would be highly unusual, since when BRDs are used to improve catch specificity, they are usually mandated for use across the entire fishery (Broadhurst et al. 2012). However, the easily attachable design of our bent-tunnel devices could permit managers to require these devices only in areas where rockfish bycatch is known to be high, or where the tolerance for bycatch is low, such as in Rockfish Conservation Areas. A site-specific adoption rule could represent a compromise which would enable prawn fishers to access RCAs, where prawn catches are sometimes high (BF, personal observations) while maintaining the purpose of the protected areas as refuges from rockfish extraction (Yamanaka and Logan 2010).

In summary, we found that, as expected, simple reductions in entrance size did not reduce fish bycatch in the spot prawn fishery. In fact, if the fishery increased effort to compensate for the reduced prawn catches of these modified traps, fish bycatch could actually increase compared to the current fishery. However, tunnel-based BRDs appeared to be effective at excluding fishes, but in their current form also result in lower prawn catches that may be unacceptable to the fishing industry. Our *in situ* observations of deployed traps point to a potential modification to the tunnel BRDs which could mitigate the loss in prawn catch, but a redesigned BRD would need to be tested at a wide scale to verify its effectiveness. Recent studies have investigated gear modifications to reduce bycatch in traps, but these usually focus on enhancing escape rates of non-target species (though various forms of escape hatches; e.g. Boutson et al. 2009, Johnson 2010, Bury 2011) rather than preventing their entry. The much lower volume of work on BRD development for traps, compared to trawl and longline fisheries, may stem from the perceived high selectivity of traps. However, not all traps are highly selective (Alverson et al. 1994), and in some ecosystems and regions, trapping is a major contributor to overall catch and bycatch (Mahon and Hunte 2001, Shester and Micheli 2011). Our study demonstrates that BRDs have the potential to reduce bycatch, even for gear with relatively high specificity, but the design of such devices should be underpinned by a thorough understanding of the behaviour, distribution, and physical characteristics of the target and non-target species.

6. General Discussion

In this thesis I described a research and development process for BRDs in commercial fisheries. I conducted a global-scale analysis of work conducted by other researchers to solve the problem of elasmobranch bycatch in longline fisheries, and then focused in detail on a BRD research and development program for a local fishery. In assessing my own devices as well as the devices produced by others, I endeavoured to consider devices based on three criteria: the gear's effect on bycatch, the gear's effect on target catch, and the practicality of the gear for use in the commercial fishery. My hope is that this analysis will influence the way in which future researchers approach the development of new BRDs, as well as the manner by which they develop assessment programs prior to use in fisheries. This discussion places the findings of my thesis chapters in the context of the global effort to reduce bycatch in fisheries, and outlines how collaboration and technological innovation can build on my methods to facilitate future BRD development in other systems.

In the zeal to reduce bycatch in fisheries it is important to remember that it is still not clear that selective fishing would maintain ecosystem integrity more effectively than non-selective fishing (Hall et al. 2000, Zhou 2008). For example, selective depletion of large size classes (common in commercial fishing) can have a greater negative impact on fish stocks and ecosystem resilience than balanced exploitation across size spectrums (Law et al. 2012). Highly selective fishing can affect biodiversity metrics such as species richness and evenness more strongly than non-selective fishing (Rochet et al. 2011, Garcia et al. 2012). However, when I discuss bycatch as a problem to be mitigated (e.g., in the case of rockfish bycatch in prawn traps, seabirds on longlines, turtles in trawl nets or longlines, or cetaceans in any fishing gear), I deal with situations where retention of non-target species has been accepted as being undesirable due to conservation concern, species charisma, or other reasons.

Setting targets for BRD performance

The question of whether a given BRD is a success or failure can be difficult to answer, even with comprehensive study. For example, magnets and electropositive BRDs have been reported as success stories in popular media based on laboratory testing (Shapiro 2012). However, my analysis (Chapter 2) demonstrated that they are not yet effective based on the three dimensions of gear performance, which I repeatedly revisit in my thesis. My prawn trap BRDs would be deemed a success by conservationists concerned with rockfish, but the opinion of fishers and managers may differ based on the modified gear's negative impact on target catch. Resolving these conflicting opinions is an important aspect of BRD research.

Bycatch reduction is only one aspect of what should be explicit targets for evaluating BRD effectiveness. Hall and Mainprize (2005) predicted that if BRDs were fully implemented, global bycatch could be reduced between 25-64%. However, a substantial proportion of the gears they assessed did not report their target catch rates relative to unmodified gear. I found a similar deficiency in Chapter 2, with many papers not reporting target catch, or not adequately considering the practicality of the modified gear for use in commercial fisheries. I contend that three simultaneous BRD targets (i.e., bycatch, target catch and practicality) should be set to allow a full evaluation of BRD gear.

How should BRD targets be set? To determine clear objectives, future BRD projects should begin with a collaborative exercise between scientists, fishers, and managers to establish target goals for catch composition and gear performance. In some fisheries, such as Gulf of Mexico shrimp trawls which took sea turtles as bycatch, the priority was clear from the onset. Sea turtles were listed as endangered species in the US in 1978, making elimination of turtle bycatch through the use of Turtle Excluder Devices (TEDs) a major priority (Margavio et al. 1996). In the case of the prawn-rockfish system, the initial impetus was a reported commercial bycatch rate of 0.045 rockfish per trap (Rutherford et al. 2010). In determining whether my BRDs were successful at achieving conservation targets, there must be agreement on how many rockfish are acceptable to catch as bycatch, as well as a target for an acceptable reduction in prawn catch. This clarity is important; part of the difficulty in introducing TEDs to prawn fisheries in the United States was unclear or contradictory objectives (Tucker et al. 1997). Remarkably, clear bycatch reduction targets are rarely explicit in bycatch literature; it is generally taken as self-evident that bycatch must be reduced to as low a level as possible. While this sentiment is understandable, the lack of clear goals can make it difficult to assess the effectiveness of BRDs in a fishery.

The roles of researchers and fishers in BRD development and testing

The importance of combining at-sea testing and *in situ* observations of fishing gears is shown throughout my thesis. The use of TrapCam (Chapter 3) served two purposes: it provided me with ideas on potential BRD designs that were based on the movement and behaviours of animals observed interacting with traps under natural conditions, and it facilitated the evaluation of my BRDs by directly comparing the catch process with that occurring in unmodified traps (Chapter 4). The apparatus enabled me to show us where in the capture process the BRDs were inhibiting prawn entry, giving potential options for revised designs. The utility of *in situ* video observation has been recognized for evaluating the performance and ecological impacts of fishing gear (Auster et al. 1996, Olla et al. 2000, Munro and Somerton 2002), and this thesis demonstrated their utility for testing BRDs as well. Future innovations in miniaturized cameras may permit this approach in gears where this was previously impossible (such as in hook-and-line fisheries).

Trained scientists are a critical component of performing unbiased assessments of novel BRDs (Jennings and Revill 2007). Fisher perceptions of the impact of their fishing activities can underestimate overall impact of their fishery (Tucker et al. 1997). In addition, fishers usually lack the formal training needed to set up unbiased field experiments – a role which scientists can fulfill. My own experience with the BC prawn fishery reflects this reality; I had many informal interactions with fishers who believed they had developed a gear modification that could eliminate rockfish bycatch in the prawn fishery, but were unwilling or unable to share evidence or gear designs. Therefore, combining direct observation with an unbiased sampling program remains the most effective way to assess the fishing performance of modified gears.

However, my thesis also highlights the importance of collaboration in the process of developing BRDs. One way to accomplish this is to involve fishers in the research, development, and assessment of any BRD technology. Notwithstanding the example above, there is ample evidence that fishers themselves are the most likely candidates to produce effective BRD technology (Hall et al. 2007). The presence of fishers on R&D teams serves three purposes. First, their experience of working with gear enables them to provide unique and practical input. Second, they can provide material support and funding for BRD testing. Third, their presence on the development team can provide researchers with an essential link to individuals in the fishery, facilitating the uptake of new gears and earning goodwill within the fishery to continue conservation research. Gears with an advocate within a commercial fishery are far more likely to be successfully integrated into the fishery itself than gears without fisher input (Watson et al. 1999, Campbell and Cornwell 2008). In addition, fishers are more likely to perceive the value of reducing bycatch if they are convinced that their activities are causing substantial mortality of the non-target species (Tucker et al. 1997). Turtle excluder devices (TEDs) are perhaps the most famous example of a successful class of BRD, but government-designed models faced substantial opposition when first introduced (Jenkins 2012). By contrast, TEDs designed by fishers were used with far less resistance, and even with some enthusiasm within the fishery (Jenkins 2010).

Providing incentives for fisher participation in BRD development should be a priority. Data-sharing within fisheries can be difficult, but given the right motivation fishers have shared data to reduce bycatch in fisheries (Griffith 2008, Howell et al. 2008). The WWF Smartgear Prize is an example of such an effort; it offers cash prizes to three promising BRD inventions in each round of competition, as determined by a panel of expert judges (World Wildlife Fund 2011). This competition has drawn 358 submissions of novel BRDs across the five competitions held so far (Mike Osmond, WWF Smartgear competition director, pers. comm.), and four of the five grand prizes have been issued to projects which have at least one fisher on the design team.

New technologies will enable quick and inexpensive fabrication of BRD prototypes, as well as rapid dissemination of new designs across the globe. Future studies could employ rapid prototyping through the use of 3-D printers. In order to build devices using a 3-D printer, the device must be rendered in a Computer Assisted Design (CAD) format, which is then translated through software to a product that can be manufactured on one's desktop (Dietsch 2011). This concept has existed for many years (Sachs et al. 1990), but only recently have the costs of desktop-based 3-D printers become accessible to non-commercial users. At the time of writing, desktop 3-D printers capable of "printing" the curved-tunnel BRDs tested in Chapter 5 can be purchased for USD $1000 (Dietsch 2011), and this cost will decline over time. While it may seem unlikely that fishers would engage in 3D rendering in the near-term, it is conceivable that as technology advances and such activities become mainstream, fishers might do so in the coming decades.

Conclusion

The results presented in this thesis, and the points considered in this discussion, lead to three key recommendations for future researchers in the field of bycatch mitigation. First, prior to embarking on any bycatch reduction project, clear goals about bycatch levels, target catch composition, and practicality of use (including cost) should be set in consultation with fishers and with managers. Second, an incentive structure must be devised to encourage fishers to share information on techniques and tools that they have developed to mitigate bycatch in their fishery. Third, researchers should develop and implement a testing strategy to assess the devices under field conditions, ideally with an *in situ* observation component to allow for further refinement of the gear at the conclusion of testing.

Technological development affects all fields of human activities, and fisheries are no different. Given the constraints imposed by regulations and the growing necessities of conservation, there will be an ongoing effort to improve the specificity of fishing gear. However, the enthusiasm for BRDs should be tempered by a commitment to conduct sound, objective research on the effectiveness of these devices. Collaborative design and assessment teams represent the most promising method for developing BRDs and ultimately deploying them within fisheries, but these assessments must be underpinned by clear objectives for the fishing gear. I hope that my thesis brings attention to these challenges, and that future researchers will follow this process to ensure that BRDs become a widespread and viable tool for sustainability, rather than a band-aid solution that fails to protect marine resources.

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Appendix A. Supporting material for Chapter 2

Appendix A.1. Search Terms and exclusion details. We performed our literature search using the database on Bycatch.org, Web of Science, and the Aquatic Sciences as Fisheries Abstracts Database Guide (ASFA) on March 7, 2013. For bycatch.org, we explored all papers including non-field studies, with the bycatch species field toggled to Elasmobranchs. Using Web of Science, we searched using the terms: TS=(shark OR elasmo*) AND TS=(bycatch OR discard) AND TS=(deterr* OR BRD OR Bycatch reducti*). Using ASFA, the syntax was all((shark OR elasmo*) AND (bycatch OR discard) AND (deterr* OR BRD OR Bycatch Reducti*)). The criteria for including papers in the meta-analysis are outlined in Figure 2.1.

File A.2. Excel spreadsheet listing all papers which appeared during our literature search, including detailed reasons for exclusion for each paper not included in the final analysis.

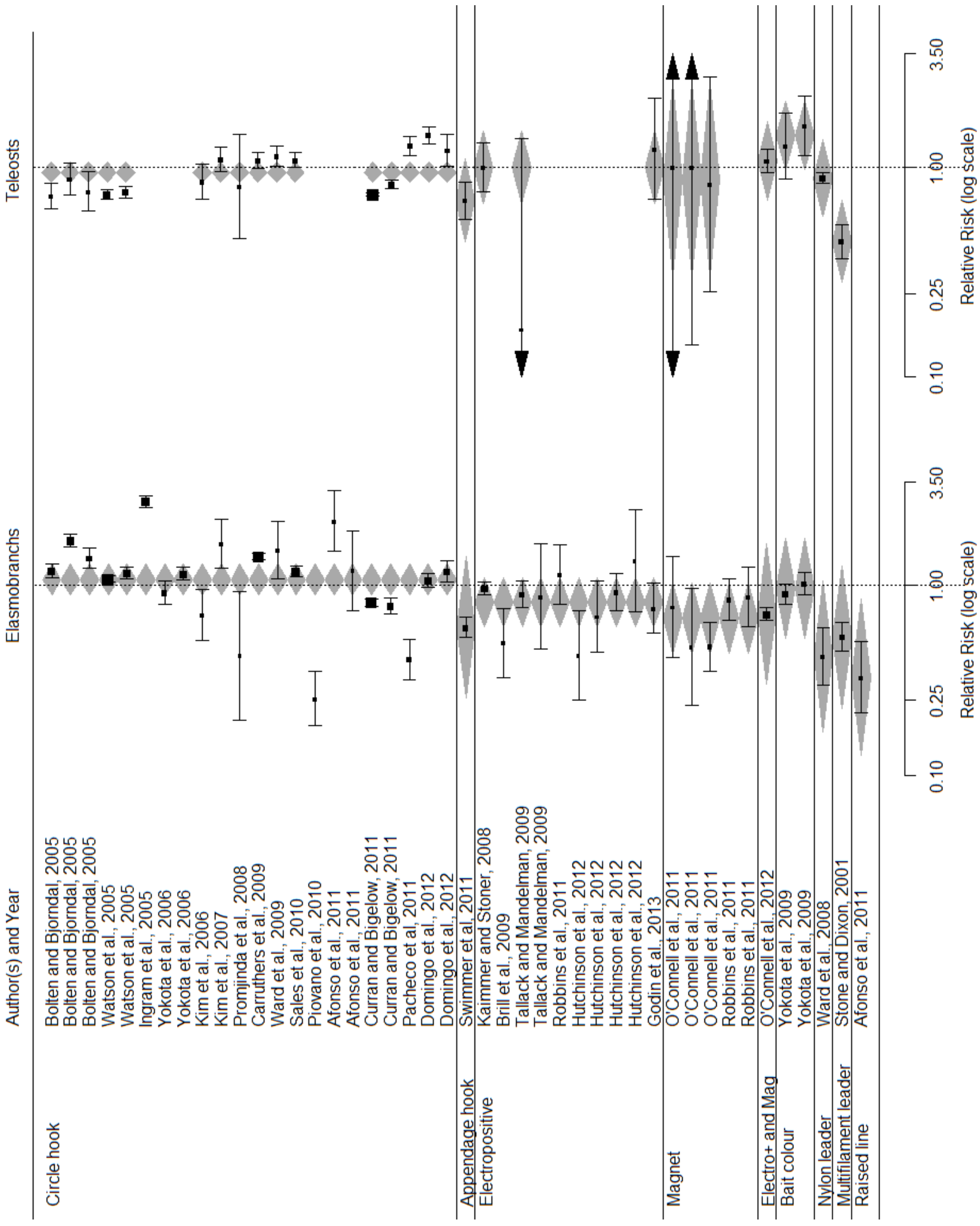

Figure A.3. Forest plot of each study. Black dots and bars represent the effect size for the individual study and 95% CI, respectively. Grey polygons represent the fitted effect size for each type of BRD. The relative size of black dot represents the weighting of that study.

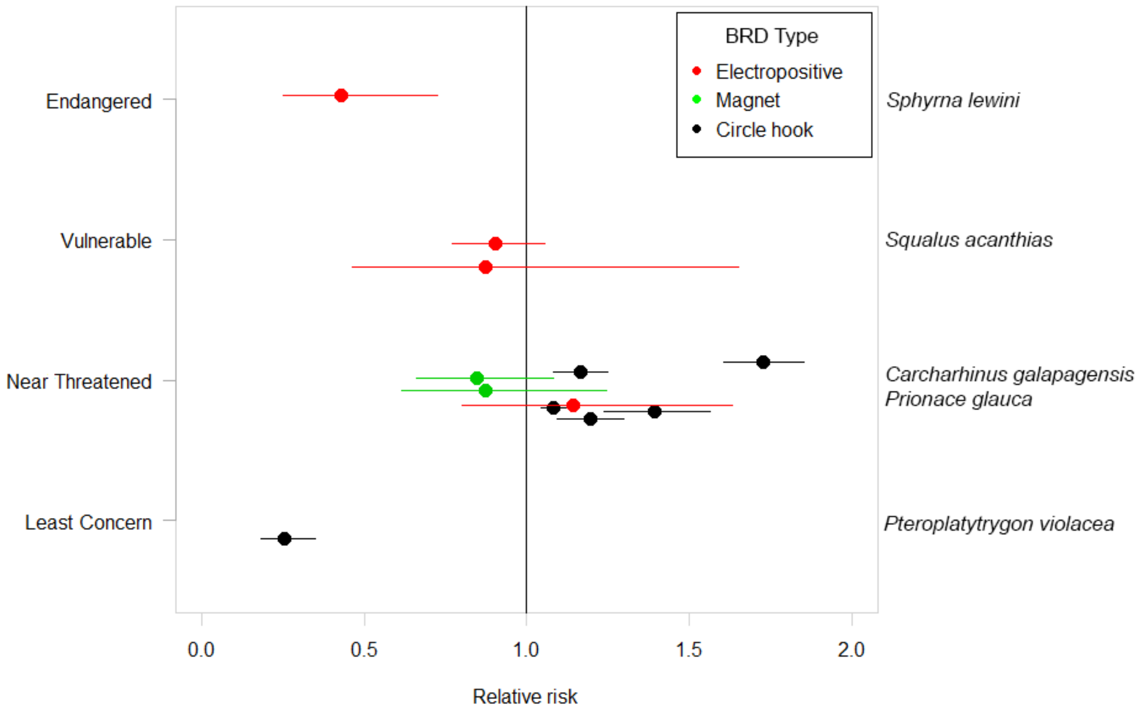

Figure A.4. Effects sizes of studies reporting a single species of shark or ray caught as bycatch, presented by IUCN Red List status. Data points are colour-coded by BRD type, and species within each status group are listed in italics. Bars represent 95% CI.

Studies included in each IUCN threat level:

Endangered: (Hutchinson et al., 2012)

Vulnerable: (Tallack and Mandelman, 2009)

Near Threatened: (Bolten and Bjorndal, 2005, Watson et al., 2005, Robbins et al., 2011)

Least Concern: (Piovano et al. 2010)

Table A.5. Full list of all shark and ray species reported across BRD types.

BRD Type	Order	Family	Species
Circle hook	Squaliformes	Somniosidae	*Zameus squamulosus*
	Lamniformes	Pseudocarchariidae	*Pseudocarcharias kamoharai*
		Alopiidae	*Alopias pelagicus*
			Alopias superciliosus
			Alopias vulpinus
			Alopias spp.
		Lamnidae	*Isurus oxyrinchus*
			Isurus paucus
			Lamna ditropis
			Lamna nasus
	Orectolobiformes	Ginglymostomatidae	*Ginglymostoma cirratum*
	Carcharhiniformes	Triakidae	*Mustelus canis*
			Mustelus norrisi
			Iago garricki
		Carcharhinidae	*Carcharhinus acronotus*
			Carcharhinus brevipinna
			Carcharhinus falciformis
			Carcharhinus granulose
			Carcharhinus isodon
			Carcharhinus leucas
			Carcharhinus limbatus
			Carcharhinus longimanus
			Carcharhinus luecus
			Carcharhinus obscurus
			Carcharhinus plumbeus
			Carcharhinus signatus
			Carcharhinus spp.
			Galeocerdo cuvier
			Prionace glauca
			Rhizoprionodon terraenovae
		Sphyrnidae	*Sphyrna lewini*
			Sphyrna lewini/S. zigaena
			Sphyrna mokarran
			Sphyrna zygaena
			Sphyrna spp.
	Myliobatiformes	Dasyatidae	*Dasyatis violacea*
			Pteroplatytrygon violacea
		Myliobatidae	*Manta birostris*

			Manta spp.
Appendage hook	Lamniformes	Pseudocarchariidae	*Pseudocarcharias kamoharai*
		Alopiidae	*Alopias spp.*
	Carcharhiniformes	Carcharhinidae	*Carcharhinus falciformis*
			Carcharhinus longimanus
		Sphyrnidae	*Sphyrna zygaena*
	Myliobatiformes	Dasyatidae	*Pteroplatytrygon violacea*
		Myliobatidae	*Mobula* spp.
Electropositive	Squaliformes	Squalidae	*Squalus acanthias*
	Lamniformes	Alopiidae	*Alopias pelagicus*
		Lamnidae	*Isurus oxyrinchus*
			Lamna nasus
	Carcharhiniformes	Carcharhinidae	*Carcharhinus galapagensis*
			Carcharhinus plumbeus
			Galeocerdo cuvier
			Prionace glauca
		Sphyrnidae	*Sphyrna lewini*
	Rajiformes	Rajidae	*Raja eglanteria*
			Raja rhina
	Myliobatiformes	Dasyatidae	*Dasyatis* spp.
		Gymnuridae	*Gymnura* spp.
Magnet	Carcharhiniformes	Triakidae	*Mustelus canis*
		Carcharhinidae	*Carcharhinus acronotus*
			Carcharhinus galapagensis
			Carcharhinus limbatus
			Carcharhinus plumbeus
			Negaprion brevirostris
			Rhizoprionodon terraenovae
		Sphyrnidae	*Sphyrna tiburo*
	Rajiformes	Rajidae	*Raja eglanteria*
	Myliobatiformes	Dasyatidae	*Dasyatis americana*
Combined E+/Mag	Squaliformes	Squalidae	*Squalus acanthias*
	Lamniformes	Lamnidae	*Lamna nasus*
	Carcharhiniformes	Carcharhinidae	*Prionace glauca*
	Torpediniformes	Torpedinidae	*Torpedo nobilana*
	Rajiformes	Rajidae	*Amblyraja radiata*
			Dipturus laevis
			Leucoraja ocellata
			Malacoraja senta

Bait colour	Squaliformes	Somniosidae	*Zameus squamulosus*
	Lamniformes	Alopiidae	*Alopias pelagicus*
			Alopias superciliosus
			Alopias vulpinus
		Lamnidae	*Isurus oxyrinchus*
			Lamna ditropis
	Carcharhiniformes	Carcharhinidae	*Prionace glauca*
	Myliobatiformes	Dasyatidae	*Dasyatis matsubarai*
			Pteroplatytrygon violacea
Monofilament nylon	Lamniformes	Alopiidae	*Alopias pelagicus*
			Alopias superciliosus
		Lamnidae	*Isurus oxyrinchus*
	Carcharhiniformes	Carcharhinidae	*Carcharhinus falciformis*
			Carcharhinus longimanus
			Carcharhinus tilstoni
			Carcharhinus spp.
			Galeocerdo cuvier
		Sphyrnidae	*Sphyrna* spp.
Multifilament nylon	Lamniformes	Lamnidae	*Isurus oxyrinchus*
	Carcharhiniformes	Carcharhinidae	*Prionace glauca*
	Myliobatiformes	Dasyatidae	*Pteroplatytrygon violacea*
Raised	Orectolobiformes	Ginglymostomatidae	*Ginglymostoma cirratum*
	Carcharhiniformes	Carcharhinidae	*Carcharhinus acronotus*
			Carcharhinus leucas
			Carcharhinus limbatus
			Galeocerdo cuvier
		Sphyrnidae	*Sphyrna lewini*
	Myliobatiformes	Dasyatidae	*Dasyatis americana*
		Myliobatidae	*Manta birostris*

Appendix B. Supporting material for Chapter 3

Appendix B.1. Total list of bycatch species caught in spot prawn research trap surveys.

Group	Common name	Scientific name
Other	Sponges	Porifera (Phylum)
	Anemone	Actiniaria (Order)
	Stony corals	Scleractinia (Order)
Molluscs	Divaricate nutclam	*Acila castrensis*
	Pacific bobtail squid	*Rossia pacifica*
	Octopus	Octopoda (Order)
	Smoothskin octopus	*Benthoctopus leioderma*
	Giant pacific octopus	*Enteroctopus dofleini*
	Pacific red octopus	*Octopus rubescens*
	Oregon hairy triton	*Fusitriton oregonensis*
Crustaceans	Branchiopods	Branchiopoda (Class)
	Eualid shrimp	*Eualus* spp.
	Shortscaled eualid	*Eualus suckleyi*
	Spiny-side shrimp (spiny lebbeid)	*Lebbeus groenlandicus*
	Sidestripe shrimp	*Pandalopsis dispar*
	Pink shrimp	*Pandalus eous*
	Coonstripe shrimp	*Pandalus danae*
	Pink shrimp (flexed)	*Pandalus goniurus*
	Humpback shrimp	*Pandalus hypsinotus*
	Pink shrimp (smooth)	*Pandalus jordani*
	Yellowleg shrimp	*Pandalus montagui*
	Bluespot shrimp	*Pandalus stenolepis*
	Slender bladed shrimp	*Spirontocaris holmesi*
	Dana's bladed shrimp	*Spirontocaris lamellicornis*
	Spirontocaris	*Spirontocaris* spp.
	Bristly crab	*Acantholithodes hispidus*
	Brown box crab	*Lopholithodes foraminatus*
	Squat lobster	*Munida quadrispina*
	Right-handed hermits	Paguridae (Family)
	Scaled crab	*Placetron wosnessenskii*
	Graceful rock crab	*Cancer gracilis*
	Dungeness crab	*Cancer magister*
	Red rock crab	*Cancer productus*
	Inshore tanner crab	*Chionoecetes bairdi*
	Redclaw crab	*Chorilia longipes*
	Pacific lyre crab	*Hyas lyratus*
	Graceful decorator crab	*Oregonia gracilis*
	Argids	*Argis* spp.
	Arctic argid	*Argis dentata*
	Crangons	*Crangon* spp.
	Paracrangons	*Paracrangon* spp.
Echinoderms	Sea lilies and feather stars	Crinoidea (Class)
	Starfish	Asteroida (Class)
	Rose starfish	*Crossaster papposus*
	Mud star	*Ctenodiscus crispatus*
	Leather star	*Dermasterias imbricata*
	Swift’s star	*Gephyreaster swifti*
	Sand star	*Luidia foliolata*

	Vermillion starfish	*Mediaster aequalis*
	Pink short-spined star	*Pisaster brevispinus*
	Purple starfish	*Pisaster ochraceus*
	Sunflower starfish	*Pycnopodia helianthoides*
	Solasterid seastars	Solasteridae (Family)
	Fish-eating star	*Stylasterias forreri*
	Phrynophiurida	Phrynophiurida (Order)
	Basket stars	Euryalina (Suborder)
	Sea urchins	Echinoidea (Class)
	Green urchin	*Strongylocentrotus droebachiensis*
Fish - other than rockfish	Spiny dogfish	*Squalus acanthias*
	Poachers	Agonidae (Family)
	Northern spearnose poacher	*Agonopsis vulsa*
	Blacktip poacher	*Xeneretmus latifrons*
	Sculpins	Cottidae (Family)
	Roughback sculpin	*Chitonotus pugetensis*
	Dusky sculpin	*Icelinus burchami*
	Threadfin sculpin	*Icelinus filamentosus*
	Spotfin sculpin	*Icelinus tenuis*
	Pacific staghorn sculpin	*Leptocottus armatus*
	Blackfin sculpin	*Malacocottus kincaidi*
	Sailfin sculpin	*Nautichthys oculofasciatus*
	Soft sculpin	*Psychrolutes sigalutes*
	Cabezon	*Scorpaenichthys marmoratus*
	Greenling	Hexagrammidae (Family)
	Kelp greenling	*Hexagrammos decagrammus*
	Whitespotted greenling	*Hexagrammos stelleri*
	Lingcod	*Ophiodon elongatus*
	Red brotula	*Brosmophycis marginata*
	Plainfin midshipman	*Porichthys notatus*
	Pacific cod	*Gadus macrocephalus*
	Deepsea cods	Moridae (Family)
	Walleye pollock	*Theragra chalcogramma*
	Decorated warbonnet	*Chirolophis decoratus*
	Shiner perch	*Cymatogaster aggregata*
	Pile perch	*Rhacochilus vacca*
	Northern ronquil	*Ronquilus jordani*
	Pricklebacks	Stichaeidae (Family)
	Eelpouts	Zoarcidae (Family)
	Pacific sanddab	*Citharichthys sordidus*
	Speckled sanddab	*Citharichthys stigmaeus*
	Flathead sole	*Hippoglossoides elassodon*
	Slender sole	*Lyopsetta exilis*
	English sole	*Parophrys vetulus*
Fish - rockfish	Copper rockfish	*Sebastes caurinus*
	Darkblotched rockfish	*Sebastes crameri*
	Splitnose rockfish	*Sebastes diploproa*
	Greenstriped rockfish	*Sebastes elongatus*
	Quillback rockfish	*Sebastes maliger*
	Yelloweye rockfish	*Sebastes ruberrimus*
	Sharpchin rockfish	*Sebastes zacentrus*

Appendix C. Supporting material for Chapter 4

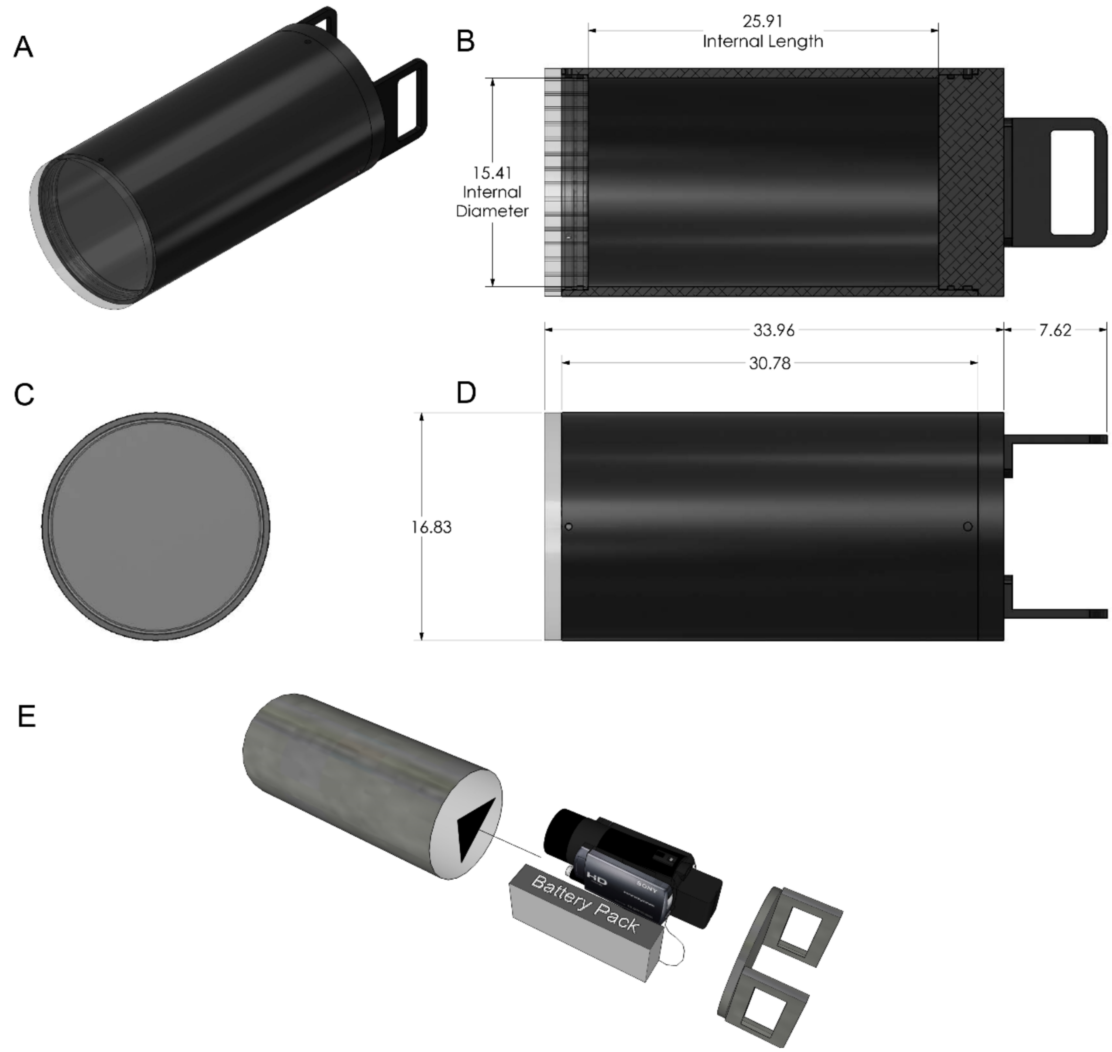

Figure C.1. Schematic of the pressure case showing A) an isometric perspective, B) a side view with internal dimensions (cm), C) a front view, D) a side view showing external dimensions, and E) a perspective demonstrating the orientation of the camcorder (camcorder model from Google 3D warehouse: HDR-CX550E) and battery pack within the pressure case. Pressure case models were supplied by A.G.O. Environmental Electronics, Victoria, British Columbia, Canada.

File C.2. Google Sketchup version 7.1 (http://sketchup.google.com) model of the TrapCam apparatus.

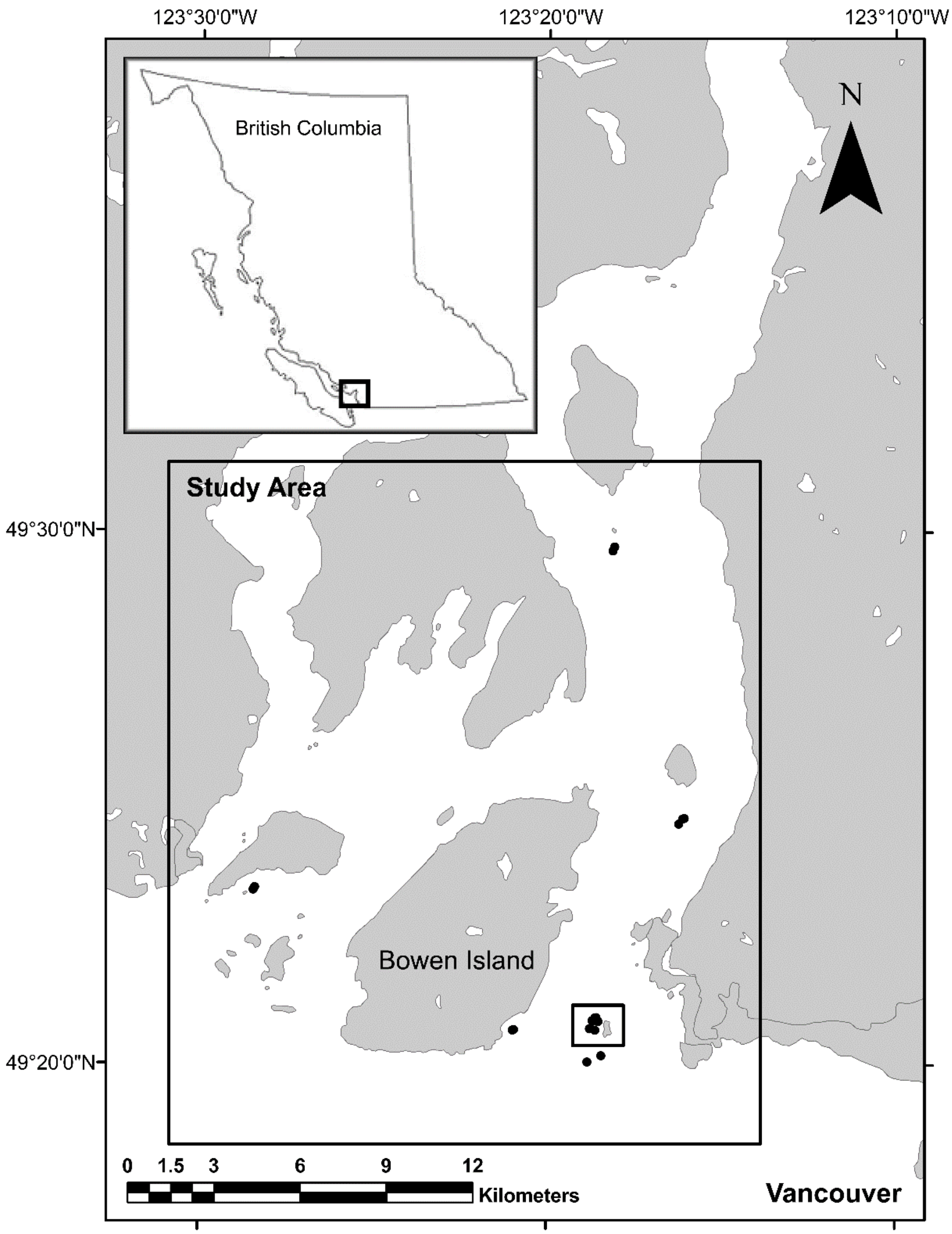

Figure C.3. Map of study area in Howe Sound, near Vancouver, British Columbia. Black dots indicate locations of gear deployment. Videos analysed in the present publication were from the cluster of videos recorded near Passage Island (small black square within study area).

Video C.4. Video demonstrating assembly and deployment of TrapCam.

Video C.5. Video showing underwater footage collected by TrapCam. This is a 20-second clip sampled 4 hours into a deployment.

Video C.6. Video showing a quillback rockfish within the prawn trap. The fish attempts to escape by swimming upward against the trap mesh. This 18-second clip was sampled 6.5 hours into a deployment.

Appendix D. Supporting material for Chapter 5

Video D.1. Video demonstrating the differences in movement between prawns and rockfish, and our rationale in developing the curved-tunnel BRDs.

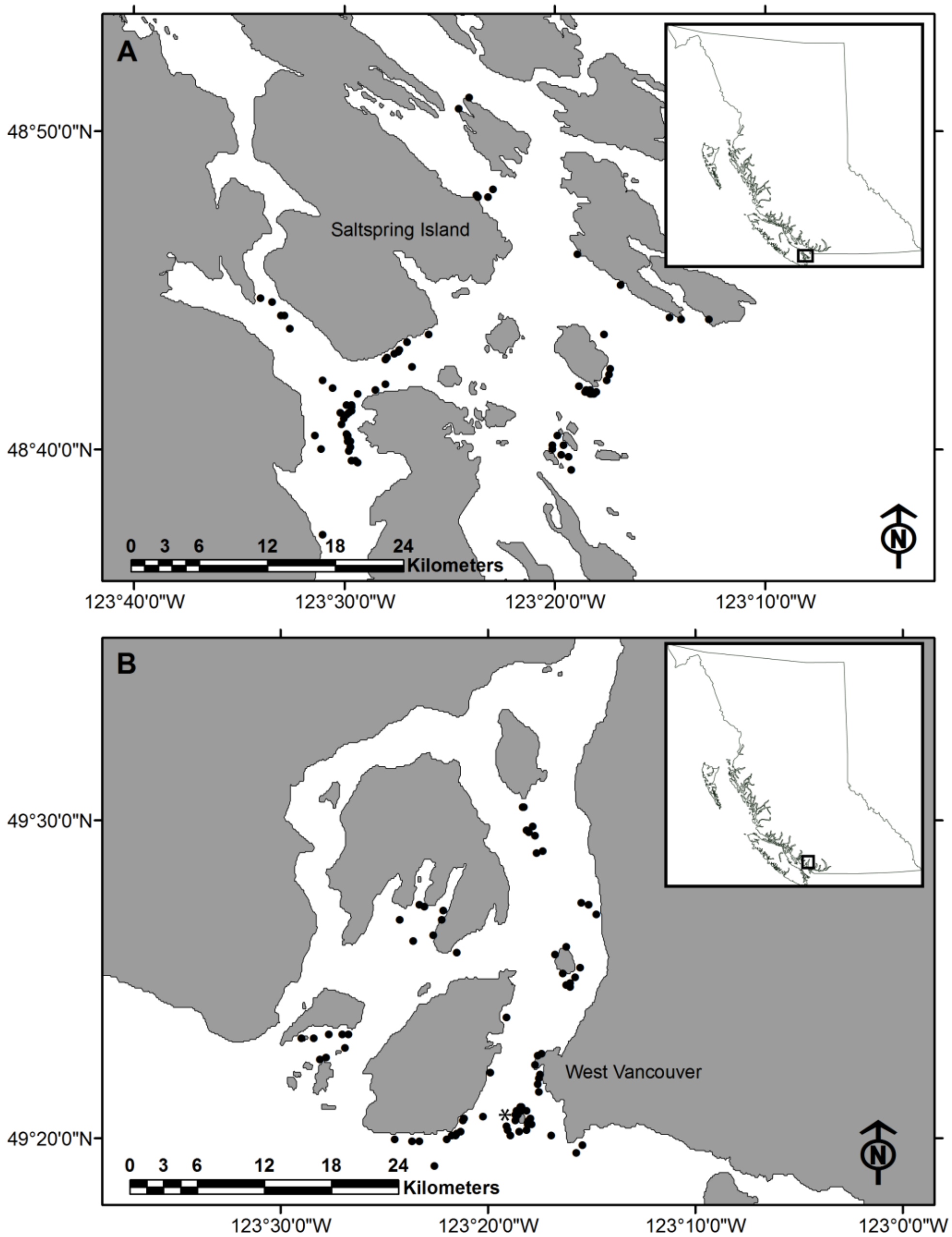

Figure D.2. Map of the study areas in (A) the southern Gulf Islands, near Sidney, British Columbia, and (B) Howe Sound, near Vancouver, British Columbia. Black dots indicate locations of gear deployments. The black star in (B) indicates the location of the camera-equipped trap deployments.

Table D.3. Summary of models used in the study of effectiveness of bycatch reduction devices on spot prawn traps.

Response	Fixed effects tested	Random effect	Distribution Assumed	R function
Count of fish per trap (overall)	Trap variant, fishing region	String ID	Negative binomial	glmmADMB
Count of fish per trap – Howe Sound	Trap variant	String ID	Negative binomial	glmmADMB
Count of fish per trap – Gulf Islands	Trap variant	String ID	Negative binomial	glmmADMB
Weight of prawn catch per trap (overall)	Trap variant, fishing region	String ID	Normal	Lme
Weight of prawn catch per trap – Howe Sound	Trap variant	String ID	Normal	Lme
Weight of prawn catch per trap – Gulf Islands	Trap variant	String ID	Normal	Lme
Log(fish body mass) per fish	Trap variant, fishing region	String ID	Normal	Lme
Body depth of fish	Trap variant, fishing region	String ID	Normal	Lme
Proportion of catch in trap variant (Catch comparison)	Prawn carapace length	String ID	Proportional data – binomial	glmmPQL
Proportion of fish catch in trap variant (Catch comparison)	Fish body depth	String ID	Proportional data – binomial	glmmPQL

Table D.4. Summary data collected from deployments of our video apparatus. All deployments occurred in Howe Sound at depths between 85 and 95 m.

Video ID	Trap type	Video Length (h)	# Approach	Total entry attempts	Successful entries	Failed entries	Proportion of successes per video
1	Control	12.9	347	37	22	15	0.59
2	Control	12.1	202	117	47	70	0.40
3	Control	13.0	877	291	111	180	0.38
4	5-Ring	12.2	232	136	13	123	0.10
5	5-Ring	12.9	722	196	51	145	0.26

Table D.5. Species caught in research prawn traps. The species in bold type is the sole target of the fishery under study.

Phylum	Class	Order	Family	Species	Common name
Cnidaria	Hexacorallia	Actiniaria			Anemone
Mollusca	Gastropoda	Littorinimorpha	Ranellidae	*Fusitriton oregonensis*	Oregon triton
		Nudibranchia	Tethydidae	*Melibe leonina*	Hooded nudibranch
	Bivalvia	Pterioida	Pectinidae		Scallop
	Cephalopoda	Octopoda	Octopodidae	*Enteroctopus dofleini*	Pacific giant octopus
Arthropoda	Malacostraca	Decapoda	Hippolytidae	*Lebbeus groenlandicus*	Spiny lebbeid
			Pandalidae	*Pandalus danae*	Coonstripe shrimp
				Pandalus eous	Pink shrimp
				Pandalus hypsinotus	Humpback shrimp
				Pandalus platyceros	**Spot prawn**
			Galatheidae	*Munida quadrispina*	Squat lobster
			Lithodidae	*Acantholithodes hispidus*	Spiny lithode crab
			Paguridae		Hermit crab
			Cancridae	*Metacarcinus magister*	Dungeness crab
				Cancer productus	Red rock crab
			Majidae	*Pugettia producta*	Kelp crab
				Chorilia longipes	Longhorn decorater crab
			Oregoniidae	*Chionoecetes bairdi*	Tanner crab
Echinodermata	Asteroidea	Valvatida	Goniasteridae	*Mediaster aequalis*	Vermilion star
		Forcipulatida	Asteriidae	*Pycnopodia helianthoides*	Sunflower seastar
			Curculionoidea	*Stylasterias forreri*	Fish eating star
	Echinoidea	Echinoia	Strongylocentrotidae	*Strongylocentrotus franciscanus*	Red urchin
	Holothuroidea	Molpadiida	Caudinidae	*Caudina arenicola*	Sweet potato cucumber
Chordata	Actinopterygii	Gadiformes	Gadidae	*Gadus macrocephalus*	Pacific cod
				Theragra chalcogramma	Walleye pollock
		Batrachoidiformes	Batrachoididae	*Porichthys notatus*	Plainfin midshipman
		Scorpaeniformes	Sebastidae	*Sebastes elongatus*	Greenstriped rockfish
				Sebastes emphaeus	Puget Sound rockfish
				Sebastes maliger	Quillback rockfish
				Sebastes miniatus	Vermilion rockfish
			Hexagrammid-ae	*Ophiodon elongatus*	Ling cod
				Hexagrammos decagrammus	Kelp greenling
				Hexagrammos stelleri	Whitespotted greenling
				Zaniolepis latipinnis	Longspine combfish
			Cottidae		

		Hemilepidotus hemilepidotus	Red irish lord
		Hemilepidotus spinosus	Brown irish lord
		Leptocottus armatus	Pacific staghorn sculpin
		Myoxocephalus polyacanthocephalus	Great sculpin
	Agonidae	*Agonopsis vulsa*	Northern spearnose poacher
Perciformes	Zoarcidae	*Lycodes pacificus*	Blackbelly eelpout
	Stichaeidae	*Chirolophis decoratus*	Decorated warbonnet
Pleuronectiformes	Pleuronectidae	*Lyopsetta exilis*	Slender sole

Table D.6. Parameter estimates for GLMMs (fish catch) and LMEs (prawn catch), examining numbers of fish per trap and weight of prawns per trap, respectively, across trap variants.

		Fish catch per trap (number)			
Region	**Trap design**	**Parameter estimate**	**Standard error**	**z-value**	**p-value**
Combined	Unmodified	-2.09	0.25	-8.46	< 0.001
	7.0 cm	-0.26	0.25	-1.03	0.304
	6.4 cm	-0.46	0.35	-1.32	0.187
	4-ring	-0.26	0.30	-0.87	0.386
	5-ring	-1.18	0.42	-2.82	0.005
	7-ring	-1.14	0.42	-2.72	0.007
Gulf Islands	Unmodified	-2.35	0.29	-8.23	< 0.001
	7.0 cm	0.05	0.28	0.18	0.854
	6.4 cm	-0.27	0.41	-0.66	0.509
	4-ring	0.34	0.33	1.01	0.314
	5-ring	-1.07	0.55	-1.96	0.050
	7-ring	-1.29	0.62	-2.08	0.037
Howe Sound	Unmodified	-2.61	0.38	-6.90	< 0.001
	7.0 cm	-2.03	1.03	-1.96	0.049
	6.4 cm	-0.93	0.63	-1.47	0.143
	4-ring	-1.95	0.75	-2.59	0.001
	5-ring	-1.52	0.63	-2.39	0.017
	7-ring	-1.24	0.56	-2.20	0.028
		Prawn catch per trap (weight)			
Region	**Trap design**	**Parameter estimate**	**Standard error**	**t-value**	**p-value**
Combined	Unmodified	260.4	28.4	9.18	< 0.001
	7.0 cm	-92.4	20.1	-4.59	< 0.001
	6.4 cm	-216.1	24.1	-8.95	< 0.001
	4-ring	-225.4	21.1	-10.68	< 0.001
	5-ring	-189.4	21.1	-8.99	< 0.001
	7-ring	-245.3	21.2	-11.57	< 0.001
Gulf Islands	Unmodified	154.1	25.3	6.1	< 0.001
	7.0 cm	-3.5	17.2	-0.2	0.840
	6.4 cm	-33.5	23.9	-1.4	0.161
	4-ring	4.4	22.3	0.2	0.842
	5-Ring	-41.4	21.9	-1.9	0.060
	7-Ring	-80.0	22.4	-3.6	< 0.001
Howe Sound	Unmodified	617.5	33.3	18.56	< 0.001
	7.0 cm	-175.4	37.3	-4.71	< 0.001
	6.4 cm	-390.3	27.3	-10.47	< 0.001
	4-ring	-406.5	31.4	-12.94	< 0.001
	5-ring	-331.7	31.6	-10.50	< 0.001
	7-ring	-394.4	31.5	-12.51	< 0.001

Table D.7: GLMM parameters for the relationships between catch rate versus body size across each trap variant, for fishes (proportion of total numbers) and for prawns (proportion of total weight).

		Fishes			
Trap variant	**Model**	**Parameter**	**Estimate**	**Standard error**	**P-value**
7.0 cm	Constant	β_0	-0.3	0.2	0.175
6.4 cm	Constant	β_0	-1.9	0.5	< 0.001
4-ring	Constant	β_0	-0.9	0.3	0.004
5-ring	Linear	β_0	9.8	2.9	0.002
		β_1	-3.7	0.8	< 0.001
7-ring	Constant	β_0	-1.9	0.6	< 0.001
		Prawns			
Trap variant	**Model**	**Parameter**	**Estimate**	**Standard error**	**P-value**
7.0 cm	Quadratic	β_0	-4.5	2.2	0.037
		β_1	2.3	1.2	0.049
		β_2	-0.3	0.2	0.029
6.4 cm	Constant	β_0	-2.3	0.2	< 0.001
4-ring	Quadratic	β_0	-7.9	2.6	0.002
		β_1	3.7	1.4	0.007
		β_2	-0.5	0.2	0.005
5-ring	Linear	β_0	1.0	0.3	0.003
		β_1	-0.6	0.1	< 0.001
7-ring	Linear	β_0	2.5	0.4	< 0.001
		β_1	-1.1	0.1	< 0.001

Video D.8: *In situ* video comparing prawn entries between unmodified traps and BRD-equipped traps.

Appendix D.9: 'Back-of-the-envelope' calculation of adjusted bycatch

When a BRD results in reductions in target catch (relative to unmodified gear), a potential mitigation strategy for fishers might be to increase effort to make up for lost catch. This additional effort will also catch non-target species which could result in an increase in overall bycatch rates, even if the rate per deployment is lower for gear with BRD than for unmodified gear. We performed a "back-of-the-envelope" calculation to assess this possibility for our modified trap variants. Using the overall mean number of fish and mean weight of prawns caught per trap, we calculated the proportion of fishing effort that would be required using each BRD to maintain prawn catch at the same level as with as the unmodified traps. For example, if a BRD-equipped trap caught 50% fewer prawns than an unmodified one, then a doubling of fishing effort (i.e., two BRD-equipped traps) would be required to maintain prawn catch at the level of the unmodified gear. We then calculated an adjusted fish bycatch for each BRD type by multiplying the fish bycatch rate of that BRD type by the proportion of effort required to maintain catch (given the prawn catch of that BRD type, relative to unmodified traps), divided by the fish bycatch rate of unmodified traps. For example, if the BRD mentioned above caught 25% fewer fish than unmodified gear (but 50% less prawns), then the doubling of effort needed to compensate for lower prawn catch would result in an adjusted bycatch that is 1.5 times that of the unmodified gear.

Appendix E. DVD: Supplementary Files

File A.2. Excel spreadsheet listing all papers which appeared during our literature search, including detailed reasons for exclusion for each paper not included in the final analysis. *File A2 - Spreadsheet of papers included in MA.xlsx*

File C.2. Google Sketchup version 7.1 (http://sketchup.google.com) model of the TrapCam apparatus. *File C2 - Sketchup model of TrapCam apparatus.skp*

Video C.4. Video demonstrating assembly and deployment of TrapCam. *VideoC4.mp4*

Video C.5. Video showing underwater footage collected by TrapCam. This is a 20-second clip sampled 4 hours into a deployment. *VideoC5.mp4*

Video C.6. Video showing a quillback rockfish within the prawn trap. The fish attempts to escape by swimming upward against the trap mesh. This 18-second clip was sampled 6.5 hours into a deployment. *VideoC6.mp4*

Video D.1. Video demonstrating the differences in movement between prawns and rockfish, and our rationale in developing the curved-tunnel BRDs. *VideoD1.mp4*

Video D.8: In situ video comparing prawn entries between unmodified traps and BRD-equipped traps. *VideoD8.mp4*