

Comparison of underwater visual methods for assessing temperate rocky reef fish communities and the effectiveness of spatial marine conservation areas

by

Lily Anne-Marie Burke  
BSc, University of Victoria, 2008

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of the Requirements for the Degree of

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## Abstract

Precise and accurate species abundance and distribution data are important for making effective ecological conservation and management decisions. These data are often challenging to obtain, especially in marine environments where the logistical and technical difficulties of working underwater can limit the precision and accuracy of detection. The chosen survey methodology, along with the study design, will determine the extent to which species' spatial or temporal variability in abundance and distribution may be investigated. Different observational methods may yield different results. I explore how the methodology used to collect sample measurements of fish abundance and diversity in marine environments can influence your understanding of the focal population and the effectiveness of spatial marine conservation measures.

I compare inshore rockfish abundance and fish diversity estimates between paired towed video and baited video surveys and between dive and baited video surveys conducted on temperate rocky-reefs in the nearshore Northeast Pacific on the coast of British Columbia, Canada. I test if the baited video survey data yield equivalent insight to those data derived from the methods commonly used in shallow (dive surveys) and deeper waters (towed video surveys). Paired dive and baited video surveys took place inside and outside of spatial marine conservation areas designated for inshore rockfish called Rockfish Conservation Areas. I test whether the baited video data generate the same conclusions about Rockfish Conservation Area effectiveness as data derived from the dive surveys, and whether the Rockfish Conservation Areas have greater inshore

rockfish abundance and fish diversity than paired locations outside the conservation areas.

I find similar inshore rockfish abundance estimates between towed and baited video, but baited video surveys detect a greater number of unique species than the towed video surveys. The dive surveys detect greater inshore rockfish abundance and fish diversity than the baited video surveys, but the baited video data yield equivalent insight on Rockfish Conservation Area effectiveness to data derived from the dive surveys.

I find little evidence that inshore rockfish recovery is influenced by Rockfish Conservation Area protection. When data were combined across all sites sampled, Rockfish Conservation Areas did not produce more inshore rockfish, bigger rockfish, or greater fish diversity than paired sites outside of Rockfish Conservation Areas, whether measured using a dive survey or a baited video survey. However, I did observe a positive effect of Rockfish Conservation Area protection for some of the individual Rockfish Conservation Areas surveyed that rated as having a high Conservation Score. This suggests certain Rockfish Conservation Areas may be effective conservation areas for inshore rockfish recovery.

The differences I observe in inshore rockfish abundance and fish diversity between the paired surveys reveals the methodology used can influence species abundance and diversity estimates. Baited video surveys are a low cost and effort methodology that can be used to examine inshore rockfish abundance and fish diversity over rocky reefs from nearshore waters down to depths greater than 20 m, and to monitor the effectiveness of spatial marine conservation areas.

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## **Chapter 1 Introduction and Thesis Goals**

### **1.1 Introduction**

I test how the methodology used to collect sample measurements of fishes in marine environments can influence our understanding of the focal population. My research contributes to knowledge on non-destructive visual methods used to assess fish populations and spatial marine conservation measures by comparing visual methods used to survey inshore rockfish (*Sebastes spp.*) over rocky reefs and by assessing the effectiveness of Rockfish Conservation Areas (RCA) in British Columbia (BC), Canada. This knowledge is critical as the methodology used may influence the precision and accuracy of data collected, and these data are important for making effective ecological conservation and management decisions, and for evaluating management actions. Further, this study documents RCA progress since establishment and assesses how RCAs are contributing to rebuilding rockfish populations. My findings have direct implications for management by providing recommendations on visual survey methods that can be used for monitoring marine fish populations of conservation concern. This introductory chapter provides a brief overview of the topics discussed in this thesis and outlines the thesis goals and objectives.

### **1.2 Methods matter**

Ecology is, at its heart, the quest to understand the spatial and temporal dynamics of living organisms - hence surveys of population abundance and distribution, community structure, and their variability in space and time are foundational. Due to time and funding constraints, it is challenging to survey an entire population of interest in the wild,

especially in remote environments. Therefore, we sample the population with repeated measurements and these sample data need to be both accurate and precise to reflect true population values.

The accuracy of sample measurements reflects how close the measurement values are to the true population value or to a reference value while precision is the closeness of the repeated measurements to each other. Systematic error (or bias) consistently over- or underestimates the accuracy of sample measurements. Precision is influenced by the random error, the variability or random variation, observed in the sample measurements. Both systematic and random errors lead to measurement uncertainty. Increasing the number of measurements sampled from the population can improve the precision of sample measurements (and decrease the random error) but systematic errors have a net direction and magnitude and increasing the sample size does not eliminate the effect of bias.

Precise and accurate data are often challenging to obtain in marine environments where the logistical and technical difficulties of working underwater can limit the precision and accuracy of detection (Thompson and Mapstone 1997; Blanchard, Maxwell, and Jennings 2008; Pais et al. 2014). Measurement uncertainty from systematic and random errors may be caused by species-specific factors, such as cryptic coloration (Sale and Douglas 1981; Willis 2001) and secretive behaviour (Kulbicki 1998; MacNeil et al. 2008), as well as extrinsic factors including sea state (Sale and Douglas 1981), site heterogeneity (Edgar and Barrett 1999), observer effects (Edgar, Barrett, and Morton 2004), and survey methodology (Willis, Millar, and Babcock 2000). Detection heterogeneity in study species may result in biased abundance and diversity estimates (Thompson and Mapstone

1997; Ackerman and Bellwood 2000; Ward-Paige, Flemming, and Lotze 2010), erroneous species abundance and distribution inferences (Sale and Sharp 1983; Edgar, Barrett, and Morton 2004), and consequently ill-informed management decisions (Jennings and Polunin 1995; Pelletier et al. 2008; Monk et al. 2012).

Observational methodologies used in marine environments can be divided into two categories: destructive and non-destructive methods. Destructive methods, such as extractive trawl, trap, and hook-and-line surveys sample without replacement, removing organisms from an area. The use of destructive methods is often dictated by the seabed topography of the survey area (Zimmermann et al. 2003), and these methods are rarely used to assess effectiveness of marine spatial closures due to their impact on the species and/or assemblages within the protected area. Additionally, for studies evaluating the effectiveness of marine spatial closures, removal of a protected species through extractive sampling may negatively influence the response to spatial protection.

Non-destructive techniques collect sample data with replacement, avoiding the problem of organism removal on the measured response of marine spatial closure effectiveness while preventing adverse impacts to protected populations or sensitive habitats. Non-destructive (or non-extractive) techniques generally consist of visual observational methods. Visual methods have proven ideal when surveying fishes of low abundance occurring in high-relief rocky areas (For example: Richards 1986; Gratwicke and Speight 2005; Tessier et al. 2005; Anderson and Yoklavich 2007; Laidig, Watters, and Yoklavich 2009; Rooper, Hoff, and De Robertis 2010; Jankowski, Graham, and Jones 2015), provide fish abundance and distribution data in areas closed for fishing (Gardner and Struthers 2013), and therefore, can be used to evaluate responses to spatial marine

conservation measures (Willis, Millar, and Babcock 2000; Langlois et al. 2006; Gardner and Struthers 2013).

In general, there are two classes of non-destructive (or non-extractive) survey methodologies used in marine systems: *active* detection relies on moving across the study area *e.g.* a towed camera or *passive* detection relies on animals moving into a stationary detection frame, *e.g.* a fixed camera trap. In shallow waters (< 20 m), the most common survey method is an active underwater visual survey utilizing a self-contained underwater breathing apparatus (SCUBA), hereafter referred to as a *dive survey (DS)*. Dive surveys typically provide a measure of spatial variability (number of individuals observed within a fixed area or water column volume) as divers move through the study area and record individuals observed along transects or at specific points. Repeated DS at fixed survey locations are also used to monitor temporal variability (*e.g.* see Partnership for Interdisciplinary Studies of Coastal Oceans dive monitoring program, which has been ongoing since 1999, [www.piscoweb.org](http://www.piscoweb.org)).

Dive surveys are subject to inherent bias and random error that may influence species detection, from underestimating cryptic and/or small fish densities (Ackerman and Bellwood 2000; Willis 2001; Bozec et al. 2011) to species-specific behavioural responses to the divers' presence underwater (Kulbicki 1998; Samoilys and Carlos 2000; Watson and Harvey 2007; MacNeil et al. 2008; Bozec et al. 2011; Dickens et al. 2011; Pais and Cabral 2017). Additional factors influencing species detection during DS are observer experience (Thompson and Mapstone 1997; Bernard et al. 2013), diver swim speed (Lincoln Smith 1988), site and sea state heterogeneity (Sale and Douglas 1981; Cheal and Thompson 1997; Edgar and Barrett 1999; MacNeil et al. 2008), and survey area (Sale and

Sharp 1983; Cheal and Thompson 1997). Further, DS are constrained by human physiology that greatly restricts survey effort through maximum depth and survey duration limitations.

Reflecting the technical and logistical limitations of DS, video-based surveys are becoming increasingly popular for enumeration of marine and aquatic biodiversity. Rapid technological evolution has made video systems economical and easy to build and deploy. Myriad video-based techniques such as remotely operated vehicle (ROV), towed video (TV), and stationary drop video and survey designs are now routinely employed to document species and habitat use (see review by Pelletier and Mallet 2014). The capacity to deploy video systems to great depths and for long time periods makes possible expansive temporal and spatial coverage and replication. Remotely operated vehicle and TV surveys are examples of *active* underwater video surveys while baited stationary drop video (BV) is used in *passive* underwater video-based surveys (Mallet and Pelletier 2014).

Trade-offs between underwater survey methodologies that affect precision and accuracy of data collected are mediated by repeatability and financial constraints. The chosen approach, along with the study design, will also determine the extent to which species' spatial or temporal variability may be investigated. The active methodologies, such as ROV, TV and DS, are typically employed to evaluate the variability in habitat and species spatial distribution. However, DS lack the spatial coverage ROV and TV surveys permit and are restricted to shallow waters. Video-based surveys sacrifice the fine-grained spatial heterogeneity data of DS reflecting the camera's narrower field of view and depth of focus in addition to being unable to consistently census cracks,

crevices and lacunae - favoured holding habitats for many species. Remotely operated vehicle and TV surveys address DS limitations as they are not restricted by depth.

However, when compared to a TV survey, ROVs are more expensive and logistically challenging to deploy, limiting sample size, upon which the precision and confidence in statistical estimates hinges.

In comparison to the active methodologies, a single passive BV survey captures temporal variability within a survey volume defined by the technical specifications of the camera rig (baited/unbaited, lights, camera field of view, *etc.*) and local conditions (*e.g.* visibility). Simple BV rigs can be economical to build, tailored for optimal performance in local conditions (Harvey et al. 2003; Watson and Huntington 2016), and are logistically modest to deploy relative to a DS, ROV or TV allowing for the collection of a larger body of data relative to the active surveys. With fewer cost restrictions, BV surveys potentially allow for greater sample sizes and thus, improved precision in statistical estimates.

Given the trade-offs and sources of variability between methodologies - and the importance to effective conservation of understanding the reliability of data that informs decisions - I seek to identify biases inherent to the active and passive survey methodologies. More specifically, I test whether passive BV survey data yield similar fish abundance and community measures to data derived from the methods commonly used in shallow (DS) and deeper (TV) waters. I conducted paired TV-BV and DS-BV surveys to estimate rockfish abundance and fish diversity over temperate rocky-reefs in the nearshore Northeast Pacific Ocean along the coast of British Columbia, Canada. I compared alpha and gamma fish diversity estimates observed at sampling sites between

paired surveys. Sampling sites are the minimum area in terms of space and time visually surveyed by each method. Alpha and gamma diversity both record the number of species observed at a site but are differentiated by scale. In the paired method comparisons, alpha diversity is the number of species observed at a site with each method while gamma diversity is the total number of species recorded across all sites for each method.

### **1.3 Monitoring spatial marine conservation measures**

Spatial marine conservation measures, such as marine protected areas (MPAs), fisheries closures, and harvest refugia are closed areas where there is partial or total protection from fishing and other extractive practices (Blyth-Skyrme et al. 2006; Dudley 2008). These conservation measures are strategies used to conserve and restore depleted populations of marine fishes and to support the sustainability of fisheries (Lauck et al. 1998; Hilborn et al. 2004; Roberts, Hawkins, and Gell 2005; Spalding, Fish, and Wood 2008). Spatial marine conservation measures are management tools used to increase the size, abundance and diversity of species protected within and sometimes adjacent to them (Mosquera et al. 2000; Halpern and Warner 2002; Halpern 2003; Alcala et al. 2005; Lester et al. 2009; Babcock et al. 2010). However, effectiveness of marine conservation areas may vary widely (Allison, Lubchenco, and Carr 1998; Hilborn, Micheli, and De Leo 2006; Mora et al. 2006; Claudet et al. 2008; Edgar 2011).

A potentially significant source of variance affecting the perceived efficacy of spatial marine conservation measures is measurement uncertainty from random and systematic error, as it can be difficult to precisely and accurately estimate species diversity and abundance with underwater surveys. Different observational methods may yield different results, making performance estimates of spatial marine conservation measures

equivocal. With the proliferation of underwater video as a survey and research tool (Mallet and Pelletier 2014), underwater survey methodologies have expanded beyond conventional diver-based assessment of fish abundance and size, making it even more important to understand methodology-driven systematic and random error.

I compared inshore rockfish abundance and fish diversity estimates between paired DS and BV surveys inside and outside spatial marine conservation areas designated for inshore rockfish conservation in the Northeast Pacific Ocean on the coast of British Columbia, Canada to address both a methodological and conservation question. Given the trade-offs between DS and BV methodologies, I ask whether the BV data generate the same conclusions about conservation area effectiveness as data derived from DS, and whether the marine conservation areas have greater inshore rockfish abundance than paired locations outside the conservation areas. Equivalence between DS and BV data would suggest that BV surveys could be used in place of DS for spatial marine conservation area monitoring, yielding more data for less investment.

#### **1.4 Rockfish and Rockfish Conservation Areas**

Inshore rockfish of the genus *Sebastes* include black rockfish (*S. melanops*), China rockfish (*S. nebulosus*), copper rockfish (*S. caurinus*), quillback rockfish (*S. maliger*), tiger rockfish (*S. nigrocinctus*), brown rockfish (*S. auriculatus*), blue rockfish (*S. mystinus*), and yelloweye rockfish (*S. ruberrimus*) (Yamanaka and Logan 2010; Love, Yoklavich, and Thorsteinson 2002). These species are typically found proximate to high rugosity rocky substrates such as boulder fields down to 200 m of depth (Love et al. 1990; Love, Yoklavich, and Thorsteinson 2002; Haggarty, Shurin, and Yamanaka 2016). Inshore rockfish are philopatric (tending to remain near a particular rocky site or area),

grow to large sizes, reach sexual maturity at a late age and experience variable recruitment that is dependent upon environmental conditions, making them particularly vulnerable to overfishing (Parker et al. 2000; Love, Yoklavich, and Thorsteinson 2002).

Rockfish possess closed swimbladders (physoclistous), trapping expanding gasses causing internal injury (barotrauma) when rapidly pulled to the surface in nets or on lines, making catch and release ineffective and placing a conservation premium on non-capture survey methods. High site fidelity and small home range sizes make rockfish particularly susceptible to localized overfishing (Love et al. 1990; Parker et al. 1995; Love, Yoklavich, and Thorsteinson 2002), but also make them suitable for recovery using spatial protection strategies (Carr and Reed 1993; Yoklavich 1998; Parker et al. 2000; Hamilton et al. 2010).

Rockfish Conservation Areas (RCAs) are spatially explicit permanent fisheries closures enacted by the federal Fisheries and Oceans Canada (DFO) in response to sharp declines in inshore rockfish catches during the 1990s (Yamanaka and Logan 2010; Haggarty 2013). Rockfish Conservation Areas prohibit commercial and recreational fishing gear that targets rockfish or leads to rockfish by-catch (Yamanaka and Logan 2010). Rockfish Conservation Areas are not no-take conservation areas as certain recreational and commercial fishing activities (such as prawn traps and off-bottom net gear) are permitted. Reduced fishing in RCAs should theoretically enhance the likelihood of an individual rockfish reaching sexual maturity and lead to increased abundance and biomass of rockfish inside of RCAs (Love et al. 1990; Yamanaka and Logan 2010; Hilborn et al. 2004).

Evaluations of spatial marine conservation areas such as RCAs or Marine Protected Areas for rockfish have yielded inconclusive results to date. Larger rockfish and greater rockfish biomass have been found inside Californian reserves relative to outside of reserves (Hamilton et al. 2010; Keller et al. 2014). In older reserves, larger rockfish and greater rockfish biomass were observed, while a 1 year old reserve showed no difference from outside areas (Paddack and Estes 2000). In British Columbia, there is little indication rockfish recovery is influenced by RCA protection (Marliave and Challenger 2009; Cloutier 2011; Haggarty, Shurin, and Yamanaka 2016). No significant increase in rockfish density has been observed as a result of RCA protection (Marliave and Challenger 2009; Cloutier 2011; Haggarty, Shurin, and Yamanaka 2016) nor have differences in the size structure of rockfishes inside RCAs and unprotected areas been found (Haggarty, Shurin, and Yamanaka 2016).

### **1.5 Study area**

I compared inshore rockfish abundance and fish diversity estimates between paired surveys in the Northeast Pacific Ocean on the central and south coastal regions of BC, Canada. The south coast of BC is characterized by developed, urban areas and inshore rockfish populations were historically subject to heavy fishing pressure reducing observed inshore rockfish abundance and fish diversity. In contrast, the central coast of BC is far less developed and has experienced less fishing pressure resulting greater observed abundance and diversity of fishes.

## **1.6 Study goals**

This thesis had two goals: 1) contribute to knowledge on the methods used to assess inshore rockfish populations, and 2) assess the ecological effectiveness of RCAs with multiple methods. This study had the following objectives:

- 1) Compare inshore rockfish abundance and fish diversity estimates generated by active and passive survey methods.
- 2) Assess if the survey method used for RCA monitoring influenced measured efficacy.
- 3) Test for evidence of RCA effectiveness as measured by increased inshore rockfish abundance and/or body size and increased fish diversity inside RCAs when compared to outside of RCAs.

## **1.7 Thesis structure**

In this thesis I explore how the methodology used to collect sample measurements of fishes in marine environments can influence your understanding of the focal population and the effectiveness of spatial marine conservation measures. Chapter 2 compared inshore rockfish abundance and fish diversity estimates between paired active (TV and DS) and passive (BV) surveys conducted over temperate rocky-reefs in the nearshore Northeast Pacific in coastal BC, Canada. I was specifically interested in testing if the BV survey data yielded equivalent insight to those data derived from the methods commonly used in shallow (DS) and deeper waters (TV). Chapter 3 evaluated results from paired DS and BV surveys inside and outside of spatial marine conservation areas designated for inshore rockfish conservation in the Northeast Pacific Ocean on the south coast of BC to determine whether the BV data generate the same conclusions about conservation area

effectiveness as data derived from DS. Further, this chapter assessed the effectiveness of RCAs at rebuilding inshore rockfish populations - whether greater inshore rockfish abundance and fish diversity were observed inside the protected areas when compared to unprotected areas outside of RCAs. Chapter 4 synthesizes the key results of my research, discusses limitations, and concludes the thesis.

## **Chapter 2 Fish on Film: an underwater method comparison between towed and baited video and baited video and dive surveys**

### **2.1 Introduction**

Precise and accurate species abundance and distribution data are important for making effective ecological conservation and management decisions. Such data are necessary to understand how living organisms are affected by spatial and temporal environmental changes (Cheung et al. 2009). These data are often challenging to obtain in the wild, especially in marine environments where the logistical and technical difficulties of working underwater can both limit the precision and accuracy of detection (Thompson and Mapstone 1997; Blanchard, Maxwell, and Jennings 2008; Pais et al. 2014).

Bias and reduced precision may be caused by species-specific factors, such as cryptic coloration (Sale and Douglas 1981; Willis 2001) and secretive behaviour (Kulbicki 1998; MacNeil et al. 2008), extrinsic factors including sea state (Sale and Douglas 1981), site heterogeneity (Edgar and Barrett 1999), observer effects (Edgar, Barrett, and Morton 2004), and survey methodology (Willis, Millar, and Babcock 2000). Detection heterogeneity in study species may result in biased abundance and diversity estimates (Thompson and Mapstone 1997; Ackerman and Bellwood 2000; Ward-Paige, Flemming, and Lotze 2010), erroneous species abundance and distribution inferences (Sale and Sharp 1983; Edgar, Barrett, and Morton 2004), and consequently ill-informed management decisions (Jennings and Polunin 1995; Pelletier et al. 2008; Monk et al. 2012).

In general, there are two classes of non-extractive (or non-destructive) survey methodologies used in marine systems: *active* detection relies on moving across the study area *e.g.* a towed camera or *passive* detection relies on animals moving into a stationary detection frame, *e.g.* a fixed camera trap. In shallow waters (< 20 m), the most common survey method is an active underwater visual survey utilizing a self-contained underwater breathing apparatus (SCUBA), hereafter referred to as a *dive survey* (DS). Dive surveys typically provide a measure of spatial variability (number of individuals observed within a fixed area or water column volume) as divers move through the study area and record individuals observed along transects or at specific points. Repeated DS at fixed survey locations are also used to monitor temporal variability (*e.g.* see Partnership for Interdisciplinary Studies of Coastal Oceans dive monitoring program, which has been ongoing since 1999, [www.piscoweb.org](http://www.piscoweb.org)).

Dive surveys have been employed for more than six decades (Brock 1954) and remain the standard for inshore marine surveys. However, DS are subject to inherent sampling error and biases that may influence species detection, from underestimating cryptic and/or small fish densities (Ackerman and Bellwood 2000; Willis 2001; Bozec et al. 2011) to species-specific behavioural responses to the divers' presence underwater (Kulbicki 1998; Samoilys and Carlos 2000; Watson and Harvey 2007; MacNeil et al. 2008; Bozec et al. 2011; Dickens et al. 2011; Pais and Cabral 2017). Additional factors influencing species detection during DS are observer experience (Thompson and Mapstone 1997; Bernard et al. 2013), diver swim speed (Lincoln Smith 1988), site and sea state heterogeneity (Sale and Douglas 1981; Cheal and Thompson 1997; Edgar and Barrett 1999; MacNeil et al. 2008), and survey area (Sale and Sharp 1983; Cheal and

Thompson 1997). Further, DS are constrained by human physiology that greatly restricts survey effort through maximum depth and survey duration limitations.

Responding to technical and logistical limitations of DS, video-based techniques are becoming increasingly popular for enumeration of species abundance, distribution, and biodiversity underwater, as technological progress has made video systems more economical and easier to deploy (see review by Mallet and Pelletier 2014). Video systems can be deployed rapidly, to great depths and for long time periods making possible expansive temporal and spatial coverage and replication. Remotely operated vehicle (ROV) and towed video (TV) surveys are examples of active underwater video surveys while baited stationary drop video (BV) is used in passive underwater video-based surveys (Mallet and Pelletier 2014).

Trade-offs between underwater survey methodologies that affect precision and accuracy of data collected are mediated by repeatability and financial constraints. The chosen approach, along with the study design, will also determine the extent to which species' spatial or temporal variability may be investigated. The active methodologies, such as ROV, TV and DS, are typically employed to evaluate the variability in habitat and species spatial distribution. However, DS lack the spatial coverage that the ROV and TV surveys permit and are restricted to shallow waters. Video-based surveys sacrifice the fine-grained spatial heterogeneity data of DS reflecting the camera's narrower field of view and depth of focus in addition to being unable to consistently census cracks, crevices and lacunae - favoured holding habitats for many species. Remotely operated vehicle and TV surveys address DS limitations as they are not as restricted by depth. However, when compared to a TV survey, ROVs are more expensive and logistically

challenging to deploy, limiting sample size, upon which the precision and confidence in statistical estimates hinges.

In comparison to active methodologies, a single BV survey captures temporal variability within a survey volume defined by the technical specifications of the camera rig (baited/unbaited, lights, camera field of view, *etc.*) and local conditions (*e.g.* visibility). Simple passive BV rigs can be very economical to build, tailored for optimal performance in local conditions (Harvey et al. 2003; Watson and Huntington 2016), and are logistically modest to deploy relative to a DS, ROV or TV allowing for the collection of a larger body of data relative to the active surveys. With fewer cost restrictions, BV surveys potentially allow for greater sample sizes and thus, improved precision in statistical estimates.

Given the trade-offs and sources of variability between methodologies - and the importance to effective conservation of understanding the reliability of data that informs decisions - I seek to identify biases inherent to the active and passive survey methodologies. More specifically, I test whether passive BV survey data yield similar fish abundance and community measures to data derived from active TV and DS survey methods. I conducted concurrent multi-method surveys to estimate inshore rockfish abundance and fish diversity over temperate rocky-reefs in the nearshore Northeast Pacific Ocean along the coast of British Columbia, Canada.

Inshore rockfish of the genus *Sebastes* were the focal species of the surveys. These include the black rockfish (*S. melanops*), China rockfish (*S. nebulosus*), copper rockfish (*S. caurinus*), quillback rockfish (*S. maliger*), tiger rockfish (*S. nigrocinctus*), brown rockfish (*S. auriculatus*), blue rockfish (*S. mystinus*) and yelloweye rockfish (*S.*

*ruberrimus*) (Yamanaka and Logan 2010, Love et al. 2002). Inshore rockfish are typically found proximate to structurally complex rocky substrates in nearshore waters within 200 m depth (Love et al. 1990; Love, Yoklavich, and Thorsteinson 2002). In addition to inshore rockfish, many fishes are found over temperate rocky-reefs such as other demersal fishes (e.g. *Ophiodon elongatus* and *Hexagrammos spp.*) and cryptic, benthic species (e.g. *Scorpaenichthys marmoratus*). The relief and complexity of the rocky-reefs provide abundant spatial heterogeneity and are ideal locations to compare variability between active and passive surveys.

A growing number of studies using non-extractive survey methods have been conducted in recent years to evaluate rockfish abundance and distribution along with assessing the efficacy of rockfish spatial recovery strategies (Marliave and Challenger 2009; Rooper, Hoff, and De Robertis 2010; Cloutier 2011; Karpov, Bergen, and Geibel 2012; Lotterhos, Dick, and Haggarty 2014; Starr et al. 2015; Haggarty, Shurin, and Yamanaka 2016). However, no studies have compared the strengths and limitations between non-extractive passive and active methods utilizing video for surveying rockfish.

I compared inshore rockfish abundance estimates generated by active and passive survey methods. I hypothesized if passive underwater video data are commensurate with active DS and TV data there would be a positive relationship between inshore rockfish abundance estimates across methods. Alternatively, because of constraints to the number of fish that can be viewed in the BV field of view, I hypothesized BV surveys will reach a maximum point of inshore rockfish abundance while DS and TV abundance values will continue to increase.

I also compared fish diversity estimates between paired active and passive surveys at a site-level and across all sites sampled. I hypothesized if passive underwater video data are commensurate with active DS and TV data there would be a positive relationship between species richness observed at the site-level between paired methods. In the comparison of diversity across all sites, I hypothesized BV surveys would better detect predatory and scavenging species than the TV and DS as the bait would attract these species into the BV field of view. I hypothesized the DS would be better than the BV for capturing cryptic species because divers can search complex habitats and census cracks, crevices and lacunae in ways that cameras cannot.

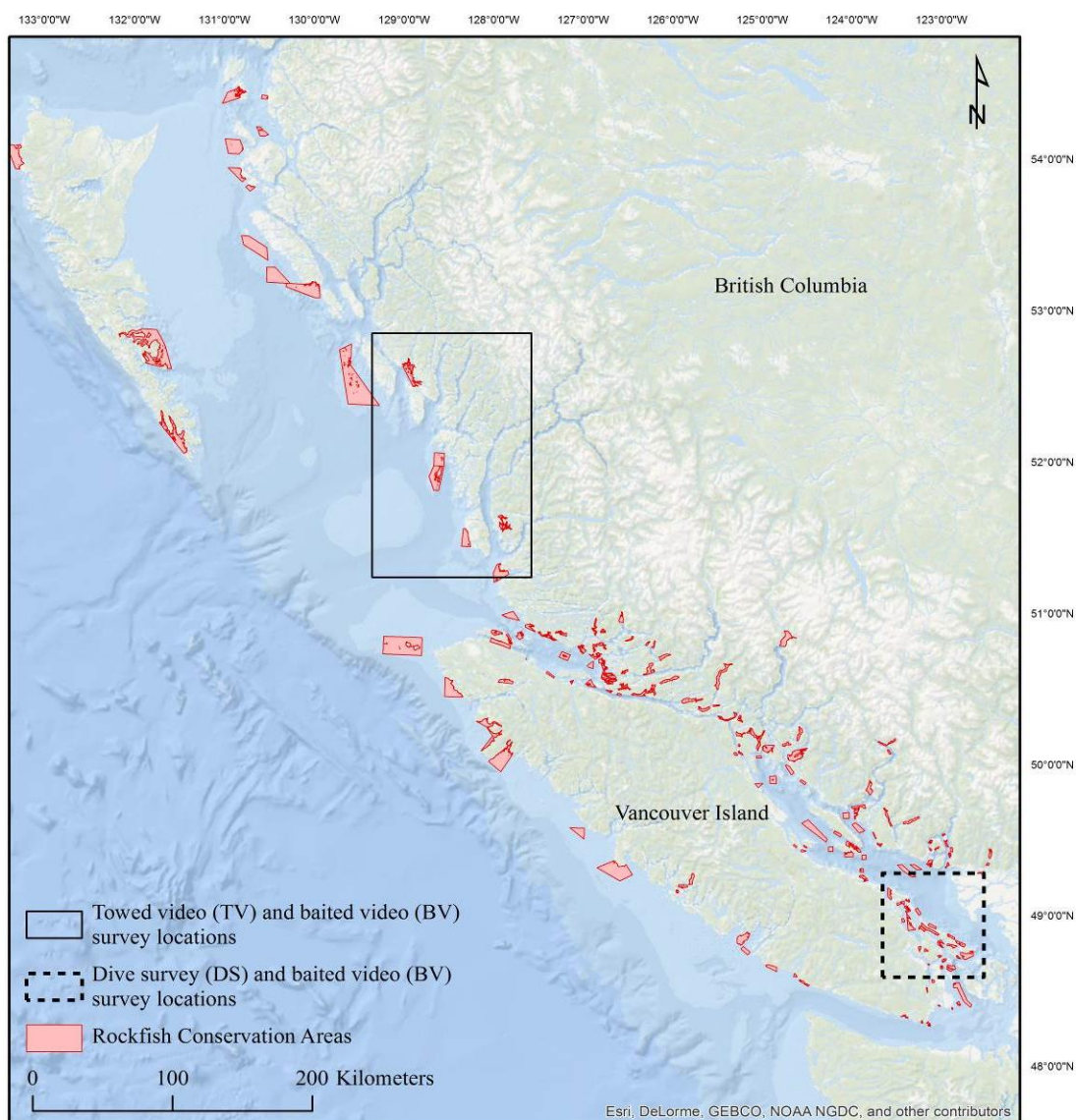
Further, I evaluated the influence of spatial heterogeneity on the method specific data to determine: i) the abiotic variables that best explain inshore rockfish abundance and fish diversity; and ii) are the variables that best explain inshore rockfish abundance and fish diversity similar between active and passive survey methods. As the two active methodologies, DS and TV, collect high-resolution data on species location and habitat, I hypothesize estimates derived from DS and TV methods to be more influenced by spatial heterogeneity than BV surveys.

Previous research has found a BV survey to be more efficient in terms of statistical power, survey personnel needed, and boat resources than DS (Watson et al. 2005), but this result is not consistent across studies (Colton and Swearer 2010). I provide a general assessment of the effort required to conduct a single sample of each method and include a summary of factors that influence method performance such as environmental conditions and species characteristics.

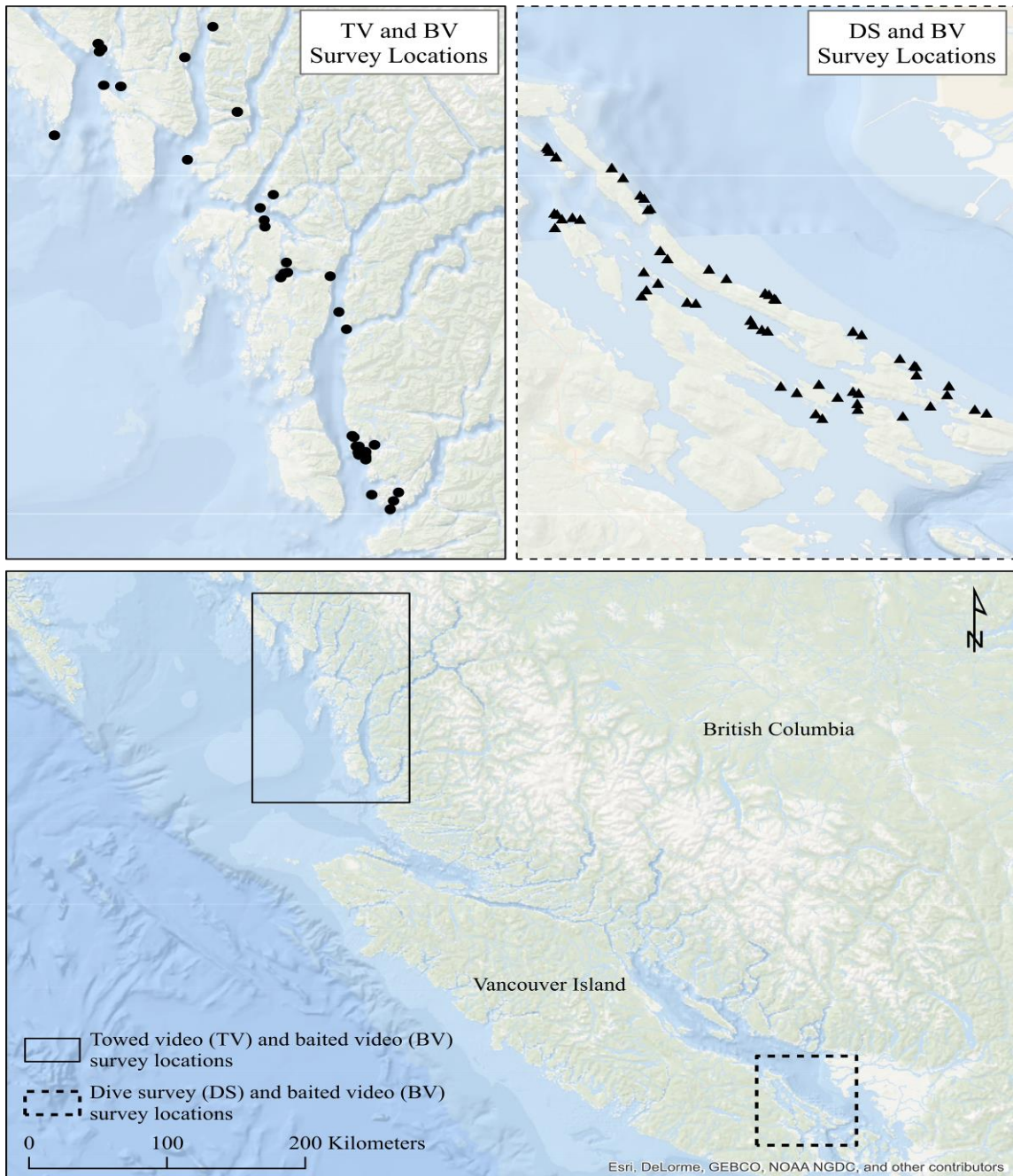
## 2.2 Methods

### 2.2.1 Study system and species

I conducted 90 surveys on nearshore rocky reefs in coastal British Columbia (BC), Canada. Paired TV and BV surveys were conducted at 34 locations in August 2015, while paired DS and BV surveys were conducted at 56 locations from September–November 2015 (Figures 2-1 and 2-2).



**Figure 2-1 Study locations of paired towed and baited video surveys and paired dive and baited video surveys on the coast of British Columbia, Canada.**



**Figure 2-2 Location of sampling sites on the coast of British Columbia, Canada.**

Rockfish are philopatric (tending to remain near a particular rocky site or area), long-lived (up to 118 years), grow to large sizes and reach sexual maturity at a late age, all of which makes these species particularly vulnerable to overfishing (Love et al. 1990, Parker et al. 1995; Love et al. 2002). Rockfishes can be extremely fecund and sporadic larval recruitment has been observed, with success dependent upon favourable oceanographic conditions (Love, Yoklavich, and Thorsteinson 2002). Rockfish possess closed (physoclistous) swimbladders that trap expanding gasses causing internal injury (barotrauma) when rapidly pulled to the surface in nets or on lines, making catch and release ineffective and placing a conservation premium on non-capture survey methods.

Rockfish are important in Canadian fisheries and culturally significant to indigenous groups of British Columbia's coast. Inshore rockfish are targeted in commercial, recreational, and Indigenous fisheries (Yamanaka and Logan 2010; Zetterberg, Watson, and O'Brien 2012; Eckert et al. 2018). After the creation of a rockfish commercial fishery in the 1970s many rockfish populations experienced sharp declines and numerous populations of *Sebastes spp.* have been since been overfished (Parker et al. 2000, Love et al., 2002, Yamanaka and Logan 2010). Of the inshore species, the quillback rockfish is listed as *Threatened* by the Committee on the Status of Endangered Wildlife in Canada and the yelloweye rockfish is listed as a *Species of Special Concern* by the Species at Risk Act. In response to conservation concerns, Fisheries and Oceans Canada (DFO) implemented management measures in the early 2000s that restricted rockfish catch. As part of these management measures, DFO established spatial closures called Rockfish Conservation Areas (RCAs) along the BC coast to promote rockfish recovery.

To compare the TV and BV surveys, I stratified candidate sampling sites based on expected densities and diversity of inshore rockfish from observational data collected by the Central Coast Indigenous Resource Alliance (CCIRA) on the Central Coast of BC (Frid et al. 2016). Sites were then selected *ad hoc* from the high diversity and density stratum. Sampling sites were therefore intended to be representative of this stratum only. I collected inshore rockfish abundance and fish diversity data at these sites via TV and BV.

For the DS and BV comparison, I included sites inside and outside RCAs in order to evaluate what effect RCA status had on inshore rockfish abundance and fish diversity (Chapter 3). I used bathymetry derivatives in ArcGIS (ArcMap 10.2) to match geophysical profiles inside and outside of RCAs. I combined seafloor features (*i.e.* rocky substrate, slope, and curvature) generated on a 20 m grid to identify suitable rockfish habitat (bathymetry derivatives provided by D. Haggarty, DFO). To enable dive surveys all sites were within 20 m depth. I used the NOAA Sampling Design Tool to randomly select survey sites targeting rocky substrate with similar slope and curvature. All paired “inside - outside” sites were more than 300 m apart.

### **2.2.2 Active field methods**

#### **Towed video survey**

TV systems are linked to a vessel by a coaxial cable and towed at low speed along transects of predefined trajectory and/or size. TV surveys may sample large areas in short time periods, increasing the potential spatial coverage of surveys. TV systems may be deployed on the seabed or in the water column and have been utilized from the inshore to depths > 2000 m (Shortis et al. 2007; Jones et al. 2009). TV surveys, in addition to

benthic habitat mapping and monitoring, are often used to census demersal fishes (Assis, Narváez, and Haroun 2008; Carbines and Cole 2009). In off-bottom TV surveys, like I used, the video system maintains a position above the seabed. Optimally, the substrate-camera distance is held constant, rendering a consistent field of view and thus area surveyed per unit time. Off-bottom TV surveys conducted over rocky substrate can be challenging given the vertical camera position is typically in constant motion as operators respond in real time to the vertically variable habitat of rocky reefs, the preferred inshore rockfish habitat. Limitations of TV surveys are associated with sea state heterogeneity (Sheehan, Stevens, and Attrill 2010), limited probability to detect cryptic species (Assis, Narváez, and Haroun 2008) and the influence of the TV system on individual behaviors, which remains unknown.

To conduct the TV surveys, I followed the survey protocol developed by CCIRA and utilized a Deep Blue Pro video camera (Ocean Systems, Inc., Everett, WA, USA) towed by 10 m research vessel at 1 knot. The active TV system consisted of the camera, an Aqualight Pro dive light and two forward-pointing lasers fixed 10 cm apart to provide a known scale in the video images. To steady and orient the camera in a forward and downward facing position, a 6.8 Kg lead ball was attached above the adjustable strain relief on the umbilical cable and a drift fin was secured below the laser base. The camera was connected to a topside 18 cm LCD video monitor and video recorder via 210 m of umbilical live-feed coaxial cable.

A Lowrance Elite-4 CHIRP with an Airmar P66 transducer collected location and depth data of the TV during deployment. The location and depth, along with date and time, were overlaid on the video image. Date and time were collected every two seconds

and serial port monitor software (rs232 data logger, [www.eltima.com](http://www.eltima.com)) logged and exported these data to an onboard computer as a text file. I used a second onboard computer with a video capture device (Pinnacle Dazzle, [www.pinnacle.com](http://www.pinnacle.com)) to capture and save the video image. The TV system was manually maintained ~1-2 m off-bottom by using the live video feed to manipulate the umbilical accordingly. Video transects began near shore at ~10 m depth and extended perpendicularly across depth contours to a maximum depth of 200 m. Linear distance of transects ranged between 80 to 1682 m.

A single transcriber processed all TV video in QuickTime (version 10.4; QuickTime Player) following a standardized video annotation protocol developed by CCIRA. Each transect was divided into segments of similar depth profiles and homogenous seabed terrain. A new segment commenced with each significant change in depth and/or seabed terrain. Seabed terrain characteristics recorded during video transcription include rocky substrate, high relief and high complexity and proportions of these characteristics were used in TV analyses. I calculated the proportions of seabed terrain characteristics by adding the number of occurrences a characteristic was observed and dividing by the total number of observations per site. Relief accounted for vertical changes in the seabed while complexity recorded the amount of surface irregularity and the number of crevices observed in the seabed. Together, relief and complexity were used as a measure of topographic heterogeneity, combining both structural relief and roughness. The median depth of a TV transect per site was also used in the TV analyses.

All fish observed were enumerated and identified to the lowest taxonomic level possible. Data recorded for each fish observation include: time, location, depth, and seabed terrain characteristics. Screenshots in each habitat segment were taken every 30-

seconds. In ImageJ (version 1.50d; Rasband 1997-2016), the screenshots were used to determine an average width per segment using the distance between the scaling lasers. Segment width was multiplied by segment length to calculate a survey area per segment and the individual segments per site were added to provide a total survey area per site. Inshore rockfish density was calculated by dividing the total number of inshore rockfish observed by the total survey area per site ( $\# \text{ inshore rockfish} / \text{TV survey area m}^2$ ). For the diversity metrics, the number of different species observed at each site in the TV was recorded.

### **Dive survey**

At each site, a DS was completed wherein two SCUBA divers descended at predetermined coordinates and surveyed (using a compass) along an isobath running parallel to the adjacent shore. Survey depths were within 20 m and average transect depth ranged from 8.5 to 17.8 m. The first diver attached a 30 m transect line to their weight belt and the line was drawn out behind the divers as they proceeded along the isobath. Once the first 30 m transect was completed the divers swam back to the transect start while reeling in the transect line. The divers then swam perpendicular to transect direction 5 m shallower and completed a second transect following the opposite compass bearing to the first transect. Search effort was standardized by maintaining a swim speed of  $4 \text{ m min}^{-1}$ .

The first diver recorded all observed fish  $> 10 \text{ cm}$  in length by species within 1.5 m on each side of the transect line. Fish length was estimated visually and recorded for each fish in 10 cm size bins. The second diver followed the first and recorded habitat data as per Pacunski and Palsson (2001) and Haggarty, Lotterhos, and Shurin (2017) scoring

relief, complexity, substrate type, and depth—recording data every 2 m. Relief accounted for vertical changes in the seabed while complexity recorded the amount of surface irregularity and the number of crevices observed in the seabed. Together, complexity and relief were used as a measure of topographic heterogeneity observed along the dive transects, combining both structural relief and roughness.

Total inshore rockfish density was calculated by dividing the number of inshore rockfish observed per site by survey volume (# inshore rockfish / 540 m<sup>3</sup>). Proportions of habitat parameters (rocky substrate, high relief and high complexity) were calculated by adding the number of occurrences of a habitat parameter observed every 2 m along the two transects and dividing by the total number of habitat observations per site (30). Mean transect depth was calculated by taking the average of depth observed every 2 m along the two transects.

### **2.2.3 Passive field methods**

#### **Baited video survey**

Passive underwater video surveys depend on motile organisms moving into the field of view of the camera to be enumerated. Unlike active systems, which use mobile detection instruments, passive systems use spatially fixed detection instruments. Passive systems may be either autonomous or tethered to the vessel facilitating real-time monitoring and data collection. Passive underwater video surveys have successfully assayed spatial-temporal variations in fish assemblages (Harvey et al. 2007; Stobart et al. 2007; Colton and Swearer 2010; Langlois et al. 2010; Birt, Harvey, and Langlois 2012; Santana-Garcon, Newman, and Harvey 2014), examined the influence of habitat characteristic on species' distributions (Watson et al. 2005; Harvey et al. 2007; Stobart et al. 2007;

Dorman, Harvey, and Newman 2012; Ebner and Morgan 2013), and tested the efficacy of marine protected areas (MPAs) and fishery closures on abundance and biodiversity (Cappo, Speare, and De'Ath 2004; Willis, Millar, and Babcock 2000; Willis 2001; Dorman, Harvey, and Newman 2012; Lowry et al. 2012; Gardner and Struthers 2013). Visibility has been found to affect sampling efficacy of a passive video system (Cappo et al. 2003; Ebner and Morgan 2013) and passive systems may be less successful in detecting cryptic, reef-associated fishes (Willis 2001; Watson et al. 2005; Stobart et al. 2007; Colton and Swearer 2010; Lowry et al. 2012).

Bait may increase the probability of organisms entering the camera field of view of a passive video system and has proven useful in areas with low fish densities and to sample targeted fish species that may be wary of divers' presence (Willis, Millar, and Babcock 2000a; Watson et al. 2005; Watson and Harvey 2007; Harvey et al. 2007; Langlois et al. 2010; Watson et al. 2010; Dorman, Harvey, and Newman 2012). Baited video surveys use an attractant placed close to the camera to lure fish into the field of view with 'soak times' in past studies ranging from 8 up to 120 minutes (Mallet and Pelletier 2014) - I used 30 minutes (soak times recommended by Stobart et al. 2007; Gladstone et al. 2012).

Baited systems may introduce biases associated with differential species or even individual-specific responses to bait (Hardinge et al. 2013). Further, limitations on the number of fishes that may appear in the camera field of view at any one time complicates enumeration accuracy (Willis, Millar, and Babcock 2000; Cappo et al. 2003; Harvey et al. 2007; Dorman, Harvey, and Newman 2012; Hardinge et al. 2013). An additional challenge of BV surveys is the inability to calculate the realized sampling volume because the dispersion of the bait attractant plume is unknown, as are individual and

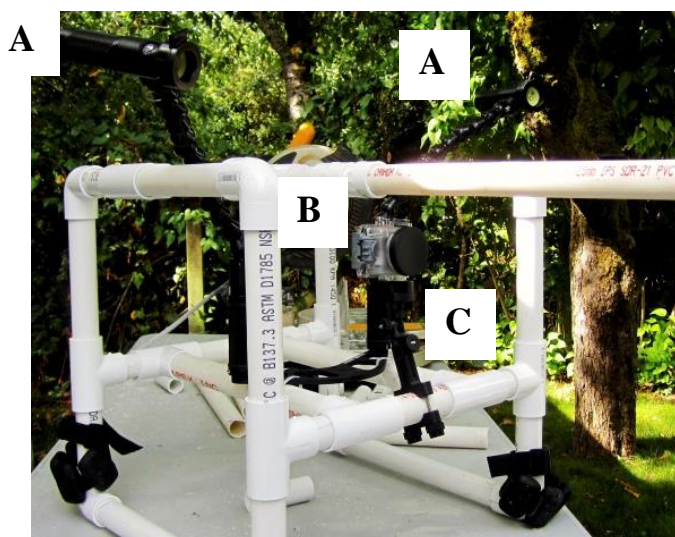
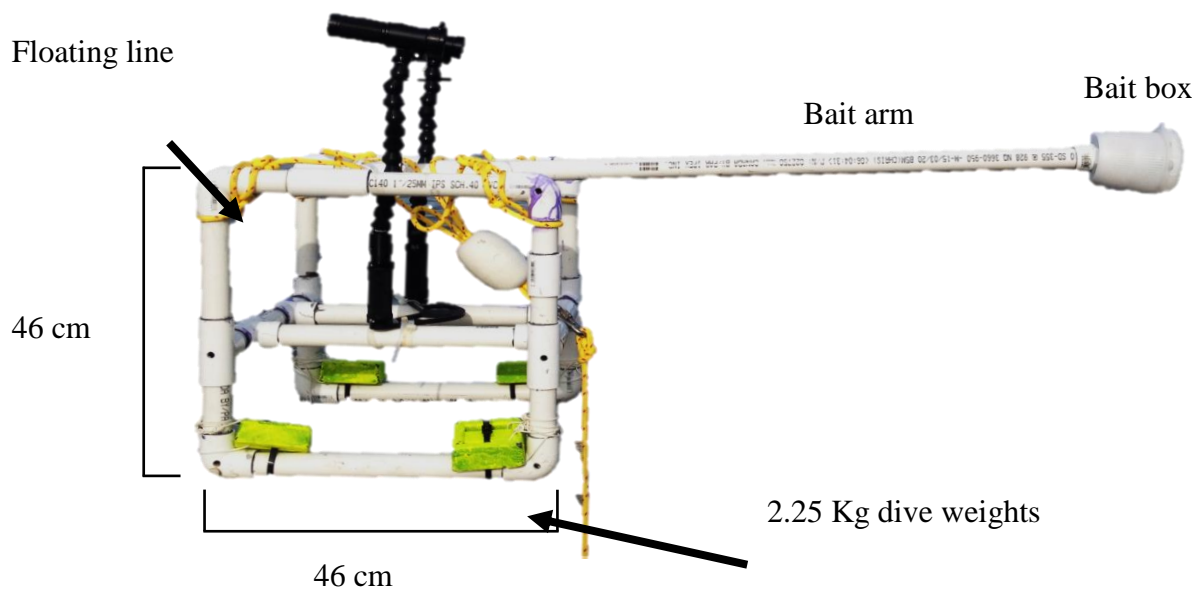
species-specific responses to it (Willis, Millar, and Babcock 2000; Willis and Babcock 2000; Heagney et al. 2007). This last point precludes measures of absolute density and therefore, BV surveys are limited to relative abundance measures.

I deployed a BV system i) at a randomly selected point on each of the TV transects and ii) at the starting point of the first dive transect at each site. The TV and dive surveys were always carried out prior to the BV sampling to eliminate potential bias associated with the effect of bait. I deployed the BV at least 30 minutes after the conclusion of the active surveys to minimize possible bias related to the towed camera and diver effect on fish behaviour. One 30-minute BV survey was conducted following each TV and dive survey.

The BV system consisted of a waterproof Intova camera (Intova Sport HD; intova.com) mounted on a polyvinyl chloride (1 1/2" / 25 mm PVC) frame with 2.25 Kg dive weights attached to each of the four legs. The camera was pointed forward towards the bait arm with dual lights attached to the frame also pointing forward. A 1 m bait arm (PVC) extended from the frame. At its distal end I secured a perforated plastic bait container typical of those used in prawn and crab traps. At rest the camera sits ~ 50 cm off the bottom. Fresh bait was used for each deployment and consisted of approximately 150 grams of 3 minced Pacific Herring (*Clupea harengus*) mixed with 1 cup of commercial prawn and crab bait pellets (FirstMate Prawn & Crab Bait).

The BV was tethered topside via a polypropylene rope attached to a 32 cm diameter commercial surface buoy (Dan-fender B40 balloon fender). Following BV deployment the support vessel remained >100 m away from the deployment area, engines off. Camera resolution was set to 1080p@30fps and the photo quality to Super Fine while all other

camera modes were left in the factory default Auto setting. Sample sites were located more than 100 m apart to minimize potential overlap of bait odour plumes ensuring each site was independent of others (Willis and Babcock 2000; Cappo, Speare, and De'Ath 2004; Harvey et al. 2007; Heagney et al. 2007).



**A.** Big Blue Video Lights

**B.** Intova Sport HD camera

**C.** Intova Bar-Pole Mount & BTS  
– quick release adapter

**Figure 2-3 Baited video platform built using 1"/25 mm diameter PVC and Oatey Purple Primer and Regular PVC Cement.**

I used the average number of individuals of the focal species (inshore rockfish *Sebastes spp.*) observed in the video sample (MeanCount) to quantify inshore rockfish abundance (Conn 2011). MeanCount was estimated by randomly sampling 60 frames from each 30-minute video (Schobernd, Bacheler, and Conn 2014; Bacheler and Shertzer 2015). All inshore rockfish observed in each frame were identified to the lowest taxonomic level possible. Inshore rockfish MeanCount was calculated by adding the total number inshore rockfish observed in the 60 randomly selected frames and dividing by 60.

To quantify BV fish diversity I used the video metric MaxN. For every species identified in the 30-minute sample, the maximum number of individuals appearing in-frame simultaneously (*i.e.* the “snapshot” where the maximum number conspecifics appearing simultaneously appear) is recorded as that species’ MaxN. MaxN records every species observed in the 30-minute sample and avoids the recounting of individuals that move in and out of the static field of view (Willis, Millar, and Babcock 2000; Harvey et al. 2007).

For the BV surveys, I noted deployment depth of the camera and coarse categorical measures of seabed terrain characteristics observed in the camera field of view. Rocky substrate, complexity and seafloor relief were considered High if rocky substrate comprised >50% of the substrate within the camera field of view, this substrate had some or many crevices, and a seafloor relief greater than 2 m or more, respectively (based on habitat parameters recorded as per Pacunski and Palsson (2001) and Haggarty, Lotterhos, and Shurin (2017)). Alternatively, substrate was classified as Low if rocky substrate was found in less than 50% of the camera field of view, complexity was rated Low if none or

few crevices were observed, and if the vertical relief was less than 2 m, relief was recorded as Low.

#### **2.2.4 Statistical analysis**

##### **Inshore rockfish abundance and fish diversity comparisons**

I pooled inshore rockfish species (*Sebastes melanops*, *S. nebulosus*, *S. caurinus*, *S. maliger*, *S. nigrocinctus*, *S. auriculatus*, *S. mystinus*, and *S. ruberrimus*) data within each site for each paired survey method and analyzed the pooled data as a group because observations of inshore rockfish species were uncommon across all methods. All inshore rockfish > 10 cm were included in the pooled data and no distinction was made between adult and juvenile inshore rockfish. An exploratory analysis was conducted to remove outliers and detect multicollinearity among independent variables *sensu* Zuur, Ieno, and Elphick (2010). Two sites were removed from the DS and BV analyses and 4 sites were removed from the TV and BV comparisons due to the survey duration being cut short by equipment malfunction.

To test my hypothesis that passive BV data are commensurate with active TV and DS data, I examined inshore rockfish abundance and species richness estimates generated by BV surveys at each site relative to those values generated by paired TV and DS. This will allow me to determine the degree of consensus in inshore rockfish abundance and species richness between active and passive surveys. For the TV-BV survey comparisons, only inshore rockfish observed within 50 m depth along the TV transect were included in the TV-BV comparisons. Species richness estimates included all fish observed at each site with each paired method. I used tests of correlation to quantify the strength and direction

of correspondence for both inshore rockfish abundance and species richness estimates between the paired surveys.

I used the Mann-Whitney U test to compare mean inshore rockfish abundance estimates and mean diversity metrics derived with the R package *vegan* (Oksanen et al. 2011) between paired surveys. Fish diversity metrics and analyses included all fish, not only inshore rockfish, observed by the TV, DS, and BV methods. The following community indices were calculated for each method across all sites:

- Species richness,  $S$  (count of number of different species observed at each site with each method)
- Shannon-Wiener diversity index  $H' = -\sum P_i \log_e(P_i)$  (Shannon and Weaver 1963);
- Pielou evenness index  $J' = H' / \ln(S)$  (Pielou 1975), where  $P_i$  is the relative abundance of the  $i$ th taxon in a sample containing  $S$  taxa.

### **Abiotic drivers of inshore rockfish abundance and fish diversity**

To test my hypothesis that estimates derived from the active methodologies, TV and DS, are more influenced by spatial heterogeneity than passive BV, I used an information-theoretic approach to identify the abiotic variables that best explain rockfish abundance and species richness with each survey method. Abiotic predictor variables included four seabed terrain characteristics (rocky substrate, relief, complexity and depth) (Table 2-1).

I standardized (mean = 0, standard deviation = 1) continuous variables prior to modeling to allow comparison of effect sizes. I combined predictors into candidate sets of variables *sensu* Burnham and Anderson (2002) and competed sets of generalized linear

models (GLM) in R (version 1.0.136, R Core Team 2017). A GLM was chosen because it allows for a non-normal error distribution of the response variable.

The Akaike information criterion (AIC; Akaike 1973) for the global model, which included all variables, was compared against the AIC of the same model minus one block of variables (Table 2-2). All possible combinations of variable blocks were tested, including the blocks in isolation. When competing models were deemed not to be different ( $\Delta \text{AIC} < 2$ ), I selected the most parsimonious model. The model for each survey method that had the lowest AIC score (and highest AIC weight) was identified as the top model for predicting inshore rockfish abundance and species richness. To validate the top model, I used a visual inspection of quantile-quantile plots, residuals versus the fitted (predicted) values plots, and correlation values between variables to verify the assumptions of normality and homoscedasticity of the residuals, and variable independence, respectively. The regression outputs for inshore rockfish abundance and species richness analyses are in Appendix B.

#### ***Towed video and baited video inshore rockfish abundance analyses***

To test the abiotic variables that best explain TV rockfish abundance, I modeled inshore rockfish density against rocky substrate, relief, complexity, and depth. I measured rockfish density as the number of inshore rockfish observed during a TV survey divided by the transect area. This analysis included all inshore rockfish and seabed terrain characteristics observed at the site along the entire transect length. A Gamma distribution (inverse link) was used in the GLMs, as the data were continuous, always positive and non-normally distributed. One site where zero inshore rockfish were observed was removed from the analyses and thus parameter estimates are contingent on inshore rockfish being present at a site.

For the BV analyses, I modeled the MeanCount of inshore rockfish abundance against rocky substrate, relief, complexity, and depth. MeanCount was rounded to the nearest integer and these count data were always positive and non-normally distributed. A negative binomial distribution (log link) was used for this analysis as a Poisson distribution yielded variance in rockfish counts that exceeded the mean and thus were over-dispersed.

#### ***Dive and baited video inshore rockfish abundance analyses***

To test the influence of abiotic variables on DS inshore rockfish abundance, I modeled the number of inshore rockfish observed per site against rocky substrate, relief, complexity, and depth. A negative binomial distribution (log link) was used for this analysis as a Poisson distribution yielded variance in rockfish counts that exceeded the mean and thus were over-dispersed.

For the BV surveys paired with the DS, at 37 out of the 56 sites surveyed, the MeanCount of inshore rockfish abundance was zero. Therefore, I used the presence and absence of inshore rockfish observed at a site as the response variable to test the influence of abiotic variables on rockfish presence. I modeled inshore rockfish presence and absence against rocky substrate, relief, complexity, and depth. A binomial distribution (logit link) was used in the GLMs, as the data were binary, a 0 or 1 (inshore rockfish being absent or present, respectively), and non-normally distributed.

#### ***Towed, dive and baited video species richness analyses***

To test the influence of abiotic variables on species richness, I modeled species richness (number of unique fish species, not only including inshore rockfish) observed at each site with each method against rocky substrate, relief, complexity, and depth. These data were

positive and non-normally distributed therefore the DS and BV models conformed to a Poisson distribution (log link).

For the TV surveys, transect area varied between survey locations. To take the difference in survey area into account species richness observed on a TV survey was divided by TV transect area at that site. I used a Gamma distribution (inverse link) as the species density data were continuous and positive.

**Table 2-1 Predictors hypothesized to explain inshore rockfish abundance and fish diversity for towed video, dive and baited video surveys.**

Survey method	Category		Predictor	Description
TV	Habitat	Topographic heterogeneity	ROCK	Proportion of rocky substrate on TV transect
			COMPLEX	Proportion of high complexity on TV transect
			RELIEF	Proportion of high relief on TV transect
	Bathymetry		DEPTH	Median depth in meters
DS	Habitat	Topographic heterogeneity	ROCK	Proportion of rocky substrate every 2 m along two 30 m transects
			COMPLEX	Proportion of high complexity every 2 m along two 30 m transects
			RELIEF	Proportion of high relief every 2 m along two 30 m transects
	Bathymetry		DEPTH	Average depth in metres every 2 m along two 30 m transects.
BV	Habitat	Topographic heterogeneity	ROCK	Amount of rocky substrate in camera field of view. High if rocky substrate > 50 % camera field of view, Low if rocky substrate in < 50% camera field of view.
			COMPLEX	High complexity having some or many crevices, Low complexity having none or few crevices.
			RELIEF	High vertical relief if seafloor relief > 2 m or more, Low vertical relief if seafloor relief < 2 m.

**Table 2-2 Explanatory variables were grouped into model sets according to the type of influence hypothesized to explain towed video, dive and baited video inshore rockfish abundance or fish diversity.**

Model set	#	Model variables	Hypothesis: rockfish abundance predicted by:
Global Model		ROCK + COMPLEX + RELIEF + DEPTH	All habitat and depth factors
Habitat	1	ROCK + COMPLEX + RELIEF	All habitat
	2	ROCK	Rocky substrate only
	3	RELIEF + COMPLEX	Topographic heterogeneity
Bathymetry	4	DEPTH	Depth only
Substrate + Bathymetry	5	ROCK + DEPTH	Rock substrate and depth
Topographic heterogeneity + Bathymetry	6	COMPLEX + RELIEF + DEPTH	Topographic heterogeneity and depth

### Comparison of effort between paired methods

I determined the minimum number of personnel needed to conduct a single survey as well as a categorical measure of the level of training needed and the approximate cost of the equipment used. I also estimated how much time it took to complete a single sample using each method.

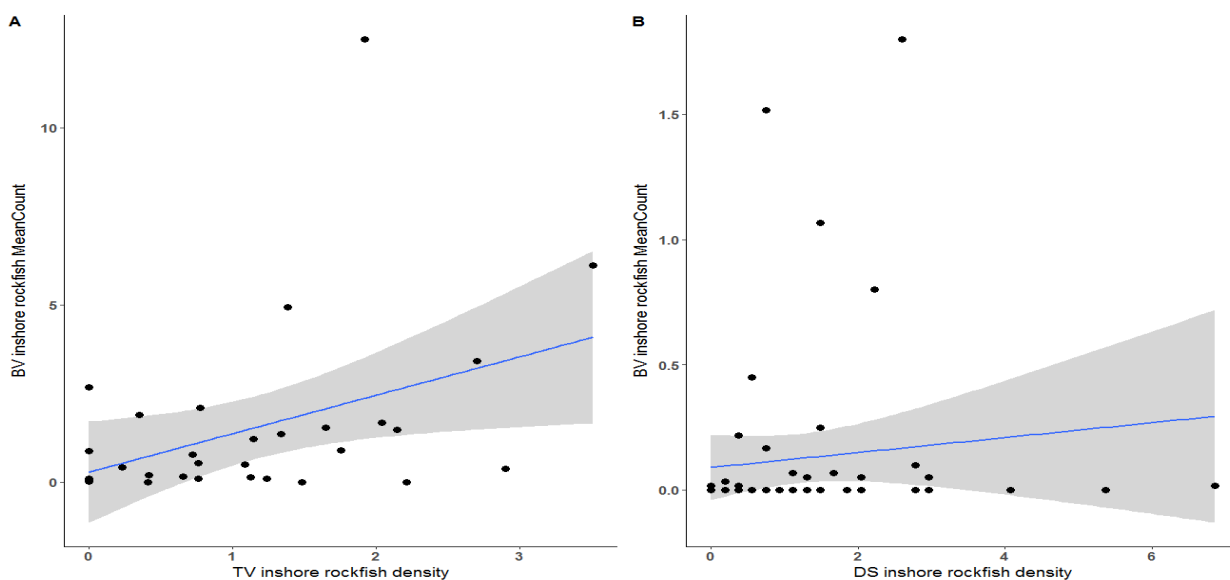
## 2.3 Results

### 2.3.1 Comparison of inshore rockfish abundance and fish diversity estimates

A Spearman correlation test between the paired TV – BV surveys showed a significant positive monotonic relationship between inshore rockfish abundance detected with the TV and BV surveys (Spearman's rank correlation  $S = 2897.7$ ,  $p\text{-value} = 0.05$ ) (Figure 2-4 A). However, the strength of association between TV and BV inshore rockfish abundance was weak ( $\rho = 0.36$ ). A significant and positive, weak monotonic relationship ( $\rho =$

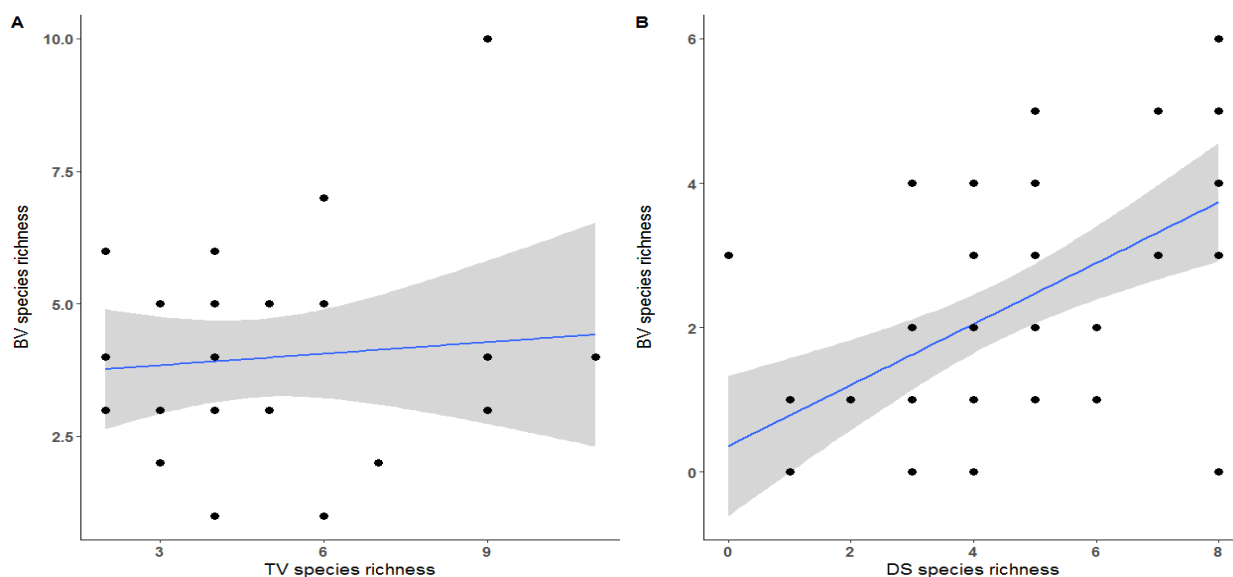
0.32) was also observed between the DS and BV surveys (Spearman's rank correlation,  $S = 17815$ ,  $p\text{-value} = 0.02$ ) (Figure 2-4 B).

When compared to both the TV and the DS, BV failed to detect inshore rockfish at sites where rockfish were observed by the active survey methods. This was more common in the DS-BV comparison. Even at sites where the greatest inshore rockfish density was observed with the DS, no inshore rockfish were detected by the BV (bottom right, Figure 2-4 B). At some of the sampling sites BV MeanCount was much higher than the MeanCount observed for the majority of sites (*e.g.* BV MeanCount > 10 in the BV-TV comparison, Figure 2-4 A). This indicates the BV surveys did not reach a maximum point of inshore rockfish abundance while the DS and TV abundance values continued to increase.



**Figure 2-4 A test of correlation was used to quantify the relationship of inshore rockfish abundance observed by paired A) TV and BV and B) dive and BV surveys.**

No significant monotonic relationship was observed between TV and BV species richness ( $\rho = -0.07$ ; Spearman's rank correlation  $S = 4831.2$ ,  $p\text{-value} = 0.69$ ) (Figure 2-5 A). The Spearman correlation test between the DS and BV paired surveys showed a significant positive monotonic relationship between species richness detected at each site with the DS and the BV surveys (Spearman's rank correlation  $S = 17815$ ,  $p\text{-value} = 0.02$ ) (Figure 2-5 B). However, the strength of the relationship between DS and BV species richness was weak ( $\rho = 0.45$ ). Similar to the relationships observed between the paired methods for inshore rockfish abundance, BV failed to detect fish at sites where fish were observed by the active survey methods. Congruence in species richness observed between the paired surveys was highest at sites where BV detected the greatest number of species.



**Figure 2-5** A test of correlation was used to quantify the relationship of species richness observed by paired A) TV and BV and B) dive and BV surveys.

Even though comparison of TV and BV inshore rockfish abundance and species richness showed little congruence between paired surveys at a site-level, no significant differences were observed between the mean TV and BV inshore rockfish abundance and fish diversity estimates generated across all sites (Figure 2-6, A – D, Table 2-3). When examining the list of all species observed on the TV and BV surveys (Tables 2-4 and 2-5), the BV surveys detected more unique genera and species than the TV surveys. This difference is largely attributed to small, pelagic species observed by the BV (*e.g.* the surf perches, Embiotocidae). The TV surveys failed to detect small, pelagic species other than Puget Sound rockfish, and mobile predators (*e.g.* *Squalus suckleyi*). More rockfish went unidentified to the species level by the BV when compared to the TV (BV unidentified 17 rockfish to the TVs 13) (Table 2-4).

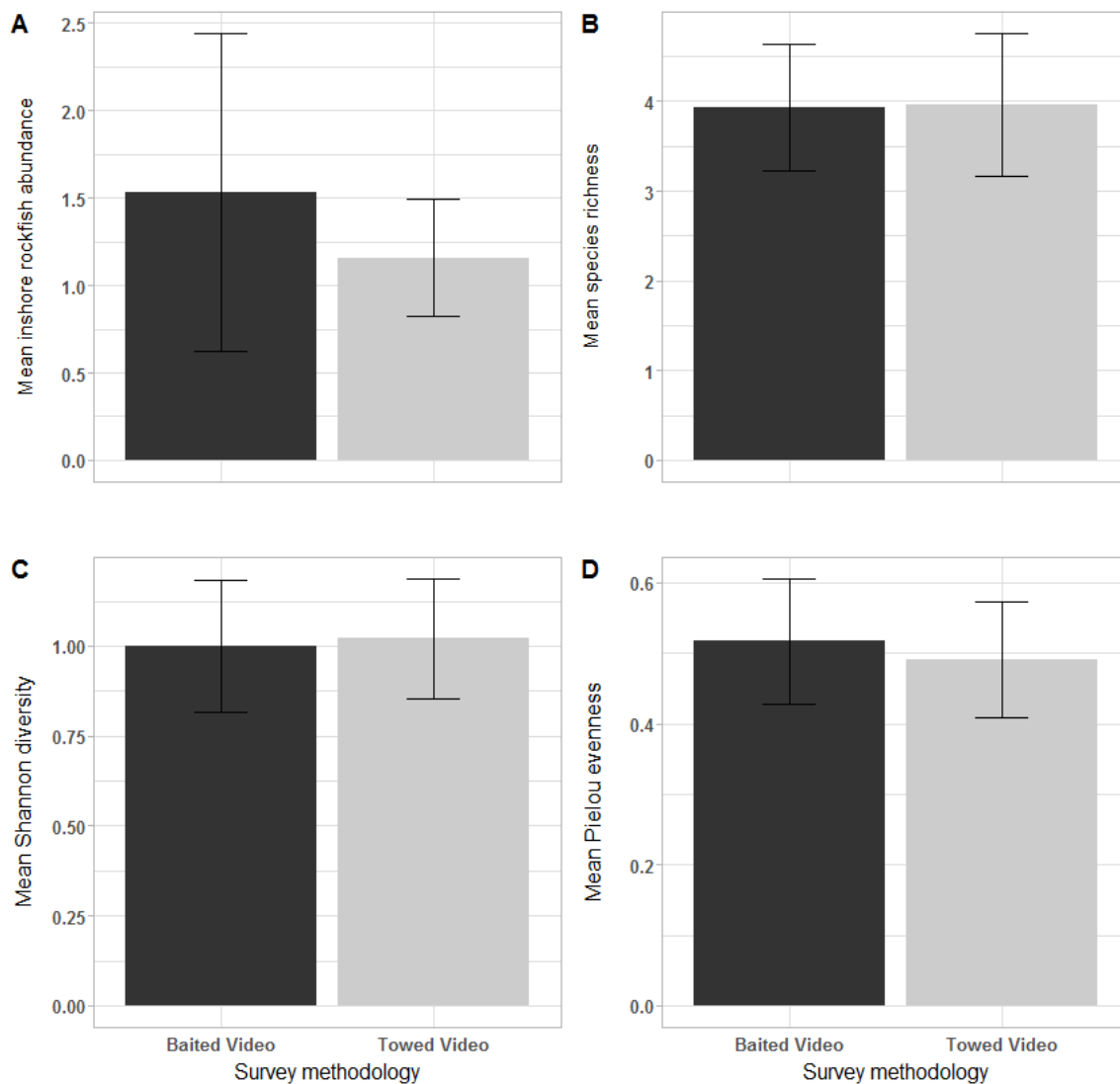
Significant differences were observed between DS and BV mean inshore rockfish abundance and average Shannon-Wiener and species richness diversity estimates (Figure 2-7, A – C, Table 2-3). The DS observed more inshore rockfish, other rockfish species, and a greater number of fish species and diversity as measured using the Shannon-Wiener diversity index. In the DS-BV comparison, BV failed to detect rockfish species (*i.e.* black and yellowtail rockfishes) and other demersal fish species such as Cabazon (*Scorpaenichthys marmoratus*) and painted greenling (*Oxylebius pictus*) that were observed with the DS (Tables 2-4 and 2-5). No difference was observed between the DS and BV surveys in species evenness as measured using the Pielou evenness index (Figure 2-7, D).

In the TV and BV comparison, quillback rockfish (*Sebastes maliger*) was the most abundant inshore rockfish observed for both methods (Table 2-4). Copper rockfish (*S.*

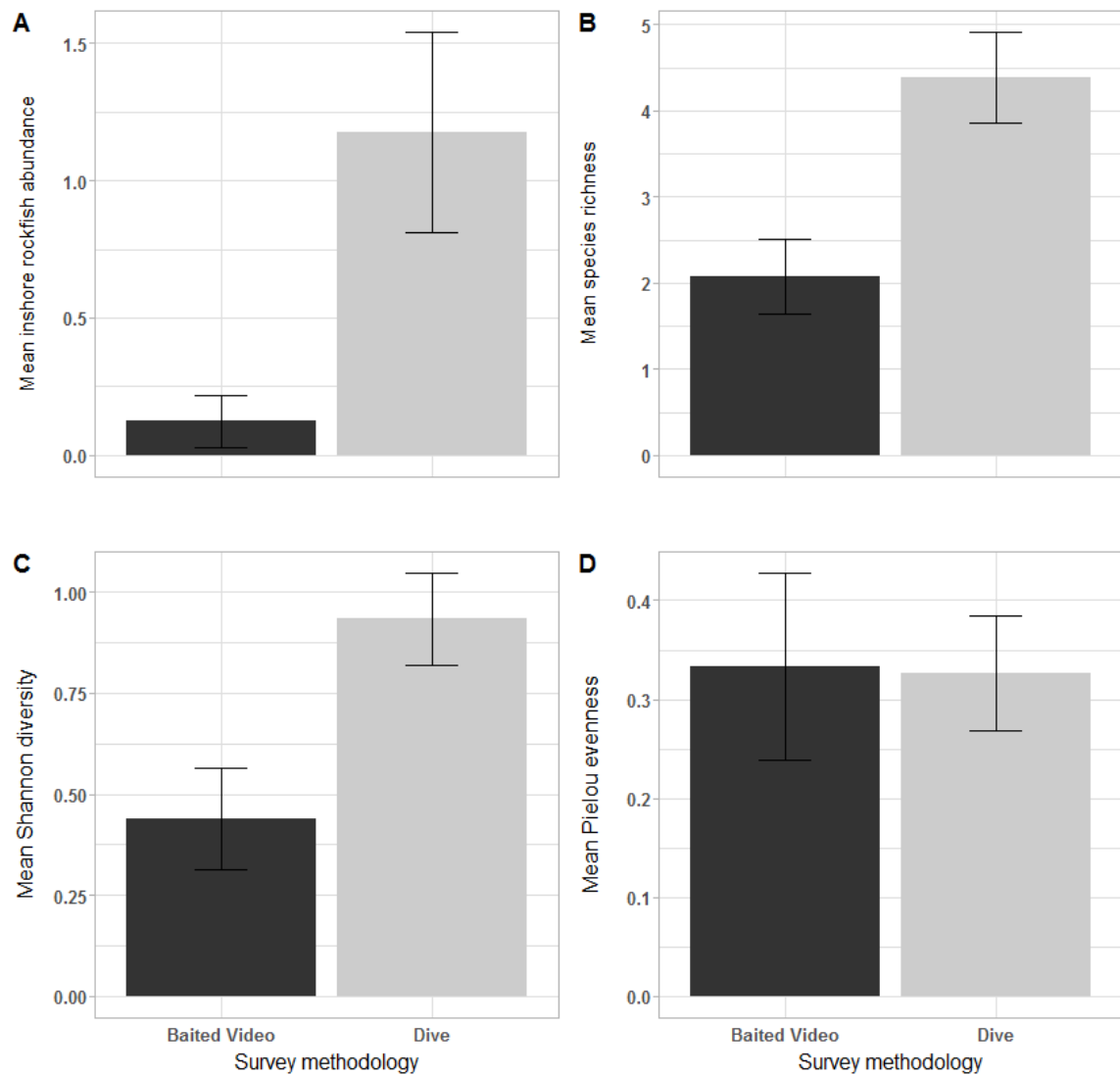
*caurinus*) was the most abundant inshore rockfish detected by both methods in the DS and BV comparison (Table 2-4).

**Table 2-3 Mean and standard deviation of inshore rockfish abundance and fish diversity estimates observed by paired TV and BV surveys and by paired dive and BV surveys along with significance between paired surveys.**

Estimate	Method	Mean +/- SD	Mann Whitney U test
Rockfish abundance	TV	1.16 +/- 0.94	W = 484.5, p-value = 0.61
	BV	1.53 +/- 2.55	
	DS	1.18 +/- 1.37	W = 2492, p-value = 4.74e-11
	BV	0.12 +/- 0.36	
Species richness	TV	3.97 +/- 2.22	W = 439.5, p-value = 0.88
	BV	3.93 +/- 1.96	
	DS	4.39 +/- 1.98	W = 2359, p-value = 2.21e-08
	BV	2.07 +/- 1.64	
Shannon-Wiener diversity index	TV	1.02 +/- 0.47	W = 457, p-value = 0.92
	BV	0.99 +/- 0.51	
	DS	0.93 +/- 0.43	W = 2268.5, p-value = 4.99e-07
	BV	0.44 +/- 0.47	
Pielou evenness index	TV	0.49 +/- 0.23	W = 340, p-value = 0.40
	BV	0.52 +/- 0.24	
	DS	0.33 +/- 0.22	W = 1294, p-value = 0.35
	BV	0.33 +/- 0.36	



**Figure 2-6 Mean and 95% confidence intervals of A) BV MeanCount inshore rockfish and TV inshore rockfish per 100m<sup>2</sup> within 50 m depth, B) species richness, C) Shannon-Wiener diversity, and D) Pielou evenness between the TV and BV surveys.**



**Figure 2-7 Mean and 95% confidence intervals of A) BV MeanCount inshore rockfish and DS inshore rockfish per 100m<sup>3</sup>, B) species richness, C) Shannon-Wiener diversity, and D) Pielou evenness between the DS and BV surveys**

**Table 2-4 Rockfish detected during TV, DS and BV surveys on the coast of British Columbia, Canada. The numbers of TV and DS rockfish are the sum of the number of individuals observed per site with each method. TV numbers are rockfish observed within 50 m depth. BV rockfish numbers are the sum of rockfish MaxN observed per site.**

Phylum Class Order	Species	Common Name	TV	BV	DS	BV
Chordata Actinopterygii Scorpaeniformes	<i>Sebastes spp.</i>	Rockfishes	13	17	3	-
	<i>Sebastes auriculatus</i>	Brown Rockfish*	-	-	25	8
	<i>Sebastes brevispinis</i>	Silvergray Rockfish	-	3	-	-
	<i>Sebastes caurinus</i>	Copper Rockfish*	20	21	250	33
	<i>Sebastes elongatus</i>	Greenstriped Rockfish	3	-	-	-
	<i>Sebastes emphaeus</i>	Puget Sound Rockfish	42	3	203	11
	<i>Sebastes entomelas</i>	Widow Rockfish	5	1	-	-
	<i>Sebastes flavidus</i>	Yellowtail Rockfish	62	102	7	-
	<i>Sebastes maliger</i>	Quillback Rockfish*	186	116	62	6
	<i>Sebastes melanops</i>	Black Rockfish*	7	15	6	-
	<i>Sebastes miniatus</i>	Vermilion Rockfish	-	2	-	-
	<i>Sebastes nebulosus</i>	China rockfish*	16	6	-	-
	<i>Sebastes nigrocinctus</i>	Tiger Rockfish*	3	1	-	-
	<i>Sebastes ruberrimus</i>	Yelloweye Rockfish*	10	2	-	-
Number of rockfish individuals			367	289	556	58
Number rockfish species			11	12	7	4

\* Inshore rockfish pooled and used in analyses

**Table 2-5 Fishes (other than rockfish) detected during the TV, DS, and BV surveys on the coast of British Columbia, Canada. The numbers of TV and DS fish are the sum of the number of individuals observed per site with each method. TV numbers are fish observed within 50 m of depth. BV fish numbers are the sum of fish MaxN observed per site.**

Phylum Class Order	Species	Common Name	TV	BV	DS	BV
Chordata Actinopterygii Scorpaeniformes	<i>Scorpaenichthys marmoratus</i>	Cabezon	-	-	1	-
	<i>Hemilepidotus hemilepidotus</i>	Red Irish Lord	-	4	-	-
	<i>Ophiodon elongatus</i>	Lingcod	23	11	29	19
	<i>Hexagrammos decagrammus</i>	Kelp Greenling	99	18	132	12
	<i>Hexagrammos stelleri</i>	Whitespotted Greenling	-	-	5	2
	<i>Oxylebius pictus</i>	Painted Greenling	-	-	7	-
Pleuronectiformes	Unidentified species	Flatfishes	4	8	2	3
Perciformes	<i>Cymatogaster aggregate</i>	Shiner Perch	-	1	3745	864
	<i>Rhacochilus vacca</i>	Pile Perch	-	26	586	87
	<i>Brachyistius frenatus</i>	Kelp Perch	-	400	141	62
	<i>Embiotoca lateralis</i>	Striped Sea Perch	-	-	520	41
Gasterosteiformes	<i>Aulorhynchus flavidus</i>	Tube-snout	-	-	-	30
Chondrichthyes Squaliformes	<i>Squalus suckleyi</i>	Pacific Dogfish	-	1	-	-
Chimaeriformes	<i>Hydrolagus colliei</i>	Spotted Ratfish	13	38	-	-
Number of fish individuals			139	507	5168	1120
Number of fish species			4	9	10	9

### **2.3.2 Abiotic drivers of inshore rockfish abundance**

#### **Towed video and baited video surveys**

Complexity and relief were included in the top TV and BV inshore rockfish abundance models (Table 2-6). For both TV and BV, less rockfish were observed at sites with higher relief. Increasing complexity at a site led to more inshore rockfish detected during the 30-minute BV survey, but less inshore rockfish observed along the TV transects. The top BV model carried 55% of the AIC weight of evidence while there was more model uncertainty for the TV models with the 80% of the AIC weight of evidence spread among the four top models. Depth and rocky substrate were included in the top TV models. More inshore rockfish were observed during the TV transects at deeper sites while the relationship between TV rockfish abundance and rocky substrate varied within the found top TV models.

#### **Dive and baited video surveys**

A similar relationship was observed between DS and BV inshore rockfish abundance and the predictors related to topographic heterogeneity, complexity and relief (Table 2-6). An increase in the complexity and relief observed at a site led to more inshore rockfish observed on both the dive and BV surveys. Depth was also included in the top DS and BV models but the relationship was inconsistent between methods. More inshore rockfish were detected on the DS as depth increased while less rockfish were observed at deeper sites during the BV surveys.

**Table 2-6 Model selection of generalized linear models of inshore rockfish abundance observed with A) TV and B) BV surveys and with C) dive and D) BV surveys.**

#	Rockfish abundance ~ Model variables	AICc	$\Delta$ AICc	AIC weight
<b>Paired towed video and baited video</b>				
A) Towed video surveys				
3	-COMPLEX – RELIEF	64.41	0	0.25
4	DEPTH	64.42	0.01	0.25
1	-COMPLEX + RELIEF + ROCK	65.16	0.75	0.17
2	-ROCK	65.74	1.33	0.13
6	-COMPLEX + RELIEF + DEPTH	66.54	2.13	0.09
5	-ROCK + DEPTH	66.95	2.54	0.07
global	-COMPLEX + RELIEF + ROCK + DEPTH	67.39	2.98	0.06
B) Baited Video surveys				
3	COMPLEX – RELIEF	75.4	0	0.55
6	COMPLEX – RELIEF – DEPTH	77.7	2.29	0.17
1	COMPLEX – RELIEF – ROCK	78.43	3.03	0.12
2	ROCK	79.55	4.15	0.07
global	COMPLEX – RELIEF – ROCK – DEPTH	80.82	5.42	0.04
4	-DEPTH	81.18	5.78	0.03
5	ROCK - DEPTH	82.18	6.78	0.02
<b>Paired dive surveys and baited video surveys</b>				
C) Dive surveys				
6	COMPLEX + RELIEF + DEPTH	307.15	0	0.47
3	COMPLEX + RELIEF	308.15	1	0.28
global	COMPLEX + RELIEF + ROCK + DEPTH	309.61	2.47	0.14
1	COMPLEX + RELIEF + ROCK	310.02	2.87	0.11
2	ROCK	319.08	11.93	0.00
4	DEPTH	319.48	12.33	0.00
5	ROCK + DEPTH	320.07	12.92	0.00
D) Baited video surveys				
4	-DEPTH	68.86	0	0.37
6	COMPLEX + RELIEF – DEPTH	70.02	1.16	0.21
5	ROCK – DEPTH	71.09	2.23	0.12
3	COMPLEX + RELIEF	71.11	2.25	0.12
global	COMPLEX + RELIEF – ROCK – DEPTH	71.73	2.88	0.09
2	ROCK	72.32	3.46	0.06
1	COMPLEX + RELIEF – ROCK	73.41	4.55	0.04

### **2.3.3 Abiotic drivers of species richness**

#### **Towed video and baited video surveys**

Predictors relating to topographic heterogeneity were included in the top TV and BV species richness models (Table 2-7). More species were observed during the TV and BV surveys as the proportion of high relief observed at a site increased. Similar to the relationship found between TV and BV inshore rockfish abundance and complexity, increasing complexity at a site led to more species detected during the 30-minute BV survey, but less species observed along the TV transects. Rocky substrate was also included in the top TV and BV species richness models, but the relationship varied. As the amount of rocky substrate increased at a site, more species were detected during a BV survey while less were observed on a TV transect. Depth was included in the 3<sup>rd</sup> top BV model that contained 17% of the AIC weight of evidence and BV species richness increased as deployment depth of the BV increased.

#### **Dive and baited video surveys**

For the DS and BV surveys, complexity and relief were included in the top DS and BV species richness models (Table 2-7). More fish species were detected at complex, high relief sites for both DS and BV surveys. Rocky substrate was included in the 2<sup>nd</sup> best DS and BV species richness models that carried 22% and 21% of the AIC weight of evidence, respectively. Less fish species were observed on both the dive and BV surveys as the proportion of rocky substrate increased. For BV species richness, depth was included in the top model and less species were observed at deeper sites.

**Table 2-7 Model selection of generalized linear models of species richness observed with A) TV and B) BV surveys and with C) dive and D) BV surveys.**

#	Species richness ~ Model variables	AICc	$\Delta$ AICc	AIC weight
<b>Paired towed video and baited video</b>				
A) Towed video surveys				
3	-COMPLEX + RELIEF	13.1	0	0.39
2	-ROCK	14.8	1.7	0.17
1	-COMPLEX + RELIEF + ROCK	15.22	2.12	0.14
6	-COMPLEX + RELIEF – DEPTH	15.57	2.47	0.11
4	DEPTH	15.68	2.58	0.11
5	-ROCK – DEPTH	17.35	4.25	0.05
global	-COMPLEX + RELIEF – ROCK – DEPTH	17.93	4.83	0.04
B) Baited video surveys				
2	ROCK	113.19	0.00	0.38
3	COMPLEX + RELIEF	114.58	1.39	0.19
4	DEPTH	114.83	1.64	0.17
5	ROCK + DEPTH	115.35	2.16	0.13
6	COMPLEX + RELIEF + DEPTH	116.79	3.60	0.06
1	COMPLEX + RELIEF	117.18	3.99	0.05
global	COMPLEX + RELIEF + ROCK + DEPTH	119.54	6.35	0.02
<b>Paired dive surveys and baited video surveys</b>				
C) Dive surveys				
3	COMPLEX + RELIEF	217.51	0	0.54
S1	COMPLEX + RELIEF - ROCK	219.27	1.76	0.22
6	COMPLEX + RELIEF - DEPTH	219.82	2.31	0.17
global	COMPLEX + RELIEF – ROCK - DEPTH	221.7	4.19	0.07
2	ROCK	230.04	12.53	0.00
4	DEPTH	230.52	13.01	0.00
5	ROCK + DEPTH	232.16	14.65	0.00
D) Baited video surveys				
6	COMPLEX + RELIEF - DEPTH	183.9	0	0.63
global	COMPLEX + RELIEF - ROCK - DEPTH	186.06	2.17	0.21
4	- DEPTH	187.73	3.83	0.09
5	ROCK - DEPTH	188.73	4.84	0.06
3	COMPLEX – RELIEF	193.83	9.93	0.00
1	COMPLEX – RELIEF + ROCK	195.88	11.99	0.00
2	ROCK	198.1	14.21	0.00

### 2.3.4 Comparison of effort between paired methods

More personnel were required to conduct a single TV and DS survey than a BV survey (Table 2.8). Dive surveys are typically conducted in pairs and a third person is needed to act as a dive tender. During the TV surveys, one person was responsible for manually adjusting the height of the TV over the substrate while another maintained the computer equipment and TV cable, and a third drove the boat. The BV rig that I used could be deployed and retrieved with one person, but it is easier with two – one to drive the boat and another for BV deployment and pick-up.

More training was needed to conduct the DS than either the TV or BV surveys. Scientific diver certification (*e.g.* Canadian Association for Underwater Science) along with a number of first aid courses are prerequisites for conducting research-level DS. Training for the TV survey involved instruction on setting up the TV electronics along with the transducer used to collect location and depth data during the TV deployment. Little training was needed to conduct the BV survey although, both the TV and BV surveys required expertise to identify the taxa for annotation of the video imagery. More complex and expensive equipment was needed for the TV survey than the equipment needed to conduct a BV survey. However, the TV could access depths down to 200 m while the BV was restricted to 60 m depth (based on the Intova camera specifications). The TV rig also included lasers, allowing for fish size and survey area calculations. DS equipment costs were related to the SCUBA and safety equipment needed for the divers.

A sample collected by both active and passive survey methods required time in the field to conduct the surveys and time in the lab for data entry and for the video methods, time for imagery annotation. A single TV survey (including the survey time and video annotation) took almost 2 hours more than a single BV survey. Set up and retrieval of the

TV equipment took more time than a BV survey and TV video annotation can take up to 60 minutes more than annotating a 30-minute BV survey. DS took less time than the BV surveys because the DS needed no time for video annotation and little to no processing time for data entry.

A detailed overview of how the methods deployed compare under different environmental conditions, to survey species with certain characteristics along with considerations for survey logistics and survey metrics are found in Appendix 1.

**Table 2-8 Comparison of effort between paired methods**

	<b>Towed video</b>	<b>Dive surveys</b>	<b>Baited video</b>
<b>Minimum personnel needed</b>	3	3	1
<b>Training needed</b>	Moderate	High	Low
<b>Equipment cost</b>	~ \$5,000	~\$3,000	~\$1,500
<b>Time (minutes)</b>			
Survey preparation: setting up video equipment, gearing up in SCUBA equipment	30	20	15
Survey: including equipment launch and 30 minute video survey; dive ascent, descent and safety stop	40	45	35
Retrieval: pick up of video equipment and clean up, dive equipment gear down	30	30	15
Video annotation	150	N/A	90
Data entry	30	20	20
<b>Total time (minutes)</b>	280	115	175

## 2.4 Discussion

BV and TV surveys were about equally reliable at estimating mean inshore rockfish abundance, but BV better estimated fish diversity across all sites. The DS outperformed the BV surveys for both inshore rockfish abundance and fish diversity metrics. Consistent with my expectations, I observed a positive relationship between active and passive paired surveys for inshore rockfish abundance, but the relationship was weak without a clear pattern in abundance between the paired methods. At a site-level, a positive relationship was found between the paired DS-BV surveys for species richness, but there was little congruence between site-level TV-BV species richness.

### **Towed video – baited video comparison**

BV detected a greater number of unique species (21 species) across sampling locations than TV (15 species), although mean diversity metrics were similar between the TV and BV surveys. TV failed to detect small, pelagic species such as the surfperches (Family Embiotocidae) and mobile predators like the Pacific dogfish (*Squalus suckleyi*). Previous research found BV better at detecting predatory and scavenging species, where bait was needed to attract these rarely occurring species into the camera field of view (Willis and Babcock 2000; Cappel, Speare, and De'Ath 2004; Harvey et al. 2007). These species may display avoidance behaviour towards boat noise and/or the TV itself, biasing TV abundance and diversity estimates, and have been reported by other researchers (Assis, Narváez, and Haroun 2008). Pelagic species observed during BV deployment were indifferent to the camera platform, but present in, or passed through the camera field of view during the survey while mobile predators actively sought out the bait at the BV platform.

A comparison of active and passive survey methods focussed on assessing Australian seabream (*Pagrus auratus*) abundance found TV surveys to generate the least biased abundance and biomass estimates, in addition to detecting greater species richness than BV (Morrison and Carbines 2006). However, these surveys were conducted over low relief, in a relatively homogenous environment. In contrast, my surveys took place over structurally complex and highly heterogeneous rocky reefs - preferred rockfish habitat (Love et al. 1990; Love, Yoklavich, and Thorsteinson 2002). I found seabed relief and complexity contributed to TV underestimating inshore rockfish abundance. Unlike previous results on rockfish-habitat associations (Love, Yoklavich, and Thorsteinson 2002; Anderson and Yoklavich 2007; Haggarty, Shurin, and Yamanaka 2016), TV inshore rockfish abundance decreased as seabed relief and complexity increased. Successful TV operation is difficult over high relief habitat due to the need of rapid manual vertical adjustment of the camera. The up-and-down movement of the camera during deployment to prevent the camera from snagging on rocks may have biased TV abundance and diversity estimates. Further, TV is unable to effectively survey microhabitats such as cracks and crevices (Holmes et al. 2013; Goetze et al. 2015), which likely led to underrepresentation of cryptic and crevice-dwelling fishes.

For the BV surveys paired with the TV, the BV captured the known relationship between inshore rockfish abundance and seabed complexity (Love et al. 1990; Love, Yoklavich, and Thorsteinson 2002; Haggarty, Lotterhos, and Shurin 2017). However, as the proportion of high relief observed in the BV field of view increased less inshore rockfish were detected while greater species richness was observed. Deployment of the BV over high relief terrain can cause the BV platform to settle on a slight angle. This can

cause BV camera orientation to face upwards and/or downwards and can also lead to the field of view being partially obstructed by boulders and/or benthic algae. A change in BV orientation may improve or worsen BV sampling performance for certain species. For example, an upward facing orientation would capture pelagic species better than demersal species (negatively influencing inshore rockfish abundance estimates) while a downward facing orientation would detect benthic species.

When compared to the TV, more rockfish went unidentified to species level by the BV. BV surveys depended upon bait at the platform to lure fish close to the camera. The unidentified BV rockfish were too far away to identify to species but their silhouette was visible enough to allow identification to genus. Fish detected in the BV survey are either attracted to the bait plume or the camera station, can also be attracted to other fish aggregating at the station, or may be indifferent to the BV but observed in the field of view. Antagonistic behaviour between fishes may have repelled these unidentified rockfish from coming closer to the BV, or caused competition for inspection of the BV (Armstrong, Bagley, and Priede 1992).

### **Dive survey – baited video comparison**

In other DS and BV comparisons, BV detected greater numbers of fish than the DS (Willis, Millar, and Babcock 2000; Gardner and Struthers 2013). The variability in my results could be attributed to focal species-specific responses to the methodology used and to differences in how the surveys were executed. Compared to the DS, BV failed to detect inshore rockfish at many of the sites sampled; underestimating inshore rockfish abundance, suggesting both BV site-level and mean inshore rockfish estimates are less reliable than the DS abundance results. However, BV did pick up the same pattern in

dominant inshore rockfish species that was observed during the DS and a positive relationship was observed between site-level DS and BV species richness.

The better performance of the DS to estimate both inshore rockfish abundance and fish diversity estimates relative to BV surveys may stem from the divers being able to observe and record individuals *in situ* whereas identification of individuals with the BV can be influenced by the video sample (*e.g.* how far the fish is away from the camera). Several studies have found divers are better than cameras at observing cryptic species because divers can search complex habitats and census cracks, crevices and lacunae in ways that cameras cannot (Watson et al. 2005; Stobart et al. 2007). Indeed, across all sites sampled the DS observed more rockfish species, other demersal fishes, and cryptic benthic species than the BV surveys, but both methods detected similar species of pelagic fishes. In other dive-video comparisons, diver visual methods have obtained greater abundance and species richness estimates than video methods (Watson et al. 2005; Colton and Swearer 2010), due to the advantages of the human eye over video technologies but this observation is not consistent between all diver-video comparisons (Willis and Babcock 2000; Willis, Millar, and Babcock 2000).

There is an upper limit to the number of fish that is visible in the BV field of view at a given moment, which may cause the BV to truncate abundance where densities are very high. This will produce conservative contrasts between areas with high and low fish density, but may result in failure to detect a difference between two areas where densities are different, but so high as to saturate the field of view in both places. I hypothesized the BV surveys would reach a maximum point of abundance, restricted by the BV field of view. There was no indication that the BV surveys reached a maximum point of inshore

rockfish abundance in the paired method comparisons. The DS surveys did however detect higher numbers of pelagic species than the BV surveys indicating the BV field of view may be more constrained than the DS area.

Fish detection with the BV is influenced by bait plume dynamics, local hydrology, and species-specific response characteristics to the bait (Armstrong, Bagley, and Priede 1992; Cappo, Speare, and De'Ath 2004; Harvey et al. 2007; Colton and Swearer 2010; Terres et al. 2015). BV may introduce biases from the behaviours adopted by fish in response to the bait and the dynamics of attraction or repulsion are largely unknown (Sainte-Marie and Hargrave 1987; Armstrong, Bagley, and Priede 1992). Seasonal, reproductive and lunar patterns of activity in swimming speed, fish behaviour, and fish appetite likely affect fish attraction to bait, as will conspecific attraction, curiosity, predator presence or absence and home range sizes (Munro 1974; Newman and Williams 1995). Strongly site-attached and territorial species, like some rockfish (*e.g. Sebastes nigrocinctus* and *S. ruberrimus*), may not have been detected if the BV settled outside of their territory.

Reflecting the motile nature of the DS, the DS may have observed a larger area and more habitat types than the BV surveys, which could explain the greater DS inshore rockfish abundance and fish diversity estimates (Lomolino 2000; Gratwicke and Speight 2005). The DS covered 540 m<sup>3</sup> per site. For BV deployments if the radius of bait detection was approximately 10 m like that estimated in Willis, Millar, and Babcock (2000), the effective sampling area was only 314 m<sup>2</sup>. The BV only allowed for relative fish abundance (MeanCount) comparisons and BV abundance results could not be expressed in terms of density because of the unknown size of the bait plume. Bait plume area is influenced by prevailing rates of advection and turbulent diffusion thus BV

sampling area may vary widely in comparison to the relatively static DS area. Variation in dispersal of the bait plume is likely to contribute to within-area variance. Modeling the radius of attraction of the bait plume (Merritt et al. 2011; Heagney et al. 2007), combined with the knowledge of fish reactions to bait and their swimming speeds will allow for more direct density estimates and between-site and methodology relative density comparisons.

In the DS and BV comparisons, seabed terrain characteristics were included in the top BV inshore rockfish abundance and species richness models even though BV sacrifices the fine-grained spatial heterogeneity data, a consequence of the stationary nature of the video method. Complexity and relief were positive predictors of inshore rockfish abundance and species richness for both the DS and BV surveys. Inshore rockfish are found over structurally complex rocky substrates (Love et al. 1990; Love, Yoklavich, and Thorsteinson 2002; Haggarty, Lotterhos, and Shurin 2017) and the positive relationship between diversity and topographic heterogeneity is an ecological pattern found in both terrestrial and aquatic ecosystems (Beatty 1984; Cusson and Bourget 1997; L. A. Levin et al. 2001; Harman, Harvey, and Kendrick 2003; Willis and Anderson 2003; Shumway, Hofmann, and Dobberfuhl 2007). Topographic heterogeneity is thought to increase overall diversity by increasing the number of available niches, allowing for more ways to utilize environmental resources while decreasing species exclusion by reducing interspecific competition, and reducing mortality by providing refuge space from predators (Levin and Paine 1974; Huston 1979; Johnson 2007). The BV captured the known relationship between inshore rockfish and seabed terrain characteristics along with

positive relationship between diversity and topographic heterogeneity, indicating a BV survey may be used to capture fish-habitat associations.

Rockfish partition habitats by depth (Richards 1986, 1987; Matthews 1990; Yoklavich et al. 2000; Anderson and Yoklavich 2007; Laidig, Watters, and Yoklavich 2009; Ingram and Shurin 2009; Haggarty, Shurin, and Yamanaka 2016) and fish diversity and abundance is often distributed along a depth gradient with many species only occurring within certain depth ranges (Bacheler et al. 2016; Jankowski, Gardiner, and Jones 2015; Galaiduk et al. 2017). Both the DS and BV surveys were limited in that they only assessed inshore rockfish species within shallow depth ranges that typically include juvenile quillback and copper rockfishes. On the DS, more inshore rockfish were observed as depth increased while inshore rockfish were less likely to be present on the BV surveys at deeper sites. The variation between methods in the relationship between inshore rockfish abundance and depth may be a result of the BV luring fish from adjacent habitat and introducing variation into observed species-environment relationships (Moore, Harvey, and Van Niel 2009; Galaiduk et al. 2017).

Due to the stationary nature of the BV surveys, areas with low abundance of inshore rockfish may have negatively biased BV rockfish detection. Many fish populations show density-dependent habitat selection with ranges contracting as abundance declines (Marshall and Frank 1995). At low fish abundance, fish may occur only in small areas that have optimal habitat. If the BV was deployed at a location with few rockfish, the bait used may not have been an effective enough attractant to lure the fish from their rocky crevice and therefore, micro-site state (and attendant variation/environmental stochasticity) may introduce stochasticity to BV data. To sample fish populations of low

abundance, longer BV deployments and/or increasing the number of samples may be needed.

## **2.5 Conclusion**

My results indicate that a BV survey can be used as a low cost and effort method for examining rockfish abundance and fish diversity over rocky reefs from the nearshore subtidal waters down to depths greater than 20 m. The BV surveys were logistically simpler to organize, required fewer skilled field personal, fewer hours in the field as compared to conducting a TV or dive survey, and can be deployed from a smaller, less costly vessel than either the TV or DS. BV surveys can be conducted rapidly, generating more data over greater temporal and spatial scales than a TV survey or a DS. BV surveys can be used as an assessment technique to evaluate species after disturbance or easily employed by non-government organizations seeking to initiate an inexpensive community-monitoring program.

The BV platform was more economical than the TV (cost ratio ~1:5) and did not experience the same challenges as the TV when deployed over structurally complex rocky reefs. Using BV as a survey method eliminates DS biases (*e.g.* influence of a diver on fish behaviour) and provides a means of visually enumerating fish at depths inaccessible to SCUBA divers and therefore, can access a greater range of rockfish depth gradients in addition to avoiding the health and safety concerns associated with the DS. However, my results suggest that the abundance of the survey population influenced BV inshore rockfish detection probability. Increasing the sample size and survey duration may improve BV rockfish estimates and increase congruence between BV and DS. Like the TV surveys, the BV survey provides permanent records that can be repeatedly

examined, but also requires technical expertise and post-survey time process video data after the surveys while DS requires little post-survey data processing. BV platforms can be developed to include live-feed topside, allowing for a decrease in post-survey time to process video data. The differences in results between paired surveys reveals that the methodology used to sample populations matters and highlights the need for carefully planned probabilistic sampling designs guided by power analysis, using a methodology that meets research objectives and the physical characteristics of the study location while considering the life-history and behaviour of the study species. The summary table of how the methods I deployed rate for different environmental conditions, species characteristics, survey logistics and survey metrics can be used to help identify a suitable method to meet research and monitoring objectives.

## **Chapter 3 Comparison of dive and baited video surveys for evaluating spatial marine conservation measures**

### **3.1 Introduction**

Spatial marine conservation measures such as marine protected areas (MPAs), fisheries closures, and harvest refugia, are strategies used to conserve and restore depleted populations of marine fishes and to support the sustainability of fisheries (Lauck et al. 1998; Hilborn et al. 2004; Roberts, Hawkins, and Gell 2005; Spalding, Fish, and Wood 2008). Spatial marine conservation measures are management tools used to increase the size, abundance and diversity of species protected within and sometimes adjacent to them (Mosquera et al. 2000; Halpern and Warner 2002; Halpern 2003; Alcala et al. 2005; Lester et al. 2009; Babcock et al. 2010). However, effectiveness of marine conservation areas may vary widely (Allison, Lubchenco, and Carr 1998; Hilborn, Micheli, and De Leo 2006; Mora et al. 2006; Joachim Claudet et al. 2008; Edgar 2011).

A potentially significant source of variance affecting the perceived efficacy of spatial marine conservation measures are sampling error and bias, as it can be difficult to precisely and accurately estimate species diversity and abundance with underwater surveys. Different observational methods may yield different results, making performance estimates of spatial marine conservation measures equivocal. With the proliferation of underwater video as a survey and research tool (Mallet and Pelletier 2014), underwater survey methodologies have expanded beyond conventional diver-based assessment of fish abundance and size, making it even more important to understand methodology-driven systematic and random error.

Diver-based surveys have been employed for more than six decades (Brock 1954) and remain a standard for marine research. Dive surveys (DS) typically provide a measure of spatial variability as divers move through the study area and record individuals observed along transects or at specific points. Repeated DS at fixed survey locations are also used to monitor temporal variability (*e.g.* see Partnership for Interdisciplinary Studies of Coastal Oceans dive monitoring program which has been ongoing since 1999, [www.piscoweb.org](http://www.piscoweb.org)). Like any survey methodology, DS are subject to inherent sampling error and biases that may influence species detection, from underestimating cryptic and/or small fish densities (Ackerman and Bellwood 2000; Willis 2001; Bozec et al. 2011) to species-specific behavioral responses to the divers' presence underwater (Kulbicki 1998; Samoilys and Carlos 2000; Watson and Harvey 2007; MacNeil et al. 2008; Bozec et al. 2011; Dickens et al. 2011; Pais and Cabral 2017). Additional factors influencing species detection during DS are observer experience (Thompson and Mapstone 1997; Bernard et al. 2013), diver swim speed (Lincoln Smith 1988), site and sea state heterogeneity (Sale and Douglas 1981; Cheal and Thompson 1997; Edgar and Barrett 1999; MacNeil et al. 2008), and survey area (Sale and Sharp 1983; Cheal and Thompson 1997). Further, DS are constrained by human physiology that greatly restricts survey effort through maximum depth and survey duration limitations.

Reflecting the technical and logistical limitations of DS, video-based surveys are becoming increasingly popular for enumeration of marine and aquatic abundance and biodiversity. Rapid technological evolution has made video systems economical and easy to build and deploy. Myriad video-based techniques such as remotely operated vehicle (ROV), towed video, and stationary drop video and survey designs are now routinely

employed to document species and habitat use (see review by Pelletier and Mallet 2014). The capacity to deploy video systems to great depths and for long time periods makes possible expansive temporal and spatial coverage and replication.

A common underwater video survey technology is the baited remote underwater video survey (BV). For more than two decades, BV surveys have been used to examine population dynamics of marine species (Ellis and DeMartini 1995). Like its extremely popular terrestrial analogue, the camera trap (Burton et al. 2015), a BV survey is a stationary method that uses an attractant placed close to the camera to act as an olfactory and/or visual lure. Stationary survey technologies like the BV depend upon motile organisms moving into the field of view of the camera to be enumerated.

Video-based systems, such as the BV, address DS limitations as survey effort is not nearly as restricted by depth and survey duration (Willis and Babcock 2000; Willis, Millar, and Babcock 2000; Westera, Lavery, and Hyndes 2003; Harvey et al. 2007). Further, BV surveys are logistically simpler to organize, require fewer field personnel with less specialized training, and fewer hours in the field as compared to the requirements of a DS of equal sampling coverage. BV surveys also avoid the health and safety concerns specific to SCUBA diving. BV surveys provide permanent records that can be repeatedly examined, but also necessitate technical expertise and time post-survey to process video data.

The survey methodology used for monitoring spatial marine conservation areas has potential random and systematic errors that affect the precision and accuracy of the data collected. It is important to identify measurement uncertainty associated with each method to understand the reliability of your results and to make sound conclusions. BV

surveys have recorded more fish, and fish of greater size ranges than DS, revealing significant differences in abundance and size of fish between protected and unprotected sites (Gardner and Struthers 2013). Alternatively, DS may observe greater species diversity and richness at both the species and family taxonomic level relative to BV surveys (Colton and Swearer 2010; Lowry et al. 2012; Langlois et al. 2010). Species-specific differences between DS and BV surveys are also well documented, with DS recording more cryptic species and reef-associated species than BV surveys, while more mobile fishes and generalist carnivores were detected on BV surveys (Colton and Swearer 2010; Lowry et al. 2012; Langlois et al. 2010).

The methodology used for monitoring marine conservation areas must also have the requisite accuracy and precision to discriminate change between protected and unprotected sites, preventing type II statistical errors (retaining a false null hypothesis of no conservation area effect) (Schluter and Whitlock 2009). Further, the discrimination power of the methodology used will also be affected by the magnitude of difference between protected and unprotected areas. The methodological errors associated with fish abundance and diversity estimates must be less than the among-treatment differences in order to be able to assess marine conservation area effectiveness.

I compared fish abundance between paired DS and BV surveys inside and outside spatial marine conservation areas designated for inshore rockfish conservation in the Northeast Pacific Ocean on the south coast of British Columbia, Canada to address both a methodological and conservation question. Given the trade-offs between DS and BV methodologies, I ask whether BV data generate the same conclusions about conservation area effectiveness as data derived from DS, and whether the marine conservation areas

have greater inshore rockfish abundance than paired locations outside the conservation areas. Equivalence between DS and BV data would suggest that BV surveys could be used in place of DS for spatial marine conservation area monitoring, yielding more data for less investment.

Inshore rockfish of the genus *Sebastes* include black rockfish (*S. melanops*), China rockfish (*S. nebulosus*), copper rockfish (*S. caurinus*), quillback rockfish (*S. maliger*), tiger rockfish (*S. nigrocinctus*), brown rockfish (*S. auriculatus*), blue rockfish (*S. mystinus*), and yelloweye rockfish (*S. ruberrimus*) (Love, Yoklavich, and Thorsteinson 2002; Yamanaka and Logan 2010). These species are typically found proximate to high rugosity rocky substrates such as boulder fields down to 200 m of depth (Love et al. 1990; Love, Yoklavich, and Thorsteinson 2002; Haggarty, Shurin, and Yamanaka 2016). Inshore rockfish are philopatric (tending to remain near a particular rocky site or area), grow to large sizes, reach sexual maturity at a late age and experience variable recruitment that is dependent upon environmental conditions, making them particularly vulnerable to overfishing (Parker et al. 2000; Yamanaka and Logan 2010)

Rockfish possess closed swimbladders (physoclistous) that trap expanding gasses causing internal injury (barotrauma) when rapidly pulled to the surface in nets or on lines, making catch and release ineffective and placing a conservation premium on non-capture survey methods. High site fidelity and small home range sizes make rockfish particularly susceptible to localized overfishing (Love et al. 1990; Parker et al. 2000; Love, Yoklavich, and Thorsteinson 2002), but also make them suitable for recovery using spatial protection strategies (Carr and Reed 1993; Yoklavich 1998; Parker et al. 2000; Hamilton et al. 2010).

Rockfish Conservation Areas (RCAs) are spatial fisheries closures enacted by the federal Fisheries and Oceans Canada (DFO) in response to sharp declines in inshore rockfish catches during the 1990s (Yamanaka and Logan 2010; Haggarty 2013). Rockfish Conservation Areas prohibit commercial and recreational fishing gear that targets rockfish or leads to rockfish bycatch (Yamanaka and Logan 2010), but do permit some recreational and commercial fishing activities (such as prawn traps and off-bottom net gear). Reduced fishing in RCAs should theoretically enhance the likelihood of an individual rockfish reaching sexual maturity and lead to increased abundance and biomass of rockfish inside of RCAs (Love et al. 1990; Yamanaka and Logan 2010; Hilborn et al. 2004).

I also compared fish diversity estimates between paired DS and BV surveys inside and outside of RCAs to evaluate the indirect effects RCA protection has on fish diversity. Even though RCAs are set aside for inshore rockfish protection, they also provide conservation benefit for other species caught with fishing gear prohibited for use within RCAs (e.g. *Ophiodon elongates* caught with hook and line fishing gear). Biological responses to spatial marine conservation measures are variable among fishes (Mosquera et al. 2000). A positive reserve response is generally observed for species targeted by fisheries and for large species while no reserve response has been found for non-targeted species (Mosquera et al. 2000; Côté, Mosqueira, and Reynolds 2001; Claudet et al. 2006). Additionally, a negative reserve response has been observed for non-targeted species, perhaps due to greater predation pressure within the protected area (McClanahan et al. 1999).

Evaluations of spatial marine conservation areas such as RCAs or Marine Protected Areas for rockfish have yielded inconclusive results to date. Larger rockfish and greater rockfish biomass have been found inside Californian reserves relative to outside of reserves (Hamilton et al. 2010; Keller et al. 2014). In older reserves, larger rockfish and greater rockfish biomass were observed, while a 1 year old reserve showed no difference from outside areas (Paddack and Estes 2000). In British Columbia, there is little indication rockfish recovery is influenced by RCA protection. No significant increase in rockfish density has been observed as a result of RCA protection (Marliave and Challenger 2009; Cloutier 2011; Haggarty, Shurin, and Yamanaka 2016) nor have differences in the size structure of rockfishes inside RCAs and unprotected areas been found (Haggarty, Shurin, and Yamanaka 2016).

Most studies evaluating rockfish response to spatial marine conservation areas have used DS (Paddack and Estes 2000; Hamilton et al. 2010; Marliave and Challenger 2009; Cloutier 2011; Haggarty, Lotterhos, and Shurin 2017) with the exception of a bottom trawl study and a ROV survey in California (Keller et al. 2014; Karpov, Bergen, and Geibel 2012), and a more recent and extensive assessment of BC RCA effectiveness conducted with a ROV (Haggarty, Shurin, and Yamanaka 2016). Dive surveys are limited by diver physiology to shallow depths. Bottom trawls can access deeper areas but are destructive and are not effective over rocky habitats. ROVs address dive and bottom trawl limitations as they are not restricted by depth and can be used in structurally complex habitats. However, ROVs are expensive and logistically challenging to deploy, limiting sample size, upon which the precision and confidence in statistical estimates hinges. Likewise to ROVs, BV systems can address these challenges while also being

economical to build and tailored for optimal performance in local conditions (Harvey et al. 2003; Watson and Huntington 2016). With fewer cost restrictions, BV surveys potentially allow for greater sample sizes and thus, improved precision in statistical estimates.

I evaluated the relative effectiveness of DS and BV surveys for monitoring RCA efficacy by comparing inshore rockfish abundance and fish diversity estimates between paired DS and BV surveys inside and outside RCAs. I asked: 1) Does the survey method used affect measured RCA efficacy? Further, I seek to assess the response of inshore rockfish and fish diversity to RCA establishment by asking: 2) Is there evidence of RCA effectiveness as measured by increased inshore rockfish abundance and/or body size and increased fish diversity inside RCAs using DS and BV surveys?

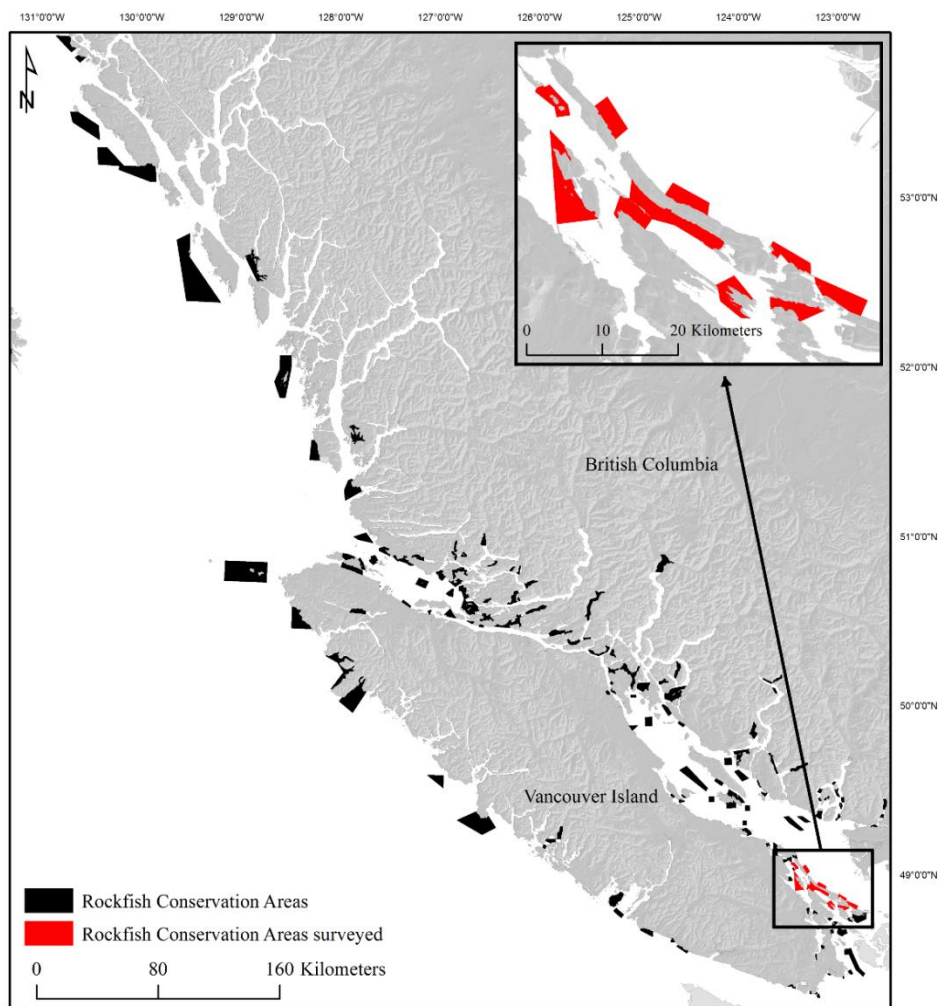
## **3.2 Methods**

### **3.2.1 Study locations**

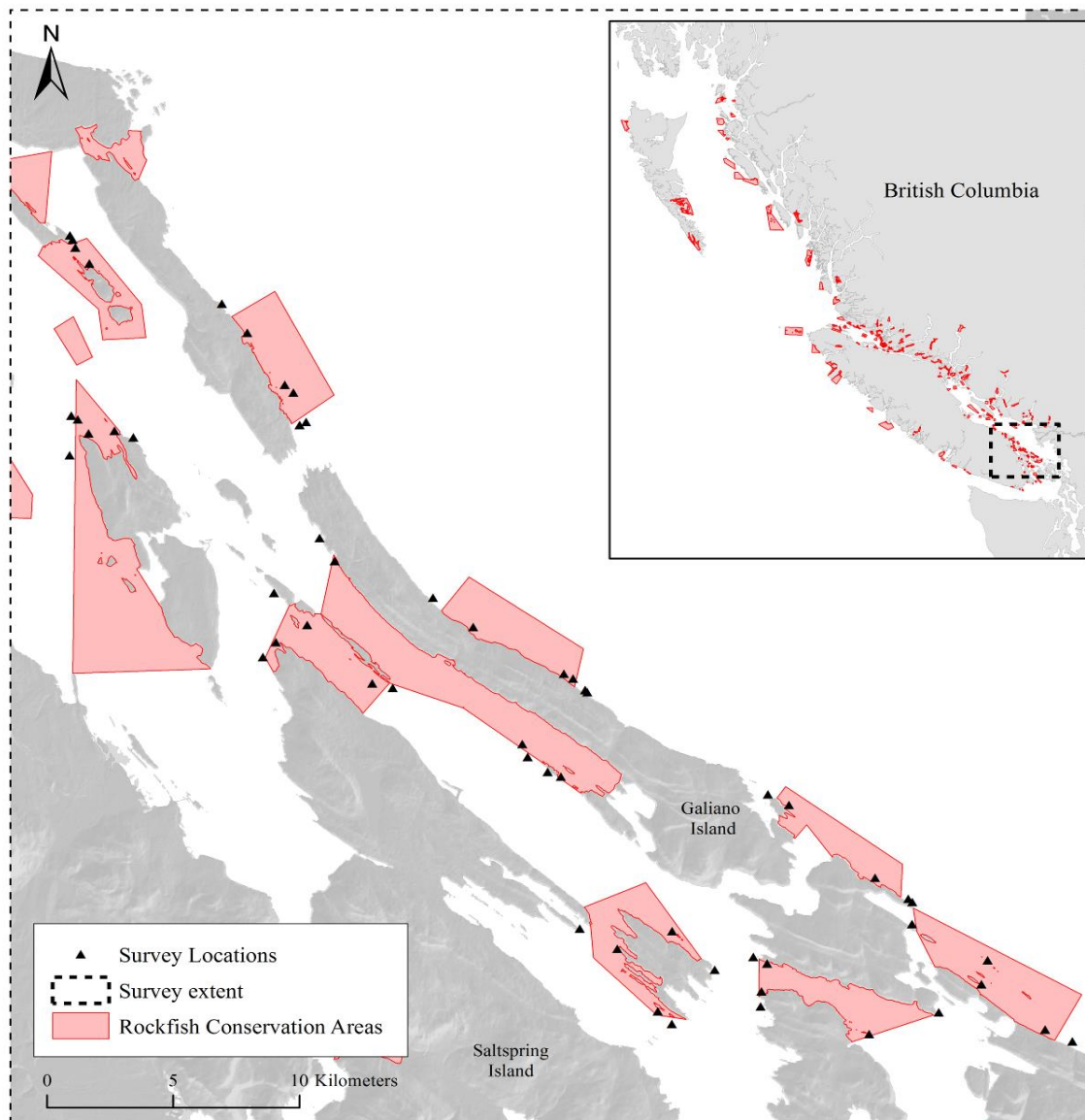
I compared data collected from inside the RCAs to data collected at geophysically similar sites adjacent to the RCAs that are open to fishing. There were no comparable survey data from before the RCAs were established. Thus, I was constrained to the survey design known as Control-Impact to test for RCA effects (Underwood 1992), which compares data outside of RCAs (control) to inside RCAs (impact). I conducted paired DS - BV surveys at 56 locations over nearshore rocky reefs on the south coast of British Columbia from September to November 2015 (Figures 3-1 and 3-2). Half of the sites were located inside RCAs ( $n = 28$ ) and half of the sites were outside of RCAs ( $n = 28$ ). A total of ten RCAs were surveyed with up to three survey sites per RCA.

I selected sampling sites with as similar of a habitat structure as possible since habitat features are an important source of variability for fish communities (Parnell et al. 2016; Claudet and Guidetti 2010; Miller and Russ 2014), and rockfish species segregate by habitat type and depth (Richards 1986; Matthews 1990; Anderson and Yoklavich 2007; Laidig, Watters, and Yoklavich 2009; Haggarty, Shurin, and Yamanaka 2016). Incorporating habitat type and depth in evaluation of spatial marine conservation areas allows us to determine if an observed response is related to protection or to intrinsic structural features within the protected areas (Pelletier et al. 2008; Claudet and Guidetti 2010).

I used bathymetric data in ArcGIS (ArcMap 10.2) to match geophysical profiles inside and outside of RCAs. I combined seafloor features (*i.e.* rocky substrate, slope, and curvature) generated on a 20 m grid to identify suitable rockfish habitat (bathymetry derivatives provided by D. Haggarty, DFO). To enable dive surveys, all sites were within 20 m depth. I used the NOAA Sampling Design Tool to randomly select survey sites targeting rocky substrate with similar slope and curvature. All paired “inside - outside” sites were more than 300 m apart. To avoid edge effects (Kellner et al. 2007; Starr et al. 2015), RCA survey sites were located at least 100 m inside the RCA boundary, and reference sites were between 100 - 1000 m from the nearest RCA boundary.



**Figure 3-1 Rockfish Conservation Areas (RCA) designated across the British Columbia coast. RCAs surveyed during this study are in red**



**Figure 3-2 Survey locations inside (n = 28) and outside (n = 28) of ten Rockfish Conservation Areas surveyed during this study.**

### 3.2.2 Field methods

#### Dive survey

At each site, a DS was completed wherein two SCUBA divers descended at predetermined coordinates and surveyed (using a compass) along an isobath running parallel to the adjacent shore. Survey depths were within 20 m and average transect depth ranged from 8.5 to 17.8 m. The first diver attached a 30 m transect line to their weight belt and the line was drawn out behind the divers as they proceeded along the isobath. Once the first 30 m transect was completed the divers swam back to the transect start while reeling in the transect line. The divers then swam perpendicular to transect direction 5 m shallower and completed a second transect following the opposite compass bearing to the first transect. Search effort was standardized by maintaining a swim speed of 4 m min<sup>-1</sup>.

The first diver recorded all observed fish > 10 cm in length by species within 1.5 m on each side of the transect line. Fish length was estimated visually and recorded for each fish in 10 cm size bins. The second diver followed the first and recorded habitat data as per Pacunski and Palsson (2001) and Haggarty, Lotterhos, and Shurin (2017) scoring relief, complexity, substrate type, and depth—recording data every 2 m. Relief accounted for changes in the vertical relief along transects while complexity recorded the amount of surface irregularity and the number of crevices observed in the seabed. Together, complexity and relief were used as a measure of topographic heterogeneity observed along the dive transects, combining both structural relief and roughness.

Total inshore rockfish density was calculated by dividing the number of inshore rockfish observed per site by survey volume (# inshore rockfish / 540 m<sup>3</sup>). Proportions of habitat parameters (rocky substrate, high relief and high complexity) were calculated by

adding the number of occurrences of a habitat parameter observed every 2 m along the two dive transects and dividing by the total number of habitat observations per site (30). Mean transect depth was calculated by taking the average of depth observed every 2 m along the two transects.

### **Baited video survey**

Unlike DS, which use mobile detection, BV systems use spatially fixed detection instruments and an attractant placed close to the camera to lure fish into the field of view. The 'soak times', or survey duration, in past studies ranges from 8 up to 120 minutes (Mallet and Pelletier 2014) - I used 30 minutes (soak times recommended by Stobart et al. 2007; Gladstone et al. 2012). Bait may increase the probability of fish entering the camera field of view of a BV system and has proven useful in areas with low fish densities and to sample targeted fish species that may be wary of divers' presence (Willis, Millar, and Babcock 2000; Watson et al. 2005; Watson and Harvey 2007; Harvey et al. 2007; Langlois et al. 2010; Watson et al. 2010; Dorman, Harvey, and Newman 2012). However, baited systems may introduce biases associated with differential species- or even individual-specific response to bait (Harvey et al. 2007; Hardinge et al. 2013). Further, limitations on the number of fish that may appear in the camera field of view at any one time complicates enumeration accuracy (Willis, Millar, and Babcock 2000; Cappo et al. 2003; Harvey et al. 2007; Dorman, Harvey, and Newman 2012; Hardinge et al. 2013). A further challenge of BV surveys is the inability to calculate the realized sampling volume because the dispersion area of the bait attractant plume is unknown, as are individual and species-specific responses to it (Willis, Millar, and Babcock 2000;

Willis and Babcock 2000; Heagney et al. 2007). This last point precludes measures of absolute density and therefore, BV surveys are limited to relative abundance measures.

The BV system consisted of a waterproof Intova camera (Intova Sport HD; intova.com) mounted on a polyvinyl chloride (PVC) frame with 2.25 Kg dive weights attached to each of the four legs (Figure 3 – 3). The camera was pointed forward towards the bait arm, with dual lights attached to the frame also pointing forward. A 1 m bait arm (20 mm diameter PVC) extended from the frame. At its distal end I secured a perforated plastic bait container typical of those used in prawn and crab traps. At rest, the camera sits ~ 50 cm off the bottom. Fresh bait was used for each deployment and consisted of approximately 150 grams of 3 minced Pacific Herring (*Clupea harengus*) mixed with 1 cup of commercial prawn and crab bait pellets (FirstMate Prawn & Crab Bait).

The BV was tethered topside via a polypropylene rope attached to a 32 cm diameter commercial surface buoy (Dan-fender B40 balloon fender). Following BV deployment, the support vessel remained >100 m away from the deployment area, engines off. Camera resolution was set to 1080p@30fps and the photo quality to Super Fine, while all other camera modes were left in the factory default Auto setting. Sampling sites were located more than 100 m apart to minimize potential overlap of bait odour plumes ensuring each site was independent of others (Willis and Babcock 2000; Cappo, Speare, and De'Ath 2004; Harvey et al. 2007; Heagney et al. 2007). Dive surveys were carried out prior to the BV sampling to eliminate potential bias associated with the effect of bait. I deployed the BV at the starting point of the first diver transect and at least 30-minutes after the conclusion of the DS to minimize possible bias related to the SCUBA diver effect on fish behaviour. One 30-minute BV was conducted following each DS.

I used the average number of individuals of the focal species (inshore rockfish *Sebastes spp.*) observed in the video sample (MeanCount) to quantify rockfish abundance (Conn 2011). MeanCount was estimated by randomly sampling 60 frames from each 30-minute video (Schobernd, Bacheler, and Conn 2014; Bacheler and Shertzer 2015). All inshore rockfish observed in each frame were identified to the lowest taxonomic level possible. Inshore rockfish MeanCount was calculated by adding the total number inshore rockfish observed in the 60 randomly selected frames and dividing by 60.

To quantify BV fish diversity I used the video metric MaxN. For every species identified in the 30-minute sample, the maximum number of individuals appearing in-frame simultaneously (*i.e.* the “snapshot” where the maximum number conspecifics appearing simultaneously appear) is recorded as that species’ MaxN. MaxN records every species observed in the 30-minute sample and avoids the recounting of individuals that move in and out of the static field of view (Willis, Millar, and Babcock 2000; Harvey et al. 2007).

For the BV surveys, I noted deployment depth of the camera and coarse categorical measures of seabed terrain characteristics observed in the camera field of view. Rocky substrate, complexity and seafloor relief were considered High if rocky substrate comprised >50% of the substrate within the camera field of view, this substrate had some or many crevices, and a seafloor relief greater than 2 m or more, respectively (based on habitat parameters recorded as per Pacunski and Palsson (2001) and Haggarty, Lotterhos, and Shurin (2017). Alternatively, substrate was classified as Low if rocky substrate was found in less than 50% of the camera field of view, complexity was rated Low if none or

few crevices were observed, and if the vertical relief was less than 2 m, relief was recorded as Low.

### 3.2.3 Statistical analysis

I pooled inshore rockfish species (*Sebastes melanops*, *S. nebulosus*, *S. caurinus*, *S. maliger*, *S. nigrocinctus*, *S. auriculatus*, *S. mystinus*, and *S. ruberrimus*) data within each site for each method and analyzed the pooled data as a group because observations of inshore rockfish species were too low to allow for species-specific analyses. All inshore rockfish > 10 cm were included in the pooled data and no distinction was made between adult and juvenile inshore rockfish. An exploratory analysis was conducted to remove outliers and detect multicollinearity among independent variables *sensu* Zuur, Ieno, and Elphick (2010). Two sites were removed from the analyses due to the DS being cut short by equipment malfunction.

I compared fish diversity between RCA ‘treatments’ using metrics from the R package *vegan* (Oksanen et al. 2011). The following community indices were calculated for all of the different species observed at each site with each method inside and outside of the RCAs:

- Species richness,  $S$  (count of the number of different species observed at each site with each method, not only including inshore rockfish);
- Shannon-Wiener diversity index  $H' = -\sum P_i \log_e(P_i)$  (Shannon and Weaver 1963);
- Pielou evenness index  $J' = H' / \ln(S)$  (Pielou 1975), where  $P_i$  is the relative abundance of the  $i$ th taxon in a sample containing  $S$  taxa.

I used the Mann-Whitney U to test for significant differences in pooled inshore rockfish mean abundance and fish diversity metrics inside and outside of RCAs between survey methods. I also compared inshore rockfish abundance inside and outside of RCAs by the individual RCA surveyed using a log Response Ratio (RR) (Equation 1). RR are often used to evaluate marine reserve effectiveness (Hamilton et al. 2010; Hedges, Gurevitch, and Curtis 1999) and has been used to evaluate RCA effectiveness (Haggarty, Shurin, and Yamanaka 2016). I used the RR to compare the mean density of inshore rockfish inside to outside of the RCAs.

#### **Equation 1 Log Reserve Ratio used to evaluate RCA effectiveness**

$$\log \text{ Reserve Ratio} = \log((\mu \text{ Rockfish Abundance}_{IN} + 1)/(\mu \text{ Rockfish Abundance}_{OUT} + 1))$$

where *Rockfish Abundance<sub>IN</sub>* is the mean inshore rockfish abundance observed per transect volume inside of RCAs and *Rockfish Abundance<sub>OUT</sub>* is the mean inshore rockfish abundance observed per transect volume outside of RCAs.

#### **Evaluation of RCA protection on dive survey rockfish size**

Fish size was only recorded during the DS, as the BV system did not allow for data collection on fish size. I compared the length frequency of copper and quillback rockfish found inside and outside of RCAs as these two rockfish species were most frequently observed (copper, n = 250 and quillback, n = 62) during the DS. The length frequency

was measured in 10 cm size bins and I used the Kolmogorov-Smirnov test and length frequency histograms to examine differences in size based on RCA protection status.

### **Influence of RCA protection on inshore rockfish abundance and species richness**

I used an information-theoretic approach to test hypotheses about the influence of abiotic variables on inshore rockfish abundance and species richness observed during the DS and BV surveys. Abiotic predictor variables included 4 seabed terrain characteristics (complexity, relief, and rocky substrate), depth, and RCA protection status (Table 3-1). I standardized (mean = 0, standard deviation = 1) continuous variables prior to modeling to allow comparison of effect sizes. I combined predictors into candidate sets of variables *sensu* Burnham and Anderson (2002) and compared sets of generalized linear models (GLM) in R (version 1.0.136, R Core Team 2017). A GLM was chosen because it allows for a non-normal error distribution of the response variable.

The Akaike information criterion (AIC; Akaike 1973) for the global model, which included all variables, was compared against the AIC of the same model minus one block of variables (Table 3 - 2). All possible combinations of variable blocks were tested, including the blocks in isolation. When competing models were deemed not to be different ( $\Delta \text{AIC} < 2$ ), I selected the most parsimonious model. The model for each survey method that had the lowest AIC weight was identified as the top model for predicting rockfish abundance. To validate the top model, I used a visual inspection of quantile-quantile plots, residuals versus the fitted (predicted) values plots, and correlation values between variables to verify the assumptions of normality and homoscedasticity of

the residuals, and variable independence, respectively. The regression output for inshore rockfish abundance and species richness analyses is in Appendix C.

**Table 3-1 Predictors hypothesized to explain inshore rockfish abundance and fish diversity inside and outside of Rockfish Conservation Areas near the Southern Gulf Islands, British Columbia, Canada.**

Method	Category		Predictor	Description
DS	Habitat	Topographic heterogeneity	ROCK	Proportion of rocky substrate observed every 2 m along two 30 m transects
			RELIEF	Proportion of high relief observed every 2 m along two 30 m transects
			COMPLEX	Proportion of high complexity observed every 2 m along two 30 m transects
	Bathymetry		DEPTH	Average depth in metres observed every 2 m along two 30 m transects.
	Protection		RCA	Site located inside RCA or outside RCA
BV	Habitat	Topographic heterogeneity	ROCK	Amount of rocky substrate in camera field of view. High if rocky substrate in > 50 % camera field of view, Low if rocky substrate < 50% camera field of view.
			RELIEF	High vertical relief if seafloor relief > 2 m or more, Low vertical relief is seafloor relief < 2 m.
			COMPLEX	High complexity having some or many crevices, Low complexity having none or few crevices.
	Bathymetry		DEPTH	Depth of BV deployment in meters
	Protection		RCA	Site is located in RCA or outside RCA

### *Inshore rockfish abundance analyses*

To test the influence of abiotic variables on DS rockfish abundance, I modeled the number of rockfish observed at each site against complexity, relief, rocky substrate, depth, and RCA protection. These data were always positive and non-normally distributed therefore I initially presumed these data to conform to a Poisson distribution (log link). However, the Poisson distribution yielded variance in rockfish counts that exceeded the mean and were over-dispersed necessitating a negative binomial distribution (log link).

For the BV analyses, at 37 out of the 56 sites surveyed, the MeanCount of rockfish abundance was zero. Therefore, I used the presence and absence of rockfish observed at a site as the response variable to test the influence of abiotic variables on rockfish presence. I modeled rockfish presence and absence against rocky substrate, relief, complexity, depth and RCA protection. A binomial distribution (logit link) was used in the GLMs, as the data were binary, a 0 or 1 (rockfish being absent or present, respectively), and non-normally distributed.

### *Species richness analyses*

To test the influence of abiotic variables on species richness, I modeled species richness (# of unique fish species, not only including inshore rockfish) observed at each site with each method against rocky substrate, relief, complexity, depth, and RCA protection.

These data were positive and non-normally distributed therefore the models conformed to a Poisson distribution (link = “log”).

**Table 3-2 Explanatory variables were grouped into model sets according to the type of influence hypothesized to explain inshore rockfish abundance or fish diversity.**

<b>Model set</b>	<b>#</b>	<b>Model variables</b>	<b>Hypothesis: rockfish abundance predicted by:</b>
Global Model		ROCK + RELIEF + COMPLEX + DEPTH + RCA	All habitat, depth, and protection factors equally
Habitat	1	ROCK + RELIEF + COMPLEX	All habitat equally
	2	ROCK	Rock substrate only
	3	RELIEF + COMPLEX	Rugosity only
Bathymetry	4	DEPTH	Depth only
Protection	5	RCA	Protection only
Habitat + Bathymetry	6	ROCK + COMPLEX + RELIEF + DEPTH	All habitat and depth
Habitat + Protection	7	ROCK + COMPLEX + RELIEF + RCA	All habitat and protection
Bathymetry + Protection	8	DEPTH + RCA	Depth and protection
Substrate + Bathymetry	9	ROCK + DEPTH	Rock substrate and depth
Substrate + Protection	10	ROCK + RCA	Rock substrate and protection
Substrate + Bathymetry + Protection	11	ROCK + DEPTH + RCA	Rock substrate, depth, and protection factors equally
Topographic heterogeneity + Bathymetry	12	COMPLEX + RELIEF + DEPTH	Topographic heterogeneity and depth
Topographic heterogeneity + Protection	13	COMPLEX + RELIEF + RCA	Topographic heterogeneity and protection
Topographic heterogeneity + Bathymetry + Protection	14	COMPLEX + RELIEF + DEPTH + RCA	Topographic heterogeneity, depth, and protection factors equally

### **3.3 Results**

#### **3.3.1 Comparison of dive and baited video RCA efficacy**

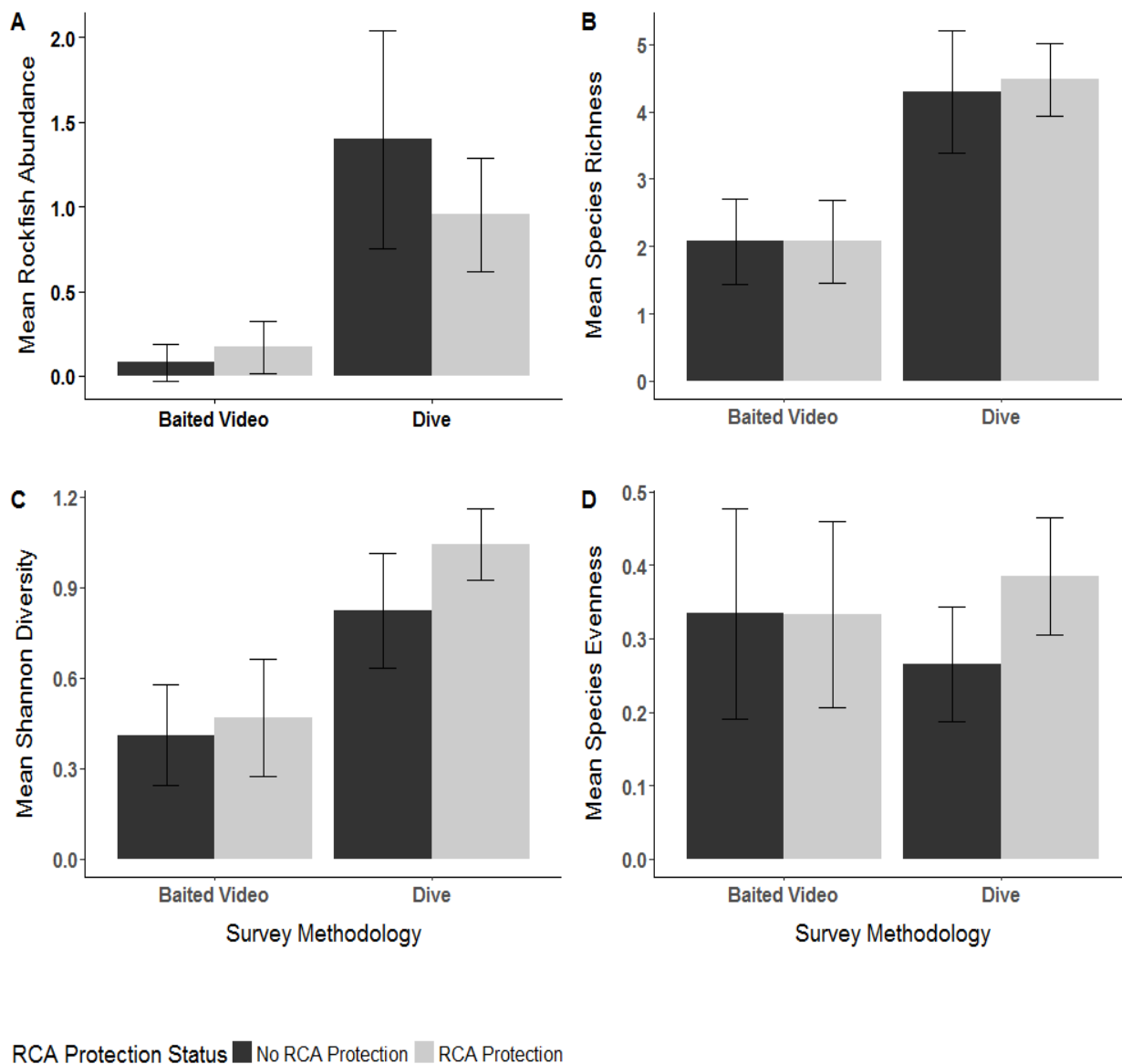
There were no statistically significant differences in inshore rockfish abundance or fish diversity inside and outside RCAs, as measured by both methods (Figure 3-3, A-D). Dive surveys observed significantly more inshore rockfish and a greater number of fish species and diversity as measured using the Shannon-Wiener diversity index (Table 3- 3) than the BV, but the impact of RCA protection was consistent between methods. Dive surveys and BV surveys detected the same dominant inshore rockfish species inside and outside of the RCAs (Table 3-4). Greater variance was observed for DS mean inshore rockfish abundance estimates inside and outside of RCAs than the BV mean rockfish abundance estimates. This is likely due to the high number of sites (37 out of the 54 analyzed) where the BV surveys did not detect rockfish when compared to the DS. The rockfish abundance pattern observed inside and outside of the 10 RCAs surveyed was also similar between methods (Figure 3-4).

#### **3.3.2 Evidence of RCA effectiveness**

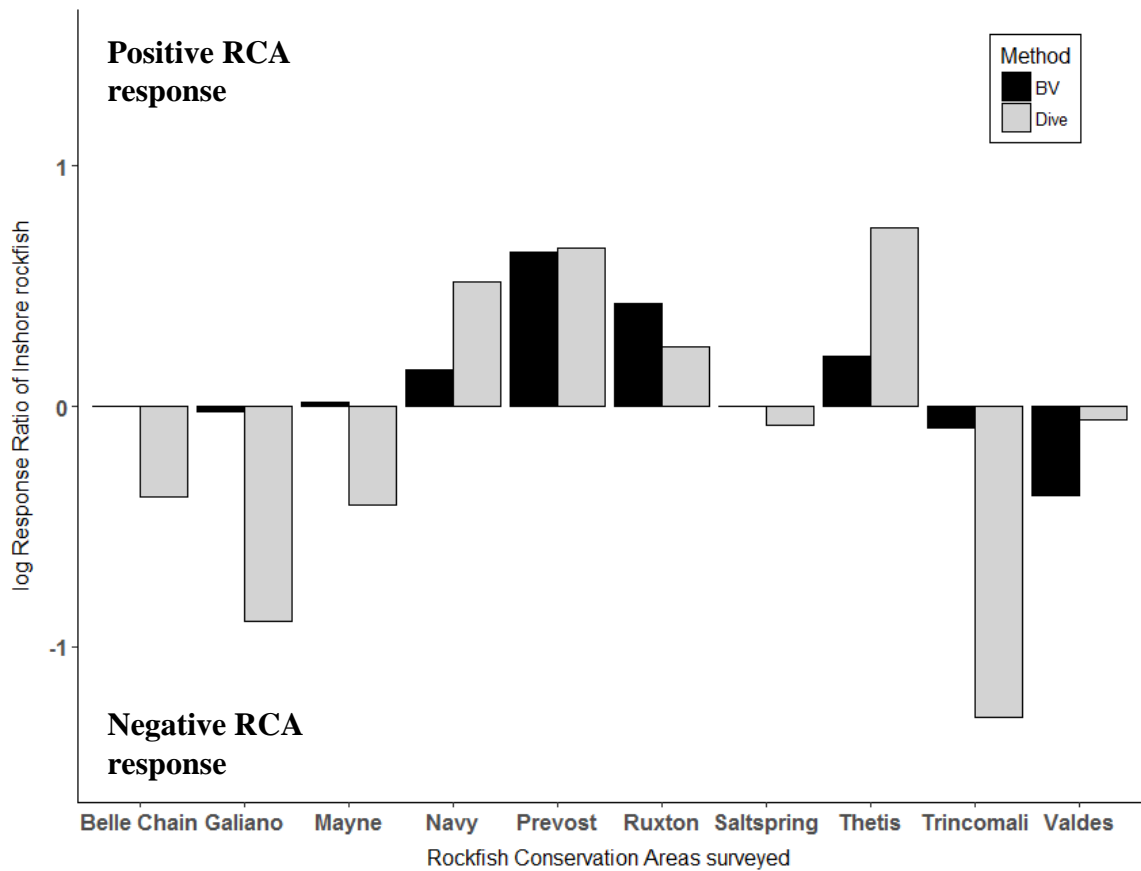
Dive survey mean rockfish abundance was higher outside of RCAs than inside of RCAs, but this difference was not statistically significant (Table 3-3; Figure 3-3 A). The DS detected marginally greater species richness inside RCAs and both the DS and the BV surveys had higher Shannon-Wiener diversity inside RCAs relative to outside, but again, these differences were not significant. A higher DS taxonomic evenness was observed inside of RCAs than outside of RCAs, while BV species richness and taxonomic evenness was identical inside and outside of RCAs.

The average RR for inshore rockfish was greater than zero at 4 of the RCA surveyed, indicating higher densities inside these RCAs than outside. At six of the RCAs surveyed,

the average RR for inshore rockfish was less than zero, indicating lower densities of inshore rockfish inside these RCAs relative to outside (Figure 3-4).



**Figure 3-3 Mean and 95% confidence intervals of A) BV inshore rockfish MeanCount and DS inshore rockfish per 100 m<sup>3</sup>, B) species richness, C) Shannon-Wiener diversity, and D) Pielou Evenness between DS and BV surveys inside and outside of RCAs**



**Figure 3-4 Differences in the log Reserve Ratio of inshore rockfish abundance inside and outside RCAs revealed by dive (DS; grey bars) and baited video (BV; black bars) surveys. Positive values indicate more rockfish observed inside the RCA than outside. Negative values indicate more rockfish observed outside the RCA than inside.**

**Table 3-3 Mean and standard deviation of inshore rockfish abundance and fish diversity metrics inside and outside of Rockfish Conservation Areas observed during the DS and BV surveys along with significance of the Mann Whitney test.**

Estimate	Method	Outside RCA	Inside RCA	Inside RCA compared to Outside RCA
		Mean +/- SD	Mean +/- SD	Mann Whitney (two-sample Wilcoxon test)
Rockfish abundance	DS	1.40 +/- 1.71	0.95 +/- 0.89	W = 337.5, p-value = 0.6452
	BV	0.29 +/- 0.26	0.17 +/- 0.42	W = 420.5, p-value = 0.25
Species richness	DS	4.30 +/- 2.43	4.48 +/- 1.42	W = 391, p-value = 0.65
	BV	2.07 +/- 1.69	2.07 +/- 1.69	W = 366.5, p-value = 0.98
Shannon-Wiener diversity index	DS	0.82 +/- 0.50	1.04 +/- 0.32	W = 461, p-value = 0.097
	BV	0.41 +/- 0.44	0.47 +/- 0.44	W = 384, p-value = 0.73
Pielou evenness index	DS	0.27 +/- 0.21	0.39 +/- 0.21	W = 482, p-value = 0.020
	BV	0.33 +/- 0.38	0.33 +/- 0.33	W = 244, p-value = 0.96

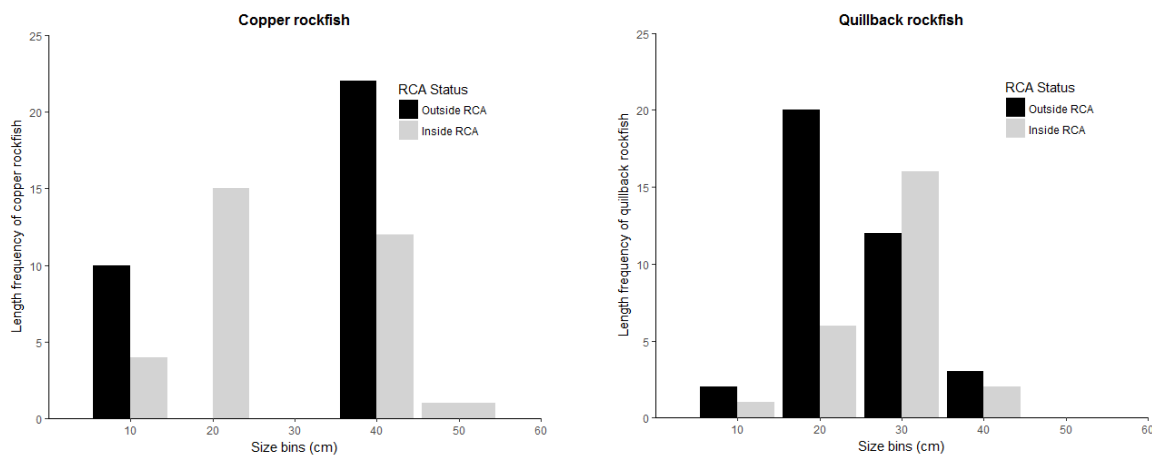
**Table 3-4 Fish detected during DS and BV surveys near the Southern Gulf Islands, British Columbia, Canada. BV numbers are the sum of MaxN per species.**

Phylum Class Order	Species	Common Name	DS		BV	
			In	Out	In	Out
Chordata Actinopterygii Scorpaeniformes	<i>Sebastes</i> spp.	Rockfish	1	2	-	-
	<i>Sebastes auriculatus</i> *	Brown Rockfish	19	6	7	1
	<i>Sebastes caurinus</i> *	Copper Rockfish	89	161	21	12
	<i>Sebastes emphaeus</i>	Puget Sound Rockfish	105	98	-	11
	<i>Sebastes flavidus</i>	Yellowtail Rockfish	7	-	-	-
	<i>Sebastes maliger</i> *	Quillback Rockfish	25	37	4	2
	<i>Sebastes melanops</i> *	Black Rockfish	6	-	-	-
Total rockfish individuals			252	304	32	26
Total rockfish species			7	5	3	4
	<i>Scorpaenichthys marmoratus</i>	Cabezon	1	-	-	-
	<i>Ophiodon elongatus</i>	Lingcod	9	20	10	9
	<i>Hexagrammos decagrammus</i>	Kelp Greenling	67	65	5	7
	<i>Hexagrammos stelleri</i>	Whitespotted Greenling	2	3	1	1
	<i>Oxylebius pictus</i>	Painted Greenling	4	3	-	-
Pleuronectiformes	Unidentified species	Flatfishes	1	1	2	1
Perciformes	<i>Cymatogaster aggregate</i>	Shiner Perch	2485	1260	210	654
	<i>Rhacochilus vacca</i>	Pile Perch	237	349	56	31
	<i>Brachyistius frenatus</i>	Kelp Perch	33	108	8	54
	<i>Embiotoca lateralis</i>	Striped Sea Perch	258	262	28	13
Gasterosteiformes	<i>Aulorhynchus flavidus</i>	Tube-snout	-	-	-	30
Total fish individuals (including rockfish)			3349	2375	352	826
Total fish species (including rockfish)			17	14	11	13

\* Inshore rockfish pooled and used in analyses

### Evaluation of RCA protection on dive survey rockfish size

No significant differences were observed in the length frequency of copper and quillback rockfish inside and outside of RCAs (Two-sample Kolmogorov-Smirnov test: copper rockfish  $D = 0.017$ ,  $p\text{-value} = 1$ ; quillback rockfish  $D = 0.31$ ,  $p\text{-value} = 0.10$ ). At the 10 and 40 cm size bins, more copper and quillback rockfish were found outside of RCAs than inside (Figure 3-5). More 20 cm copper rockfish were observed inside RCAs than outside. In contrast to copper rockfish, more 20 cm quillback rockfish were observed outside RCAs than inside. No copper rockfish were detected in the 30 cm size bin while 30 cm quillback were observed both inside and outside of RCAs. The largest inshore rockfish (at 50 cm) was observed inside the RCAs while no inshore rockfish > 40 cm were found outside of RCAs.



**Figure 3-5 Length frequency histograms of copper rockfish (left) and quillback rockfish (right) inside and outside of RCAs.**

### **Influence of RCA protection on inshore rockfish abundance**

RCA protection status was included in the 4<sup>th</sup> top DS model that carried 10% of the AIC weight of evidence and in the 3<sup>rd</sup> top BV model that carried 12% of the AIC weight of evidence (Table 3-5). There was a positive relationship between DS and BV inshore rockfish and RCA protection. This indicates more inshore rockfish were observed inside RCAs relative to outside on the DS and BV inshore rockfish presence was more likely within an RCA than outside. Though, the AIC weight of evidence for the models containing RCA protection status was low for both DS and BV models.

Predictors relating to topographic heterogeneity were included in the top DS and BV inshore rockfish abundance models, carrying 36% and 13% of the AIC weight of evidence for the DS and BV models, respectively. For both DS and BV inshore rockfish abundance, abundance increased as the proportion of complexity and relief observed at a site increased. Depth was also included in both the DS and BV top model but the relationship varied between methods. More inshore rockfish were detected on the DS as depth increased while less inshore rockfish were observed at deeper sites during the BV surveys.

**Table 3-5 Selection of top generalized linear models of inshore rockfish abundance for A) dive surveys and B) baited video surveys.**

#	Inshore rockfish abundance ~ Model variables	AICc	$\Delta$ AIC	AIC <sub>w</sub>
A) Dive surveys				
12	COMPLEX + RELIEF + DEPTH	307.15	0	0.36
3	COMPLEX + RELIEF	308.15	1	0.22
6	COMPLEX + RELIEF + ROCK + DEPTH	309.61	2.47	0.11
14	COMPLEX + RELIEF + DEPTH + RCA	309.62	2.47	0.10
1	COMPLEX + RELIEF + ROCK	310.02	2.87	0.09
B) Baited video surveys				
4	-DEPTH	68.86	0	0.23
12	COMPLEX + RELIEF – DEPTH	70.02	1.16	0.13
8	-DEPTH + RCA	70.14	1.29	0.12
9	ROCK – DEPTH	71.09	2.23	0.08
3	COMPLEX + RELIEF	71.11	2.25	0.07

### **Influence of RCA protection on species richness**

RCA protection status was included in the 3<sup>rd</sup> DS and BV top species richness models and carried 14% and 15% of the AIC weight of evidence for the DS and BV models, respectively (Table 3-6). However, the relationship between RCA protection status and species richness varied between the survey methods. More species were observed on the DS within RCAs while less species were detected in the 30-minute BV survey inside of RCAs relative to outside.

Complexity and relief were both included in the top DS and BV species richness models. Greater species richness was observed for both methods as complexity and relief increased. Rocky substrate was included the 2<sup>nd</sup> top DS model that carried 17% of the AIC weight of evidence. Less species were detected on the DS as the proportion of rocky substrate observed along the dive transects increased. Depth was also included in the top BV model and increasing depth decreased the number of fish species observed during BV surveys.

**Table 3-6 Model selection of generalized linear models of species richness for A) dive surveys and B) baited video surveys.**

#	Species richness ~ Model variables	AICc	$\Delta$ AIC	AIC <sub>w</sub>
A) Dive surveys				
3	COMPLEX + RELIEF	217.51	0	0.40
1	COMPLEX + RELIEF - ROCK	219.27	1.76	0.17
13	COMPLEX + RELIEF + RCA	219.59	2.08	0.14
12	COMPLEX + RELIEF - DEPTH	219.82	2.31	0.13
7	COMPLEX + RELIEF - ROCK + RCA	221.4	3.89	0.06
6	COMPLEX + RELIEF - ROCK - DEPTH	221.7	4.19	0.05
B) Baited video surveys				
12	COMPLEX + RELIEF - DEPTH	183.9	0	0.48
6	COMPLEX + RELIEF - ROCK - DEPTH	186.06	2.17	0.16
14	COMPLEX + RELIEF - DEPTH - RCA	186.29	2.39	0.15
4	-DEPTH	187.73	3.83	0.07
global	COMPLEX + RELIEF - ROCK - DEPTH - RCA	188.59	4.69	0.05
9	ROCK - DEPTH	188.73	4.84	0.04

### 3.4 Discussion

Baited video data yielded equivalent insight on RCA effectiveness to data derived from the DS. However, the DS produced higher estimates of inshore rockfish abundance and fish diversity metrics than the BV surveys. RCA protection showed no significant effect on rockfish abundance and size, or fish diversity. Even though the DS recorded greater rockfish abundance than BV surveys, the BV surveys detected a representative sample of *Sebastes spp.* and also observed the same pattern in dominant inshore rockfish as the DS. In other DS and BV comparisons used to evaluate conservation area effectiveness, BV detected greater numbers of fish than the DS but, like our results, the differences between DS and BV fish abundance did not affect measured marine conservation area efficacy (Willis, Millar, and Babcock 2000; Gardner and Struthers 2013). The variability in our

results to the DS-BV fish abundance observed by Willis, Millar, and Babcock (2000) and Gardner and Struthers (2013) could be attributed to focal species-specific responses to the methodology used and to differences in how the surveys were executed. Our results show that compared to the DS, BV failed to detect inshore rockfish at many sites sampled; suggesting both BV site-level and mean inshore rockfish abundance estimates are less reliable than the DS results. With so few inshore rockfish detected by the BV, it is difficult to attribute differences to spatial marine conservation area effects, as opposed to chance.

The better performance of the DS to estimate both inshore rockfish abundance and fish diversity relative to BV surveys may stem from the divers being able to observe and record individuals *in situ* whereas identification of individuals with the BV can be influenced by the video sample (*e.g.* how far the fish is away from the camera). Several studies have found divers are better than cameras at observing cryptic species because divers can search complex habitats and census cracks, crevices and lacunae in ways that cameras cannot (Watson et al. 2005; Stobart et al. 2007). Indeed, our DS observed more rockfish species, other demersal fishes, and cryptic benthic species than the BV surveys, but both methods detected similar species of the smaller pelagic schooling fishes (*e.g.* Embiotocidae).

In other dive-video comparisons, diver-based visual methods have obtained greater abundance and species richness estimates than video methods (Watson et al. 2005; Colton and Swearer 2010), due to the advantages of the human eye over video technologies but this observation is not consistent between all diver-video comparisons (Willis and Babcock 2000; Willis, Millar, and Babcock 2000). In areas of high fish

densities, the BV may truncate abundance as there is an upper limit to the number of fish that may be visible in the BV field of view at a given moment. This will produce conservative contrasts between areas with high and low fish density, but may result in failure to detect a difference between two areas where densities are different, but so high as to saturate the field of view in both places. The DS surveys detected higher numbers of pelagic species than the BV surveys indicating the BV field of view may be more constrained than the DS area.

Fish detection with the BV is influenced by bait plume dynamics, local hydrology, and species-specific response characteristics to the bait (Armstrong, Bagley, and Priede 1992; Cappo, Speare, and De'Ath 2004; Harvey et al. 2007; Colton and Swearer 2010; Terres et al. 2015). Baited video may introduce biases from the behaviours adopted by fish in response to the bait and the dynamics of attraction or avoidance are largely unknown (Sainte-Marie and Hargrave 1987; Armstrong, Bagley, and Priede 1992). Seasonal, reproductive and lunar patterns of activity in swimming speed, fish behaviour, and fish appetite likely affect fish attraction to bait, as will conspecific attraction, curiosity, predator presence or absence and home range sizes (Munro 1974; Newman and Williams 1995). Strongly site-attached and territorial species, like some rockfish (*e.g. Sebastes nigrocinctus* and *Sebastes ruberrimus*), may not have been detected if the BV settled outside of their territory.

Reflecting the motile nature of the DS, the DS may have surveyed a larger area and more habitat types than the BV surveys, which could explain the greater DS rockfish abundance and fish diversity estimates (Lomolino 2000; Gratwicke and Speight 2005). DS covered 540 m<sup>3</sup> per site. For BV deployments if the radius of bait detection was

approximately 10 m like that estimated in Willis, Millar, and Babcock (2000), the effective sampling area was only 314 m<sup>2</sup>. BV only allows for relative fish abundance comparisons due to the unknown size of the bait plume and thus, area surveyed. Bait plume area is influenced by prevailing rates of advection and turbulent diffusion thus BV sampling area may vary widely in comparison to the relatively static DS area. Variation in dispersal of the bait plume is likely to contribute to within-area variance. Modeling the radius of attraction of the bait plume (Merritt et al. 2011; Heagney et al. 2007) is needed to estimate BV sample area, and combined with the knowledge of fish reactions to bait and their swimming speeds will allow for more direct density estimates and between-site and methodology density comparisons.

Due to the stationary nature of the BV surveys, the low abundance of rockfish on the South Coast may have negatively biased BV rockfish detection. The south coast of BC is characterized by developed, urban areas and fish populations here were historically subject to heavy fishing pressure reducing observed fish abundance and diversity. Many fish populations show density-dependent habitat selection with ranges contracting as abundance declines (Marshall and Frank 1995). At low fish abundance, fish may occur only in small areas that have optimal habitat. If the BV was deployed at a location with few rockfish, the bait used may not have been an effective enough attractant to lure the fish from their rocky crevice. Therefore, micro-site state (and attendant variation/environmental stochasticity) may introduce stochasticity to BV data, which is reflective of more zeros observed in the BV data.

Seabed terrain characteristics were included in the top BV inshore rockfish abundance and species richness models even though BV sacrifices the fine-grained spatial

heterogeneity data, a consequence of the stationary nature of the video method.

Complexity and relief were positive predictors of inshore rockfish abundance and species richness for both the DS and BV surveys. Inshore rockfish are found over structurally complex rocky substrates (Love et al. 1990; Love, Yoklavich, and Thorsteinson 2002; Haggarty, Lotterhos, and Shurin 2017) and the positive relationship between diversity and topographic heterogeneity is an ecological pattern found in both terrestrial and aquatic ecosystems (Beatty 1984; Cusson and Bourget 1997; Levin et al. 2001; Harman, Harvey, and Kendrick 2003; Willis and Anderson 2003; Shumway, Hofmann, and Dobberfuhl 2007). Topographic heterogeneity is thought to increase overall diversity by increasing the number of available niches, allowing for more ways to utilize environmental resources while decreasing species exclusion by reducing interspecific competition, and reducing mortality by providing refuge space from predators (Levin and Paine 1974; Huston 1979; Johnson 2007). The BV captured the known relationship between inshore rockfish and seabed terrain characteristics as well as the positive association between diversity and topographic heterogeneity, indicating a BV survey may be used to capture fish-habitat associations.

Rockfish partition habitats by depth (Richards 1986; Richards 1987; Matthews 1990; Yoklavich et al. 2000; Yoklavich, Love, and Forney 2007; Laidig, Watters, and Yoklavich 2009; Ingram and Shurin 2009; Haggarty, Shurin, and Yamanaka 2016; Anderson and Yoklavich 2007) and fish diversity and abundance is often distributed along a depth gradient with many species only occurring within certain depth ranges (Bacheler et al. 2016; Jankowski, Graham, and Jones 2015; Galaiduk et al. 2017). Both the DS and BV surveys were limited in that they only assessed inshore rockfish species

within shallow depth ranges that typically include juvenile quillback and copper rockfishes. On the DS, more inshore rockfish were observed as depth increased while inshore rockfish were less likely to be present on the BV surveys at deeper sites. The variation between methods in the relationship between inshore rockfish abundance and depth may be a result of the BV luring fish from adjacent habitat and introducing variation into observed species-environment relationships (Moore, Harvey, and Van Niel 2009; Galaiduk et al. 2017).

### **Monitoring Rockfish Conservation Area Effectiveness**

I found no significant indication inshore rockfish recovery is influenced by RCA protection nor did I find evidence of increasing species richness inside of RCAs. After 12 to 15 years of RCA protection, when data were combined across all sites sampled, Rockfish Conservation Areas did not produce more inshore rockfish, bigger rockfish, or greater fish diversity than paired sites outside of Rockfish Conservation Areas, whether measured using a DS or a BV survey. These results align with previous research evaluating RCA effectiveness along the BC coast. No significant increase in rockfish density has been observed as a result of RCA protection (Marliave and Challenger 2009; Cloutier 2011; Haggarty, Shurin, and Yamanaka 2016), nor have differences in the size structure of rockfishes inside RCAs and unprotected areas been found (Haggarty, Shurin, and Yamanaka 2016). Although, (Cloutier 2011) found rockfish densities were 1.2-2.3 times higher in 15 RCAs than nearby areas with no protection, this difference was not significant.

There are a number of reasons that may explain the lack of RCA effectiveness. Additional rockfish management measures were put in place at the time of RCA establishment (Yamanaka and Logan 2010). In addition to RCA implementation, Fisheries and Oceans Canada reduced the commercial Total Allowable Catch for inshore rockfish and decreased the recreational daily catch limits. Rockfish and all other commercial groundfish fisheries were integrated under the Commercial Groundfish Initiative and all rockfish catch, including targeted and by-catch, were accounted for with increased monitoring measures (Davis 2008). The equivalence I observed between protected and unprotected sites may be from increases in inshore rockfish both inside and outside of the RCAs as a result of these additional management measures.

Sufficient time may not have elapsed for inshore rockfish growth and recruitment to replenish depleted stocks inside of RCAs. Rockfish response to protection is expected to be slow because the same life-history traits that make them vulnerable to overfishing, such as longer generation times and sporadic recruitment success, means it may take longer to observe effects of spatial marine conservation measures (Karpov, Bergen, and Geibel 2012; Starr et al. 2015). Lag times for recovery after implementation of conservation action are observed in both terrestrial and aquatic ecosystems. For example, full recovery of a river after weir removal is suggested to take as long as 80 years (Bednarek 2001), while development of key habitat resources (*e.g.* tree hollows and fallen timber) after revegetation requires at least 100 years to develop (Vesk et al. 2008). Our surveys were conducted 12 – 15 years after RCA establishment and positive reserve effects have been found for some species of Californian rockfish after 5 years (Hamilton et al. 2010), but these rockfish are relatively short lived and mature early (Love,

Yoklavich, and Thorsteinson 2002). Rockfish with older lifespans have not shown a positive reserve effect (Keller et al. 2014) and Starr et al. (2015) predicted that it could take 20 years or more to detect a significant response of rockfish as a result of spatial marine conservation measures.

Certain recreational and commercial fishing activities permitted within RCAs, such as commercial and recreational prawn fishing by trap or mid-water trawling, may be indirectly leading to inshore rockfish catch. In addition, recreational non-compliance has been hypothesized as a factor that may be impeding inshore rockfish recovery inside of RCAs (Marliave and Challenger 2009; Haggarty, Martell, and Shurin 2016), as compliance with regulations influences effectiveness of marine conservation areas. Haggarty, Martell, and Shurin (2016) found the recreational fishing effort within many RCAs has not changed since implementation. Using terrestrial camera traps at RCAs within the Southern Gulf Islands, Lancaster et al. (2017) showed similar rates to Haggarty (2016) of non-compliance in RCAs, while a survey of recreational fishers determined fisher knowledge of RCA regulations was low, with many fishers unsure of RCA locations. If illegal fishing, and fishing permitted within RCA boundaries are imposing high inshore rockfish mortality, RCA protection may be inadequate to allow rockfish recovery.

RCA effectiveness is likely influenced by a combination of factors. Haggarty (2015) developed a Conservation Score based on research by Edgar et al. (2014) and Mora et al. (2006) that highlighted the influence of cumulative effects on MPA effectiveness. Haggarty combined habitat metrics (*i.e.* seabed terrain characteristics) along with estimates of recreational compliance, RCA size, commercial compliance, inshore

rockfish by-catch in permitted fisheries, and connectivity into a single Conservation Score. This score was used to evaluate RCA effectiveness and designed to be an adaptive spatial management tool to improve RCAs. RCAs with higher Conservation Scores were likely to be more effective. For both DS and BV surveys, I observed a positive effect of RCA protection for RCAs in our study that were rated as having a higher Conservation Score (*e.g.* Ruxton-Pylades, Prevost), while more rockfish were found outside of RCAs that were rated with the lowest Conservation Score (*e.g.* Trincomali, Galiano). This suggests certain RCAs may be effective conservation areas for inshore rockfish recovery. Although, the inshore rockfish abundance – Conservation Score comparisons were inconsistent as I observed more inshore rockfish inside than outside the Thetis-Kuper RCA, which received a low Conservation Score (17) and more inshore rockfish outside than inside the Mayne Island North RCA that had a Conservation Score of 19 out of 24.

### **3.5 Conclusion**

BV data yielded equivalent insight on RCA effectiveness to data derived from the DS indicating BV could be used as a low cost and low effort method for monitoring marine conservation area effectiveness. The BV surveys were logistically simpler to organize, required fewer skilled field personal, and fewer hours in the field as compared to conducting a DS. BV surveys can be conducted rapidly, generating more data over greater temporal and spatial scales than a DS. I surveyed only a small fraction of inshore rockfish depth ranges, as inshore rockfish can be found in depths more than 200 m (Love, Yoklavich, and Thorsteinson 2002). Even though the BV surveys I conducted were within the DS depth range, a BV survey can access deeper depths and provides a means of surveying fish at depths inaccessible to SCUBA divers. Therefore, BV can access a

greater range of rockfish habitat, in addition to avoiding the health and safety concerns associated with a DS.

Comparison between DS and BV rockfish abundance and fish diversity estimates demonstrates that the method chosen to collect data can influence estimates of abundance and diversity. Many of the BV survey limitations I discussed are relatively easy to address. Increasing the number of BV samples and/or BV survey duration may improve BV inshore rockfish abundance and fish diversity estimates and congruence between DS and BV surveys. Although I was unable to determine fish size from the simple BV platform I developed, the BV rig could be improved through the addition of stereo-video techniques (*i.e.* two video cameras) and software used to obtain accurate length estimates of fish (Harvey and Shortis 1998).

It is critical to evaluate RCAs and assess if RCA protection is contributing to rebuilding inshore rockfish populations. Evaluation of spatial marine conservation areas is necessary for management plans, and needed to implement adaptive management of protected areas and/or species. Monitoring of spatial marine conservations areas is essential as it prevents ineffective areas from creating a false sense of security and detracting from other forms of fisheries management.

## Chapter 4 Conclusion

### 4.1 Thesis overview

This study compared multiple non-destructive visual methods used to survey fish in marine environments to meet two goals: 1) contribute to knowledge on the methods used to assess inshore rockfish populations, and 2) assess the ecological effectiveness of RCAs with multiple methods.

### 4.2 Underwater visual method comparison

I compared inshore rockfish abundance and fish diversity estimates generated by paired active and passive surveys to test whether the passive BV survey data yielded similar inshore rockfish abundance and fish diversity metrics to data derived from active TV and DS. I found similar inshore rockfish abundance estimates between TV and BV surveys, but the BV surveys detected higher fish diversity. In the dive and baited video comparison, the dive surveys detected greater inshore rockfish abundance and fish diversity than the baited video surveys. BV detected a wider range of species than the TV such as pelagic fishes and mobile predators, but the DS observed more rockfish species, other demersal fishes, and cryptic benthic species than the BV. The BV, along with the DS, captured the known relationship between inshore rockfish abundance and seabed relief and complexity as well as the widely accepted positive relationship between species diversity and topographic heterogeneity that the TV surveys did not. Our results indicate BV is a low cost and effort methodology that can be used for examining inshore rockfish abundance and fish diversity over rocky reefs from nearshore subtidal waters down to depths greater than 20 m. Congruence between the TV-BV results suggests BV could be

used in place of TV surveys. Due to the differences I observed between DS-BV inshore rockfish abundance and fish diversity, BV cannot be used as a substitute for DS but instead could supplement DS by surveying deeper waters. Further, an increase in BV survey duration or sample effort may improve BV performance and congruence between DS-BV results.

### **4.3 Comparison of visual methods used to assess RCA protection**

To assess the ecological effectiveness of RCAs with multiple visual methods, I compared inshore rockfish abundance and fish diversity estimates between paired DS and BV surveys. I conducted surveys on the south coast of BC inside and outside of ten RCAs to address both a methodological and a conservation question. One aim was to determine whether the BV data generated the same conclusions on the effectiveness of spatial marine conservation measures as data derived from the DS. A secondary aim was to assess RCA effectiveness as measured by increased inshore rockfish abundance and size, and fish diversity inside RCAs when compared to outside of RCAs. BV data yielded equivalent insight on RCA effectiveness to data derived from the DS. However, the DS produced higher estimates of inshore rockfish abundance and fish diversity metrics than the BV surveys. The BV surveys did detect a representative sample of *Sebastes spp.* and also observed the same pattern in dominant inshore rockfish as the DS. I found little indication inshore rockfish recovery is influenced by RCA protection. When data were combined across all sites sampled, RCAs did not produce more inshore rockfish, bigger quillback or copper rockfish, or greater fish diversity than similar areas outside of RCAs, whether measured using a DS or a BV survey. However, I did observe a positive effect of RCA protection for some of the individual RCAs surveyed that rated as having a higher

Conservation Score (as developed by Haggarty 2015), while more rockfish were found outside of RCAs rated with the lowest Conservation Score. This indicates certain RCAs may be effective conservation areas for inshore rockfish recovery.

The equivalence of RCA effectiveness as measured by DS and BV surveys suggests the BV could be used as a rapid, non-destructive assessment tool for monitoring the effectiveness of spatial marine conservation areas. Monitoring of spatial marine conservation measures is critical as it evaluates management actions and determines which objectives are being met. Results from monitoring can be used to implement adaptive management of marine conservation areas (*i.e.* promoting compliance regulations through education). Inshore rockfish response to protection is expected to be slow because the same life-history traits that make them vulnerable to overfishing such as longer generation times and sporadic recruitment success, means it may take longer to observe effects related to spatial marine conservation measures. This lag in recovery means long-term monitoring is critical to separate a conservation area response from the natural spatial and temporal variability observed in marine environments. In addition, results from long-term monitoring can feed into adaptive management strategies to improve management measures. Implementation of long-term monitoring programs is often restricted by the funds needed to conduct surveys. BV can address this limitation as it is a low cost and low effort non-destructive methodology that can quantify the response of inshore rockfish to spatial marine conservation measures.

#### **4.4 Recommendations**

The differences in inshore rockfish abundance and fish diversity estimates between the paired surveys reveals the methodology used to sample the focal population matters. It is important to test method performance to understand how the methodology used may influence sample measurements, especially when assessing management actions.

Accounting for the detection probability of survey species in the underwater methodology used will allow for more accurate estimates of population abundance. Methods have been developed to account for DS detection probability, and include distance sampling (Kulbicki et al. 2010), mark-recapture (MacNeil, Tyler et al. 2008), and the double-observer techniques (Bernard et al. 2013). Little research accounting for detection probability with TV and BV surveys has taken place and could be developed. Additional comparisons between methodologies such as evaluating how an increase in the number of BV samples and/or increasing the survey duration may improve congruence between DS and BV surveys to identify if BV could be used in place of DS, not only to supplement DS data.

I observed inconsistent associations between BV rockfish abundance and the seabed terrain characteristics. In addition, some of the associations I found were different than the expected and previously observed relationship between rockfish abundance and seabed terrain characteristics. The differences in the strength and direction of habitat predictors in BV models may be a result of the BV luring fish from adjacent habitat and introducing variation into observed species-environment relationships (Moore, Harvey, and Van Niel 2009; Galaiduk et al. 2017). To capture a more accurate representation of rockfish-habitat associations, more BV samples could be deployed for shorter survey

duration. Additional research examining the impact of different BV deployments could be initiated to look into these questions.

Long-term monitoring of RCAs is essential to evaluate their effectiveness. The BV survey has potential to be a low cost and effort methodology to monitor RCA effectiveness, but survey limitations such as the inability to calculate the realized sample volume and measurement of fish size would need to be addressed. A current velocity probe could be used with the BV to determine bait plume size and the BV platform could be improved through the addition of stereo-video techniques and software to obtain accurate length estimates of fish. The positive rockfish response observed for some of the RCAs surveyed indicates certain RCAs may be effective conservation areas for rockfish recovery. However, I did observe inconsistencies in the rockfish abundance – Conservation Score (as developed by Haggarty 2015) comparisons. Continued monitoring of RCAs with low and high Conservation Scores would address these inconsistencies. In addition, our results suggest management actions should target RCAs with low Conservation Scores to improve their effectiveness.

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## Appendices

### Appendix A Comparison of towed video, dive, and baited video methods for surveying different environmental conditions, species characteristics, survey logistics and survey metrics.

Parameter			Method rating	Condition
Environmental characteristics	Depth	< 20 m	TV, DS or BV	I deployed the BV easily down to 60 m. For deeper deployments, the low-cost BV platform I developed would need to be evaluated.
		> 20 m	TV or BV	
	Substrate	Rock	TV, DS or BV	Depends upon relief and complexity of survey area and survey species characteristics.
		No rock	TV, DS or BV	
	Relief	Low	TV, DS or BV	DS may outperform BV in high relief but the BV platform could be designed to ensure it rested upright over high relief areas.
		High	DS or BV	
	Complexity	Low	TV, DS or BV	Depends upon species characteristics and survey objective as BV may underperform DS in complex areas if surveying cryptic and/or territorial species.
		High	DS or BV	
	Current	Low	TV, DS or BV	Additional weight may need to be added to BV platform. Use of BV avoids health and safety concerns of a DS in high current.
		High	BV > DS	
	Visibility	Poor	DS > BV	Depends on species characteristics as BV lures fish to the camera. If survey species are attracted to the BV, the BV could perform equally well to the DS in poor visibility.
		Good	TV, DS or BV	
	Sea state	Poor	BV > TV, DS	Our BV platform was still challenging to deploy and retrieve when seas were > 1 m. Sea state influences TV performance as boat movement translates to the TV cable making it difficult to maintain TV altitude. Use of BV or TV avoids health and safety concerns of a DS in poor sea conditions.
		Good	TV, DS or BV	

Parameter		Method rating	Condition	
Species characteristics	Size	< 10 cm	DS	DS better estimates small fishes because DS observe fish <i>in situ</i> and can search complex habitats in ways cameras cannot.
		> 10 cm	TV, DS or BV	
	Abundance	Low	DS > BV	To survey populations of low abundance with the BV, I recommend increasing the number of samples and/or survey duration.
		High	TV, DS, BV	
	Benthic		TV, DS > BV	BV observed benthic species ( <i>e.g.</i> flatfish) but the smaller, cryptic benthic species were difficult to identify to species with the BV. TV did not detect small, pelagic species. During our surveys the field of view of the BV did not seem to truncate pelagic species abundance as expected.
	Demersal		TV, DS or BV	
	Pelagic		DS or BV	
	Cryptic		DS > BV > TV	DS better estimates cryptic fishes because DS observe fish <i>in situ</i> and can search for fishes in ways cameras cannot.
	Schooling species		DS > BV > TV	BV captured schooling species but abundance may be truncated for very large schools because of BV field of view.
	Territorial species		TV, DS > BV	BV detection of territorial species depends on BV being deployed in territory of species. May require increasing the number of samples if surveying territorial species with a BV.
Mobile predators/scavengers		BV > TV, DS	BV better surveyed mobile predators and scavengers than TV and DS surveys.	

Parameter		Method rating	Condition	
Logistics	Survey duration	TV, DS or BV	Dependent upon survey objectives. All methods I used could be deployed for varying survey durations. The minimum BV duration I used was 30-minutes.	
	Difficulty of survey	Low	BV	The BV survey was easy to deploy and retrieve. TV and the equipment associated with the TV survey was challenging to execute reliability and consistently. DS surveys were easy to conduct but depend more upon environmental conditions and were physically taxing
		High	TV > DS > BV	
	Observer training and/or experience	Low	BV > TV, DS	Little training and/or experience needed to conduct a BV survey.
		High	TV, DS or BV	
	Equipment cost	Low	BV	BV costs dependent upon characteristics of survey environment. The BV platform was < \$200 and the camera equipment (camera, lights, camera stand and accessories) was ~\$1500. The BV platform deployed successfully in rocky reef environments with varying relief and complexity.
		High	TV > DS > BV	
	Total project costs	Low	BV	A BV rig and survey can be conducted at a very low cost.
		High	TV, DS or BV	
	Vessel size	Small	BV > DS	The BV could be deployed from a small vessel (14-20 feet in length). DS can also take place from a small vessel but require additional vessel specifications that a BV does not. The TV system I used would be difficult to deploy from a small vessel as I relied heavily on the vessel power to conduct the TV survey.
		Large	TV, DS or BV	
	Data availability after survey	Immediate	DS	Both TV and BV require post-survey video annotation. Depending upon BV metric used (MeanCount or MaxN), the time required for annotation can be short. A benefit of using a video survey is that there is a permanent record of the survey that can be used for future research.
		Processing required	TV, BV	
		High	TV, DS or BV	

Parameter		Method rating	Condition
Survey metrics	Survey area estimate	DS > TV > BV	It is challenging to estimate the area surveyed by the BV. It can be done, but may require estimates of current flow at the survey site and a stereo camera set-up to estimate area in BV field of view.
	Species densities estimates	DS > TV > BV	Most precise for DS because survey area not affected by the changing field of view of the TV. If BV survey area not estimated, BV only provides a measure of relative abundance.
	Relative abundances	TV, DS or BV	
	Estimating detection probability	DS > TV, BV	Methods developed for DS that can estimate detection probability. Still needs to be done for TV and BV.
	Community estimates	DS > BV > TV	TV underestimated small, pelagic species and both TV and BV underestimated cryptic species.
	Species - habitat association	DS > TV, BV	TV and DS move across a study area and record individuals over habitat. BV detection relies on luring fish into the camera field of view, which may obscure species-habitat associations.
	Fish length /biomass estimates	DS > TV, BV	I was not able to estimate fish size with the BV platform I developed, but using a stereo video set-up would address this limitation.

**Appendix B Supplementary information from Chapter 2**

**Table B-1 Regression model output for the paired surveys inshore rockfish abundance and species richness analyses.**

<b>Towed video inshore rockfish abundance regression model output</b>										
	<b>Intercept</b>	<b>Complexity</b>	<b>Relief</b>	<b>Depth</b>	<b>Rocky substrate</b>	<b>df</b>	<b>logLik</b>	<b>AICc</b>	<b>delta</b>	<b>AIC weight</b>
3	1.14	-0.3	-0.0022	-	-	4	-27.49	64.41	0	0.25
4	1.08	-	-	0.19	-	3	-28.8	64.42	0.01	0.25
1	1.19	-0.59	0.02	-	0.36	5	-26.47	65.16	0.75	0.17
2	1.06	-	-	-	-0.07	3	-29.46	65.74	1.33	0.13
6	1.15	-0.29	0.03	0.12	-	5	-27.16	66.54	2.13	0.09
5	1.08	-	-	0.18	-0.04	4	-28.76	66.95	2.54	0.07
global	1.2	-0.59	0.05	0.12	0.37	6	-26.08	67.39	2.98	0.06
<b>Towed video species richness regression model output</b>										
3	1.75	-0.34	0.03	-	-	4	-1.86	13.1	0	0.39
2	1.7	-	-	-	-0.14	3	-4	14.8	1.7	0.17
1	1.76	-0.47	0.04	-	0.18	5	-1.54	15.22	2.12	0.14
6	1.75	-0.35	0.01	-0.07	-	5	-1.72	15.57	2.47	0.11
4	1.69	-	-	0.00028	-	3	-4.44	15.68	2.58	0.11
5	1.7	-	-	-0.03	-0.15	4	-3.98	17.35	4.25	0.05
global	1.76	-0.48	0.03	-0.07	0.17	6	-1.41	17.93	4.83	0.04

<b>Baited video inshore rockfish abundance regression model output (paired with towed video surveys)</b>										
	<b>Intercept</b>	<b>Complexity</b>	<b>Relief</b>	<b>Depth</b>	<b>Rocky substrate</b>	<b>df</b>	<b>logLik</b>	<b>AICc</b>	<b>delta</b>	<b>AIC weight</b>
3	-1.61	2.05	-0.12	-	-	4	-32.79	75.4	0	0.55
6	-1.33	2.11	-0.34	-0.26	-	5	-32.42	77.7	2.29	0.17
1	-1.58	2.09	-0.11	-	-0.08	5	-32.79	78.43	3.03	0.12
2	-0.85	-	-	-	1.03	3	-36.25	79.55	4.15	0.07
global	-1.12	2.35	-0.35	-0.32	-0.43	6	-32.31	80.82	5.42	0.04
4	0.25	-	-	-0.27	-	3	-37.07	81.18	5.78	0.03
5	-0.66	-	-	-0.13	0.95	4	-36.18	82.18	6.78	0.02
<b>Baited video species richness regression model output (paired with towed video surveys)</b>										
2	1.10	-	-	-	0.30	2	-54.36	113.19	0.00	0.38
3	1.10	0.31	0.06	-	-	3	-53.81	114.58	1.39	0.19
4	1.28	-	-	0.05	-	2	-55.18	114.83	1.64	0.17
5	1.01	-	-	0.07	0.32	3	-54.19	115.35	2.16	0.13
6	0.99	0.28	0.17	0.10	-	4	-53.56	116.79	3.60	0.06
1	1.06	0.26	0.06	-	0.10	4	-53.76	117.18	3.99	0.05
global	0.93	0.21	0.16	0.11	0.13	5	-53.47	119.54	6.35	0.02

<b>Dive survey inshore rockfish abundance regression output</b>										
<b>Model #</b>	<b>Intercept</b>	<b>Complexity</b>	<b>Relief</b>	<b>Depth</b>	<b>Rocky substrate</b>	<b>df</b>	<b>logLik</b>	<b>AICc</b>	<b>delta</b>	<b>AIC weight</b>
6	1.66	0.42	0.2	0.22	-	5	-147.95	307.15	0	0.47
global	1.69	0.41	0.19	-	-	4	-149.67	308.15	1	0.28
3	1.66	0.41	0.2	0.21	0.04	6	-147.91	309.61	2.47	0.14
1	1.68	0.38	0.2	-	0.12	5	-149.38	310.02	2.87	0.11
4	1.81	-	-	-	0.32	3	-156.3	319.08	11.93	0.00
5	1.82	-	-	0.23	-	3	-156.5	319.48	12.33	0.00
2	1.8	-	-	0.17	0.25	4	-155.63	320.07	12.92	0.00
<b>Dive survey species richness regression output</b>										
3	1.45	0.17	0.1	-	-	3	-105.51	217.51	0	0.54
1	1.45	0.19	0.1	-	-0.06	4	-105.23	219.27	1.76	0.22
6	1.45	0.17	0.1	-0.01	-	4	-105.5	219.82	2.31	0.17
2	1.45	0.19	0.1	0	-0.06	5	-105.23	221.7	4.19	0.07
4	1.48	-	-	-	0.06	2	-112.9	230.04	12.53	0.00
5	1.48	-	-	0.03	-	2	-113.14	230.52	13.01	0.00
global	1.48	-	-	0.02	0.05	3	-112.84	232.16	14.65	0.00

<b>Baited video inshore rockfish abundance regression output (paired with the dive surveys)</b>										
<b>Model #</b>	<b>Intercept</b>	<b>Complexity</b>	<b>Relief</b>	<b>Depth</b>	<b>Rocky substrate</b>	<b>df</b>	<b>logLik</b>	<b>AICc</b>	<b>delta</b>	<b>AIC weight</b>
5	-0.78	-	-	-0.64	-	2	-32.31	68.86	0	0.37
9	-1.14	0.38	1.20	-0.62	-	4	-30.6	70.02	1.16	0.21
14	-0.84	-	-	-0.62	0.08	3	-32.3	71.09	2.23	0.12
12	-1.1	0.59	1.10	-	-	3	-32.31	71.11	2.25	0.12
10	-0.81	0.72	1.31	-0.74	-0.74	5	-30.24	71.73	2.88	0.09
7	-1.03	-	-	-	0.50	2	-34.04	72.32	3.46	0.06
13	-1.03	0.67	1.12	-	-0.15	4	-32.29	73.41	4.55	0.04
<b>Baited video species richness regression output (paired with the dive surveys)</b>										
6	0.49	0.53	0.02	-0.36	-	4	-87.54	183.9	0	0.63
global	0.55	0.59	0.03	-0.37	-0.14	5	-87.41	186.06	2.17	0.21
4	0.71	-	-	-0.38	-	2	-91.75	187.73	3.83	0.09
5	0.55	-	-	-0.35	0.24	3	-91.13	188.73	4.84	0.06
3	0.52	0.65	-0.01	-	-	3	-93.68	193.83	9.93	0.00
1	0.46	0.58	-0.02	-	0.14	4	-93.53	195.88	11.99	0.00
2	0.46	-		-	0.49	2	-96.93	198.1	14.21	0.00

**Appendix C Supplementary information from Chapter 3**

**Table C-1 Regression model output for the paired surveys inshore rockfish abundance and species richness analyses**

**Chapter 3 - Dive survey inshore rockfish abundance regression output, RCA protection**

Model #	Intercept	RCA Status	Complexity	Relief	Rocky substrate	Depth	df	logLik	AICc	delta	AIC weight
12	1.66	-	0.42	0.2	-	0.22	5	-147.95	307.15	0	0.36
3	1.69	-	0.41	0.19	-	-	4	-149.67	308.15	1	0.22
6	1.66	-	0.41	0.2	0.04	0.21	6	-147.91	309.61	2.47	0.11
14	1.7	-0.07	0.4	0.2	-	0.23	6	-147.92	309.62	2.47	0.10
1	1.68	-	0.38	0.2	0.12	-	5	-149.38	310.02	2.87	0.09
13	1.68	0.02	0.42	0.19	-	-	5	-149.66	310.58	3.43	0.07
global	1.7	-0.08	0.39	0.2	0.05	0.22	7	-147.87	312.18	5.03	0.03
7	1.69	-0.01	0.37	0.2	0.12	-	6	-149.38	312.55	5.41	0.02
2	1.81	-	-	-	0.32	-	3	-156.3	319.08	11.93	0.00
4	1.82	-	-	-	-	0.23	3	-156.5	319.48	12.33	0.00
8	2.02	-0.45	-	-	-	0.25	4	-155.38	319.57	12.42	0.00
10	1.99	-0.38	-	-	0.32	-	4	-155.48	319.77	12.62	0.00
9	1.8	-	-	-	0.25	0.17	4	-155.63	320.07	12.92	0.00
11	1.99	-0.44	-	-	0.24	0.2	5	-154.53	320.31	13.17	0.00
5	2.02	-0.38	-	-	-	-	3	-156.99	320.46	13.31	0.00

**Chapter 3 - Dive survey species richness regression output, RCA protection**

<b>Model #</b>	<b>Intercept</b>	<b>RCA Status</b>	<b>Complexity</b>	<b>Relief</b>	<b>Rocky substrate</b>	<b>Depth</b>	<b>df</b>	<b>logLik</b>	<b>AICc</b>	<b>delta</b>	<b>AIC weight</b>
3	1.45	1.45	-	0.17	0.1		3	-105.51	217.51	0	0.40
1	1.45	1.45	-	0.19	0.1	-0.06	4	-105.23	219.27	1.76	0.17
13	1.46	1.42	0.07	0.18	0.09	-	4	-105.39	219.59	2.08	0.14
12	1.45	1.45	-	0.17	0.1	-	4	-105.5	219.82	2.31	0.13
7	1.45	1.41	0.07	0.2	0.1	-0.06	5	-105.08	221.4	3.89	0.06
6	1.4	1.45	-	0.19	0.1	-0.06	5	-105.23	221.7	4.19	0.05
14	1.45	1.42	0.07	0.18	0.09	-	5	-105.37	222	4.49	0.04
global	1.42	1.41	0.07	0.2	0.1	-0.06	6	-105.07	223.93	6.43	0.02
2	1.45	1.48	-	-	-	0.06	2	-112.9	230.04	12.53	0.00
4	1.46	1.48	-	-	-	-	2	-113.14	230.52	13.01	0.00
5	1.46	1.47	0.03		-	-	2	-113.23	230.7	13.2	0.00
9	1.39	1.48	-	-	-	0.05	3	-112.84	232.16	14.65	0.00
10	1.45	1.47	0.03	-	-	0.06	3	-112.87	232.23	14.72	0.00
8	1.39	1.47	0.03	-	-	-	3	-113.11	232.71	15.2	0.00
11	1.41	1.47	0.03	-	-	0.05	4	-112.81	234.44	16.93	0.00

**Chapter 3 – Baited video inshore rockfish abundance regression output, RCA protection**

<b>Model #</b>	<b>Intercept</b>	<b>RCA Status</b>	<b>Complexity</b>	<b>Relief</b>	<b>Substrate</b>	<b>Depth</b>	<b>df</b>	<b>logLik</b>	<b>AICc</b>	<b>delta</b>	<b>AIC weight</b>
4	-0.78	-		-	-	-0.64	2	-32.31	68.86	0	0.23
12	-1.14	-	0.38	1.20	-	-0.62	4	-30.6	70.02	1.16	0.13
8	-1.09	0.59	-	-	-	-0.61	3	-31.83	70.14	1.29	0.12
9	-0.84	-	-	-	0.08	-0.62	3	-32.3	71.09	2.23	0.08
3	-1.1	-	0.59	1.10	-	-	3	-32.31	71.11	2.25	0.07
5	-1.05	0.68	-	-	-	-	2	-33.7	71.64	2.78	0.06
6	-0.81	-	0.72	1.31	-0.74	-0.74	5	-30.24	71.73	2.88	0.05
14	-1.41	0.53	0.41	1.12	-	-0.59	5	-30.25	71.75	2.9	0.05
2	-1.03	-	-	-	0.50	-	2	-34.04	72.32	3.46	0.04
13	-1.41	0.61	0.60	1.03	-	-	4	-31.81	72.43	3.58	0.04
11	-1.08	0.60	-	-	-0.01	-0.61	4	-31.83	72.48	3.62	0.04
global	-1.07	0.63	0.81	1.23	-0.88	-0.71	6	-29.77	73.33	4.47	0.02
1	-1.03	-	0.67	1.12	-0.15	-	4	-32.29	73.41	4.55	0.02
10	-1.27	0.61	-	-	0.39	-	3	-33.52	73.51	4.66	0.02
7	-1.29	0.65	0.76	1.06	-0.31	-	5	-31.73	74.72	5.86	0.01

**Chapter 3 – Baited video species richness regression output, RCA protection**

<b>Model #</b>	<b>Intercept</b>	<b>RCA Status</b>	<b>Complexity</b>	<b>Relief</b>	<b>Rocky substrate</b>	<b>Depth</b>	<b>df</b>	<b>logLik</b>	<b>AICc</b>	<b>delta</b>	<b>AIC weight</b>
12	0.49	-	0.53	0.02	-	-0.36	4	-87.54	183.9	0	0.48
6	0.55	-	0.59	0.03	-0.14	-0.37	5	-87.41	186.06	2.17	0.16
14	0.51	-0.04	0.52	0.03	-	-0.36	5	-87.52	186.29	2.39	0.15
4	0.71	-	-	-	-	-0.38	2	-91.75	187.73	3.83	0.07
global	0.56	-0.02	0.59	0.03	-0.14	-0.37	6	-87.4	188.59	4.69	0.05
9	0.55	-	-	-	0.24	-0.35	3	-91.13	188.73	4.84	0.04
8	0.74	-0.07	-	-	-	-0.39	3	-91.68	189.84	5.95	0.02
11	0.59	-0.09	-	-	0.26	-0.35	4	-91.01	190.83	6.93	0.02
3	0.52	-	0.65	-0.01	-	-	3	-93.68	193.83	9.93	0.00
1	0.46	-	0.58	-0.02	0.14	-	4	-93.53	195.88	11.99	0.00
13	0.52	0.01	0.65	-0.01	-	-	4	-93.67	196.16	12.27	0.00
2	0.46	-	-	-	0.49	-	2	-96.93	198.1	14.21	0.00
7	0.46	-0.01	0.58	-0.02	0.14	-	5	-93.53	198.32	14.42	0.00
10	0.48	-0.05	-	-	0.50	-	3	-96.9	200.27	16.37	0.00
5	0.78	0.03	-	-	-	-	2	-99.8	203.84	19.94	0.00