CONDUCTING REEF FISH SURVEYS THROUGH A NEW LENS: THE TRANSFORMATIVE POTENTIAL OF INNOVATIVE TECHNOLOGIES AND COMMUNITY-BASED MONITORING METHODS.

A THESIS SUBMITTED TO THE GRADUATE DIVISION OF THE UNIVERSITY OF HAWAIʻI AT HILO IN
PARTIAL FULFILLMENT OF THE REQUIREMENT OF THE DEGREE OF
MASTER OF SCIENCE
IN
TROPICAL CONSERVATION BIOLOGY AND ENVIRONMENTAL SCIENCE

DECEMBER 2023

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Acknowledgements

I want to first thank my advisor Dr. John Burns and committee members Dr. Randy Kosaki and Dr. Grady Weyenberg for their willingness to support me. This project wouldn’t have been possible without their support, encouragement, and trust in me. Their advice and knowledge are what made this project a success, and I am certain I will carry this with me throughout my professional career and it will continue to help me grow as a scientist.

Furthermore, I want to thank the entire Multiscale Environmental Graphical Analysis (MEGA) Lab for providing such an incredible working, brainstorming and discussion-based lab space with so many like-minded individuals. Being part of such an epic team not only made the project more memorable but is truly what made this project happen. I couldn’t have asked for a better group of scientists to work alongside.

I am extremely grateful for my family and friends for their patience and support throughout my entire academic career. They always stuck with me, trusted the process, and gave me the confidence needed to get to this point in my academic career.

I would also like to extend my gratitude to all faculty and students of the TCBES program and the Graduate Assistant position for the opportunity and funding to pursue my passion and further my education. And I’d like to thank ChangeHawaii for additional funding to complete the project, and Aqualink and Fathom for the camera system that made this project possible.

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List of Abbreviations and Symbols

OC SCUBA: open circuit SCUBA

CCR: closed circuit rebreather

RUV: Remote Underwater Video

DOV: Diver Operated Video

UVC: Underwater Visual Census

SPC: Stationary Point Count

PCA: Principal Component Analysis

LMM: Linear Mixed Effects Model

Abstract

Coral reef fishes in Hawaiʻi are economic pillars both as subsistence and commercial food sources, and for ecotourism, with greater than 70% of the value being associated with subsistence activities. Current monitoring efforts are heavily focused on using a single method of open circuit (OC) SCUBA diving by certified scientific divers. These methods have been scrutinized in recent years for potential biases, and perhaps more importantly, for excluding many fishers in the community from collecting data that is useful for the management of nearshore fisheries. This thesis describes a comprehensive methods analysis of the standard OC SCUBA-based coral reef fish surveys in comparison to innovative approaches including permanent live-streaming remote underwater video (RUV) system, snorkeling, and freediving. This approach was designed to address the need for more accurate methodology that can be incorporated into inclusive community survey efforts to enhance reef monitoring. We utilized advanced statistical approaches to directly compare species richness, size estimation, and abundance among these different methodologies. Results from principal component analysis (PCA) and linear mixed effects models (LMM) showed that RUV, snorkeling, and freediving offer robust alternatives to OC SCUBA-based surveys and provide accurate data collection without the necessity of SCUBA. The novel live-streaming RUV approach displayed promising capabilities for enabling in-depth insights into ecological processes on coral reefs and continuous data acquisition. Snorkeling and freediving showcased their suitability in sizing and counting fish in shallow reef environments, and the potential for encouraging community engagement and local knowledge integration. This study highlights the transformative potential of innovative methods, encouraging collaboration between researchers and communities for effective coral reef conservation through advancement of coral reef fish monitoring efforts.

Introduction

Coral reefs are some of the most ecologically and culturally valuable ecosystems on our planet, supporting a diverse range of marine life and providing essential resources that humans depend on for livelihood and well-being (Hourigan et al., 1988; Cole et al., 2008). Coral reef fish, in particular, are a critical resource for many island nations and a primary source of protein for global food consumption. Unfortunately, reef ecosystems are under threat due to anthropogenic exploitation and the impacts of climate change (Kittinger et al., 2015). The Western Pacific has experienced a decline of up to 50% in coral reefs, with an annual decrease of 1-2% (Bruno & Selig, 2007). Such deterioration is attributed to a combination of global factors, including rising temperatures and ocean acidity, as well as local factors such as unsustainable fishing practices and nutrient runoff (Hughes et al., 2003; Magel et al., 2019). The ecological consequences of these declines are significant, affecting all components of coral reef ecosystems and creating substantial challenges to marine biodiversity conservation. Hawaiʻi, albeit less severe than other tropical regions, has also exhibited degradation of coral reef resources. Recent reports estimate the economic value of protecting and restoring Hawaiian coral reefs at $33.57 billion, underscoring the importance of these ecosystems for both environmental and economic well-being (Bishop et al., 2011). Given the ongoing environmental changes and their escalating impacts on reef ecology, it is critical we empower coastal communities with the necessary resources to conserve these systems. It will become necessary to enhance our monitoring efforts in an attempt to connect the vitality of reef ecosystems and the livelihoods reliant on their associated fisheries at the local level.

While acknowledging the significance of all reef species in maintaining a healthy coral reef ecosystem, coral reef fishes serve an array of diverse interests, encompassing commercial, recreational, and subsistence activities (Pooley, 1993; Dalzell & Schug, 2002). Many coastal communities in Hawaiʻi heavily rely on coral reef fisheries, with estimates suggesting values ranging from $10.3 to $16.4 million, with over 70% of the value attributed to subsistence activities (Grafeld et al., 2017). Cultural traditions have long intertwined with fish monitoring on the Hawaiian Islands where indigenous knowledge has been passed down through generations, encompassing insights into spawning seasons, habitat preferences, and sustainable harvesting practices (Manu 1992; Kahaulelio 2005). Scientifically, data on coral reef fish in Hawaiʻi has been primarily collected through underwater visual censuses (UVCs), most often via belt transect or stationary point count (SPC) involving a team of SCUBA divers who identify, count, and estimate fish sizes (Minton et al., 2015; Fukunaga et al., 2019; Pacific Islands Fisheries Science Center, 2023. SCUBA-based UVCs have been utilized for decades, although concerns have been raised regarding their accuracy, potential biases, and exclusivity of the non-scientific community in the process.

Two of the most utilized SCUBA-based UVCs among scientific organization and management agencies are the visual belt transect and stationary point count (Caldwell et al., 2016). Visual belt transects involve laying a transect at a predetermined heading and location in which a diver swims along visually identifying, counting, and sizing fish species within a defined width or belt parallel to the transect line. Stationary point count (originally pioneered by Bohnsack and Bannerot, 1986) involves a diver rotating at a stationary point and visually identifying, counting, and sizing fish species within a predetermined cylinder often 15 meters in diameter. Although they are the two most common methods utilized, many scientists are critical of these studies and argue they disturb the survey area and introduce errors in diversity estimates (Dickens et al., 2011; Wilson et al., 2018). The critics further determined the expertise and experience of divers may significantly influence survey results ultimately affecting the assessment of major fish families such as Acanthuridae, Labridae, Pomacentridae and Holocentridae (Williams et al., 2006). Such inaccuracies can have profound implications for the management decisions concerning reef fish and present a need for further analysis assessing the accuracy and effectiveness.

In response to these limitations, researchers have been actively seeking less invasive methods for conducting coral reef fish surveys to minimize bias in estimates used for characterization of fish assemblages. Studies in Hawaiʻi have investigated the use of open circuit (OC) versus closed circuit rebreather (CCR) SCUBA surveys as potential alternatives, with CCR demonstrating promise in providing more accurate biomass estimates for specific fish families

(Gray et al., 2016). However, the adoption of CCR presents challenges such as increased training requirements, safety considerations, and equipment costs. Consequently, alternative approaches utilizing underwater video surveys have gained popularity due to recent advancements in camera affordability and functionality (Rowcliffe and Carbone, 2008; Rovero et al., 2013). Since the year 2000, the mention and use of cameras has seen 50% annual growth in peer reviewed papers, but when comparing video-based methods to their terrestrial counterparts, there is a notable lag in coral reef ecology to adopt these digital tools (Rowcliffe and Carbone 2008; Lowry et al. 2011; Sun et al. 2018). One common approach is the use of diver-operated video (DOV) surveys, where divers record video while swimming along transects for later analysis (Lindfield et al., 2014; Larson et al., 2022). Interestingly, this approach has been shown to reduce errors and inaccuracies in data and yielded significantly different results when compared to standard belt transects. While DOV surveys represent one alternative, they remain exclusive to the scientific diving community and still present potential bias due to the presence of divers. Furthermore, the analysis of the digital video requires hours of additional video analysis following fieldwork activities. There is still a need for methods that can reduce bias and increase efficiency of data collection to rapidly characterize reef fish assemblages.

To address the limitations associated with divers and increase data efficiency while minimizing disturbances, scientists have been developing improved remote stationary camera stations. This technological advancement aims to reduce the need for divers and allows for increased data collection effort, ultimately minimizing environmental disturbances. While promising, current remote video surveys are constrained by limitations like soak time, battery life, continuous repositioning, and ocean conditions (Parker et al. 2016; Whitmarsh et al. 2018; Follana-Berna et al. 2019). It is also crucial to acknowledge that many current remote video surveys involve the use of baited cameras, which alter natural fish behavior and provide misleading baselines for fish assemblages (Harvey et al. 2004; Cappo, 2006; Parker et al., 2016; Asher 2017; Whitmarsh et al., 2017, 2018; Langlois et al., 2020). To overcome this challenge, unbaited camera surveys have been explored, showing promise specifically in capturing non-carnivorous fish species more accurately (Cappo, 2006; Harvey & Shortis, 2006; Mallet &

Pelletier, 2014; Erickson et al., 2022). However, the constant repositioning of cameras still poses difficulties and compromises baseline data, resulting in excessive hours of video analysis. In order to effectively progress camera systems, there is a strong need for advanced methods utilizing long-term or permanently positioned remote cameras to explore the intricacies of reef fish assemblages.

In summary, the exclusive nature of diver-based surveys which require specific certifications and affiliations, inadvertently excludes valuable knowledge from local communities, fishers, and community scientists who possess profound connections to reef resources (Friedlander et al. 2000; Kobori et al. 2016). To harness the insights of these knowledgeable individuals, particularly in regions like Hawaiʻi and the Pacific, alternative approaches with fewer equipment and training barriers should be explored. Snorkeling has been largely recognized for its utility in freshwater ecosystems for assessing fish diversity, distribution, and abundance, however it has been overlooked in the context of accurately assessing coral reef fish assemblages (Slaney & Martin 1987; Johnson et al. 2007). The feasibility of snorkeling stems from its accessibility and simplicity, requiring minimal equipment and offering a non-invasive approach that minimizes disturbances to aquatic life. And to further complement the snorkel efforts, freedivers are often overlooked despite presenting a unique perspective to observe the reef. With minimal equipment, the observers can remain conspicuous during observations and provide new insight into reef fish ecology. The significance of incorporating freediving and snorkeling becomes evident when evaluating the numerous islands across the Pacific. Many of these islands, despite having limited resources, are dedicated to monitoring, and managing their coral reefs.

The present study aims to contribute to the advancement of coral reef monitoring efforts by introducing minimally invasive survey methods: snorkeling, freediving, and remote underwater video (RUV) using a permanently mounted Pan-Tilt Zoom live-streaming video camera system. These methods are compared with the conventional open circuit (OC) SCUBA surveys to evaluate efficacy, accuracy, and limitations with a focus on assessing diversity, size, and abundance of observed reef fishes. We hypothesize that these methods complement each other, offering new perspectives for reef fish surveys. These developments aim to bridge the gap between fishermen and resource management, involving spearfishers and snorkelers in community-based scientific methodologies. Notably, the RUV system provides an unprecedented number of observations across diel and seasonal scales without the limitations of discrete time points inherent in diver-based methods. By enhancing community involvement and leveraging local fishers' knowledge alongside technological advancements, we aim to establish robust baselines for future conservation efforts in the Pacific region.

Methods

Site Description

Data collection for this study was conducted over a two-week period in May and June of 2023, in the nearshore reef environment at Keāhole Point, in Kailua Kona, Hawaiʻi (Figure 1). Keāhole Point is located as the most Westward point of the Hawai’i Island and is characterized as a highly exposed area to wind and swell. Notably, the location experiences high fishing pressure from shore fishermen and occasional spearfishermen due to the exposed ocean conditions.   The nearshore reef structure is unique in that the area has three sets of pipelines with 1.2-meter diameter pipes that deliver deep sea water from up to 900 meters depth and surface seawater. The pipes were originally installed for the purpose of generating energy utilizing the temperature difference between the pipes in a process referred to as ocean thermal energy conversion. After many years, the pipes were repurposed and are now used to supply numerous aquaculture facilities with both deep and surface seawater. A total of 15 paired surveys were conducted on concurrent days, for a total of 60 surveys utilizing the following methods: OC SCUBA, snorkeling, freediving, and RUV surveys. All surveys were conducted at the same precise location directly above the permanently mounted camera system.. Scale bars and markers were deployed at the survey location to establish a 15-meter diameter cylinder with the remote camera positioned at the center of the cylinder. These scale bars allowed for accurate sizing and the markers facilitated accurate distance determination of the cylinder during both visual *in-situ* and RUV surveys.

Survey Methods

To characterize fish assemblages, we employed a stationary point count (SPC) survey (Brock 1954; Ayotte et al. 2015). Each survey duration was 15 minutes, divided into two periods: the first 5 minutes were dedicated to species enumeration, and the subsequent 5-15 minutes were used for counting and sizing. Species observed during the initial 5 minutes were considered instantaneous fish and given priority for counting and sizing. Species observed between 5 and 15 minutes were also recorded but considered non-instantaneous fish. During the sizing period, fish were systematically counted in the order of observation and sized to the nearest centimeter. In cases where a species was present during the enumeration period but absent during the tallying period, the number initially observed was recorded. Survey methods, (OC SCUBA, free-dive, snorkel, and RUV), were randomly selected and conducted in two-time brackets (morning and afternoon) on each day of surveying.

SCUBA fish surveys were conducted using standard stationary point count (SPC) method (Ayotte et al. 2015). Divers positioned themselves adjacently and simultaneously performed two SPC surveys measuring 15 m in diameter. During observation, divers rotated in a stationary position to avoid disturbance of fishes and minimize double counting. Data collected by the individual surveying the area directly above the camera was used for data analysis. Upon completion of the survey the divers exited the water and waited a minimum of 30 minutes to allow for settling and minimizing disturbance for the subsequent survey.

Snorkel methods were consistent with SCUBA, with divers performing standard SPC surveys, but entirely from the surface.

Freedive survey methods differed slightly from OC SCUBA and snorkel methods. Surveyors positioned themselves directly above the mounted camera and were assigned 180 degrees of the cylinder to observe. Surveyors dived down to a depth of approximately 1 meter above the benthic substrate and began surveying within their assigned 180-degree section of the 15-meter diameter cylinder. Upon reaching the bottom, surveyors remained stationary for

30 seconds and enumerated all fish species. After surfacing, surveyors recorded their observations and remained at the surface for 1-2 minutes while their partner dove down. Surveyors repeated this process until a total survey time of 5 minutes was achieved between the two divers. After the five-minute species enumeration period, divers initiated the counting and sizing period, systematically recording the number of individuals of each species from the surface for the remaining 10 minutes.

RUV surveys were conducted using modified SPC surveys, with the camera programmed to emulate a diver. The survey time was recorded using OBS Studio Version 29.1.2 and saved to be observed at the conclusion of the day. Throughout the 15-minute survey period, the camera rotated, completing an entire rotation every five minutes. The first rotation (5 minutes in duration) was the species identification period, and the second and third rotation (10 minutes in duration) was the counting and sizing period. For video analysis, the footage was paused and played as needed during the entire clip (Assis et al., 2013). The surveyor identified all fish to the species level and attempted to size and count fish within the cylinder using scale bars and boundary markers placed on the reef. As no single consensus method for counting using video currently exists (Cappo 2006; Harvey & Shortis 2006; Conn 2011; Whitmarsh 2018; Fairweather & Huveneers 2017), we developed a novel approach for this study. With a 360-degree rotating point of view, specific species were systematically chosen in order of observation. Individual species were counted and sized during a single rotation, either during the 5–10-minute period or 10–15-minute period. This systematic approach aimed to minimize the possibility of double-counting as the camera pans over the same areas repeatedly throughout the duration of the survey. Upon completion of all RUV surveys, three community scientists repeated the surveys following the described methods to test the efficacy and robustness of the novel approach.

All surveys were conducted by four highly trained and experienced divers. The two OC SCUBA surveys were conducted by the same two divers throughout the study, while the free-dive and snorkel divers were randomly selected. Video analysis was performed by a single individual, and then repeated by three additional community science surveyors to allow for comparison of potential observer bias and test the efficacy of the proposed novel approach. By using OC SCUBA as the reference method and maintaining consistency in the survey team, we aimed to reduce the potential scope of observer bias, while randomization in divers for the new methods introduced a controlled level of variability.

Descriptive Statistical Analysis

We computed the mean and standard deviation of species richness for each of the four survey methods employed in the study. This allowed us to discern the central tendency and variability of species richness associated with each method. Additionally, we examined species richness with respect to the diurnal timeframes in which the surveys were conducted: morning and afternoon. By calculating the mean and standard deviation of species richness separately for these periods, we gained insight into potential diurnal shifts in species richness. To explore whether the order in which surveys were conducted had an impact on species richness, we computed the mean and standard deviation of species richness for each survey order. This analysis illuminated any trends or variations that might emerge based on survey sequence and potential disturbance because of preceding surveys. Finally, we evaluated the daily variability in species richness by calculating mean and standard deviation for each survey day. The mean and standard deviation values for each day's species richness provided a comprehensive view of the potential temporal fluctuations.

PERMANOVA

To assess the potential differences in mean species richness among the four survey methods, we employed a combination of multivariate and pairwise statistical techniques. Specifically, we utilized the PERMANOVA (Permutational Multivariate Analysis of Variance) analysis via the ‘adonis’ function to examine the overall variance in species richness attributed to the different survey methods. This approach is robust to the heterogeneity of multivariate data and provides an effective means to partition and quantify the contributions of categorical factors, such as survey methods, to the observed variability in species richness (Anderson 2001). Furthermore, to discern specific differences between the survey methods, we conducted pairwise comparisons using t-tests. Given the potential for inflated Type I error rates due to multiple comparisons, we applied a Bonferroni correction to mitigate this concern (Dunn 1961). This correction is particularly suitable when dealing with multiple pairwise comparisons, as in our case where we compared each survey method against the others, ensuring that the probability of any false positive findings is controlled across the entire set of comparisons. In addition to comparing species richness across survey methods, we assessed the potential variations in species richness across survey times, survey orders and time periods (i.e. morning and afternoon) of surveys. We again employed PERMANOVA analysis to assess the contributions of these various factors to differences in species richness.

Linear Mixed Effects Model

Linear Mixed Effects Models (LMM) were used to assess differences and variations in the sizing and counting of individual fish species among the four distinct survey methods. The application of LMM is well-suited for our investigation as it accommodates fixed effects, such as survey method, within hierarchical groups, while simultaneously incorporating random effects associated with fish presence and absence across repeated measures (Zuur et al. 2009). The utilization of Gaussian distribution in the LMM analysis facilitated the modeling of continuous variables representing both fish counts and sizing, ensuring compatibility with the underlying data structure. This approach allowed us to comprehensively capture the impact of survey methods on the observed differences in fish counts and sizes. The count data was log-transformed to account for the pronounced right skew.

Principal Component Analysis

Principal Component Analysis (PCA) examined variability in the presence and absence of fish at the species level among the various methods we used for this study. PCA is a powerful visualization tool that serves as a dimensionality reduction technique enabling the comprehension of complex relationships and observation of patterns within multivariate data. Each principal component represents a linear combination of the original variables, and by plotting these data points along the axes we can observe if they cluster, overlap or separate. The

PCA generated eigenvectors and eigenvalues, enabling the interpretation of variance patterns within the dataset. Specifically, the separation or clustering of survey methods and the overlap of fish species across these methods were evaluated based on the extracted principal components. This approach provided a visual and quantitative means to explore potential associations and variations between methodological factors and fish-related outcomes. The PCA included all fish species that were observed on 5% (minimum of 3 out of the 60 total) or greater of all surveys.

Community Science Analysis

Descriptive statistics and PERMANOVA were conducted to assess potential differences in mean species richness, size, and counts. To enhance the robustness of our analysis, the size and abundance data underwent bootstrapping. By generating multiple datasets through bootstrapping, we aimed to ensure a more accurate assessment of size and abundance differences, thereby strengthening the validity of our findings.

All statistical analyses were processed in R Software (Version 4.1.1.) All statistical modeling was conducted using the lme4, vegan, FactoMiner, and facto extra package (Pinheiro & Bates 2000, Irnawati et al. 2021). All plots were created with ggplot2 package and tables were generated using sjPlot package (Wickham 2009).

Results

Species Richness

A total of 21 families and 81 instantaneous species were observed (Appendix Table A1). Analysis of species richness revealed a notable increase during the initial five minutes of the surveys, after which the curve reached a plateau over the subsequent 10 minutes (Appendix Figure A1). Among the survey methods involving a diver (OC SCUBA, freediving, snorkel), 76% of the species were instantaneously recorded within the first five minutes of the survey period, distributed as follows: freedive = 81%, OC SCUBA = 76%, and snorkel = 74%. In the case of the

RUV method, 92% of the species were observed instantaneously within the first five minutes. Overall, mean species richness was significantly different among the survey methods (pseudo $F_{3,56} = 6.59$, $R^2 = 0.26$, p-value = 0.002). Freediving yielded the highest mean species richness at 23 ± 3 species, while RUV surveys generated the lowest at 18 ± 2 species (Figure 2A). Pairwise t-tests indicated statistically significant differences in species richness between freedive and RUV (p-value = 0.001), OC SCUBA and RUV (p-value = 0.020), and snorkel and freedive (p-value = 0.048). However, no significant differences were observed between OC SCUBA and freedive (p-value = 1.000), OC SCUBA and snorkel (p-value = 0.471), and snorkel and RUV (p-value = 1.000). No significant differences were detected in mean species richness between the morning and afternoon survey periods (Figure 2B) (pseudo $F_{1,58} = 0.25$, $R^2 = 0.004$, p-value = 0.64), the random order of survey occurrence (Figure 2C) (pseudo $F_{7,52} = 1.22$, $R^2 = 0.14$, p-value = 0.30), nor the survey number (Figure 2D) (pseudo $F_{14,45} = 0.54$, $R^2 = 0.14$, p-value = 0.903).

Linear Mixed Effects Model (Size)

Results of multiple models determined random factors (fish species) have the greatest influence on sizing individual fish among surveys. Based on the significantly low Marginal R-sq value of the fixed effects, less than 1% of variation can be attributed to the fixed effects (i.e. survey period, survey order, etc.) among methods with all species included in the model (Table 1). Size estimates among methods demonstrated a consistent trend across models with a negative estimate for freedive, and positive estimate for RUV and snorkel. A high number of species observed on less than 5% of surveys for each method, such as the *Carcharhinus melanopterus* and small cryptic species, resulted in a large range within the 95% confidence intervals. When these species were excluded from the model, the range in confidence interval decreased, however a strong right skew appears in RUV. The skew is a result of large individuals of one single species, S*carus rubroviolaceus*. Excluding *Scarus rubroviolaceus* from the model resulted in non-skewed estimates and reduced range in confidence interval among methods. As species with a low rate of observation among methods and *Scarus rubroviolaceus* were removed, the accuracy and precision of predicted values among models increased (Figure 3).

Linear Mixed Effects Model (Abundance)

Results of multiple models determined random factors (fish species) have the greatest relationship in explaining the variance of counting individuals among methods. Based on the significantly low Marginal R-sq value of the fixed effects, less than 2% of variation can be attributed to the sizing among the best-fit models (Table 2). Count estimates among methods demonstrated a consistent trend across models with a negative estimate for freedive, and positive estimate for RUV and Snorkel. Again, a high number of species observed for many on less than 5% of surveys for each method resulted in a large range in 95% confidence intervals. When these species were excluded from the model, the range in confidence interval decreased, however a significant difference was still detected with RUV and freedive. The significant difference is a result of the high observation rates of schooling fishes, such as *Acanthurus nigrofuscus* and *Zebrasoma flavescens* (Figure 4).

Principal Component Analysis

The eigenvalues corresponding to each component provide the percentage of variance captured by the data along those dimensions. The first component (PCA1) exhibited an eigenvalue of 3.51, accounting for 7.64% of the total variance, while the second component (PCA2) possessed an eigenvalue of 3.29, contributing to 7.16% of the total variance. The cumulative percentages of variance revealed by these components was 14.80%, respectively. The resulting PCA plot unveiled a notable clustering of all four methods, suggesting a remarkable similarity among them (Figure 5A). Nearly all species demonstrated no significant change in detection between methods (Figure 5B). The divergence observed can be attributed to the presence of the species *Abudefduf abdominalis*, which were predominantly associated with snorkel surveys and conversely, *Chromis vanderbilti*, *Halichoeres ornatissimus,* and *Stethojulis balteata* which were predominantly absent during snorkel surveys due to their cryptic behavior and small size. When these influential species were removed from the PCA, there was an increase in overlap and even greater clustering of the centroid points among methods (Figure 5C). These specific species occurrences likely contribute to the variation observed in the separation of the snorkel method in the PCA plot.

Community Science Video Surveys

Bootstrapping results determined a significant difference in sizing among community scientists in comparison to the reference observer ($F_{3,1296}$ = 3.85, p-value = 0.009). Pairwise t-tests indicated significant difference between only one observer and the reference observer (Figure 6A). There was no significant difference among abundance counts by community science observers in comparison to the reference observer ($F_{3,1296}$ = 1.63, p-value = 0.181) (Figure 6B).

Discussion

This study introduces an innovative RUV approach for characterizing fish assemblages and highlights the utility of snorkeling and freediving-based surveys as adaptable alternatives for coral reef fish surveys across Hawaiʻi and the broader Pacific region. The findings underscore the importance of revisiting potential biases inherent in OC SCUBA-based surveys (Williams et al. 2006; Colvocoresses & Acosta 2007; Dickens et al. 2011; Pyle et al. 2015; Gray et al. 2016). While numerous studies have reported increases in fish abundance with alternative methods like closed circuit rebreather (CCR) SCUBA or diver-operated video surveys, it's essential to consider the factor of contextual differences between study sites (Lindfield et al. 2014; Gray et al. 2016; Schramm et al., 2020; Larson 2022). The controlled environment of this study, where all surveys were conducted at the same location and under the same conditions, isolates the methodological factor as the primary driver of survey outcomes and variability among the methods. Unlike previous studies that span spatial gradients, this research eliminates confounding variables, such as habitat variability and site-specific fishing pressures (Watson et al. 2005, 2010). The findings provide strong evidence that all methods proposed are complementary to one another and can be summed up as follows. The majority of fishes show no change in observation among methods, other than cryptic species such as *Chromis vanderbilti, Halichoeres ornatissimus,* and *Stethojulis balteata* which are best observed on OC SCUBA and freedive, and surface-dwelling fishes such as *Abudefduf abdominalis* are best observed on snorkel. Overall, sizing is comparable among all methods and RUV is best for observing large herbivorous species such as *Scarus rubroviolaceus.* Freedive and OC SCUBA are best for counting cryptic species, and snorkel and OC SCUBA are best for counting schooling species (Appendix Table B1).

The species saturation curve displayed a plateau at the 5-minute mark for the three methods involving divers, contributing to 77% of species detection. Likewise, RUV displayed a plateau at 5 minutes providing strong evidence that the programmed rotation of the camera system is consistent with a diver and was set at a reasonable rate for observation. Among the most frequently observed species was *Acanthurus nigrofuscus* observed on all 60 surveys, and *Zebrasoma flavescens* on 59 out of the 60 surveys, followed by *Naso literatus* and *Ctenochaetus strigosus*. The Surgeonfish (Acanthuridae) were the most diverse family observed with 16 species being recorded, followed by wrasses (*Labridae)* with 12 species, and butterflyfishes (*Chaetodontidae*) with 11 species. Notably, *Chromis vanderbilti* exhibited the most pronounced methodological divergence most likely due to its size. Ranging from 2 to 4 cm, this species challenged confident identification by divers from the surface during snorkel surveys. Nevertheless, divers were able to identify these fish at the family level and note their high abundance, as they ranked as the most prevalent species across all surveys with a mean abundance of 139 individuals per survey. Other species, including *Abudefduf abdominalis*, also demonstrated slight separation due to their behavior, contingent on the surveyor's position within the water column. Notably at this site, these species often inhabit surface waters, extending beyond the cylinder's confines. However, during snorkel surveys they displayed downward movement toward the reef within the cylinder likely influenced by the diver's presence. Several species were excluded from the PCA analysis due to their low detection rates and frequent occurrence of zero values, stemming from their small size, cryptic behavior, or transient tendencies. It is important to acknowledge that these findings from this study do not pertain to cryptic and highly transient species due to their infrequent detection. The timing of surveys emerged as a pivotal consideration, preventing potential biases linked to the morning and evening herbivore activity. The consistent species richness observed across various times, survey order, and methods strongly supports the chosen survey time as an accurate portrayal of the survey site.

Our investigation into the size estimation among different survey methods highlighted intriguing differences that stem from the varying perspectives and approaches inherent to each technique. Both snorkeling and RUV consistently yielded larger size estimates in comparison to OC SCUBA, while freediving consistently resulted in slightly smaller size estimates. This top-down perspective while snorkeling and wide-angle perspective through the RUV inherently introduces a degree of distortion, as the distance between the observer and the fish can compromise accurate size estimation. Conversely, the unique approach of freediving offers a distinct advantage. By diving beneath the water's surface, freedivers gain a more direct line of sight to the fish and their surroundings similar to OC SCUBA. This perspective enables them to assess the true size of different individuals within the fish assemblage, using their direct observation of fish at close range as a guide for sizing from the surface. The small variance in sizing among methods suggests well trained fish surveyors are capable of sizing fish via each method. Additionally, the results of sizing across community scientists further supported these findings and provided strong evidence for the potential of these innovative approaches. Although there was a significant difference between one community scientist and the reference observer this was not unexpected as experience level is known to result in inconsistent sizing across surveyors and is more likely the cause than the method in this scenario (Williams et al., 2006).

Our investigation into the abundance estimation across different survey methods revealed noteworthy differences obtained through each technique. Both snorkeling and RUV consistently yielded higher abundance estimates compared to OC SCUBA, while freediving resulted in slightly lower abundance estimates. These variations can be primarily attributed to the differences in observer proximity to the fish, as well as the potential for double counting and the absence of a true reference for accurate abundance estimation. While RUV provides insights into relative abundance trends, it may lead to inflated or deflated abundance estimates due to the challenges associated with schooling fishes and lack of ability to decipher between individuals within the school. In contrast, snorkeling and freediving emerge as promising techniques, as they offer a more accurate assessment of abundance by mitigating the risk of overestimation and providing a more direct line of sight from both above and at the bottom. Although the freediving method presents additional physical challenges and potential barriers to entry to conduct the surveys safely and effectively, it opens the door to integrate freedivers and spearfishers into conservation and monitoring activities. It's also noteworthy that the explanation for the decrease in estimates by freediving is in part due to the divide of the cylinder and fish transiting throughout the entirety of the cylinder. This scenario results in the assumption that the partner may have already counted individuals which ultimately leads to individuals being unaccounted for. Results from the community science replicates remained consistent with the findings among methods showing significant difference among surveyors and consistent trends in oversizing by under sizing by individual surveyors. Knowing there is an inherent bias between surveyors, we hope by diversifying the methods in which we collect the data there is potential for reducing this variation and overall improving our estimates as a whole.

The findings of this study emphasize that permanent RUV systems are a valuable investment, enabling long-term monitoring with extensive data collection capabilities. Current limitations involving divers or constant camera repositioning hinder widespread adoption of these techniques (Follana-Berna et al. 2019). The implementation of permanent RUV systems addresses these limitations, offering a resolution for the constraints placed on video observations in coral reef ecosystems. The capability to continuously collect data from these systems without diver involvement or repositioning accelerates the potential for in-depth understanding of reef dynamics. These systems offer the potential for increased observations, providing insight on fish assemblages patterns and capturing seasonal, diel, and ocean-condition-related shifts (Erickson et al. 2022). With the rapid evolving capabilities of on-board artificial intelligence, there is strong potential for real-time species identification and enumeration which would provide an unprecedented number of datasets to enhance monitoring efforts (Ditria et al. 2022). Furthermore, they afford opportunities for novel behavioral insights that could inform management strategies and uncover potential biases present in other monitoring approaches (Aguzzi et al. 2015). Additionally the ability for the community scientists to become proficient with the method suggests the use of the camera system as both a training tool and monitoring method. Due to lack of necessary equipment many young and novice surveyors have access to, utilizing the camera becomes a great alternative to develop accurate species identification, sizing and counting techniques. As technology advances, it's pertinent we keep up in the marine environment and remain relevant regarding innovative methods that can harness current technological capabilities. Overcoming the primary obstacle of being able to mount remote cameras long term, we can now observe the reef in a completely new perspective and this advancement bridges a gap in video-based monitoring, positioning it as a strong contender for improved coral reef management strategies and serve as long-term ocean observation stations.

The integration of snorkeling and freediving as alternative survey methods expands the inclusivity of coral reef assessments. Although previous studies have suggested snorkeling causes a disturbance and potentially inaccurate data, this study finds no evidence in coral reef habitat (Dearden et al. 2010). And while snorkeling accuracy may be variable dependent on depth and visibility in the water, the findings from this study suggest strong evidence of suitability for shallow reef environments (~8 meters in depth) with moderate to excellent visibility (> than 10 meters). Freediving, on the other hand, enhances the depth at which observers can study fish behavior and species composition without the need for SCUBA equipment and opens the door to integrate spearfishers and community scientists. This study represents the foundation for the involvement of freediving with coral reef fish monitoring as there is currently no mention of free diving in any peer reviewed publications. These findings provide the first scientific evidence that spearfishers and community scientists are highly capable and can provide comparable data, therefore should be invited as stakeholders, and greatly utilized in monitoring efforts. With the increase in Community-Based Subsistence Fishing Areas (CBSFA) such as Hā'ena and management plans such as "Try Wait" at Kaʻūpūlehu' in Hawaiʻi, these findings are pivotal in allowing local resource users to become the local resource monitors and provide meaning scientific data to support their existing knowledge (Minton et al. 2015, DAR 2023). Unlike SCUBA diving, which often demands specialized training and certifications, both methods have an anecdotal track record of engaging local and community scientists, enabling broader participation. The empowerment of local stakeholders such as the indigenous peoples, spearfishermen, and avid ocean users, to contribute to monitoring endeavors aligns with community-based conservation initiatives and strengthens the overall effectiveness of resource management (Friedlander et al. 2000; Tissot 2007).

Conclusion

In conclusion, the current state of coral reefs and fisheries calls for innovative and inclusive approaches to monitoring that can bridge the gap between traditional scientific methods and local knowledge. Our study showcases the transformative potential of permanent remote underwater video systems, snorkeling, and freediving in revolutionizing the way we understand and conserve coral reef ecosystems. Through rigorous comparisons, we have demonstrated that these novel methods not only provide accurate and reliable data but also overcome the limitations and biases associated with conventional SCUBA-based surveys. RUV systems, with their ability to continuously monitor reef dynamics, offer a pathway to unprecedented insights into fish behavior, distribution, and seasonal variations. The integration of snorkeling and freediving not only enhances community involvement but also harnesses the profound insights of local stakeholders, who are often excluded from the scientific process. We recommend the increased implementation of snorkel and freediving methods combined into one method to allow divers to survey from the surface and diving to depth to identify cryptic species as needed. Additionally due to the more consistent results of sizing and counting with freediving, the perspective gained by diving to depth throughout the survey has proven pivotal when it comes to having a true perspective and accurate representation throughout the survey. By expanding the options of survey techniques, we are not only advancing our scientific understanding but also empowering communities to take an active role in reef conservation. Our findings emphasize the urgent need for collaboration between researchers, local communities, and policymakers to create a unified and rounded approach to coral reef management. As we stand at the crossroads of environmental challenges and the growing momentum of technological advancements, it is our duty to embrace innovation, inclusivity, and community engagement. By embracing these techniques alongside established methods, researchers and managers can develop a more comprehensive understanding of coral reef fish assemblages, ultimately bolstering conservation strategies and safeguarding these vital marine ecosystems for future generations.

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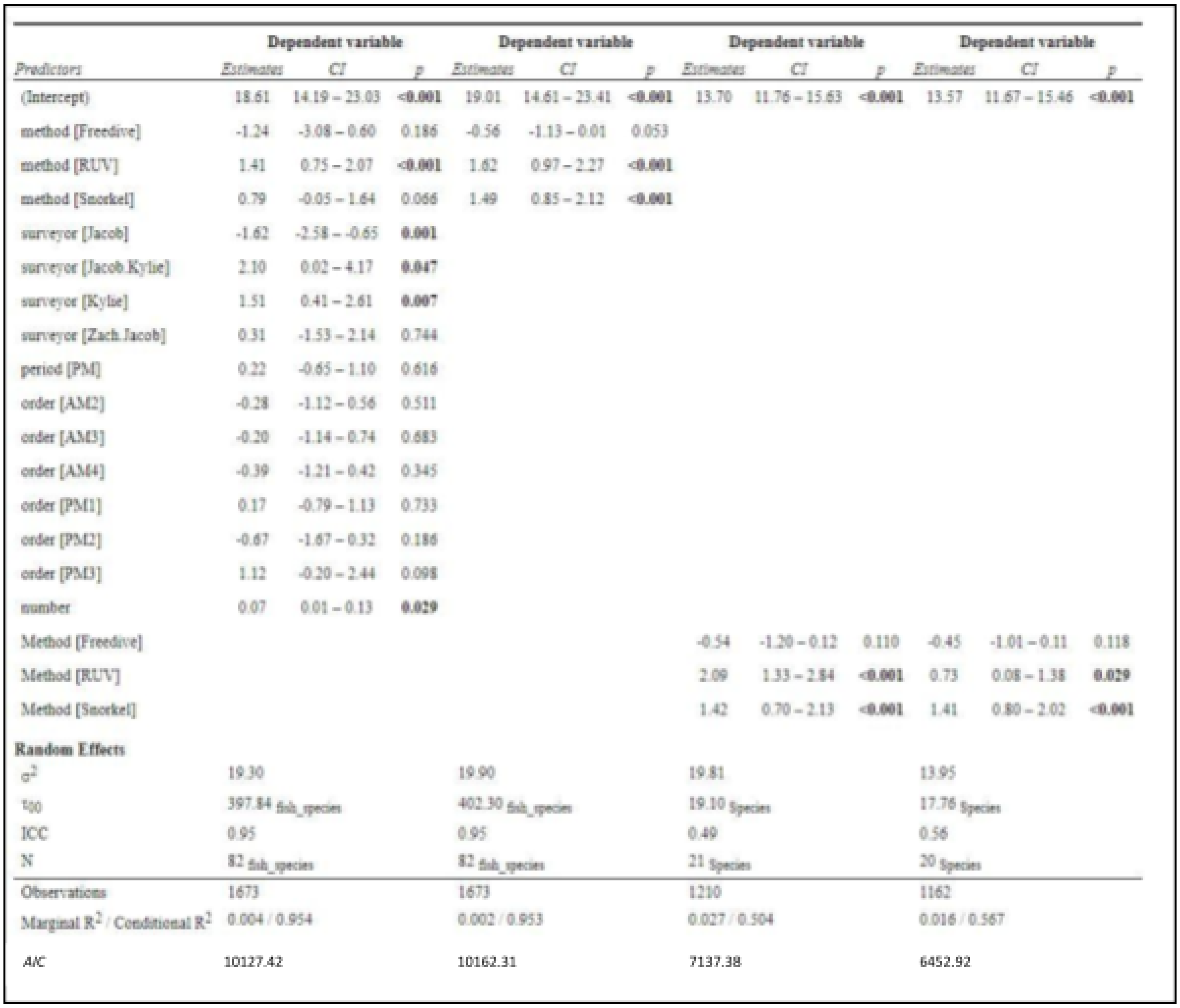
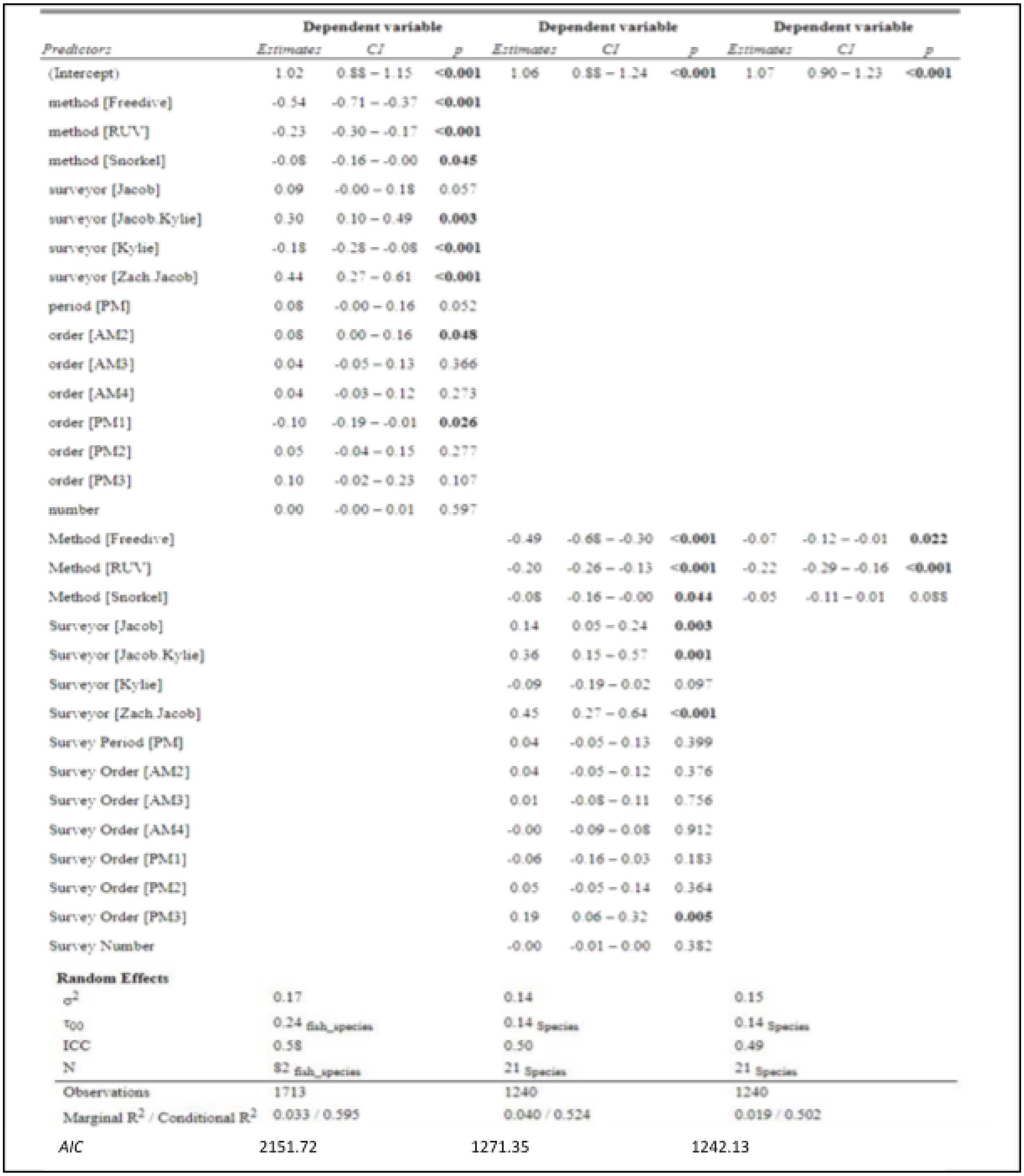
List of Tables

Table 1. Results of linear mixed effects model (LMM) explaining variation in the sizing of fish species among the four survey methods. Model 1 includes all species observed among methods. Model 2 includes all fish species with only random variables in the model, Model 3 excludes species observed in less than three out of fifteen surveys per method, and Model 4 further excludes Scarus rubroviolaceus.



	Dependent variable			**Dependent variable**			**Dependent variable**			**Dependent variable**		
Predictors	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*
(Intercept)	18.61	14.19 – 23.03	**<0.001**	19.01	14.61 – 23.41	**<0.001**	13.70	11.76 – 15.63	**<0.001**	13.57	11.67 – 15.46	**<0.001**
method [Freedive]	-1.24	-3.08 – 0.60	0.186	-0.56	-1.13 – 0.01	0.053						
method [RUV]	1.41	0.75 – 2.07	**<0.001**	1.62	0.97 – 2.27	**<0.001**						
method [Snorkel]	0.79	-0.05 – 1.64	0.066	1.49	0.85 – 2.12	**<0.001**						
surveyor [Jacob]	-1.62	-2.58 – -0.65	**0.001**									
surveyor [Jacob.Kylie]	2.10	0.02 – 4.17	**0.047**									
surveyor [Kylie]	1.51	0.41 – 2.61	**0.007**									
surveyor [Zach.Jacob]	0.31	-1.53 – 2.14	0.744									
period [PM]	0.22	-0.65 – 1.10	0.616									
order [AM2]	-0.28	-1.12 – 0.56	0.511									
order [AM3]	-0.20	-1.14 – 0.74	0.683									
order [AM4]	-0.39	-1.21 – 0.42	0.345									
order [PM1]	0.17	-0.79 – 1.13	0.733									
order [PM2]	-0.67	-1.67 – 0.32	0.186									
order [PM3]	1.12	-0.20 – 2.44	0.098									
number	0.07	0.01 – 0.13	**0.029**									
Method [Freedive]							-0.54	-1.20 – 0.12	0.110	-0.45	-1.01 – 0.11	0.118
Method [RUV]							2.09	1.33 – 2.84	**<0.001**	0.73	0.08 – 1.38	**0.029**
Method [Snorkel]							1.42	0.70 – 2.13	**<0.001**	1.41	0.80 – 2.02	**<0.001**
Random Effects												
σ^2	19.30			19.90			19.81			13.95		
τ_{00}	397.84 $_{\text{fish\_species}}$			402.30 $_{\text{fish\_species}}$			19.10 $_{\text{Species}}$			17.76 $_{\text{Species}}$		
ICC	0.95			0.95			0.49			0.56		
N	82 $_{\text{fish\_species}}$			82 $_{\text{fish\_species}}$			21 $_{\text{Species}}$			20 $_{\text{Species}}$		
Observations	1673			1673			1210			1162		
Marginal R^2 / Conditional R^2	0.004 / 0.954			0.002 / 0.953			0.027 / 0.504			0.016 / 0.567		
AIC	10127.42			10162.31			7137.38			6452.92		

Table 2. Results of Log-transformed linear mixed effects model (LMM) explaining variation in the counting of fishes among the four survey methods. Model 1 includes all species observed and all fixed variables. Model 2 excludes species observed in less than three out of fifteen surveys per method and includes all fixed variables. Model 3 excludes species observed in less than three out of fifteen surveys per method and excludes all fixed variables.



	Dependent variable			**Dependent variable**			**Dependent variable**		
Predictors	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*	*Estimates*	*CI*	*p*
(Intercept)	1.02	0.88 – 1.15	**<0.001**	1.06	0.88 – 1.24	**<0.001**	1.07	0.90 – 1.23	**<0.001**
method [Freedive]	-0.54	-0.71 – -0.37	**<0.001**						
method [RUV]	-0.23	-0.30 – -0.17	**<0.001**						
method [Snorkel]	-0.08	-0.16 – -0.00	**0.045**						
surveyor [Jacob]	0.09	-0.00 – 0.18	0.057						
surveyor [Jacob.Kylie]	0.30	0.10 – 0.49	**0.003**						
surveyor [Kylie]	-0.18	-0.28 – -0.08	**<0.001**						
surveyor [Zach.Jacob]	0.44	0.27 – 0.61	**<0.001**						
period [PM]	0.08	-0.00 – 0.16	0.052						
order [AM2]	0.08	0.00 – 0.16	**0.048**						
order [AM3]	0.04	-0.05 – 0.13	0.366						
order [AM4]	0.04	-0.03 – 0.12	0.273						
order [PM1]	-0.10	-0.19 – -0.01	**0.026**						
order [PM2]	0.05	-0.04 – 0.15	0.277						
order [PM3]	0.10	-0.02 – 0.23	0.107						
number	0.00	-0.00 – 0.01	0.597						
Method [Freedive]				-0.49	-0.68 – -0.30	**<0.001**	-0.07	-0.12 – -0.01	**0.022**
Method [RUV]				-0.20	-0.26 – -0.13	**<0.001**	-0.22	-0.29 – -0.16	**<0.001**
Method [Snorkel]				-0.08	-0.16 – -0.00	**0.044**	-0.05	-0.11 – 0.01	0.088
Surveyor [Jacob]				0.14	0.05 – 0.24	**0.003**			
Surveyor [Jacob.Kylie]				0.36	0.15 – 0.57	**0.001**			
Surveyor [Kylie]				-0.09	-0.19 – 0.02	0.097			
Surveyor [Zach.Jacob]				0.45	0.27 – 0.64	**<0.001**			
Survey Period [PM]				0.04	-0.05 – 0.13	0.399			
Survey Order [AM2]				0.04	-0.05 – 0.12	0.376			
Survey Order [AM3]				0.01	-0.08 – 0.11	0.756			
Survey Order [AM4]				-0.00	-0.09 – 0.08	0.912			
Survey Order [PM1]				-0.06	-0.16 – 0.03	0.183			
Survey Order [PM2]				0.05	-0.05 – 0.14	0.364			
Survey Order [PM3]				0.19	0.06 – 0.32	**0.005**			
Survey Number				-0.00	-0.01 – 0.00	0.382			
Random Effects									
σ^2	0.17			0.14			0.15		
τ_{00}	0.24 $_{\text{fish\_species}}$			0.14 $_{\text{Species}}$			0.14 $_{\text{Species}}$		
ICC	0.58			0.50			0.49		
N	82 $_{\text{fish\_species}}$			21 $_{\text{Species}}$			21 $_{\text{Species}}$		
Observations	1713			1240			1240		
Marginal R^2 / Conditional R^2	0.033 / 0.595			0.040 / 0.524			0.019 / 0.502		
AIC	2151.72			1271.35			1242.13		

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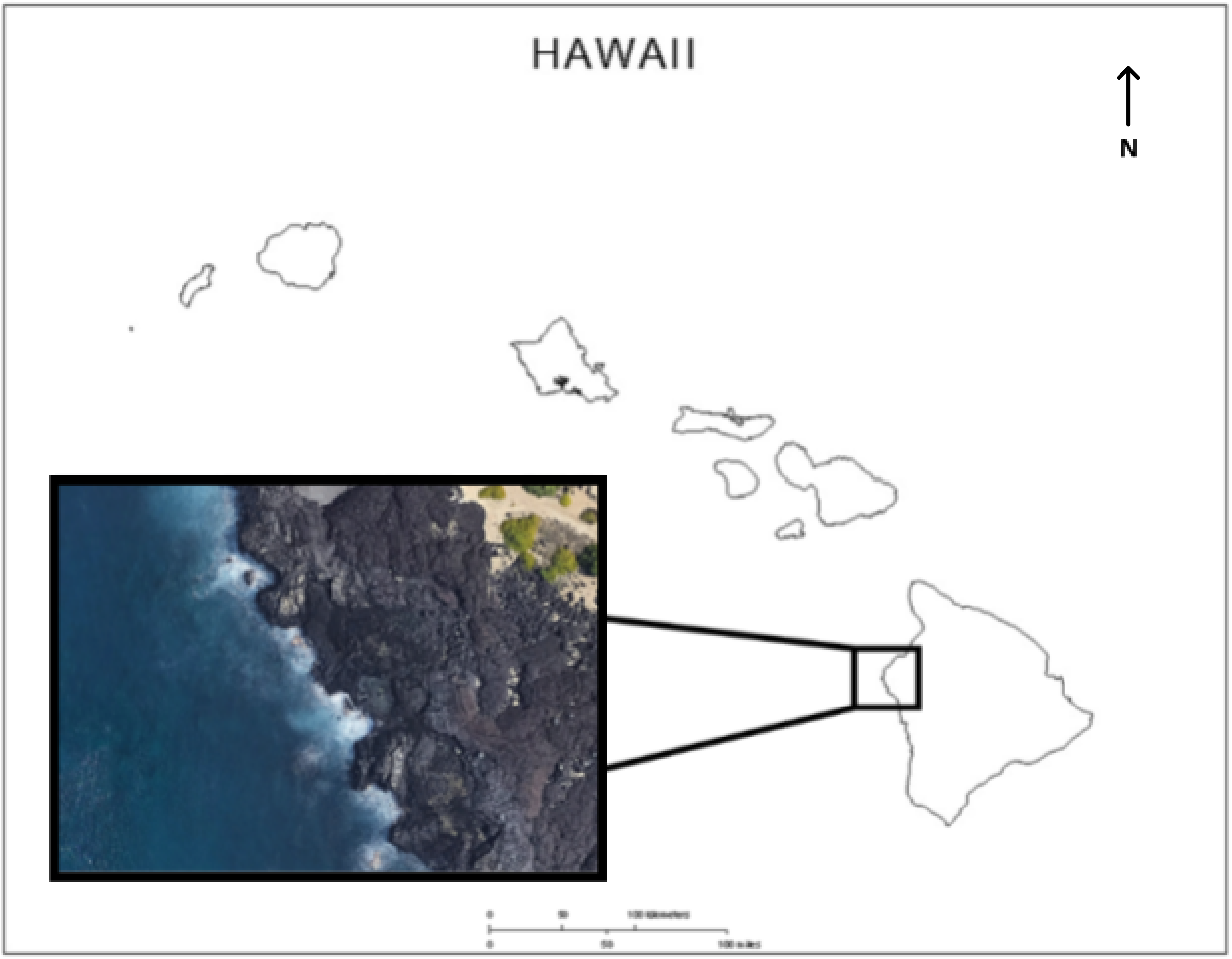

Figure 1. Map depicting the geographic location of the study site, located at Keāhole Point in Kailua Kona, Hawaiʻi.

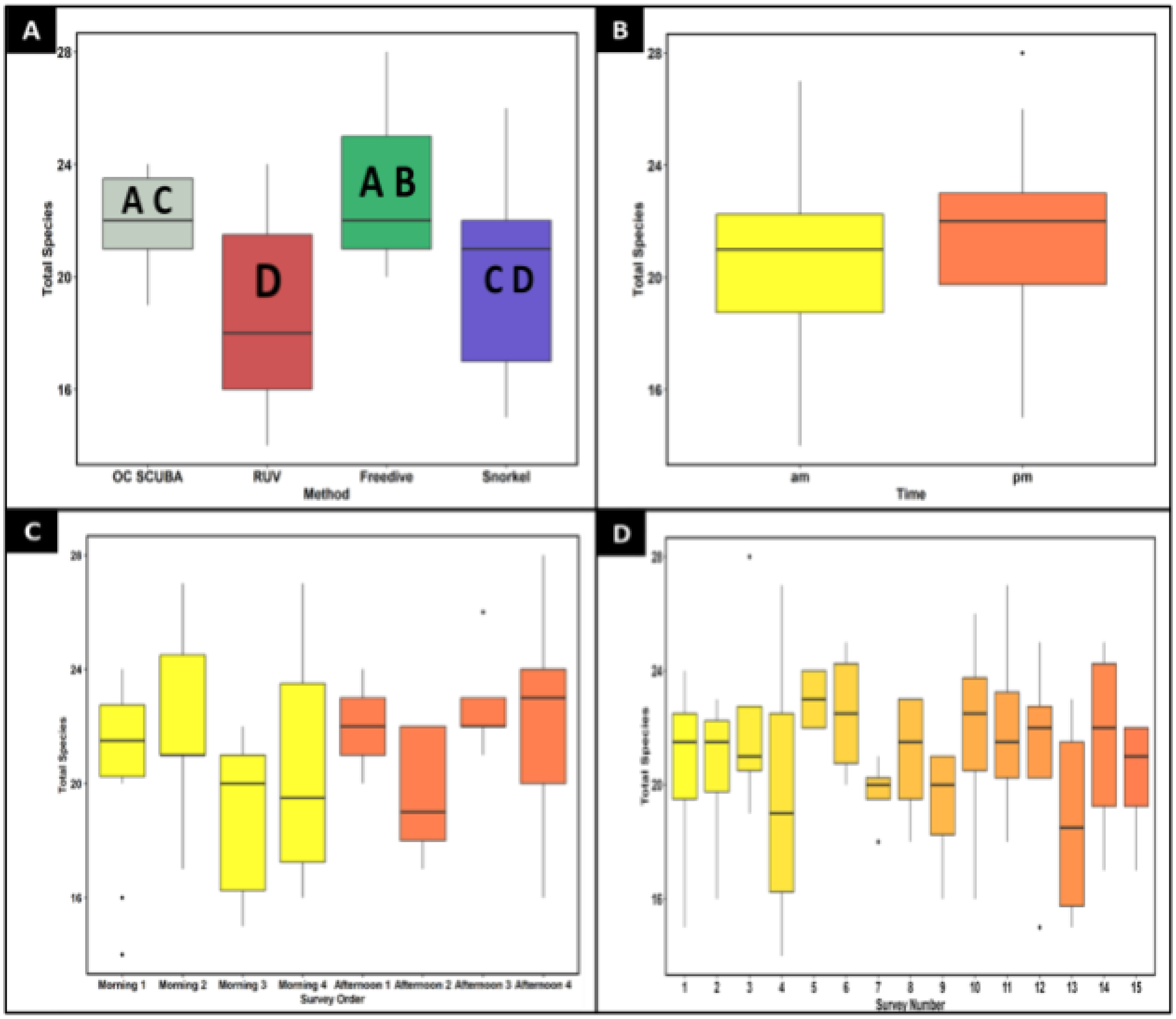

Figure 2. Boxplots showing variation of the mean ± standard deviation of species richness (A) among four survey methods, (B) between survey time periods, (C) the order of survey methods, and (D) among survey samples. Letter groupings are results of PERMANOVA pairwise comparison and represent significantly different groupings of methods.

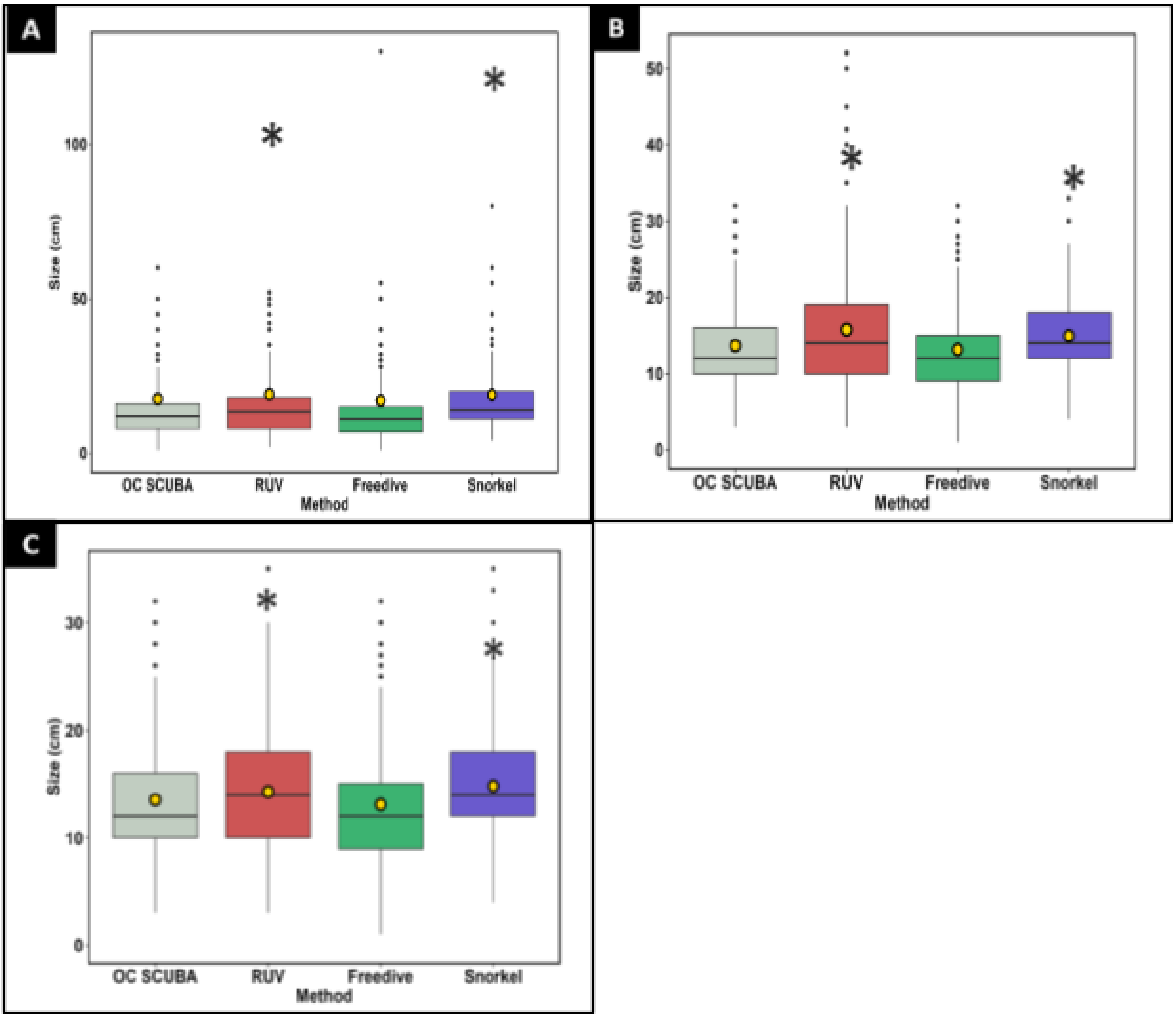

Figure 3. Box plots showing (A) mean size and size variance of all fish species observed among different survey methods (RUV p-value <0.001, freedive p-value <0.001, snorkel p-value <0.001), and (B) mean size and variance after removing species observed less than five percent of all surveys (RUV p-value <0.001, freedive p-value= 0.110, snorkel p-value <0.001), and (C) mean size and variance after removing *Scarus rubroviolaceus* (RUV p-value <0.029, freedive p-value= 0.110, snorkel p-value <0.001), . The yellow dot positioned on the chart represents the predicted mean size derived from the linear mixed effects model and asterisks denote significant difference.

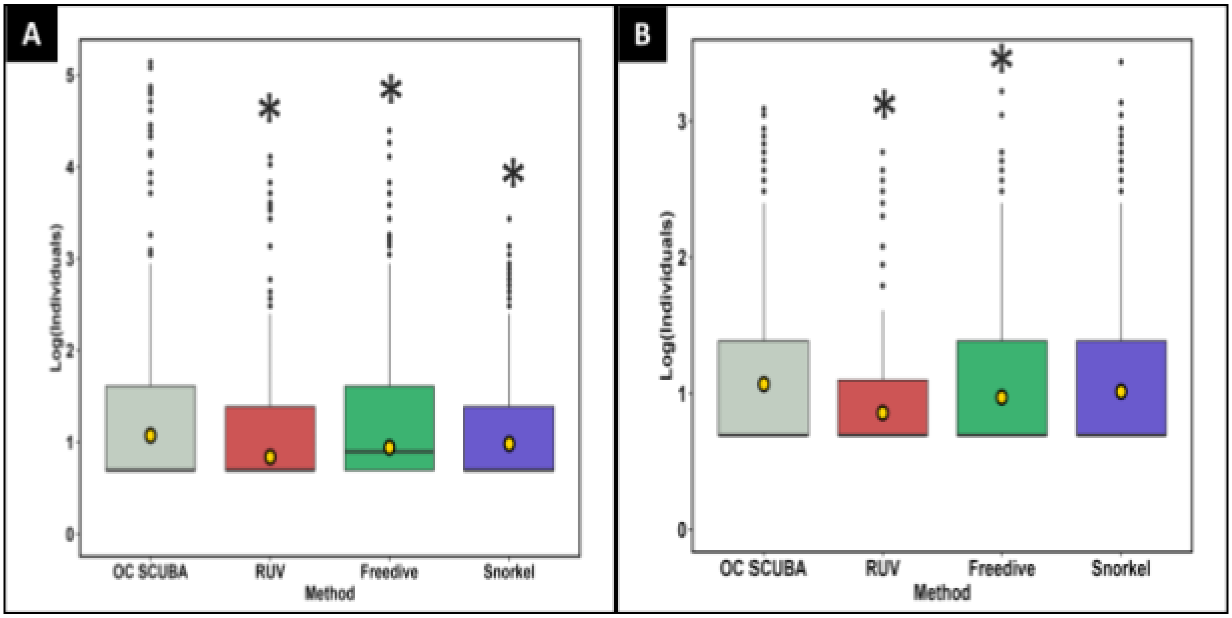

Figure 4. Box plots showing (A) Log-transformed mean number of individuals and variance of all fish species observed among different survey methods (n=15, RUV p-value= 0.00, freedive p-value= 0.00, snorkel p-value= 0.00), and (B) Log-transformed mean number of individuals and variance after removing species observed less than five percent of all surveys (n=15, RUV p-value= 0.00, freedive p-value= 0.00, snorkel p-value= 0.08). The red dot positioned on the chart represents the predicted mean number of individuals derived from the linear mixed effects model.

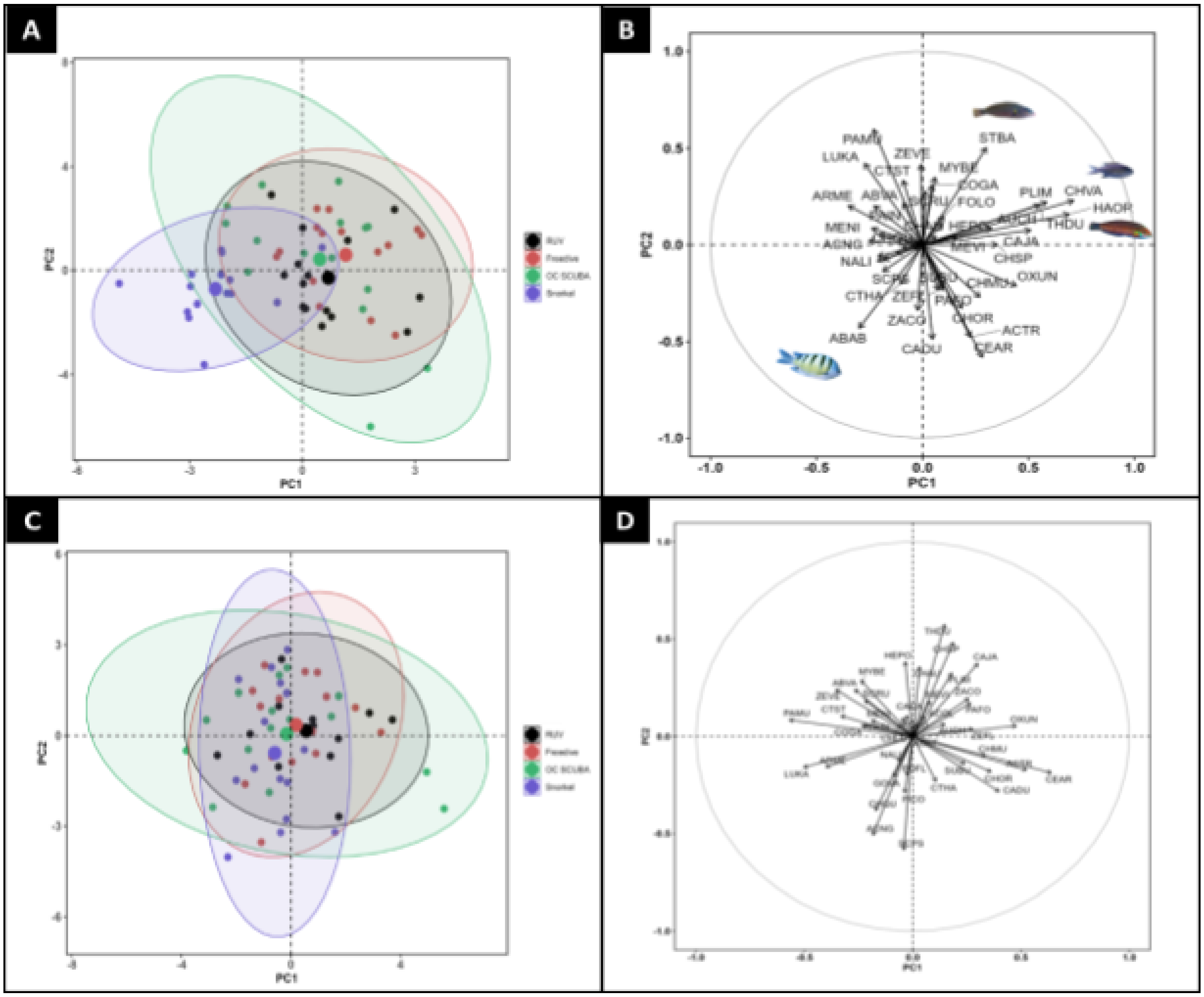

Figure 5. (A) Principal component analysis plot of fish survey methods and (B) contribution of individual fish species to Principal components. (C) Principal component analysis plot of fish survey methods excluding the most influential species (*Stethojulis balteata, Chromis vanderbilti, Halichoeres ornatissimus, Abudefduf abdominalis*) and (D) contribution of individual fish species to Principal components.

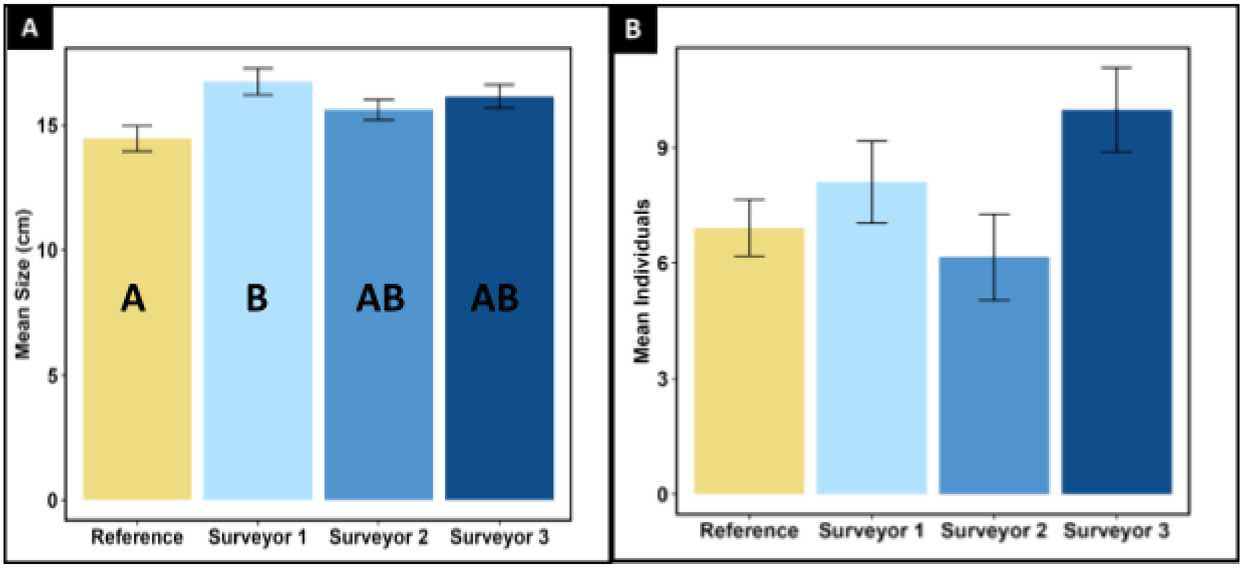

Figure 6. Results of PERMANOVA pairwise carried out on the bootstrapped (r=1300) mean ± standard deviation of (A) sizing among community scientists, and (B) abundance counts among community scientists.

Appendix A

Table A1. List of all species observed among the four survey methods utilized at Keahole Point, Hawaiʻi.

Boxfishes (*Ostraciidae*)	
(OSME)	*Ostracion meleagris*
Butterflyfishes (*Chaetodontidae*)	
(CHAU)	*Chaetodon auriga*
(CHLU)	*Chaetodon lunula*
(CHMI)	*Chaetodon miliaris*
(CHMU)	*Chaetodon multicinctus*
(CHOR)	*Chaetodon ornatissimus*
(CHQU)	*Chaetodon Quadramaculatus*
(CHRE)	*Chaetodon reticulatus*
(FOFL)	*Forciper flavissimus*
(FOLO)	*Forciper longirostris*
(HEPO)	*Hemitaurichthys polylepis*
Damselfishes (*Pomacentridae*)	
(ABAB)	*Abudefduf abdominalis*
(ABSO)	*Abudefduf sordidus*
(ABVA)	*Abudefduf vaigiensis*
(CHHA)	*Chromis hanui*
(CHOV)	*Chromis ovalis*
(CHVA)	*Chromis vanderbilti*
(PLIM)	*Plectroglyphidodon imparipennis*
(PLJO)	*Plectroglyphidodon johnstonianus*
Eels (Muraenidae)	
(GYFL)	*Gymnothorax flavimarginatus*
(GYME)	*Gymnothorax meleagris*
(ECNE)	*Echidna nebulosi*
Emperorfishes (*Lethrinidae*)	
(MOGR)	*Monotaxis grandoculis*
Filefishes (*Microdesmidae*)	
(CADU)	*Cantherhines dumerilii*
(PEAS)	*Pervagor aspricaudus*
Goatfishes (*Mullidae*)	
(MUFL)	*Mulloidichthys flavolineatus*
(PACY)	*Parapeneus cyclostomus*
(PAIN)	*Parapeneus insularis*
(PAMU)	*Parapeneus multifasciatus*
Groupers (*Serranidae*)	
(CEAR)	*Cephalopholis argus*
Hawkfishes (*Cirrhitidae*)	
(PAAR)	*Paracirrhites arcatus*
(PAFO)	*Paracirrhites forsteri*
Jacks (*Carangidae*)	
(CAME)	*Caranx melampygus*
(SCLY)	*Scomberoides lysan*
Moorish Idol (*Zanclidae*)	
(ZACO)	*Zanclus cornutus*
Parrotfishes (*Scaridae*)	
(CACA)	*Calotomus carolinus*
(CHPE)	*Chlorurus perspicillatus*
(CHSP)	*Chlorurus spilurus*
(SCPS)	*Scarus psittacus*
(SCRU)	*Scarus rubroviolaceus*
Porcupinefishes (*Diodontidae*)	
(DIHY)	*Diodon hystrix*
Pufferfishes (Tetraodontidae)	
(ARTH)	*Arothron meleagris*
(CAAM)	*Canthigaster amboinensis*
(CAJA)	*Canthigaster jactator*
Sharks (*Carcharhinidae*)	
(CARME)	*Carcharhinus melanopterus*
Snappers (Lutjanidae)	
(APFU)	*Aphareus furca*
(LUKA)	*Lutjanus kasmira*
Soldierfishes (*Holocentridae*)	
(MYBE)	*Myripristis berndti*
Surgeonfishes (*Acanthuridae*)	
(ACBL)	*Acanthurus blochii*
(ACLE)	*Acanthurus leucopareius*
(ACNF)	*Acanthurus nigrofuscus*
(ACNG)	*Acanthurus nigroris*
(ACOL)	*Acanthurus olivaceus*
(ACTH)	*Acanthurus thompsoni*
(ACTR)	*Acanthurus triostegus*
(ACXA)	*Acanthurus xanthopterus*
(CTHA)	*Ctenochaetus hawaiiensis*
(CTST)	*Ctenochaetus strigosus*
(NALI)	*Naso lituratus*
(NAUN)	*Naso unicornis*
(ZEFL)	*Zebrasoma flavescens*
(ZEVE)	*Zebrasoma velifer*
Triggerfishes (*Balistidae*)	
(MENI)	*Melichthys niger*
(MEVI)	*Melichthys vidua*
(RHRE)	*Rhinecanthus rectangulus*
(SUBU)	*Sufflamen bursa*
Wrasses (*Labridae*)	
(COGA)	*Coris gaimard*
(GOVA)	*Gomphosus varius*
(HAOR)	*Halichoeres ornatissimus*
(LAPH)	*Labroides phthirophagus*
(OXUN)	*Oxycheilinus unifasciatus*
(PSOC)	*Pseudocheilinus octotaenia*
(STBA)	*Stethojulis balteata*
(THBA)	*Thallasoma ballieui*
(THDU)	*Thalassoma duperrey*
(THPU)	*Thalassoma purpureum*
(THQU)	*Thalassoma quinquevittatum*

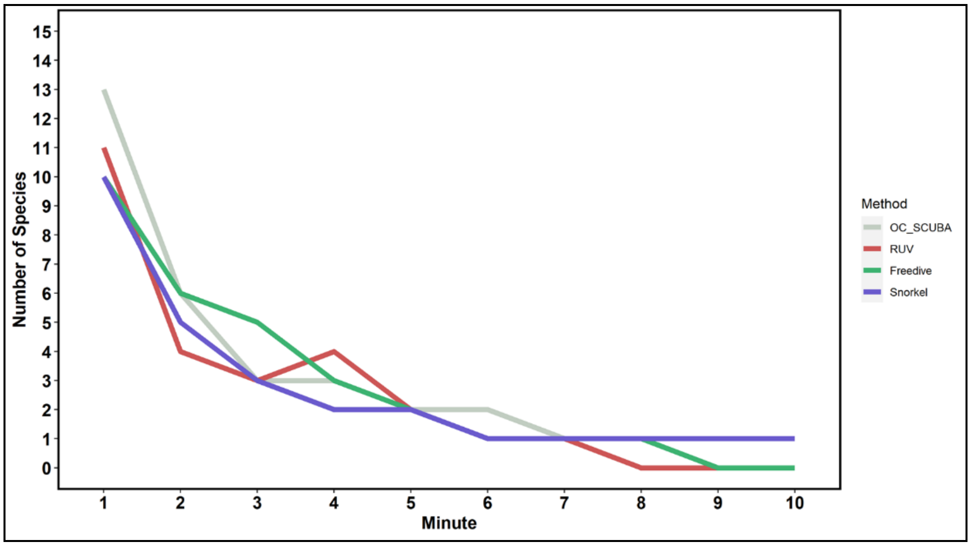

Figure A1. Species saturation curve depicting the accumulation of observed species over a 10-minute survey period among all four methods.

Appendix B

Table B1. Summary list of Pros and Cons of each individual method based on results of PERMANOVA, Linear Mixed Effects Model, and Principal Component Analysis.

OC Scuba PROS

- ✓ Worldwide application and established methods among surveyors.
- ✓ Best method for observing cryptic species such as *Chromis vanderbilti* and Labridae species.
- ✓ Best method for sizing across all species in general.

OC Scuba CONS

- − Barriers to entry due to required certifications and equipment costs.
- − Results in potential bias results due to a change in fish behavior as a response to bubbles and large surveyor.
- − Limited by weather conditions and often requires access to a boat to sufficiently survey multiple site.

RUV PROS

- ✓ Increased observation of large herbivorous fishes such as *Scarus rubroviolaceus.*
- ✓ Sizing of fishes is consistent across species when compared to OC Scuba.
- ✓ Allows for increased observations that are not limited by weather and ocean conditions and provide on-board AI with large datasets.
- ✓ Real time ecological observations that can be matched to oceanographic data and serve as ocean observation stations.

RUV CONS

- − Increased difficulty with counting cryptic species such as *Chromis vanderbilti.*
- − Increased difficulty with counting schooling species such as *Acanthurus nigrofuscus.*
- − Time requirement for troubleshooting and continuous maintenance of camera system.

Freedive PROS	Freedive CONS
✓ Increased observation of cryptic species such as *Chromis vanderbilti* and Labridae species.	⊖ Increased difficulty with counting schooling species such as *Acanthurus nigrofuscus*.
✓ Complimentary method for counting and sizing cryptic species	⊖ Increased barrier to entry due to physical demand and safety precautions.
✓ Allows for involvement of spearfishers and fishers who rely on the resource for subsistence food source.	⊖ Limited by weather conditions and accessibility of dive site.

Snorkel PROS	Snorkel CONS
✓ Increased observation in surface dwelling fishes such as *Abudefduf abdominalis*.	⊖ Increased difficulty with observing, sizing, and counting cryptic species such as *Chromis vanderbilti*.
✓ Complimentary method for counting schooling fishes such as *Acanthurus nigrofuscus*.	⊖ Ability of survey is limited by ocean conditions such as depth and visibility.
✓ Allows for increased participation by community scientists.	⊖ Limited by weather conditions and accessibility of dive site.