CHARACTERIZATION OF A NEW STEREO-VIDEO TOOL TO SURVEY DEEP WATER BENTHIC FISH ASSEMBLAGES WITH COMPARISON TO A REMOTELY OPERATED VEHICLE

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by

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CERTIFICATION OF APPROVAL

I certify that I have read Characterization of a new stereo-video tool to survey deep water benthic fish assemblages and comparison with a remotely operated vehicle by Christian Thomas Charles Denney, and that in my opinion this work meets the criteria for approving a thesis submitted in partial fulfillment of the requirement for the degree Master of Science in Biology: Marine Biology at San Francisco State University.

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Increasing use of ecosystem-based management strategies, which are often applied to broad geographic areas and preclude extractive activities, are creating a need for rapid, cost-effective monitoring of large areas. Visual surveys are increasingly being used to meet this need. In this thesis, I examine a new tool for surveying fish assemblages in deep-water habitat: a stereo-video Lander. In Chapter 1, I evaluate the utility of using a new stereo-video Lander for surveying fish communities. In Chapter 2, I compare the video Lander with a Remotely Operated Vehicle (ROV), and evaluate the strengths and weakness of each technique. In characterizing the new stereo-video Lander as a tool, I found that there was a negligible effect of bait. The rotating camera system yielded density estimates slightly lower than those determined by a stationary camera but that the rotating camera system produced less variance with the same number of surveys. In comparing the Lander and the ROV, both measured similar densities for most species. Because of the similarity in results and ability to quickly perform surveys and move on to new areas, the Lander represents a new option when considering visual tools for deep-water research.

I certify that the Abstract is a correct representation of the content of this dissertation.

Chair, Thesis Committee   Date

PREFACE AND/OR ACKNOWLEDGEMENTS

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Introduction

The California Current Ecosystem (CCE) in the eastern Pacific ocean contains several different benthic habitat types, which are distributed across a wide geographic area on and near the continental shelf off the U.S. West Coast. The largest of these is the soft-bottom shelf habitat, characterized by large areas of low relief, small (< 2 mm) sediment grain size, and often including transient features such as sand waves and ripples caused by bottom currents (Davis et al. 2013). These soft habitats can be interspersed with small deep-water rocky reefs, which can be comprised of a variety of hard-bottom substrates that range from gravel and boulder beds to contiguous-bedrock outcroppings. These reefs often have high relief and rugosity and can host large aggregations of fishes (Carlson and Straty 1981).

The CCE is a highly productive and economically important marine ecosystem and is host to a wide range of large, valuable fisheries such as market squid (*Doryteuthis opalescens*), Sardine (*Sardinops sagax*), and Dungeness crab (*Metacarcinus magister*) that are collectively worth over $100 million in 2010 dollars in California alone (Huyer 1983, Sweetnam 2010, Black et al. 2011). Hard-bottom habitat in particular is home to many demersal species important in commercial and recreational fisheries (Love et al. 1998, Willis and Anderson 2003, Anderson and Yoklavich 2007). Examples of some of the most valuable hard-bottom fisheries include the commercial live-groundfish fishery of about 250 tons annual catch and the recreational groundfish fishery, of which the two most important species (Black Rockfish; *Sebastes melanops* and Lingcod; *Ophiodon elongatus*) have a combined catch of over 1000 tons along the west coast (Figueira and Coleman 2010, Sweetnam 2010). Despite hosting these large fisheries, shallow-water rocky reefs make up a tiny minority (~6%) of continental shelf habitat (Johnson et al. 2015).

Despite comprising such a small amount of the available habitat, rocky reefs make up a disproportionate amount of the Essential Fish Habitat (EFH) for large number of demersal fish species (PFMC 2005, 2011). Species such as Yelloweye (*Sebastes ruberrimus*) and Cowcod Rockfish (*S. levis*), whose populations are depressed as a result of overfishing, can be found primarily on these hard bottom reefs. Although these reefs are important habitat for a variety of species, they can be difficult to survey effectively. They are often too deep for the use of SCUBA surveys and bottom trawls are easily lost when towed over high-relief rocky habitats. Remotely Operated Vehicles (ROVs) and Human Occupied Vehicles (HOVs), which are fully capable of surveying rocky habitats, are expensive and logistically difficult to utilize.

The difficulty of studying high-relief habitats is especially problematic because many of the groundfish species that utilize them are particularly susceptible to overfishing. Many species of groundfish, especially rockfishes, are slow growing, late maturing, and long lived (Love et al. 2002). These traits, in combination with the fact that the fisheries for these species can be large and valuable, mean that high quality data are of extra importance in managing these fisheries.

Stock assessments are the primary means by which fish stocks are monitored and managed on the west coast. At their core, stock assessments are models that help fisheries managers estimate the total number of new recruits into a fishery, total number of mortalities from natural causes (e.g., disease, predation), commercial or recreational take, and spawning biomass (Hilborn and Walters 1992). As a science-based management tool, they include a series of data requirements and assumptions (such as the availability of accurate life history data, an understanding of survey tool selectivity, etc.) to provide good estimates of stock status (usually defined as a ratio of fished to unfished biomass) and to better estimate the recruitment/mortality relationship (Hilborn and Walters 1992, Gallucci et al. 1995). In addition to the requirements and assumptions that many stock models carry, the use of high quality data sets can provide greater validity to the analysis. When a new tool or technique is proposed with the intention of informing future stock assessments, the qualities, characteristics, strengths, and limitations of the data the tool collects must be well understood, to fit in the larger framework of the stock assessment.

One of the primary data types used in stock assessments is fishery-dependent catch and effort data from the fishing fleet. When placed in historical context, these data can help understand how a stock is responding to fishing pressure. However, fisheries-independent sampling techniques are becoming more and more common as they allow for better control over sampling and study design. To conduct appropriately rigorous fisheries-independent monitoring, a variety of techniques have been used in different situations. Fisheries-independent survey methods currently used include: hook and line fishing (e.g., Wendt and Starr 2009), submersibles (e.g., Grimes et al. 1982, Ralston et al. 1986, Yoklavich et al. 2002), ROVs (e.g., De Marignac et al. 2009, Knight et al. 2014, Lindholm et al. 2015), trawl surveys (Wathne 1977, Gunderson and Sample 1980, Shaw et al. 2000), and acoustic surveys (e.g., Hampton 1992, Starr et al. 1996, Starr and Thorne 1998).

Each of these current tools has its advantages and disadvantages. Hook and line methods allow you to physically handle the fish for accurate identification and measurement. However, they are labor and time intensive and are typically extractive to a greater or lesser degree, even when used with catch and release methods. Usually, hook and line methods are best suited for relatively shallow, near-shore habitats and are more difficult in deeper, offshore waters. These fishing surveys can also have species- or size-selectivity biases depending on the gear type, hook size, and bait type used (Ralston et al. 1986, Willis et al. 2000a).

Trawl surveys similarly allow the handling of fish for accurate measurements and collection of physical samples. They are relatively inexpensive and can be used to survey large geographic regions with ease. Perhaps the largest advantage of trawl surveys is the long history of use (e.g., Coleman 1986, Shaw et al. 2000, Bradburn and Keller 2014). Trawl data have been rigorously tested and examined over many decades, and have a solid history for comparison in many environments and ecosystems. They do, however, have some downsides. Trawl surveys are even more extractive than hook and line surveys, having the potential to catch a greater number of individual fish as well as not being compatible with catch and release techniques. Additionally, trawls are poor at surveying complex, rocky bottoms and work better either on soft substrates or in the mid-water due to the risk of gear loss.

Acoustic surveys attempt to identify and estimate fish biomass using high resolution multi-beam sonar (MacLennan and Simmonds 1992). These acoustic techniques are able to rapidly and cost effectively survey large areas and numbers of fish. However, species identification can be difficult and acoustic surveys are mostly restricted to midwater species, as acoustic shadows and reflections make interpretation of fish near the bottom difficult.

ROVs and Submersibles have a different set of advantages and disadvantages. They are both visual, non-extractive tools, which make them suitable for use in protected areas and complex habitats that are not amenable to trawl or acoustic surveys. ROVs and submersibles are well suited to collecting information on species distributions, habitat associations, and relative abundance in many habitat types and locations. However they are also both expensive and slow to operate over large survey areas. For this reason ROVs and submersibles are most often used for detailed surveys of a small area of particular interest (e.g., Pearcy et al. 1989, Yoklavich et al. 2002).

These common tools leave a gap in survey capability: the ability to rapidly, and cost-effectively survey large areas of complex habitat, much of which may be protected from extractive sampling. In light of a need for a survey technique to fill in this gap, we designed a new survey tool: the stereo-video Lander. The stereo-video Lander is a drop-camera system mounted on a weighted aluminum frame. The cameras and lights are mounted on a bar, which rotates a full 360 degrees giving complete coverage of the drop site. The Lander is tethered to the boat for power and to permit a live video feed. The Lander system is capable of setting in complex bottom types and quickly surveying an area before being recovered and redeployed. The stereo-video Lander is inexpensive to build and operate relative to other remote or human occupied video tools. Similar camera systems have been used extensively in Australia (BRUVS, Langlois et al. 2006, Watson et al. 2009, Lowry et al. 2012) and in Oregon (Hannah and Blume 2012, 2014, Easton et al. 2015), although usually with differences such as the use of stationary cameras or a non-stereo camera setup.

In the first chapter of my thesis I provide an analysis of the capabilities of the Lander and evaluate various protocols to develop best practices for assessing demersal fish in rocky habitat using a rotating, stereo drop camera system. I seek to examine the accuracy of the stereo-video length measurements and the effects of bait, soak time, and rotating cameras on data collected by stationary drop camera systems. How these factors influence video surveys will help choose methodologies most appropriate to specific scientific questions.

In the second chapter, I attempt to characterize the differences between the video Lander and a more traditional ROV tool, providing advice on how researchers could select one tool versus the other for a particular question, as well as addressing larger questions about sampling along the Eastern Pacific continental shelf. Exploring the details of how the video Lander collects data and how those data compare with data from other, better understood tools will hopefully help video Lander collected data fit into the larger stock assessment framework as a means of filling a data-poor gap in current sampling. Towards these goals, I compare the community composition, length distribution, and species-specific density estimates between the video Lander and a ROV.

Chapter 1: Characteristics and methodologies for the use of a rotating stereo-video Lander

Introduction

A variety of tools have been used to survey benthic fish communities, such as trawls, hook-and-line fishing gear, and remotely operate vehicles (ROVs). These existing survey methodologies leave a capability gap for an inexpensive, logistically simple tool that can effectively survey complex rocky habitats. Still-photo Landers and single-camera video Landers are effective tools for surveying rocky reef fishes (e.g., Willis et al. 2000a, Harvey et al. 2007, Easton et al. 2015). In addition, stereo-video cameras can accurately size fish without the need to physically handle the individuals (Colton and Swearer 2010, Taylor et al. 2013, Hannah and Blume 2016). Given this history of use of similar systems, we designed a stationary stereo video system with added capability of rotating cameras to allow for increased survey area. The Lander is a non-destructive and non-extractive tool designed for use in complex rocky substrates that are difficult to survey with other techniques. This tool is similar to other drop camera systems used elsewhere such as Baited Remote Underwater Video Cameras (BRUVs) in Australia and single camera systems in Oregon (Hannah and Blume 2012, De Vos et al. 2013), with a few important modifications.

With any new tool, it is necessary to do basic testing on how the tool operates, to validate the efficacy of the tool in different physical environments, and to quantify the strengths and weakness relative to other techniques. Various types of remote video Landers and drop cameras have been used during the past several years and a variety of testing has been done to assess their performance. For example, Hannah (2012) established that a duration of 5 minutes was sufficient to allow sediment to clear and ensure that fish were not double counted based on a qualitative assessment of the video. However, the Lander they used had a non-rotating camera system and the qualitative nature of the determination leaves room for further analyses.

Hannah (2014) also examined the influence of bait on the data collected by a stereo-video Lander and found that bait produced changes in species composition and increased apparent relative abundance of some groundfish species. Additionally, they found that estimates of fish lengths where the Lander was baited were more accurate as a result of individuals coming closer to the cameras. Other researchers have shown that the type of bait, current strength, and target species all influenced interactions with the observations made using baited camera systems (e.g., Dorman et al. 2012, Taylor et al. 2013, Wraith et al. 2013). However, most of these studies were conducted with long drop durations, ranging from 15 minutes to over an hour, and in some cases the drop duration was chosen specifically to allow time for the bait to more effectively attract fishes (Hannah and Blume 2014). Shorter duration drops likely reduce the impacts of bait, and other related factors such as current speed, on species composition, but may still allow for increased accuracy in measurements by bringing fish closer to the cameras.

Harvey and colleagues (Harvey and Shortis 1998, Harvey et al. 2002, Watson et al. 2010) published a variety of work on the accuracy of stereo-video systems, finding that measurement accuracy can be as good as 1% of target length. However, measurement accuracy is dependent on the exact configuration of the stereo system and the particulars such as camera model and video resolution. Factors such as the particulars of the housing window material, separation of the cameras, camera axis, and pixel size of the digital sensor can all impact stereo measurements and calibration. For example, cameras that are too closely placed will have less accuracy than cameras farther apart, and the measurements will be most accurate along the same axis as camera separation. Higher resolution cameras also will result in more accurate measurements as it is easier to identify and mark the exact tip of the head and tail on the fish in different camera views with higher resolution video. For these reasons, measurement accuracy on any new stereo video system should be tested and calibrated rather than relying on published values in the literature.

Most of the tools used to assess fish populations only allow for estimates of relative abundance (average number of fish per survey), rather than metrics that can be extrapolated to obtain absolute abundance (e.g., density estimates). This is a result of two main factors. Single cameras often are unable to accurately measure the area surveyed (especially for side facing transects such as those used by submersibles) whereas most of the stereo systems deployed in the past (especially BRUVs) used bait and long soak times, which would artificially increase the density of fish over natural levels. This means that it is difficult to compare survey results among different tools. We designed the new stereo camera setup and used short soak durations to enable estimates of fish density, which will increase the utility of this tool for incorporation into stock assessments and comparisons with other visual tools, such as ROVs.

In light of the previous work on video Landers and stereo systems, I developed several objectives for testing our video Lander. My first objective was to evaluate the effect of soak time on species counts and species richness. I tested these effects by measuring species accumulation and changes in fish abundance, using the response metric of MaxN, over time. MaxN is traditionally used to identify the minimum number of unique individuals of a given species present on a video survey, through quantification of the maximum number of fish observed on a single video frame, using stationary camera systems. For our rotating system, I defined MaxN as the greatest number of individuals seen on a single full rotation. As a second objective, I evaluated the effect of bait presence or absence on density estimates and species richness of fishes. My third objective was to explore the accuracy of the fish length measurements produced by our specific stereo configuration. My fourth objective was to identify the best ways to calculate the area surveyed for a rotating system, to enable calculations of fish density (i.e., number fish observed per unit area).

For the first objective, I hypothesized that MaxN values would approach an asymptote over time, denoting diminishing returns with longer drop durations. I further expected that the point at which the accumulation curve levels off was likely to be greater than the five minutes used by Hannah et al. (2012) but less than the 1 – 1.5-hour-long drops frequently used in BRUV surveys.

For the second objective, I hypothesized that, during shorter duration drops, bait would not have a significant impact on fish abundance or density. Many baited systems are used with extremely long, hour-plus soak times to try and perform a comprehensive species census of an area. However, I proposed that longer soak times would result in diminishing returns on the number of species observed and that for our species of interest (primarily rockfishes, Table 1) I can obtain good estimates of abundance/density in shorter drops without bait attracting inflated numbers from surrounding areas.

For the third objective, I hypothesized that angle and distance from the cameras would have a significant impact on the accuracy of length measurements, with targets that are closer and more parallel to the cameras producing more accurate measurements. I also hypothesized that accuracy of measurements typical of those taken in the field (largely parallel, between 1 – 2.5 m) would have errors of less than 5% of target length.

For the fourth objective, I attempted to establish a species-specific, area-surveyed metric. The video Lander surveys the fish present in a specific location, however, different fish species are identifiable at different maximum distances based on coloration patters and behavior, so that different species have different effective survey areas. Therefore, using a single area for all species could result in underestimating the density of one species by including more area than was actually effectively surveyed for that species while overestimating the density of another by artificially decreasing the effective area surveyed.

Methods

Lander design and operation: The video Lander consisted of an aluminum frame 1.5 m tall, 1 m in diameter, and weighing 45 kg, with 70 kg of lead weight added during operations. This frame houses two Deep Sea Power and Light (DSPL) cameras with 620 TV line (TVL) resolution. The cameras were mounted on a rotating bar, with both cameras facing out from the Lander in the same direction, and with each camera toed-in towards the center of the bar approximately 5 degrees from pointing straight ahead. Additionally, two DSPL LED lights outputting 3000 lumens at a color temperature of

5000K were mounted above the cameras. Finally, the electronics bottle that housed the video recording devices (two Stack LTD™ DVRs with removable 32GB storage cards) as well as the electronics necessary to handle the power and a data stream to the boat was mounted behind and above the rotating bar (Fig. 1). The video Lander was deployed from the deck of a fishing vessel using the main winch and was supported on a high-tensile-strength line. Additionally, an umbilical that supplies power and allows data transfer up and down from the electronics bottle was attached to the video Lander. The umbilical was rated to support the weight of the video Lander in the case of an emergency.

During this project, the video Lander was deployed in field surveys to characterize fish assemblages off central California in depths of 70 m to 230 m (Table 2). Each time the video Lander was put over the side of the boat was referred to as a deployment. Within a single deployment, the video Lander was picked up and placed at several different locations, within 1 km of each other. Each of these locations within a deployment is referred to as a “drop”. Each drop lasted for 8 minutes, plus time for the sediment to clear in the beginning of the drop, which was usually 1-2 minutes. The cameras on the video Lander completed one 360-degree rotation approximately every minute (each rotation is known as a sweep) with a total of 8 sweeps per drop. Video was recorded on the Lander but the video feed was also transmitted up the umbilical, which allowed for real time viewing by the science crew and captain on the boat.

Data collection: Once video was collected at sea, it was returned to the lab for analysis. I used a program called EventMeasure (v. 4.42, SeaGIS) to mark and measure the total length of each fish in three-dimensional space. Properly calibrated, the software is able to plot any point identified by the user in both cameras in three-dimensional space and calculate distances between two such points, through the principle of co-linearity (Harvey and Shortis 1995). When the user identifies the head and tail of fish in both cameras, this allows the software to generate a straight-line length estimate to the nearest 1 mm, although data were rounded to the nearest 10 mm for analyses. For each species, the sweep with the greatest number of identified individuals was used as the total count for that species on a drop. For stationary underwater visual systems, fish have traditionally been counted using the metric of MaxN, which is defined as the maximum number of individuals per species observable in a single camera frame (e.g., Willis et al. 2000, Heagney et al. 2007, Watson et al. 2010). I used our modified MaxN to select the sweep with the greatest number of individuals and only those individuals were measured for length estimates and exported to our database.

Optimal soak duration: Previous work with BRUVs often included soak times of over an hour (Langlois et al. 2006, Watson et al. 2009, De Vos et al. 2013) while trying to get complete species counts. Hannah and Blume (2012), however, found that a shorter, five-minute duration was sufficient for getting accurate counts of their species of interest (rockfishes). Hannah also reported that bait-effects (i.e., attraction to the bait canister) were noticeable after 12 minutes of soak time (Hannah and Blume 2014). For this project, I were primarily concerned with accurate counts of our species of interest so I elected to employ the shorter durations used by Hannah et al. (2014). However, I wanted to examine the impact of drop duration in a more quantitative way than previous work had done. Optimal drop duration comes from a combination of factors including biological data such as fish behavior, environmental conditions such as current speed and water clarity, logistics such as the sampling needs of the project, and the realities of vessel operations with a tethered tool. Due to the constraints listed above, it was necessary to identify a drop duration that enabled us to collect high quality, useful data, while minimizing post-collection processing and still allowing the captain to maintain station over the Lander in inclement weather. For vessel operational reasons, and because of the time used by Hannah, I started with an initial duration of 12 minutes and examined how MaxN counts and species accumulation curves changed with drop time up to that limit. Where the accumulation curve for each species reached an asymptote was determined to be the time required to get a count of a particular species.

Efficacy of baiting: Stationary video systems have most often been used with bait and relatively long deployment times to get estimates of fish length, community composition, and population indices (e.g., Langlois et al. 2006, Lowry et al. 2012, De Vos et al. 2013, Easton et al. 2015). However, I used substantially lesser drop durations than some of the baited surveys, and wanted to test whether baiting affected estimates of fish density and species richness. For our study, bait consisted of 150 grams of market squid (*Doryteuthis opalescens*, roughly three squid), cut into large pieces and put into a plastic jar which was then zip tied to the base of the Lander, below the line of sight of the cameras. Bait was replaced after every deployment. To test the effect of bait, I conducted 13 baited and 15 unbaited drops. I then performed a t-test on the number of species and total number of fish observed. In addition to the direct comparison, I also performed regression analysis on the number of fish of each species counted in each sweep from a different set of 34 baited drops to determine if time was a significant predictor of which sweep would contain the MaxN value for baited drops. Finally, I compared the species accumulation curves between baited and unbaited drops to see if baited drops had greater species diversity than unbaited drops.

Measurement error: All measurements contain some degree of error from a variety of sources. In our measurement of fish lengths, error comes from two primary places: the accuracy of the software and the accuracy of the video analyst in placing the points that the software uses. The accuracy of the software is related to the physical parameters of the system, including camera type, the quality of the calibration, and the position of the target in relation to the cameras. To test the total error of our system, I brought the video Lander to the Monterey Bay Aquarium Research Institute (MBARI) test tank. A target of known size was put in the water at a variety of ranges and angles relative to the Lander. This video was brought back to the lab for analysis where more than 400 length measurements were determined and compared with the known length to estimate measurement error expressed as a percent of target length. Based on the distribution of calculated errors, I used a beta-distributed GLM to determine the error as a function of distance of the target from the cameras and horizontal angle relative to the camera bar. The predicted error model from the GLM was then used to filter length measurements from our field deployments that had a predicted error of greater than 10%.

Survey area: For side-facing video transects or outward-facing point-count tools, it is usually difficult to estimate the area surveyed because of the difficulty in calculating the distance of organisms from the cameras/observer and the maximum distances at which observations can be reliably made. Because of the Lander's ability to accurately measure positions in three-dimensional space, it was possible to determine the effective distance from the Lander at which different species could be accurately observed and identified. This allowed us to create an estimate of the area surveyed for each species. To do this, I created two different metrics: a minimum detection distance and a maximum reliable detection distance. The minimum detection distance reflects the minimum distance at which a fish can be seen and identified and is a product of the physical shape of the Lander and the view field of the cameras (Fig. 2). Because it relates to the physical parameters of the Lander, the minimum detection distance is common to all species observed. Minimum detection distance was calculated in the Monterey Bay Aquarium Research Institute (MBARI) test tank by measuring the distance from the cameras that an object could be observed while on the bottom. The minimum detection distance was 81 cm from the cameras, and 104 cm from the center of the main pillar of the video Lander (Fig. 2). Fish up off the bottom but within this minimum detection distance were excluded from count data. The maximum reliable detection distance, however, is unique to each species; I defined it to be the distance from the Lander at which 95% of all individuals of each species Ire observed. This max cutoff distance allowed us to avoid an artificial drop in density, which could arise from using survey areas where I could only identify a particular species under ideal circumstances but where most individuals would not be identifiable.

Besides determining the maximum reliable detection distance for each species, I also explored the number of drops required for the maximum reliable detection distance to stabilize, or after which further sampling would not change the maximum reliable detection distance. This demonstrates that our maximum reliable detection distance metric represents an estimate of the distance at which a given species is consistently identifiable. to determine the number of drops required, I used a bootstrapping technique to randomly select an increasing number of drops a thousand times, from three drops to the maximum number of drops containing a given species. For each bootstrapped drop sample, I calculated the maximum reliable detection distance for each group of drops, for each of the thousand bootstrapped replicates. The average maximum reliable detection distance for the thousand replicates was then plotted against number of drops selected. The number of drops required to determine a "stable" maximum reliable detection distance was determined based on the slope of the curve. When the maximum reliable detection distance reached an asymptote, it was considered stable.

Rotating vs. stationary cameras: The video Lander has a rotating camera system but I also wanted to understand how the data collected by our rotating system would compare with data collected by a stationary camera system. To compare these two different techniques, I generated a stationary data set from the rotating data. I calculated the amount of time required (15 seconds) for the Lander's cameras to sweep 90 degrees, as an analogue for a stationary system. I then recorded only fish that were observed during that time period of each rotation and calculated MaxN of each drop of rotating and stationary cameras. This meant that the simulated stationary data might have a different MaxN sweep than the rotating data. I then calculated density for each method by using the relative areas surveyed (i.e., fish counts were divided by 1/4th the area for the stationary cameras as compared with the rotating cameras). Finally, I compared densities and data variability between the "stationary" data and the full rotational data. Total fish density was calculated by summing the density values for the 14 species of interest (Table 1), which was then compared with a paired t-test between the rotating and stationary camera data sets as well as using a bootstrapped mean difference value for the density estimates of each species.

Results

Optimal soak duration: For each of the 14 species of interest, I plotted the average proportion of final density observed after each sweep (Appendix A). I also calculated the average of all 14 species of interest together (Fig. 3). This figure and those for each species demonstrate that the density for most species reached an asymptote, as determined by when best fit curve reached 85% of the maximum observed value, after approximately eight minutes (Appendix A, Fig. 3). The remaining 15% increase took the final 33% of the soak time, which demonstrated the diminishing returns of longer drops. Similarly, the average number of species observed per drop leveled off as a percentage of max value after approximately four minutes (Fig. 4). Vermilion Rockfish reached the 85% of max value point the soonest at four minutes. The species that reached this point the slowest were Copper Rockfish, Canary Rockfish, and Cowcod at 10 minutes (Table 3).

Efficacy of baiting: A total of 13 species were observed across all baited drops with an average of 4.3 species per drop (N = 13, SD = 1.9). Eleven species were observed on unbaited drops with an average of 4.4 species per drop (N = 15, SD = 2.3). The maximum number of species seen on a single unbaited drop was nine, compared with eight for baited. The minimum number of species seen on a single unbaited drop was one compared with two for baited. Three species (Greenspotted Rockfish, Widow Rockfish, and Halfbanded Rockfish) were observed only on baited drops whereas one species (Bocaccio) was observed only on unbaited drops. There was no significant difference between the mean number of observed species on baited and unbaited drops (Two Sample Student's t-test, t_{26} = -0.113, p = 0.9107).

The greatest number of fish observed on an unbaited drop was 26, whereas the greatest number of fish observed on a baited drop was 39. The fewest observed fish on an unbaited drop was nine, whereas the fewest on a baited drop was six. There was no significant difference between the total mean number of fish observed on baited vs. unbaited drops (Welch's t-test, $t_{13.741} = 0.43465$, $p = 0.67$).

On the baited drops, there also was no effect of time on the likelihood of a sweep containing MaxN (Appendix B). The only species for which there was a significant correlation with time (implying a potential impact of bait) was Canary Rockfish, which was positively correlated with time, and Lingcod and Yelloweye Rockfish, which were negatively correlated with time.

Measurement error: A total of 421 measurements of a target of known size were calculated in the MBARI test tank at a variety of distances from ~0.5 m to ~2.5 m, spanning horizontal angles of between 0 and 90 degrees relative to the cameras (0 degrees being perfectly parallel and 90 degrees being perfectly perpendicular). Measurement error ranged from <0.01% to 31.79% of target length with a mean value of 2.89%. Distances to target and angle of target were used as predictor variables in a beta-distributed GLM to create a model of predicted measurement error. The identity link function was used. The model obtained was as follows:

$$Error = \frac{e^{(-5.135+3.4*10^{-4}*A+5.03*10^{-4}*D+\ 7.53*10^{-6}*AD)}}{1+e^{(-5.135+3.4*10^{-4}*A+5.03*10^{-4}*D+\ 7.53*10^{-6}*AD)}}$$

where A represents the horizontal angle of the target and D represents the distance to the target. The pseudo R-squared value for this model was 0.32. This model was then applied to more than 7000 fish length measurements in our database, given our field estimates of distance to the target and the horizontal angle to the cameras. Results indicated that >95% of fish measurements contained a predicted error of less than 5% of measured body length (Fig. 5), with only 46 measurements having a predicted error greater than 10%.

Survey area: For the species of interest observed by the video Lander, the maximum reliable detection distance value fell between 2.3 and 3.7 m (Table 4) with the greatest maximum reliable detection distance being Lingcod, at 3.7 m, whereas the smallest maximum reliable detection distance was at 2.3 m for Starry Rockfish. The maximum reliable detection distance stabilized for most species after approximately 40 drops, with a maximum of 55 drops for Lingcod and a minimum of 33 drops for Pygmy Rockfish (Table 4).

Rotating vs. stationary cameras: A total of 261 drops was used to test the difference between rotating and stationary cameras. The stationary cameras yielded a significantly greater total fish density (of the 14 species of interest) than the rotating cameras (Paired t-test, $t_{112} = -3.30$, $p = 0.0013$). The mean total density of the 14 species of interest combined for rotating cameras was 0.90 +/- 0.09 fish/m^2. The mean total density for 14 species of interest combined for the stationary cameras was 1.09 +/- 0.11 fish/m^2. The bootstrapped mean difference values for the 14 species indicated a tendency for slightly greater densities on the stationary camera, with Bocaccio, Copper Rockfish, Lingcod, Starry Rockfish, Vermilion Rockfish, Yelloweye Rockfish, and Yellowtail Rockfish being significantly greater using the stationary cameras (Fig. 7), although the difference was only about 0.01 fish/m^2 for all of those species except Vermilion Rockfish. Mean variance for the fish observed with stationary cameras was 1.79 times greater than those observed with rotating cameras.

Discussion

The soak duration for an underwater video survey tool is an important methodological choice. Longer soaks can potentially increase the number of species detected and increasing the number of individuals observed because fish are drawn in from a larger area covered by the bait plume, requiring larger areas to be used in the density calculations. The tradeoff of longer soak times is increasing the logistical difficulty of data collection (primarily holding position over the Lander) in the field as well as increasing the time and cost of collection and analysis during post-processing, while also reducing number of samples over a given time in the field. Additionally, for highly mobile species, greater soak times may overestimate densities if bait plume dispersion is not well understood and corrected for (Ward-Paige et al. 2010). It is therefore important to find the point at which increased soak times result in diminished returns and are no longer worth the tradeoffs in time and cost while still providing an accurate estimate of fish populations. Optimal soak duration was one of the most troublesome methodological questions to address for this project, mostly because it was informed not just by the biology, ecology, and behavior of the fishes but also by the logistics of the vessel and ability to hold position over the drop site in occasionally adverse weather conditions. By calculating the point at which species accumulation curves leveled and when MaxN estimates indicated diminishing returns, I could set lower bounds on drop duration to accommodate the logistical concerns of the vessel and data collection/analysis. This information, in conjunction with the captain’s input on the ability to hold location over a given survey site led to our drop duration of eight minutes.

Many underwater video systems have been baited (e.g., Heagney et al. 2007, Colton and Swearer 2010, De Vos et al. 2013), which necessitated disentangling the effects of bait attraction and environmental factors on community assemblage and density. As a result, the common practice for these types of tools has mostly settled on using relative abundance (i.e., MaxN) as the metric of abundance, which restricts direct comparisons to data collected using similar tools (e.g. ROVs) and different methodologies (e.g., strip transects). However, because our Lander was tethered to the boat, which logistically limited the length of the soak time, I had significantly lesser drop times than most other stationary drop camera studies. As a result, there was less bait-plume dispersion and I saw no impacts of baiting on numbers of fish observed or number of species observed. Similarly, during our baited drops, I could detect no impact of time on the number of fish observed. Our suspicion is that the bait acted only to draw fish already present slightly closer to the cameras for easier identification. Our lesser drop times (and corresponding small bait plume), combined with the fact that I rarely observed fish attempting to feed off of the bait bottle, reinforces this idea that the numbers and composition of fish observed in our study were less impacted by the use of bait than in other studies. Because of this lack of significant bait effect, I believe our density estimates are not inflated from bait attraction of more distant fishes and are therefore more likely to represent true density values than a similar tool using bait.

Video surveys have the distinct disadvantage of not having the survey targets in hand for identification or sizing. The problem of identification can only be solved with increased training and improved video cameras and lighting, but the problem of sizing has traditionally been approached with the use of paired lasers (e.g., Deakos 2010, Rohner et al. 2011, 2015). The use of calibrated stereo-video photogrammetry, however, allows for more accurate sizing than paired lasers (Harvey et al. 2002). To obtain accurate fish length measurements, it is necessary to calibrate the stereo-video system based on the particulars of that system. I performed all of our calibrations and size testing in the MBARI test pool, which allowed for easy use of a variety of targets and multiple tests. However, water clarity and lighting were both significantly better in the test tank relative to open water deployments and it is therefore likely that calculated measurements represent an ideal situation. However, our tests show that sizing is usually not negatively impacted until extreme combinations of the angle of the fish to the camera and the distance of the individual from the camera were reached. Although these limits were most likely lower in field use, an analysis of our actual measurements indicated that in the vast majority of cases, video analysts took measurements only in the cases of conservative angles and distances, and the predicted error was usually less than 2% of target length, giving some amount of room for larger errors in the field.

Conceptually, there are three factors that should influence the accuracy of sizing a target. Two of them: distance and angle, have been accounted for in the model described here. The third is more difficult to quantify directly and is also closely related to the cameras used: the ability of the video reader to accurately and precisely locate and mark the tip of the nose and tail in the video. However, it also is likely that this ability is relatively consistent across targets and any error it introduces is of similar magnitude regardless of target distance or angle. The best way to deal with this source of error is during tool design when picking the cameras. Higher definition cameras will likely result in an increased ability to mark these points, resulting in decreased error. Preliminary testing with HD GoPro cameras has shown error rates of between 0.5% and 1% (unpublished data). As a final note, the in-pool testing was done before I knew the full range of maximum reliable detection distances I would encounter in the field. As a result, the distance testing does not quite capture the full range of distances at which I actually measured fish. However, distance on its own (i.e., without a high angle in addition) seemed to have a relatively small impact on measurement accuracy. Because most length measurements were determined at low angle and the fact that most species fell within the tested limit, I do not believe that this had a large impact on the accuracy of the vast majority of measurements. For those few measurements taken at extreme distance (most notably Lingcod), the conservative angles used mean that the relationship between angle, distance, and measurement error would need to undergo a drastic change between 3.5 and 4 meters distance for the actual measurement errors to fall outside of acceptable limits. For these reasons, I do not believe that measurement error is a major issue for even the most distant of measured fish in this study.

Because I was attempting to calculate actual fish density values, I needed to establish the area surveyed by a single drop. The camera recorded video in a doughnut shape with the outer limit being defined by the limit of visibility. Individuals of different species vary widely in their ability to be identified at a given distance from the cameras based on size, morphology, coloration, markings, and swimming patterns. It was therefore necessary to establish a different functional radius of detectability for each species. To omit extreme situations of exceptional lighting or visibility situations, I chose to use the distance at which 95% of a particular species had been identified (the maximum reliable detection distance). This metric, however, relies on having a large enough sample size to capture the variability in identifying individuals of a given species. By examining how the maximum reliable detection distance changed with increased sampling effort, I could pinpoint the number of samples (containing a particular species) that were necessary to achieve a stable estimate. The point at which the maximum reliable detection distance stopped changing with increased samples (approached the asymptote) was determined to represent the “true”, stable estimate for identifiable distance for each species (Table 4, critical slope value). For many species tested, I had more than 200 observations, and rarely did I have fewer than 100. For all species of interest, the maximum reliable detection distance estimate stabilized at between 30 and 50 samples, indicating that for this metric I had more than enough samples to be confident that our calculated distance was representative for each species.

When using a maximum reliable detection distance, it is important to remember that it is meant to be an average value, meaning that it is not calculated per each drop based on water conditions or other specific environmental factors. For any individual drop or survey, water clarity, the presence of large rock walls and other factors could all influence the area visible to the cameras. On average, these changes should not influence the density estimates derived from maximum reliable detection distances as they become included in the overall variability of species-level identification. Testing the exact influences of these conditions was outside the scope of this project. For this reason, maximum reliable detection distance should not be used to calculate area surveyed for individual drops or from studies with a small number of surveys, as small deviations could create large discrepancies under these circumstances. In the future, with more granular habitat complexity measurements and a turbidity meter, the factors influencing maximum reliable detection distances could be better understood. Even with these physical measurements however, there will be difficulty in quantifying how aspects of the fish itself other than size, such as body shape, pattern, and color influence the ability to identify a given species.

In testing the rotating camera design of the video Lander, I compared fish densities from the full rotating data set to data created by using only a fraction of each rotation (and thus a fraction of the surveyed area), thus simulating a stationary camera. I found that the simulated stationary cameras estimated slightly greater densities in aggregate (on the order of 0.01 fish/m^2), and that the difference was significant for about half of the species I tested. This differences was unexpected. One possible explanation is that recent simulation work has shown that limited-view tools can exhibit a non-linear relationship between estimated and real density (Campbell et al. 2015), with larger non-linearity when real density is increased. Besides density, the rotating camera system exhibited dramatically reduced variability (Fig. 8) compared with our simulation of a fixed camera. Because the two techniques were both stereo visual surveys, but with a larger/smaller area surveyed, decreased variability is what I would expect from the rotating camera system. The same number of surveys of a larger area should result in less variability compared with surveys of a smaller area because of increased homogeneity in surveys with larger area. For example, a rotating system will always detect fish hiding in the lee of the tool, whereas a stationary system will occasionally see the fish in the lee, and occasionally miss them depending on orientation to the current, resulting in larger or smaller density values. This allows a rotating camera system to achieve a less variable estimate of population parameters with fewer samples. For these reasons, I believe that a rotating camera system or other means of 360° viewing will provide better density estimates.

It is necessary for these types of tests and calibrations to be performed for every new video system. For example, measurement accuracy is dependent on the quality of cameras used, so the formula for predicted measurement error I produced is unlikely to be perfectly applicable for other video systems. Similarly, maximum reliable detection distances are dependent on the type of cameras, lights, and on environmental factors, so studies using different cameras or in a different environment would need to calculate their own maximum reliable detection distances for density calculations.

Overall, the extensive testing I conducted demonstrates how various practices for a tethered, rotating, stereo-video Lander affect data collection. My testing of the video Lander indicated that it is capable of recording accurate length estimates, that reasonably accurate fish counts can be obtained in relatively short amounts of time, that survey area can be estimated for each species and that with shorter drop durations, bait has a negligible impact on species counts or density estimates.

I believe that this tool represents a cost-effective way of sampling large, rugose marine habitats in a timely fashion while collecting useful data that allows for comparison with other visual survey tools. By changing several of the use protocols, different questions can be asked. For these reasons, the Lander represents a flexible, inexpensive new tool for surveying rugose, deep-water habitats.

Chapter 1 Tables

Table 1: Species of interest for this project. Composed of a combination of the most abundant species and those of importance to the commercial or recreational fishery.

Common Name	Latin name
Bocaccio	*Sebastes paucispinus*
Canary Rockfish	*Sebastes pinniger*
Copper Rockfish	*Sebastes caurinus*
Greenspotted Rockfish	*Sebastes chlorostictus*
Halfbanded Rockfish	*Sebastes semicinctus*
Lingcod	*Ophiodon elongatus*
Pygmy Rockfish	*Sebastes wilsoni*
Rosy Rockfish	*Sebastes rosaceus*
Squarespot Rockfish	*Sebastes hopkinsi*
Starry Rockfish	*Sebastes constellatus*
Vermilion Rockfish	*Sebastes miniatus*
Widow Rockfish	*Sebastes entomelas*
Yelloweye Rockfish	*Sebastes ruberrimus*
Yellowtail Rockfish	*Sebastes flavidus*

Table 2: Number of Lander deployments by year.

Year	Number
2012	35
2013	176
2014	204
2015	44

Table 3: Number of rotations required to reach 85% of MaxN value as assessed at the end of the 12-rotation drop.

Species	Rotations to reach 85% of MaxN
Bocaccio	7
Canary Rockfish	10
Copper Rockfish	10
Cowcod	10
Greenspotted Rockfish	6
Lingcod	6
Pygmy Rockfish	6
Squarespot Rockfish	6
Starry Rockfish	6
Vermilion Rockfish	3
Widow Rockfish	8
Yelloweye Rockfish	7
Yellowtail Rockfish	5

Table 4: List of model parameters and final maximum reliable detection distances for the most commonly observed species. The model described is the best-fit log model of maximum reliable detection distances with increasing number of samples. Critical slope value is the number of drops after which the slope of the best-fit line falls below 0.0005. Maximum reliable detection distances are in meters. Graphs of maximum reliable detection distance as a function of number of surveys can be found in Appendix A.

Species	Coefficient	Intercept	Critical Slope	Max. reliable detection distance	Effective Area Sampled (m^2)
Bocaccio	0.994	4.439	45	3.454	35.4
Canary Rockfish	0.843	3.851	41	3.021	26.6
Copper Rockfish	1.066	3.672	46	2.636	19.8
Greenspotted Rockfish	0.645	3.485	36	2.847	23.4
Halfbanded Rockfish	0.795	3.532	40	2.753	21.7
Lingcod	1.507	5.197	55	3.713	41.2
Pygmy Rockfish	0.537	2.939	33	2.410	16.2
Rosy Rockfish	0.738	3.106	39	2.382	15.8
Squarespot Rockfish	0.765	3.097	39	2.368	15.6
Starry Rockfish	0.992	3.310	45	2.350	15.2
Vermilion Rockfish	0.681	3.741	37	3.064	27.4
Widow Rockfish	0.746	4.088	39	3.390	34.0
Yelloweye Rockfish	1.345	4.326	52	3.026	26.7
Yellowtail Rockfish	1.074	4.664	46	3.620	39.1

Chapter 1 Figures

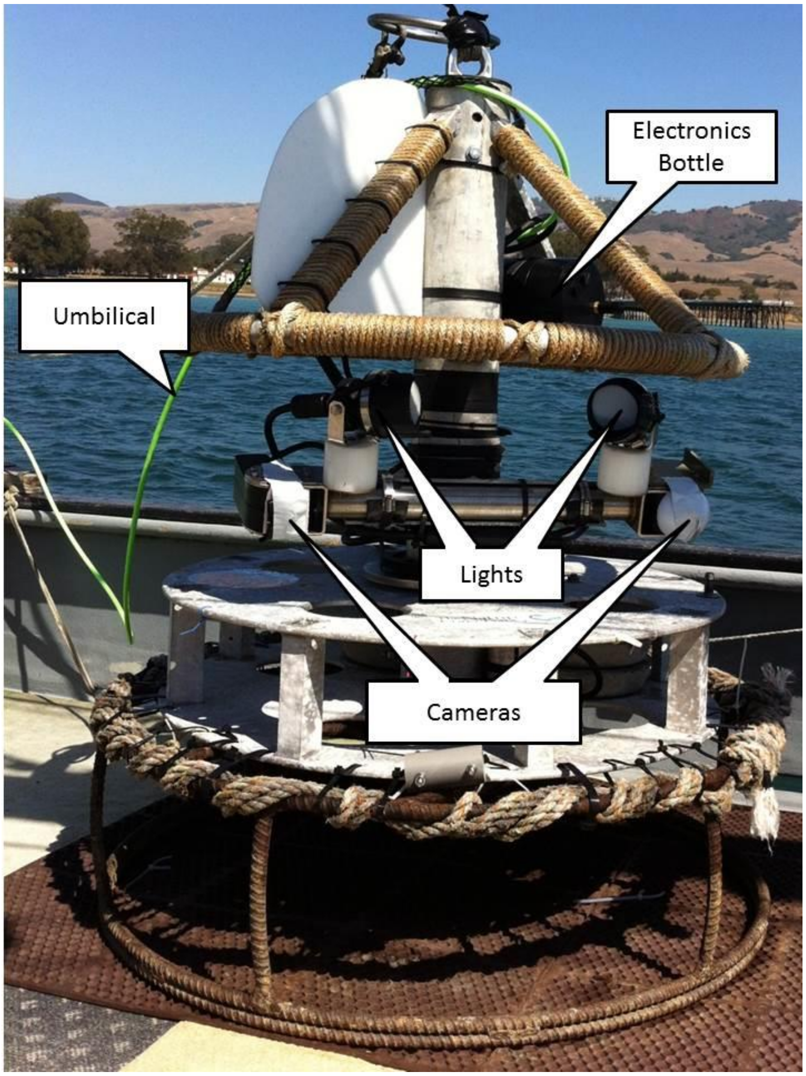

Figure 1: Lander Design. Primary Lander frame is attached to a converted crab pot with heavy-duty zip ties and breakaway line. The camera bar rotates around the central post at a rate of 1 rotation/minute. Lights are mounted on the camera bar directly above each camera. The electronics bottle contains the DVRs and converts the power and signal carried by the umbilical to the actions required for the lights, cameras, and motor.

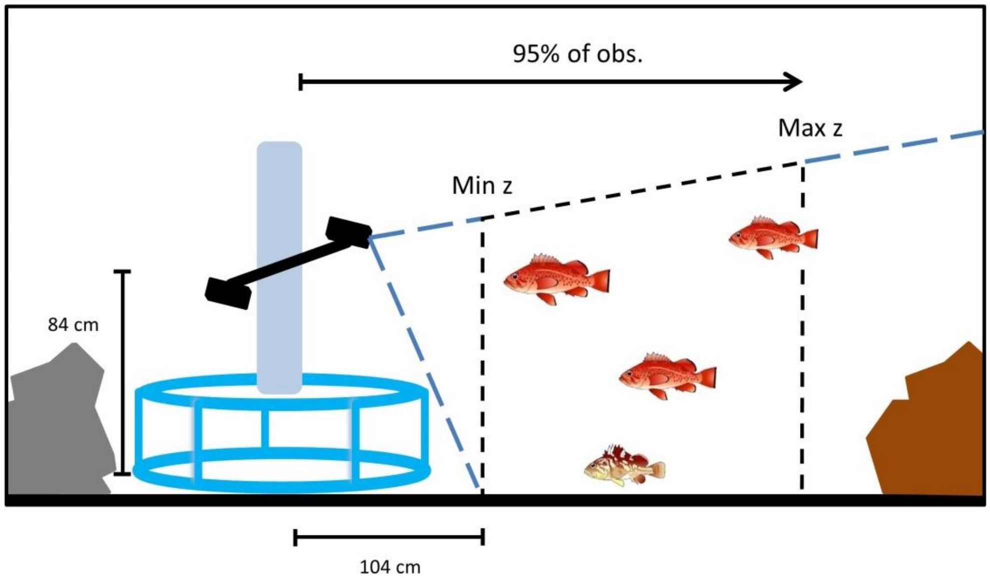

Figure 2: Survey area diagram. Minimum detection distance is determined by the shape of the Lander and the closest distance at which the bottom can be observed based on that shape. Maximum detection distance (Max Z) is determined on a species by species basis.

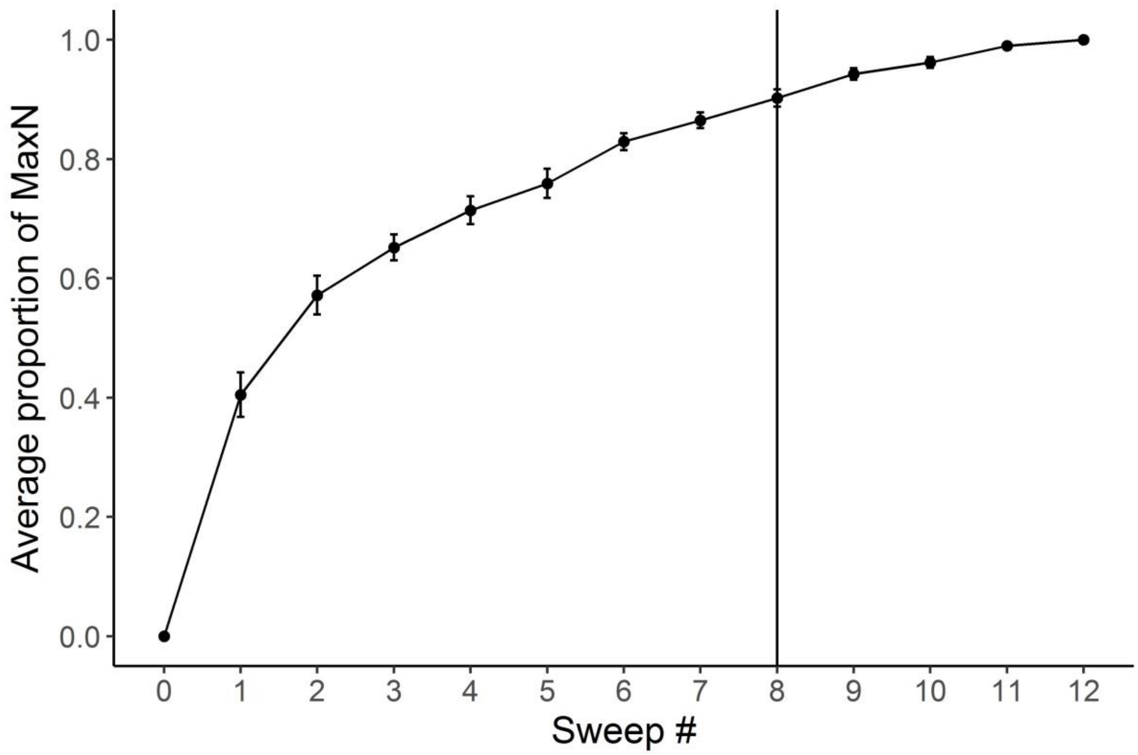

Figure 3: Average proportion of Maximum number (MaxN) for all species of interest achieved with increasing number of rotations. At eight minutes, approximately 85% of MaxN is achieved. A version of this figure specific to each species of interest can be found in Appendix A.

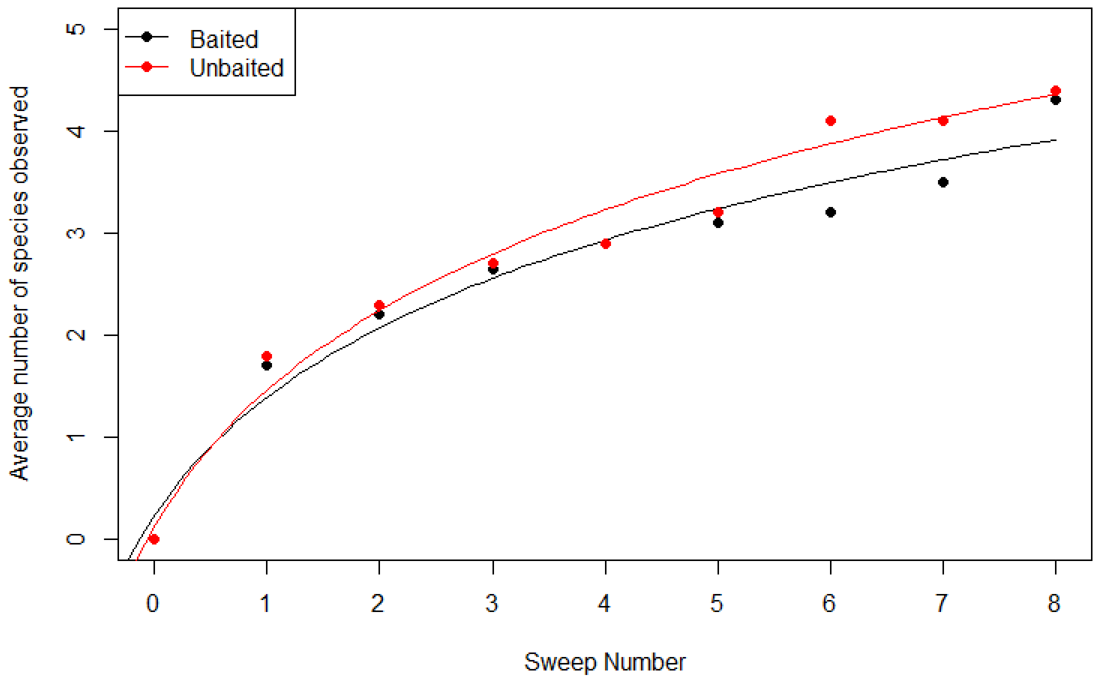

Figure 4: Average number of species observed per drop at increasing number of rotations for baited and unbaited drops. Best-fit lines are shown with a log fit.

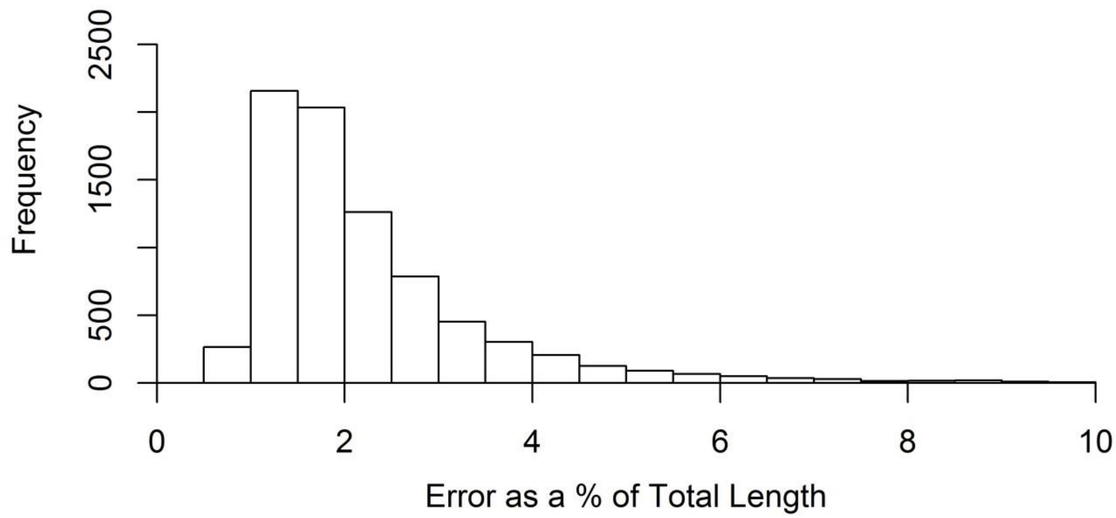

Figure 5: Histogram of errors as a percent of total length. Error determined by applying model from test tank measurements to measured fish in our database. Errors were trimmed at 10% of target length, only 46 lengths were trimmed. N = 7917. Nearly 60% of all measurements had a predicted error of less than 2% of body length.

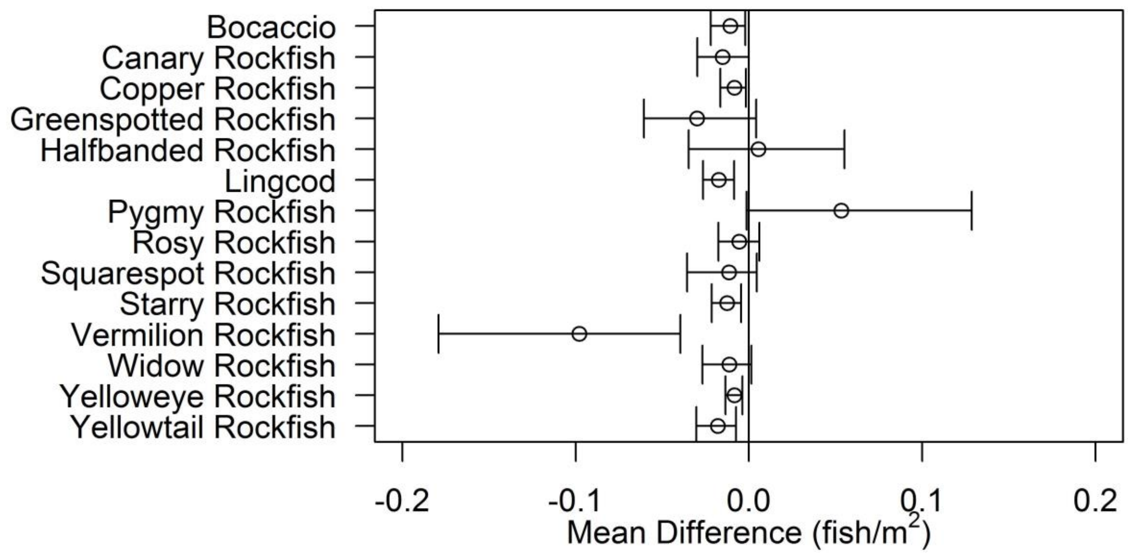

Figure 6: Bootstrapped mean differences between stationary and rotating methodologies with 95% CI bars. Positive values indicate a greater rotating camera density; negative values indicate a greater stationary camera density.

Chapter 2: Comparison between two different remote visual tools: a rotating stereo-video Lander and a remotely operated vehicle (ROV)

Introduction

Submersibles and remotely operated vehicles (ROVs) have historically been used to visually survey deep, rocky, benthic habitats, and have often been used to ground truth other techniques such as acoustic surveys or diver transects (Greene and Alevizon 1989, Starr et al. 1996, Nasby-Lucas et al. 2002). Both techniques also have been used to ascertain small scale habitat associations for deep-water benthic fishes (Yoklavich et al. 2000, 2002, Parry et al. 2003, Anderson and Yoklavich 2007, Anderson et al. 2009, Lindholm et al. 2015) and to estimate the relative abundance of fish and invertebrate populations (Love et al. 1994, Yoklavich et al. 2002, 2007, Johnson et al. 2003, Pinkard et al. 2005). However, it can be extremely expensive and logistically complex to operate these survey tools across wide geographic ranges.

Single and stereo-video drop-camera systems can represent a cheaper and more logistically simple alternative to ROVs and submersibles, especially for surveying rocky or complex-bottom habitats (Ellis and DeMartini 1995, Willis et al. 2000a, Hannah and Blume 2012, Easton et al. 2015). Stereo video systems allow accurate size determination of fish without collecting the individuals (Harvey et al. 2002, Colton and Swearer 2010, Lowry et al. 2012, Hannah and Blume 2016).

We designed a stereo-video Lander in an attempt to create a tool that is capable of surveying these habitats quickly and inexpensively. When developing a new tool, it can be useful to know how it compares with older tools used for similar tasks. Video Landers have previously been compared with diver surveys (Colton and Swearer 2010, Lowry et al. 2012) but have not been extensively compared with other video survey techniques such as ROVs or human occupied vehicles (HOVs). As an unmanned video-survey tool meant for rocky-bottom habitats, ROVs are perhaps the closest existing analogue to the video Lander, but have several important methodological differences such as typically being used in strip-transect surveys as opposed to single-point surveys.

As a visual survey tool, ROVs are able to survey a larger area during a single transect than a video Lander would survey during a particular survey. Additionally, as a mobile tool, they have the ability to navigate to specific features of interest and around obstacles. However these advantages come with several tradeoffs. ROVs require skilled pilots to fly them, especially over complex terrain, and often require relatively large vessels as research platforms. Video Landers, on the other hand, can be logistically simple to operate and do not require a particularly skilled deck crew to manage. Video Landers also can be launched off of nearly any vessel with a winch and block arrangement, allowing for the use of smaller, cheaper vessels. However, the downsides of video Landers are that they tend to survey smaller areas and are unable to precisely target specific seafloor features to survey, although many more independent replicate surveys can often be completed for the same sampling effort.

Given these differences and potential trade-offs, my objective was to compare and contrast fish assemblage data collected by a video Lander and a ROV in the same area and at nearly the same time. I used a video Lander described by Starr et al. (2016) and the Beagle ROV owned by Marine Applied Research and Exploration (MARE) in two locations in Monterey Bay to survey the same areas at the same time of year. I chose three primary indicators to compare the two tools: fish density, fish length distributions, and fish community composition. Because these two tools are both used for visual surveys and they were deployed in the same locations at the same time of year, I expected that the differences between them would be minor and non-significant. However, I did expect that the methodological differences would reveal situations or questions for which one tool or the other was better suited.

Methods

Tools: The video Lander consisted of an aluminum frame 1.5 m tall, 1 m in diameter, and weighed 45 kg, with 70 kg of lead weight added during operations. The frame housed two Deep Sea Power and Light (DSPL) cameras with 620 TV line (TVL) resolution. The cameras were mounted on a rotating bar, with both cameras facing out from the Lander in the same direction, with each camera turned in towards the center of the bar approximately 5 degrees from pointing straight ahead. Additionally, two DSPL LED lights outputting 3000 lumens at a color temperature of 5000K were mounted above the cameras. Finally, the frame contained an electronics bottle that housed the video recording devices (two Stack LTD™ DVRs with removable 32GB storage cards) and the electronics necessary to provide commands from the surface to the video Lander on the bottom. The electronics bottle was mounted behind and above the rotating bar. The video Lander was deployed from the deck of a fishing vessel using the main winch and was supported by a high-tensile-strength line. Additionally, the video Lander used an umbilical that supplied power and allowed data transfer up and down from the electronics bottle. This umbilical was rated to support the weight of the video Lander in the case of an emergency.

The Beagle ROV was 204 kg and 1.5 m x 0.75 m x 0.75 m (LxHxW). It had a forward facing standard definition (SD) camera for navigation and is equipped with the same DSPL cameras for identification and sizing as the video Lander, mounted facing forward. The ROV carried two 200 W dimmable Nuytco hydrargyrum medium-arc iodide (HMI) lights. The vehicle was capable of 82 kg of forward thrust, 30 kg of vertical thrust, and 18 kg of lateral thrust. It operated on 8 kW of 220 volt AC power transferred through the umbilical. It was rated to 1000 m of depth, dependent on umbilical length. It was controlled from a converted communications van that weighed ~900 kg.

Study site and field operations: I used the video Lander and ROV Beagle to survey two sites in Monterey Bay: Portuguese Ledge (~36.6°N, 121.9°W) and a section of the shelf break just south of the Monterey submarine canyon (~36.7°N, 121.9°W) (Fig. 1). These two sites contained a range of different rocky habitats that were representative of hard bottom areas along the west coast of the US, including boulder and cobble fields, bedrock, and rock ridges. At each of these two sites, four target lines were designated. Three ROV transects and between 3 and 5 Lander deployments were conducted along each target line. I performed 24 Lander deployments and 31 ROV strip transects along the eight target lines designated for the Lander-ROV comparison study. ROV transects were 2 m wide strip transects, the length being determined by the length of the target line, usually ~1 km. A Lander deployment was defined as being the time between when the Lander was deployed off the boat and recovered. During a single deployment, typically 4 – 6 video surveys were conducted along the target line. These surveys were referred to as “drops”. Each drop consisted of the Lander resting on the seafloor and performing eight full rotations with each rotation taking one minute to complete. Rotations were counted after the sediment had cleared from touchdown on the bottom, typically 1-2 minutes. Each drop was at least 50 m away from the previous drop.

Data collection: Data were post-processed in the lab for habitat features and fish counts. The ROV video was analyzed as a continuous video transect. Fish were recorded in the middle of the frame of the forward facing video cameras. Each fish was marked with the time code when it was observed. Because of the greater number of fish observed, most species were measured with paired lasers.

Habitats were time stamped with the time code at which they began and ended in the video. These time codes were matched with fish time codes to associate fish with the habitat. Habitats were similarly identified by the Greene et al. (1999) habitat categories to record the primary and secondary habitat types. The area surveyed by a single ROV replicate is the entire length of a transect in a 1 m swath. The area surveyed on a single ROV transect was usually ~1000 m^2.

For the Lander video, every fish was identified to the lowest identifiable taxonomic group. Fish for which identification was questionable were marked for review by another reviewer and identified according to the consensus decision. Eight full rotations were watched, with every fish marked in every rotation. For fish identified down to species, the rotation with the greatest number of individuals observed (MaxN) was the only rotation counted. Each fish on the rotation containing the MaxN value was measured with the stereo video cameras. To maximize the ability to assign length estimates, the video was rewound or advanced to find the frame most amenable to stereo length measurement. This measurement provided an estimate of length, distance from the cameras, and angle relative to the cameras. Individual fish for which it was not possible to determine length (typically because the entire body was not visible) were marked with a single point in both cameras to estimate the distance from the cameras for use in the maximum reliable detection distance (the distance within which 95% of individuals of a given species were observed across all surveys).

Habitat was recorded using the 60%/40% two letter code method described by Greene et al. (1999) in which the dominant and secondary habitat types were recorded based on percentage of area covered by each habitat type. Habitat codes were appended to the record of each fish. The area surveyed was the summed aggregate of the doughnut shapes between the minimum detection distance (the nearest distance that both cameras could see the bottom) and the maximum reliable detection distance (the species-specific distance at which 95% of individuals were identified) from all drops on a deployment. Depending on the species, the area surveyed by a deployment was usually between 100 and 200 m^2, whereas the area surveyed by any individual drop was between 20 and 40 m^2, depending on species.

Community composition: To compare the community compositions determined by the Lander and the ROV, I created a non-metric multidimensional scaling (nMDS) plot of square-root transformed aggregated-species densities, with each individual survey (deployment for the Lander, transect for the ROV) being a sample in Primer (PRIMER v6). Additionally, I created an nMDS plot based on a Bray-Curtis Similarity matrix to compare how similar or different the aggregated densities were between the two tools. Also, I performed a SIMPER analysis to determine which species affected differences among the groups. To determine whether the factors of site and tool were significant in determining community level differences, I performed a PERMANOVA test on site and tool.

Fish density: To compare fish densities between the two tools, all the Lander drops from a given deployment were combined into a single survey. Thus, fish observations and area surveyed were summed; this resulted in a single density estimate for each species on each survey for each tool (i.e., 4 Lander deployments and three ROV transects). Also, I analyzed the fish densities between the two tools grouped by benthic habitat type, with habitats being defined as “hard” or “soft”. Hard habitats consisted of rock ridge, boulder, cobble, and mixed substrates, whereas soft substrates consisted of sand or mud with >60% coverage of the bottom (Greene et al. 1999).

I compared species level densities using a bootstrapped mean difference values from the target lines. Bootstrapping was conducted by selecting a random sample of deployment level densities (for the Lander) and transect level densities (for the ROV) for a given species. The number of randomly selected samples was determined by the number of surveys conducted by each tool (n_{Lander} = 24, n_{ROV} = 31). The mean density of the ROV samples was subtracted from the mean density of the Lander samples, producing a density difference value in which positive values represent greater Lander densities and negative values represent greater ROV densities. This sampling procedure was performed 1000 times for each species and a mean difference value and 95% confidence intervals were calculated from these replicate samples. Total fish densities on hard and soft habitat types were compared using Welch's t-tests as the variances between the ROV and Lander samples were non-homogenous.

Variability and sampling effort: To compare how well each tool captured the variability present in the fish populations and how much sampling effort each tool required to make an estimate of variability, I calculated the precision (standard error/mean) and coefficient of variation (CV) for each deployment or transect and bootstrapped the values across the sampling effort spectrum from a minimum of three samples (transects for the ROV, deployments for the Lander) up to the maximum available number of samples (31 transects for the ROV, 24 deployments for the Lander) and graphed the mean values with 95% confidence intervals according to number of samples conducted.

***Mean length*:** Length of fish determined using the two tools was collected in slightly different ways. Because of the increased amount of video collected by the ROV transects compared with the Lander drops and the corresponding increase in time to watch and collect the data in the lab, fish lengths from the ROV were calculated using paired lasers spaced 10 cm apart rather than the stereo-video cameras (which were slower but more precise). I determined lengths for three species (Canary Rockfish, Yelloweye Rockfish, and Cowcod) that were of particular interest using the paired-laser technique and the stereo-video technique. Fish lengths from the Lander data set were calculated using the stereo-video cameras and SeaGIS software. Because of this, differences in mean length could be attributed either to differences in the lengths of the fishes observed or to differences between the two techniques. To address this issue, mean length estimates for Yelloweye Rockfish, Cowcod, and Canary Rockfish were compared between the estimates of the paired laser and the stereo camera measurements. Because those three species were measured with both techniques on each fish in the ROV data, it was possible to test the impact of the two techniques on estimated fish length. Lengths for the rest of the species were compared between the stereo measurements from the Lander and the paired laser measurements from the

ROV using Welch's t-tests because variance between the two samples was non-homogenous in most cases.

Results

Summary statistics: The Lander data consisted of 24 deployments on the eight target lines, with observations of 1,785 fish of 28 species, and an average fish density of 2.92 +/- 0.41 fish/m^2 (Table 2, mean +/- standard error). The ROV data consisted of 31 transects on the eight target lines and observations of 12,537 fish of 25 species, and an average fish density of 2.48 +/- 0.47 fish/m^2 (Table 1, mean +/- standard error).

Community composition: The nMDS plot indicated clustering both by tool and by site (Fig. 2). Site (Bay Shelf (BS) and Portuguese Ledge (PL)) cluster along the first axis whereas the two tools separate out slightly less clearly along the second axis. Lander samples were an average of 48.3% similar to each other, and most of the similarity in Lander samples was driven by Copper Rockfish, Lingcod, Canary Rockfish, and Pygmy Rockfish (Table 3). ROV samples were an average of 49.5% similar to each other and most of the similarity in ROV samples was driven by Halfbanded Rockfish, Lingcod, Rosy Rockfish, and Greenspotted Rockfish (Table 4). The two tools were on average 56.2% dissimilar from each other, and differences between the two were primarily driven by Halfbanded Rockfish, Squarespot Rockfish, Pygmy Rockfish, and Canary Rockfish, of which Halfbanded and Squarespot Rockfishes were more abundant in the ROV surveys and Pygmy and Canary Rockfishes were more abundant in the Lander surveys (Table 5). Overall, more than 1/3 of the difference between the two groups was driven by schooling dwarf-rockfishes, although the relative abundances of the three rockfishes (Halfbanded, Squarespot, and Pygmy Rockfishes) indicated no consistent pattern.

Bay Shelf samples were on average 51.0% similar to each other, with most of the similarity being driven by Lingcod, Greenspotted Rockfish, Canary Rockfish, and Copper Rockfish (Table 6). Portuguese Ledge samples were on average 55.3% similar to each other with most of the similarity being driven by Rosy Rockfish, Pygmy Rockfish, Copper Rockfish, Yellowtail, and Halfbanded Rockfish (Table 7). The two sites had an average dissimilarity of 58.1%, with most of the difference driven by Halfbanded Rockfish, Squarespot Rockfish, Canary Rockfish, Pygmy Rockfish, Greenspotted Rockfish, and Bocaccio, which were all more abundant at Bay Shelf sites except Pygmy Rockfish (Table 8).

The PERMANOVA test indicated significant differences between site and tool respectively (Site: Pseudo-F_1 = 8.0222, P = 0.001; Tool: Psuedo-F_1 = 12.243, P = 0.001). Interaction between site and tool however was non-significant (Pseudo-F_1 = 1.3041, P = 0.235)

Fish density: There was no significant difference in estimates of the total fish density between the Lander and ROV either on hard habitat (Welch's t-test, $t_{39.711} = -1.629$, $p = 0.1112$) or soft habitat (Welch's t-test, $t_{33.094} = -1.1939$, $p = 0.241$). For the fourteen species of interest on hard substrate, bootstrapped 95% confidence intervals around the mean density difference show no significant differences (Fig. 3). Similarly, there were no significant differences on soft substrate (Fig. 3).

Variability and sampling effort: The CV values as a function of sampling effort for all four guilds, on hard and soft habitats, leveled off at between 5 and 10 samples for the Lander and the ROV (Figs. 4 & 5). Similarly, the improvements in precision diminished after 5 to 10 samples (Figs. 6 & 7). For all guild/habitat type combinations, the 95% confidence intervals overlap for the CV, except for the *Sebastomus spp*. guild on soft habitat. Dwarf Rockfishes on soft bottom indicated increased values for both metrics in those cases. In most other cases, although the difference was not significant, the Lander values were consistently less for CV and precision across the sampling effort range. On soft substrate, benthic solitary species observed with the ROV had the greatest CV whereas dwarf rockfishes observed on the ROV had the lowest CV. Semi-pelagic rockfishes and schooling dwarf-rockfishes had similar CV between the two tools, while benthic solitary species and Sebastomus species were the most different, although only Sebastomus species had non-overlapping confidence intervals (Fig. 4).

These patterns were similar for the estimates of precision among the four guilds (Fig. 6). On hard substrate, Sebastomus species observed by the ROV had the greatest CV and SE values while benthic solitary species observed by the video Lander had the lowest CV and SE. Of note is that the rate of change with increased sample size for both CV and SE was similar between both tools across all guilds and substrate types, with differences ocurring mostly in in the initial values.

Mean length: Over half of all tested species (9 of 15) had significantly different length distributions when compared with the KS test using Holm-Bonferroni corrected alpha levels (Table 9). For the three species for which paired laser and stereo-video measurements were collected in the ROV data (Canary Rockfish, Yelloweye Rockfish, and Vermilion Rockfish), the ROV stereo-video length distributions were significantly greater than the ROV paired laser measurements (KS $\text{test}_{\text{Canary}}$, D = 0.717, **p <<0.01**; KS $\text{test}_{\text{Yelloweye}}$, D = 0.348, p = <0.001; KS $\text{test}_{\text{Vermilion}}$, D = 0.75, p = 0.003), with the difference (stereo estimate – paired laser estimate) in mean length being 7 cm, 8 cm, and 8 cm for the three species, respectively. For each of the three species, the difference in mean length between the Lander stereo estimates and the ROV paired laser estimates was 6 cm, 11 cm, and 14 cm, respectively (Table 9). Stereo length distributions for Yelloweye Rockfish and Vermilion Rockfish collected with the Lander and the ROV data sets were not significantly different (KS $\text{test}_{\text{Yelloweye}}$, D = 0.319, p = 0.213; KS $\text{test}_{\text{Vermilion}}$, D = 0.474, p = 0.195). Canary Rockfish however did have significantly different length distributions between Lander and ROV stereo measurements (KS test, $D = 0.223$, $p < 0.001$) with the Lander lengths being greater.

Discussion

Community composition varied significantly between the two tools. Site had the larger effect on differences in community composition, but tool also was significant. The difference in community composition between sites was unsurprising as the Bay Shelf and Portuguese ledge sites have different geology with the latter having higher relief and more rock ridges compared to the low relief and boulder/cobble fields of the shelf. With these habitat differences, the main drivers of dissimilarity mostly match the expected relative abundances on the major habitat types, particularly the fact that Halfbanded, Squarespot, and Canary Rockfishes were all more abundant on the boulder and cobble fields of the Bay Shelf, whereas Pygmy Rockfishes were more abundant on the steeper rock ridges of Portuguese Ledge (Yoklavich et al. 2000). The difference in composition between the tools, however, was harder to explain. Mean total fish densities pooled across all species were not significantly different between the two tools, on either hard or soft habitat. Furthermore, the bootstrapped density differences indicated that only three species were significantly different in abundance between hard and soft bottom habitat. One clue as to the cause of the tool difference can be found by comparing the species account for much of the Lander-ROV differences with those which explain most of the site level differences. I found similar species in the top five positions of the SIMPER analysis (Tables 5 & 8). Even beyond similar species, the relative abundances of each tool matched the relative abundances of the two sites. Specifically, the relative abundances observed by the Lander for species driving the Lander-ROV differences were similar to the abundances of those species at Portuguese Ledge, and the relative abundances observed by the ROV were similar to those species at the Bay Shelf site. These patterns indicate that one possible explanation for the difference is differential ability of the tools on the two habitat types. It is possible that the Lander, which sets down in a single place, has fewer issues with navigating the complex topography found at Portuguese Ledge whereas the ROV does a better job over the relatively flatter and less complex structure of the Bay Shelf.

In the literature, estimates of species-specific density, as opposed to abundance, are rare, with even fewer coming from the central coast of California. Yoklavich et al. (2007) reported an average density of 0.00033 fish/m^2 for Cowcod in southern California which is less than the density I estimated with the Lander (0.0014 fish/m^2, Table 2) and the estimate made with the ROV (0.0012 fish/m^2, Table 1). Starr et al. (1996) did not report by species but found a total fish density of 0.03 fish/m^2, working off the coast of

Oregon as compared with our study in which I estimated the total fish density using the ROV to be 0.730 fish/m^2 and the total density of fish for the Lander to be 1.441 fish/m^2. Meanwhile Nasby-Lucas et al. (2002), also working off the central Oregon coast, reported densities for three of our species of interest: Pygmy Rockfish, Lingcod, and Yellowtail Rockfish. On hard substrates, they reported a density of 0.0003 fish/m^2(Pygmy Rockfish), 0.000001 fish/m^2(Lingcod), and 0.00002 fish/m^2(Yellowtail Rockfish). I estimated 0.02 fish/m^2(Pygmy Rockfish), 0.0001 fish/m^2(Lingcod), and 0.008 fish/m^2(Yellowtail Rockfish) with the ROV and 0.07 fish/m^2(Pygmy Rockfish), 0.03 fish/m^2(Lingcod), 0.01 fish/m^2(Yellowtail Rockfish) with the Lander. Although there is an obvious large difference in the densities found in our survey and those of earlier submersible surveys of fish density, it is difficult to ascribe a cause to the differences. These surveys were taken hundreds of kilometers apart and separated by well over a decade, with both of these surveys being conducted near the low point in rockfish populations post-overfishing (Love et al. 1998).

The graphs of precision and CV plotted against sample size were perhaps the most important comparison between the two tools. Although mean density was not significantly different between the two tools, there was a difference in how variability changed with sampling effort. Although it appears that variability is well estimated after a similar number of samples for both tools, the Lander can achieve that number of samples in a lesser amount of time at sea. This is because a single Lander deployment takes far less time than the equivalent sample unit for the ROV (transects). The tradeoff is that an ROV transect covers a given area in greater detail. For a targeted study on a particular type of habitat in an area, which has high quality benthic maps, the video Lander is likely to collect similar data to the ROV in a lesser amount of time. However, in a study that is less targeted to a particular benthic type, or for which high-resolution benthic maps are lacking, the ability of the ROV to survey a greater area will probably produce better results relative to pinpoint sampling of the Lander or other stationary video techniques.

In addition to the increased time in the field to collect the data, the increase in area covered by the ROV leads to a corresponding increase in number of fish observed and post-processing time. It is this difference that finally gives us an idea of how I might choose between the two when designing a new project. The ROV provides better broad scale coverage of a study site, covering more area and recording greater numbers of fish, at the expense of increased time both in the field and in post-processing. The video Lander can provide fast, targeted surveys of field sites and offers decreased post-processing times.

Fish lengths estimated using the video Lander were significantly greater than those measured by the ROV. Although it is possible that larger fish were observed on the Lander surveys, maybe as a result of quieter operation due to the absence of thrusters or the fact that it is stationary, it is more likely that differences were due to the methods used to measure fish lengths from each tool. Whereas the ROV was equipped with a stereo-video system, most fish lengths were estimated using the parallel lasers mounted on the vehicle. Video analysts chose to provide a coarse estimate of fish lengths from ROV data because of the logistical difficulties associated with the greater number of fish observed by the ROV (nearly an order of magnitude more observations with the ROV than with the Lander). For three species, however, ROV video analysts measured fish using both paired laser and stereo-video measurements. The results of those comparisons indicated that the magnitude of the differences in length estimates were similar to those observed between the Lander stereo-video and the ROV paired laser for all species. One possible explanation for this discrepancy is that video analysts had trouble extrapolating the larger fish lengths accurately using the paired lasers which were only 10 cm apart. This could explain why larger species were more different and smaller species more similar between the two tools (Table 6). Further analyses of differences in length estimates using paired-laser/stereo-video length have indicated no directional bias between techniques. Paired-laser estimates for individual fish did result in increased variability (length estimates from paired lasers ranging from ~20 cm over to ~ 20 cm under the stereo-video estimates) with the magnitude of the variability greatly dependent on species (Kline, Shrestha, Starr, unpublished data 2016). Despite these large variances, ~70% of measurements were within 5 cm of the stereo-video estimates. Kline et al. (2016) described significant differences among mean lengths of seven of ten species examined using the two techniques. I found significant differences among lengths of thirteen out of fourteen species when comparing paired laser and stereo video measurements. Where the tested species were the same (Copper Rockfish, Canary Rockfish, Yellowtail Rockfish, Vermilion Rockfish, and Yelloweye Rockfish), my data and those of Kline et al. (2016) indicated mean lengths calculated from the stereo-video measurements were greater than those calculated by the paired-lasers, although magnitudes varied widely. In the future, to deal with the difficulties of measuring large numbers of fish with stereo video tools, a subsample of the fish could be measured using the stereo cameras as opposed to attempting to measure every observed fish.

In my comparison of the two tools, there were a few metrics of individual species that were significantly different from one tool to another, but there was no pattern of particular guilds or fish with similar characteristics being different between the two. I did find a significant difference in the lengths collected for several species, but these differences were likely due to the fact that the ROV lengths were largely determined with paired laser estimates. Sizes of species that were determined using with stereo-video measurements from both tools indicated no significant differences between the two. Community composition did vary significantly between the two tools. These differences were largely driven by dwarf, schooling species, which were seen in greater numbers on the ROV. And although total community composition did differ, the site variability was similar between the two tools, suggesting that although the overall communities were different, data from the two tools indicated similar changes in those communities when going from one site to the next. These differences between the ROV and Lander emphasize the importance of having a wide variety of survey tools and techniques available. However, it is just as important to have good comparative studies between those techniques to understand how they compare and under which circumstances a particular tool will be the correct choice. It is also likely that for large-scale studies over a wide range of benthic habitats, that a combination of several tool types will produce the most accurate estimates of population parameters.

Overall, data from the ROV and stereo-video Lander were broadly similar. Whereas there were significant differences in community composition and in the density of a few species, because of the lack of a clear pattern in these differences, it seems unlikely that either tool is better at estimating true population densities, but rather that each tool is more or less effective for particular species. As such, tool choice should be driven either on the basis of particular species of interest in the case of studies of individual species or on the basis of area covered and survey speed in the case of broad community surveys. Because the Lander has faster survey times, decreased logistical complexity, and decreased variability, it can more effectively cover large areas, particularly when studies are targeted at a particular habitat type and high quality maps are available to target said habitat type. The ROV is better suited to detailed surveys of a particular location where the question of interest is not oriented around a specific habitat type but rather the full range and variability of habitat at a site. Alternatively, the ROV is effective when a specific habitat is of interest but high quality maps of where that habitat occurs are unavailable and a large amount of searching on the bottom is necessary to find the features of interest. Understanding the specifics of how a tool operates and how those characteristics interact with the scientific questions is vital to making sure the most effective tool for a given study is used. I believe that the Lander represents a viable new tool that can operate in ways that make it better suited than existing tools for specific scientific questions and studies while providing data that are similarly representative of the populations of interest.

Chapter 2 Tables

Table 1: Number and density of observed species for the ROV across 31 transects at 8 target lines.

Common Name	Scientific Name	Number Observed	Mean Density (fish/100m^2)
Bocaccio	*Sebastes paucispinus*	222	4.27
Canary Rockfish	*Sebastes pinniger*	503	6.89
Copper Rockfish	*Sebastes caurinus*	254	2.10
Cowcod	*Sebastes levis*	34	0.12
Flag Rockfish	*Sebastes rubrivinctus*	73	0.23
Greenspotted Rockfish	*Sebastes chlorostictus*	441	3.75
Greenstriped Rockfish	*Sebastes elongatus*	114	0.47
Halfbanded Rockfish	*Sebastes semicinctus*	167	26.92
Lingcod	*Ophiodon elongatus*	609	3.97
Longnose skate	*Raja rhina*	3	0.01
Pink surfperch	*Zalembius rosaceus*	33	0.10
Pygmy Rockfish	*Sebastes wilsoni*	83	4.06
Rosy Rockfish	*Sebastes rosaceus*	545	4.34
Shortbelly Rockfish	*Sebastes jordani*	1	0.00
Speckled Rockfish	*Sebastes ovalis*	8	0.09
Spotted Ratfish	*Hydrolagus colliei*	13	0.10
Squarespot Rockfish	*Sebastes hopkinsi*	214	10.77
Starry Rockfish	*Sebastes constellatus*	165	0.71
Stripedfin Ronquil	*Rathbunella hypoplecta*	1	0.00
Stripetail Rockfish	*Sebastes saxicola*	1	0.00
Vermilion Rockfish	*Sebastes miniatus*	135	0.79
Widow Rockfish	*Sebastes entomelas*	14	1.23
Wolf Eel	*Anarrhichthys ocellatus*	10	0.03
Yelloweye Rockfish	*Sebastes ruberrimus*	144	1.31
Yellowtail Rockfish	*Sebastes flavidus*	164	0.73

Table 2: Number and density of observed species for the video Lander, across 24 deployments on 8 target lines.

Common Name	Scientific Name	Number Observed	Mean Density fish/100 m^2
Big Skate	*Raja binoculata*	1	0.05
Blackeye Goby	*Rhinogobiops nicholsii*	3	0.38
Bocaccio	*Sebastes paucispinus*	45	2.35
Canary Rockfish	*Sebastes pinniger*	350	24.76
Chilipepper	*Sebastes goodie*	1	0.14
Copper Rockfish	*Sebastes caurinus*	188	16.13
Cowcod	*Sebastes levis*	2	0.14
Flag Rockfish	*Sebastes rubrivinctus*	14	0.70
Greenspotted Rockfish	*Sebastes chlorostictus*	170	12.57
Greenstriped Rockfish	*Sebastes elongatus*	33	0.75
Halfbanded Rockfish	*Sebastes semicinctus*	200	18.57
Lingcod	*Ophiodon elongatus*	163	7.77
Pacific Hagfish	*Eptatretus stoutii*	17	0.70
Pacific Sanddab	*Citharichthys sordidus*	14	1.79
Pygmy Rockfish	*Sebastes wilsoni*	171	21.04
Rosethorn Rockfish	*Sebastes helvomaculatus*	2	0.26
Rosy Rockfish	*Sebastes rosaceus*	58	7.32
Shortbelly Rockfish	*Sebastes jordani*	14	2.05
Spotfin Sculpin	*Icelinus tenuis*	1	0.13
Spotted Ratfish	*Hydrolagus colliei*	2	0.10
Squarespot Rockfish	*Sebastes goodie*	71	8.40
Starry Rockfish	*Sebastes constellatus*	35	3.90
Swordspine Rockfish	*Sebastes ensifer*	2	0.26
Vermilion Rockfish	*Sebastes miniatus*	59	3.95
Widow Rockfish	*Sebastes entomelas*	74	4.38
Wolf-eel	*Anarrhichthys ocellatus*	2	0.10
Yelloweye Rockfish	*Sebastes ruberrimus*	31	2.12
Yellowtail Rockfish	*Sebastes flavidus*	62	3.27

Table 3: SIMPER analysis results for drivers of similarity in Lander deployments. Listed in order of largest driver of similarity to smallest driver of similarity. Av.Abund: average abundance across all surveys, based on square root transformed data. Av.Sim: average of the bray Curtis similarity value. Sim/SD: ratio of the similarity of each species divided by the SD of all contributions. Contrib%: percentage of total similarity contributed by the given species. Cum.%: cumulative Contrib%.

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Copper Rockfish	0.23	10.77	3.30	22.31	22.31
Canary Rockfish	0.23	5.79	0.96	12.01	34.32
Pygmy Rockfish	0.22	5.34	0.83	11.06	45.39
Lingcod	0.14	5.31	1.86	11.01	56.40
Greenspotted Rockfish	0.17	4.76	0.79	9.86	66.26
Halfbanded Rockfish	0.18	3.91	0.65	8.10	74.36
Rosy Rockfish	0.12	3.21	0.76	6.65	81.01
Vermilion Rockfish	0.09	2.59	0.87	5.37	86.39
Starry Rockfish	0.09	2.02	0.66	4.19	90.58

Table 4: SIMPER results for drivers of similarity between ROV transects. Listed in order of largest driver of similarity to smallest driver of similarity. Av.Sim: average of the bray Curtis similarity value. Sim/SD: ratio of the similarity of each species divided by the SD of all contributions. Contrib%: percentage of total similarity contributed by the given species. Cum.%: cumulative Contrib%.

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Halfbanded Rockfish	0.27	6.56	0.95	13.25	13.25
Lingcod	0.14	6.43	1.95	12.99	26.24
Greenspotted Rockfish	0.11	5.50	1.58	11.12	37.35
Rosy Rockfish	0.13	4.95	1.20	9.99	47.35
Squarespot Rockfish	0.24	4.78	1.01	9.65	57.00
Canary Rockfish	0.14	3.74	0.96	7.55	64.56
Copper Rockfish	0.09	3.52	1.63	7.11	71.67
Starry Rockfish	0.07	2.74	1.22	5.54	77.21
Pygmy Rockfish	0.10	2.63	0.58	5.31	82.52
Vermilion Rockfish	0.06	2.51	1.26	5.06	87.58
Bocaccio	0.13	2.45	0.89	4.94	92.52

Table 5: Similarity Percentage(SIMPER) results for drivers of dissimilarity between Lander deployments and Remotely Operated Vehicle (ROV) transects. Av.Sim: average of the bray Curtis similarity value. Sim/SD: ratio of the similarity of each species divided by the SD of all contributions. Contrib%: percentage of total similarity contributed by the given species. Cum.%: cumulative Contrib%.

Species	Mean Lander Abund.	Mean ROV Abund.	Mean Diss	Diss /SD	Contrib %	Cum.%
Halfbanded Rockfish	0.18	0.27	7.62	1.10	13.55	13.55
Squarespot Rockfish	0.10	0.24	6.26	0.97	11.14	24.69
Pygmy Rockfish	0.22	0.10	5.93	1.22	10.55	35.24
Canary Rockfish	0.23	0.14	5.85	1.23	10.40	45.64
Copper Rockfish	0.23	0.09	4.73	1.46	8.41	54.06
Greenspotted Rockfish	0.17	0.11	4.31	1.24	7.66	61.72
Bocaccio	0.06	0.13	3.49	0.98	6.21	67.93
Rosy Rockfish	0.12	0.13	3.33	1.28	5.92	73.85
Lingcod	0.14	0.14	2.77	1.29	4.93	78.79
Yellowtail Rockfish	0.07	0.05	2.58	1.02	4.59	83.38
Starry Rockfish	0.09	0.07	2.47	1.36	4.39	87.76
Widow Rockfish	0.07	0.02	2.45	0.58	4.37	92.13

Table 6: Similarity Percentage (SIMPER) results for drivers of similarity between Bay Shelf survey sites. Abund: average of the relative abundance of each speices. Sim: average of the bray Curtis similarity value. Sim/SD: ratio of the similarity of each species divided by the SD of all contributions. Contrib%: percentage of total similarity contributed by the given species. Cum.%: cumulative Contrib%.

Species	Abund	Sim	Sim/SD	Contrib %	Cum.%
Lingcod	0.2	8.1	3.07	15.89	15.89
Greenspotted Rockfish	0.18	7.06	1.62	13.84	29.73
Canary Rockfish	0.25	6.63	1.25	12.99	42.72
Copper Rockfish	0.17	5.47	1.54	10.72	53.44
Halfbanded Rockfish	0.24	4.84	0.96	9.49	62.92
Squarespot Rockfish	0.24	3.94	0.67	7.73	70.65
Yelloweye Rockfish	0.09	3.32	1.73	6.51	77.16
Bocaccio	0.15	2.45	0.72	4.80	81.96
Rosy Rockfish	0.11	2.03	0.73	3.99	85.95
Vermilion Rockfish	0.06	1.52	0.92	2.98	88.93
Starry Rockfish	0.06	1.46	0.86	2.87	91.8

Table 7: Similarity Percentage (SIMPER) results for drivers of similarity between Portuguese Ledge survey sites. Abund: average of the relative abundance of each speices. Sim: average of the bray Curtis similarity value. Sim/SD: ratio of the similarity of each species divided by the SD of all contributions. Contrib%: percentage of total similarity contributed by the given species. Cum.%: cumulative Contrib%.

Species	Abund	Sim	Sim/SD	Contrib%	Cum.%
Rosy Rockfish	0.15	7.72	2.02	13.95	13.95
Pygmy Rockfish	0.21	6.83	1.21	12.35	26.3
Copper Rockfish	0.14	5.96	1.52	10.78	37.08
Yellowtail Rockfish	0.13	5.06	1.37	9.14	46.21
Halfbanded Rockfish	0.21	4.96	0.73	8.96	55.17
Lingcod	0.08	3.73	2.12	6.74	61.91
Vermilion Rockfish	0.09	3.71	1.46	6.7	68.61
Starry Rockfish	0.09	3.43	1.2	6.2	74.81
Squarespot Rockfish	0.1	3.28	0.95	5.94	80.75
Greenspotted Rockfish	0.08	2.55	0.93	4.6	85.35
Canary Rockfish	0.09	2.40	0.75	4.34	89.69

Table 8: SIMPER analysis of dissimilarity between Bay Shelf and Portuguese Ledge sites. Av.Sim: average of the bray Curtis similarity value. Sim/SD: ratio of the similarity of each species divided by the SD of all contributions. Contrib%: percentage of total similarity contributed by the given species. Cum.%: cumulative Contrib%.

Species	Mean BS Abund.	Mean PL Abund.	Mean Diss	Diss /SD	Cont%	Cum.%
Halfbanded Rockfish	0.24	0.21	7.3	1.01	12.55	12.55
Squarespot Rockfish	0.24	0.1	6.02	0.88	10.35	22.9
Canary Rockfish	0.25	0.09	5.49	1.36	9.44	32.34
Pygmy Rockfish	0.1	0.21	4.95	1.12	8.52	40.87
Greenspotted Rockfish	0.18	0.08	3.66	1.19	6.3	47.16
Bocaccio	0.15	0.03	3.44	0.92	5.92	53.09
Lingcod	0.2	0.08	3.43	1.75	5.89	58.98
Yellowtail Rockfish	0.01	0.13	3.42	1.42	5.89	64.87
Rosy Rockfish	0.11	0.15	3.27	1.53	5.62	70.49
Yelloweye Rockfish	0.09	0.01	2.26	1.60	3.89	74.37
Copper Rockfish	0.17	0.14	2.24	1.50	3.86	78.23
Widow Rockfish	0.07	0.02	1.99	0.54	3.43	81.66

Table 9: Differences in mean length and mean tested p-values for species of interest between the Lander and the ROV. The Lander used stereo video measurements whereas the ROV used paired laser measurements. Three species were measured on the ROV with both techniques. Comparisons of fish lengths from those three species can be found in the Ch. 2 figures section. Komogorov-Smirnov tests were used to determine whether length distributions were significantly different. Significant differences are bolded. The alpha level was determined by a Holm-Bonferroni adjustment.

Species	Lander (cm)	ROV (cm)	Diff. (cm)	K-S test D_{val}	P-value
Bocaccio	47	44	3	0.203	0.39
Canary Rockfish	40	33	7	0.731	**<<0.01**
Copper Rockfish	43	33	10	0.660	**<<0.01**
Greenspotted Rockfish	29	22	7	0.486	**<<0.01**
Greenstriped Rockfish	21	19	2	0.265	0.152
Halfbanded Rockfish	9	10	-1	0.402	**<<0.01**
Lingcod	51	45	6	0.336	**<<0.01**
Pygmy Rockfish	10	11	-1	0.410	**<<0.01**
Rosy Rockfish	20	16	4	0.388	**<<0.01**
Squarespot Rockfish	16	14	2	0.436	<0.01
Starry Rockfish	28	21	7	0.625	**<<0.01**
Vermilion Rockfish	45	34	11	0.833	**<<0.01**
Widow Rockfish	31	25	6	0.544	0.01
Yelloweye Rockfish	39	25	14	0.535	<0.01
Yellowtail Rockfish	34	36	-2	0.356	<0.01

Chapter 2 Figures

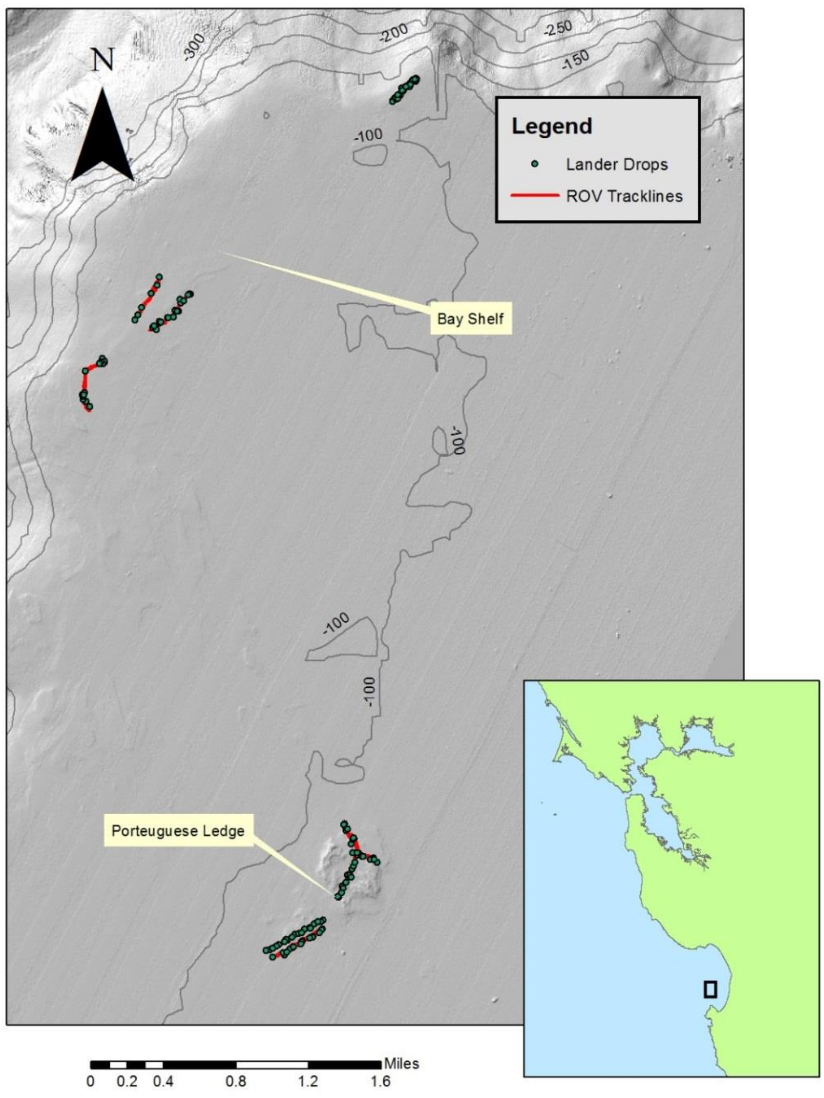

Figure 1: Map of Lander-ROV locations for comparison field surveys. These surveys were conducted in June of 2014. Portuguese ledge surveys were the four target lines near the Portuguese ledge feature whereas the Bay Shelf surveys consisted of the four target lines spread along the shelf edge.

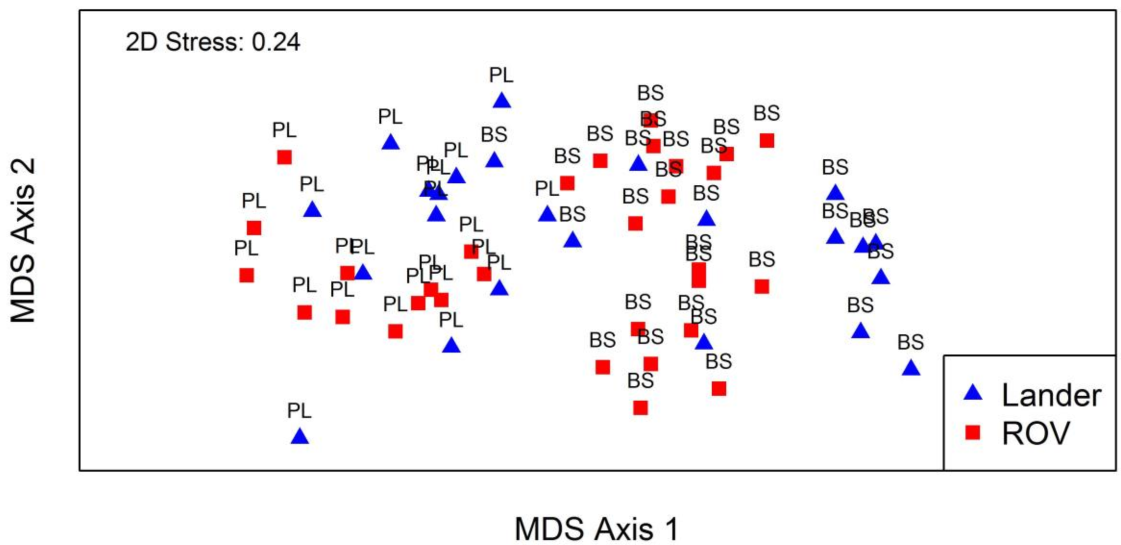

Figure 2: nMDS plot of community composition based on relative species densities (square root transformed) between Lander and ROV. Each point is a particular transect (ROV) or deployment (Lander). Points are labeled by which site they occurred at (PL: Portuguese Ledge, BS: Bay Shelf).

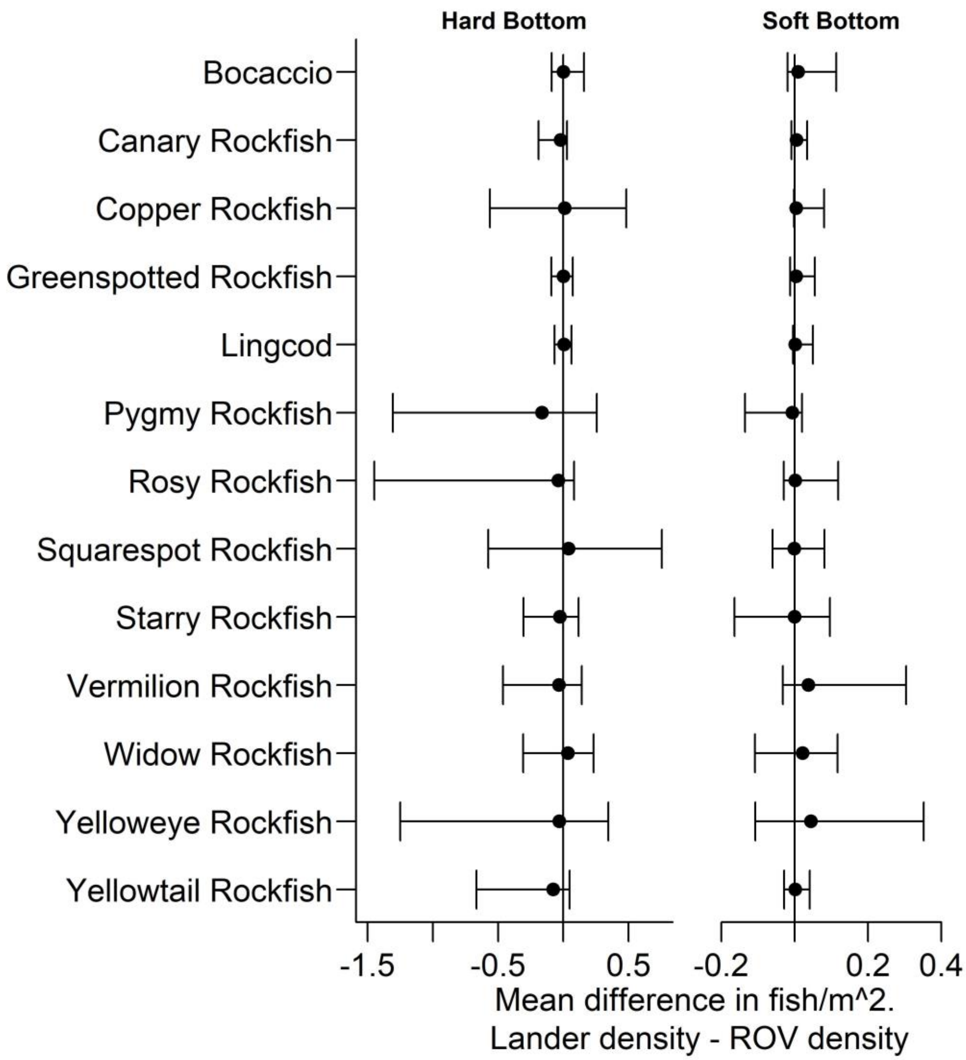

Figure 3: Bootstrapped mean difference between Lander and ROV density values by species for the 14 species of interest. Calculated by subtracting ROV density from Lander density so positive values indicate greater Lander density, 95% CI bars shown.

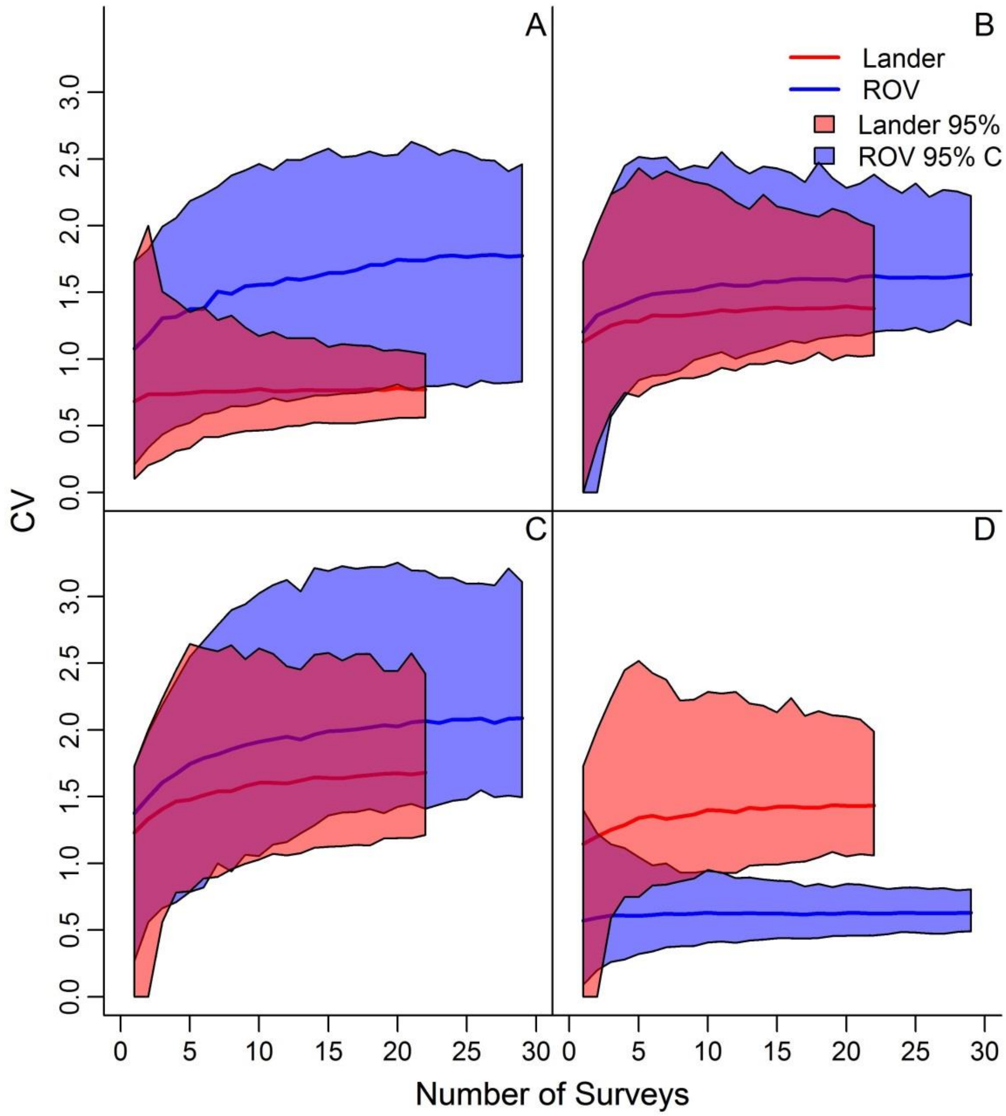

Figure 4: Soft Bottom. Bootstrapped CV as a function of sample size with 95% CI shaded is shown for each of the four species guilds. A) Benthic Solitary species, B) Semi-Pelagic rockfishes, C) Schooling Dwarf rockfishes, and D) Sebastomus spp. Benthic Solitary guild consists of Lingcod, Copper, Vermilion, and Yelloweye Rockfishes. Sebastomus guild consists of Greenspotted, Rosy, and Starry Rockfishes. Semi-Pelagic guild consists of Widow, Bocaccio, Yellowtail, and Canary Rockfishes. Dwarf Rockfish guild consists of Halfbanded, Squarespot, and Pygmy Rockfishes.

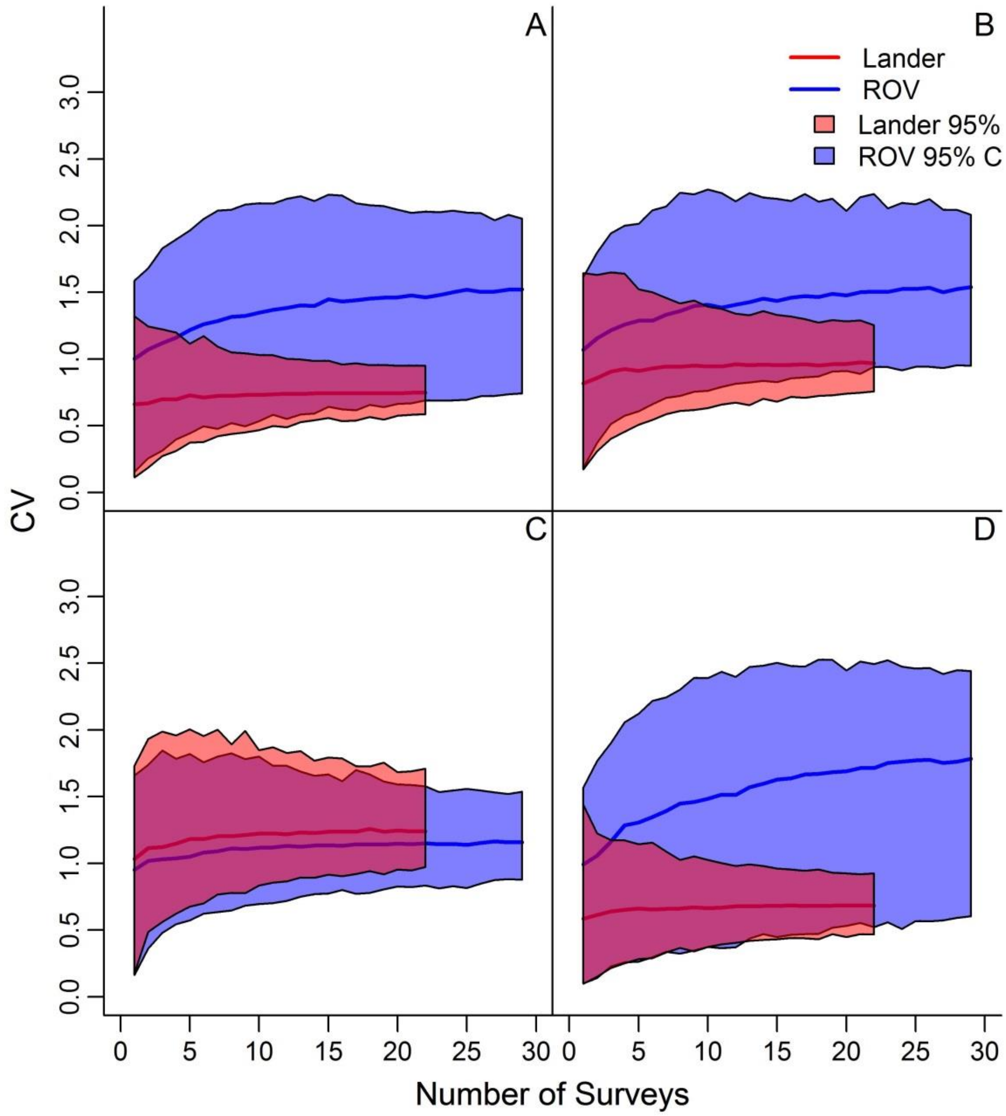

Figure 5: Hard Bottom. Bootstrapped CV as a function of sample size with 95% CI shaded is shown for each of the four species guilds. A) Benthic Solitary species, B) Semi-Pelagic rockfishes, C) Schooling Dwarf rockfishes, and D) Sebastomus spp. Benthic Solitary guild consists of Lingcod, Copper, Vermilion, and Yelloweye Rockfishes. Sebastomus guild consists of Greenspotted, Rosy, and Starry Rockfishes. Semi-Pelagic guild consists of Widow, Bocaccio, Yellowtail, and Canary Rockfishes. Dwarf Rockfish guild consists of Halfbanded, Squarespot, and Pygmy Rockfishes.

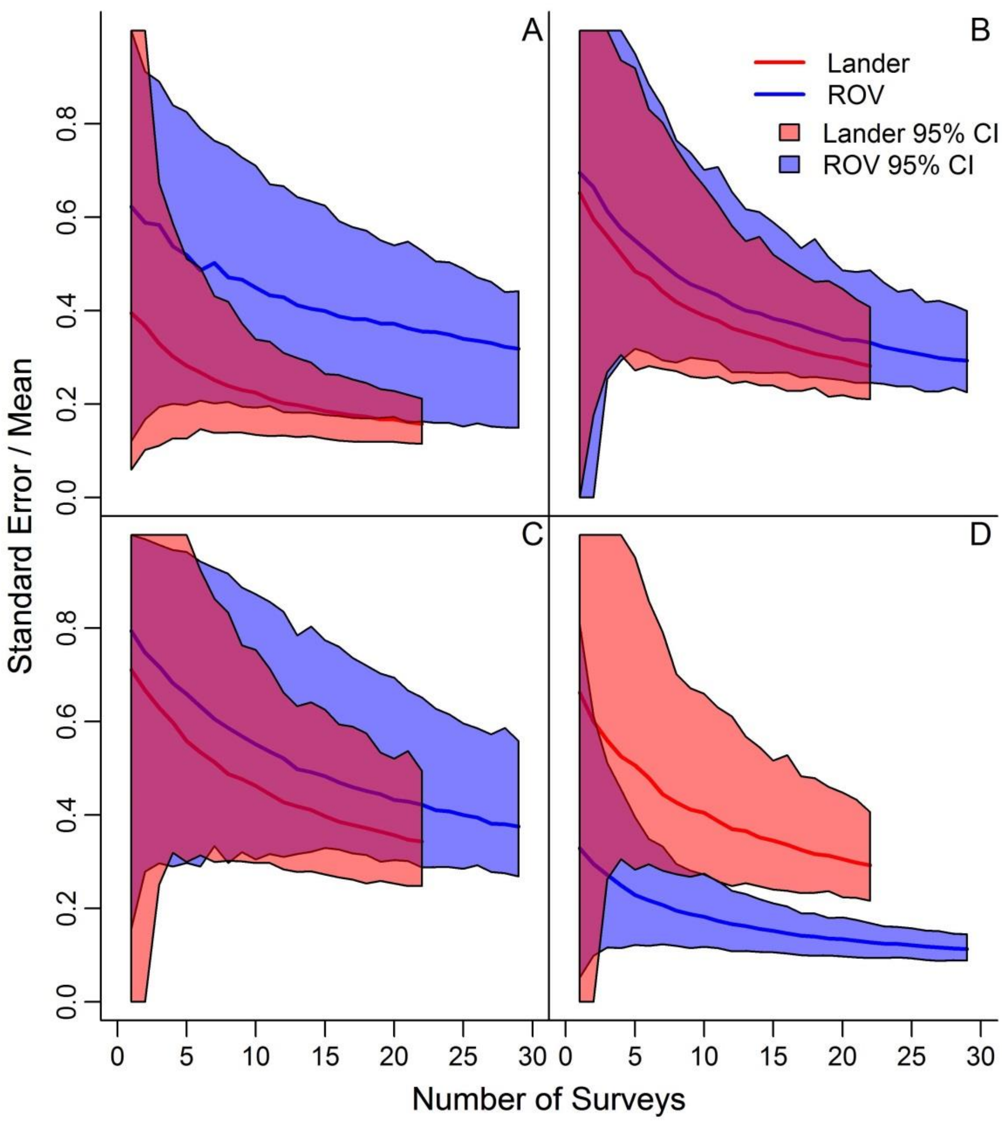

Figure 6: Soft Bottom. Bootstrapped Precision as a function of sample size with 95% CI shaded is shown for each of the four species guilds. A) Benthic Solitary species, B) Semi-Pelagic rockfishes, C) Schooling Dwarf rockfishes, and D) Sebastomus spp. Benthic Solitary guild consists of Lingcod, Copper, Vermilion, and Yelloweye Rockfishes. Sebastomus guild consists of Greenspotted, Rosy, and Starry Rockfishes. Semi-Pelagic guild consists of Widow, Bocaccio, Yellowtail, and Canary Rockfishes. Dwarf Rockfish guild consists of Halfbanded, Squarespot, and Pygmy Rockfishes.

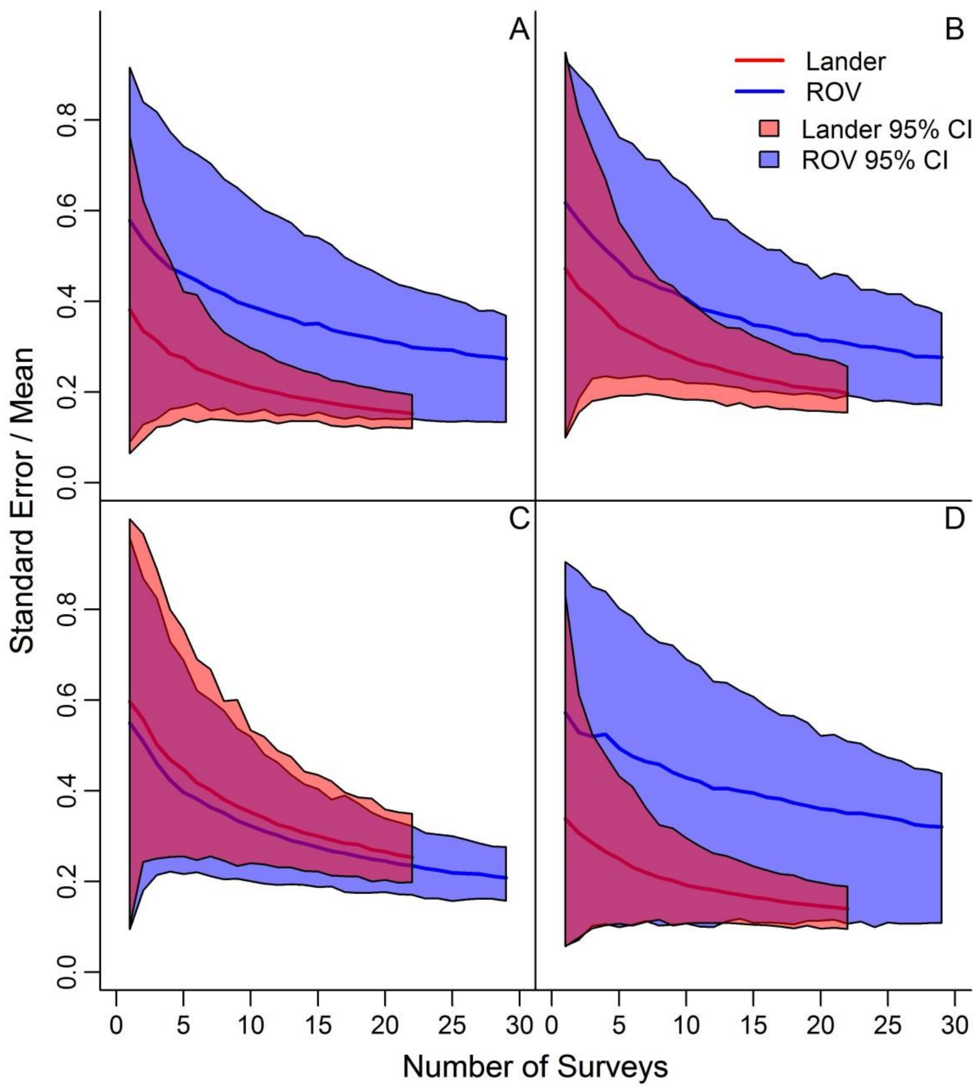

Figure 7: Hard Bottom Bootstrapped standard error divided by the mean (SE/Mean) as a function of sample size with 95% CI shaded is shown for each of the four species guilds. A) Benthic Solitary species, B) Semi-Pelagic rockfishes, C) Schooling Dwarf rockfishes, and D) Sebastomus spp. Benthic Solitary guild consists of Lingcod, Copper, Vermilion, and Yelloweye Rockfishes. Sebastomus guild consists of Greenspotted, Rosy, and Starry Rockfishes. Semi-Pelagic guild consists of Widow, Bocaccio, Yellowtail, and Canary Rockfishes. Dwarf Rockfish guild consists of Halfbanded, Squarespot, and Pygmy Rockfishes.

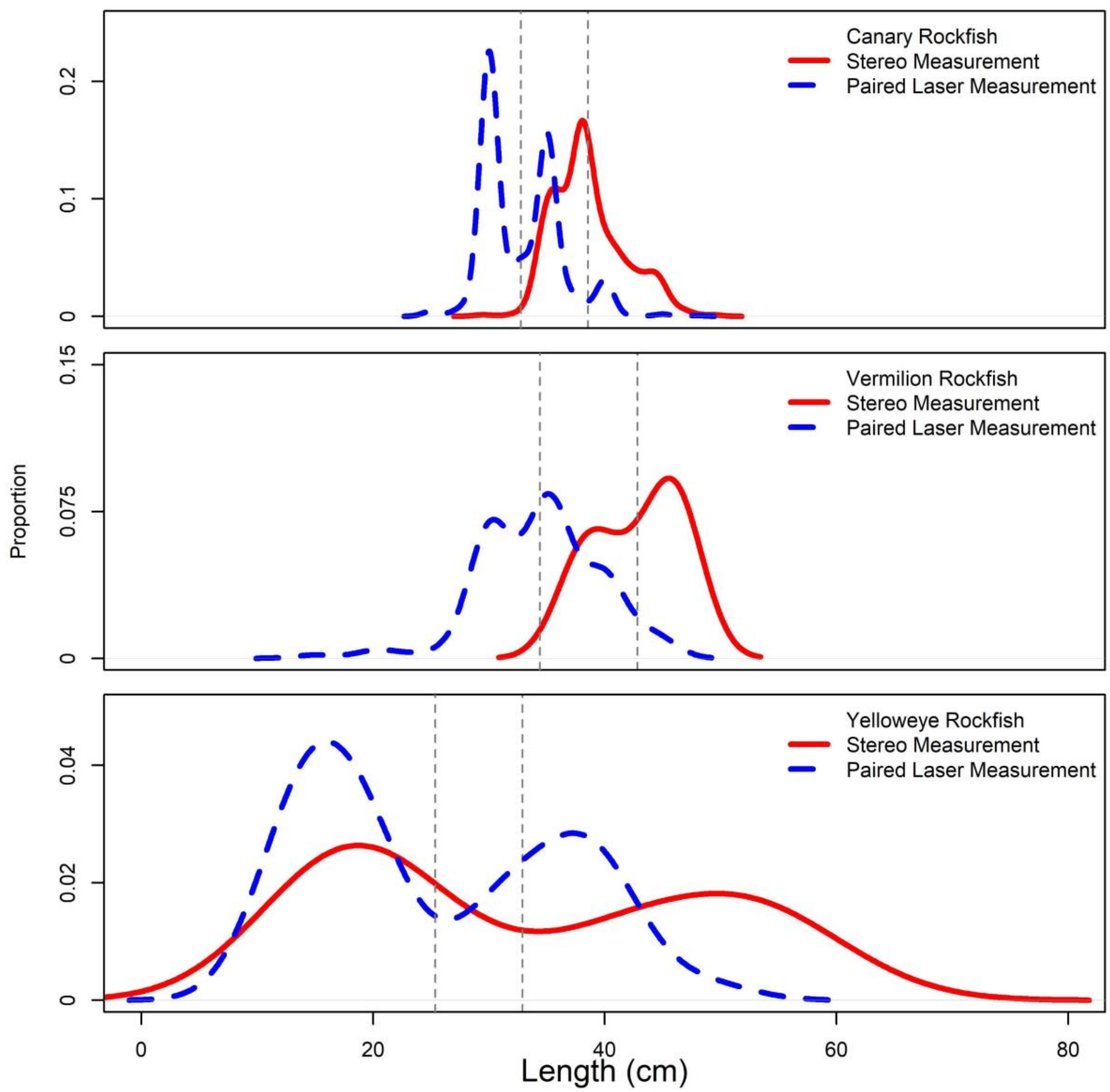

Figure 8: Length distributions of the three species for which stereo measurements and paired laser measurements were recorded. Each fish was measured with both techniques, so these distributions represent the same fish. Red is the stereo video estimate, blue is the paired laser estimate. Vertical lines represent the mean estimate.

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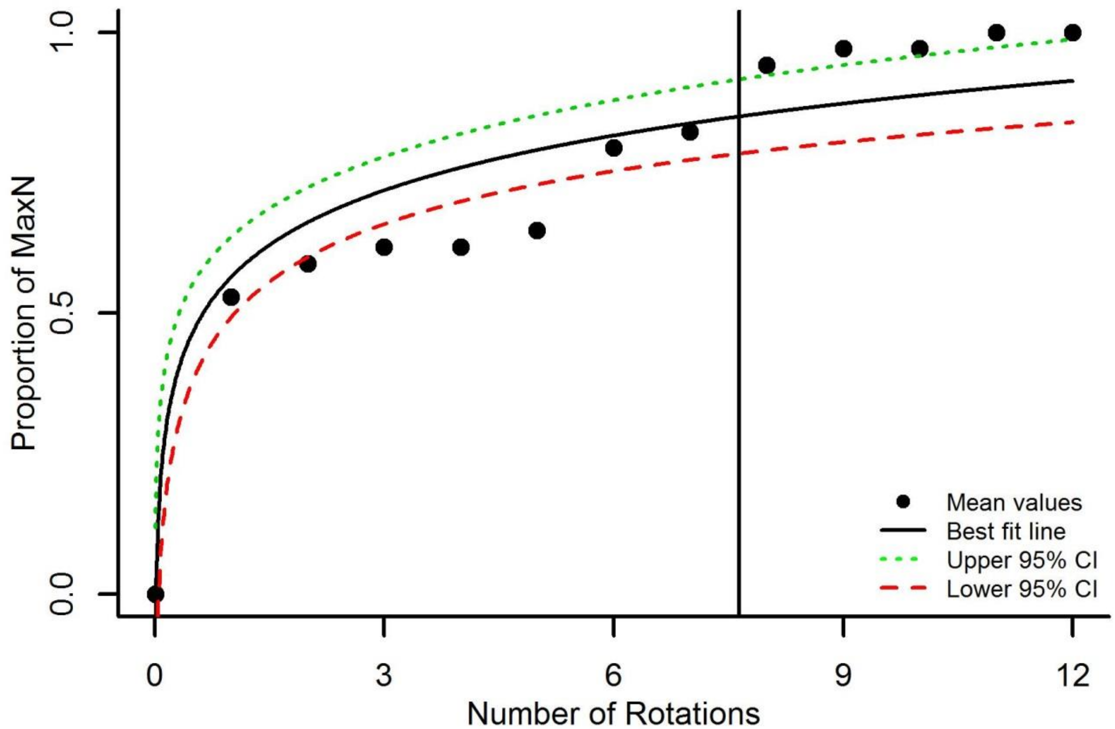
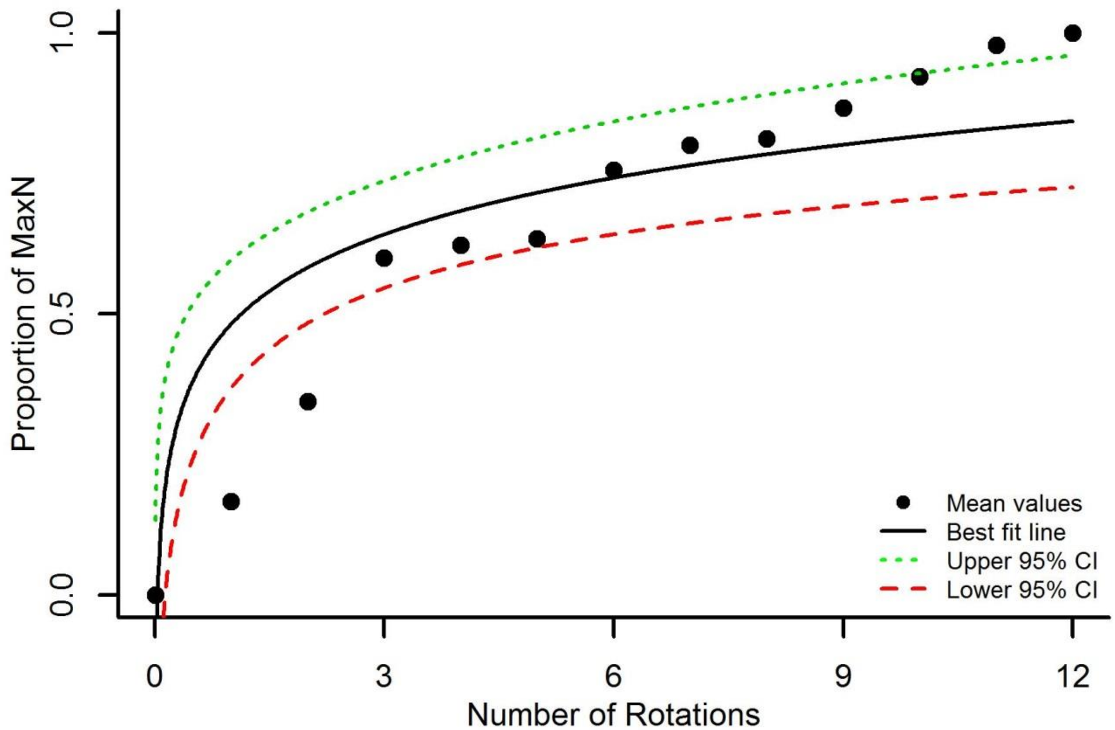
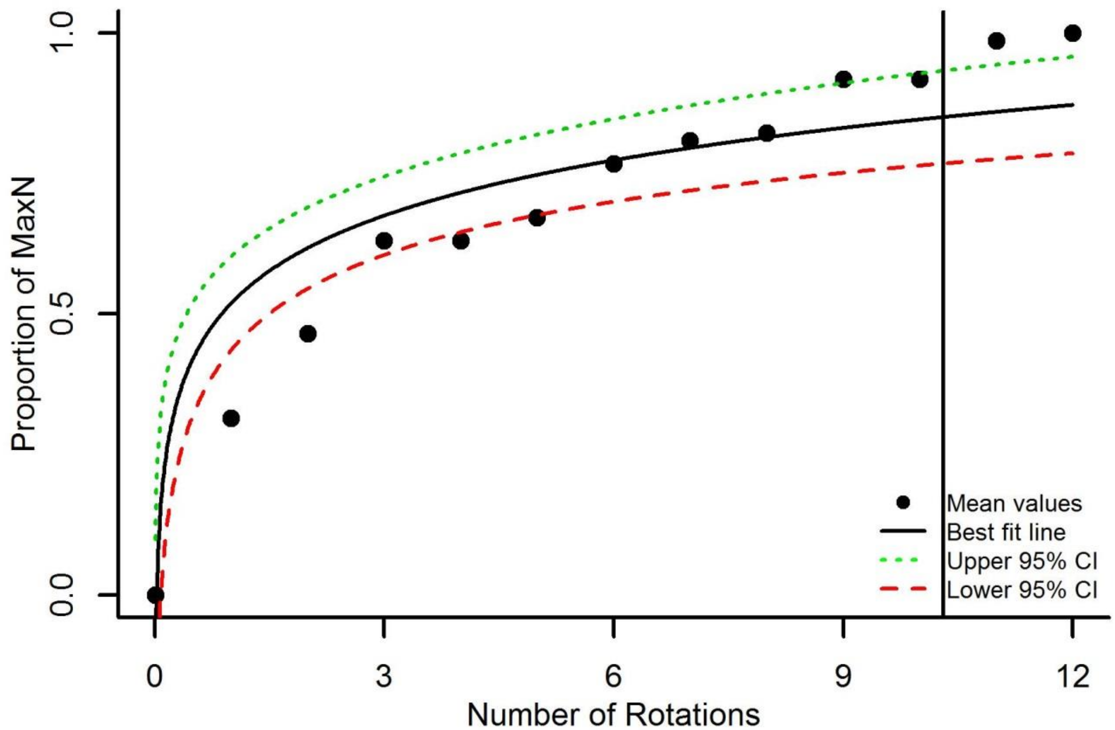
Appendix A: MaxN over time for species of interest



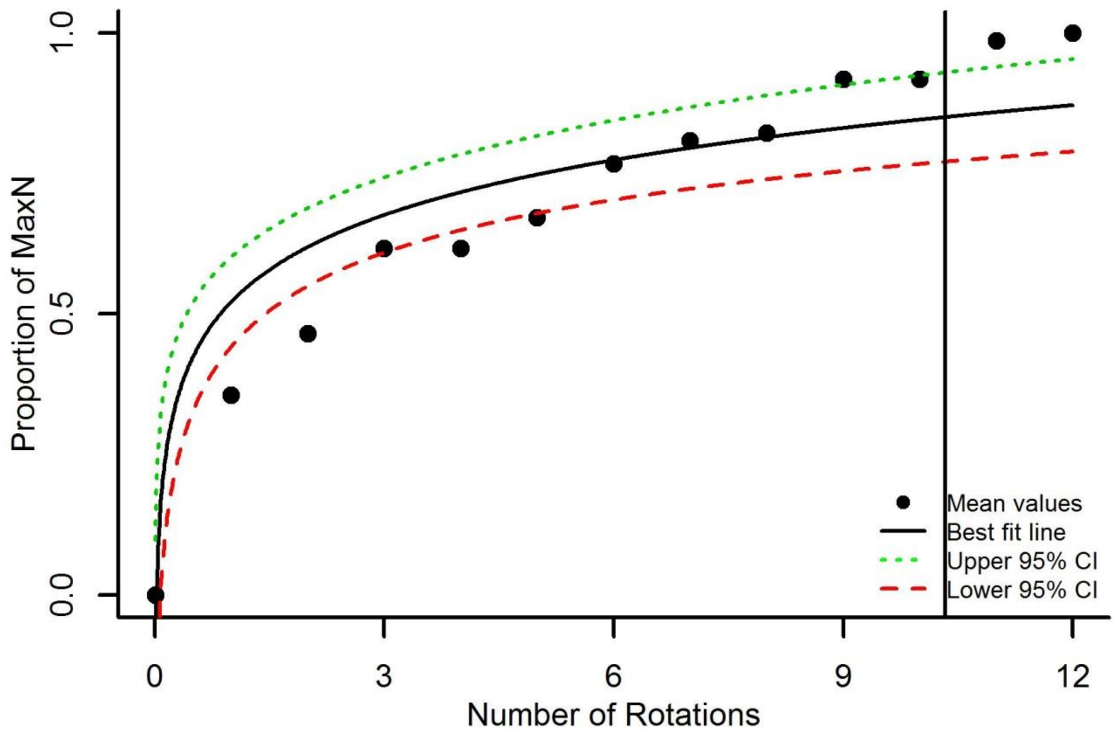
Appendix A Figure 1: Proportion of MaxN at 12 minutes achieved with increasing number of rotations for Bocaccio, *Sebastes paucispinus.* Time when 85% of MaxN is marked by the vertical lines. Dashed lines are +/- SE curves.



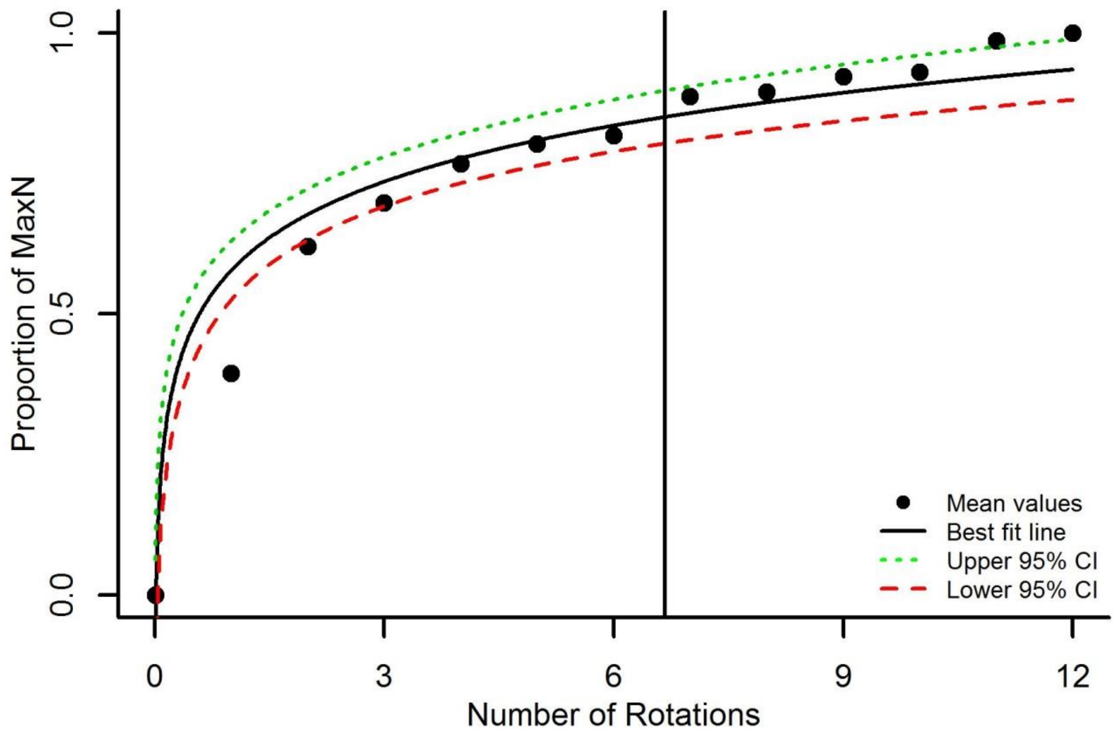
Appendix A Figure 2: Proportion of MaxN at 12 minutes achieved with increasing number of rotations for Canary Rockfish, *Sebastes pinniger.* Time when 85% of MaxN is marked by the vertical lines. Dashed lines are +/- SE curves.



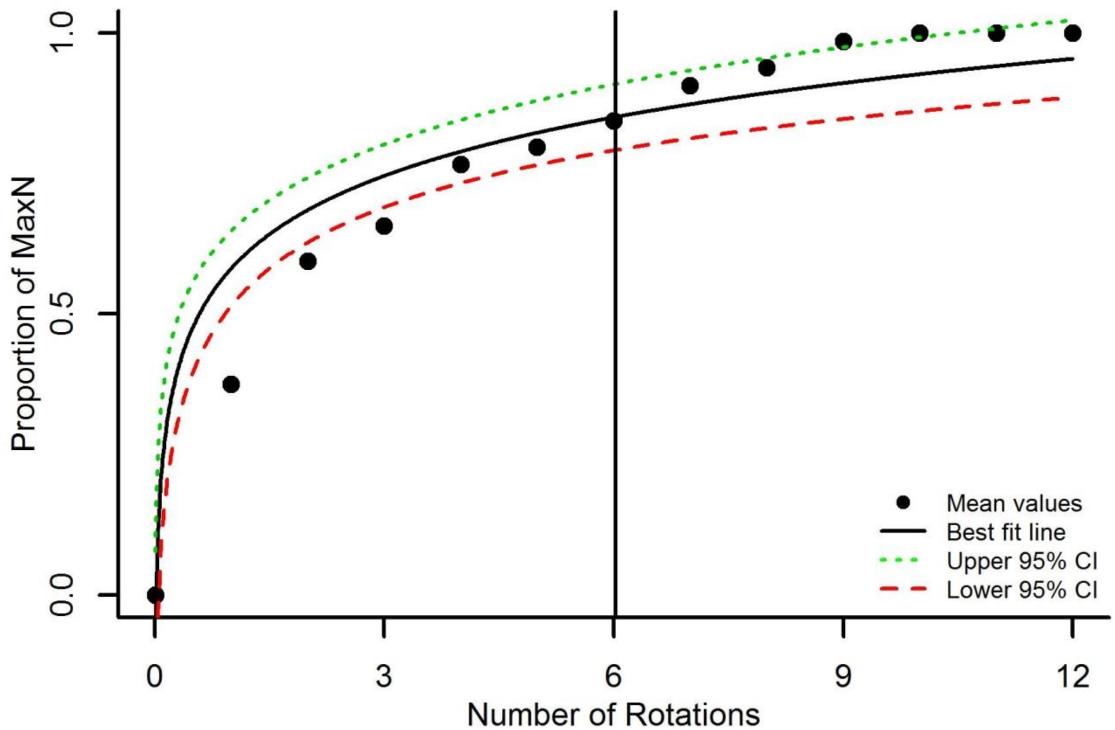
Appendix A Figure 3: Proportion of MaxN at 12 minutes achieved with increasing number of rotations for Copper Rockfish, *Sebastes caurinus.* Time when 85% of MaxN is marked by the vertical lines. Dashed lines are +/- SE curves.



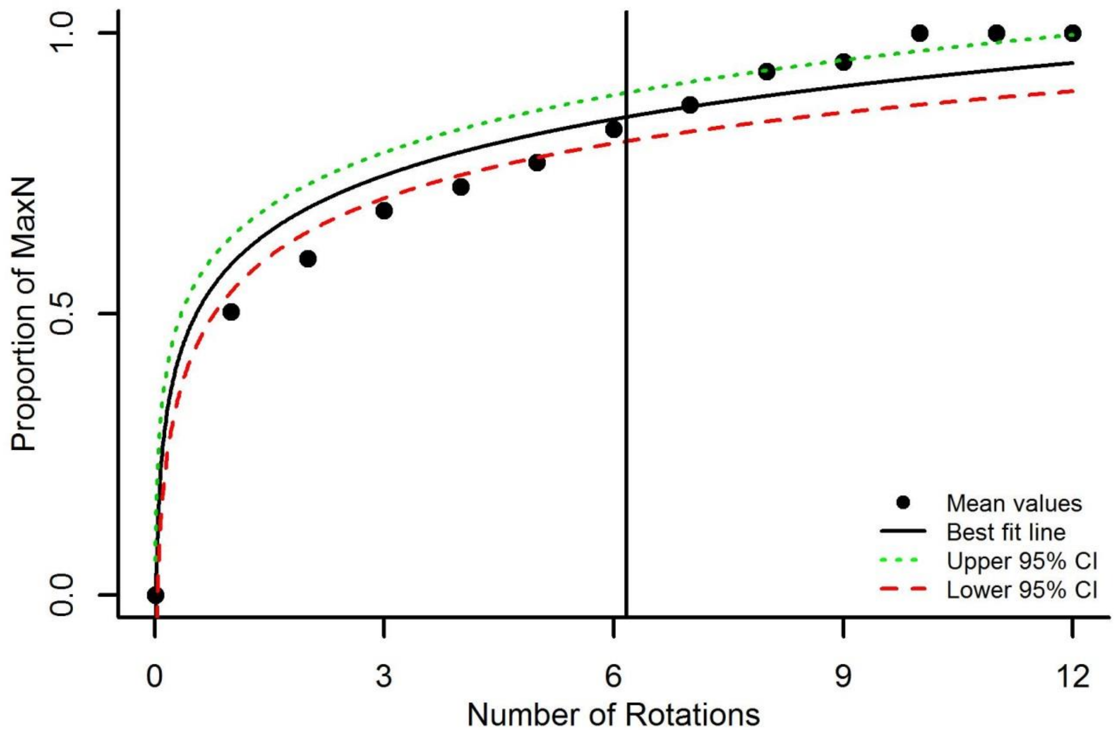
Appendix A Figure 4: Proportion of MaxN at 12 minutes achieved with increasing number of rotations for Cowcod, *Sebastes levis.* Time when 85% of MaxN is marked by the vertical lines. Dashed lines are +/- SE curves.



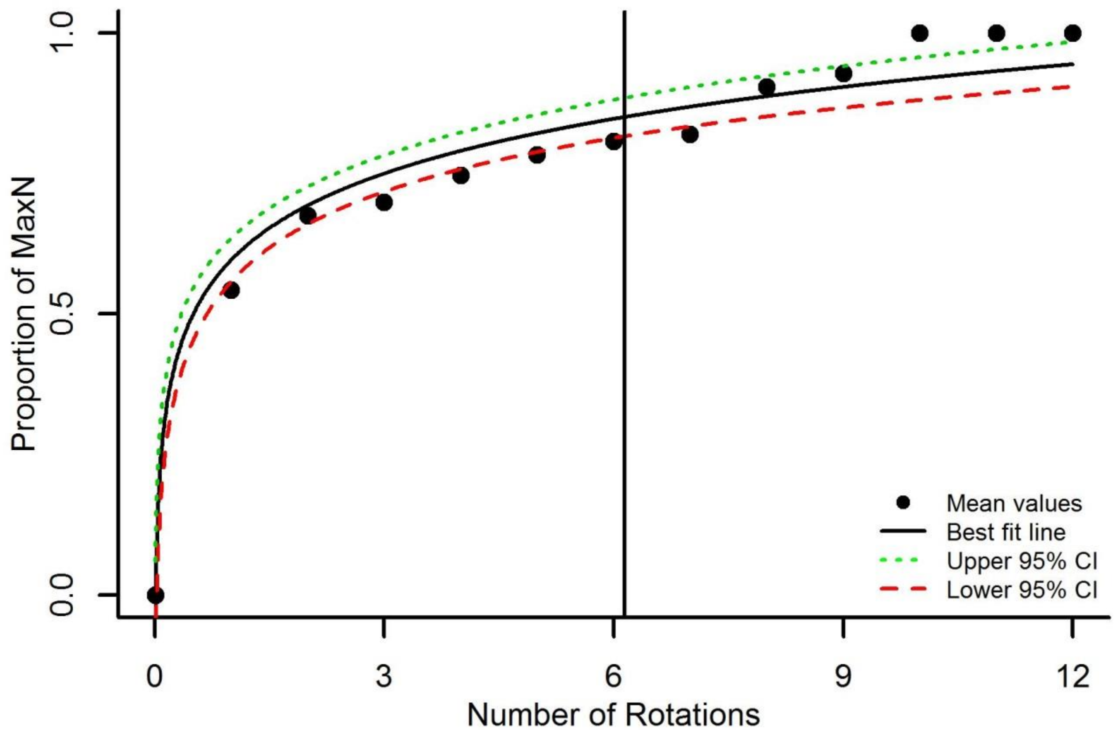
Appendix A Figure 5: Proportion of MaxN at 12 minutes achieved with increasing number of rotations for Greenspotted Rockfish, *Sebastes chlorostictus.* Time when 85% of MaxN is marked by the vertical lines. Dashed lines are +/- SE curves.



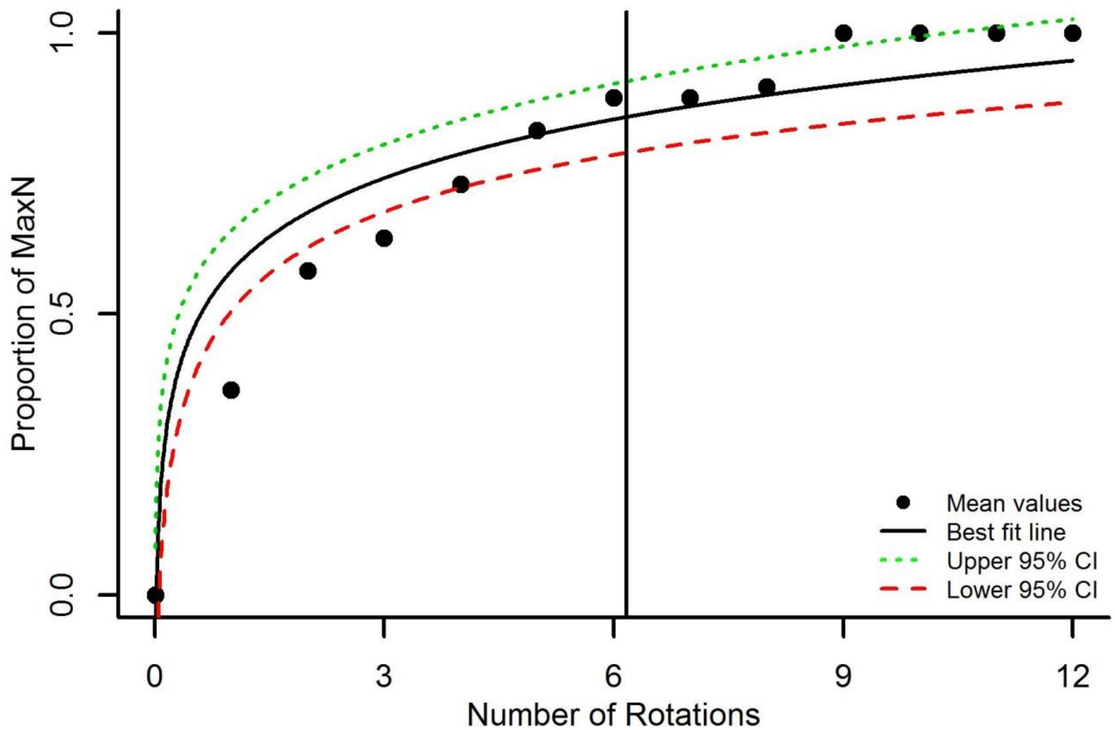
Appendix A Figure 6: Proportion of MaxN at 12 minutes achieved with increasing number of rotations for Lingcod, *Ophiodon elongatus.* Time when 85% of MaxN is marked by the vertical lines. Dashed lines are +/- SE curves.



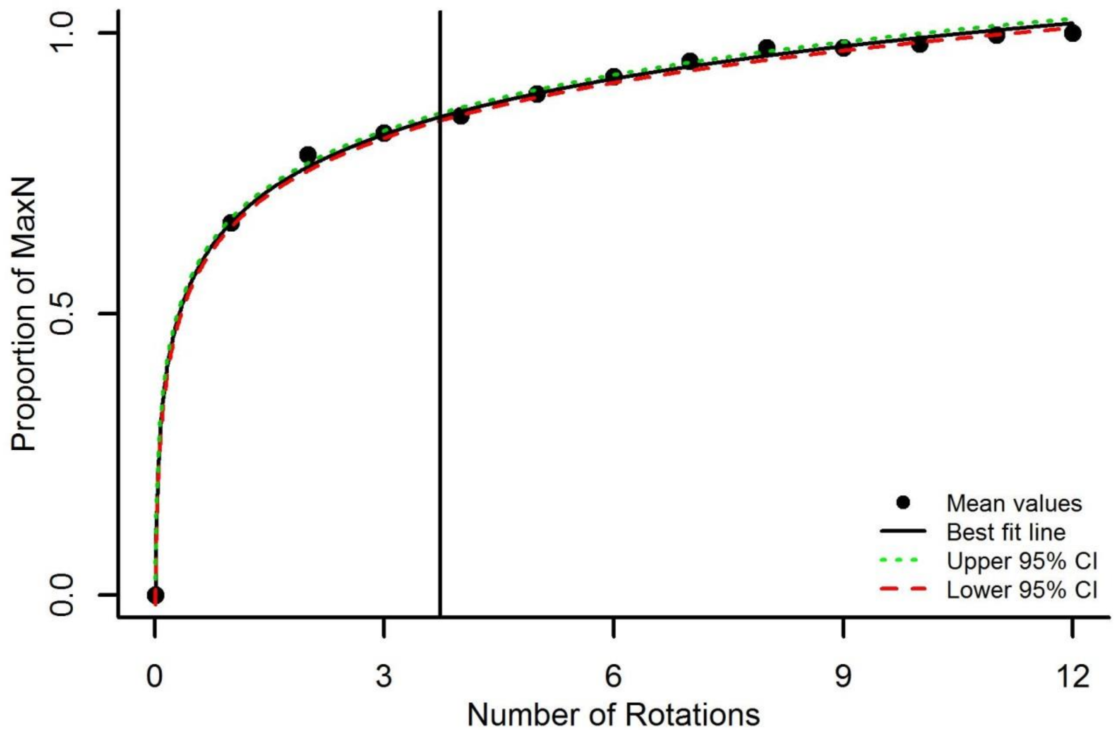
Appendix A Figure 7: Proportion of MaxN at 12 minutes achieved with increasing number of rotations for Pygmy Rockfish, *Sebastes wilsoni.* Time when 85% of MaxN is marked by the vertical lines. Dashed lines are +/- SE curves.



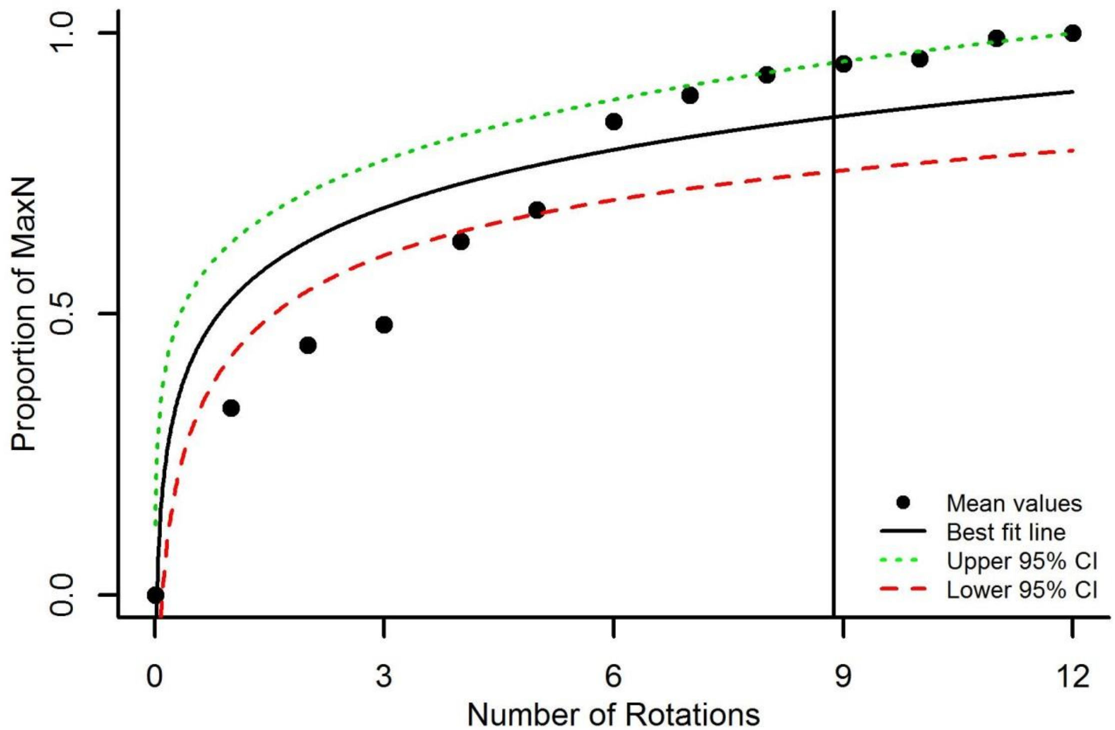
Appendix A Figure 8: Proportion of MaxN at 12 minutes achieved with increasing number of rotations for Squarespot Rockfish, *Sebastes hopkinsi.* Time when 85% of MaxN is marked by the vertical lines. Dashed lines are +/- SE curves.



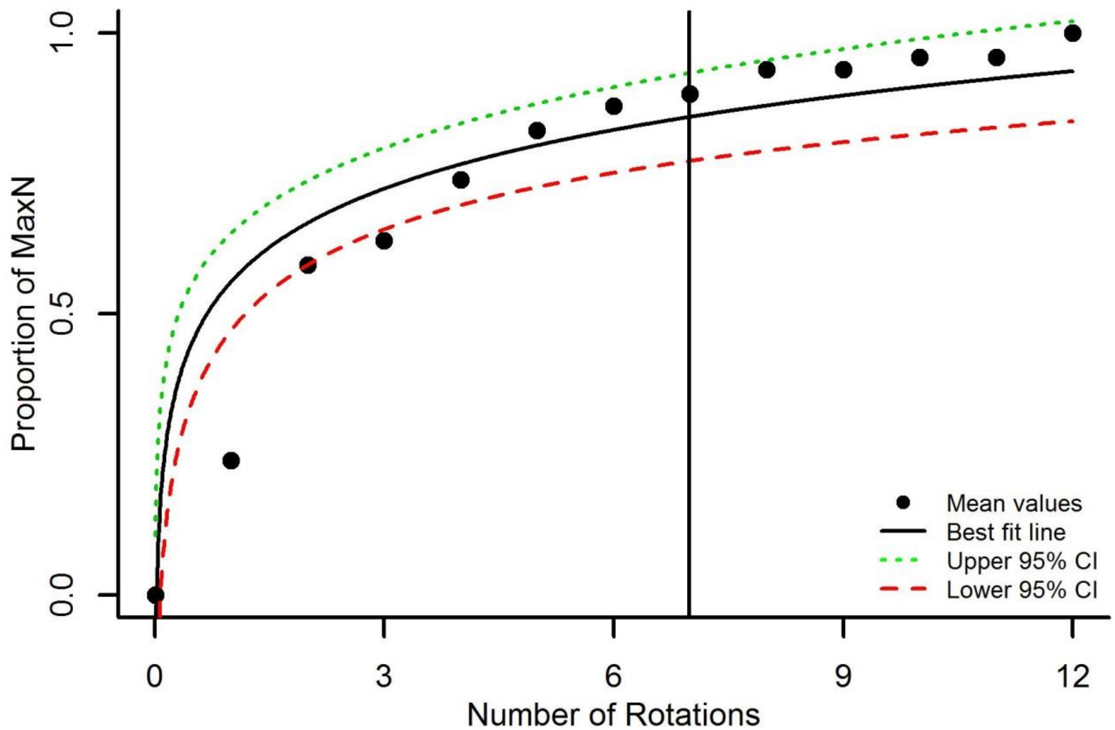
Appendix A Figure 9: Proportion of MaxN at 12 minutes achieved with increasing number of rotations for Starry Rockfish, *Sebastes constellatus*. Time when 85% of MaxN is marked by the vertical lines. Dashed lines are +/- SE curves.



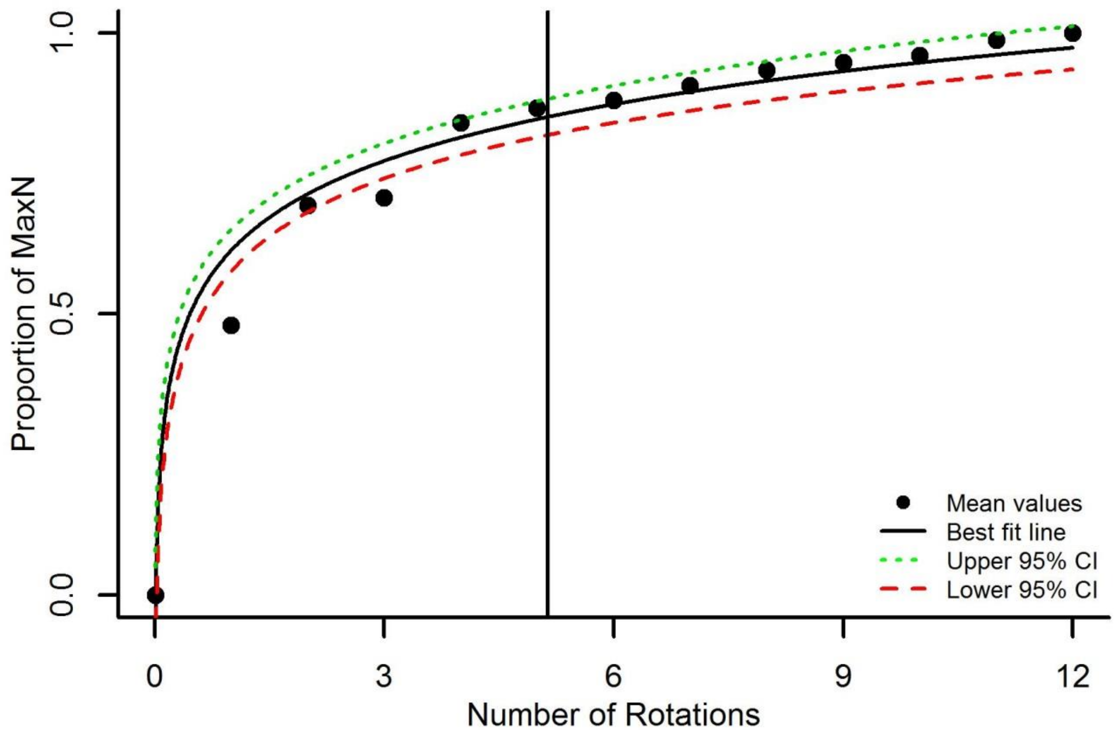
Appendix A Figure 10: Proportion of MaxN at 12 minutes achieved with increasing number of rotations for Vermilion Rockfish, *Sebastes miniatus.* Time when 85% of MaxN is marked by the vertical lines. Dashed lines are +/- SE curves.



Appendix A Figure 11: Proportion of MaxN at 12 minutes achieved with increasing number of rotations for Widow Rockfish, *Sebastes entomelas.* Time when 85% of MaxN is marked by the vertical lines. Dashed lines are +/- SE curves.

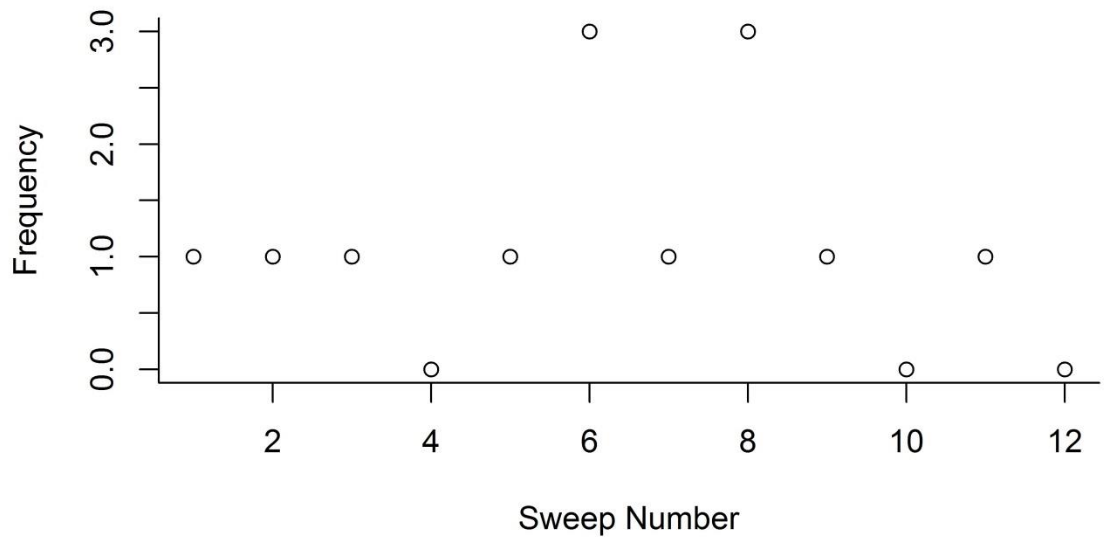


Appendix A Figure 12: Proportion of MaxN at 12 minutes achieved with increasing number of rotations for Yelloweye Rockfish, *Sebastes ruberrimus.* Time when 85% of MaxN is marked by the vertical lines. Dashed lines are +/- SE curves.

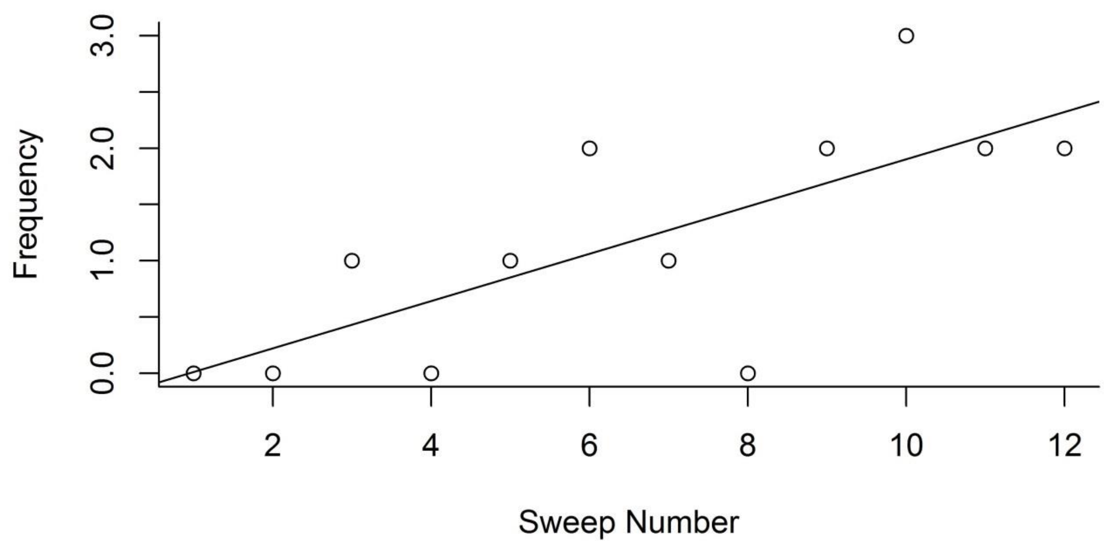


Appendix A Figure 13: Proportion of MaxN at 12 minutes achieved with increasing number of rotations for Yellowtail Rockfish, *Sebastes flavidus.* Time when 85% of MaxN is marked by the vertical lines. Dashed lines are +/- SE curves.

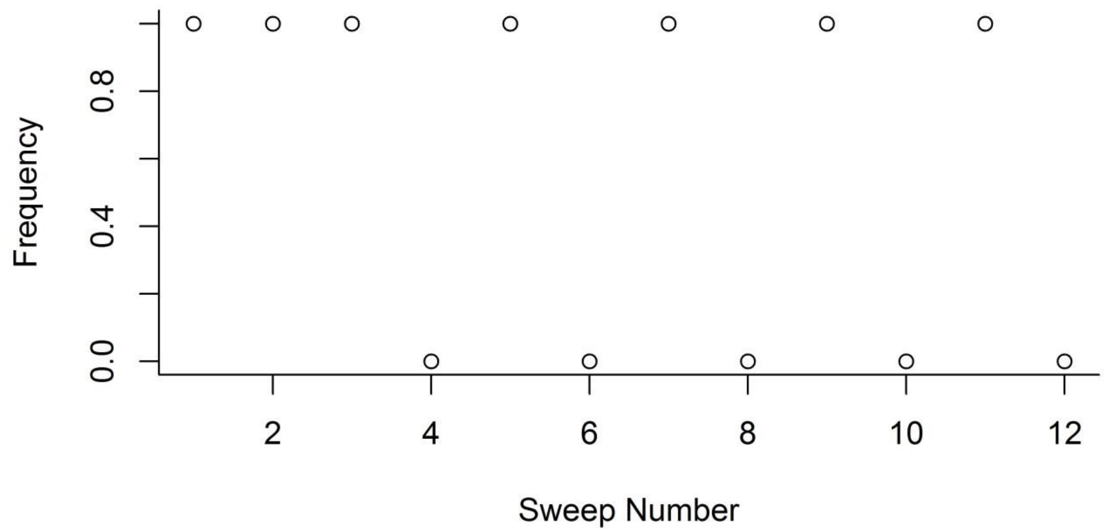
Appendix B: Sweep Containing MaxN



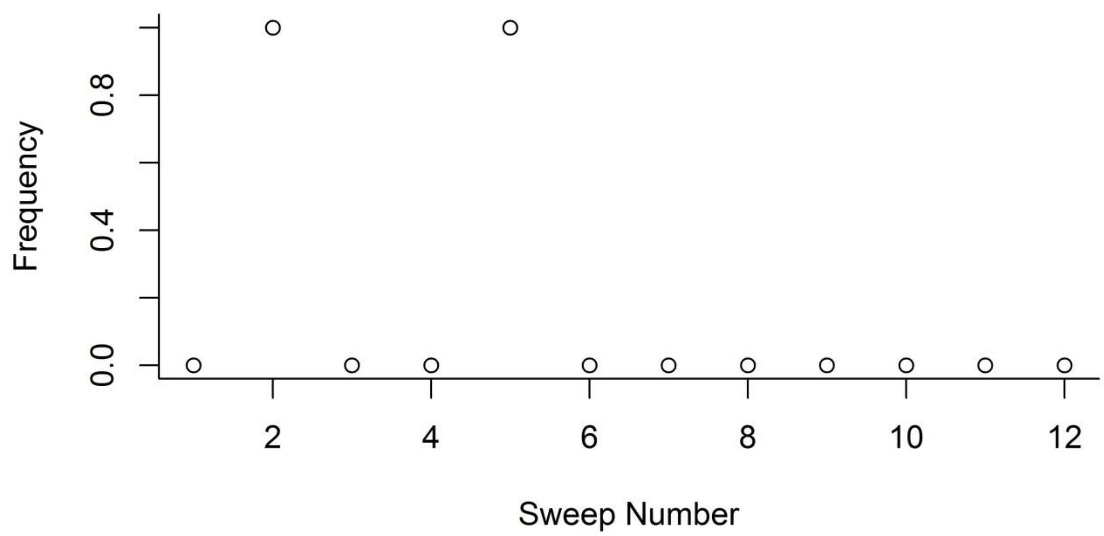
Appendix B Figure 1: Bocaccio Rockfish. No significant relationship between sweep number and frequency of that sweep containing the MaxN value.



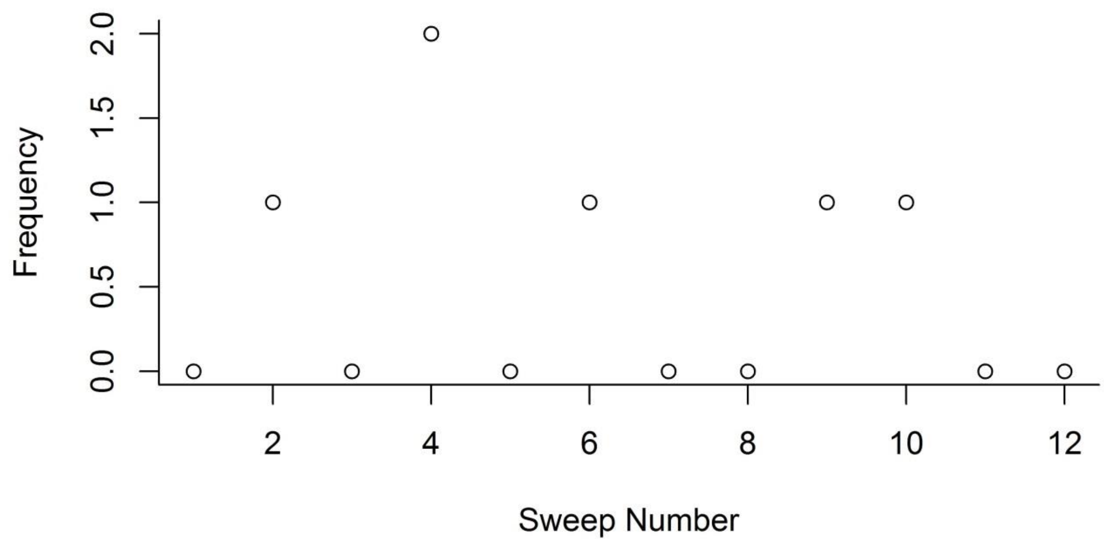
Appendix B Figure 2: Canary Rockfish. Significant relationship between sweep number and frequency of that sweep containing the MaxN value, **$p = 0.007$**, $r^2 = 0.493$.



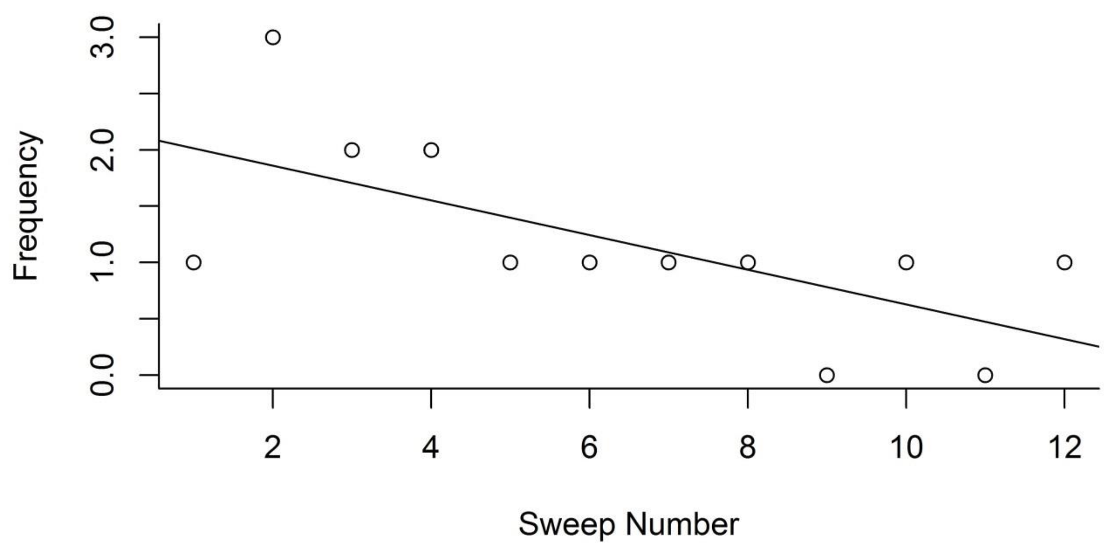
Appendix B Figure 3: Copper Rockfish. No significant relationship between sweep number and frequency of that sweep containing the MaxN value.



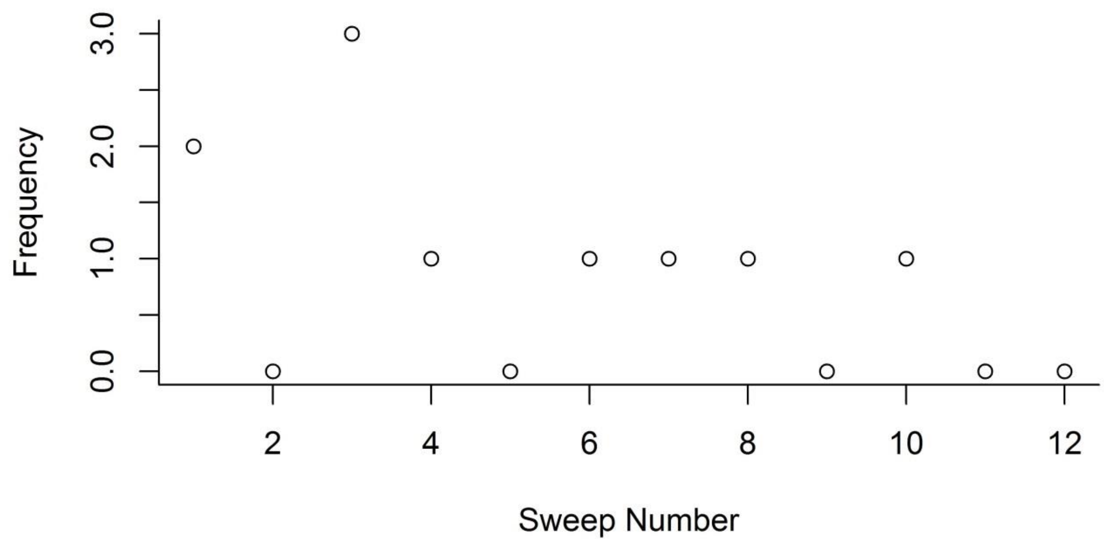
Appendix B Figure 4: Cowcod. No significant relationship between sweep number and frequency of that sweep containing the MaxN value.



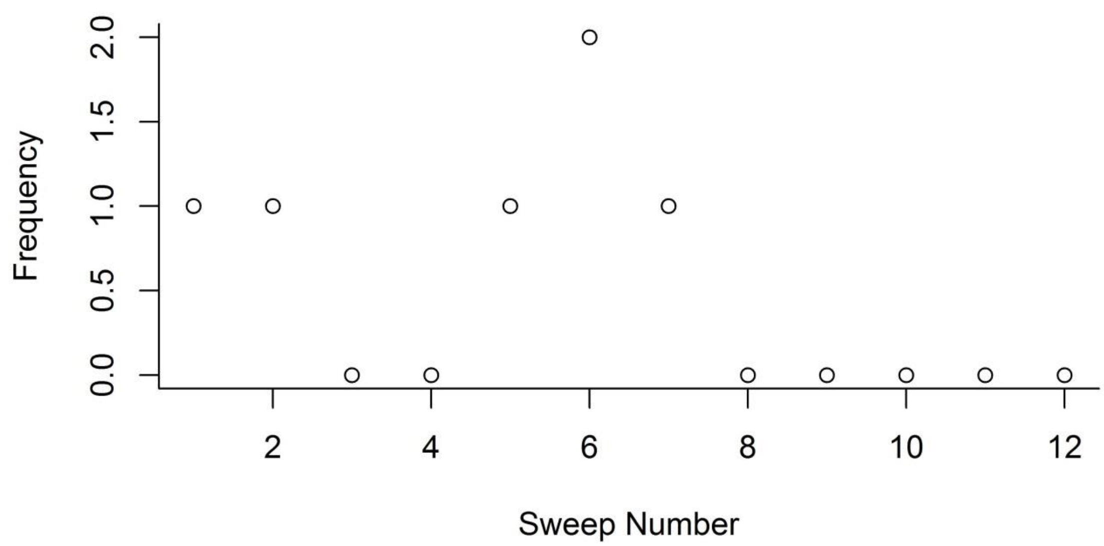
Appendix B Figure 5: Greenspotted Rockfish. No significant relationship between sweep number and frequency of that sweep containing the MaxN value.



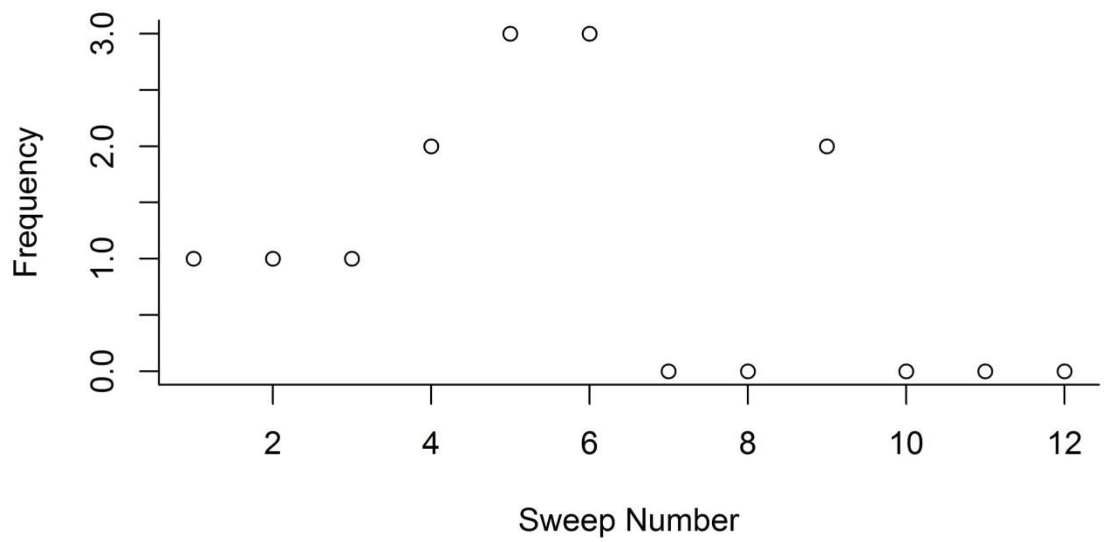
Appendix B Figure 6: Lingcod. Significant relationship between sweep number and frequency of that sweep containing the MaxN value, **p = 0.006**, r^2 = 0.508.



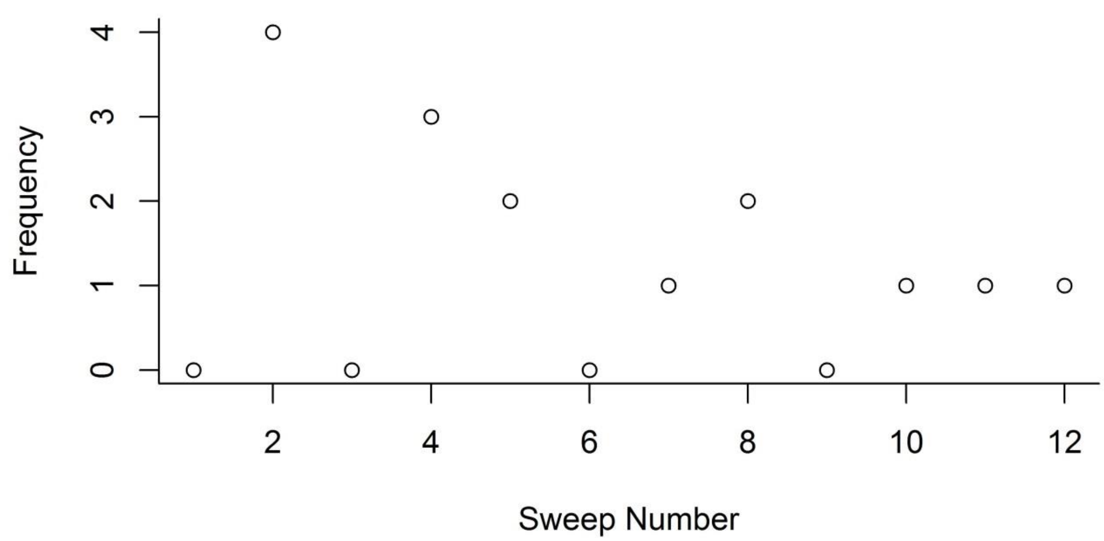
Appendix B Figure 7: Pygmy Rockfish. No significant relationship between sweep number and frequency of that sweep containing the MaxN value.



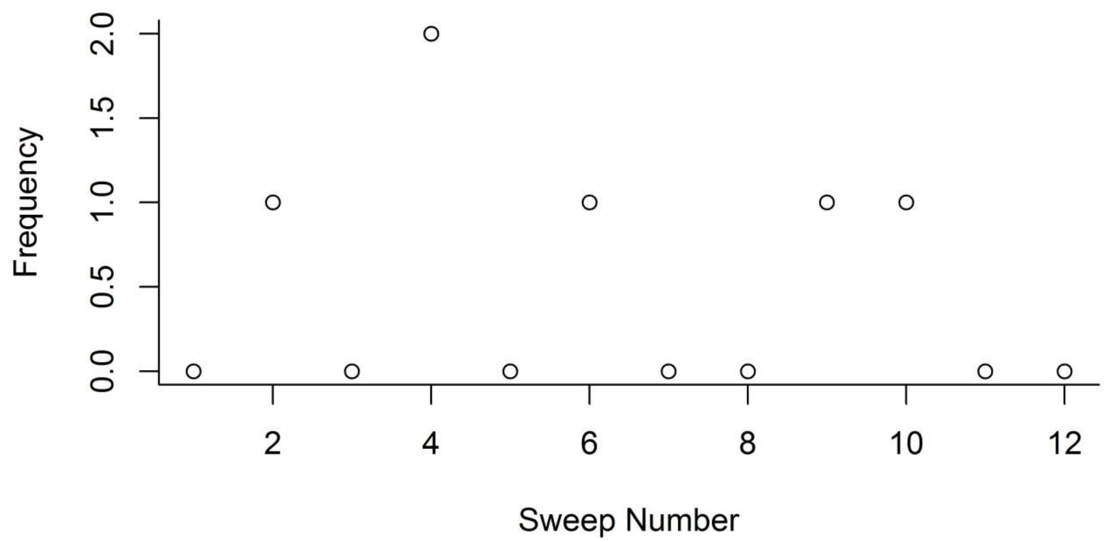
Appendix B Figure 8: Squarespot Rockfish. No significant relationship between sweep number and frequency of that sweep containing the MaxN value.



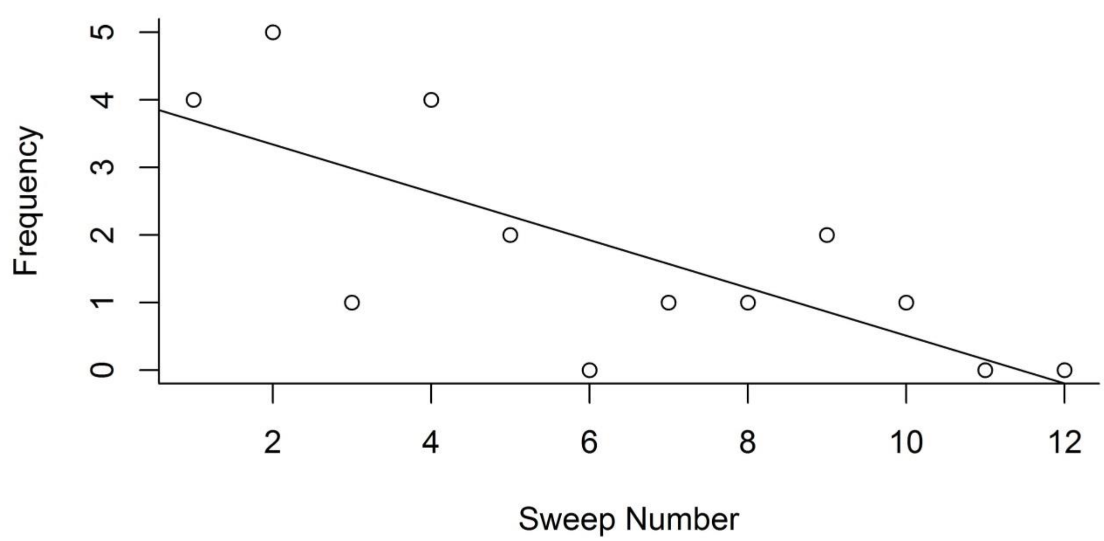
Appendix B Figure 9: Starry Rockfish. No significant relationship between sweep number and frequency of that sweep containing the MaxN value.



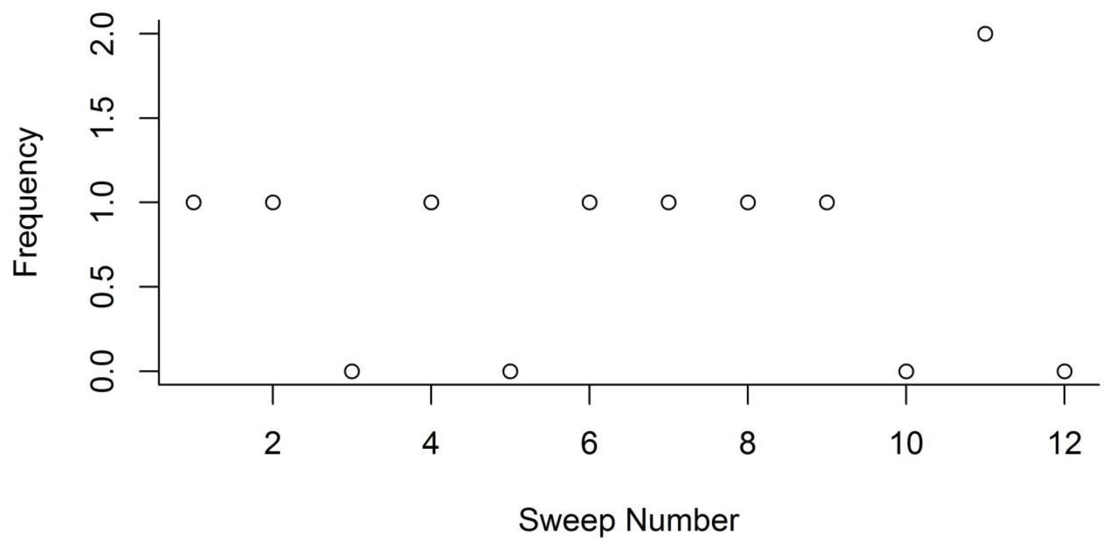
Appendix B Figure 10: Vermilion Rockfish. No significant relationship between sweep number and frequency of that sweep containing the MaxN value.



Appendix B Figure 11: Widow Rockfish. No significant relationship between sweep number and frequency of that sweep containing the MaxN value.



Appendix B Figure 12: Yelloweye Rockfish. Significant relationship between sweep number and frequency of that sweep containing the MaxN value, p-value = 0.018, r^2 = 0.386.



Appendix B Figure 13: Yellowtail Rockfish. No significant relationship between sweep number and frequency of that sweep containing the MaxN value.