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Evaluation of MPA effects on small-scale fisheries: A long-term landings-based monitoring approach

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ABSTRACT

Small-scale commercial fisheries represent a significant economic activity that can be affected by coastal Marine Protected Areas (MPAs). Official fishery landings data can serve as an effective means of evaluating the effects of MPAs on fisheries and harvested species, as they are available over long periods and do not incur any costs. However, the use of long-term landings as a solid baseline for pre-MPA conditions has been rare. In this study, we applied a Before-After-Control-Impact (BACI) design to long-term landings time series to assess the economic and ecological effects of a coastal multi-zone MPA in Portugal. We compared the landings and income per unit of effort (LPUE and IPUE) inside and outside the MPA after its implementation, and within the MPA before and after the implementation. Our results showed that the MPA had a positive influence on the LPUE and IPUE of the local fleet (54 landed taxa in the assemblage), based on significant positive trends inside the MPA after implementation, but not before or outside. We found significant positive responses to protection in four taxa with the highest LPUE: *Octopus vulgaris*, *Conger conger*, Soleidae, and Rajidae. The MPA's small no-take zones likely enhance species with small home ranges and their spillover, and, together with the controlled number of fishing licenses, contribute to positive MPA outcomes. However, the LPUE of *Muraena helena* and *Diplodus vulgaris* declined significantly inside the MPA between before and after MPA implementation, which could be attributed to enhanced inter-species competition. Despite encouraging LPUE trends within the MPA, the study revealed that prices evolved in a more favourable manner outside than inside the MPA, suggesting that future research in this topic may be necessary to ensure the proper valuation of fishing resources within this and other MPAs. Despite the common limitations of landings data, our study demonstrates that comparing long-term landings from pre- and post-MPA periods using a BACI design can be an efficient monitoring solution for budget and data-limited coastal MPAs.

1. Introduction

Commercial fishing plays a significant role in global food security, and supports various economic activities across the seafood value chain (Dyck and Sumaila, 2010; Garcia and Rosenberg, 2010; HLPE, 2014), while it is one of the most detrimental activities for marine species and habitats (Pauly et al., 1998; Chuenpagdee et al., 2003; Crowder et al., 2008; Froese et al., 2018; Hilborn et al., 2020). Marine Protected Areas (MPAs) help reduce the negative impacts on marine life by prohibiting extractive activities–namely fishing–in their no-take zones or limiting fishing effort and destructive gears in their partially protected zones (Horta e Costa et al., 2016; Grorud-Colvert et al., 2021; Blampied et al., 2023). However, displacement due to conservation measures can negatively affect commercial fishers if leading to the loss of historical fishing grounds, increased costs, or more intense competition elsewhere (Jones, 2008; Mangi and Austen, 2008; Batista et al., 2011; Gall and Rodwell, 2016; Ban et al., 2019). Nevertheless, the benefits of MPAs to fisheries can mitigate socio-economic impacts (Halpern et al., 2004; Goñi et al., 2010; Ban et al., 2019; Costello, 2024). MPAs can promote the growth of individuals of commercial species, which can result in increased reproductive outcomes and the rebuilding of spawning stocks (Tetreault and Ambrose, 2007; Lester et al., 2009; Carbonara et al.,

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2022). Additionally, benefits can extend to fisheries outside of MPAs through spillover from no-take to adjacent partially protected or to unprotected areas (Rowley, 1994; Goñi et al., 2008, 2011; Di Lorenzo et al., 2016). In the strictly regulated partially protected areas, restrictions on gear and fishing effort can have a positive impact on harvested fish species (Zupan et al., 2018; Smallhorn-West et al., 2022), benefiting the permitted fisheries (Blyth-Skyrme et al., 2006). The benefits provided to fisheries, as well as the design and management of MPAs, including co-design and enforcement, are important factors that shape attitudes towards MPAs, such as acceptability and compliance, and hence MPA effectiveness (Mangi and Austen, 2008; Lédée et al., 2012; Gall and Rodwell, 2016; Giakoumi et al., 2018).

Despite the critical role of fisheries in both affecting and being affected by MPAs, economic assessments of MPAs have been less common than ecological monitoring (Gill et al., 2017; O`Leary et al., 2021). The objective evaluation of the economic effects of MPAs on commercial fisheries has typically relied on both fishery-independent (i.e., scientific surveys) and fishery-dependent methods (i.e., data from commercial vessels) (Goñi et al., 2011; Di Lorenzo et al., 2016; Pennino et al., 2016). Fishery-dependent methods are less common but more adequate for economic evaluations of fisheries, as often providing long-term, year-round data, while fishery-independent data show distinct selectivity compared to the gears of the assessed fishery and incur high costs, resulting in their limited coverage (in time and space) (Di Lorenzo et al., 2016; Pennino et al., 2016). In contrast to fishery-dependent methods, fishery-independent methods adhere to systematic random sampling and provide data on non-target and prohibited species, which are unreported in commercial fisheries, thus facilitating a more objective ecological evaluation (Batista et al., 2009; Di Lorenzo et al., 2016; Pennino et al., 2016).

Among fishery-dependent methods, official fishery landings, in particular, are a cost-effective and readily available means of monitoring fishery production over time (Pauly et al., 2013). They require no fieldwork and are often publicly accessible, making them a useful tool for economic, as well as ecological, monitoring of MPAs, especially in areas where funding is limited (Pauly et al., 2013; Fortibuoni et al., 2017; Gill et al., 2017). However, fishery landings have limitations, including imprecise georeferencing of fishing activity, underreporting or mislabelling of catches, and variability due to changes in legislation, reporting requirements, and technology (Pauly et al., 2002, 2013; Blyth-Skyrme et al., 2006; Rodríguez-Cabello et al., 2008; Batista et al., 2009; Holmes et al., 2013; Bradley et al., 2019). Despite these drawbacks, fishery landings have been used to evaluate the economic and ecological effects of MPAs on fisheries (Wilcox and Pomeroy, 2003; Smith et al., 2006; Mangi et al., 2011; Kerwath et al., 2013; Batista et al., 2015; Lenihan et al., 2021). Despite their long-term potential, relatively few studies have included more than a couple of landings years from the before-MPA period when assessing protection effects (Yamasaki, 2002; Smith et al., 2006; Kerwath et al., 2013; Rife et al., 2013; Rolim & Ávila-da-Silva, 2016). But solid before-MPA data are crucial for monitoring, as they should secure a relevant baseline for evaluating changes over time (Guidetti, 2002; Pelletier et al., 2008; Lester et al., 2009). To effectively monitor protection effects and account for temporal and spatial variability, Smokorowski and Randall (2017) recommended incorporating a minimum of three years of data before and six years after the implementation of an MPA into a Before-After-Control-Impact (BACI) monitoring design. When well-designed, BACI is considered one of the most powerful approaches for environmental impact assessments, but so far, it has been underused to evaluate MPA effects on fisheries (Underwood, 1992; Smokorowski and Randall, 2017; Ban et al., 2019). There is a particular lack of studies that would include long-term landings data from the before-MPA period in a BACI design (maximum of 4 years of before-MPA landings, in Rife et al., 2013).

In this study, our main objective was to evaluate the protection effects on small-scale artisanal multi-gear fisheries, focusing on Parque Natural Sudoeste Alentejano e Costa Vicentina (PNSACV), the largest multi-zone coastal MPA in Portugal (ca 290 km^2), established in 2011. We aimed to assess the impact of the MPA on local fisheries and on the fish resources they target. To achieve this, we conducted a fishery-dependent monitoring approach using official fishery landings, which were readily available and provided a long time series of data, allowing us to compare fishery production before and after the implementation of the MPA. We compared landings of local fisheries from inside and outside the MPA from 10 years before and 10 years after the MPA implementation using a BACI analysis. The presented approach based on landings data could also serve as a low-cost model for monitoring other MPAs while contributing to the economic and ecological assessments of MPAs.

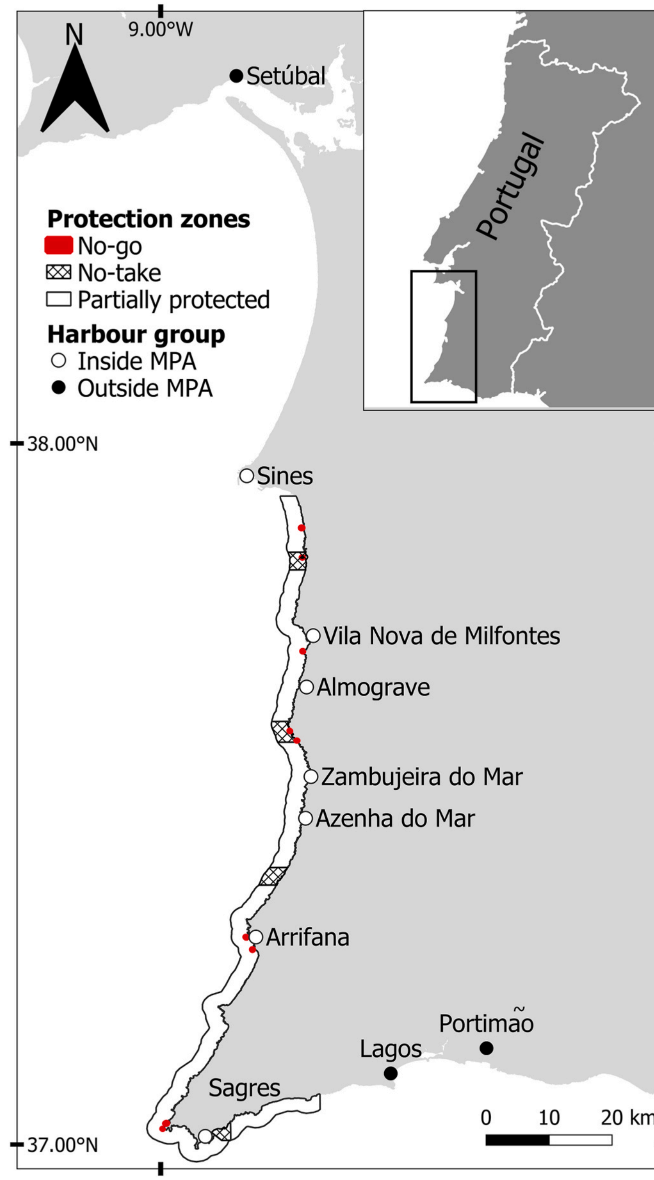

Fig. 1. Map of the study area showing the zonation of Parque Natural Sudoeste Alentejano e Costa Vicentina (PNSACV) MPA (red dots: no-go; dashed polygons: no-take: open polygon: partially protected area) and fishing harbours. Open circles: ○ Inside MPA, i.e., harbours with fishing fleet operating inside the MPA (Sines, Vila Nova de Milfontes, Almograve, Zambujeira do Mar, Arrifana, Sagres) Filled circles: ● Outside MPA, i.e., harbours with fishing fleet operating outside the MPA (Setúbal, Lagos, Portimão).

2. Material and methods

2.1. Study area

The PNSACV MPA, with an area of approximately 290 km^2, is the largest Portuguese coastal MPA, extending up to 2 km (1.1 nautical mile, NM) from coast (Fig. 1). In 2011, a management plan was implemented, defining zones with different protection measures (PCM, 2011), including no-go, no-take, and partially protected zones (Fig. 1). The MPA was primarily established for conservation purposes, while it also aimed at promoting economic growth and human well-being by supporting activities that are consistent with conservation goals. The no-go zone comprises nine rocky islets, where access to all extractive and recreational activities is prohibited within a 100 m radius of each islet; total area of the no-go zones is approximately 0.6 km^2. The no-take zone prohibits all types of fishing, except for commercial hand harvesting of the goose barnacle in the intertidal area of the nearshore cliffs (*Pollicipes pollicipes*, not assessed in this study). The no-take zone covers four delimited areas, accounting for approximately 23.5 km^2, or 8.1% of the MPA (Gonçalves et al., 2021). Finally, the partially protected zone covers the largest portion of the MPA, accounting for approximately 266.0 km^2 (91.7% of the MPA), but has few limitations on top of the national fishing law for mainland Portugal. The disparities between the partially protected zone and the national law for commercial fishing encompass: i) clam dredging—prohibited in the partially protected zone, yet allowed beyond the MPA; ii) bottom longlining—banned up to 0.9 km (0.5 NM) from shore within the partially protected zone, but allowed without limitations beyond the MPA, iii) shore angling—temporal closures for three commercial species (*Diplodus sargus*, *Diplodus vulgaris*, *Labrus bergylta*) in the partially protected zone, but not in the areas beyond the MPA. In analogy to the MPA's outside, in the partially protected zone, other commercial gears may fish without spatial limitations (commercial angling and traps), or approximately 0.5 km (¼ NM) from shore (purse seines, gillnets, trammel nets) and 0.9 km (0.5 NM) from shore (pots). Both inside the MPA and outside up to 1.85 km (1 NM) from coast, most commercial gears (gillnets, trammel nets, traps and pots) can only be used by vessels with a length overall (LOA) of up to 9 m. The restrictions to commercial fishing, as well as recreational fishing, are summarized in Table A.1 of Appendix A. Furthermore, a specific license is needed to operate within the partially protected zone of the MPA, which was only assigned to vessels registered in the neighbouring harbours in 2010, with the aim of limiting the fishing effort within the MPA (PCM, 2011). When the MPA was implemented in 2011, 287 professional fishing vessels held this license, which is only transferable to local residents or the direct descendants of the vessel owner, and otherwise expires upon the cessation of fishing activity, while new licenses are not granted.

This research incorporated ten fishing harbours, for which fishery landings were available. Based on their location inside the MPA, six of these harbours were classified as within the MPA ('inside'), including major harbours: Sagres; minor harbours: Vila Nova de Milfontes, Zambujeira do Mar, Azenha do Mar; and very small harbours: Almograve and Arrifana. To allow for a control-impact comparison and to control for environmental variability, we chose control harbours located outside and at both extremities of the MPA, including two major harbours: Setúbal (in the north, distance to MPA ca 70 km) and Portimão (in the South, distance to MPA ca 30 km) (Fig. 1). Two remaining harbours are located at the edges of the MPA: the Sines harbour in the north is approximately 6 km away from the MPA, and the Lagos harbour in the south about 15 km. In 2021, face-to-face interviews were held with owners or captains of commercial fishing vessels (hereafter called 'fishers') at the fishing docks of these harbours. Fishers were approached in the order of their arrival from regular fishing operations and interviewed once they agreed to participate. Most fishers (91.7%) belonging to the local fleet (focus of this study: LOA ≤9 m, see section 2.2) interviewed at the Sines harbour (n = 12, corresponding to 20 % of the harbour's active local fleet in 2020) claimed to fish inside the MPA, with 75 % doing so with a high frequency (authors' unpublished results). In contrast, at the Lagos harbour, most fishers (81.4 %) operating local vessels (n = 16, corresponding to 33.3 % of the harbour's active local fleet in 2020) stated that they never fish inside the MPA. The remaining 18.6 % of fishers interviewed at Lagos claimed to fish inside the MPA occasionally or rarely. Based on this information, Sines was classified as an 'inside' harbour, similar to a previous report by Castro et al. (2020) that evaluated the PNSACV effects in small-scale fisheries, while Lagos as an 'outside' harbour. Table A.2 of Appendix A provides information on harbours sizes, based on mean annual landings and number of active local vessels using them.

The study area consists of rocky and soft-bottom habitats both inside and outside the MPA, and depths below 60 m. In the no-take zone of the MPA, rocky reefs and soft bottoms are roughly equally represented (11.5 km^2 vs. 12.0 km^2), while soft bottom substrates (174.1 km^2) dominate over rock (91.5 km^2) in the partial protection (Gonçalves et al., 2021). The no-go zone is situated at depths below 20 m, and mostly consists of rocky bottom (0.4 km^2 out of 0.6 km^2). In the control area, marine habitats are not mapped in as much detail as those inside the MPA, but rocky reefs are more abundant nearby the south control harbours (Lagos and Portimão), whereas soft bottom substrates dominate over rocky bottom in the north control area (Setúbal) (https://emodnet.ec.europa.eu). The Setúbal harbour is situated within a large estuary (Sado).

2.2. Data collection

The landings data utilized in this study are official landings originating from fish auctions, where catches have to be declared and sold (Pita and Gaspar, 2020). These auctions occur at the landing docks of major and some minor (e.g., Villa Nova de Milfontes, Zambujeira do Mar) harbours. The landed catch is weighted per species or groups by auction personnel. Directorate-General for Natural Resources, Safety and Maritime Services (DGRM), the national competent entity, collects the landings data from the national auction network. The obtained landings data included details such as year, vessel code, vessel length overall (LOA) interval (LOA: ≤7 m, 7–9 m, 9–15 m, >15 m), gear type (trawl, purse-seine, multi-gear), harbour of landings, taxa, landed quantity in kg, average off-vessel price in EUR, and the number of landing days of each vessel per year and harbour. The landings dataset covered 10 years before (2001–2010) and 10 years after (2011–2020) the MPA establishment. The multi-gear category includes vessels licensed for various gears, the most common being traps and pots, gillnets, trammel nets, and less common longlines (Alexandre, 2019; Szynaka et al., 2021, 2022; Henriques et al., 2023). Traps and pots capture mainly *Octopus vulgaris*, but also *Conger conger* and *Muraena helena*, while nets and longlines catch a variety of demersal and benthic species (Alexandre, 2019; Szynaka et al., 2021). While traps, pots and longlines represent highly selective fishing gear that demonstrate low impact on habitats and species, gillnets and trammel nets are less selective and more impactful (Horta e Costa et al., 2016). However, details on fishing operations and gear are currently not discriminated in the landings dataset but reported under a single–multi-gear–category. Multi-gear vessels alternate their gear between fishing trips and seasons, and few utilize two gear types (gillnets and trammel nets) within a single trip (Szynaka et al., 2021, 2022). During the interviews held at the fishing docks in 2021 (see section 2.1), 54.7 % of local fishers interviewed inside the MPA (total interviewed n = 42, active local fleet comprised 145 vessels in 2020) and 41.0 % outside (total interviewed 39, local fleet 262 vessels) indicated that traps and pots with octopus as the main target species were their main fishing gear. Concurrently, only 35.7 % inside reported that their primary gear were nets (gillnets and trammel nets), compared to 66.7 % outside, suggesting potential differences in the gear composition, its selectivity and impact between inside and outside the MPA (author's unpublished data).

2.3. Data analysis

As the purpose of this study was to assess the protection effects on fisheries, for the harbours inside the MPA, only landings from licensed vessels (i.e. allowed to fish in the partially protected zone) were considered. For all harbours, we retained landings only for vessels using multi-gear and up to 9 m LOA, as these are the vessels constituting local fishing fleet allowed to operate inside the MPA and in the outside control area (see section 2.1). Such an approach allowed for exclusion of fishing activity not authorized within the MPA from landings classified as inside. It is anticipated that fishers from the vicinity who were granted privileges for the MPA through licenses would preferentially fish there, and that multi-gear vessels up to 9 m LOA would utilize the nearshore area (up to 1 NM), from which larger vessels are legally excluded. Furthermore, vessel's size is a limiting factor of the travel distance and time spend at sea, and smaller vessels show more dependence on coastal waters (Guyader et al., 2013; Damasio et al., 2016). The fishing grounds of local vessels operating from the harbours of the studied MPA are typically situated in the harbour vicinity, inside the MPA (Monteiro et al., 2020). These vessels typically demonstrate fidelity to their home harbour and established fishing grounds. We excluded trawling, as it is only authorized further offshore than 11.1 km (6 NM) according to the national fisheries law, and purse-seining, although allowed inside the partially protected zone of the MPA, it targets small pelagic fish (*Sardina pilchardus*, *Trachurus trachurus*, *Scomber colias*) not aimed for protection within this coastal MPA (MADRP, 2000; PCM, 2011; Castro, N. et al., 2021).

For the harbours outside the MPA, to maintain consistency in the number of vessels between temporal periods, only the vessels that fished in the outside area during the after-MPA period were selected. Therefore, we monitored the same fishing vessels throughout the study period, which also contributed to reduce possible variations in fishing operations among distinct vessels. Portuguese fisheries regulations define legal limits on the maximum number of fishing gears per vessel, and vessel size limits the maximum number of gears that fishers can manage in a fishing operation (DGRM, 2023a). Small vessels are limited to few hours at sea, necessitating daily landings of their catch. Thus, fishing operations (if at night or day, average fishing hours, hauling times, etc.) have remained typically stable in the study area and over time for each fishing gear.

As our goal was to infer about potential protection effects on the local fishing fleet and the species it harvests within the MPA, we retained data for 54 demersal and benthic taxa landed in the MPA, that had a previous record in the MPA (Horta e Costa et al., 2018; Castro, J. J. et al., 2021), and/or habitat and depth preferences consistent with those within the MPA (i.e., depth <60 m). These are the species that are most expected to benefit from protection measures. For the same reason, we excluded pelagic species from the analysis because: i) these species are caught in schools and may bias results, ii) protection measures are less likely to benefit this group, due to their mobility and low association with small coastal areas (Claudet et al., 2010, Horta e Costa et al., 2013a). Additionally, three species–*Sepia officinalis*, *Diplodus annularis* and *Myliobatis aquila*–were excluded because of their disproportionately higher landings outside than inside the MPA, likely due to the influence of the Sado estuary, located in the Setúbal area, where high landings occurred. This estuary serves as spawning and nursery grounds for *S. officinalis* and *D. bellottii* (which was likely mislabelled as *D. annularis* in the landings data, a species not common in this area) (Neves et al., 2008, 2009; Vinagre et al., 2010). The ray species *M. aquila* also exhibits an affinity to estuarine habitats (Compagno, 1986; La Mesa et al., 2017). This approach to species scope ensured that the same community was compared between inside and outside the MPA across temporal periods, contributing for indirect control of the habitat effect.

We analysed the landings of the local fleet for the entire coastal species community (of the 54 retained taxa, according to the criteria explained above), followed by a focus analysis on selected groups and particular species. Among these, we chose the species with the highest landings: (1) *Octopus vulgaris* (Common octopus), and (2) *Conger conger* (European conger), and the most commercially valuable taxa: (3) genus *Mullus* (Red mullets), (4) genus *Dicentrarchus* (Seabasses), (5) family Soleidae (Soles, the main landed taxa being *Microchirus* spp. and *Pegusa lascaris*), and (6) family Sparidae (Sparids). Within the Sparidae family, we were interested in the protection effect on seabreams from the genus (6.1) *Diplodus* (main landed species being *D. sargus*, but not reported separately for the Sagres harbour until 2005), (6.2) *Pagrus* and (6.3) *Pagellus*, and (6.4) *Sparus aurata* (Gilt-head bream). The species (6.5) *Diplodus vulgaris* (Two-banded sea bream) was analysed separately due to its low market price, contrary to the other species in its genus, as well as its importance in landings and reporting separate from the genus (Fig. 2), but also because of its previously reported potential negative protection effects in the MPA area (Belackova et al., 2023). We also included (7) Rajidae (Skates), which are important in landings and extremely vulnerable to overfishing due to their life-history traits (slow growth, late maturity, and low fecundity) (Dulvy et al., 2000; Stevens et al., 2000), and (8) *Muraena helena* (Mediterranean moray), which may benefit from the no-take zones situated in rocky reefs, given its territorial behaviour and strong association to rocky habitats (Böhlke et al., 1989; Matić-Skoko et al., 2011). Each group or species can be found in one or both of the MPA's habitats: rocky reefs (*C. conger*, *M. helena*), soft bottom substrates (Rajidae, Soleidae), or both rock and soft bottom (genus *Diplodus*, *Pagrus* spp., *D. labrax*, *S. aurata*, *O. vulgaris*, *Pagellus* spp., *Mullus* spp.) (Mangold, 1983; Saldanha, 1997; Froese and Pauly, 2024). The full list of taxa and taxa belonging to each group can be consulted in Table A.4 of Appendix A.

For the purpose of our analysis, we used the division of the landings by protection status: inside the MPA (protected) and outside (unprotected), as depicted in Fig. 1 (see section 2.1 on harbours splitting and additional criteria applied to the data in section 2.3). In each year and protection status category, we calculated the landings per unit effort (LPUE) by dividing each vessel's landings by its annual landing days. The landing days represent the most consistent variable available for the entire landings time series and were used as a fishing effort estimate in previous research that analysed official landings (Sonderblohm et al., 2014; Batista et al., 2015; Baptista et al., 2016; Leitão et al., 2016; Bueno-Pardo et al., 2020), although they do not account for the efficiency of individual fishing operations (including factors such as the number and type of gear deployed, soak times, and technological advancements) (McCluskey and Lewison, 2008; Eigaard et al., 2014, De la Puente et al., 2020). We then calculated the mean annual LPUE (± standard error, in kg vessel^{-1} landing day^{-1}) of all vessels by protection status. We differentiated between two time periods: before the MPA implementation, from 2001 to 2010, and after, from 2011 to 2020. This allowed us to create four LPUE time series: inside-before, inside-after, outside-before, and outside-after.

We aimed to determine whether improved LPUE due to the MPA resulted in economic benefits in the form of increased income per unit of effort (IPUE). We also kept the results for prices, to help us understand the influence of price changes on IPUE, to try to disentangle MPA from external effects. For this purpose, we performed price inflation adjustments, utilizing the Consumer Price Index (CPI) of unprocessed food between 2001 and 2020 (INE, 2021). We adjusted the prices for inflation by recounting the CPI to CPI in 2020 ($CPI_{Yn,2020}$), the last year of our analysis, which served to recount all prices to prices in 2020 based on Equation (1). We then calculated real (i.e. inflation adjusted) IPUE of a single vessel as a product of vessel's LPUE and real prices (Equation (2)). We also calculated real weighted mean prices (EUR_{2020} kg^{-1}) using real prices and landings quantity as weights. We then calculated the real mean annual IPUE (± standard error, in EUR_{2020} vessel^{-1} landing

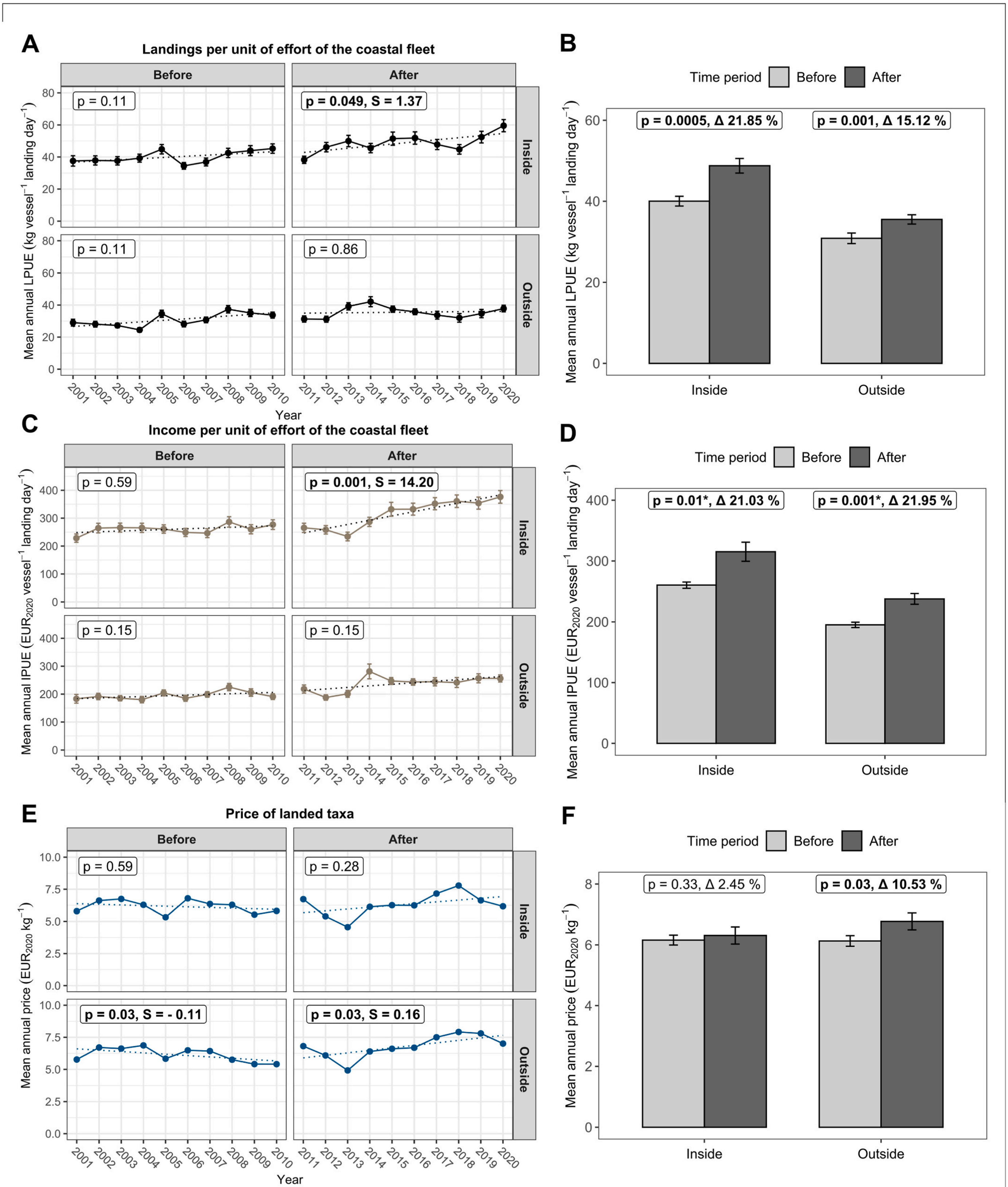

Fig. 2. Trends and barplots of mean annual landings per unit of effort (LPUE), income per unit of effort (IPUE) and prices of landed taxa. A, B) LPUE, C, D) IPUE, and E, F) Prices. Protection status: Inside or Outside the Parque Natural Sudoeste Alentejano e Costa Vicentina Marine Protected Area (PNSACV MPA). Prices and IPUE were adjusted for inflation and expressed in EUR of the year 2020 (EUR_{2020}). For significant results of the Mann-Kendall trend test ($p < 0.05$, in bold), we expressed the Sen's slope estimate (S). Dotted trend lines indicate directions of the trends based on linear regression. In barplots, for significant results from Student's t-test or Wilcoxon test, we also show the % change (△ for increase) between before and after the MPA implementation. When Wilcoxon test applied, we show *, otherwise we used Student's t-test. Error bars represent standard errors.

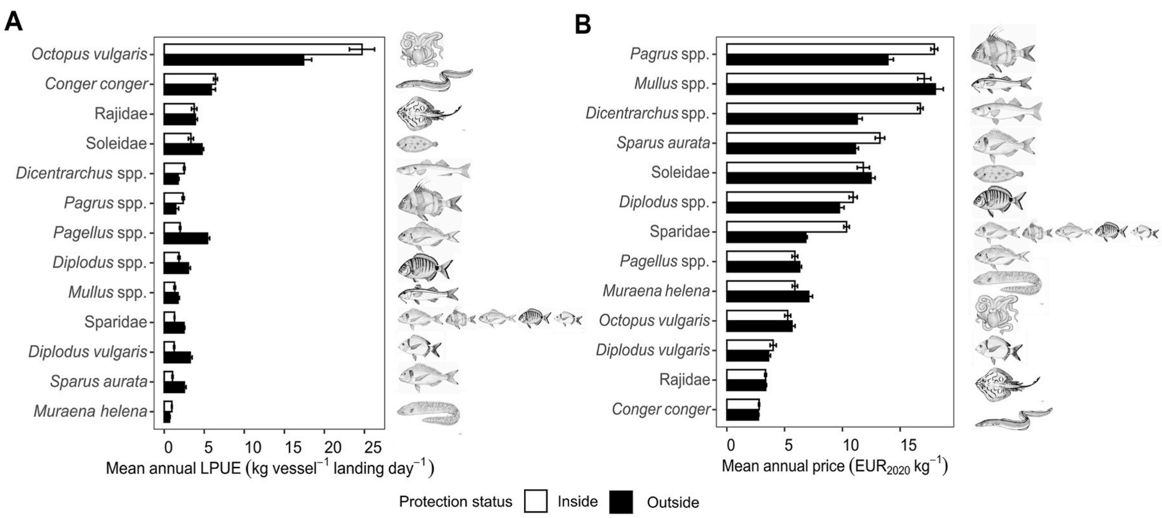

Fig. 3. Barplots showing the importance of the groups and species selected for the evaluation of protection effects of Parque Natural Sudoeste Alentejano e Costa Vicentina Marine Protected Area (PNSACV MPA) during the study period (2001–2020): Mean annual A) landings per unit of effort (LPUE) and B) prices. Prices were adjusted for inflation and expressed in EUR of the year 2020 (EUR_{2020}). Protection status: Inside or Outside the MPA. Error bars stand for standard errors. The taxa reported in the official landings included in each group can be consulted in Table A.4 of Appendix A. Illustrations reproduced with permission, source: Food and Agriculture Organization of the United Nations.

day^{-1}) and real mean annual prices ($\pm$ standard error, in EUR_{2020} kg^{-1}) of all vessels, similar to the calculation for LPUE.

$$\text{Price}_{\text{Real}}\ (EUR_{2020}\ kg^{-1}) = \text{Price}\ (EUR_{Yn}\ kg^{-1}) / CPI_{Yn,\ 2020} \quad (1)$$

$$\text{IPUE}_{\text{Real}}\ (EUR_{2020}\ vessel^{-1}\ landing\ day^{-1}) = \text{LPUE}\ (kg\ vessel^{-1}\ landing\ day^{-1}) \times \text{Price}_{\text{Real}}\ (EUR_{2020}\ kg^{-1}) \quad (2)$$

We utilized the Mann-Kendall trend test to examine significant time series trends in mean annual LPUE, mean annual IPUE, and prices (Mann, 1945; Kendall, 1975; Venables and Ripley, 2002; Chandler and Scott, 2011). When a significant trend was detected, we calculated the Sen's trend estimate (S) (Mann, 1945; Sen, 1968; Chandler and Scott, 2011; Fortibuoni et al., 2017). Prior to conducting the Mann-Kendall trend test, we checked autocorrelation estimates plots and confirmed that there was no autocorrelation in each time series. To aid in visualizing potential trends, we displayed linear regression lines along each time series (Teixeira and Cabral, 2009; Fogliarini et al., 2021). We also conducted two-sample tests to examine differences in mean annual LPUE, mean annual IPUE and prices between the before and after periods by protection status (inside vs. outside). We used the parametric Student's t-test or non-parametric Wilcoxon test, depending on the result of the normality test (Shapiro-Wilk test) and homogeneity of variances (Bartlett's test) (Montgomery and Runger, 2003; Dytham, 2011; Gardener, 2017); the results for the normality and homogeneity are available in Table A.3 of Appendix A. The visual representation for the two-sample tests consisted of barplots that displayed the mean annual LPUE, IPUE, and price in each period and protection status, and their respective standard errors. We repeated the statistical analysis for the local fleet landings of the entire coastal species community (species included are explained above in 2.3), and of particular species and groups of species (also detailed above in 2.3). For simplicity, for particular species and groups of species, we only presented the IPUE and price results of the two-sample tests, i.e., by period and protection status, and not the trend test. Data analysis was performed using R version 4.2.2 (R Core Team, 2022). Mean LPUE and price of the taxa and groups of taxa inside and outside the MPA by period (before, after) can be consulted in Tables A.4 and A.5 of Appendix A.

Table 1
Results of the Mann-Kendall trend test for mean annual landings per unit of effort (LPUE in kg vessel^{-1} landing day^{-1}) of selected groups and species. For significant results ($p < 0.05$, in bold), we expressed the Sen's slope estimate (S). Protection status: Inside or Outside of Parque Natural Sudoeste Alentejano e Costa Vicentina Marine Protected Area (PNSACV MPA), Period: Before and After the MPA implementation.

	Mann-Kendall trend test p-value, Sen' s trend estimate (S)			
	Inside		Outside	
	Before	After	Before	After
Conger conger	0.86	0.15	0.15	**0.01 (S = - 0.33)**
Mullus spp.	0.11	0.11	0.15	0.86
Octopus vulgaris	0.28	0.21	0.72	0.59
Rajidae	**0.049 (S = 0.13)**	**0.049 (S = 0.50)**	**0.007 (S = 0.22)**	0.47
Soleidae	0.86	**0.03 (S = 0.19)**	0.59	0.72
Dicentrarchus spp.	0.15	0.28	0.37	0.28
Muraena helena	0.86	0.37	0.21	0.11
Sparidae	0.28	1.00	0.28	0.72
- *Diplodus* spp.	0.07	0.47	0.11	0.21
- *Diplodus vulgaris*	0.07	**0.01 (S = - 0.04)**	0.11	0.28
- *Pagellus* spp.	0.72	1.00	0.47	0.21
- *Pagrus* spp.	0.21	0.72	0.21	0.07
- *Sparus aurata*	0.72	0.15	0.47	0.37

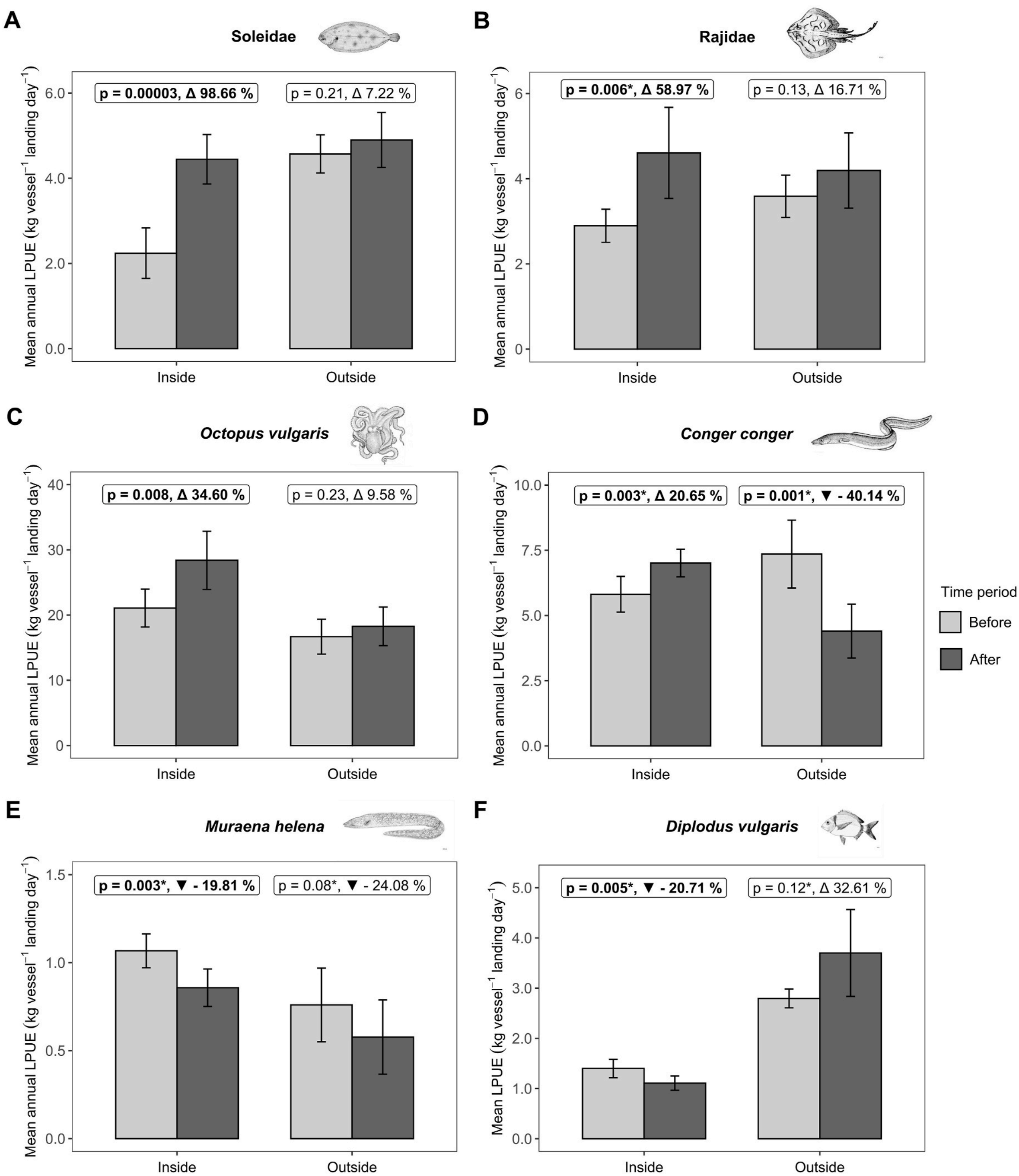

Fig. 4. Barplots of mean annual landings per unit of effort (LPUE) inside and outside the Parque Natural Sudoeste Alentejano e Costa Vicentina Marine Protected Areas (PNSACV MPA). A) Soleidae, B) Rajidae, C) *Octopus vulgaris,* D) *Conger conger*, E) *Muraena helena*, F) *Diplodus vulgaris*. Period: Before and After the MPA implementation. Groups and species with significant results ($p < 0.05$, in bold) are displayed, based on results from Student's t-test or Wilcoxon test. For significant results we also show the % increase (△) or decrease (▾) between before and after the MPA implementation. When Wilcoxon test applied, we show *, otherwise we used Student's t-test. Error bars represent standard errors. Illustrations reproduced with permission, source: Food and Agriculture Organization of the United Nations.

3. Results

3.1. Local fleet landings

The mean annual LPUE of the local fleet exhibited a significant positive trend inside the MPA in the after period (i.e. from 2011 to 2020), with a mean annual increase of 1.37 kg vessel^{-1} landing day^{-1} (Mann-Kendall test, $p = 0.049$; Fig. 2). Similarly, the mean annual IPUE of the local fleet showed a significant positive trend inside the MPA after implementation (Mann-Kendall test, $p = 0.001$, $S = 14.20$; Fig. 2). In contrast, the LPUE and IPUE of the local fleet did not show any significant trends inside the MPA before implementation (i.e. from 2001 to 2010) or outside the MPA in both the before and after implementation periods (Mann-Kendall test, $p \geq 0.05$; Fig. 2).

According to the two-sample tests comparing periods, both the mean annual LPUE and IPUE of the local fleet increased significantly not only inside but also outside the MPA between the before and after implementation periods (LPUE: Student's t-test, IPUE: Wilcoxon test, $p < 0.05$; Fig. 2). Nevertheless, this increase in mean annual LPUE of the local fleet was relatively greater inside than outside the MPA,

Table 2
Results of two-sample tests for mean annual landings per unit of effort (LPUE, in kg vessel^{-1} landing day^{-1}), income per unit of effort (IPUE, in EUR$_{2020}$ vessel^{-1} landing day^{-1}) and price (in EUR$_{2020}$ kg^{-1}) of selected groups and species. Price was adjusted for inflation and expressed in EUR of the year 2020 (EUR$_{2020}$). Protection status: Inside or Outside the Parque Natural Sudoeste Alentejano e Costa Vicentina Marine Protected Area (PNSACV MPA). For significant results ($p < 0.05$) from Student's t-test or Wilcoxon test, we indicate the significant increase (△) or decrease (▼) between before and after the MPA implementation. Results from Wilcoxon test marked with *, otherwise we used Student's t-test.

Two-sample tests results (p-value) for mean annual values of LPUE, IPUE and price (p-values)

	LPUE		IPUE		Price	
	Inside	Outside	Inside	Outside	Inside	Outside
Conger conger	△ 0.003 *	▼ 0.001 *	△ 0.02	▼ 0.001	0.19	0.25
Mullus spp.	△ 0.0001	△ 0.00002	△ 0.0005*	△ 0.00003	▼ 0.03	▼ 0.03 *
Octopus vulgaris	△ 0.008	0.23	△ 0.003*	△ 0.003	△ 0.04	△ 0.01
Rajidae	△ 0.006 *	0.13	△ 0.01*	0.32	▼ 0.007	▼ 0.002
Soleidae	△ 0.00003	0.21	△ 0.00009	0.13	▼0.0002 *	▼ 0.005 *
Dicentrarchus spp.	0.13	0.08 *	0.09	0.37 *	0.07 *	0.18 *
Muraena helena	▼ 0.003 *	0.08 *	0.17	0.43	△ 0.00004	△ 0.0002
Sparidae	0.33	△ 0.001	0.11	△ 0.04 *	▼ 0.0003	0.26
- *Diplodus* spp.	0.09	0.14	0.37	0.33	▼ 0.000004	▼ 0.000005 *
- *Diplodus vulgaris*	▼ 0.005 *	0.12 *	▼ 0.00002	▼ 0.09	▼ 0.0000005	▼ 0.000006
- *Pagellus* spp.	0.37 *	0.14 *	▼ 0.0008 *	0.32 *	▼ 0.00007	0.34
- *Pagrus* spp.	0.35	0.10 *	0.43	0.10 *	▼ 0.04 *	0.30
- *Sparus aurata*	△ 0.002	△ 0.02	0.07	△ 0.003	▼ 0.009 *	0.49

corresponding to an increase of 21.85 % vs. 15.12 %, respectively (Fig. 2). On the other hand, the increase in mean annual IPUE of the local fleet was similar inside and outside the MPA (21.03 % vs. 21.95 %) because of a significant rise in mean annual prices outside (but not inside) between the before and after implementation periods, which improved the IPUE outside the MPA (Fig. 2).

3.2. Landings of groups and species

Regarding groups and species selected for the analysis, *Octopus vulgaris* had the highest mean annual LPUE both inside and outside the MPA (Fig. 3). The second highest LPUE belonged to *Conger conger*, followed by the families Soleidae and Rajidae both inside and outside, and *Pagellus* spp. outside the MPA (Fig. 3). Both inside and outside the MPA, the most valued taxa included *Pagrus* spp., *Mullus* spp., *Dicentrarchus* spp., *Sparus aurata*, Soleidae and *Diplodus* spp. (Fig. 3).

When analysing trends in groups and species, we found that two families associated with soft bottom habitats–Soleidae and Rajidae–showed positive LPUE trends inside the MPA after the implementation (i.e. inside-after), contrasting with the trends inside the MPA before the implementation (i.e. inside-before) and outside the MPA after the implementation (i.e. outside-after). Concretely, Soleidae demonstrated a significant positive trend in mean annual LPUE inside-after (Mann-Kendall test, $p = 0.03$, $S = 0.19$), but not inside-before or outside (Mann-Kendall test, $p \geq 0.05$; Table 1). In line with this trend, Soleidae increased their mean annual LPUE significantly inside the MPA between the before and after implementation periods, but not outside of it (Student's t-test, inside: $p = 0.00003$, outside: $p = 0.21$; Fig. 4). The family Rajidae showed a significant positive trend in its mean annual LPUE in the before period both inside and outside the MPA (Mann-Kendall test, $p < 0.05$), which continued as positive inside the MPA during the after period, but not outside (Mann-Kendall test, inside-after: $p = 0.049$, $S = 0.50$, outside-after: $p = 0.47$; Table 1). As a result, the mean annual LPUE of Rajidae increased significantly between the before and after periods inside the MPA, but not outside (Wilcoxon test, inside: $p = 0.006$; Student's t-test, outside: $p = 0.13$; Fig. 4). In contrast, the highly valued *Mullus* spp. showed a significant increase in its mean annual LPUE both inside and outside the MPA between the before and after periods (Student's t-test, inside: $p = 0.0001$, outside: $p = 0.00002$; Table 2).

Representing both rocky and soft-bottom habitats, the most landed species, *O. vulgaris*, increased its mean annual LPUE significantly inside the MPA between the before and after periods, whereas no change occurred outside (Student's t-test, inside: $p = 0.008$, outside: $p = 0.23$; Fig. 4). Despite that, trends in the LPUE of *O. vulgaris* were not significant inside-after (Mann-Kendall test, $p = 0.21$). Another species, associated with rocky reefs, *C. conger* demonstrated a significant negative trend in its mean annual LPUE outside-after (Mann-Kendall test, $p = 0.01$, $S = -0.33$), but not inside the MPA both before and after the implementation, nor outside-before (Mann-Kendall test, $p \geq 0.05$; Table 1). There was a significant decrease in the mean annual LPUE of *C. conger* outside between the before and after periods, which contrasted with its significant increase inside the MPA (Wilcoxon test, inside: $p = 0.003$, outside: $p = 0.001$; Table 2 and Fig. 4). Contrary to *C. conger*, another rocky reef species, *M. helena*, showed a decreasing pattern in its mean annual LPUE between the before and after periods both inside and outside the MPA, which was only significant inside (Wilcoxon test, inside: $p = 0.003$, outside: $p = 0.08$; Fig. 4).

We also found significant results in the mean annual LPUE of the Sparidae family that showed a significant increase outside the MPA between the before and after periods, but not inside (Student's t-test, inside: $p = 0.33$, outside: $p = 0.001$; Table 2). The overall result in the mean annual LPUE of the Sparidae family (increase outside) was influenced by the contrasting significant results in the LPUE of *Sparus aurata* and *Diplodus vulgaris* (Table 2). The highly valued *S. aurata* demonstrated an increase in its mean annual LPUE both inside and outside the MPA between the before and after periods (Wilcoxon test, $p < 0.05$; Table 2). Belonging to the same family, the rather low valued *D. vulgaris* displayed a significant negative trend in the mean annual LPUE inside-after, which resulted in its decrease inside between the before and after periods (Mann-Kendall test, $p = 0.01$, $S = -0.04$; Wilcoxon test, $p = 0.005$; Tables 1 and 2). No other significant results were found in the mean annual LPUE of the remaining groups and species selected for the analysis (Tables 1 and 2).

Based on two-sample tests, the results for mean annual IPUE were validated to be the same as for LPUE for Soleidae, Rajidae, *C. conger*, *Mullus* spp., and Sparidae (Table 2). For Soleidae and Rajidae, this meant a significant increase in mean annual IPUE between the before and after periods inside, but not outside the MPA (Table 2). For *C. conger*, the mean annual IPUE showed a significant increase inside the MPA between the before and after periods, contrasting with a significant decrease outside (Table 2). *Mullus* spp. increased its IPUE both inside and outside the MPA between the before and after periods, whereas Sparidae showed an increase in mean annul IPUE only outside, but not inside (Table 2). In the cases where results in mean annual IPUE and LPUE differed, this was a result of mean annual prices significantly evolving in an opposite way than the LPUE. This was the case of *M. helena*, *S. aurata* and *Pagellus* spp. inside, and *O. vulgaris* and *D. vulgaris* outside, as apparent in Table 2.

Mean annual prices of groups and species showed the same results inside as outside the MPA, except for Sparidae who experienced a significant drop in prices inside the MPA between the before and after periods (Student's t-test, $p = 0.0003$), but not outside (Student's t-test, $p = 0.26$; Table 2). This was due to the decrease in mean annual prices of all the five taxa from the Sparidae family inside the MPA (*Diplodus* spp., *D. vulgaris*, *Pagellus* spp., *Pagrus* spp., *S. aurata*), while only two taxa from this family (*Diplodus* spp. and *D. vulgaris*) decreased their mean annual prices outside (Table 2). Both inside and outside the MPA, we also witnessed a significant decrease in the mean annual prices of Soleidae, Rajidae and *Mullus* spp., contrasting with a significant increase in mean annual prices of *O. vulgaris* and *M. helena* (Table 2). Given the high importance of *O. vulgaris* on landings, a greater significant increase in its prices outside (23.9 %) compared to inside (18.8 %) had a major influence on the local fleet results at community level in both mean annual price and IPUE (see Fig. 2). A lower significant decrease in mean annual price of Soleidae outside (−15.1 %) than inside (−24.1 %) also contributed substantially to the results at the community level.

4. Discussion

Based on a Before-After-Control-Impact (BACI) design applied to long-term official landings, we found positive protection effects on harvested species and local fisheries associated with a coastal MPA. The local fleet landings and income per unit of effort (LPUE and IPUE) both showed significant positive trends inside the MPA after the implementation, in contrast with no such significant trends inside the MPA before the implementation or outside the MPA. Since trends were different inside and outside the MPA, we hypothesise that the no-take zones and license limitations in the partially protected zone play a key role in protecting the harvested demersal and benthic species, as the other conservation measures are less effective. Indeed, by excluding all types of fishing and other extractive activities, no-take zones are expected to deliver the highest conservation outcomes (Sciberras et al., 2015; Giakoumi et al., 2017; Sala et al., 2018; Ferreira et al., 2022), and export fish to adjacent waters (Goñi et al., 2011; Di Lorenzo et al., 2016; Hamilton et al., 2021). Previous research demonstrated positive protection effects of one of the MPA's no-take zone on the biomass of target species, as well as increased abundances of coastal species within both no-take and adjacent partially protected areas, based on fishery-independent surveys (experimental netting, stereo-baited video) and few years of data (Castro, J. J. et al., 2021; Belackova et al., 2023). Partially protected areas can enhance densities and biomass, particularly of target species, although to a lesser extent than in no-take zones, while yielding variable results depending on the fishing restrictions, and other factors such as age, size and connectivity (Sciberras et al., 2015; Giakoumi et al., 2017; Zupan et al., 2018; Ferreira et al., 2022). Partially protected areas provide more positive effects when limiting the numbers of less impactful fishing gears (e.g., traps and pots, lines, gillnets) and excluding highly impactful gears (e.g., trawling, trammel nets) (Sciberras et al., 2015; Zupan et al., 2018), and when implemented for an extended period of time (Ferreira et al., 2022). In our case, limitations to fishing gear are minimal in the MPA's partially protected zone compared to the open access area, but 10 years of controlled number of fishing licenses has likely helped to reduce fishing pressure (Horta e Costa et al., 2013, Giakoumi et al., 2017; Anderson et al., 2019; Wright, 2022). The number of fishing vessels holding a license for the MPA dropped by almost 30 % since the MPA implementation (287 licensed vessels in 2011 vs. 204 in 2020) as part of the licenses expired with activity cessation, while new licenses are not granted. MPAs that combine no-take zones with limited licensing have been demonstrated to improve fishery resources (Russ et al., 2004; Batista et al., 2015; Sousa et al., 2018; Hogg et al., 2019).

Outside the MPA, a significant increase in the local fleet's LPUE was also found in the after MPA period, but to a smaller magnitude than inside. This suggests that additional factors, independent of protection, affect the landings both inside and outside. Possible reasons include improvements in fisheries management that promote more sustainable exploitation of marine resources (Worm et al., 2009; Melnychuk et al., 2016; Hilborn et al., 2020), such as the national seasonal closures of Rajidae fisheries since 2011 (Mar, 2016), regional weekend closures of octopus fisheries between 2019 and 2020 (Mar - Gabinete da Ministra, 2019), and a delayed effect of the 2006 regulation of recreational fisheries (Diogo et al., 2020). Another possible reason is the spillover or larval export from the MPA to the surrounding areas (Di Lorenzo et al., 2020; Hamilton et al., 2021). On the other hand, increasing LPUE also reflects the overall decline in the number of fishing vessels, with smaller vessels exiting fisheries at a faster rate, while larger vessels with greater capacity become more important (Castro et al., 2020).

Our findings suggest that local fishers did not lose but, in the contrary, improved their income (IPUE) through an increase in LPUE within the MPA. However, a cost analysis needs to be conducted to determine whether net economic benefits or losses exist, as costs of fishing may have increased due to the establishment of the MPA (Mangi and Austen, 2008; Batista et al., 2011; Gall and Rodwell, 2016; Ban et al., 2019). A review conducted by Costello (2024) found no evidence of negative economic effects of MPAs on fisheries, contradicting previous concerns regarding the ability of MPAs' economic benefits to offset their associated costs (Jones, 2008; Mangi and Austen, 2008; Gall and Rodwell, 2016).

As another economic effect, we reveal that market prices followed a significant positive trend outside-after, but not inside-after, improving local fleet revenue (IPUE) outside the MPA. These results are opposite to those of the LPUE and suggest that prices may evolve in a more favourable manner in unprotected than protected areas, which can undermine the MPAs' relative profitability to fisheries and, contrary to the MPAs objectives, stimulate overfishing to compensate for lost profits. Multiple market factors influence prices on the supply side, such as: landed quantities, fish sizes and alternative sources (aquaculture), and on the demand side, number of buyers and consumer preferences (Pinnegar et al., 2006; Sjöberg, 2015). These factors should be independent from protection effects, except for fish abundance and size that tend to increase in successful MPAs (Lester et al., 2009, Horta e Costa et al., 2013; Giakoumi et al., 2017). As scarcity (decrease in supply) generally leads to price increases (Pinnegar et al., 2006), prices may rise in unprotected areas where landings tend to decline or stagnate compared to market demand, whereas an MPA-induced increase in landed quantities (increase in supply) would lower the price. Thus, the less favourable evolution of prices inside than outside the MPA may relate to the landings quantity increase within the MPA above the local market demand in some years, while reflecting a potential need for fishery products' certification/valorisation programs that would guarantee a premium price within the MPA (Garraud et al., 2023).

Regarding the groups or species analysis, *Octopus vulgaris*, *Conger conger*, Rajidae and Soleidae were the four taxa with the highest LPUE, and they followed the LPUE trends of the overall harvested community. Such results suggest that these taxa, associated with rocky reefs (*C. conger*), soft bottom sediments (Soleidae, Rajidae) or both habitats (*O. vulgaris*), benefit from the MPA and contribute to the overall trends. This aligns with previous research that heavily fished species are more prone to positive responses to protection (Lester et al., 2009, Horta e Costa et al., 2013; Giakoumi et al., 2017; Sørdalen et al., 2022). As one of the most impactful measures within this MPA seems to be the presence of no-take zones and that these are relatively small (each <6 km^2), positive protection outcomes were more expected in species with small home ranges (Abecasis et al., 2014, 2015; Pereira et al., 2017; Di Franco et al., 2018). Indeed, predators of greater mobility are not expected to be efficiently protected by the small no-take zones (Kramer and Chapman, 1999; Le Quesne and Codling, 2009; Weeks et al., 2017). From the analysed taxa, small home ranges have been previously evidenced for *O. vulgaris* (Mereu et al., 2015; Arechavala-Lopez et al., 2019) and *C. conger*, as well as for *Muraena helena* (Böhlke et al., 1989; Matić-Skoko et al., 2012; Pereira et al., 2017), and two species from the *Diplodus* family, *Diplodus sargus* and *D. vulgaris* (Abecasis et al., 2013; Belo et al., 2016; Di Franco et al., 2018).

Despite inshore-offshore reproduction related movements (Mangold, 1983), *O. vulgaris* is a sedentary and very fast-growing species (Boletzky and Mangold, 1973; Domain et al., 2000), and may have benefited from the MPA, same as reported from another Portuguese MPA (Luiz Saldanha Marine Park) that also combines no-take area and license limitations (Horta e Costa et al., 2013; Batista et al., 2015). Octopus has become the most important species in terms of landings and revenues for multi-gear Portuguese fisheries (Pita et al., 2015; INE, 2019; DGRM, 2023b), with landings increasing since the 1970s, but dropping below their regional historical mean between 2013 and 2018 (i.e. during the after MPA period of this study) (Sonderblohm et al., 2014; Pita et al., 2015, 2021; INE, 2019). Our results demonstrated that these years of poor *O. vulgaris* landings–likely related to poor species recruitment due to environmental conditions (Otero et al., 2008; Sonderblohm et al., 2014, 2017)–, were less severe inside the MPA, as we observed a significant rise in its LPUE within the MPA between the before and after periods, but not outside of it. Despite octopus opportunistic behaviour and common thrive even in fished locations associated to predators reduction (Aronson, 1991; Caddy and Rodhouse, 1998; Arechavala-Lopez et al., 2019), its short life span and non-overlapping generations make it sensitive to overfishing, which is more likely to occur when recruitment falls due to environmental conditions (Otero et al., 2008; Sonderblohm et al., 2014, 2017). Thus, during critical periods, it may even benefit more from MPAs that include no-take areas and partial protection (Grorud-Colvert et al., 2021).

Here, we also found potential positive outcomes in LPUE of *C. conger* (significant increase inside and decrease outside the MPA between the before and after periods), but negative in LPUE of *M. helena* (significant decrease inside the MPA between the before and after periods). Indirect negative effects of MPAs supposedly result from enhanced competition or predation, which in our case may relate to increased competition between the two nocturnal territorial predators *C. conger* and *M. helena*, who show overlapping habitats and diets (Micheli et al., 2004; Matić-Skoko et al., 2012, 2014; Pereira et al., 2017). Species with low mobility are susceptible to negative effects if competition arises, although they are also the group that displays most positive effects (Micheli et al., 2004; Cabral et al., 2019). Thus, the poor swimming abilities of *M. helena* that lead to its very small home ranges (0.19 ha), half of those of *C. conger*, may pose a competitive disadvantage (Böhlke et al., 1989; Pereira et al., 2017). Similarly, increased competition between *D. vulgaris* and *D. sargus* is one of the possible reasons for accentuated spatial splitting (Sala and Ballesteros, 1997; Gonçalves and Erzini, 1998; Osman and Mahmoud, 2009), which was one of the possible reasons for previously reported higher abundance of *D. vulgaris* in the partially protected area, instead of the dominance of *D. sargus* in one of this MPA's no-take areas (Belackova et al., 2023). The concentration of *D. vulgaris* in the fished (partially protected) area may explain its significant LPUE decrease within the MPA recorded here between the before and after periods. Although in this study, we did not find any protection effects in the LPUE of *Diplodus* spp., which includes *D. sargus* as the main species.

Another of the most landed groups of this study, the Soleidae family, displayed a potential positive response to the MPA. However, studies conducted in another Portuguese MPA (Luiz Saldanha Marine Park) suggest diverse responses to protection within the Soleidae family, by showing that whereas *Solea senegalensis* declined from before to after periods both inside and outside the protected area (Abecasis et al., 2014; Sousa et al., 2018; Martínez-Ramírez et al., 2021), *Microchirus azevia* responded to the no-take area positively and *Pegusa lascaris* preferred the area with the lowest protection (Sousa et al., 2018). In this study, *Microchirus* spp. and *P. lascaris* represented the most landed taxa from the Soleidae family, contributing to the group positive results within the MPA. Previous research found significant increases in abundance of *P. lascaris* in the partially protected zone of our study MPA, adjacent to one of the no-take zones (Castro et al., 2021). This suggests that this species may benefit from fished areas of the MPA due to predation reduction (Blyth-Skyrme et al., 2006; Link et al., 2015) and/or opportunistic feeding on fishery discards and benthic fauna that proliferates in disturbed habitats, as reported in other flatfish species (Jennings, 2001; Fanelli et al., 2022). Thus, our findings emphasize the importance of large multi-zone MPAs, such as PNSACV, where synergistic effects of different protection measures can occur, such as those from no-take zones and license limitations in the partial protection.

Our results also highlight the role of MPAs for vulnerable taxa such as Rajidae, who are highly susceptible to overfishing due to their large body size, slow growth, late maturity, and low fecundity (Dulvy et al., 2000; Stevens et al., 2000). Although species of Rajidae can travel over long distances and perform seasonal migrations, previous studies also suggest high seasonal site fidelity in most individuals (Hunter et al., 2006; Ellis et al., 2011; Simpson et al., 2020; Leeb et al., 2021). In line with our results, consistent with positive MPA effects, a significant increase in the abundance of *Raja undulata* was previously reported from the partially protected zone adjacent to the MPA's no-take zone (Castro, J. J. et al., 2021). Previous research in European waters demonstrated positive responses to spatial restrictions in some Rajidae species, due to both seasonal and full protection (Wiegand et al., 2011; Sousa et al., 2018), as well as trawling exclusions (Dimech et al., 2008; Rodríguez-Cabello et al., 2008; Serrano et al., 2011).

To enhance conservation and fisheries through MPA design, it is crucial to understand the MPAs' impacts on a wide range of taxa (Cabral et al., 2019). Official landings enable this, as they include extensive datasets on a vast array of taxa, covering species typically scarce in non-fishing scientific surveys (e.g., nocturnal species such as *C. conger* and *M. helena*). However, historical reporting may occur at higher than species level (Pauly et al., 2013), preventing long-term analyses for some species. For this reason, we could not assess the MPA effects on the fisheries of *D. sargus*, a valuable commercial species known to benefit from small MPAs and no-take zones in this and other European MPAs (Horta e Costa et al., 2013, Di Lorenzo et al., 2014; Aspillaga et al., 2016; Belo et al., 2016; Belackova et al., 2023). Certainly, using groups instead of species obscures results when species from the same group experience distinct changes.

There are various shortcomings associated with landings data, imprecise or absent georeferencing of fishing activity being a key issue preventing fishing grounds location (Blyth-Skyrme et al., 2006; Rodríguez-Cabello et al., 2008; Bradley et al., 2019). To address this constraint, we categorized the harbours of landings by protection status and implemented a set of criteria to maximize the exclusion of activity occurring outside the MPA from the landings classified as inside, by selecting spatially relevant fishing fleet (length overall $\leq$9 m, license for the MPA), gear category (multi-gear), and species (coastal species reported from the MPA). As the results obtained unveiled differences consistent with MPA effects, they lend credibility to our approach, which allowed for a spatial approximation of the fishing activity occurring within the MPA and the control area. Nevertheless, this approximation cannot fully preclude the possibility that landings occur within the MPA, while the fishing activity takes place outside. Although the applied methodology aimed to minimise such instances, the uncertainty regarding the spatial origin of landings remains a limitation, highlighting the necessity of fishing grounds reporting/tracking in small-scale fisheries to facilitate efficient management. Furthermore, due to the lack of georeferencing, relevant covariates such as habitat, depth or concrete gear cannot be linked to the utilized landings dataset. While the proportion of each habitat and/or gear type may differ among locations (inside and outside), with potential influence in catch composition, this factor is accounted for when comparing respective data and their trends among different periods (before and after). However, specific threats, other than fishing, may affect distinct habitats, and related fishery resources differently (e.g., climate change) (Trégarot et al., 2024).

As another limitation, same as in other countries, landings data in Portugal underestimate true catches due to the high proportion of bycatch and discards, but also the practice (illegal) to sell outside of the official fish auctions (Gonçalves et al., 2007; Batista et al., 2009; Leitão et al., 2014; Zejnilovic et al., 2023). Apart from that, some fishermen exceed the number of gears and soak-times allowed by law (Sonderblohm et al., 2014; Szynaka et al., 2022; Henriques et al., 2023), introducing unknown variables of fishing effort that may affect landings statistics. In our study, we assumed these limitations to have the same effect on landings inside as outside the MPA. We suggest that the quality of monitoring can be enhanced by providing more detail in the reporting (e.g. geolocation of fishing grounds, concrete gear, fishing effort, habitat, depth, sizes and abundances of individuals, species level). An obligatory vessel tracking system would facilitate geolocation and fishing effort estimates (Henriques et al., 2023), and their integration with ocean bottom maps for habitat and depth estimates. Finally, we recommend that future research focuses on the relationship between protection and market prices, as prices may undermine the profitability and acceptability of MPAs to fisheries.

Despite these limitations, we demonstrated that landings data can be effectively used to detect long-term outcomes of MPAs on fisheries and harvested species. To date, there has been a lack of studies employing long-term landings from the pre-MPA period, particularly when coupled with a BACI design, which belongs between the most effective methods for environmental impact assessment (Underwood, 1992; Smokorowski and Randall, 2017; Ban et al., 2019). When evaluating long-term protection effects, a balanced number of years before and after the MPA enhances the statistical power, but monitoring typically suffers from a lack of sufficient data from the pre-MPA period to match the post-MPA (Shaw and Mitchell-Olds, 1993; Smokorowski and Randall, 2017). In this study, we used 10 years of data preceding the MPA and 10 years following it, compared in a BACI design.

Although research has shown that the positive impacts of MPAs on human well-being outweigh the negative (Ban et al., 2019; Costello, 2024), negative perceptions of MPAs prevail between fishers, often reflecting their personal interests and concerns (Pita et al., 2011; Jentoft & Pascual-fernandez 2012; Ban et al., 2019). Ultimately, results originating in fisheries, e.g. based on official landings and costs of fishing operations, can serve as reliable evidence of the MPAs net economic benefits and engage fishers in the dialog about their effectiveness, helping to change fishers' preconceived images of MPAs (Jentoft & Pascual-fernandez 2012; Ban et al., 2019).

5. Conclusion

Our study demonstrates that analysing long-term landings from both pre- and post-MPA implementation using a BACI design can be a valuable approach for monitoring MPA effects on small-scale local fisheries and harvested species; however it is underutilized to date. Our conclusions regarding the protection effects rely on the contrasting results in LPUE inside and outside the MPA during the after implementation period, as well as between the periods before and after implementation within the MPA, but not outside of it. Thus, our monitoring approach and the fact that landings data, as in Portugal, are typically accessible at no cost may facilitate long-term monitoring of budget-limited MPAs. Furthermore, the taxonomic groups and species identified as benefiting from the studied MPA, their high importance for fisheries, and movement patterns can help define monitoring indicators of protection effects in other MPAs of the North-east Atlantic.

Declaration of generative AI and AI-assisted technologies in the writing process

During the preparation of this work the author(s) used Paperpal (https://paperpal.com) in order to refine language. After using this tool, the author(s) reviewed and edited the content as needed and take(s) full responsibility for the content of the publication.

CRediT authorship contribution statement

Adela Belackova: Writing – original draft, Visualization, Methodology, Formal analysis. **Luis Bentes:** Writing – review & editing, Methodology, Conceptualization. **Lene Buhl-Mortensen:** Writing – review & editing. **Bárbara Horta e Costa:** Writing – review & editing, Validation, Supervision, Methodology, Conceptualization. **Jorge Manuel dos Santos Gonçalves:** Writing – review & editing, Validation, Supervision, Methodology, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ocecoaman.2024.107502.

Data availability

Data used in this research can be obtained from Directorate-General for Natural Resources, Safety and Maritime Services (DGRM) by email ESTAT@dgrm.mm.gov.pt.

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