



Potential protection effects in a small marine reserve: a demersal community analysis from stereo baited remote underwater video

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ABSTRACT: Monitoring is essential for efficient management of marine protected areas (MPAs) but often limited financially. Using low-cost stereo baited remote underwater video, we evaluated potential protection effects in the demersal assemblage of a marine reserve in European Atlantic temperate waters (southwestern Portugal). Due to a lack of before-MPA data, we compared multiple sites from the reserve inside and outside, investigated time trends, and assessed the potentially confounding effects of physical habitat. We found greater biomass of the legal-sized group (i.e. target above legal minimum landing size) inside the reserve than outside in the most recent year as opposed to the first year and a significant time increase in its abundance and biomass inside but not outside, indicating different trends in protection levels. On the contrary, biomass of the non-target group decreased significantly in both protection levels through time. One of the most targeted species, the legal-sized *Diplodus sargus*, reached significantly higher biomass inside than outside while also manifesting positive protection signs in abundance and length. Two legal-sized taxa showed no (*Diplodus vulgaris*) and negative (Mugilidae) protection responses, possibly due to low value and mobility of the latter. We detected positive responses in non-target wrasses *Labrus bergylta* and *Symphodus* spp., highlighting incidental fishing interest. Habitat did not interfere with protection effects. We suggest that community analysis based on stereo measurements renders sensitive protection indicators and offers a more thorough approach than prevailing abundance-focused analysis. The proposed low-cost monitoring solution can serve as a model to other MPAs.

KEY WORDS: Reserve effect · Stereo video · Community biomass · Demersal fish · Legal-sized species · Marine protected area · MPA · Stereo baited remote underwater video · SBRUV

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1. INTRODUCTION

Apart from main conservation objectives, marine protected areas (MPAs) commonly aim to rebuild fished populations and promote sustainable fisheries (Gell & Roberts 2003, Halpern & Warner 2003, Weigel et al. 2014). Local fisheries can benefit from MPAs through increased catches inside partially protected zones and in areas adjacent to marine reserves, which get enriched by the export of target (commercial) species at various life stages (Gell & Roberts 2003, Halpern & Warner 2003, Lenihan et al. 2021). However, theoretical and empirical studies indicate

that highly exploited and depleted fisheries are more likely to benefit from MPAs than well-managed, moderately exploited fisheries (Goñi et al. 2010, Kerwath et al. 2013, Buxton et al. 2014, Rassweiler et al. 2014), and not all MPAs result in fisheries benefits (Lédée et al. 2012, Cinner et al. 2014, Pita et al. 2020).

Well-managed MPAs rely on monitoring, which evaluates performance and supports adjustments of protection measures (Pomeroy et al. 2005, Fox et al. 2014, Gallacher et al. 2016). Researchers have widely reported eventual positive protection effects of marine reserves (i.e. no-take areas or zones of MPAs), mainly in densities, sizes, and biomass of target spe-

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cies (Tetreault & Ambrose 2007, Taylor et al. 2012, Horta e Costa et al. 2013, Malcolm et al. 2018), while numerous studies also found no or equivocal effects of MPAs for target species (Bacheler et al. 2016, Horta e Costa et al. 2020, Pickens et al. 2021) and no or negative effects for non-target (i.e. non-commercial) species (Mosquera et al. 2000, Micheli et al. 2004, Tetreault & Ambrose 2007, Horta e Costa et al. 2013). Various factors influence the success of MPAs, especially sound planning that optimises MPA design and size based on species habitat requirements and mobility (Le Quesne & Codling 2009, Agardy et al. 2011), level of protection (Horta e Costa et al. 2016, Giakoumi et al. 2017, Zupan et al. 2018), effective management (Gill et al. 2017, Grorud-Colvert et al. 2021), legal enforcement (Jennings et al. 1996, Giakoumi et al. 2017), and stakeholder involvement (Himes 2003, Giakoumi et al. 2018), the last 2 influencing the level of compliance with conservation measures.

Given the frequent lack of before-MPA data, most monitoring studies follow the after control–impact approach, i.e. a comparison between unprotected and protected locations after MPA establishment (Lester et al. 2009). However, such an approach assumes that control and impact locations were similar before the impact, which may not always be the case, leading to weak inference in some cases (Smokorowski & Randall 2017). Thus, in the absence of before-MPA data, control–impact studies must disentangle natural differences between control and impact locations, such as eventual differences in habitat (Miller & Russ 2014, Di Franco et al. 2021). Including multiple control locations and years assessed may improve the ability to detect protection effects (Underwood 2002). The after control–impact approach becomes more robust when pursued continuously for the long term, as contrasting time trends may occur in different protection levels: abundance increases over time in the most strongly protected areas, while it stagnates or decreases in unprotected areas (Alcala et al. 2005, Stobart et al. 2009, Sackett et al. 2014, Malcolm et al. 2018).

The lack of financial resources represents a barrier to increased monitoring of MPAs, especially in low-income countries (Gill et al. 2017, Bohorquez et al. 2019); cost-efficient sampling methods may facilitate monitoring worldwide. Lately, stereo baited remote underwater video (SBRUV) has experienced a rapid expansion, thanks to technical advances and declining prices of video equipment (Cappo et al. 2003, Mallet & Pelletier 2014, Unsworth et al. 2014, Letessier et al. 2015). Because of the remote character

of the method, SBRUV enables collecting multiple samples simultaneously, decreasing time and costs at sea (Watson et al. 2005, Langlois et al. 2010). It also avoids changes in fish behaviour due to the presence of divers in an underwater visual census (Francour et al. 1999, Watson & Harvey 2007), which is the traditional method of fish community surveys in MPAs. Apart from that, the technique combines several features worthwhile for MPA monitoring, especially (1) being non-extractive, which preserves species and habitats, in line with conservation principles (Malcolm et al. 2007, Unsworth et al. 2014); (2) being baited, which allows sampling of greater numbers of carnivorous, i.e. mostly target, species, which best indicates the effects of fishing exclusion (Willis & Babcock 2000, Claudet et al. 2006, Harvey et al. 2007, Goetze et al. 2015); and (3) being stereo, which results in more precise and accurate measurements compared to divers' estimates, increasing the quality of length and biomass data (Harvey et al. 2002, Letessier et al. 2015). While SBRUV represents a standardised MPA monitoring tool in Oceania (Goetze et al. 2015, Malcolm et al. 2018, Harasti et al. 2019), it remains underused in Europe and many other regions (Text S1 and Table S1 in the Supplement at www.int-res.com/articles/suppl/m708p079_supp.pdf).

So far, most baited video studies opted for a single camera, and more than a third of SBRUV studies did not explore the full potential of the stereo method, i.e. they analysed abundance but not length data (Whitmarsh et al. 2017). A similar pattern has existed in baited video monitoring of MPAs, where more than half of the articles used single video (Text S1). More than three-quarters of SBRUV studies in MPA monitoring skipped length measurements completely, or when they measured organisms, they did so only for some species (less than 10); 79% of studies did not include biomass in the analysis (Text S1). The above outlined patterns partially reflect that single-video technology preceded stereo-video development and that camera costs have only declined recently. Other reasons still limit the use of stereo video nowadays, especially the high costs of stereo annotation software and greater time required for analysis of stereo than single video, due to manual length measurements. However, baited video studies that do not include measurements miss the possibility to calculate community indicators of biomass and classify the assemblage relative to the legal minimum landing size of the organisms. But biomass can respond to protection in a more sensitive way than abundance, length, and richness (Tetreault & Ambrose 2007, Lester et al. 2009), and legal-sized target species (i.e.

target above legal minimum landing size) represent the key indicator of protection effects (Erzini et al. 1996, 2006, Di Franco et al. 2009). Dividing the fish community by the commercial status and legal minimum landing size of the organisms can also inform fisheries management about the efficiency of restrictions to fishing gears in the different zones of MPAs.

In this study, we investigated the protection effect of the Ilhotes do Martinhal marine reserve, established in 2011 in southwestern Portugal. Since its implementation, the marine reserve has lacked systematic monitoring and sufficient legal enforcement. Some illegal fishing activity, both small-scale commercial (octopus traps) and recreational (spear-fishing), has taken place inside the marine reserve (A. Belackova and other researchers pers. obs.; testimonials from local SCUBA divers). We aimed at understanding if, despite illegal fishing, this marine reserve provided protection effects in the demersal fish and cephalopod community. Previous research suggested that target seabreams *Diplodus* spp. may benefit from protection (Gil Fernández et al. 2016), but it lacked sufficient temporal and spatial replication. To improve the robustness of our control-impact design, we included multiple control locations from outside the marine reserve (i.e. area open to fishing with very few restrictions) and data from different seasons in 2 years (Underwood 2002, Willis et al. 2003a). Using the video images, we also examined the effects of physical habitat, to remove con-

founding effects (Miller & Russ 2014). Our SBRUV method had low entry costs: it consisted of low-cost action cameras in combination with a video-processing freeware (Neuswanger et al. 2016); its lower costs can contribute to long-term monitoring of this or other MPAs. Here, we explored the full potential of SBRUV by incorporating length measurements of the entire community in video processing; this also allowed us to analyse the effects of protection based on the relevance of organisms for fisheries management (target above legal minimum landing size, target below legal minimum landing size, non-target). To our knowledge, this is the first study in Europe that applied SBRUV in MPA assemblage monitoring.

2. MATERIALS AND METHODS

2.1. Study area

Established in 1995, the Parque Natural do Sudoeste Alentejano e Costa Vicentina (PNSACV) is the largest Portuguese coastal MPA, covering over ca. 290 km² of the southwestern coast of Portugal (Ministério do Ambiente e Recursos Naturais 1995). In 2011, the PNSACV's management plan implemented a zonation that included 4 marine reserves (i.e. no-take zones) and a large partially protected zone (called complementary zone in the management plan) (Presidência do Conselho de Ministros 2011) (Fig. 1). The marine

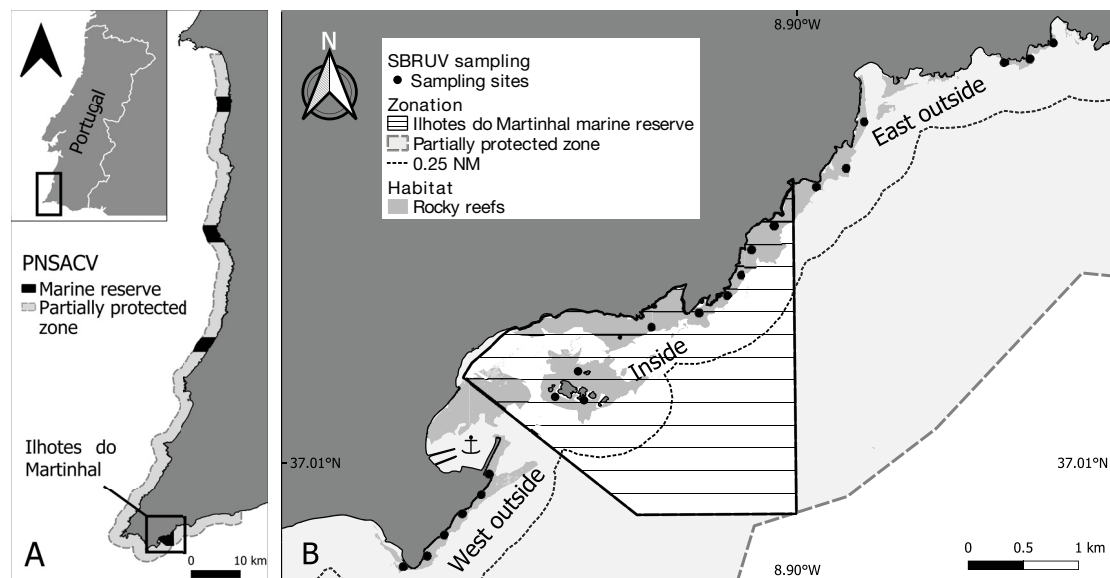


Fig. 1. (A) Location of the Ilhotes do Martinhal marine reserve within Parque Natural Sudoeste Alentejano e Costa Vicentina (PNSACV), southwest of Portugal; (B) rocky reefs sampled with stereo baited remote underwater video (SBRUV) inside the Ilhotes do Martinhal marine reserve and outside (i.e. in the partially protected zone <¼ nautical mile [NM] from the coastline). Outside sampling sites were situated both to the west and to the east of the marine reserve

reserves are closed to all fishing activities, except the commercial hand harvest of goose barnacle *Pollicipes pollicipes*, which is not evaluated in this study. Ilhotas do Martinhal represents one of the marine reserves of PNSACV; it covers ca. 3.9 km² of infralittoral rocky reefs and sandy bottom (Fig. 1).

The partially protected zone of PNSACV is open to fishing, with very few additional restrictions compared to the national fishing law, which applies outside of PNSACV. Hereafter, we only focused on the partially protected zone in the nearshore (<¼ nautical mile [NM] from coast), where rocky reefs—our target habitat—are located. In these rocky reefs, commercial traps and angling are allowed for licensed commercial fishing boats, but bottom longlines are forbidden (<½ NM; the only fishing gear whose allowed distance is different from the national fishing law). The other commercial gears used in this region are prohibited by the national fishing law in the nearshore both inside and outside of PNSACV (Table S2).

Recreational fishing, i.e. angling and spearfishing, is authorised in the partially protected zone, with daily catch limits slightly lower compared to the national fishing law. These include daily limits of fish and cephalopods per recreational fisher: (1) 7.5 kg in the partially protected zone vs. 10.0 kg for shore angling and 15.0 kg for spearfishing outside of PNSACV, (2) maximum of 2 specimens of the wrasse *Labrus bergylta* in the partially protected zone vs. no limit outside of PNSACV, and (3) maximum of 2 specimens of *Octopus vulgaris* in the partially protected zone vs. the same rule being valid only on weekends outside of PNSACV (Table S2). In addition, in the partially protected zone but not outside of PNSACV, seasonal closures apply to recreational (since 2009) and commercial (since 2011) shore angling of 2 target seabreams, *Diplodus sargus* and *D. vulgaris*, between 1 February and 15 March, and 1 non-target wrasse, *L. bergylta*, between 1 March and 31 May (Table S2). These closures partially coincide with the species' spawning seasons (Gonçalves et al. 2003, Villegas-Ríos et al. 2013, Abecasis et al. 2015a) but do not apply to other recreational or commercial fisheries allowed in the nearshore of the partially protected zone (i.e. spearfishing, traps, and vessel-based angling).

In addition, rocky reefs outside of PNSACV are too far away and have different habitat conditions. Thus, the

selected reefs from the partially protected zone provide the only adequate controls to the marine reserve, showing comparable depth profiles, habitats, and complexity.

2.2. Sample collection

The SBRUV system consisted of 2 small action cameras in waterproof housings, tightened on a wooden bar at 0.4 m distance from each other, facing the bait basket at an 8° inward angle (Fig. 2). Such distance between cameras provides sufficient accuracy for sampling of demersal organisms at distances below 5 m (Boutros et al. 2015). For each video sample, we estimated visibility as the distance from the cameras to the most remote measured object (mostly fish). Mean \pm SE visibility in our samples corresponded to 3.1 ± 0.09 m and maximum visibility to 4.8 m (the same in 2016 and 2018). We excluded samples that showed water visibility below 1.5 m (4.1 %) or had most of the field of view obstructed by rocky features within the 1.5 m distance from cameras (4.1 %). Low-cost action cameras (see Table S3 for approximate costs) are suitable for stereo-video sampling under such limited visibility, as they provide low measurement errors at short distances (Letessier et al. 2015). We used 3 SBRUV structures during sampling with GoPro Hero3-type cameras. Camera setup included medium field of view and full high definition (Langlois et al. 2020). Given their reduced size and low weight, the SBRUV structures are easy to transport and operate from a small vessel (ca. 6 m in length).

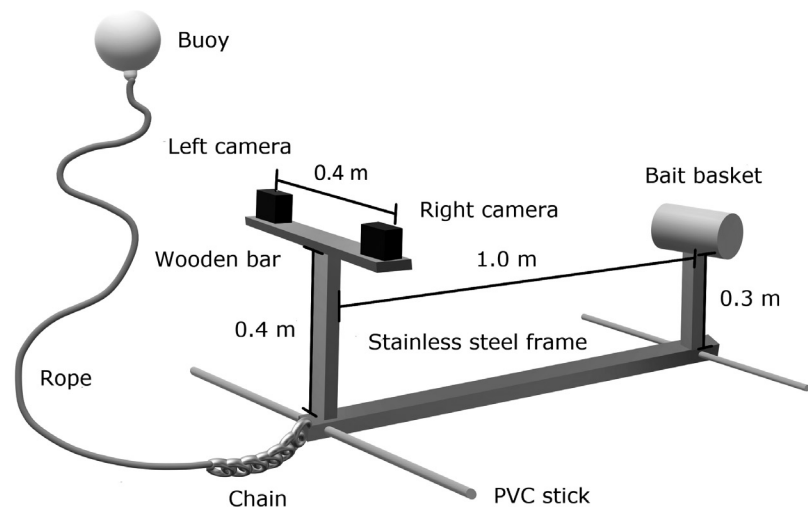


Fig. 2. Stereo baited remote underwater video structure used in marine reserve monitoring. Image created entirely by A. Belackova in Paint3D

Using SBRUV, we sampled demersal organisms at multiple rocky reefs inside the marine reserve (i.e. in Ilhotas do Martinhal) and in the surrounding outside (i.e. in the nearshore partially protected zone) (Fig. 1). Sampling took place during the same months of winter (March) and spring (April and May) of 2016 and 2018, on days with favorable sea conditions. Each month, 1 sampling campaign was held during 1 to 2 consecutive days. During each day, we sampled sites from both inside and outside (Fig. 1). This way, we guaranteed equivalent sea conditions (water visibility and temperature) for both protection levels, as the study area is small. Between years, sea water temperatures were comparable, varying between 14.0 and 16.2°C in 2016 and between 15.0 and 16.5°C in 2018. At each site, 1 SBRUV structure was deployed from a vessel, reaching depths between 7 and 18 m both inside and outside (mean depth \pm SE; inside: 11.07 \pm 0.41 m, outside: 11.31 \pm 0.46 m). The same sites were sampled in 2018 as in 2016, providing equal depth profiles. The rocky reefs reach up to ca. 25 m in depth both inside and outside, but these depths are not accessible for sampling from a vessel.

To ensure cost-efficient sampling, the SBRUV method allows sampling several sites simultaneously; this requires a separation distance between sites to guarantee sample independence (i.e. avoiding overlap of bait plumes and fish swimming between 2 deployed SBRUVs) (Cappo et al. 2007, Langlois et al. 2010, Goetze et al. 2015). We kept 250 m separation distance between sites, as recommended for 30 min video (Cappo et al. 2003, Langlois et al. 2020). The selection of sampling sites was random, while respecting the condition of minimum separation distance. Each underwater video sample lasted 30 min, based both on a local study that found this soak time sufficient to allow entry of most of the demersal species and to capture 95% of all abundance estimate events (Capaz 2013) and on other studies from temperate regions (Stobart et al. 2007, Harasti et al. 2015). Sampling was always done during daylight (between 09:00 and 17:00 h) to avoid crepuscular behaviour (Hobson et al. 1981, Harvey et al. 2007). Equal quantities (~200 g) and proportions of small pelagic fish (*Sardina pilchardus*, *Trachurus trachurus*, *Scomber colias*) and mussels (*Mytilus galloprovincialis*) were used as bait throughout the experiment (Willis & Babcock 2000, Capaz 2013).

Our sampling design may be seen as pseudoreplicated according to Hurlbert (1984), as a single marine reserve was evaluated using random sampling sites as replicates. However, we sampled sites situ-

ated at multiple rocky reefs both inside the marine reserve and outside, to guarantee sufficient representation of each protection level. The control sites sampled outside were located both to the west and to the east of the marine reserve (Fig. 1). Thus, including marine reserve replication would be desirable, but it was outside the scope of this study, and not done in the majority of other studies in the field of reserve monitoring.

2.3. Video processing

A freeware for Mac (VidSync 1.661) was used for video annotation (Neuswanger et al. 2016). Video processing included calibration (i.e. synchronisation, distortion correction, and 3-dimensional calibration) (Neuswanger et al. 2016), species identification and abundance counts, fish total length and cephalopod mantle length measurements (Horta e Costa et al. 2013, Harasti et al. 2018, Wilson et al. 2018), and physical habitat complexity categorisation. Abundance counts and length measurements were performed within the maximum visibility range of each sample. Abundance was estimated using MaxN—the maximum count of individuals of a species in a frame (Cappo et al. 2003, Watson et al. 2005). MaxN represents the standard metric in stationary video analysis (Whitmarsh et al. 2017, Langlois et al. 2020), although it tends to underestimate abundances at high species densities (Cappo et al. 2003, Schobernd et al. 2014, Stobart et al. 2015). However, the issue of high densities should not impact our study, as in our records, MaxN counts corresponded to low densities for all species, except for *Coris julis* (Table S4). Video annotation began as soon as the SBRUV structure landed on the sea bottom. Along the entire 30 min video, we annotated all potential MaxN frames for each species, i.e. frames that showed a higher number of individuals compared to the previous annotation. We added a buffer of 30 s to each potential MaxN frame, to let entire groups of fish enter the field of view while still avoiding double counts (Stobart et al. 2007, Whitmarsh et al. 2017). From the potential MaxN frames, we selected the one with the highest number of individuals. In line with the MaxN method (Langlois et al. 2020), we selected a MaxN frame with the maximum number of measurable individuals of a species to obtain lengths, i.e. a frame where the highest number of total fish length or cephalopod mantle length was visible in both videos (Stobart et al. 2007, Sackett et al. 2014). We only measured individuals swimming under less than a

Table 1. Physical habitat complexity categories (Hab1–4) of stereo baited remote underwater video samples collected in 2016 and 2018 inside and outside the Ilhotas do Martinhal marine reserve, assigned based on the character of sediment observed in the video field of view. The levels of criteria (surface type, abundance of hiding places) used to distinguish habitat categories and description of each habitat category are displayed

	Hab1	Hab2	Hab3	Hab4
Criteria				
Surface type	Flat	Flat to irregular	Irregular to highly irregular	Highly irregular
Abundance of hiding places	Very few	Few	Some	Common
Description	Rock mixed with sand, flat mother rock	Moderately structured mother rock	Highly structured mother rock or mixed with small or medium-sized boulders	Big boulders

25° angle relative to the cameras (Letessier et al. 2015). Physical habitat complexity was assessed visually (Polunin & Roberts 1993). Each sample was assigned one of the 4 physical habitat complexity categories, based on surface type and abundance of hiding places in the rocky sediment visible in the camera's field of view (Table 1) (Roberts & Ormond 1987, Horta e Costa et al. 2013). Additional observation of the habitat, performed before and after the SBRUV structure landing and at its retrieval, allowed completion of the habitat assessment. The surface type irregularity and abundance of hiding places increased from habitat 1 (hab1) towards habitat 4 (hab4). Thus, hab1 represented the least complex habitat formed by mixed sediment or flat mother rock and hab4 the most complex habitat formed by big boulders (Table 1). We provide a protocol for monitoring of MPAs with SBRUV and free software in Text S2.

2.4. Data analysis

Because the SBRUV system was designed for demersal organisms, we excluded pelagic (i.e. *Boops boops*, *T. trachurus*) and cryptobenthic (i.e. Gobiidae, Blenniidae) species (Watson et al. 2005, Holmes et al. 2013, Horta e Costa et al. 2013). Due to large schools (>200 individuals) that acted as outliers, *Sarpa salpa* was also removed from the analysis. Frequency of occurrence was calculated for each demersal taxa, based on the equation:

$$\text{Frequency of occurrence (\%)} = (\text{no. of samples containing the species} / \text{total no. of samples}) \times 100$$

Species richness (i.e. number of taxa) was expressed for each sample (Clarke & Gorley 2015). For each species, biomass was calculated based on the length–weight relationship from the nearest region (Gonçalves et al. 1997, Morato et al. 2001, Morey

et al. 2003, Froese & Pauly 2019). Species (or taxa) were classified based on their local commercial fishing status and size compared to the legal minimum landing size (Table 2), resulting in 3 groups: legal-sized (i.e. target organisms above legal minimum landing size), sublegal-sized (i.e. target organisms below legal minimum landing size), and non-target (i.e. organisms without commercial interest). Target species without a defined legal minimum landing size (i.e. *Oblada melanura*, *Muraena helena*, *Plectro rhinchus mediterraneus*) were assigned to the legal-sized group. Individual biomass was summed to provide species biomass (Willis et al. 2003b). Group abundance (MaxN) and group biomass were obtained as the sum of the abundance and biomass of all species belonging to the same group (i.e. legal-sized, sublegal-sized, non-target). Tables S4–S6 contain mean abundance, length, and biomass for groups and species.

The effects of protection (i.e. inside, outside), year (i.e. 2016, 2018), and habitat (i.e. hab1, hab2, hab3, hab4) on the response variables of richness, abundance, length, and biomass were tested for each group. Primer-E version 6.4.7.0 with PERMANOVA+ was used in species abundance and species biomass multivariate analysis (Anderson 2005). To decrease the dominance of species, data were square-root transformed (Clarke & Gorley 2015). A 3-way crossed main PERMANOVA with 9999 permutations was conducted based on the Bray-Curtis similarity matrix for factors protection, year, and habitat (Anderson 2001). A dummy species was added before similarity matrix calculation when necessary (Clarke & Gorley 2015). Pairwise PERMANOVA was conducted when the main PERMANOVA result was significant ($p < 0.05$).

Univariate analysis was developed in R version 3.4.3 (R Core Team 2021). For abundance and biomass of each group and species, the response to pro-

Table 2. Local commercial fishing status, market price, legal minimum landing size (MLS), and number of counted and measured individuals of each species sampled with stereo baited remote underwater video. Commercial status: target (i.e. with commercial interest) and non-target (i.e. without commercial interest). (–) MLS or market price are not defined for the species. MaxN: maximum count of individuals of a species in a frame

Species or taxon	Commercial status	Market price ^a (EUR kg ⁻¹)	MLS ^b (cm)	Counted (total MaxN)	Measured
<i>Coris julis</i>	Non-target	–	–	1234	762
<i>Centrolabrus exoletus</i>	Non-target	–	–	73	71
<i>Chromis chromis</i>	Non-target	–	–	4	3
<i>Ctenolabrus rupestris</i>	Non-target	–	–	42	36
<i>Conger conger</i>	Target	3.0	58.0	3	2
<i>Diplodus sargus</i>	Target	7.8	15.0	409	302
<i>Diplodus vulgaris</i>	Target	1.7	15.0	439	323
<i>Diplodus cervinus</i>	Target	11.6	15.0	18	15
<i>Dicentrarchus labrax</i>	Target	17.7	36.0	16	13
<i>Labrus bergylta</i>	Non-target	–	–	35	30
<i>Labrus mixtus</i>	Non-target	–	–	2	2
<i>Loligo vulgaris</i>	Target	12.1	10.0	5	4
Mugilidae	Target	1.8	20.0	52	36
<i>Mullus surmuletus</i>	Target	16.1	15.0	4	4
<i>Muraena helena</i>	Target	7.0	–	4	3
<i>Oblada melanura</i>	Target	–	–	31	20
<i>Octopus vulgaris</i>	Target	6.5	0.75 kg	19	17
<i>Pagrus auriga</i>	Target	14.6	–	9	8
<i>Pagrus pagrus</i>	Target	16.8	20.0	3	3
<i>Plectorhinchus mediterraneus</i>	Target	5.5	–	1	0
<i>Pseudocaranx dentex</i>	Non-target	–	–	6	5
<i>Sepia officinalis</i>	Target	6.1	10.0	5	4
<i>Serranus atricauda</i>	Non-target	–	–	9	9
<i>Serranus cabrilla</i>	Non-target	–	–	159	149
<i>Sparus aurata</i>	Target	11.5	19.0	1	1
<i>Spondyllosoma cantharus</i>	Target	2.1	23.0	5	5
<i>Symphodus</i> spp.	Non-target	–	–	24	23
<i>Trisopterus luscus</i>	Target	3.2	17.0	3	2
Total				2615	1852

^aMean market price at fish auction per kilogram, calculated from data provided by Directorate-General for Natural Resources, Safety and Maritime Services for Lagos and Sagres harbours in 2016 and 2017

^bIn centimeters, unless noted otherwise

tection was assessed through inside:outside ratios, assuming depth and environmental conditions (water visibility and temperature) being the same in both protection levels. Ratios allow testing and quantifying differences between 2 treatments (e.g. control vs. impact) (Hedges et al. 1999) and were used in previous studies, including protection effects assessments (Hamilton et al. 2010, Miller et al. 2012, Abecasis et al. 2015b). The inside:outside ratios were expressed as the mean of inside:outside seasonal (winter, spring) ratios. Due to the lack of sufficient data for single target species, numerous species were grouped together as other legal-sized and other sub-legal-sized (Table 3). Length ratios were calculated only for the most common species, i.e. legal-sized *D. sargus* and *D. vulgaris*, sublegal-sized *D. vulgaris*, and non-target *C. julis* and *Serranus cabrilla*. We

expected that protection effects may intensify the longer the time since protection (Molloy et al. 2009, Giakoumi et al. 2017) and potentially lead to different patterns between 2016 and 2018 inside than outside. To unveil differences in time trends between protection levels, mean year (2018:2016) ratios were also calculated for group and species abundance and biomass, based on seasonal ratios.

The ln of the mean seasonal ratios and their 95 % CIs (CI = $1.96 \pm \text{SE of the ln ratio}$) were used to assess differences between protection levels and years (Hedges et al. 1999, Hamilton et al. 2010, Miller et al. 2012, Abecasis et al. 2015b). The response variables were significantly greater inside than outside when $\ln \text{ ratio} \pm 95\% \text{ CI} > 0$ and, vice versa, significantly lower inside than outside when $\ln \text{ ratio} \pm 95\% \text{ CI} < 0$. The same principle applied when interpreting year

Table 3. Frequency of occurrence (FO) and number of sampled individuals (N, between brackets) of target species above legal minimum landing size (legal-sized, LS+), target species below legal minimum landing size (sublegal-sized, LS-), and non-target species, by protection level (inside, outside). Totals per group (legal-sized, sublegal-sized, non-target) and sub-group (other legal-sized, other sublegal-sized) in **bold**. (–) No data available

	FO in % (N)					FO in % (N)	
	Inside		Outside			Inside	Outside
	LS+	LS–	LS+	LS–			
Target species	100.0 (243)	84.2 (105)	95.7 (275)	82.6 (151)	Non-target species	100.0 (673)	98.0 (915)
<i>Diplodus sargus</i>	94.7 (134)	15.8 (15)	82.6 (132)	19.6 (21)	<i>Coris julis</i>	100.0 (531)	98.0 (703)
<i>Diplodus vulgaris</i>	65.8 (53)	68.4 (76)	71.7 (79)	80.4 (115)	<i>Serranus cabrilla</i>	84.2 (67)	86.0 (92)
Mugilidae	18.4 (11)	–	34.8 (25)	–	<i>Labrus bergylta</i>	52.6 (20)	28.0 (15)
Other target (sub-group)	47.4 (43)	10.5 (7)	37.0 (37)	13.0 (9)	<i>Ctenolabrus rupestris</i>	42.1 (17)	42.0 (25)
<i>Oblada melanura</i>	23.7 (14)	–	6.5 (17)	–	<i>Centrolabrus exoletus</i>	36.8 (19)	34.0 (54)
<i>Diplodus cervinus</i>	23.7 (9)	–	8.7 (5)	2.2 (1)	<i>Symphodus</i> spp.	31.6 (14)	18.0 (10)
<i>Pagrus auriga</i>	10.5 (5)	2.6 (1)	2.2 (2)	–	<i>Serranus atricauda</i>	7.9 (3)	12.0 (6)
<i>Dicentrarchus labrax</i>	7.9 (7)	2.6 (1)	4.3 (2)	4.3 (3)	<i>Chromis chromis</i>	2.6 (1)	6.0 (3)
<i>Muraena helena</i>	5.3 (2)	–	4.3 (2)	–	<i>Labrus mixtus</i>	2.6 (1)	2.0 (1)
<i>Octopus vulgaris</i>	5.3 (2)	18.4 (7)	4.3 (2)	13.0 (6)	<i>Pseudocaranx dentex</i>	–	2.0 (6)
<i>Loligo vulgaris</i>	5.3 (3)	–	–	2.2 (1)			
<i>Sepia officinalis</i>	2.6 (1)	–	6.5 (3)	–			
<i>Pagrus pagrus</i>	2.6 (1)	2.6 (2)	–	–			
<i>Plectorhinchus mediterraneus</i>	2.6 (1)	–	–	–			
<i>Mullus surmuletus</i>	–	–	6.5 (4)	–			
<i>Conger conger</i>	–	–	4.3 (2)	–			
<i>Sparus aurata</i>	–	–	2.2 (1)	–			
<i>Spondyllosoma cantharus</i>	–	5.3 (3)	–	4.3 (2)			
<i>Trisopterus luscus</i>	–	–	–	2.2 (2)			

ratios. Inside:outside ratios were calculated for pooled data from both years and for each year separately; year ratios were calculated for each protection level separately. Back-transformed ln ratios and their SEs were used in figures (Hamilton et al. 2010); thus, the reference value for back-transformed ratios in figures is 1 (instead of zero for ln ratios).

The effect of physical habitat complexity was analysed using generalised linear models (GLMs). Data distributions were chosen based on the discrete or continuous character of each dataset, while verifying the model fit using the diagnostic plots of model assumptions (i.e. homogeneity of variances, normality of residuals, independence of samples) and checking for overdispersion (Pekár & Brabec 2016). Negative binomial distribution was selected for abundance and gamma distribution for biomass of community datasets and individual species (Pekár & Brabec 2016). GLMs were also used to test for differences in richness between protection, year, and habitat—estimated with Poisson distribution (Pekár & Brabec 2016). When the main effect indicated significant differences (GLM, $p < 0.05$), a pairwise comparison was made using the χ^2 test of least square means (Searle et al. 1980) or interaction means in the event of a significant interaction (De Rosario-Martinez 2013). To understand whether physical complexity differs be-

tween the inside and outside, differences between the proportions of habitat categories in protection levels were tested using the χ^2 test of independence (Maindonald & Braun 2010).

With the purpose of testing a viable long-term monitoring method, we assessed the equipment, fieldwork, and laboratory costs of the SBRUV. When calculating costs per sample, we considered the average number of valid samples obtained per day of sampling and the average time needed for 1 stereo-video sample analysis (Table S3) (Perkins et al. 2013, La Manna et al. 2021).

3. RESULTS

We analysed 88 SBRUV samples that complied with the minimum standards of visibility and field of view, corresponding to 44 h of stereo video. The mean time of 1 SBRUV sample (i.e. 30 min recording) analysis was approximately 4.6 h. We obtained data from 38 samples from inside and 50 from outside, which corresponded to 28 samples from 2016 and 60 from 2018. In 2016, 4 of the outside samples lacked measurements (due to calibration or camera failure).

We identified 29 demersal taxa (26 fish and 3 cephalopods) from 15 families; most taxa belonged to

Table 4. Responses to protection, year, and habitat in assemblage abundance and biomass from PERMANOVA. * $p < 0.05$

Group of species	Variable	Protection	Year	Habitat	Protection \times Year	Protection \times Habitat
Legal-sized	Abundance	<0.01*	0.01*	0.32	0.01*	0.87
	Biomass	0.11	0.07	0.31	0.23	0.72
Sublegal-sized	Abundance	0.54	0.29	0.43	0.65	0.55
	Biomass	0.45	0.89	0.78	0.49	0.97
Non-target	Abundance	0.71	0.02*	0.03*	0.04*	0.19
	Biomass	0.99	<0.01*	0.66	0.15	0.28

the Sparidae (9 species) and Labridae (6 species) families. We counted 2615 individuals, of which we were able to measure 1852 (70.8%) (Table 2). Legal-sized individuals belonged to 16 taxa, sublegal-sized to 10 taxa, and non-target to 10 taxa. In both protection levels, legal-sized individuals (inside: $N = 243$, outside: $N = 275$) prevailed over sublegal-sized individuals (inside: $N = 105$, outside: $N = 151$). Legal-sized individuals dominated for *Diplodus sargus* and Mugilidae, while sublegal-sized individuals prevailed for *Octopus vulgaris* and *D. vulgaris*, independently from protection level (Table 3).

3.1. Response to protection

We encountered 24 taxa inside and 27 outside in the 38 samples gathered inside and 50 samples outside (Table 3). In both protection levels, the same species dominated the community: legal-sized *D. sargus* and *D. vulgaris*, sublegal-sized *D. vulgaris*, and non-target *Coris julis* and *Serranus cabrilla* (Table 3). The majority (10 of 16) of the legal-sized species had greater frequencies of occurrence inside than outside (Table 3). These included several commercially valuable species, such as *D. sargus* and various rarely detected but highly priced target species (e.g. *D. cervinus*, *Pagrus auriga*, *Dicentrarchus labrax*) (Tables 2 & 3). From the non-target species, *Labrus bergylta* and *Symphodus* spp. also had greater frequencies of occurrence inside the reserve, while the dominant *C. julis* and *S. cabrilla* showed similar frequencies in both protection levels. Two legal-sized taxa of relatively low market value, Mugilidae and *D. vulgaris*, occurred with lower frequencies inside than outside (Tables 2 & 3).

We did not find any significant differences in richness for the tested factors (GLM, protection: $p = 0.44$, year: $p = 0.27$, habitat: $p = 0.10$). The interaction between protection and year was also non-significant for richness (GLM, protection \times year: $p = 0.06$, protec-

tion \times habitat: $p = 0.74$). The multivariate abundance of legal-sized and non-target species interacted significantly between protection and year (PERMANOVA, legal-sized: $p = 0.01$, non-target: 0.04; Table 4). Pairwise tests confirmed a significant protection effect in each of the years in the multivariate abundance of legal-sized and non-target species, but these effects showed an opposite direction in each year (pairwise PERMANOVA, $p < 0.05$).

Based on the inside:outside ratios, groups of species (i.e. legal-sized, sublegal-sized, non-target) responded to protection in an opposite way in 2016 than in 2018, showing ln ratios < 0 in 2016 but > 0 in 2018 — causing non-significant pooled ratios for both years (Fig. 3). In 2016, the 3 groups attained significantly lower values of abundance and/or biomass inside than outside: biomass of the legal-sized group was 35 % lower inside than outside, abundance and biomass of the sublegal-sized group was 37 % lower, and abundance of the non-target group 20 % lower (Fig. 3). The species most responsible for the results of the groups in 2016 included legal-sized Mugilidae; legal-sized *O. vulgaris*; and non-target *L. bergylta*, *Ctenolabrus rupestris*, and *S. cabrilla*. All showed significantly lower values inside than outside.

In 2018, the legal-sized group showed significantly greater biomass inside than outside (approximately twice as great inside than outside) (Fig. 3). Legal-sized *D. sargus* contributed to the 2018 result through significantly greater biomass inside than outside.

We unveiled a potential positive protection effect in legal-sized *D. sargus* that was significant (80 % greater biomass inside than outside) in both years pooled (Fig. 4) and also when looking at each year separately (84 % in 2016 and 90 % in 2018). Both abundance and length drove the results in biomass of legal-sized *D. sargus* in 2018; both were significantly greater inside than outside in 2018 but not in 2016. In 2016, abundance and length ln ratios of legal-sized *D. sargus* were > 0 (i.e. inside $>$ outside) but not sig-

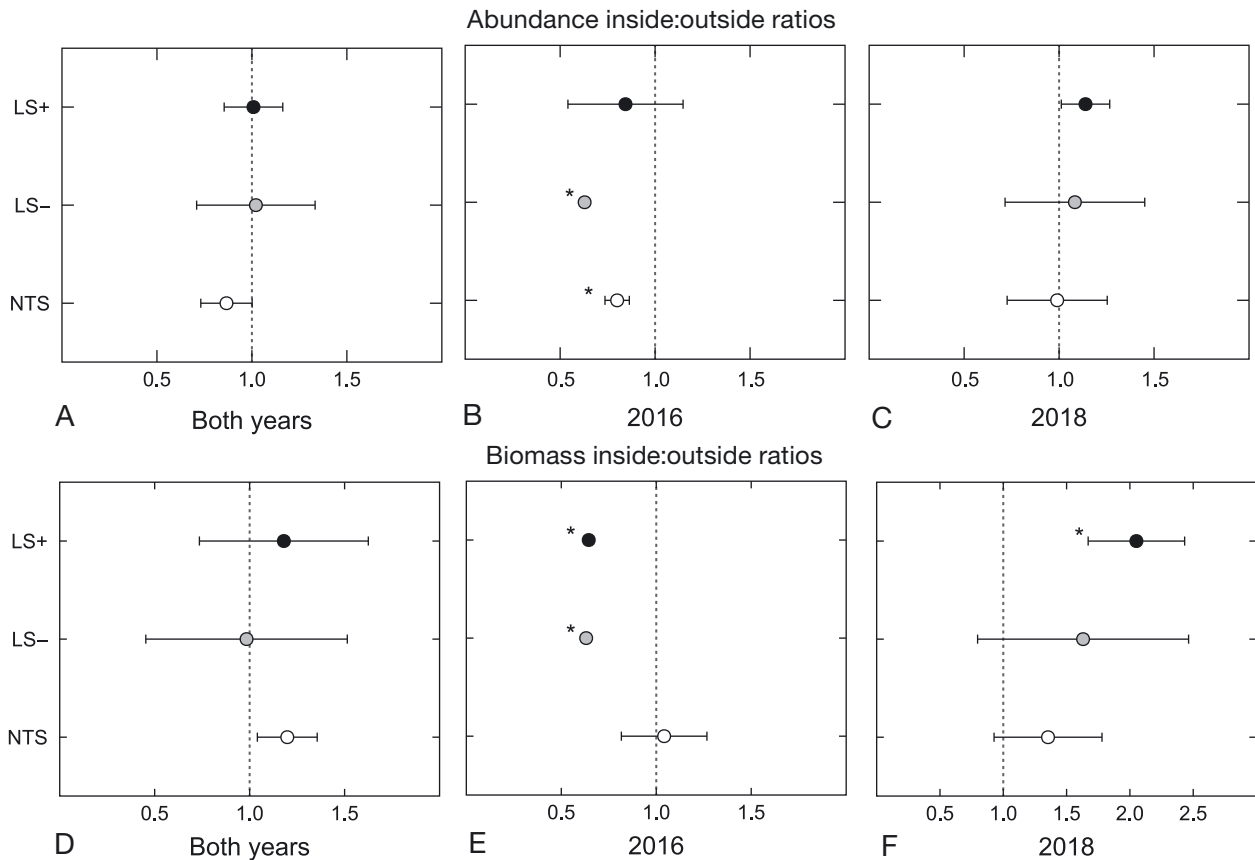


Fig. 3. Back-transformed \ln inside:outside ratios for groups of species (mean ratio \pm SE). Abundance in (A) both years pooled, (B) year 2016, (C) year 2018; biomass in (D) both years pooled, (E) year 2016, (F) year 2018. Groups of species: LS+: legal-sized (i.e. target above legal minimum landing size); LS-: sublegal-sized (i.e. target below legal minimum landing size); NTS: non-target (i.e. without commercial interest). Protection levels: inside (marine reserve); outside (partial protection ~ fished area). *Significant ratio: variable significantly greater inside than outside when \ln ratio \pm 95 % CI > 0 and, vice versa, significantly lower inside than outside when \ln ratio \pm 95 % CI < 0

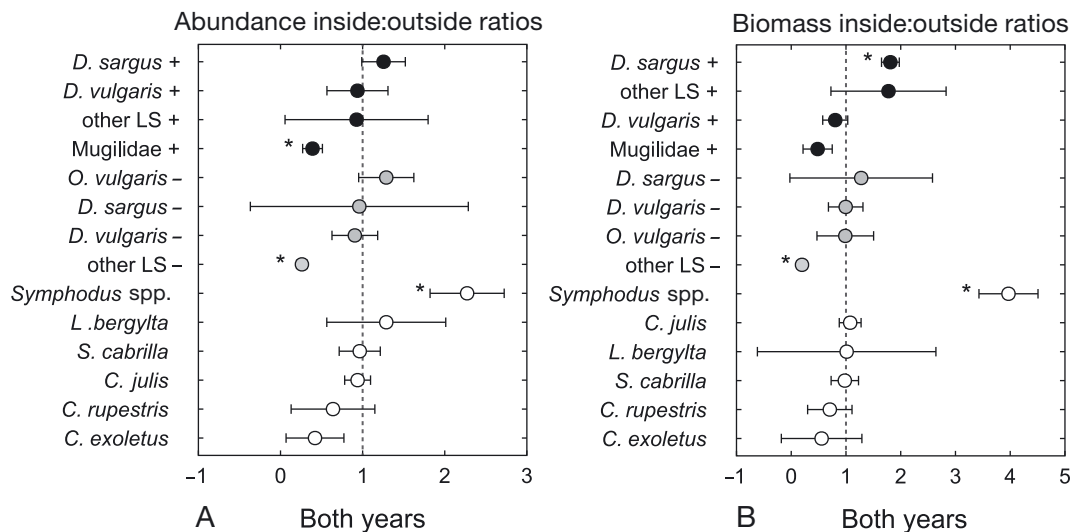


Fig. 4. Back-transformed \ln inside:outside ratios for individual species (mean ratio \pm SE). (A) Abundance in both years pooled, (B) biomass in both years pooled. Groups of species and protection levels as defined in Fig. 3. Rarely observed non-target species are not displayed. *D. sargus*: *Diplodus sargus*; *D. vulgaris*: *Diplodus vulgaris*; *O. vulgaris*: *Octopus vulgaris*; *L. bergylta*: *Labrus bergylta*; *S. cabrilla*: *Serranus cabrilla*; *C. julis*: *Coris julis*; *C. rupestris*: *Ctenolabrus rupestris*; *C. exoletus*: *Centrolabrus exoletus*. *Significant ratio (see Fig. 3 for details)

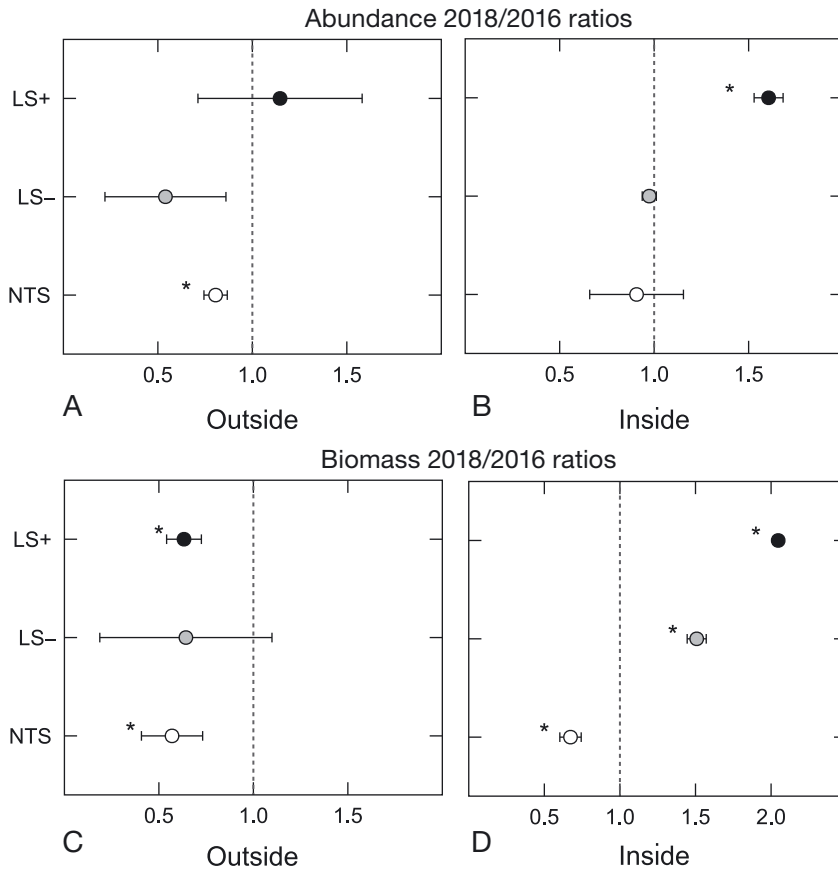


Fig. 5. Back-transformed ln year ratios for groups of species (mean ratio \pm SE). Abundance (A) outside, (B) inside; biomass (C) outside, (D) inside. Groups of species and protection levels as defined in Fig. 3. *Significant ratio

nificantly. We found an overall positive significant result in abundance and biomass of non-target *Symphodus* spp.; its biomass was almost 4 times greater inside than outside (Fig. 4). Despite that, in both years, we detected a potential negative protection effect in legal-sized Mugilidae (abundance 60 % significantly lower inside than outside) and in other sub-legal-sized species (abundance 74 % and biomass 80 % significantly lower) (Fig. 4), but the latter had relatively low frequencies of occurrence to be accurately assessed (Table 3). We did not find any significant differences in species length in both years pooled.

3.2. Response to year

Both inside and outside the reserve, differences in richness between years were non-significant (χ^2 test of interaction means, inside: $p = 0.43$, outside: $p = 0.06$), although the outside result was marginal and pointed at slightly greater richness in 2016 than in 2018 (outside: mean richness \pm SE, 2016: 7.8 ± 0.6 , 2018: 6.1 ± 0.3). Based on the year ratios, inside the reserve, biomass of the legal-sized group doubled

significantly between 2016 and 2018, while it decreased by 37 % outside (Fig. 5). Inside the reserve, abundance of the legal-sized group provided the same significant result as biomass, corresponding to a 61 % increase (Fig. 5). Other legal-sized species contributed to the legal-sized group results: their biomass grew 4 times inside, although non-significantly (Fig. 6).

As confirmed by significant year ratios, the results in legal-sized *D. sargus* were equivalent in both protection levels: an increase in biomass between 2016 and 2018 occurred both inside (42 % increase) and outside (48 % increase). However, abundance of legal-sized *D. sargus* only increased significantly inside. Outside, both abundance and length showed an increase between years, but this increase was non-significant. In contrast, legal-sized Mugilidae decreased in both protection levels, with a smaller decrease inside (48 % decrease) than outside (76 % decrease) (Fig. 6). Legal-sized *D. vulgaris* increased its abundance significantly outside the reserve but not inside (Fig. 6).

As in the legal-sized group, the sublegal-sized group had significantly greater biomass in 2018 than in 2016 inside the reserve—showing an increase by

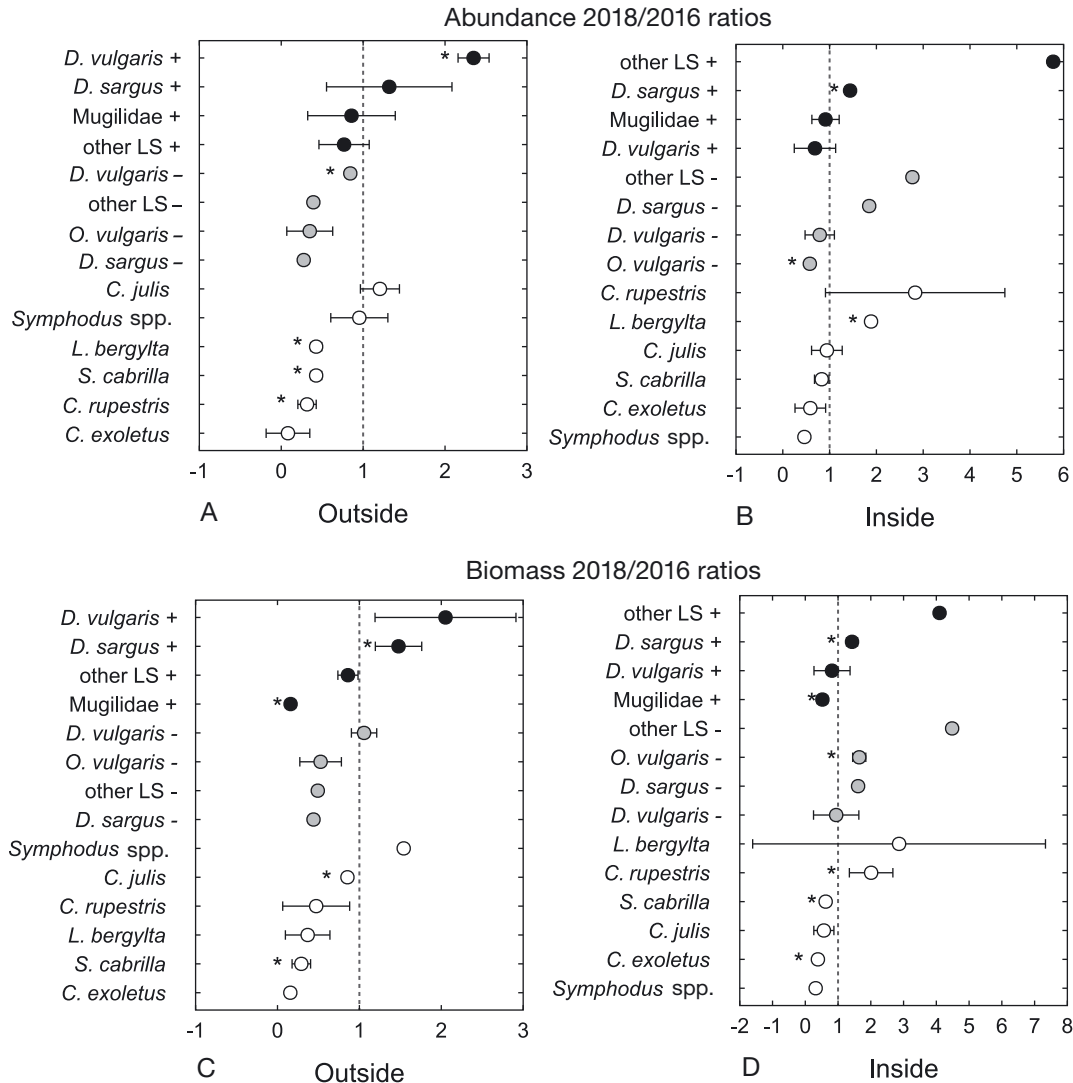


Fig. 6. Back-transformed ln year ratios for individual species (mean ratio \pm SE). Abundance (A) outside, (B) inside; biomass (C) outside, (D) inside. Groups of species and protection levels as defined in Fig. 3; species names as defined in Fig. 4. Rarely observed non-target species are not displayed. *Significant ratio

half (Fig. 5). With a 65 % significant and 61 % non-significant increase in biomass, sublegal *O. vulgaris* and *D. sargus* contributed to the result of the sublegal-sized target group. But other sublegal-sized species influenced the group's result even more, as their biomass increased 4.5 times, although non-significantly (Fig. 6).

Biomass of the non-target group decreased significantly between 2016 and 2018 in both protection levels by 40 %, and abundance decreased significantly outside (Fig. 6). Together with the majority of non-target species, *S. cabrilla* and *Centrolabrus exoletus* mostly contributed to the non-target group's results; the 2 species showed a consistent negative response to year in both protection levels (Fig. 6). Given its significant year ratios, the non-target *L.*

bergylta increased its abundance inside but decreased outside—following the same pattern as the legal-sized group (Fig. 6). There were no significant differences in length between years inside or outside.

3.3. Response to physical habitat complexity

Richness was similar across physical habitat categories (GLM, $p = 0.10$). Assemblage abundance differed significantly between habitats for non-target species (PERMANOVA, $p = 0.03$, Table 4; GLM, $p = 0.045$). Significantly lower abundance of non-target species existed in hab1 (i.e. the least complex) than in hab4 (i.e. the most complex) (χ^2 test

of least square means, $p = 0.04$). Two non-target species, *C. exoletus* (GLM, $p < 0.01$) and *C. rupestris* (GLM, $p < 0.001$), and 1 sublegal-sized species, *D. vulgaris* (GLM, $p < 0.05$), had significantly different abundance and biomass between habitats and were lower in physically less complex habitats (i.e. hab1, hab2) than in more complex habitats (i.e. hab3, hab4) (*C. exoletus*: hab1 < hab4; *C. rupestris*: hab1 < hab3, hab4; sublegal-sized *D. vulgaris*: hab1 < hab4; χ^2 test of least square

means, $p < 0.05$) (Fig. 7). We did not find any significant differences between habitats in legal-sized species.

The proportions of habitat complexity categories differed significantly between inside and outside (χ^2 test of independence, $p = 0.004$). Habitats of lower physical complexity (hab1 and hab2) were more frequent inside than outside, whereas the habitat of greatest complexity (hab4) had greater proportions outside than inside (Fig. 8).

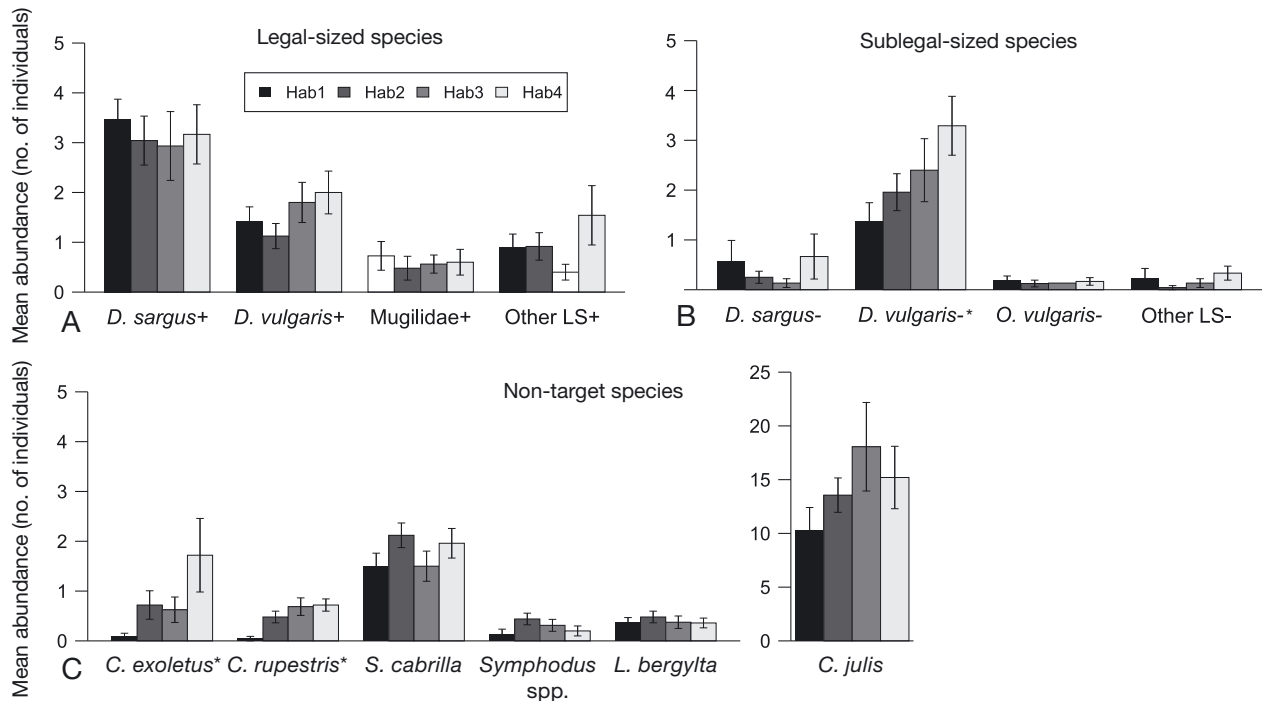


Fig. 7. Mean abundance (±SE) of species in habitat categories (Hab1–4). Abundance of (A) legal-sized species (i.e. target above legal minimum landing size) (LS+), (B) sublegal-sized species (i.e. target below legal minimum landing size) (LS-), (C) non-target species (i.e. without commercial interest). Species names as defined in Fig. 4. Rarely observed non-target species are not displayed. *Significant differences in abundance between habitat categories (χ^2 test of least square means, $p < 0.05$; *C. exoletus*: hab1 < hab4; *C. rupestris*: hab1 < hab3, hab4; sublegal-sized *D. vulgaris*: hab1 < hab4)

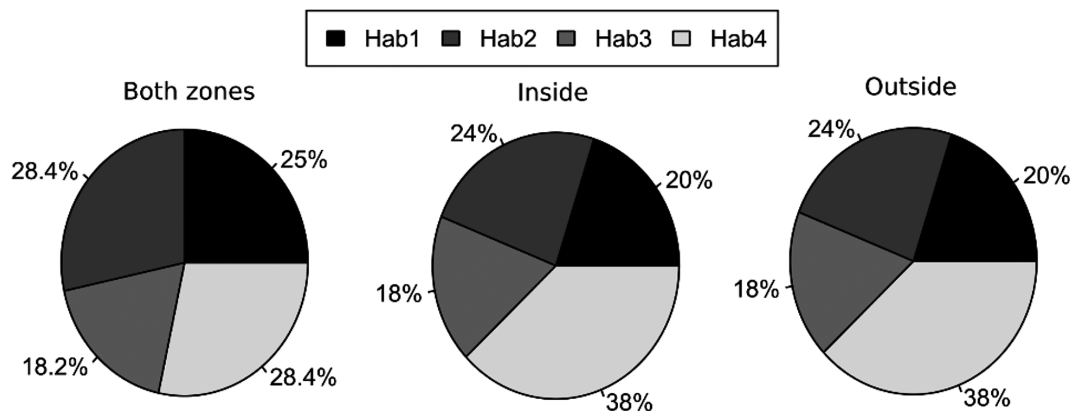


Fig. 8. Proportions of physical habitat complexity categories (Hab1–4) in both protection zones, inside (marine reserve) and outside (partial protection ~ fished area)

4. DISCUSSION

More than 5 yr after its implementation, the Ilhotes do Martinhal reserve seems to provide some positive protection effects, at least for valuable target species in the region. Responses in the legal-sized group (i.e. target above legal minimum landing size) supported potential positive protection effects: the group's abundance and biomass increased significantly inside the reserve between years but not outside, where biomass decreased. The legal-sized group's biomass was also significantly higher inside than outside in the most recent year of sampling, contrary to the first year of sampling. Legal-sized target species serve as the best indicator of protection effects that result from fisheries exclusion (Mosquera et al. 2000, Micheli et al. 2004, Claudet et al. 2006, Horta e Costa et al. 2013). In contrast to the result in the legal-sized group, we found that biomass of the non-target group decreased significantly in both protection levels between years. Non-target species—being less impacted by fisheries—mostly lack protection effects or respond negatively to protection (Mosquera et al. 2000, Micheli et al. 2004, Tetreault & Ambrose 2007, Horta e Costa et al. 2013).

In the absence of before-MPA data, the analysis of trends between years after protection is central, as it reinforces the results of the inside–outside comparison. Time analyses unveiled opposite trends inside compared to outside for the legal-sized group, but not the non-target group, suggesting that the group most sensitive to fishing exclusion (i.e. legal-sized) is benefiting from this marine reserve. Analysed as a group due to low individual species occurrences, the other legal-sized species included fish of high commercial value (e.g. *Pagrus auriga*, *Dicentrarchus labrax*, *Diplodus cervinus*). The other legal-sized species, many of which are large-bodied, contributed greatly to the community biomass; this subgroup largely influenced the time trends in the legal-sized group, by its increase inside but not outside between years.

The sublegal-sized group (i.e. target below legal minimum landing size) increased its biomass significantly inside but not outside between years, suggesting some protection benefits for juveniles of target species. However, abundance of the sublegal-sized group did not change between years, and the group did not show protection effects in the inside–outside comparison. Sublegal-sized species can experience positive reserve effects, while, like non-target species, they show more habitat complexity dependencies, which might interact with protection (Horta e Costa et al. 2013, 2020). In this study, results in the

sublegal-sized group were independent from habitat complexity, while an overall weaker protection effect occurred in the sublegal-sized compared to the legal-sized group.

The significant 40 % decline in biomass of the non-target group between years might reflect potential indirect negative protection side effects—due to enhanced competition and predation—that may affect non-target species inside and potentially also outside the reserve, e.g. through spillover (Micheli et al. 2004). However, as the decline in the non-target group was similar inside and outside, factors other than protection, such as interannual natural variability, may have influenced our results. We attributed the decrease in the non-target group mainly to *Serranus cabrilla* and *Centrolabrus exoletus*, although other non-target species contributed as well. Due to its high territoriality and low mobility, *S. cabrilla* seems susceptible to negative protection effects (García-Rubies & Zabala 1990, Micheli et al. 2004, Horta e Costa et al. 2013). Mortality from by-catch might have also influenced the decrease in non-target species outside, as *S. cabrilla* suffers from high mortality due to barotrauma when accidentally jigged (Alós 2008).

Apart from the legal-sized group, potential positive protection effects existed at the single legal-sized species level. We found a potentially positive response to protection in the legal-sized *Diplodus sargus* that had, overall, significantly 80 % greater biomass inside than outside. This species has served as a preferred protection effects indicator in coastal MPAs (D'Anna et al. 2011, Horta e Costa et al. 2013, Di Lorenzo et al. 2014, Abecasis et al. 2015b, Aspillaga et al. 2016, Belo et al. 2016). As our study highlights, the high value of legal-sized *D. sargus* for fisheries implies that this is one of the species in the region that can benefit the most from fisheries exclusion. Local recreational and commercial anglers highly appreciate *D. sargus*—it is the most abundant species in their catches, particularly in the region of this marine park (Veiga et al. 2010), and also the second most targeted species by Portuguese spearfishers (Assis et al. 2018). Previous studies from other Portuguese coastal MPAs reported positive protection effects in legal-sized *D. sargus* within less than 4 yr of protection, indicating the species' high sensitivity to permanent fishing closures and ability to quickly recover even in small areas, due to its high site fidelity (Horta e Costa et al. 2013, Abecasis et al. 2015b, Pereira et al. 2017). As in the region of our study, nearshore angling and spearfishing used to be intense at the Arrábida marine reserve (ca. 180 km north of our study area). Despite their mostly recreational character, the exclusion of such fishing tech-

niques may explain the recovery of legal-sized *D. sargus* (Horta e Costa et al. 2013). Thus, our results reinforce that highly targeted species with reduced home ranges, like *D. sargus*, can benefit from small marine reserves that, for such species, can serve as an efficient fisheries management tool. On the other hand, at Ilhotes do Martinhal, fishers may benefit from adult spillover from the marine reserve, as biomass of the legal-sized *D. sargus* increased significantly not only inside but also outside between years. However, our study was not designed to assess spillover. Spawning closures to shore angling of *D. sargus* in the outside area since 2009 potentially also contributed to its biomass increase.

Apart from biomass, legal-sized *D. sargus* showed some potential positive protection effects in abundance and length: abundance increased significantly inside but not outside between years, and both variables were significantly greater inside than outside in the second year of sampling but not in the first year. The latter results reflect a slightly delayed response in abundance and length, compared to biomass. Abundance tends to improve more slowly, as recruitment success varies with environmental conditions, and spawning biomass requires several years to build up (Russ & Alcala 1996, García-Charton et al. 2004, Shelton & Mangel 2011). Length and biomass usually give the first signs of protection effects, as they respond directly to exclusion of fishing, which targets large specimens (Erzini et al. 1996, 2006, Di Franco et al. 2009), and as fish may grow at a faster pace inside of unfished than fished areas (Carbonara et al. 2022). But it is biomass that represents the most sensitive metric of early protection effects due to its exponential increase with length, while abundance and length undergo less pronounced changes (Tetreault & Ambrose 2007, Lester et al. 2009).

At the same time, 2 non-target taxa also showed signs of potential positive protection effects: *Labrus bergylta* abundance increased significantly inside but declined outside between the 2 years, and *Symphodus* spp. had significantly greater abundance and biomass inside than outside, suggesting that some fishing interest may exist in these species, as the more exploited the species, the greater its response to protection (Lester et al. 2009). As in the legal-sized species, large-bodied non-target species (such as *L. bergylta*) can respond positively to protection, as fishermen decide to keep them when by-caught (Mosquera et al. 2000, Malcolm et al. 2018). Regional shore anglers keep most *L. bergylta* specimens and about half of *Symphodus* spp. (Veiga et al. 2010); this fishing behaviour may explain the potential positive

protection effects in both taxa. Seasonal closures apply to shore angling of *L. bergylta* during its spawning season, reflecting some subsistence, recreational, or even commercial interest in this non-target species (Ministérios da Defesa Nacional, da Agricultura, do Desenvolvimento Rural e das Pescas e do Ambiente e do Ordenamento do Território 2011, Presidência do Conselho de Ministros e Ministérios da Defesa Nacional, da Economia, da Inovação e do Desenvolvimento, da Agricultura, do Desenvolvimento Rural e das Pescas e do Ambiente e do Ordenamento do Território 2011, Villegas-Ríos et al. 2013). However, *Symphodus* spp. had relatively low frequencies of occurrence, limiting the strength of our results. Fishing interest may also exist in non-target *S. cabrilla* when by-caught, due to its larger sizes, but our results did not detect any potential positive effects of protection in this species.

Not all individual legal-sized species responded positively to protection: the second most abundant legal-sized *Diplodus vulgaris* significantly increased its abundance between years outside the reserve but not inside; legal-sized Mugilidae had significantly greater abundance outside than inside, and their abundance and biomass declined in both protection levels between years. Several mutually non-exclusive mechanisms may explain these results. (1) Low-valued target species respond weakly or negatively to protection, as they tend to be less exploited (Le Quesne & Codling 2009, Lester et al. 2009). Both *D. vulgaris* and Mugilidae have low economic values; Mugilidae even do not appear as a separate group in landing statistics, due to their very low commercial catches (Directorate-General for Natural Resources, Safety and Maritime Services 2019). Although regional commercial landings of *D. vulgaris* are similar to those of *D. sargus*, and between the largest from all demersal species (Directorate-General for Natural Resources, Safety and Maritime Services 2019), the high selectivity of allowed commercial fishing gear targeting such species in the nearshore reserve outside (i.e. hooks except bottom longlines) may lead to lower exploitation of legal-sized *D. vulgaris*, due to its lower value. Recreational fisheries definitely exhibit lesser interest in *D. vulgaris*: the species' catches by recreational anglers reach only one-third of *D. sargus* catches in number and one-sixth in weight (Veiga et al. 2010); it is also absent from the list of preferred spearfishing species, unlike *D. sargus* (Assis et al. 2018). Thus, our findings reinforced previous general results about the benefits of Ilhotes do Martinhal for the genus *Diplodus* (Gil Fernández et al. 2016), but here we showed that not all species of the genus responded to protec-

tion positively. (2) Competition due to partially overlapping diets as well as distinct feeding preferences can lead to habitat splitting between *D. vulgaris* and *D. sargus* (Sala & Ballesteros 1997, Gonçalves & Erzini 1998, Osman & Mahmoud 2009); this may also explain why legal-sized *D. vulgaris* increased outside between the 2 years but not inside, where legal-sized *D. sargus* prevailed. (3) Mugilidae represent a vagile taxa, with low residency and migratory behaviour (Arechavala-Lopez et al. 2010, Whitfield et al. 2012); due to high mobility, they might not be efficiently protected by a relatively small reserve (Kramer & Chapman 1999, Le Quesne & Codling 2009, Weeks et al. 2017). In addition, relatively low frequencies make the results in Mugilidae less representative, compared to legal-sized *D. sargus* or *D. vulgaris*.

We analysed differences in physical habitat complexity because it may confound the protection effects: habitat complexity often correlates positively with species richness and abundance (García-Charlton & Pérez-Ruzafa 2001, Miller & Russ 2014). Physical habitat complexity was significantly greater outside than inside, but habitat and protection effects did not interact. Despite lower physical complexity in the sites assessed, the marine reserve supported equal richness and community abundance as the outside area. In line with existing research, only small fishes (*C. exoletus*, *Ctenolabrus rupestris*, sublegal-sized *D. vulgaris*) showed significant habitat preferences, independently from protection level: they prioritised more complex habitats, which provide greater abundance of hiding places from predators (Scharf et al. 2006, Kovalenko et al. 2012). At the same time, our SBRUV method undersamples small benthic fish species (e.g. Blenniidae, Gobiidae) (Stobart et al. 2007, Lowry et al. 2012), and the bait attracts predators that may repel other species (Harvey et al. 2007, Goetze et al. 2015), potentially leading to an underestimated species richness. Underestimated richness may explain the lack of habitat effect in richness, as small fish species increase the richness of complex habitats, where they occupy crevices and holes.

The natural park (PNSACV) includes terrestrial protection and land use restrictions, which have hampered the economic development of the region, leaving an important role to fisheries as a source of income. Prior to the PNSACV implementation in 2011, the local community expressed disagreement with the protected area, including recreational fishing restrictions (Thaman et al. 2016). The top-down designation, the lack of community involvement, and the absence of enforcement, observed here and in many MPAs, can undermine the success of marine

reserves (Himes 2003, Thaman et al. 2016, Giakoumi et al. 2018). Pre- and post-reserve fishing pressures—both in the vicinity or illegal—influence productivity and speed of recovery (Kellner et al. 2007, McDermott et al. 2019). These management flaws and lack of compliance seemed to have diminished the productivity of the Ilhotes do Martinhal marine reserve, as illegal fishing takes place inside the reserve, mostly with commercial octopus traps and recreational spearfishing (A. Belackova and other researchers pers. obs.; testimonials from local SCUBA divers). Although as a group, legal-sized species seemed to benefit from the marine reserve, most individual legal-sized species had very low occurrences in both protection levels, preventing single species assessments. Spearfishing can regulate populations of large-bodied predatory species, e.g. *D. labrax* and *Sparus aurata* (Harmelin et al. 1995, Jouvenel & Pollard 2001, Coll et al. 2004), and moderate protection effects if practiced inside reserves.

Rare encounters may also reflect reduced densities of individual species due to habitat availability, reserve size, and efficiency of the sampling method. The marine reserve might be too small and the rocky habitat too fragmented to guarantee efficient protection for mobile species (Kramer & Chapman 1999, Afonso et al. 2009, Le Quesne & Codling 2009, Weeks et al. 2017). Furthermore, adults of some large demersal species prefer deeper reefs (e.g. *P. pagrus*) than those available in this coastal area (maximum depth of 25 m) (Manooch & Hassler 1978, Erzini et al. 1996).

In some species, low occurrences may relate to the SBRUV method. Given the stationary video that uses a single MaxN (abundance) frame per species, the method is relatively less efficient for sampling of solitary and/or cryptic species (e.g. *Octopus vulgaris*, *Sepia officinalis*, *Conger conger*, *Muraena helena*), which become less frequent, compared to social and/or more conspicuous species (e.g. legal-sized *D. sargus*, *D. vulgaris*, *Coris julis*) (Colton & Swearer 2010, Lowry et al. 2012). Furthermore, sampling during the day underestimates predatory fish that are more active at night (Helfman 1986, Abecasis & Erzini 2008, Shoji et al. 2017). However, for diver-averse species, the SBRUV still offers a better sampling method than diver-based methods (Assis et al. 2013, Goetze et al. 2015). Although previous studies found 30 min of SBRUV soak time appropriate for temperate reef fish assemblages (Harasti et al. 2015), including those in our study region (Capaz 2013), recent research recommended at least 60 min for temperate regions (Birt et al. 2021). Thus, the 30 min soak time used in this study may have underestimated richness, abundance, and biomass,

especially as large-bodied mobile fishes need more time to show up (Birt et al. 2021). Furthermore, the choice of MaxN as a metric of abundance may bias the selection of frames towards smaller individuals, and influence length and biomass data, as fish in schools tend to be smaller than solitary individuals. Yet, that potential bias applies to both protection levels.

Although SBRUV is a static monitoring gear and does not cover the same area as an underwater visual census or a remotely operated vehicle (ROV), it attracts fish from a broader area than the measured field of view (Priode & Merrett 1996, Heagney et al. 2007). Several studies showed that SBRUV better performs in sampling of carnivorous fishes than diver-related methods (Langlois et al. 2010, Watson et al. 2010, Goetze et al. 2015), as many target species (i.e. mostly carnivores) are diver adverse and bait attracted. Recent research shows that SBRUV and ROV sample similar species richness, but it is SBRUV that better samples key target species (i.e. key indicators of fisheries exclusion) (Jessop et al. 2022). Previous studies found significant differences in MaxN and species richness when comparing baited with unbaited video samples (Cappo et al. 2003, Hardinge et al. 2013, Jones et al. 2020). The bait attraction area can be modelled using variables such as currents and fish swimming speeds (Priode & Merrett 1996). In practice, such models were applied in studies on abyssal scavengers (Priode & Merrett 1996, Henriques et al. 2002), where conditions best approach theoretical assumptions on constant water flow and immediate olfactory response to bait odour in fish (Hardinge et al. 2013). In shallow coastal environments, complex water mixing and abundant food supply prevent the application of such models (Hardinge et al. 2013). Instead, shallow water studies use minimum separation distance between SBRUV samples to avoid bait plume overlap (Cappo et al. 2007, Langlois et al. 2010, Goetze et al. 2015), as also applied in this study.

Although video processing with measurements is time consuming (Holmes et al. 2013), we showed that length measurements are essential in the evaluation of protection effects with SBRUV, as they allow for grouping by size and calculating biomass; our most relevant results were in the abundance and biomass of legal-sized groups and in the biomass of legal-sized *D. sargus*. Thus, baited video monitoring of the demersal community should opt for a stereo setup to guarantee measurements. Further developments in deep learning to automate SBRUV annotation should save time and reduce costs of the stereo method (Ditria et al. 2020, Veiga et al. 2022), which still limit

its wider usage. When looking for fast results of fisheries benefits, studies can restrict the analysis to the legal-sized group, legal-sized species, and large non-target species as indicators. Here, we showed that stereo analysis of the demersal community allows for understanding the patterns of these indicator groups, providing a more complete picture of protection effects than studies that focus mainly on abundance (a common situation in monitoring of MPAs with baited video, see Text S1).

We demonstrated that our SBRUV method represents a reliable tool for reserve monitoring: it supplied sensitive indicators of protection effects in legal-sized *D. sargus* and in the legal-sized group. Furthermore, by combining low-cost video equipment with a stereo-video annotation freeware (Neuswanger et al. 2016), our approach offers a more cost-competitive solution when compared to available commercial measurement hardware and software. We believe that the low-cost SBRUV can serve as a model for other MPAs and marine reserves as well as facilitate monitoring under budget-limited conditions, while the sampling design (e.g. optimal soak time, number of samples, depth profiles) should always be assessed individually based on the study aims and sampling location features. The cost-efficient solution adds to the existing advantages of SBRUV for MPA monitoring, which include the precision of measurements (Harvey et al. 2002, Letessier et al. 2015), the attraction of target species to the bait (Claudet et al. 2006, Goetze et al. 2015), and the high number of samples rendered in a sampling day (Watson et al. 2005, Langlois et al. 2010).

Although we obtained significant results for indicator groups and species, a relatively low sample size and only 2 years of data limit the power of our analysis. Our sample size ($N = 88$) may offer a reduced power to determine protection effects across 2 protection levels and 2 years. The separation distance kept between SBRUV deployments determined the number of samples that we could collect during each campaign within the available rocky bottom area, thus influencing the final sample size. Long-term monitoring is now required to increase the inference power of our control–impact design (Smokorowski & Randall 2017) and validate our results.

Considering the lack of before-MPA data, the inside–outside and time trend comparisons represent the most comprehensive approach that can be achieved to assess protection effects from fishing closures, when also controlling for confounding habitat effects and differentiating species by commercial status and size. With only 2 years of data, we were

able to detect patterns consistent with positive protection effects for the most targeted species (legal-sized *D. sargus*) and groups assessed (legal-sized group)—recognized indicators of reserve effects in coastal reefs—as well as for some non-target species that fishermen keep in by-catch (*L. bergylta*, *Symphodus* spp.). We obtained these results despite the fact that control (outside) locations belonged to a partially protected zone and not an area without fishing restrictions (not available in nearshore continental Portugal).

Acknowledgements. We thank Isidoro Costa and Nuno Sales Henriques (members of the MARSW project), Thomas Riedinger (intern at CCMAR), Diogo Paulo (leader of the CCMAR Scientific Dive Center), and Nick Coertze (former MSc student at the University of Algarve) for help with field sampling and video analysis; Algarve Dolphin Lovers for cooperation in terms of boat and skipper services; Jason Neuswanger for making the VidSync software available as a freeware for Mac OS; and the Directorate-General for Natural Resources, Safety and Maritime Services (DGRM) for providing data on target species market prices related to the MARSW project. This study was co-funded by the Operational Programme for Sustainability and Efficient Use of Resources (PO SEUR), Fundo Ambiental (Portuguese Environmental Fund), and by the municipalities of Aljezur, Odemira, and Vila do Bispo through the project 'Sistemas de Informação e Monitorização da Biodiversidade Marinha das Áreas Classificadas do Sudoeste Alentejano e Costa Vicentina—MARSW', with grant number POSEUR-03-2215-FC-000046. MARSW project had, as beneficiary, the Liga para a Proteção da Natureza (LPN), a Portuguese NGO, in collaboration with National Institute for the Conservation of Nature and Forests (ICNF). CCMAR received support from Portuguese national funds from the Foundation for Science and Technology (FCT) through strategic projects UIDB/04326/2020, UIDP/04326/2020, and LA/P/0101/2020. A.B. was supported by PhD research fellowship UI/BD/151307/2021 through FCT (Portugal) and B.H.C. by national funds through FCT (Portugal), in agreement with the University of Algarve, in the scope of Norma Transitória DL57/2016/CP1361/CT0038.

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Editorial responsibility: Paul Snelgrove,
St. John's, Newfoundland and Labrador, Canada;
Susanne Schüller, Oldendorf/Luhe, Germany
Reviewed by: 3 anonymous referees

Submitted: December 20, 2021

Accepted: February 1, 2023

Proofs received from author(s): March 9, 2023