

RESEARCH ARTICLE

Correspondence among multiple methods provides confidence when measuring marine protected area effects for species and assemblages

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Funding information

California Department of Fish and Wildlife; California Ocean Protection Council; California Sea Grant, University of California, San Diego; David and Lucile Packard Foundation

Handling Editor: Rucha Karkarey

Abstract

1. Marine protected areas (MPAs) have become a popular tool utilised across global oceans to achieve a variety of conservation goals. Because the reasons for MPA implementation can differ, it is imperative that resource managers design and execute management strategies that allow them to effectively assess MPA performance relative to the goals they set.
2. We compared three MPA monitoring techniques commonly utilised to survey groundfish populations across different depth strata of temperate rocky reef habitat: underwater visual census (0–20 m), scientific hook and line fishing (10–50 m) and baited remote underwater video (30–100 m). We compared the strength and direction of standardised metrics, including response ratios, diversity indices and community structure, examining results through the lens of MPA performance.
3. While each of our monitoring techniques detected similar MPA effects on groundfish biomass and density aggregated across species, MPA effects for individual species varied across methods.
4. Each technique was shown to survey distinct groundfish community assemblages with varying levels of species diversity and richness.
5. *Synthesis and applications.* While each technique was found to measure similar general trends in marine protected area (MPA) performance over time, we found compelling evidence that the utilization of multiple techniques allows managers to create the most comprehensive, effective and inclusive MPA monitoring regimes.

KEYWORDS

adaptive management, baited remote underwater video, collaborative fishing, fish, long-term monitoring, marine conservation, marine protected area, underwater visual census

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1 | INTRODUCTION

In recent decades, the combined anthropogenic stressors of climate change and overfishing have negatively impacted oceans worldwide, resulting in widespread species loss and changes to ecosystem structure, resilience, and function. One of the most common conservation strategies used by resource managers to safeguard against these and other negative outcomes is the implementation of marine protected areas (MPAs), designated areas of the ocean that restrict extractive human activities. A variety of conservation goals are addressed by the creation of MPAs and ecologically connected MPA networks, such as enhancing biodiversity, preserving ecosystem structure and function, and replenishing fished stocks (Sullivan-Stack et al., 2022). Although often controversial upon implementation, research shows that properly designed and managed MPAs are capable of driving increases in the abundance and biomass of targeted (defined here as 'fished') species both within and outside of their boundaries (Caselle et al., 2015; Giakoumi et al., 2017; Hackradt et al., 2014; Hamilton et al., 2010; Lenihan et al., 2021; Stobart et al., 2009; Ziegler et al., 2022). As a result, initiatives that call for the global expansion of protected areas have gathered increased support, such as the '30×30' initiative, which calls for globally increasing protections on land and at sea to at least 30% by the year 2030 (Dinerstein et al., 2019).

As the number of global MPAs increases in response to such initiatives, so too will the number of resource managers engaging in adaptive management, an iterative approach to resource management that aims to address conservation goals through a systematic feedback loop (Allen et al., 2011). While there are a variety of factors that contribute to adaptive management, long-term monitoring is key for the iterative process to be successful (Day, 2008; Hughes et al., 2017; Nickols et al., 2019). Effective monitoring can be challenging, as MPAs and MPA networks are often designed with the overarching goal of ecosystem preservation, necessitating the protection of a variety of habitats, each of which might require a different methodology to monitor. Monitoring efforts are often constrained by the availability of technologies, expertise, and funding; hence MPA managers are required to weigh a variety of factors when designing a management plan.

Three methods commonly utilised to monitor benthic fish assemblages in global MPAs are diver-based underwater visual census (UVC; Dickens et al., 2011), baited remote underwater videos (BRUVs; Willis & Babcock, 2000), and collaborative fisheries surveys (Yochum et al., 2011). A large body of research in a variety of coastal marine systems has documented the benefits and biases of each of these methods (Colton & Swearer, 2010; Gillett et al., 2012; Hernan et al., 2022; Karnauskas & Babcock, 2012; Lowry et al., 2012; Murphy & Jenkins, 2010); Pelletier et al., 2011; Rassweiler et al., 2020; Starr et al., 2010; Watson et al., 2005; Watson & Huntington, 2016). However, the majority of these studies were purely methodological, for the purpose of refining the use of a given tool or method, and most are short in duration and occurred at single locations or small spatial scales. While some studies compare one tool to another,

notably few have compared multiple methods through the lens of a particular management question, such as MPA effectiveness, and fewer still have looked at species-specific responses to management action. This is counterintuitive to real-world applications of these methods, where data from monitoring programs are analysed across a variety of temporal and spatial scales and resource managers are often interested in species-specific trends.

In this study, we compare results from three long-term MPA monitoring programs used in the state of California, USA, to monitor fish assemblages across a range of depths throughout a large MPA network: UVC, BRUVs and collaborative, citizen science hook-and-line fishing. For the context of this study, UVC are diver-based transect surveys performed within recreational self contained underwater breathing apparatus (SCUBA) diving depth limits, BRUVs are baited, stationary video surveys that operate beyond recreational scuba diving limits, and collaborative fisheries surveys are timed hook-and-line, catch-and-release fishing surveys (see Table S1). Prior research on MPA effectiveness and ecosystem resilience has utilised data from each of these or similar monitoring programs in California separately (Eisaguirre et al., 2020; Hamilton & Caselle, 2015; Pondella et al., 2015; Starr et al., 2016; Wendt & Starr, 2009; Ziegler et al., 2022), but rarely together (but see Claisse et al., 2018; Smith et al., 2023). First, we ask if each method surveys the same species assemblage within MPAs of interest. Because many of the species encountered can be found across the depth range sampled by all three methods, we expect each surveys a similar groundfish assemblage. We then ask if each method detects similar MPA effects for groundfish abundance, biomass, and diversity, three common metrics of MPA assessments. We hypothesise that MPA responses for these metrics are similar across methods for each island.

2 | MATERIALS AND METHODS

2.1 | California's MPA network

In 1999, the state of California passed the Marine Life Protection Act (MLPA), mandating the creation of an ecologically connected, state-wide MPA network (Botsford et al., 2014; Gleason et al., 2013; Saarman et al., 2013). Scientists, resource managers, and regional stakeholders undertook an intense planning and implementation process over the following 13 years that transformed the state's existing, scattered MPAs into what is now one of the largest MPA networks in the world, encompassing 16% of state waters and a diverse assemblage of coastal habitats ranging from sandy beaches to deep rocky reefs (Gleason et al., 2013). In addition to expanding existing MPAs, the MLPA laid out a framework for the adaptive management of the network to ensure that policy and conservation goals are met (CDFW, 2016; Gleason et al., 2013). In response to such a vast undertaking, the state developed an MPA Action Plan (CDFW, 2016) which formed habitat-specific research groups to monitor the different habitat types found in the network (e.g. sandy beach, rocky intertidal,

nearshore rocky reef and kelp forests, and deep reef) and resulted in multiple MPA monitoring programs with differing methodologies that overlap spatially across the network (Marine Protected Area Monitoring Action Plan, 2018).

2.2 | Study area

The Channel Islands MPA network consists of 13 MPAs covering just over 20% of the Northern Channel Islands' waters. We analysed data from MPAs and associated reference areas at two islands: Carrington Point State Marine Reserve (SMR; no-take) and South Point SMR at Santa Rosa Island, and Anacapa Island SMR and State Marine Conservation Area (SMCA; limited-take) at Anacapa Island (Figure 1). Reference areas are similar to their associated MPA sites in habitat, depth and biogeographic influence but open to fishing. The NCI experiences a wide range of environmental conditions over a relatively small spatial scale, namely in sea surface temperature

(SST), exposure, and productivity, resulting in each island being influenced by distinct biogeographic processes (Figure 1; Hamilton et al., 2010). In the Western channel, Santa Rosa Island (Carrington Point SMR and South Point SMR) is characterised by cooler temperatures and direct exposure to the productive California current, while in the Eastern channel, Anacapa Island SMR/SMCA is influenced by the warmer and less productive waters of the Southern California Countercurrent. Exposure to fishing pressure also differs between MPAs and their associated reference sites at each island, with MPAs at Santa Rosa being located 50–69 km away from port, while MPAs at Anacapa Island are located 15 km away from port. At Santa Rosa Island both Carrington Pt. SMR and South Pt. SMR are no-take State Marine Reserves with associated reference areas open to fishing directly adjacent to MPA boundaries. Anacapa Island consists of three distinct management zones correlating with small breaks in the island (Caselle et al., 2018). On the north side there is the no-take SMR and the limited-take SMCA (where only the commercial and recreational take of CA spiny lobsters and recreational

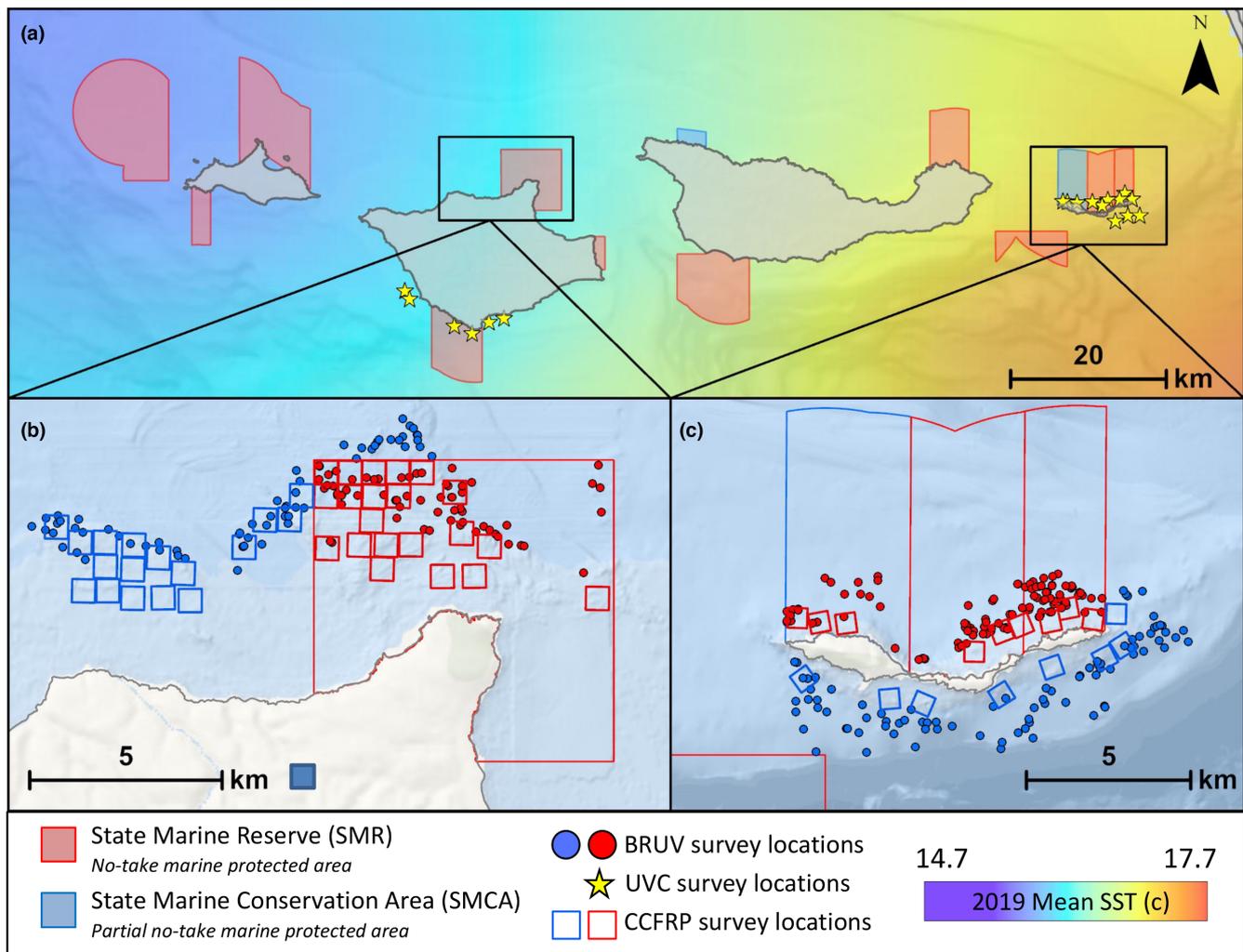


FIGURE 1 (a) The Northern Channel Islands with marine protected areas, underwater visual census (UVC) sites and 2019 mean sea surface temperature overlaid. (b) Carrington Point SMR and (c) Anacapa Island SMR/SMCA with California Collaborative Fisheries Research Program (CCFRP) sampling cells and baited remote underwater video (BRUV) survey locations overlaid.

take of pelagic fish is allowed), while the entire south side of the island is open to fishing.

2.3 | MPA monitoring techniques

In this study, we compare fish density and biomass from three MPA monitoring programs designed to survey different but overlapping depth strata of rocky reef habitat: BRUVs (30–100 m), collaborative hook-and-line fishing (CCFRP) (10–50 m) and UVC (0–20 m). All fieldwork was conducted under California Department of Fish and Wildlife scientific collecting permits SC-397, S-191210002-19126-001, and S-183620009-19134-001. In order to make the abundance and biomass metrics collected by each method comparable, we converted these metrics to response ratios, diversity indices, and community dissimilarities. We then compared the direction and strength of each metric across monitoring methods at each island.

2.3.1 | Baited remote underwater video

BRUV surveys consisted of stereo-video baited camera rigs deployed inside and outside of the Anacapa Island SMR/SMCA and Carrington Pt. SMR using a stratified random sampling approach. Using habitat and bathymetry maps, a fishnet grid of 100 m × 100 m cells was created (in ArcGIS Pro 2.1) and applied across each MPA and its adjacent reference areas. Grid cells were classified based on the amount of hard bottom and allocated into three depth bins (30–50, 50–70 and 70–100 m). Grid cells meeting baseline hard bottom criteria (>15% for surveys shallower than 70 m and >5% for surveys deeper than 70 m) were haphazardly selected to be sampled on a given survey day. BRUVs were deployed by hand to hard bottom habitats for a minimum of 30 min (Harasti et al., 2015).

SeaGIS EventMeasure (www.seagis.com.au) was used to log observations and record 3D measurements for each video. Prior to the start of each BRUV season, stereo video systems are calibrated using a SeaGIS calibration cube and CAL software program until an accuracy of <1 cm is achieved against known length models at various distances from the camera. To quantify fish abundance, we recorded the maximum number of individuals of a species present within a single video frame (MaxN) (Willis & Babcock, 2000). MaxN is considered a conservative estimate of relative abundance and has become the standard metric used in BRUV analysis worldwide (Langlois et al., 2020; Shortis et al., 2008). The total length was measured for every fish observed in the MaxN frame, except for individuals unable to be accurately measured due to camera angle, visibility, or other obstructions. Measurements were converted to biomass using an allometric length-weight conversion: $W = aTL^b$, where parameters a and b are species-specific constants, TL is the total length in cm and W is weight in grams. Parameters and constants were obtained from the literature and FishBase (Froese & Pauly, 2023; Love, 1990),

and the sum of all individual weights in the MaxN frame for a given species was used to measure biomass per BRUV survey.

2.3.2 | Collaborative hook and line surveys

Catch and release, hook and line MPA monitoring surveys at Anacapa Island SMR/SMCA and Carrington Pt. SMR were conducted by the California Collaborative Fisheries Research Program (CCFRP) using a stratified random sampling design developed in collaboration with local fisherman and academic scientists (Bonney et al., 2021; Wendt & Starr, 2009; Wilson, 1999; Yochum et al., 2011). Utilising a combination of habitat maps (distributed by the Seafloor Mapping Lab, CSU Monterey Bay) and Commercial Passenger Fishing Vessel (CPFV) captain expertise, 500 m × 500 m grid cells were placed (in ArcMap 10.5) on rocky reef habitat inside each MPA and associated reference areas. Grid cells were constrained between depths of 10 and 50 m to limit fish barotrauma and mortality (Yochum et al., 2011). Surveys were conducted from 2017 to 2020 onboard local CPFVs using volunteer anglers recruited from various fishing clubs, websites and organisations. Each day, we randomly selected four grid cells for sampling (2 inside the MPA and 2 in reference areas) at a given island. Once inside a grid cell, captains were instructed to complete three 15-min drifts over suitable rocky reef habitat, with a goal time of 45 min of fishing per cell. We evenly distributed standardised tackle (double dropper loop with squid bait, double shrimp fly with squid bait, and plastic swimbaits) among anglers and collected data on time spent fishing, location, depth and other environmental factors. Caught fishes were identified to species, measured, and sometimes tagged with a T-bar anchor tag (only if larger than 25 cm and not exhibiting signs of barotrauma) before being released. All lengths were recorded as total length (TL) to the nearest cm. Ethical approval for this work was granted by UC Santa Barbara's Institutional Animal Care and Use Committee (IACUC), protocols #856.1 and #856.2.

Groundfish density was calculated as catch per unit effort (CPUE) by dividing the total number of fishes caught by the total angler hours per grid cell per day. TL measurements for each fish were converted to biomass as described above. Biomass per unit effort (BPUE) was then calculated by dividing biomass by total angler hours per grid cell per day.

2.3.3 | Underwater visual census fish surveys

From 2017 to 2020, the Partnership for the Interdisciplinary Study of Coastal Oceans (PISCO) conducted fish community surveys on rocky reefs shallower than 25 m at South Pt. SMR and Anacapa Island SMR/SMCA (Malone et al., 2022). Between July and November, PISCO divers characterised fish community structure across three levels of the water column: benthic, midwater and kelp canopy (if present). For each 30 × 2 × 2 m transect, divers identified and sized all fishes encountered to the nearest cm (except for small-bodied, cryptic species that are not surveyed).

Transects were replicated 12 times at each site, with transect layout depending on the extent of the reef. Typically, three transects were distributed end to end at least 5 m apart for four (5, 10, 15 and 20 m) isobaths. The average fish density for a site is measured as a volume (fish per m³). Biomass density (kg per m³) was calculated from length data using the conversions described above. Data were used from 12 sites at Anacapa Island and 6 sites from Santa Rosa Island.

2.4 | Analysis

2.4.1 | Data standardisation

In order to compare species diversity and community structure among methods, abundance data from each project (CCFRP: CPUE per grid cell; UVC: density per site; BRUV: MaxN per drop) was standardised into a range between 0 and 1 in R version 4.0.2 using the 'decostand' function from the Vegan package (version 2.5-7).

2.4.2 | Community structure

To compare fish community structure across methods at each island, we applied non-metric multi-dimensional scaling (nMDS) to the range standardised abundance dataset. We removed species complexes, schooling bait fishes and young of the year (YOY) to reduce variation unrelated to MPA performance before separating the data by island. Species complexes removed from this analysis include olive and yellowtail rockfish, young-of-year (YOY) gopher, copper, and kelp rockfishes, and any bait fishes that are difficult to identify to species during UVC and video surveys but can be identified to species during hook and line surveys. YOY fishes were excluded because (a) their density is more likely to be determined by settlement processes than MPA effects, (b) they preferentially settle in habitats not sampled by each method (e.g. kelp canopy), and (c) they are too small to be sampled by hook and line surveys. We then created a Bray–Curtis dissimilarity matrix for each island and conducted nMDS using the 'metaMDS' function in 'R' to display differences in fish community structure across techniques by year and tested for significance using a one-way analysis of similarity (ANOSIM, R 'Vegan' 4.0.2). We then used a similarity percentage analysis (SIMPER, R 'Vegan' 4.0.2) to determine which fish species contributed the most to dissimilarity across methods at each island. These analyses were conducted with 54 fish species for Anacapa, and 52 fish species for Santa Rosa (see [Table S2](#)).

2.4.3 | Diversity and richness

Shannon diversity and species richness were calculated in R using the 'diversity' function ('Vegan' 4.0.2). The data were then split by island and tested for differences in the mean between methods and

between MPA and reference areas for each method. The variance in Shannon diversity between BRUVs, CCFRP, and UVC at both Anacapa and Santa Rosa were significantly different from one another, so we employed a Welch's *t*-test to compare each method using the Bonferroni *p*-value adjustment for multiple comparisons. We then tested for differences in mean Shannon diversity between MPA and reference areas within a particular method using Welch's *t*-test. The species richness data between all three methods were significantly different from normal, so we employed a pairwise Wilcoxon signed-rank test to test for differences in species richness. We then tested for differences in mean species richness between MPA and reference areas for each method using a Wilcoxon signed-rank test. All statistical tests were done in R version 4.1.3 using the 'stats' package.

2.4.4 | MPA effects

We used log-response ratios to compare the effect of MPA protection using each monitoring method's respective metric for fish density and biomass. Log-response ratios for each species were calculated in R using the formula $\text{Response Ratio} = \ln\left(\frac{X_{\text{MPA}}}{X_{\text{REF}}}\right)$, where X_{MPA} is the mean value of a variable (density or biomass) for a given method inside the MPA, and X_{REF} is the corresponding mean value of that variable for the associated reference area. Response ratios were calculated for commercially or recreationally 'targeted' (i.e. 'fished') and 'non-targeted' species groupings as well as individual species that were among the top drivers of dissimilarity from the SIMPER analysis. We summed the density and biomass for non-targeted and targeted species as we hypothesise that positive MPA effects should be greater for species that are fished relative to those that are not (Caselle et al., 2015; Hamilton & Caselle, 2015; Malone et al., 2022; Ovando et al., 2021). For visualisation purposes, the frequency of occurrence for each species per island per method was calculated by dividing the number of UVC sites, CCFRP grid cells, or BRUV surveys at an island that encountered a given species by the total number of sites, cells, or drops at that island.

3 | RESULTS

Our comparative analyses show that each monitoring method detects a different fish community and diversity values, despite the overlapping depth ranges of most of the species surveyed. However, each method generally detects similar trends in overall MPA response for targeted species when aggregated together. When considering aggregated non-targeted species, individual targeted species, and individual non-targeted species, MPA responses in both density and biomass differed among methods and islands.

3.1 | Community structure

We found that each method captured a unique groundfish assemblage at both Anacapa (2D stress 0.094, [Figure 2a](#)) and Santa Rosa

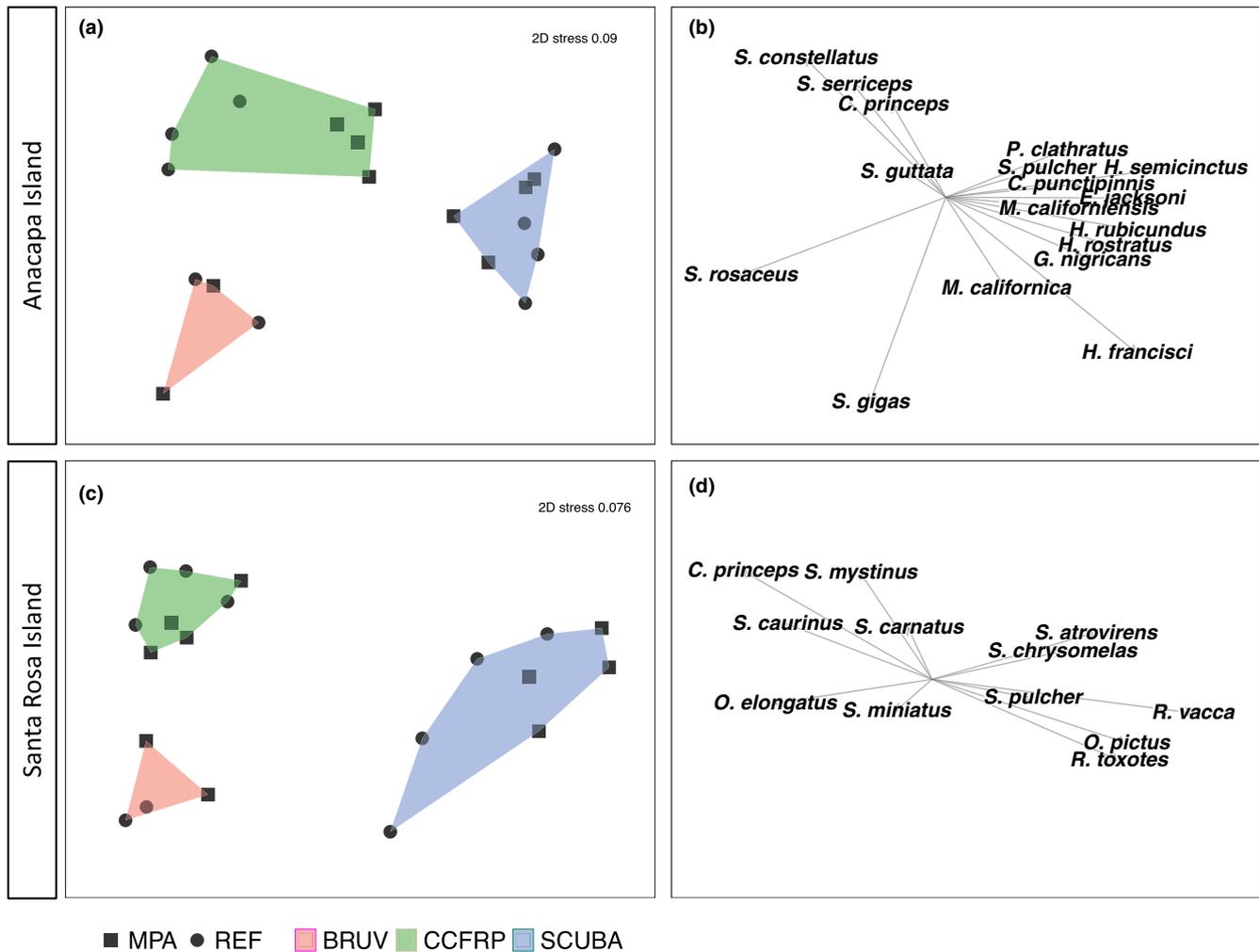


FIGURE 2 nMDS ordination and species vector overlays for Anacapa (a, b) and Santa Rosa (c, d) generated using Bray–Curtis dissimilarity matrices of range standardised abundance data. The stress values of 0.09 (a) and 0.076 (c) indicate a fair representation of dissimilarity between techniques by year. (a, c) Shaded polygons represent marine protected area (MPA) monitoring techniques: Baited remote underwater video (BRUV, red), collaborative fisheries research (CCFRP, green) and underwater visual census (UVC, blue) while point shape indicates management strategy (MPA=square and reference=circle). (b, d) The vector length is proportional to the effect each species has on matrix dissimilarity. Values adjusted for clarity. Species shown were chosen from the top drivers of dissimilarity among methods from a SIMPER analysis.

(2D stress 0.076, Figure 2c). An analysis of similarity (ANOSIM) supported these results, indicating that the fish community structure surveyed by each method was significantly different at both islands (Anacapa $R=0.87$, $p=0.0001$, see Figure S1; Santa Rosa $R=0.92$, $p=0.0001$, see Figure S2). Analysis with SIMPER identified the top drivers of dissimilarity in pairwise comparisons of the methods at both islands (Figure 2b,d; Tables S3–S8).

3.2 | Diversity and richness

At both islands, we found that the mean Shannon diversity index was significantly different across methods (Table 1; Figure 3a). For both islands, UVC surveys generally had the highest diversity, while the lowest diversity levels were associated with BRUV surveys. We

also tested for differences in mean Shannon diversity index between MPA and reference areas for each method-island pair. Diversity was significantly different between MPA and reference areas for BRUV data from Santa Rosa Island ($t(113)=2.25$, $p=0.02$, Table 1) and marginally significant for UVC data from Santa Rosa Island ($t(20)=1.85$, $p=0.07$, Table 1).

We found mean species richness was significantly different between each pair of monitoring methods with the exception of BRUV vs. CCFRP at Anacapa Island (Table 2; Figure 3b). At both islands, mean species richness was highest in the UVC data and tended to be lowest in the BRUV data. We also tested for differences in mean species richness between MPA and reference areas for each method. We found that mean species richness in the UVC data was significantly different between MPA and reference areas for both islands (p -value=0.02 for Anacapa and p -value=0.05 for Santa Rosa,

TABLE 1 *p*-values from Welch's *t*-tests comparing Shannon *H*-index between monitoring methods and Wilcoxon Signed Rank tests of species richness between MPA monitoring methods.

Comparison	Shannon diversity		Species richness	
	Anacapa	Santa Rosa	Anacapa	Santa Rosa
SCUBA vs. CCFRP	<0.0001	<0.0001	<0.0001	<0.0001
SCUBA vs. BRUV	<0.0001	<0.0001	<0.0001	<0.0001
BRUV vs. CCFRP	0.0005	<0.0001	0.10	<0.0001

Note: Values less than 0.05 are significant.

Abbreviations: BRUV, baited remote underwater video; CCFRP, California Collaborative Fisheries Research Program; MPA, marine protected area; SCUBA, self contained underwater breathing apparatus.

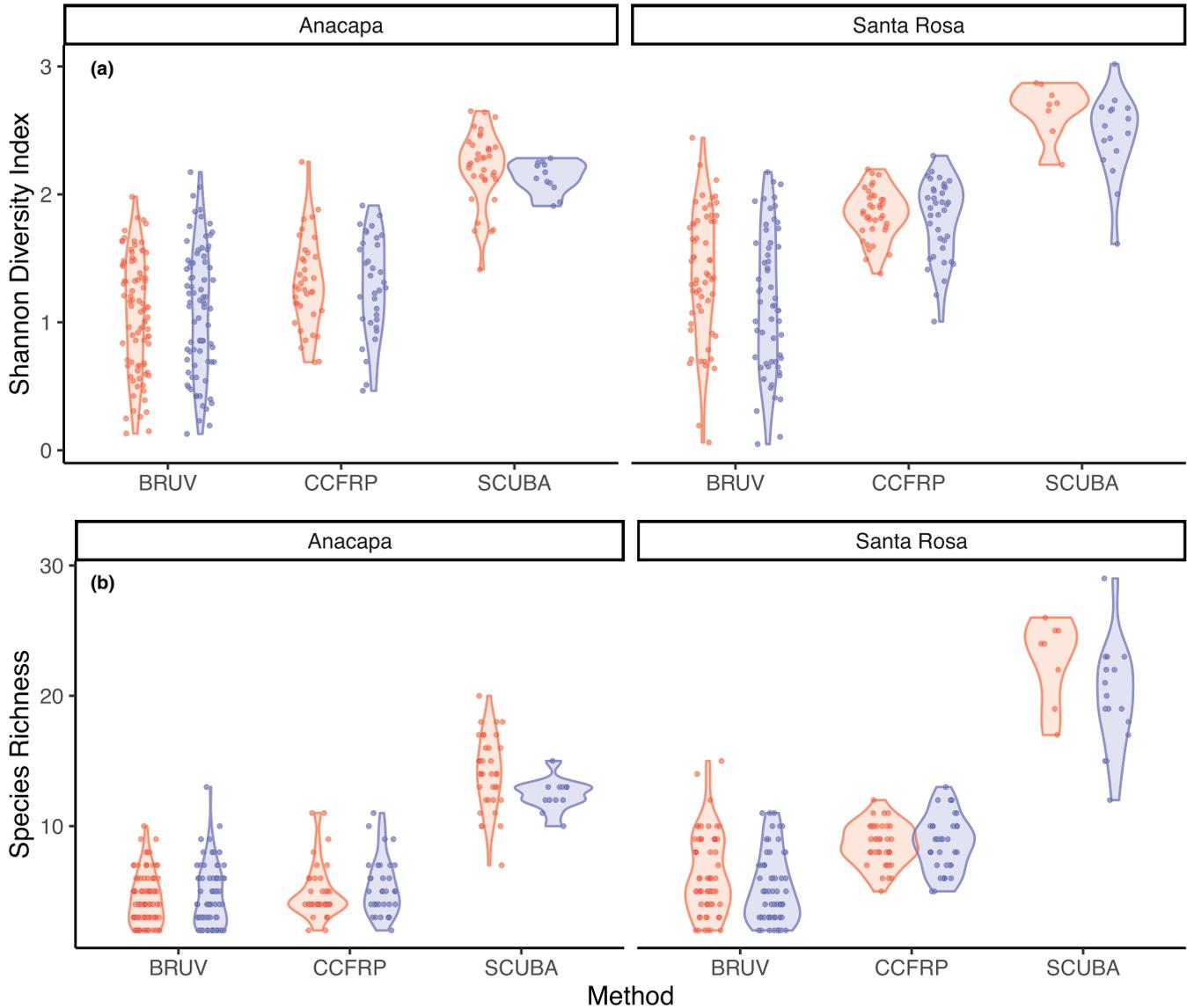


FIGURE 3 (a) Shannon Diversity Index by island and monitoring method. (b) Species Richness by island and monitoring method. For both plots, data from marine protected areas are shown in red and data from reference areas in blue.

Table 2), while CCFRP and BRUV did not show any significant differences. However, species richness between the MPA and reference areas were marginally significant for BRUV surveys at Anacapa (*p*-value=0.07, Table 2).

3.3 | MPA effects

All three methods detected positive MPA response ratios for aggregated commercially and recreationally targeted fish species density

(Figure 4) and biomass (Figure 5) at both islands. MPA responses for aggregated non-targeted species density and biomass varied across methods and islands. At Anacapa, we found that BRUVs and CCFRP both detected more negative density and biomass response ratios for individual species when compared to UVC. This was not the case at Santa Rosa, where both BRUVs and CCFRP generally detected more positive density and biomass response ratios. BRUVs and CCFRP generally detected similar results for most rockfish (*Sebastes* spp.) and ocean whitefish (*Caulolatilus princeps*) response ratios at both

TABLE 2 *p*-values from Welch's *t*-tests comparing Shannon *H*-index between MPA and Reference areas for each monitoring method and pairwise Wilcoxon Signed Rank tests of species richness between MPA and reference areas for each method.

Method	Shannon diversity		Species richness	
	Anacapa	Santa Rosa	Anacapa	Santa Rosa
SCUBA	0.20	0.07	0.02	0.05
CCFRP	0.86	0.5	0.45	0.83
BRUV	0.75	0.02	0.75	0.07

Note: Values less than 0.05 are significant.

Abbreviations: BRUV, baited remote underwater video; CCFRP, California Collaborative Fisheries Research Program; MPA, marine protected area; SCUBA, self contained underwater breathing apparatus.

islands, whereas response ratios from UVC surveys notably differed. For a given island, all three methods only agreed for a few individual species, detecting positive density and biomass response ratios for kelp bass (*Paralabrax clathratus*) at Anacapa, California sheephead (*Semicossyphus pulcher*) and copper rockfish (*Sebastes caurinus*) at Santa Rosa, and negative density and biomass response ratios for blacksmith (*Chromis punctipinnis*) at Anacapa. For other species, both targeted and non-targeted, there was high disagreement in the sign of the density and biomass response ratios between monitoring techniques and islands, although, for many of these species, there was a very low frequency of occurrence at particular islands.

4 | DISCUSSION

We found strong evidence to support the concept that there is no 'one-size-fits-all' approach to comprehensive MPA monitoring. While all three of our methods detected positive density and biomass responses for targeted fish species when aggregated, the strength and direction of species-specific responses varied by both method and island. Additionally, each sampling method was found to survey dissimilar species assemblages at both islands. Our comparison adds to a growing body of work showing species selectivity to be an important driver of variation among sampling methods (Parker

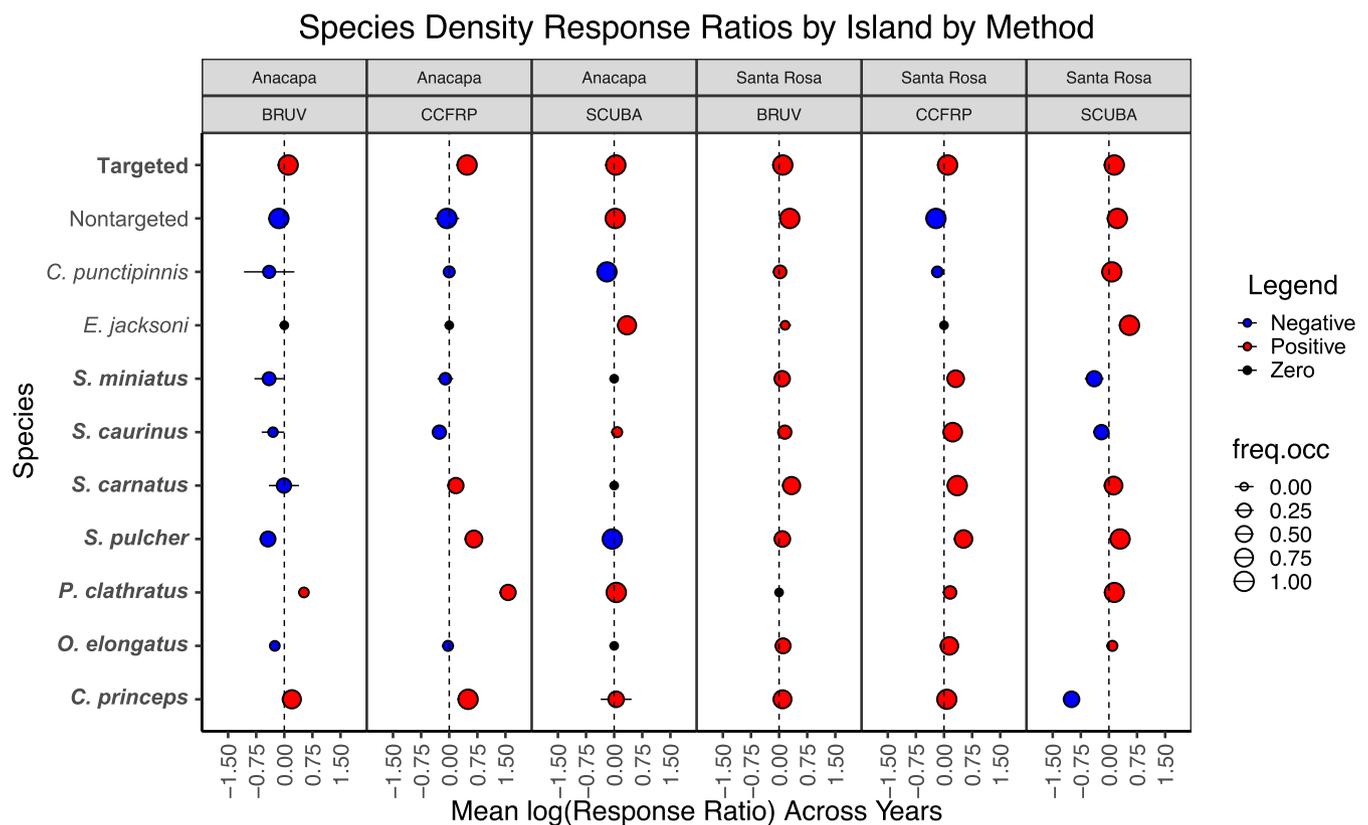


FIGURE 4 Fish density log response ratios by island by method. Species in bold are targeted. Horizontal lines represent standard error. The size of each point is scaled to the frequency of occurrence for that species per technique per island, ranging from 0 (not seen) to 1 (seen on every survey).

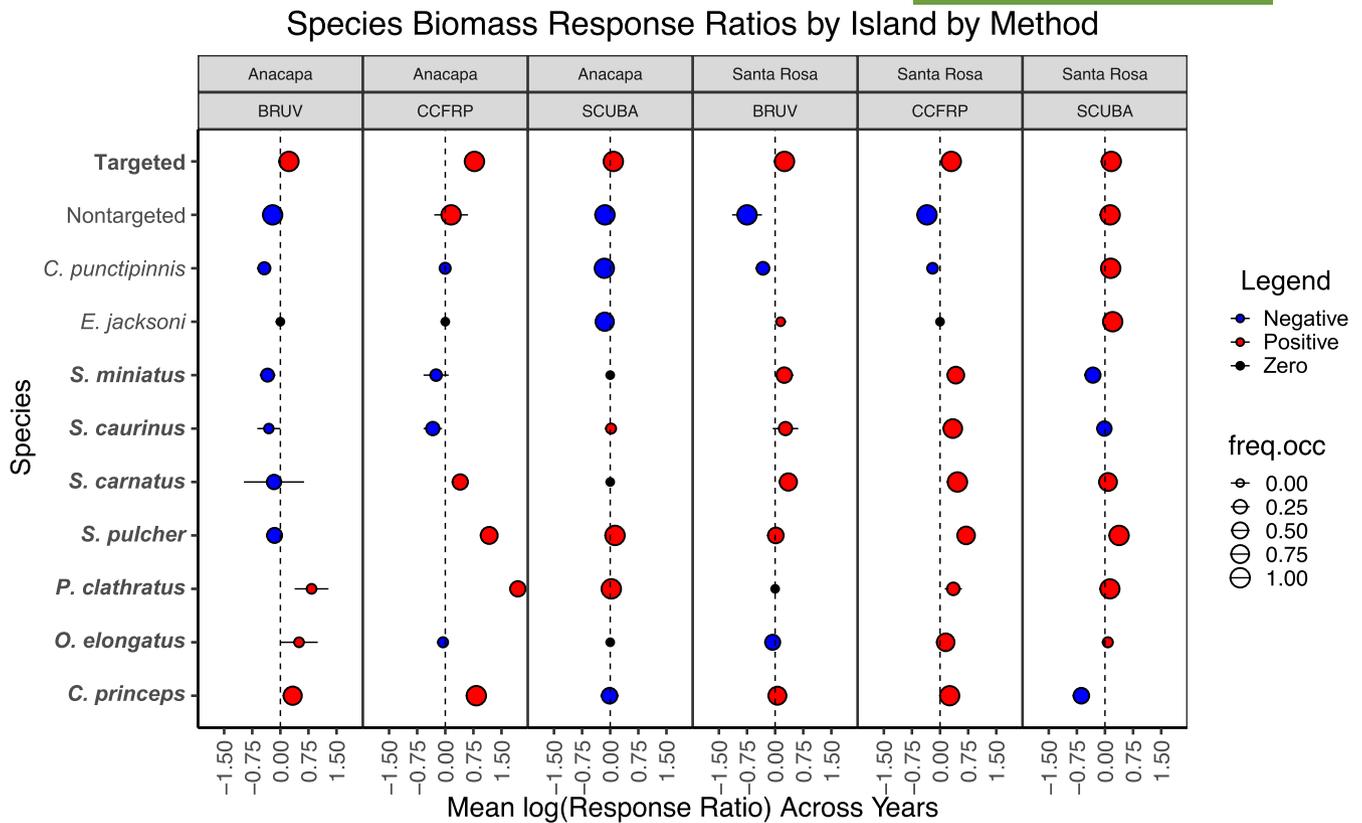


FIGURE 5 Fish biomass log response ratios by island by method. Species in bold are targeted. Solid horizontal lines represent standard error. The size of each point is scaled to the frequency of occurrence for that species per technique per island, ranging from 0 (not seen) to 1 (seen on every survey).

et al., 2016; Starr et al., 2010), and provides compelling evidence that multiple sampling techniques are required to design a comprehensive, effective, and adaptable MPA monitoring regime.

Community structure analysis indicates that each technique surveyed distinct fish assemblages at both islands, an interesting result considering surveys were often conducted on the same reefs by multiple methods. Variation in sampling methodology is likely the main driver of these differences, in particular, the presence or absence of bait, as suggested by a large body of prior work on the selectivity of fishing techniques and baited camera systems (Jessop et al., 2022; Starr et al., 2010; Watson et al., 2010). Our two baited methods, CCFRP and BRUVs, detected more similar but less rich and diverse fish species assemblages than UVC surveys. These discrepancies appear mostly driven by the presence of larger-bodied, carnivorous fishes and the absence of small-mouthed herbivores and micro-invertivores on CCFRP and BRUV surveys. Conversely, divers performing UVC were more likely to encounter a more rich and diverse fish species assemblage, as many resident kelp forest fishes are either herbivorous, small-bodied, or small-mouthed and therefore unattracted to bait, unable to take a fishing hook, or in fear of larger predatory fishes at the bait. Although not of direct consequence to local fisheries, these primarily non-targeted species fill important niches in kelp forest ecosystems, such as clearing space for kelp recruitment, controlling invasive species, and acting as prey for fishes that are of commercial and recreational interest (Bredvik

et al., 2011; House & Allen, 2022). In addition, a common analytical design used to evaluate MPA effectiveness is to compare an MPA site or sites to 'reference' sites using response ratios on some metric such as density or biomass, primarily on targeted species as those are the most likely to respond to the removal of fishing pressure. Reference sites are assumed to be similar in all regards except for fishing pressure, yet this assumption is likely untrue in most cases. A more robust inference of an MPA effect will not only include an evaluation of targeted species inside and outside of an MPA but also include a comparison of targeted to non-targeted species responses with the non-targeted species acting as controls for changing environmental conditions independent of MPA effects (Ovando et al., 2021). If an MPA is implemented with the intent to preserve ecosystem structure, function, or biodiversity, being able to account for both targeted and non-targeted species is critical to properly assessing MPA success (Day et al., 2002; Grorud-Colvert et al., 2014; Jessop et al., 2022). By utilising multiple techniques, managers can account for the biases of one with the strengths of another, resulting in more confident measures of MPA effect than one technique alone could provide.

Each technique detected positive density and biomass response ratios for aggregated targeted species at both islands. Inversely, we found no discernible pattern for grouped non-targeted species. Prior work has shown that targeted species stand to benefit more from MPAs than non-targeted species due to

the direct release from fishing pressure MPAs provide (Caselle et al., 2015; Giakoumi et al., 2017), and these results show that any of these monitoring techniques should provide managers with accurate results on general MPA effectiveness. However, our analysis indicates that differences in the community structure sampled by each method likely impact species-specific biomass and density measurements. This, in turn, results in the techniques showing conflicting patterns for a variety of important species at the same island, presenting potential issues for species-specific management. For example, *C. princeps*, a tilefish commonly encountered across the NCI, is known to be wary of divers, highly mobile, and is generally encountered on the sand-rock ecotone of rocky reefs (Bellquist et al., 2008). A voracious carnivore often found schooling in high densities, response ratios for *C. princeps* from both BRUVs and CCFRP indicated greater density and biomass inside MPAs at both islands. However, response ratios from UVC indicated there was no MPA effect at Anacapa and even greater density and biomass of *C. princeps* outside MPAs at Santa Rosa. A resource manager who chooses to only consider UVC data may make decisions based on the assumption that MPAs are not beneficial to *C. princeps*, despite evidence from other methods supporting the opposite conclusion. By taking into account the frequency of occurrence for *C. princeps*, we can parse out the influence of sampling methodology, as this species was encountered on the vast majority of BRUV and CCFRP surveys (>75%), but less than half of UVC surveys at both islands (<46%). This indicates that either baited method would provide a better measure of MPA effectiveness for *C. princeps* than UVC. Our analysis uncovered multiple examples of this phenomenon with a variety of both targeted and non-targeted species, highlighting the importance of considering multiple data sources and species-specific biology when designing an MPA monitoring plan. For example, in ecosystems where larger-bodied, highly mobile carnivorous fishes like sharks are important, such as in the seagrass meadows of the Caribbean (Gallagher et al., 2022), managers stand to benefit most from some type of BRUV monitoring, as larger-bodied predators are more frequently encountered on this technique than UVC. In contrast, in tropical MPA networks such as the Great Barrier Reef, where parrotfishes and other herbivores are crucial to mediating coral-macroalgae competition (Cheal et al., 2010) managers would likely benefit the most from an extensive UVC monitoring regime, as herbivores are less likely to be encountered on baited surveys.

Variation in our analyses on the island scale (i.e. Anacapa vs. Santa Rosa) was expected and is easily explained by biogeographic conditions, as previous work has shown there to be a strong environmental gradient across the NCI due to the mixing of the warm Southern California Countercurrent and the cold California Current (Hamilton et al., 2010). Prior analysis of UVC data has shown that many resident groundfish species at the NCI exhibit consistent thermal preferences (Freedman et al., 2020) and our results support this conclusion, as these methods encountered species where we expected them to be based on these classifications. For example, known cool water associates were encountered far more frequently in the colder waters of

Santa Rosa than they were in the relatively warm waters of Anacapa. If these species were encountered at Anacapa, it was more likely to be on the colder, deeper reefs surveyed by BRUVs and CCFRP, with multiple cool water associates in our dataset *not* encountered at all on UVC at Anacapa, despite being targeted and caught there by commercial and recreational anglers. In contrast, warm water associates such as herbivores were encountered far more frequently at Anacapa on UVC than by BRUVs or CCFRP, as well as by any of the methods at Santa Rosa, indicating that without all three techniques, resource managers would be receiving an incomplete picture of groundfish community structure, species-specific biomass, and species-specific density inside their MPAs.

We were surprised to find BRUVs to be the least diverse sampling method, given that they provide a visual reference of the reef similar to UVC. Previous work in temperate ecosystems found this method to sample a more diverse fish community than angler-based surveys (Parker et al., 2016) and prior work from the Great Barrier Reef found BRUVs measured similar levels of species richness to UVC (Cheal et al., 2021). This was not the case with our analysis, indicating that factors such as visibility, current, and equipment limitations (e.g. camera rigs tipping) may limit the utility of this method in our particular system. While both CCFRP and UVC face subsets of these limitations, samplers for both can assess conditions in real-time, deciding whether or not a survey should be performed, whereas BRUV survey success cannot be assessed until the footage is reviewed. Over a quarter (27%) of BRUV deployments were discarded before analysis for various reasons (e.g. missed habitat, rig tipped), an issue not seen with either of the other methods (Jainese, 2023). Additionally, BRUV surveys are the most spatially constrained of these methods as they are conducted from a single point, whereas UVC and CCFRP surveys cover a greater area of the reef to gather each data point. However, each method is not without limitations. UVC has a high barrier to entry, requiring extensively trained SCUBA divers and expensive equipment, and CCFRP is unable to provide detailed information on habitat structure and groundfish behaviour, which may help contextualise results. It is important for resource managers to be aware of the strengths and weaknesses of each technique, as they will ultimately decide which method is the most appropriate for a given MPA management regime.

4.1 | Impacts on future management

As global resource managers opt to protect more of the world's oceans in the coming years, many will engage in adaptive management to ensure conservation goals are being met. Prior work has demonstrated the importance of defining management goals, expectations, and metrics of success before adopting a monitoring regime (Nickols et al., 2019), as many managers may be resource, time, or personnel limited. If resources are limited, MPAs designed solely around protecting targeted fish biomass should consider prioritising either a hook-and-line or similar fisheries-dependent,

non-consumptive monitoring technique (e.g. catch and release trapping). Conversely, MPAs implemented to protect biodiversity or ecosystem structure, such as those in an eco/dive-tourism economy, might benefit most from a UVC-based monitoring scheme, while managers concerned about the indirect effects of fishing on community structure, particularly at depths beyond recreational SCUBA diving limits, could benefit from a BRUV based monitoring scheme. However, our analysis suggests that the most effective way to comprehensively monitor global MPAs and MPA networks may be to deploy a suite of complementary monitoring techniques. This study builds upon prior assessments of monitoring methodologies and provides compelling evidence that managers must consider factors such as species-specific biology and sampling selectivity when both designing monitoring regimes and analysing data resulting from monitoring efforts.

AUTHOR CONTRIBUTIONS

Christopher Honeyman, Jennifer E. Caselle, and Peter Carlson conceived the ideas and designed the methodology. Christopher Honeyman, Avrey Parsons-Field, Kathryn Davis, Conner Jainese, Peter Carlson, Jacob Eisaguirre, and Jennifer E. Caselle collected the data. Christopher Honeyman, Peter Carlson, Barbara Spiecker, and Anita Giraldo-Ospina analysed the data. Christopher Honeyman and Jennifer E. Caselle led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for submission.

ACKNOWLEDGEMENTS

This study was made possible by the support and participation of countless scientists, stakeholders, and other individuals who were instrumental in the creation of these MPA monitoring programs and integral to their success throughout the years. We extend our special thanks to the California Department of Fish and Wildlife, Ocean Protection Council, the David and Lucile Packard Foundation, and California Sea Grant for current and historic funding for each of these programs. Other project partners include the Channel Islands National Marine Sanctuary, Coastal Conservation Association, Stardust Sportfishing, Aloha Spirit Sportfishing, and the CA Nature Conservancy. This is PISCO contribution number 534.

CONFLICT OF INTEREST STATEMENT

The authors have no conflicts of interest to declare.

DATA AVAILABILITY STATEMENT

The data and code that support the findings of this study are openly available on GitHub (<https://github.com/c-honeyman/Correspondence-among-multiple-methods-provides-confidence-when-measuring-Marine-Protected-Area-effect>) and archived on Zenodo <https://doi.org/10.5281/zenodo.8361054> (Honeyman et al., 2023).

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REFERENCES

- Allen, C. R., Fontaine, J. J., Pope, K. L., & Garmestani, A. S. (2011). Adaptive management for a turbulent future. *Journal of Environmental Management, Adaptive Management for Natural Resources*, 92, 1339–1345. <https://doi.org/10.1016/j.jenvman.2010.11.019>
- Bellquist, L. F., Lowe, C. G., & Caselle, J. E. (2008). Fine-scale movement patterns, site fidelity, and habitat selection of ocean whitefish (*Caulolatilus princeps*). *Fisheries Research*, 91(2–3), 325–335. <https://doi.org/10.1016/j.fishres.2007.12.011>
- Bonney, R., Byrd, J., Carmichael, J. T., Cunningham, L., Oremland, L., Shirk, J., & Von Harten, A. (2021). Sea change: Using citizen science to inform fisheries management. *BioScience*, 71, 519–530. <https://doi.org/10.1093/biosci/biab016>
- Botsford, L. W., White, J. W., Carr, M. H., & Caselle, J. E. (2014). Marine protected area networks in California, USA. *Advances in Marine Biology*, 69, 205–251. <https://doi.org/10.1016/B978-0-12-800214-8.00006-2>
- Bredvik, J. J., Boerger, C., & Allen, L. G. (2011). Age and growth of two herbivorous, kelp forest fishes, the opaleye (*Girella nigricans*) and halfmoon (*Medialuna californiensis*). *Bulletin of the Southern California Academy of Sciences*, 110, 25–34. <https://doi.org/10.3160/0038-3872-110.1.25>
- California Department of Fish and Wildlife. (2016). *California marine life protection act master plan for marine protected areas*. California Fish and Game Commission. www.wildlife.ca.gov/Conservation/Marine/MPAs/Master-Plan
- Caselle, J. E., Davis, K., & Marks, L. M. (2018). Marine management affects the invasion success of a non-native species in a temperate reef system in California, USA. *Ecology Letters*, 21, 43–53. <https://doi.org/10.1111/ele.12869>
- Caselle, J. E., Rassweiler, A., Hamilton, S. L., & Warner, R. R. (2015). Recovery trajectories of kelp forest animals are rapid yet spatially variable across a network of temperate marine protected areas. *Scientific Reports*, 5, 14102. <https://doi.org/10.1038/srep14102>
- Cheal, A. J., Emslie, M. J., Currey-Randall, L. M., & Heupel, M. R. (2021). Comparability and complementarity of reef fish measures from underwater visual census (UVC) and baited remote underwater video stations (BRUVS). *Journal of Environmental Management*, 289, 112375.
- Cheal, A. J., MacNeil, M. A., Cripps, E., Emslie, M. J., Jonker, M., Schaffelke, B., & Sweatman, H. (2010). Coral-macroalgal phase shifts or reef resilience: Links with diversity and functional roles of herbivorous fishes on the Great Barrier Reef. *Coral Reefs*, 29, 1005–1015. <https://doi.org/10.1007/s00338-010-0661-y>
- Claissie, J. T., Blanchette, C. A., Dugan, J. E., Williams, J. P., Friewald, J., Pondella, D. J., Schooler, N. K., Hubbard, D. M., Davis, K., Zahn, L. A., Williams, C. M., & Caselle, J. E. (2018). Biogeographic patterns of communities across diverse marine ecosystems in southern California. *Marine Ecology*, 39, e12453. <https://doi.org/10.1111/maec.12453>
- Colton, M., & Swearer, S. (2010). A comparison of two survey methods: Differences between underwater visual census and baited remote underwater video. *Marine Ecology Progress Series*, 400, 19–36. <https://doi.org/10.3354/meps08377>
- Day, J. (2008). The need and practice of monitoring, evaluating and adapting marine planning and management—Lessons from the

- Great Barrier Reef. *Marine Policy*, 32, 823–831. <https://doi.org/10.1016/j.marpol.2008.03.023>
- Day, J., Hockings, M., & Jones, G. (2002). *Measuring effectiveness in marine protected areas—Principles and practice*. Australian Society for Fish Biology.
- Dickens, L. C., Goatley, C. H. R., Tanner, J. K., & Bellwood, D. R. (2011). Quantifying relative diver effects in underwater visual censuses. *PLoS ONE*, 6, e18965. <https://doi.org/10.1371/journal.pone.0018965>
- Dinerstein, E., Vynne, C., Sala, E., Joshi, A. R., Fernando, S., Lovejoy, T. E., Mayorga, J., Olson, D., Asner, G. P., Baillie, J. E. M., & Burgess, N. D. (2019). A global Deal for nature: Guiding principles, milestones, and targets. *Science Advances*, 5(4), eaaw2869. <https://doi.org/10.1126/sciadv.aaw2869>
- Eisaguirre, J. H., Eisaguirre, J. M., Davis, K., Carlson, P. M., Gaines, S. D., & Caselle, J. E. (2020). Trophic redundancy and predator size class structure drive differences in kelp forest ecosystem dynamics. *Ecology*, 101, e02993. <https://doi.org/10.1002/ecy.2993>
- Freedman, R. M., Brown, J. A., Caldow, C., & Caselle, J. E. (2020). Marine protected areas do not prevent marine heatwave-induced fish community structure changes in a temperate transition zone. *Scientific Reports*, 10, 21081. <https://doi.org/10.1038/s41598-020-77885-3>
- Froese, R., & Pauly, D. (Eds.). (2023). *FishBase*. World Wide Web electronic publication. www.fishbase.org
- Gallagher, A. J., Brownscombe, J. W., Alsdairy, N. A., Casagrande, A. B., Fu, C., Harding, L., Harris, S. D., Hammerschlag, N., Howe, W., Huertas, A. D., Kattan, S., Kough, A. S., Musgrove, A., Payne, N. L., Phillips, A., Shea, B. D., Shipley, O. N., Sumaila, U. R., Hossain, M. S., & Duarte, C. M. (2022). Tiger sharks support the characterization of the world's largest seagrass ecosystem. *Nature Communications*, 13, 6328. <https://doi.org/10.1038/s41467-022-33926-1>
- Giakoumi, S., Scianna, C., Plass-Johnson, J., Micheli, F., Grorud-Colvert, K., Thiriet, P., Claudet, J., Di Carlo, G., Di Franco, A., Gaines, S. D., García-Charton, J. A., Lubchenco, J., Reimer, J., Sala, E., & Guidetti, P. (2017). Ecological effects of full and partial protection in the crowded Mediterranean Sea: A regional meta-analysis. *Scientific Reports*, 7, 8940. <https://doi.org/10.1038/s41598-017-08850-w>
- Gillett, D. J., Pondella, D. J., Freiwald, J., Schiff, K. C., Caselle, J. E., Shuman, C., & Weisberg, S. B. (2012). Comparing volunteer and professionally collected monitoring data from the rocky subtidal reefs of Southern California, USA. *Environmental Monitoring and Assessment*, 184, 3239–3257. <https://doi.org/10.1007/s10661-011-2185-5>
- Gleason, M., Fox, E., Ashcraft, S., Vasques, J., Whiteman, E., Serpa, P., Saarman, E., Caldwell, M., Frimodig, A., Miller-Henson, M., Kirlin, J., Ota, B., Pope, E., Weber, M., & Wiseman, K. (2013). Designing a network of marine protected areas in California: Achievements, costs, lessons learned, and challenges ahead. *Ocean and Coastal Management*, 74, 90–101. <https://doi.org/10.1016/j.ocecoaman.2012.08.013>
- Grorud-Colvert, K., Claudet, J., Tissot, B. N., Caselle, J. E., Carr, M. H., Day, J. C., Friedlander, A. M., Lester, S. E., de Loma, T. L., Malone, D., & Walsh, W. J. (2014). Marine protected area networks: Assessing whether the whole is greater than the sum of its parts. *PLoS ONE*, 9, e102298. <https://doi.org/10.1371/journal.pone.0102298>
- Hackradt, C. W., García-Charton, J. A., Harmelin-Vivien, M., Pérez-Ruzafa, Á., Le Diréach, L., Bayle-Sempere, J., Charbonnel, E., Ody, D., Reñones, O., Sanchez-Jerez, P., & Valle, C. (2014). Response of rocky reef top predators (Serranidae: Epinephelinae) in and around marine protected areas in the Western Mediterranean Sea. *PLoS ONE*, 9, e98206. <https://doi.org/10.1371/journal.pone.0098206>
- Hamilton, S. L., & Caselle, J. E. (2015). Exploitation and recovery of a sea urchin predator has implications for the resilience of southern California kelp forests. *Proceedings of the Royal Society B*, 282, 20141817. <https://doi.org/10.1098/rspb.2014.1817>
- Hamilton, S. L., Caselle, J. E., Malone, D. P., & Carr, M. H. (2010). Incorporating biogeography into evaluations of the Channel Islands marine reserve network. *Proceedings of the National Academy of Sciences of the United States of America*, 107, 18272–18277. <https://doi.org/10.1073/pnas.0908091107>
- Harasti, D., Malcolm, H., Gallen, C., Coleman, M. A., Jordan, A., & Knott, N. A. (2015). Appropriate set times to represent patterns of rocky reef fishes using baited video. *Journal of Experimental Marine Biology and Ecology*, 463, 173–180. <https://doi.org/10.1016/j.jembe.2014.12.003>
- Hernan, G., Dubel, A. K., Caselle, J. E., Kushner, D. J., Miller, R. J., Reed, D., Sprague, J. L., & Rassweiler, A. (2022). Measuring the efficiency of alternative biodiversity monitoring sampling strategies. *Frontiers in Marine Science*, 9, 820790. <https://doi.org/10.3389/fmars.2022.820790>
- Honeyman, C., Carlson, P., Jainese, C., Parsons-Field, A., Eisaguirre, J., Davis, K., Giraldo-Ospina, A., Spiecker, B., & Caselle, J. (2023). Data from: Correspondence among multiple methods provides confidence when measuring marine protected area effects for species and assemblages. *Zenodo Digital Repository*: v1.1, 2023. <https://doi.org/10.5281/ZENODO.8361053>
- House, P. H., & Allen, L. G. (2022). Differences in trophic and community structure of kelp forest fishes inside and outside of three long-standing MPAs in the Southern California bight. *Bulletin of the Southern California Academy of Sciences*, 121, 1–26. <https://doi.org/10.3160/0038-3872-121.1.1>
- Hughes, B., Beas, R., Barner, A., Brewitt, K., Brumbaugh, D., Cerny-Chipman, E., Close, S., Coblenz, K., Nesnera, K., Drobnitch, S., Figurski, J., Shanahan, B., Friedman, M., Freiwald, J., Heady, K., Heady, W., Hettinger, A., Johnson, A., Karr, K., & Carr, M. (2017). Long-term studies contribute disproportionately to ecology and policy. *BioScience*, 67, 271–281. <https://doi.org/10.1093/biosci/biw185>
- Jainese, C. J. (2023). *Marine protected area effects on mesophotic rocky reef fish communities using baited remote underwater video -how deep is your BRUV-*. University of California.
- Jessop, S. A., Saunders, B. J., Goetze, J. S., & Harvey, E. S. (2022). A comparison of underwater visual census, baited, diver operated and remotely operated stereo-video for sampling shallow water reef fishes. *Estuarine, Coastal and Shelf Science*, 276, 108017. <https://doi.org/10.1016/j.ecss.2022.108017>
- Karnauskas, M., & Babcock, E. (2012). Comparisons between abundance estimates from underwater visual census and catch-per-unit-effort in a patch reef system. *Marine Ecology Progress Series*, 468, 217–230. <https://doi.org/10.3354/meps10007>
- Langlois, T., Goetze, J., Bond, T., Monk, J., Abesamis, R. A., Asher, J., Barrett, N., Bernard, A. T. F., Bouchet, P. J., Birt, M. J., Cappel, M., Currey-Randall, L. M., Driessen, D., Fairclough, D. V., Fullwood, L. A. F., Gibbons, B. A., Harasti, D., Heupel, M. R., Hicks, J., ... Harvey, E. S. (2020). A field and video annotation guide for baited remote underwater stereo-video surveys of demersal fish assemblages. *Methods in Ecology and Evolution*, 11, 1401–1409. <https://doi.org/10.1111/2041-210X.13470>
- Lenihan, H. S., Gallagher, J. P., Peters, J. R., Stier, A. C., Hofmeister, J. K. K., & Reed, D. C. (2021). Evidence that spillover from marine protected areas benefits the spiny lobster (*Panulirus interruptus*) fishery in southern California. *Scientific Reports*, 11, 2663. <https://doi.org/10.1038/s41598-021-82371-5>
- Love, M. S. (1990). *Life history aspects of 19 rockfish species (Scorpaenidae: Sebastes) from the Southern California bight*. U.S. Department of Commerce.
- Lowry, M., Folpp, H., Gregson, M., & Suthers, I. (2012). Comparison of baited remote underwater video (BRUV) and underwater visual census (UVC) for assessment of artificial reefs in estuaries. *Journal of Experimental Marine Biology and Ecology*, 416–417, 243–253. <https://doi.org/10.1016/j.jembe.2012.01.013>

- Malone, D. P., Davis, K., Lonhart, S. I., Parsons-Field, A., Caselle, J. E., & Carr, M. H. (2022). Large scale, multi-decade monitoring data from kelp forest ecosystems in California and Oregon (USA). *Ecology*, 103(5), e3630. <https://doi.org/10.1002/ecy.3630>
- Marine Protected Area Monitoring Action Plan. (2018). California Department of Fish and Wildlife and California Ocean Protection Council.
- Murphy, H. M., & Jenkins, G. P. (2010). Observational methods used in marine spatial monitoring of fishes and associated habitats: A review. *Marine and Freshwater Research*, 61, 236. <https://doi.org/10.1071/MF09068>
- Nickols, K. J., White, J. W., Malone, D., Carr, M. H., Starr, R. M., Baskett, M. L., Hastings, A., & Botsford, L. W. (2019). Setting ecological expectations for adaptive management of marine protected areas. *Journal of Applied Ecology*, 56, 2376–2385. <https://doi.org/10.1111/1365-2664.13463>
- Ovando, D., Caselle, J. E., Costello, C., Deschenes, O., Gaines, S. D., Hilborn, R., & Liu, O. (2021). Assessing the population-level conservation effects of marine protected areas. *Conservation Biology*, 35, 1861–1870. <https://doi.org/10.1111/cobi.13782>
- Parker, D., Winker, H., Bernard, A. T. F., Heyns-Veale, E. R., Langlois, T. J., Harvey, E. S., & Götz, A. (2016). Insights from baited video sampling of temperate reef fishes: How biased are angling surveys? *Fisheries Research*, 179, 191–201. <https://doi.org/10.1016/j.fishres.2016.02.025>
- Pelletier, D., Leleu, K., Mou-Tham, G., Guillemot, N., & Chabanet, P. (2011). Comparison of visual census and high definition video transects for monitoring coral reef fish assemblages. *Fisheries Research*, 107, 84–93. <https://doi.org/10.1016/j.fishres.2010.10.011>
- Pondella, D. J., Caselle, J. E., Claisse, J. T., Williams, J. P., Davis, K., Williams, C. M., & Zahn, L. A. (2015). *South Coast baseline program final report: Kelp and shallow rock ecosystems*. California Sea Grant.
- Rassweiler, A., Dubel, A. K., Hernan, G., Kushner, D. J., Caselle, J. E., Sprague, J. L., Kui, L., Lamy, T., Lester, S. E., & Miller, R. J. (2020). Roving divers surveying fish in fixed areas capture similar patterns in biogeography but different estimates of density when compared with belt transects. *Frontiers in Marine Science*, 7, 272. <https://doi.org/10.3389/fmars.2020.00272>
- Saarman, E., Gleason, M., Ugoretz, J., Airame, S., Carr, M., Fox, E., Frimodig, A., Mason, T., & Vasquez, J. (2013). The role of science in supporting marine protected area network planning and design in California. *Ocean and Coastal Management*, 74, 45–56. <https://doi.org/10.1016/j.ocecoaman.2012.08.021>
- Shortis, M., Seager, J. W., Williams, A., Barker, B. A., & Sherlock, M. (2008). Using stereo-video for deep water benthic habitat surveys. *Marine Technology Society Journal*, 42, 28–37. <https://doi.org/10.4031/002533208787157624>
- Smith, J. G., Free, C. M., Lopazanski, C., Brun, J., Anderson, C. R., Carr, M. H., Claudet, J., Dugan, J. E., Eurich, J. G., Francis, T. B., Hamilton, S. L., Mouillot, D., Raimondi, P. T., Starr, R. M., Ziegler, S. L., Nickols, K. J., & Caselle, J. E. (2023). A marine protected area network does not confer community structure resilience to a marine heatwave across coastal ecosystems. *Global Change Biology*, 00, 1–18. <https://doi.org/10.1111/gcb.16862>
- Starr, R. M., Carr, M., Malone, D., Greenley, A., & McMillan, S. (2010). Complementary sampling methods to inform ecosystem-based management of nearshore fisheries. *Marine and Coastal Fisheries*, 2, 159–179. <https://doi.org/10.1577/C08-056.1>
- Starr, R. M., Wendt, D. E., Barnes, C. L., Marks, C. I., Malone, D., Waltz, G., Schmidt, K. T., Chiu, J., Launer, A. L., Hall, N. C., & Yochum, N. (2016). Data from: Variation in Responses of Fishes across Multiple Reserves within a Network of Marine Protected Areas in Temperate Waters. *Dryad Digital Repository*, <https://doi.org/10.5061/DRYAD.6HK4H>
- Stobart, B., Warwick, R., González, C., Mallol, S., Díaz, D., Reñones, O., & Goñi, R. (2009). Long-term and spillover effects of a marine protected area on an exploited fish community. *Marine Ecology Progress Series*, 384, 47–60. <https://doi.org/10.3354/meps08007>
- Sullivan-Stack, J., Aburto-Oropeza, O., Brooks, C. M., Cabral, R. B., Caselle, J. E., Chan, F., Duffy, J. E., Dunn, D. C., Friedlander, A. M., Fulton-Bennett, H. K., Gaines, S. D., Gerber, L. R., Hines, E., Leslie, H. M., Lester, S. E., MacCarthy, J. M. C., Maxwell, S. M., Mayorga, J., McCauley, D. J., ... Grorud-Colvert, K. (2022). A scientific synthesis of marine protected areas in the United States: Status and recommendations. *Frontiers in Marine Science*, 9, 849927. <https://doi.org/10.3389/fmars.2022.849927>
- Watson, D. L., Harvey, E. S., Anderson, M. J., & Kendrick, G. A. (2005). A comparison of temperate reef fish assemblages recorded by three underwater stereo-video techniques. *Marine Biology*, 148, 415–425. <https://doi.org/10.1007/s00227-005-0090-6>
- Watson, D. L., Harvey, E. S., Fitzpatrick, B. M., Langlois, T. J., & Shedrawi, G. (2010). Assessing reef fish assemblage structure: How do different stereo-video techniques compare? *Marine Biology*, 157, 1237–1250. <https://doi.org/10.1007/s00227-010-1404-x>
- Watson, J. L., & Huntington, B. E. (2016). Assessing the performance of a cost-effective video lander for estimating relative abundance and diversity of nearshore fish assemblages. *Journal of Experimental Marine Biology and Ecology*, 483, 104–111. <https://doi.org/10.1016/j.jembe.2016.07.007>
- Wendt, D. E., & Starr, R. M. (2009). Collaborative research: An effective way to collect data for stock assessments and evaluate marine protected areas in California. *Marine and Coastal Fisheries*, 1, 315–324. <https://doi.org/10.1577/C08-054.1>
- Willis, T. J., & Babcock, R. C. (2000). A baited underwater video system for the determination of relative density of carnivorous reef fish. *Marine and Freshwater Research*, 51, 755. <https://doi.org/10.1071/MF00010>
- Wilson, D. C. (1999). *Fisheries science collaborations: The critical role of the community*. Conference on holistic management and the role of fisheries and mariculture in the coastal community, 11–13 November 1999, Tjärnö Marine Biological Laboratory, Sweden.
- Yochum, N., Starr, R. M., & Wendt, D. E. (2011). Utilizing fishermen knowledge and expertise: Keys to success for collaborative fisheries research. *Fisheries*, 36, 593–605. <https://doi.org/10.1080/03632415.2011.633467>
- Ziegler, S. L., Brooks, R. O., Hamilton, S. L., Ruttenberg, B. I., Chiu, J. A., Fields, R. T., Waltz, G. T., Shen, C., Wendt, D. E., & Starr, R. M. (2022). External fishing effort regulates positive effects of no-take marine protected areas. *Biological Conservation*, 269, 109546. <https://doi.org/10.1016/j.biocon.2022.109546>

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

Figure S1: Results from a one-way analysis of similarity (ANOSIM) run on the Bray–Curtis dissimilarity matrix of range standardised groundfish abundance data from Anacapa Island. An *R*-value of 0.872 and *p*-value of 0.0001 indicate the fish community structure surveyed by each method is significantly different.

Figure S2: Results from a one-way analysis of similarity (ANOSIM) run on the Bray–Curtis dissimilarity matrix of range standardised groundfish abundance data from Santa Rosa Island. An *R*-value of 0.929 and *p*-value of 0.0001 indicate the fish community structure surveyed by each method is significantly different.

Table S1: Table comparing each of the MPA monitoring methods compared in the manuscript. For each method, approximate depth ranges and a selection of pros and cons are given. Methods are listed on the left-hand side.

Table S2: Table providing the scientific classifications and common names of groundfish included in the community structure analysis for both Santa Rosa and Anacapa Islands. This table includes information on trophic classifications, targeted status and thermal preferences for each species, as well as which MPA monitoring techniques it was observed on.

Table S3: The top 10 drivers of dissimilarity between BRUVs and CCFRP at Anacapa Island ordered by cumulative contribution from SIMPER analysis. Species names are listed on the left, with the cumulative contribution to dissimilarity on the right.

Table S4: The top 10 drivers of dissimilarity between BRUVs and SCUBA at Anacapa Island ordered by cumulative contribution from SIMPER analysis. Species names are listed on the left, with the cumulative contribution to dissimilarity on the right.

Table S5: The top 10 drivers of dissimilarity between CCFRP and SCUBA at Anacapa Island ordered by cumulative contribution from SIMPER analysis. Species names are listed on the left, with the cumulative contribution to dissimilarity on the right.

Table S6: The top 10 drivers of dissimilarity between BRUVs and CCFRP at Santa Rosa Island ordered by cumulative contribution

from SIMPER analysis. Species names are listed on the left, with the cumulative contribution to dissimilarity on the right.

Table S7: The top 10 drivers of dissimilarity between BRUVs and SCUBA at Santa Rosa Island ordered by cumulative contribution from SIMPER analysis. Species names are listed on the left, with the cumulative contribution to dissimilarity on the right.

Table S8: The top 10 drivers of dissimilarity between CCFRP and SCUBA at Santa Rosa Island ordered by cumulative contribution from SIMPER analysis. Species names are listed on the left, with the cumulative contribution to dissimilarity on the right.

How to cite this article: Honeyman, C., Carlson, P., Jainese, C., Parsons-Field, A., Eisaguirre, J., Davis, K., Giraldo-Ospina, A., Spiecker, B., & Caselle, J. E. (2023). Correspondence among multiple methods provides confidence when measuring marine protected area effects for species and assemblages. *Journal of Applied Ecology*, 60, 2699–2712. <https://doi.org/10.1111/1365-2664.14515>