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Assessment of catches, landings and fishing effort as useful tools for MPA management

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ABSTRACT

Marine protected areas (MPAs) have been widely recognized as a tool to achieve both fisheries management and conservation goals. Simultaneously achieving these multiple goals is difficult due to conflicts between conservation (often long-term) and economic (often short-term) objectives. MPA implementation often includes additional control measures on fisheries (e.g. vessel size restrictions, gear exclusion, catch controls) that in the short-term may have impacts on local fishers' communities. Thus, monitoring fisheries catches before, during and after MPA implementation is essential to document changes in fisheries activities and to evaluate the impact of MPAs in fishers' communities. Remarkably, in contrast with standard fisheries-independent biological surveys, these data are rarely measured at appropriate spatial scales following MPA implementation. Here, the effects of MPA implementation on local fisheries are assessed in a temperate MPA (Arrábida Marine Park, Portugal), using fisheries monitoring methods combining spatial distribution of fishing effort, on-board observations and official landings statistics at scales appropriate to the Marine Park. Fisheries spatial distribution, fishing effort, on-board data collection and official landings registered for the same vessels over time were analysed between 2004 and 2010. The applicability and reliability of using landings statistics alone was tested (i.e. when no sampling data are available) and we conclude that landings data alone only allow the identification of general patterns. The combination of landings information (which is known to be unreliable in many coastal communities) with other methods, provides an effective tool to evaluate fisheries dynamics in response to MPA implementation. As resources for monitoring socio-ecological responses to MPAs are frequently scarce, the use of landings data calibrated with fisheries information (from vessels, gear distribution and on-board data) is a valuable tool applicable to many worldwide coastal small-scale fisheries.

1. Introduction

In recent decades coastal ecosystems have faced increasing anthropogenic pressures and the need to implement efficient management measures to reduce (or reverse) widespread declines in marine species, habitats and ecosystems has become widely recognised (Halpern et al., 2008; Lotze et al., 2006; Palumbi et al., 2008; Rice et al., 2012; Ruckelshaus et al., 2008). Globally, human communities and economies depend on marine resources to satisfy their needs on recreational, aesthetic, and economic dimensions, but also, importantly for food security and health. The way marine ecosystems have historically been managed has led to overfishing or depletion of many fish stocks (Halpern et al., 2008).

Ecosystem-based management (EBM) is recognized as the most appropriate approach to maintaining ecosystem structure and function (Claudet, 2011; Frid et al., 2005; Jennings and Kaiser, 1998; Murawski, 2000). However, its implementation is challenging, since ecosystems are complex and dynamic. Ecosystem assessment often requires an enormous amount of data using multidisciplinary approaches, which can be incompatible with limited availability of human and financial resources (Claudet, 2011). Marine Protected Areas (MPAs) became a mainstream management tool for

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a variety of management problems, with goals such as the sustainable management of exploited stocks and biodiversity conservation (Costanza et al., 1998; Field et al., 2006; Gerber et al., 2003; Halpern, 2003; Murawski, 2007; Roberts et al., 2001). MPAs can also contribute to the protection of essential habitats (e.g. spawning and nursery grounds) (Gell and Roberts, 2003; Nagelkerken et al., 2012), reduction in fishing mortality (Grüss et al., 2011; Mesnildrey et al., 2013) and provision of benefits to local fisheries by increasing catches in adjacent areas through spillover effects (Goñi et al., 2008; Goñi et al., 2010; Halpern et al., 2010; Harmelin-Vivien et al., 2008; Harrison et al., 2012; Roberts et al., 2001).

MPAs are often located in coastal areas where human activities (both extractive and non-extractive) are intense and ecosystem functions and services are thus under multiple pressures. They usually include some types of management measures aimed at regulating fisheries (e.g. no-take areas, gear exclusion, catch and/or effort control), that may have impacts on local fishing communities, particularly if not implemented together with other policies to compensate fishermen (Batista et al., 2011; Lester and Halpern, 2008; Mascia et al., 2010; Rees et al., 2013). Hence, MPA implementation is frequently accompanied by opposition from fishers (Christie, 2004; Rice et al., 2012), which can put the MPAs' objectives at risk (Pollnac et al., 2001). Compliance by users, including fishers, acceptance of MPA rules, cooperation in monitoring and management processes and adequate enforcement are critical components to MPA success (Claudet and Guidetti, 2010; Guidetti et al., 2008). When fishers recognize benefits from MPAs, they are more likely to accept the various regulations, leading to a greater chance of the MPAs meeting their goals (McClanahan et al., 2005, 2014). However, measurement of 'success' can be challenging due to the difficulty in establishing adequate monitoring processes that account for temporal and spatial variability in a large variety of possible responses. Measuring success is further compromised by a widespread lack of data from the period before MPA establishment (Horta e Costa et al., 2013c). Monitoring fisheries catch-per-unit-of-fishing effort before, during and after MPA implementation is essential not only to understanding the ecological responses of marine populations but also to evaluating the socio-economic impacts on fishers' communities. Interestingly, programs to monitor socio-economic responses to MPAs are rare compared with those directed at ecological responses, despite their interdependence. It has been suggested that monitoring fisheries (e.g. catches and effort) is costly to implement and to maintain during long-time frames, resulting in gaps in our knowledge of MPA impacts (both positive and negative) on fisheries (Mesnildrey et al., 2013). Coastal artisanal fisheries can be particularly difficult to accurately monitor due to the inherent diversity of artisanal fishing gears and techniques. Past assessments have been based primarily on the number of boats or the number of fishers, potentially obscuring evaluation of the actual fishing pressure on the resource (Garcia et al., 2008). For example, for static gears such as traps, both the number and the length of sets in a given area can vary greatly among individual fishers and boats. While important, these data are rarely available. Finally, the reliability of catch statistics from these artisanal fisheries is also questionable since it is usually based on officially registered landings which are rarely validated by systematic observer programs or calibrated with sampling at sea and during landing (Tzanatos et al., 2013). As a consequence, a high percentage of catches are not being accounted for in small scale fisheries (e.g. Batista et al., 2009). Yet, these are precisely the fisheries for which MPAs are promoted.

Adequate use of monitoring information may contribute to efficient management measures to promote the fulfilment of MPA goals, including the minimization of social and economic impacts and the improvement of benefits to local fishers' communities. In this context, efforts have been developed worldwide to improve the availability and accuracy of data for management. For example, the implementation of volunteer monitoring projects where fishers actively participate are becoming common around the world, and can play a role in achieving higher rates of MPA acceptance and success (Danielsen et al., 2005; Lloret et al., 2012). Moreover, some studies have been focused on the development of methods based on the integration of existing data from different sources with the aim of optimizing the utility of existing data for fisheries management (e.g. Tzanatos et al., 2013).

The aim of this study was to design, implement and test the efficiency of a fisheries monitoring scheme at a coastal temperate MPA (the Arrábida Marine Park, Portugal), that combines data from fine-scale spatial distribution of fishing effort, on-board sampling and official landings. Our approach was used to evaluate the impact of protection measures on local fisheries, by assessing how landings and revenues changed with MPA implementation. We also tested the accuracy of using landings statistics alone (i.e. a common scenario in MPA assessments in the context of artisanal fisheries). In addition, the influence of season and fishing effort in overall captures were characterized using a modelling approach to test the relevance of including these factors into fisheries monitoring methods, namely to develop models for the calibration of official landings statistics. Our results provide some of the first indications of the response of artisanal fisheries to MPAs and we suggest that the approach developed here can be applied to many coastal MPAs worldwide and also be adapted for other types of coastal fisheries monitoring and management programs.

2. Methods

2.1. Study area

The study was conducted in a coastal multiple-use MPA, the Arrábida Marine Park (AMP), adjacent to a terrestrial nature park (Fig. 1). The marine park extends along 38 km of coast (53 km^2), faces mainly south and is protected from prevailing north and northwest winds and waves, allowing for year-round human activities. Very nearshore subtidal habitats (approximately 0-10 m deep) are dominated by complex rocky reefs and outcrops resulting from the erosion of coastal calcareous cliffs. Adjacent to the reef, moving away from the cliffs, habitat is primarily sand, gradually giving way to fine sand and mud as depth increases. This region is an important biodiversity hotspot, with more than 1100 species identified (Henriques et al., 1999; Horta e Costa et al., 2013a). There is a small fishing town, Sesimbra, in the middle of the park, which has a long fishing tradition and is currently also a tourist area.

The AMP was created in 1998 but the management plan was only approved in 2005, defining eight zones subject to three protection levels: a fully-protected area (FPA: 4 km^2), four partially-protected areas (PPAs: 21 km^2) and three buffer areas (BAs: 28 km^2). The AMP objectives are broad: preserve marine biodiversity, recover habitats and promote scientific research, enhance environmental awareness and education, allow for sustainable nature oriented tourism, achieve sustainable development, and support regional economic and cultural activities, such as the traditional "hook and line" fishery (i.e. small-scale handlines, longlines, jigs). The AMP management plan defines limits and protection measures for various activities, in order to protect the local small-scale fisheries that have high local socio-economic importance. Thus, trawling, dredging, purse-seining and spearfishing are forbidden in the entire AMP and only vessels less than 7 m in length are allowed to operate in the park. There were 89, 79 and 77 vessels licensed to fish in the AMP in 2007, 2008 and 2009, respectively. These vessels are multigear; all have longline licenses and 35–40% of them also have licenses for traps and nets. Discarding in the AMP is also not

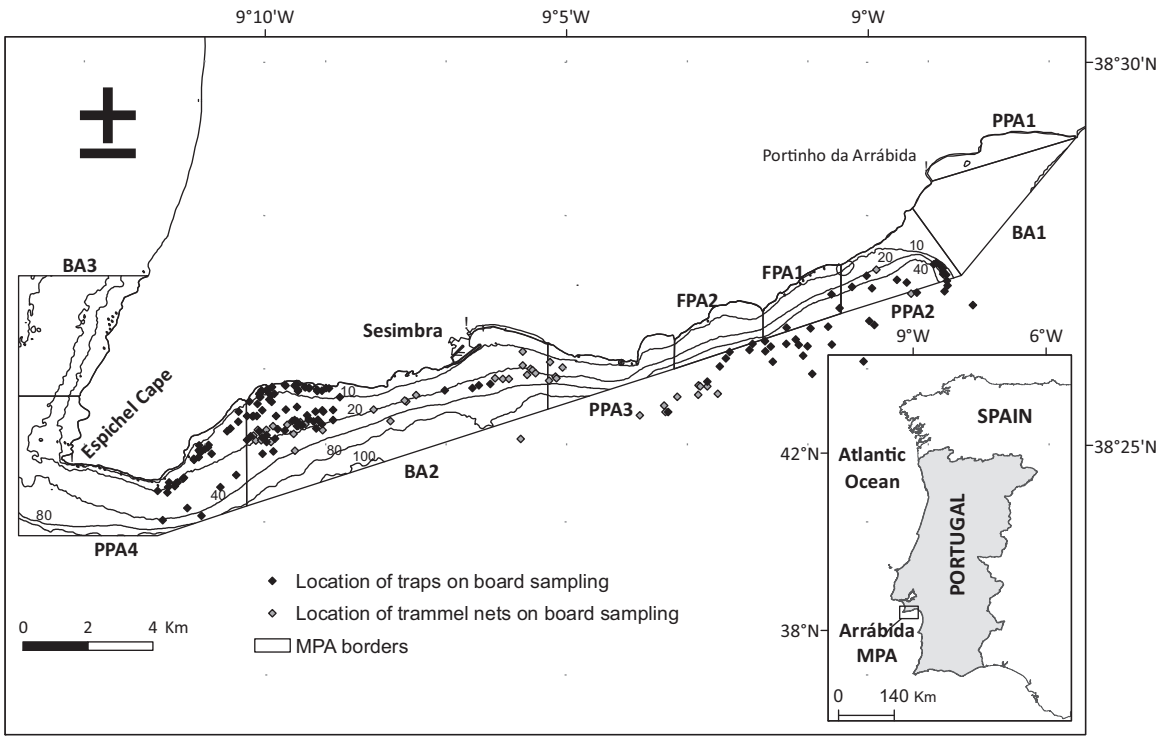

Fig. 1. Map of the Arrábida Marine Park (AMP) with location, bathymetry lines and zoning implemented by the management plan. Zoning: PPA: partially-protected area; BA: buffer area; FPA: fully-protected area. Location of the on-board surveys for traps (black squares) and trammel nets (grey squares) are also shown.

Table 1
Fishing effort sampling phases and periods, number of samples collected and indication of protection zones implemented by period (cells in grey).

Sampling phases	Sampling periods	Number of samples	MPA zones implemented				
			FPA1	FPA2	PPA1	PPA2,3,4	BAs
Before	April–November'04	7					
Implementation	Year 1 (March–August'07)	15	a				
	Year 2 (September'07–February'08)	14	a	a			
	Year 3 (November'08–August'09)	16		a			
After	September–December'09	6					

a: Implemented with PPA regulations.

allowed. Furthermore, commercial fishing licenses are only allotted to vessels registered in the fishing port inside the park and are renewed annually if active. In the partially-protected areas, only traps and jigs are allowed and only beyond 200 m offshore. In the fully-protected area, human presence is not allowed (some exceptions are authorized, for example for scientific research). Since this is a traditional fishing region, the different protection measures were implemented sequentially during a four year transition period, with all the BAs, the PPA1 and half of the FPA (with PPA regulations) established in mid-2006, and all the PPAs and the second half of the FPA (with PPAs regulations) in mid-2007. The first half of the FPA started with FPA regulations in mid-2008 and full implementation of the management plan was achieved in mid-2009 with the second half of the FPA (see Horta e Costa et al., 2013a for details).

The fishing gear and targets that are primarily used in the AMP are: traps mainly targeting octopus, *Octopus vulgaris* Cuvier, 1797; trammel nets which target species such as soles, *Solea senegalensis* Kaup, 1858 and *Solea solea* (Linnaeus, 1758) and cuttlefish, *Sepia officinalis* Linnaeus, 1758; longlines which target mostly Sparidae; and jigs used to catch cephalopods (*O. vulgaris*, *S. officinalis* and squid *Loligo vulgaris* Lamarck, 1798). Most vessels using longlines and jigs are less than 4 m total length and are operated by a single fisher, while traps and nets are used from vessels that are 5–7 m length and usually operated by two fishers (Batista, 2007; Horta e Costa et al., 2013a).

2.2. Data collection

2.2.1. Fishing effort assessment

The buoys of static gear (traps and nets) deployed within the marine park limits were surveyed and identified in all areas south of the Espichel Cape, through zigzag transects by boat (Horta e Costa et al., 2013a,c). The entire study area was sampled on each sampling day, and the location of buoys was marked using a Global Positioning System (GPS). We recorded the type of fishing gear and the name of the vessel from each buoy (mandated by Portuguese legislation).

Sampling was carried out in five periods; 'before', during 'implementation' (three periods: year 1, 2 and 3) and 'after' the implementation of the management plan. Different periods had different protection measures (Table 1). In the analysis we focussed on trap and trammel net fisheries since they were identified as

Table 2
Technical characteristics of fishing gears sampled in on board surveys. Average, standard deviation and mode of depth, number of traps or nets total length, soak time and number of set hauled per day.

Fishery		Depth (m)	N. traps/ nets total length (m) set^{-1}	Fishing time (h)	Sets day^{-1}
Traps	Av. (Std)	35.5 (26.4)	45.8 (8.3)	115.4 (100.8)	6.5 (2.1)
	Mode	16.0	40.0	48.0	8.0
Trammel nets	Av. (Std)	34.4 (27.5)	806.9 (272.1)	24.9 (6.2)	2.5 (0.6)
	Mode	17.0	1000.0	24.0	3.0

the most important and abundant in the study area, and we could accurately assess their position by buoy surveys.

2.2.2. Official landings data

Fisheries landing data were obtained from the Portuguese "Direção Geral de Recursos Naturais, Segurança e Serviços Marítimos". The dataset quantified daily landings (biomass and average price) per species for all vessels fishing in the AMP between 2004 and 2010, as well as vessel characteristics (age, length, fishing licenses). Vessels accounting for 97% of total buoys (according to our fishing effort assessment) were considered as those continuously fishing in the AMP in each period and were used in our analyses of landings. Although these vessels also fished in the boundary areas, most of their catches were probably from the AMP and their landings were considered as a good measure of landings from the AMP. Furthermore, interviews of local fishers were performed to further clarify landings data.

Five species were excluded from the analysis due to known inconsistencies in landings (Bogue *Boops boops* (Linnaeus, 1758), green crab, *Carcinus maenas* (Linnaeus, 1758), Atlantic horse mackerel *Trachurus trachurus* (Linnaeus, 1758), blue jack mackerel *Trachurus picturatus* (Bowdich, 1825) and Atlantic chub mackerel *Scomber colias* Gmelin, 1789). In addition, landings of some species were not discriminated at the species level: *Solea* spp. landings include *S. solea* and *S. senegalensis*, and *Raja* spp. landings include undulate ray, *Raja undulata* Lacepède, 1802, blond ray, *Raja brachyura* Lafont, 1871, thornback ray, *Raja clavata* Linnaeus, 1758, spotted ray, *Raja montagui* Fowler, 1910, sandy ray, *Leucoraja circularis* (Couch, 1838), cuckoo ray, *Leurcoraja naevus* (Müller and Henle, 1841) and other species of genus *Raja*.

2.2.3. Fisheries on-board sampling

Sampling of fishing effort and catches was carried out on-board commercial vessels fishing primarily with trammel nets and traps in the AMP and nearby areas. These vessels also used other gear types at particular times or for short periods. Vessels operating with jigs were also common in the AMP (Horta e Costa et al., 2013a,c), however, their small size did not allow on-board observations and thus they were not included in the study. Surveys were performed on five vessels with similar characteristics between May 2007 and April 2008 in a total of 35 fishing trips regularly distributed in time to account for possible seasonality (16 trips for traps – 3 in Year 1 and 13 in Year 2; 19 for trammel nets – all samples from Year 2). Vessels were chosen based on their fishing area (identified from our fishing effort assessment) with the aim of distributing our sampling across the study area (Fig. 1). Willingness of fishers to participate in the study was also taken into account. The fishing gear used had similar characteristics. Traps were made of an iron or steel frame with a hard plastic net stretched around it (mesh size: 3 cm, trap dimensions: 50 cm × 30 cm × 20 cm). Each trap has a bait holder and a funnel shaped opening. *Scomber colias* and *C. maenas* were the most used baits in traps, and sometimes *B. boops*, *T. trachurus* and sardine *Sardina pilchardus* (Walbaum, 1792) were also added. All bait species were dead, except *C. maenas*. Traps were set from 24 h to several consecutive days, attached to a mainline (Table 2). Trammel nets were composed of 3 panels, made of polyethylene, with an inner panel with 100 mm stretched mesh (the minimum allowed by Portuguese legislation) with ca. 50 mesh high and two outer panels with 600 mm stretched mesh with 3–4 mesh high (Table 2).

Generally, fishing trips were conducted separately for nets and traps (with the exception of five trips where both gears were used). In each survey (i.e. fishing trip sampled), two researchers accompanied one full-day fishing trip (ca. 6–8 h). For each set we measured: length of nets, number of traps or nets, haul location, depth, fishing time (total immersion time) and type of bait (for traps). Catches were separated by the fishers into specimens for selling, for their own consumption or for discarding. Discards are usually promptly thrown back to the sea and were further separated for the purpose of this study; some individuals were discarded alive (mostly small *O. vulgaris* and *Raja* spp.) and the remaining (dead) discards were preserved on ice and brought to the laboratory. All individuals were identified, measured (total length to the nearest mm) and weighed (with a dynamometer, to the nearest 5 g) on-board or at the laboratory. A total of 5521 traps (grouped into 102 sets) and 930 nets (grouped into 48 sets) were surveyed.

2.3. Data analysis

2.3.1. Fishing effort assessment

Buoy density (buoys km^{-2}) was calculated to assess fishing effort: for the total AMP and per protection zone, for the total of the five periods, and per fishing gear (traps and trammel nets) using ArcGIS 10.0 (ESRI, 2011). Data from the fisheries on-board surveys were used to calculate the average number of traps and the average length of nets per set. Analyses of variance (ANOVA for parametric data and Kruskal–Wallis for non-parametric data) were performed to test differences in fishing effort between: periods (independently of protection zone and also in each zone), zones (independently of the period). Tukey and Dunn *post hoc* tests were done respectively when applicable. Statistica 12 software (Statsoft, 2013) was used.

2.3.2. Official landings

We estimated the relative importance of species landed by each vessel in our on-board sampling per time period to evaluate the accuracy of official landings for the studied periods and area. To avoid potentially confounding seasonality, we considered data from an entire year (Before: 2004; Year 1: September 2006 to August 2007; Year 2: September 2007 to August 2008; Year 3: September 2008 to August 2009; After: September 2009 to August 2010). Aggregation of data per period allowed us to compare a full year's landings under the same protection measures. Averaged monthly landings and revenues per vessel were also calculated.

To analyse trends in species landings across periods, landings per unit of effort (LPUE, kg vessel^{-1} day^{-1}) were calculated per period for the group of species representing 95% of total landings (only data for the months sampled in the fishing effort assessment surveys were considered). Total revenue per unit effort (RPUE, € vessel^{-1} day^{-1}) was also estimated.

To analyse the relation between official landings and our sampled fishing effort, we calculated landings per gear unit (LPGU) that accounted for the average number of fishing gears set (obtained in our fishing effort assessment; kg day^{-1} trap^{-1}; kg day^{-1} 1000 m of

nets^{-1}). LPGU was calculated for *O. vulgaris* (for traps) and the soles (*Solea* spp.) and *S. officinalis*—for nets) per period. For each period, we included landings on the days when fishing effort surveys were conducted and we excluded buoys where vessels' identification was not visible. A preliminary assessment of the seasonality of landings was performed (seasons were defined based on landings trends and species biology) but no significant differences among seasons were found (ANOVA, p-value < 0.05). The exception was *S. officinalis,* which had almost no landings between June and October. ANOVA and Kruskal–Wallis tests were performed to analyse statistical significance of differences found in LPGU between periods.

2.3.3. Fisheries catches from on-board sampling

Mean, standard deviation and mode of the number of traps per set, total length of each net, number of sets per day, depth and fishing time (immersion time) were calculated. Data obtained during survey trips were analysed to calculate total catch, portion of catch for selling and for self-consumption and discards per species (in biomass), separately per fishing gear. The primary reasons for discards were also assessed (i.e. species with no commercial value, damaged or undersized individuals).

2.3.4. Comparisons between official landings and catches from on-board sampling

Unreported catches (catches retained but not sold at official fish auction plus catches discarded) were estimated by comparing data from surveys with official landings from the same day and same vessels using the most important species (*O. vulgaris* in the trap fishery and *Solea* spp., European hake *Merluccius merluccius* (Linnaeus, 1758) and *Raja* spp. in the trammel net fishery). When available, species LPUE from official landings data (kg vessel^{-1} day^{-1}) was compared with catch per unit effort from on-board observations (CPUE, kg vessel^{-1} day^{-1}), considering the same vessel, species and day.

2.3.5. Effect of effort and season on catches

To understand which factors influence the total catch of commercial species at the AMP, generalized linear models (GLMs) were run relating the response variable total catch (from the on-board sampling) to the covariates season or month (categorical), the number of traps or total net length and mean water depth (numerical). This approach is useful to identify whether these factors should be included in future monitoring plans and to estimate calibration factors to correct official landings data. Separate models were fit for the most frequently caught species per fishing gear: *O. vulgaris* for traps fishery; and *S. officinalis*, *Solea spp.*, *M. merluccius* and Rajidae for the trammel net fishery. Distribution resulted in residuals which approximated those expected under a normal distribution for all models. ANOVAs were conducted for each model to sequentially test the significance of the covariates considered. An assumption of GLMs is independence between samples and this was guaranteed in our sampling design. Different crews from different vessels may have distinct fishing strategies which could influence total catches, therefore, we included the effect of individual vessel in models as a random variable (Coelho, 2007). Generalized linear mixed models (GLMM) are extensions of GLM and allow for fixed and random effects whilst handling non-normal data (Bolker et al., 2009). GLMMs were run using the model characteristics selected above for the GLMs but including vessel as a random effect (package lme4 1.0–5, R software).

For each model, AIC (Akaike Information Criterion; (Akaike, 1974)) and R^2 (Nagelkerke's coefficient of determination; (Nagelkerke, 1991)) were calculated and used for model comparison and selection. The R^2 was modified for the GLMMs following the method described by Nakagawa et al. (2013). The marginal R^2 (i.e. variance explained by fixed factors) was compared to the conditional R^2 (i.e. variance explained by fixed and random factors) for each GLMM tested. When the random effect of the vessel had approximately zero variance, and the conditional R^2 was equal to the marginal R^2, we assumed there was no among-vessel variance in the response variable and based model selection on the GLM. All these analyses were conducted in the R 3.02 software (R Development Core Team, 2012).

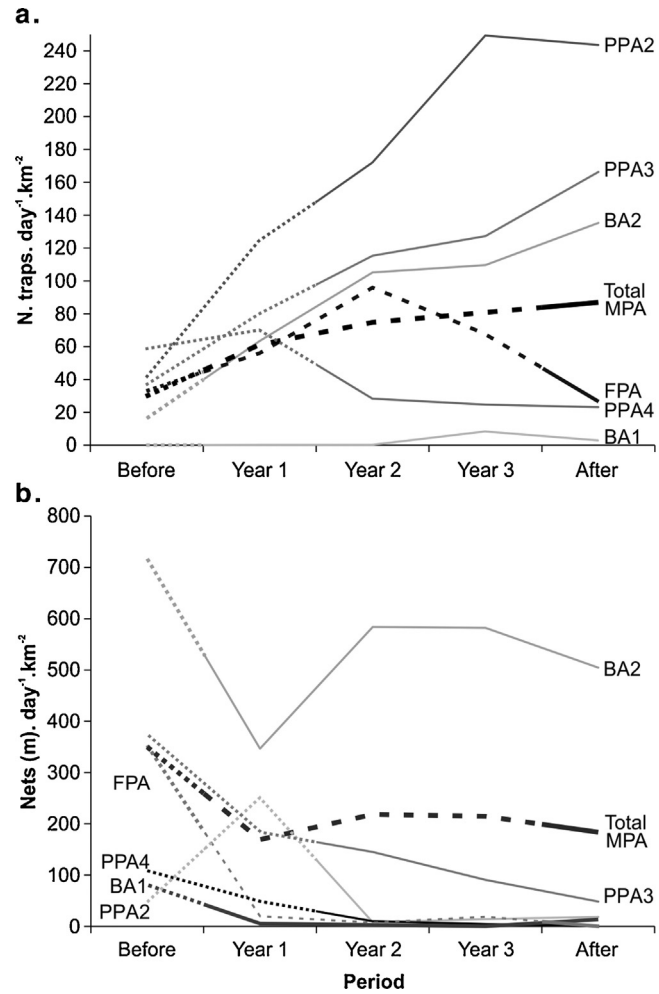

Fig. 2. Density of (a) traps and (b) nets by protection zones at the Arrábida Marine Park throughout the different periods of the implementation of the management plan: Before, Year 1, Year 2, Year 3 and After. Dotted lines: no conservation measures are implemented; dashed lines: conservation measures of FPA are partially implemented; full lines: conservation measures are fully implemented.

3. Results

3.1. Fishing effort assessment

Fishing effort varied over time and patterns depended on specific protection zone and gear type (Fig. 2a). Generally, trap density increased over time but the specific pattern of increase differed among protection zones (Fig. 2a). Trap density in PPA2, PPA3 and BA2 increased continuously whereas trap density in FPA increased between Before and Year 2 when the FPA was functioning as a partially-protected area (where traps are allowed beyond 200 m from shore) and then decreased after year 2 when the FPA was fully implemented as a no-take zone. Effort in PPA4 decreased markedly after year 1 when this area became a partially-protection zone. The variation across time periods was statistically significant with analyses of variance in each of the protection

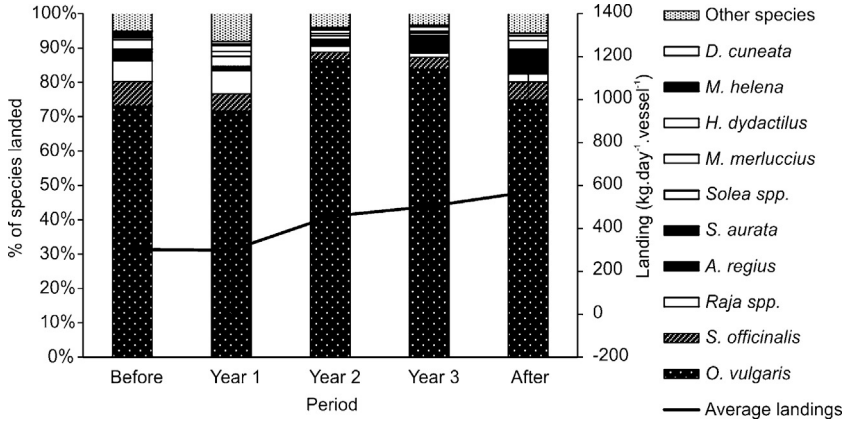

Fig. 3. Relative importance (%) of species in the overall landings at the Sesimbra port during the periods of the implementation of the management plan: before, years 1, 2, 3 and after. Averaged monthly biomass (kg) landed per vessel and respective standard deviation are also represented.

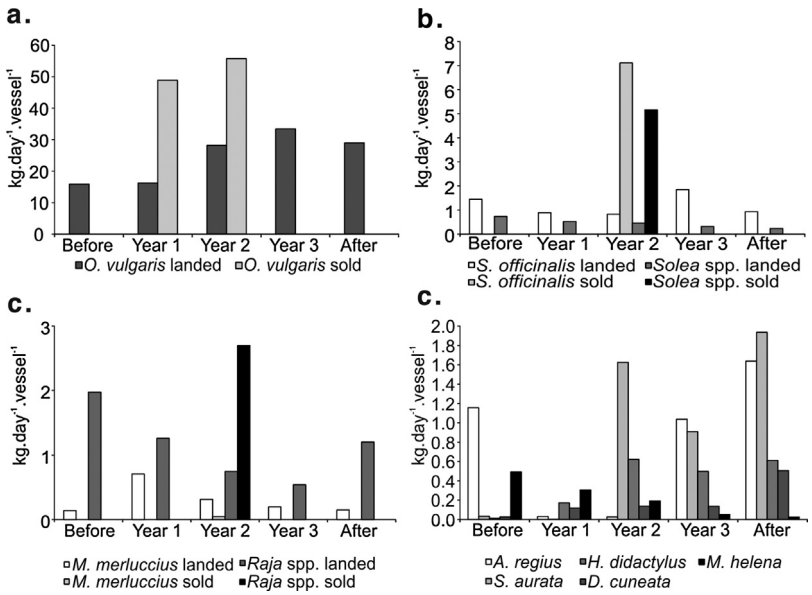

Fig. 4. Landings per unit of effort (kg day^{-1} vessel^{-1}) at the Sesimbra port for the most landed species during the periods of the implementation of the management plan: before, year 1, 2, 3 and after. Also shown are the most captured species during on-board observations (only available for year 1 for species from trammel nets fishery and for years 1 and 2 for *O. vulgaris*), captures recorded by fishers to sell (kg day^{-1} vessel^{-1}). (a) *O. vulgaris* (b) *S. officinalis*, *Solea* spp.; (c) *Raja* spp., *M. merluccius* and (d) *A. regius*, *S. aurata*, *H. didactylus*, *D. cuneata* and *M. helena*.

zones (p-value < 0.05; df = 4; PPA2 – F = 17.35; PPA3 – F = 11.71; PPA4 – F = 18.68; FPA – F = 11.40; BA1 – H = 24.92; BA2 – F = 19.13). There were also significant differences in the overall trap density both between periods (p-value < 0.05; df = 4; F = 12.84) and between zones (p-value <0.05; df = 5; H = 200.97). *Post-hoc* tests showed that trap density was significantly lower in periods Before and in Year 1 than in subsequent periods and that protection zones FPA, BA2 and PPA4 had significantly lower trap densities than PPA2 and PPA3.

Temporal patterns of net density in the trammel net fishery were almost the opposite of those of the trap fishery, with a steep decrease in trammel net density over the study period (Fig. 2b). The analyses of variance showed significant differences in fishing effort between periods (p-value <0.05; df = 4; F = 3.17) and also between zones (p-value <0.05; df = 5; H = 164.95). *Post-hoc* tests showed a decrease between Before and Year 1. BA2 had the highest density of nets in all periods, and was significantly different from the remaining zones. The analyses showed decreasing trends across zones with the exception of an increase in net density in PPA2 between Before and Year 1 and in BA2 between Year 1 and Year 2. Statistical tests performed on net density for each zone separately only detected significant difference in PPA4 (p-value <0.05; df = 4; H = 14.67), with significantly higher densities in Year 2 when compared to the following periods.

3.2. Official landings

Total landings of vessels operating in the AMP increased over time both in weight and overall revenue (Table A1). The number of vessels continuously fishing with nets and traps in the AMP decreased between the Before (22 vessels) and After (18 vessels) periods. Total revenue per unit of effort (RPUE = € vessel^{-1} day^{-1}), estimated through landings data, increased over the studied time frame (Table A.1).

Although there was some variability in the relative importance of species landed, average monthly landings per vessel (in biomass) generally mirrored the pattern of total landings, increasing over the study period (Fig. 3). Approximately 160 taxa were landed, but 95% of total landings (in biomass) were comprised of only 10 taxa (Fig. 3 and Table A.1). *Octopus vulgaris* was the most landed species in all

periods, accounting for between 71.7% (Year 1) and 86.5% (Year 3) of total landings. Total catches of species targeted by the trammel net fishery (*S. officinalis*, *Raja* spp. and *Solea* spp.) generally decreased between the Before and After periods (Fig. 3). Trends observed for LPUE were generally similar to those of total biomass landed per species (Fig. 4a–d).

Eight of the 10 most landed taxa (all except for *Halobatrachus dydactilus* and *Muraena helena*) plus European seabass, *Dicentrarchus labrax* (Linnaeus, 1758), were responsible for 95% of total revenue (Table A.1). *Octopus vulgaris* was the valuable species in terms of total revenue, followed by gilthead seabream *Sparus aurata* Linnaeus (1758), *Solea* spp. and *S. officinallis*, respectively (Table A.1).

Octopus vulgaris LPGU increased over the study period with a significant difference between the Before and After periods (p-value <0.05) (Fig. 5). *Solea* spp. did not show significant differences among periods. *Sepia officinalis* LPGU was relatively constant between Before and Year 3, with a maximum in the After period although no significant differences in LPGU among periods were detected.

3.3. Fisheries on-board sampling

3.3.1. Trap fishery

During trap sampling 41 taxa (36 species identified at species level and some individuals grouped at higher taxonomic levels) were identified. Of the biomass caught 78% was sold, 13% was discarded and the remaining was retained by fishers for personal consumption. The target species for this fishery, *O. vulgaris,* represented 91% of total catch. *Halobatrachus didactylus* (Bloch and Schneider, 1801), *Holothuroidea* and *Scorpaena notata* Rafinesque (1810) were the next most-caught taxa, although with low relative importance. Several economically valuable species were caught, but they accounted for low biomass (e.g. *S. officinalis*, *Necora puber* (Linnaeus, 1767)). For *O. vulgaris,* 93% of biomass caught was retained (8.5% of which was for fishers own consumption) and the remaining was discarded mostly alive due to being underweight (Table B1). Undersized individuals were much abundant in nearshore zones (within 200 m offshore) than offshore and consequently discards of undersized individuals were higher nearshore (ca. 8 individuals 100 traps^{-1} within 200 m offshore and 0.4 individuals. 100 traps^{-1} in offshore area).

3.3.2. Trammel net fishery

Approximately 80 taxa were identified in the trammel net fishery but 15 species comprised 80% of total biomass caught (Table C1). From the biomass caught, 44% was for sale, 34% was discarded and 22% was for fishers' own consumption. Target species were *Solea* spp. (mainly *S. senegalensis* and *S. solea*) and *S. officinalis,* which represented 30% of total biomass caught (6.9%, 4.5% and 19.1%, respectively) and 65% of total biomass sold. A high percentage of several other species (e.g. *Raja spp.*, *M. merluccius*, grey triggerfish, *Balistes capriscus* Gmelin, 1789, *O. vulgaris*) was retained. Among those species, *Raja* spp. had a high relative importance among total captures for sale (13.5% of the total biomass sold) and *B. capriscus* and *M. merluccius* were most likely to be retained for self-consumption (20.1% and 13.9% of total catches for personal consumption, respectively). *Scomber colias* and various invertebrates were the most discarded species (*S. colias*: 42.7% of total discards). In general, catches directed to fishers' self-consumption had low market value (due to species characteristics, small size or scarce number of individuals caught from a given species). Low market value or damaged condition were the major reasons for discards.

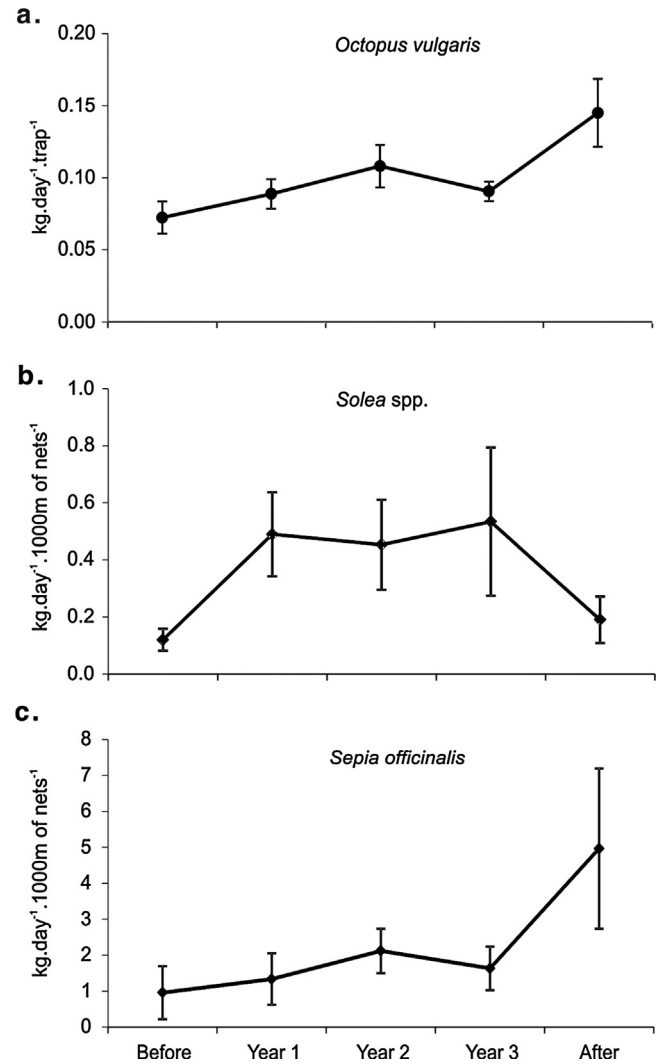

Fig. 5. *Octopus vulgaris* (a), *Solea* spp. (b) and *Sepia officinalis* (c) average landings per gear unit (LPGU), (a) kg day^{-1} trap^{-1}; (b, c) kg day^{-1}. 1000 m of nets^{-1}, and standard error at the Sesimbra port throughout the different periods of the implementation of the management plan: before, years 1, 2, 3 and after.

3.4. Comparisons between official landings and catches from on-board sampling

Targeted and high value bycatch species (e.g. *Raja* spp.) showed the highest correspondence between landed (landings data) and caught (from on-board sampling) biomass. Not surprisingly, both these groups were high in both. Other species had relatively lower on-board catches than those recorded in official landings (e.g. *M. merluccius, M. helena,* wedge sole *Dicologlossa cuneata* (Moreau, 1881)). Despite high relative importance in landings, two species, meagre *Argyrosomus regius* (Asso, 1801) and *S. aurata,* were never observed during on-board surveys as these species are caught during short periods by longlines. Some vessels fish with longlines sporadically or within short time-frames each year and this activity was not captured in our on-board program.

Our analysis of official landings from the trap fishery on the same days and vessels as on-board sampling showed that there were no registered landings for approximately 68.8% of the fishing trips observed despite the fact that in every sampled trip there were catches separated on-board for sale. An average of 72.2% of *O. vulgaris* caught (not including discards) per day was not reported in official fish auctions and corresponding official statis-

tics. CPUE (kg vessel^{-1} day^{-1}) calculated for *O. vulgaris* separated on-board for sale by the fishers was on average more than twice as high as LPUE (kg vessel^{-1} day^{-1}). Furthermore, besides *O. vulgaris*, among all catches recorded during on-board observations, only black seabream, *Spondyliosoma cantharus* (Linnaeus, 1758) and *S. officinalis* were reported in official landings (ca. 20% and 64% of total catches were delivered at official fish auction, respectively).

For the trammel net fishery, no landings were registered for approximately 16% of fishing trips in which catches were recorded on-board. The highest levels of unreported catches were for *Solea* spp., *S. officinalis* and *Raja* spp. (45.9%, 36.6% and 75.3%, respectively). CPUE of target species (*Solea* spp., *S. officinalis*) in trammel nets were much higher than LPUE (Fig. 4b). For *M. merluccius* there were no reported landings, although 15% of catches were recorded on-board as being for sale (67% of catches were for own consumption and 18% were discarded). In addition, there were records of landings for 13 more taxa, but several inconsistencies were identified: (i) species landed, but not caught during on-board observations (e.g. 127 kg of *T. trachurus* in two different days; 137 kg of *S. colias* in one day); (ii) species caught but landed under a wider taxonomic category or incorrect species designation (e.g. *Lepidorhombus boscii* landed under a category that can include the genus *Citharus* and *Lepidorhombus;* sometimes species from genus *Raja* were landed with incorrect species or *Raja* spp.); (iii) non-targeted species recorded with lower biomass than observed on-board and with lower frequency than caught (e.g. *B. capriscus*, tub gurnard *Chelidonichthys lucerna* (Linnaeus, 1758)).

3.5. Effect of effort and season on catches

The best model selected for the total catch of *O. vulgaris* (GLMM) targeted by the trap fishery explained 72.10% of total variability (Table 3). Both fishing effort and season (fixed factors) significantly influenced *O. vulgaris* total catch and 7.11% of variability was due to a random factor (vessel). Catches of this species generally increased with number of traps, until a plateau in the number of traps was reached. Octopus catches were much higher in late winter/spring than in summer (Table 3 and Fig. 6a).

Two models were selected for catches from the trammel net fishery, one using *S. officinalis* and the other using the sum of the most-caught species. The best model for *S. officinalis* (GLMM) explained 82.80% of total variability (Table 3 and Fig. 6b). There was no clear relationship between the total amount of *S. officinalis* caught and fishing effort as total catches only increased with the increase of net length below a low value above which an additional increase in net length did not increase catches. Despite this there are numerous zero catches, likely related to season which had a strong influence on catch. Catches were highest in autumn/winter (S1) and spring (S2). The random factor individual vessel also influenced *S. officinalis* captures (Table 3). In contrast, the model obtained for the most caught fishes in trammel nets (i.e. both target and relevant bycatch species) explained a small percentage of total variance (10.75%). In that GLM, only month was significant, with higher catches generally observed during cold months (Table 3 and Fig. 6c).

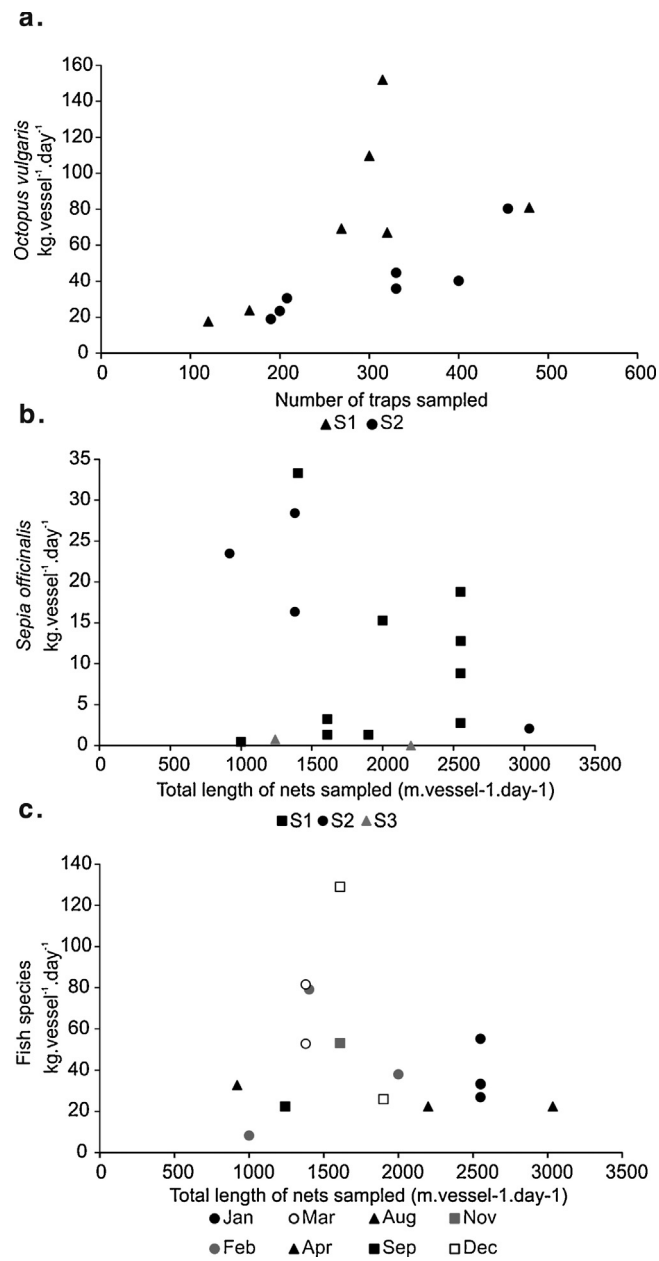

Fig. 6. Relationship of fishing effort (number of traps vessel^{-1} day^{-1} or total length (m) of nets sampled vessel^{-1} day^{-1}) and season or month on *Octopus vulgaris* (a), *Sepia officinalis* (b) and a group of fish species (c) catches observed during on-board sampling of vessels fishing in Arrábida Marine Park with traps and trammel nets. Seasons considered were S1: February–April; S2: August, September, December (a) and S1: November–February; S2: March, April; S3: August, September (b).

4. Discussion

4.1. Fishery redistribution and reporting

The assessment of catches (and consequently revenues) is a potential key tool to evaluate impacts (both positive and negative) of MPAs to local fisheries, fishers' successful adaptation and the effects of protection on fished species. The combination of methods used in this study to assess and monitor fisheries catches in different phases over the establishment of an MPA is, to our knowledge, a novel approach and has a strong potential for application in other systems.

Fishing catch and effort assessments of artisanal fisheries are essential to calibrate official reported commercial landings. Fishing effort analyses revealed that, as expected, some fishing grounds became inaccessible to particular fisheries after MPA implementation. In the case of Arrábida (AMP), approximately half of the park is off-limits to very nearshore traps and jigs (up to 200m from shore) and to trammel net fisheries, as discussed in a recent work describing the spatial adaptations of fisheries in response to this MPA (Horta e Costa et al., 2013a). In the current study, changes in fishing effort (both spatial and temporal) resulted in an increase in the number of traps within AMP, suggesting that this fishing gear

Table 3
Results of the best models obtained – generalized linear (GLM) and mixed (GLMM) models – testing captures from onboard samples, from traps and trammel nets fisheries, in relation to the best set of explanatory variables selected: fishing effort – number of traps – Ntraps – or trammel nets – Nnets (continuous variable); season or month (*O. vulgaris*: February–April; S2: August, September, December; S. officinalis: S1: November–February; S2: March, April; S3: August, September (b); fish species: month). Species considered were *Octopus vulgaris*, *Sepia officinalis* and the trammel nets most targeted fish species (*Solea senegalensis*, *Solea solea*, *Merluccius merluccius*, *Raja clavata*, *Raja undulata*, *Raja alba*, *Raja montagui* and *Raja brachyura*). The% of variance explained is given by the R^2. In the GLMMs the random variable is the vessel (random effects). Estimated degrees of freedom (edf), statistical tests Chisq (for GLMM), *F*-statistics (for GLM) and corresponding *p*-values are indicated. Significant values are in bold. Marginal (i.e. variance explained by fixed factors) and conditional (i.e. variance explained by fixed and random factors) variances are shown for GLMMs.

Fishing gear (model) species	Explanatory variables	Df	Chisq/*F*-statistics	*p*-value	% of variance explained (R^2)
Traps	Ntraps	1	19.57	$p < 0.001$	Marginal: 64.99
(GLMM)	Season	1	10.92	$p < 0.001$	Conditional: 72.10
O. vulgaris					
Trammel nets	Nnets	1	4.12	$p < 0.05$	Marginal: 49.20
(GLMM)	Season	2	19.46	$p < 0.001$	Conditional: 82.80
S. officinalis					
Trammel nets	Nnets	1	0.50	$p < 0.05$	10.75
(GLM)	Month	7	2.55	$p < 0.05$	
Fish species					

sustained most of the fishers' adaptation to AMP regulations. However, net density decreased even in the area where they are still allowed, suggesting that AMP rules were not the only factor leading to the decreased use of trammel nets. The re-direction of fishing effort from finfish (caught mostly by nets) to cephalopods (caught mostly by traps) that we observed in AMP has also been observed at national level (Moreno et al., 2014; Pilar-Fonseca et al., 2014) and has been suggested to result from a decline in national demersal finfish stocks. In the AMP, however, it is possible the remaining areas where nets were allowed are simply too small to support such fishing and thus fishers changed from nets to traps, which can be fished even in small areas.

Several factors may influence shifts in the use of fishing gears in small-scale coastal artisanal fisheries occurring inside marine protected areas, such as the one studied here. These include: (i) changes in the abundance of target species due to environmental conditions, reserve effects and/or excessive fishery pressure (e.g. our observed increase in *O. vulgaris* and decrease in trammel nets target species such *Solea* spp., *S. officinalis*), (ii) market driven factors (e.g. increase in market demand and higher prices for *O. vulgaris*); (iii) loss of fishing grounds; (iv) competition for "fishing space" (usually nets occupy larger areas than traps). Independent of specific factors and their interactions, the multigear and multispecies nature of local artisanal fisheries confers an advantage in response to spatial limitations since other options, such as moving to distant fishing grounds far from home port are less viable due to technical limitations (e.g. vessels small size and low power) (Lédée et al., 2012).

Our observed differences between actual on-board catches and subsequent reported landings are common in artisanal fisheries worldwide. Many studies have documented the high prevalence of unreported catches (Batista et al., 2009; Coll et al., 2014; Pauly et al., 2014). If we assume that official landing statistics will always be inaccurate, then only by creating means to estimate realistic levels of bycatch and the level of other unreported catches can we hope to improve fisheries management of these important artisanal fisheries (Ainsworth and Pitcher, 2005; Alverson et al., 1994; Tsagarakis et al., 2013). Accurate estimation is also key for effective MPA management and monitoring and usually requires the cooperation of local fishers.

In the present study, the ranking of target species abundance in catches and landings was similar despite differences in absolute abundance. Species other than those that are targeted, in particular the skates, were found to be economically important for local fisheries. However, despite the economic importance to the fishery, the landings data for these non-targeted species was even less reliable than for the target species, many are simply not reported. Reasons for non-reporting appear to be related to economic and socio-cultural issues such as (i) captures sold outside the fishing docks (the only legal channel for sales) often are of higher economic value since there are no middlemen and fishers avoid paying taxes; (ii) culturally, fishers used to keep some fish to sell (or give) to their neighbours and (iii) sometimes the quantities caught per species are very small and thus have low demand by middlemen in fishing docks, thus fishers opt to reserve these captures to sell (or give) or for their own consumption. To the authors knowledge these types of situations occur at the national level, although with very variable patterns and rates, depending of several factors such as the region and the fishery's m (a is a group of fishing operations targeting a specific assemblage of species, using a specific gear, during a precise period of the year and/or within the specific area), but see Batista et al. (2009) for a brief discussion on this.

Official landings did not account for discarded species or those that were typically retained for fishers' own consumption, and therefore do not reflect the true catches of the fishery (Ainsworth and Pitcher, 2005; Alverson et al., 1994; Batista et al., 2009). Some species (e.g. *M. merluccius*, *H. dydactilus*) were important both in landings and in on-board catches but were mostly discarded or kept for fishers' own consumption, while others were important in landings but were not observed in on-board catches (e.g. *S. aurata*, *A. regius*, *D. cuneata*). There are a variety of possible explanations for this: (i) landings could be from catches made by other larger vessels (fishing with gill and trammel nets or longlines) and recorded as being from AMP vessels (e.g. *M. merluccius*, *D. cuneata*) (e.g. due to the need to report at least 100 days of catches to renew the licence); (ii) species could be caught and landed by vessels fishing in the AMP but using non-licenced or other less used gears, since vessels have licenses for several fishing gears (e.g. longlines) which they only seldom use; (iii) landed species were not correctly identified (e.g. *D. cuneata* due to morphological similarity with *Solea* spp.). All these factors may explain the differences found between landings and on-board observations. Official landings were also a poor proxy of fishing effort since no landings were reported for most fishing days. Thus on-board observations and number or length of fishing gears set per day would be more appropriate measures of fishing effort to monitor these coastal fisheries.

4.2. Individual species responses

The analyses of the observed trends in landings of the most important target species are indicative of the impact MPAs are having on local fisheries since, in theory, this group of species

are responsible for most of fishers' incomes. This type of assessment is a simplification of reality, as there are many factors other than MPA implementation that can influence captures (and their economic value), such as independent, market-driven factors and natural environmental variability. However, the integration of the results obtained here with conclusions obtained in other studies can support some of the present observations as discussed below.

The increase in LPGU of *O. vulgaris* during the study, although not statistically significant, suggests that protection measures may have contributed to some recovery in abundance and/or size of this species within the AMP. Horta e Costa et al. (2013b) found more *O. vulgaris* in highly protected zones (FPA and PPAs) than in control zones, suggesting that protection can benefit the fishery outside these areas if spillover events occur. Also, overall catches of small individuals probably diminished with AMP implementation due to exclusion of fishing within 200 m from shore in PPAs, and the prohibition of spearfishing and other limitations on recreational fisheries inside the park. In fact, these limitations have been shown to benefit small-scale commercial fisheries in multiple use MPAs (Cooke and Cowx, 2004; Rocklin et al., 2011). Nevertheless, since *O. vulgaris* recruitment and year class strength are known to be strongly influenced by environmental factors, such as salinity and water temperature (Lourenço et al., 2012; Moreno et al., 2014), we expect those factors to affect population abundance over long time frames. This hypothesis is also supported by the GLM results since the factor "season" was significant for catch.

S. officinalis also showed an increasing trend in LPGU in the After period which could be a sign of positive effects of the implementation of the AMP. However, the small size of the no-take area together with the short life cycle and rapid growth of this species (Dunn, 1999; Neves et al., 2009) make it unlikely that protection alone played a major role in the observed trends. In fact, our models and those of others (Abecasis et al., 2013) indicated that environmental factors (season) were an important driver of *S. officinalis* catches. The protected area may, however, contribute to a decrease in fishing mortality of mature individuals before breeding, as *S. officinalis* individuals use this coastal area before entering the nearby Sado estuary to spawn (Abecasis et al., 2013; Neves et al., 2009). These authors also suggested that larger individuals can spawn in the adjacent coastal areas of this estuary, including areas within AMP boundaries, which could improve the positive effects of the reserve. Again, as found for *O. vulgaris*, environmental conditions are likely to contribute more strongly (and over a longer time-frame) to recruitment success and population trends, since the growth of juveniles is extremely dependent on environmental conditions (Dunn, 1999; Koueta and Boucaud-Camou, 2003).

Solea spp. LPGU was similar from Before to After periods with no significant differences through time. In any case, benefits for the *Solea* spp. fishery due to protection effects, if occurring, are unlikely to be detected through landings data *per se* since high levels of unreported catches obscure any possible sign of protection effects. The lower fishing pressure on these exploited species should likely promote high adult biomass and abundance in the "no-nets" area (PPA and FPA). For instance, Claudet et al. (2010) in a study considering several temperate reserves of Southern Europe found significant increases in density of exploited fish species inside reserves. In addition, these authors showed responses increased with time since protection and with the size of the no-take zone (Babcock et al., 2010; Claudet et al., 2008). However, in a recent study by Abecasis et al. (2014) no significant changes in mean abundance or biomass of *S. senegalensis* attributable to AMP protection levels were found, as also discussed in Halpern (2003) for other MPAs. Hence, overall predictions about the contribution of the study area to an increase in surrounding fisheries are premature. The small size of the reserve (see e.g. Claudet et al., 2008; Palumbi, 2003) and the fact that key parts of these species life-cycle do not occur inside the park make it unlikely that AMP will be a key to the sustainability of some of these species stocks. Nonetheless, the quality and availability of known nursery grounds in the nearby estuaries may be an important key component to take into consideration in conservation efforts of species such as *Solea* spp., *S. officinalis*, and other finfish (Tanner et al., 2013; Vasconcelos et al., 2011). The GLM detected the importance of fishing effort to explain the amount of *Solea* spp. catches, as expected, but season was also a significant factor which reflects the importance of species ecology in the observed trends.

Besides the above discussion on individual responses to protection, the reserve contribution to enhance the biomass available to fish (outside no-fishing areas) may be offset by the increase of fishing effort in some PPAs and BAs bordering the no-take reserve (and potentially outside the AMP borders) through "dispersal imbalance" phenomena (Abesamis and Russ, 2005 Walters et al., 2007). Thus, fishing effort occurring outside AMP boundaries should also be included in future monitoring plans in order to assess whether "fishing the line" can be affecting eventual spillover effects.

4.3. Conclusions

Here we show that vessels fishing inside the AMP apparently did not suffer a decrease in revenues during the study period as could have been initially expected due to the implementation of fisheries restrictions (Batista et al., 2011). Fishers apparently compensated for losses in some target species (such as *S. officinalis* and *Solea* spp.) with increases in the use of additional gear types targeted to other species (such as traps and with a consequent increase in *O. vulgaris* catches). While these results are encouraging, it is not clear if the current levels of resource exploitation are sustainable and if other factors might be influencing interpretation of landing trends as discussed by Horta e Costa et al. (2013b). For instance, the applied statistical models highlighted the seasonal variability in catches and the significant influence of environmental factors. Thus, a better understanding of the influence of environmental conditions, species life history characteristics and even fishers' behaviour would allow for more accurate conclusions about the importance of these factors in both captures and revenues. Nonetheless, to integrate the contribution of each factor, longer and more complete data series on catches (as well as data on fishing effort) are essential.

The manner in which protection measures in AMP may benefit some of the target species of coastal fisheries is still an open question, even if early reserve effects have been shown for finfish and some invertebrates (Horta e Costa et al., 2013b). In addition, effects on fisheries due to spillover or larval export processes (Gell and Roberts, 2003) are still unknown. Overall, the small size of the no-take zone, the high fishing effort observed in the area, the infancy of this park and challenges with enforcement of regulations likely all contribute to the lack of a clear signal in the recovery of coastal fisheries.

Despite the short duration of the on-board sampling we documented high levels of unreported catches and the large inconsistencies in official landings when compared with sampling-based data. This issue is likely a common feature to many coastal MPAs (Lescrauwaet et al., 2013) and will to be addressed to allow more accurate data on fisheries captures. A longer time series of on-board observations would allow a higher confidence in the estimated correction factors since several features could influence the level of inconsistency in official statistics (e.g. economic, cultural, seasonal, fishers behaviours). Thus, landings data could be a useful and cost-effective tool for MPA monitoring but only when calibrated by on-board sampling of catches and fishing effort collected over extended periods (to include seasonal variability), and by combining these data sources with accurate correction factors through

the application of modelling approaches. Longer sampling datasets would also allow for a better fit of the models applied here, or enable the use of fishing effort (number of traps and net's length) to estimate global catches of target species. Efforts to include fishers in MPA monitoring and increasing their awareness to the importance of reliable datasets could be advantageous since it would potentially provide higher data reliability and/ or complementary information (Leleu et al., 2014; Roman et al., 2011) with reduced costs.

The approach described here of combining fishing effort, on-board data collection and official landings proved to be an effective tool for monitoring small-scale artisanal fisheries in MPAs. However, in most cases official landings are the only datasets available. In order to be able to use these data reliably one needs to assume that biases are consistent over time (e.g. the relative importance of unreported catches among years are constant), allowing the quantification of general patterns and trends in putative catches and revenues. In addition, knowledge of fishers' behaviours and perceptions, and the socio-ecological system under study are essential. Understanding these general trends may help in the early detection of unsustainable exploitation practices thus allowing managers to implement more efficient or preventive monitoring and management measures.

Overall, landings data for coastal artisanal fisheries worldwide are very scarce, and usually unreliable and biased. When using these data to monitor or evaluate the effectiveness of MPA developing effective methodological approaches, such as the one presented in the current study, which seek to measure and account for the biases in the data will be essential.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.fishres.2015.07.020

References

Abecasis, D., Afonso, P., O'Dor, R.K., Erzini, K., 2013. Small MPAs do not protect cuttlefish (*Sepia officinalis*). Fish. Res. 147, 196–201.
Abecasis, D., Afonso, P., Erzini, K., 2014. Can small MPAs protect local populations of a coastal flatfish, *Solea senegalensis*? Fish. Manag. Ecol. 21, 175–185.
Abesamis, R.A., Russ, G.R., 2005. Density-dependent spillover from a marine reserve: long-term evidence. Ecol. Appl. 15, 1798–1812.
Ainsworth, C.H., Pitcher, T.J., 2005. Estimating illegal, unreported and unregulated catch in British Columbia's marine fisheries. Fish. Res. 75, 40–55.
Akaike, H., 1974. A new look at the statistical model identification. IEEE Trans. Autom. Control 19, 716–723.
Alverson, D.L., Freeberg, M.H., Pope, J.G., Murawski, S.A., 1994. A Global Assessment of Fisheries Bycatch and Discards. FAO Fisheries Technical Paper. FAO, Rome, p. 233.
Babcock, R.C., Shears, N.T., Alcala, A.C., Barrett, N.S., Edgar, G.J., Lafferty, K.D., McClanahan, T.R., Russ, G.R., 2010. Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. Proc. Natl. Acad. Sci. U. S. A. 107, 18256–18261.
Batista, M.I., 2007. Avaliação do impacto do Plano de Ordenamento do Parque Natural da Arrábida na pesca comercial local (Assessement of the Impacts of Arrábida Marine Park Management Plan in Local Fisheries). Faculdade de Ciências, Universidade de Lisboa, Lisboa, Portugal, p. 85.
Batista, M.I., Baeta, F., Costa, M.J., Cabral, H.N., 2011. MPA as management tools for small-scale fisheries: the case study of Arrábida Marine Protected Area (Portugal). Ocean Coast. Manag. 54, 137–147.
Batista, M.I., Teixeira, C.M., Cabral, H.N., 2009. Catches of target species and bycatches of an artisanal fishery: the case study of a trammel net fishery in the Portuguese coast. Fish. Res. 100, 167–177.
Bolker, B.M., Brooks, M.E., Clark, C.J., Geange, S.W., Poulsen, J.R., Stevens, M.H.H., White, J.S.S., 2009. Generalized linear mixed models: a practical guide for ecology and evolution. Trends Ecol. Evol. 24, 127–135.
Christie, P., 2004. Marine protected areas as biological successes and social failures in southeast Asia. Am. Fish. Soc. Symp. 42, 155–164.
Claudet, J., 2011. Marine Protected Areas—A Multidisciplinary Approach. Cambridge University Press, Cambridge, UK.
Claudet, J., Guidetti, P., 2010. Improving assessments of marine protected areas. Aquat. Conserv. 20, 239–242.
Claudet, J., Osenberg, C.W., Benedetti-Cecchi, L., Domenici, P., Garcia-Charton, J.A., Perez-Ruzafa, A., Badalamenti, F., Bayle-Sempere, J., Brito, A., Bulleri, F., Culioli, J.M., Dimech, M., Falcon, J.M., Guala, I., Milazzo, M., Sanchez-Meca, J., Somerfield, P.J., Stobart, B., Vandeperre, F., Valle, C., Planes, S., 2008. Marine reserves: size and age do matter. Ecol. Lett. 11, 481–489.
Claudet, J., Osenberg, C.W., Domenici, P., Badalamenti, F., Milazzo, M., Falcon, J.M., Bertocci, I., Benedetti-Cecchi, L., Garcia-Charton, J.A., Goni, R., Borg, J.A., Forcada, A., De Lucia, G.A., Perez-Ruzafa, A., Afonso, P., Brito, A., Guala, I., Le Direach, L., Sanchez-Jerez, P., Somerfield, P.J., Planes, S., 2010. Marine reserves: fish life history and ecological traits matter. Ecol. Appl. 20, 830–839.
Coelho, R., 2007. Biology, Population Dynamics, Management and Conservation of Deep Water Lantern Sharks, *Etmopterus spinax* and *Etmoepterus pusillus* (Chondrichthyes: Etmopteridae) in southern Portugal (northeast Atlantic). Faculdade de Ciências do Mar e do Ambiente, Universidade do Algarve, Faro, Portugal, p. 264.
Coll, M., Carreras, M., Cornax, M.J., Massutí, E., Morote, E., Pastor, X., Quetglas, A., Sáez, R., Silva, L., Sobrino, I., Torres, M.A., Tudela, S., Harper, S., Zeller, D., Pauly, D., 2014. Closer to reality: reconstructing total removals in mixed fisheries from Southern Europe. Fish. Res. 154, 179–194.
Cooke, S.J., Cowx, I.G., 2004. The role of recreational fishing in global fish crises. Bioscience 54, 857–859.
Costanza, R., Andrade, F., Antunes, P., den Belt, M., Boersma, D., Boesch, D.F., Catarino, F., Hanna, S., Limburg, K., Low, B., Molitor, M., Pereira, J.G., Rayner, S., Santos, R., Wilson, J., Young, M., 1998. Principles for sustainable governance of the oceans. Science 281, 198–199.
Danielsen, F., Burgess, N.D., Balmford, A., 2005. Monitoring matters: examining the potential of locally-based approaches. Biodivers. Conserv. 14, 2507–2542.
Dunn, M.R., 1999. Aspects of the stock dynamics and exploitation of cuttlefish, *Sepia officinalis* (Linnaeus, 1758), in the English Channel. Fish. Res. 40, 277–293.
ESRI, 2011. ArcGIS Desktop: Release 10. Environmental Systems Research Institute, Redlands, CA.
Field, J.C., Punt, A.E., Methot, R.D., Thomson, C.J., 2006. Does MPA mean 'major problem for assessments'? Considering the consequences of place-based management systems. Fish Fish 7, 284–302.
Frid, C., Paramor, O., Scott, C., 2005. Ecosystem-based fisheries management: progress in the NE Atlantic. Mar. Policy 29, 461–469.
Garcia, S.M., Allison, E.H., Andrew, N., Béné, C., Bianchi, G., De Graaf, G., Kalikoski, D., Mahon, R.L., Orensanz, L., 2008. Towards Integrated Assessment and Advice in Small-scale Fisheries: Principles and Processes. FAO Fisheries and Aquaculture Technical Paper No. 515.
Gell, F.R., Roberts, C.M., 2003. Benefits beyond boundaries: the fishery effects of marine reserves. Trends Ecol. Evol. 18, 448–455.
Gerber, L.R., Botsford, L.W., Hastings, A., Possingham, H.P., Gaines, S.D., Palumbi, S.R., Andelman, S., 2003. Population models for marine reserve design: a retrospective and prospective synthesis. Ecol. Appl. 13, S47–S64.
Goñi, R., Adlerstein, S., Alvarez-Berastegui, D., Forcada, A., Reñones, O., Criquet, G., Polti, S., Cadiou, G., Valle, C., Lenfant, P., Bonhomme, P., Pérez-Ruzafa, A., Sánchez-Lizaso, J.L., García-Charton, J.A., Bernard, G., Stelzenmüller, V., Planes, S., 2008. Spillover from six western Mediterranean marine protected areas: evidence from artisanal fisheries. Mar. Ecol. Prog. Ser. 366, 159–174.
Goñi, R., Hilborn, R., Díaz, D., Mallol, S., Adlerstein, S., 2010. Net contribution of spillover from a marine reserve to fishery catches. Mar. Ecol. Prog. Ser. 400, 233–243.
Grüss, A., Kaplan, D.M., Guénette, S., Roberts, C.M., Botsford, L.W., 2011. Consequences of adult and juvenile movement for marine protected areas. Biol. Conserv. 144, 692–702.

Guidetti, P., Milazzo, M., Bussotti, S., Molinari, A., Murenu, M., Pais, A., Spanò, N., Balzano, R., Agardy, T., Boero, F., Carrada, G., Cattaneo-Vietti, R., Cau, A., Chemello, R., Greco, S., Manganaro, A., Notarbartolo di Sciara, G., Russo, G.F., Tunesi, L., 2008. Italian marine reserve effectiveness: does enforcement matter. Biol. Conserv. 141, 699–709.
Halpern, B.S., 2003. The impact of marine reserves: do reserves work and does reserve size matter? Ecol. Appl. 13, S117–S137.
Halpern, B.S., Lester, S.E., Kellner, J.B., 2010. Spillover from marine reserves and the replenishment of fished stocks. Environ. Conserv. 36, 268–276.
Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, R., Heinemann, D., Lenihan, H.S., Madin, E.M., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R., Watson, R., 2008. A global map of human impact on marine ecosystems. Science 319, 948–952.
Harmelin-Vivien, M., Ledireach, L., Baylesempere, J., Charbonnel, E., Garciacharton, J., Ody, D., Perezruzafa, A., Renones, O., Sanchezjerez, P., Valle, C., 2008. Gradients of abundance and biomass across reserve boundaries in six Mediterranean marine protected areas: evidence of fish spillover. Biol. Conserv. 141, 1829–1839.
Harrison, H.B., Williamson, D.H., Evans, R.D., Almany, G.R., Thorrold, S.R., Russ, G.R., Feldheim, K.A., van Herwerden, L., Planes, S., Srinivasan, M., Berumen, M.L., Jones, G.P., 2012. Larval export from marine reserves and the recruitment benefit for fish and fisheries. Curr. Biol. 22, 1023–1028.
Henriques, M., Gonçalves, E.J., Almada, V.C., 1999. The conservation of littoral fish communities: a case study at Arrábida coast (Portugal). In: Almada, V.C., Oliveira, R., Gonçalves, E.J. (Eds.), Behaviour and Conservation of Littoral Fishes. ISPA, Lisboa, Portugal, pp. 473–513.
Horta e Costa, B., Batista, M.I., Goncalves, L., Erzini, K., Caselle, J.E., Cabral, H.N., Goncalves, E.J., 2013a. Fishers' behaviour in response to the implementation of a marine protected area. PloS ONE 8, e65057.
Horta e Costa, B., Erzini, K., Caselle, J.E., Folhas, H., Gonçalves, E.J., 2013b. 'Reserve effect' within a temperate marine protected area in the north-eastern Atlantic (Arrábida Marine Park, Portugal). Mar. Ecol. Prog. Ser. 481, 11–24.
Horta e Costa, B., Gonçalves, L., Gonçalves, E.J., 2013c. site fidelity and spatio-temporal distribution of artisanal fisheries before the implementation of a temperate multiple-use marine protected area. Fish. Res. 148, 27–37.
Jennings, S., Kaiser, M.J., 1998. The effects of fishing on marine ecosystems. Adv. Mar. Biol. 34, 201–352.
Koueta, N., Boucaud-Camou, E., 2003. Combined effects of photoperiod and feeding frequency on survival and growth of juvenile cuttlefish *Sepia officinalis* L. in experimental rearing. J. Exp. Mar. Biol. Ecol. 296, 215–226.
Lédée, E.J.I., Sutton, S.G., Tobin, R.C., De Freitas, D.M., 2012. Responses and adaptation strategies of commercial and charter fishers to zoning changes in the Great Barrier Reef Marine Park. Mar. Policy 36, 226–234.
Leleu, K., Pelletier, D., Charbonnel, E., Letourneur, Y., Alban, F., Bachet, F., Boudouresque, C.F., 2014. Métiers, effort and catches of a Mediterranean small-scale coastal fishery: the case of the Côte Bleue Marine Park. Fish. Res. 154, 93–101.
Lescrauwaet, A.-K., Torreele, E., Vincx, M., Polet, H., Mees, J., 2013. Invisible catch: a century of bycatch and unreported removals in sea fisheries, Belgium 1929–2010. Fish. Res. 147, 161–174.
Lester, S.E., Halpern, B.S., 2008. Biological responses in marine no-take reserves versus partially protected areas. Mar. Ecol. Prog. Ser. 367, 49–56.
Lloret, J., Muñoz, M., Casadevall, M., 2012. Threats posed by artisanal fisheries to the reproduction of coastal fish species in a Mediterranean marine protected area. Estuar. Coast. Shelf Sci. 113, 133–140.
Lotze, H.K., Lenihan, H.S., Bourque, B.J., Bradbury, R.H., Cooke, R.G., Kay, M.C., Kidwell, S.M., Kirby, M.X., Peterson, C.H., Jackson, J.B., 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. Science 312, 1806–1809.
Lourenço, S., Moreno, A., Narciso, L., González, Á.F., Pereira, J., 2012. Seasonal trends of the reproductive cycle of *Octopus vulgaris* in two environmentally distinct coastal areas. Fish. Res. 127–128, 116–124.
Mascia, M.B., Claus, C.A., Naidoo, R., 2010. Impacts of marine protected areas on fishing communities. Conserv. Biol. 24, 1424–1429.
McClanahan, T., Davies, J., Maina, J., 2005. Factors influencing resource users and managers' perceptions towards marine protected area management in Kenya. Environ. Conserv. 32, 42–49.
McClanahan, T.R., Cinner, J.E., Abunge, C., Rabearisoa, A., Mahatante, P., Ramahatratra, F., Andrianarivelo, N., 2014. Perceived benefits of fisheries management restrictions in Madagascar. Ecol. Soc. 19.
Mesnildrey, L., Gascuel, D., Le Pape, O., 2013. Integrating marine protected areas in fisheries management systems: some criteria for ecological efficiency. Aquat. Living Resour. 26, 159–170.
Moreno, A., Lourenço, S., Pereira, J., Gaspar, M.B., Cabral, H.N., Pierce, G.J., Santos, A.M.P., 2014. Essential habitats for pre-recruit *Octopus vulgaris* along the Portuguese coast. Fish. Res. 152, 74–85.
Murawski, S., 2000. Definitions of overfishing from an ecosystem perspective. ICES J. Mar. Sci. 57, 649–658.
Murawski, S.A., 2007. Ten myths concerning ecosystem approaches to marine resource management. Mar. Policy 31, 681–690.
Nagelkerke, N., 1991. A note on a general definition of the coefficient of determination. Biometrika 78, 691–692.
Nagelkerken, I., Grol, M.G., Mumby, P.J., 2012. Effects of marine reserves versus nursery habitat availability on structure of reef fish communities. PloS ONE 7, e36906.
Nakagawa, S., Schielzeth, H., O'Hara, R.B., 2013. A general and simple method for obtaining R2 from generalized linear mixed-effects models. Method Ecol. Evol. 4, 133–142.
Neves, A., Cabral, H., Sequeira, V., Figueiredo, I., Moura, T., Gordo, L.S., 2009. Distribution patterns and reproduction of the cuttlefish, *Sepia officinalis* in the Sado estuary (Portugal). J. Mar. Biol. Assoc. UK 89, 579.
Palumbi, S.R., 2003. Population genetics, demographic connectivity, and the design of marine reserves. Ecol. Appl. 13, S146–S158.
Palumbi, S.R., Mcleod, K.L., Grunbaum, D., 2008. Ecosystems in action: lessons from marine ecology about recovery, resistance, and reversibility. Bioscience 58, 33–42.
Pauly, D., Ulman, A., Piroddi, C., Bultel, E., Coll, M., 2014. 'Reported' versus 'likely' fisheries catches of four Mediterranean countries. Sci. Mar. 78, 11–17.
Pilar-Fonseca, T., Campos, A., Pereira, J., Moreno, A., Lourenço, S., Afonso-Dias, M., 2014. Integration of fishery-dependent data sources in support of octopus spatial management. Mar. Policy 45, 69–75.
Pollnac, R.B., Crawford, B.R., Gorospe, M.L.G., 2001. Discovering factors that influence the success of community-based marine protected areas in the Visayas, Philippines. Ocean Coast. Manag. 44, 683–710.
R Development Core Team, 2012. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria www.R-project.org.
Rees, S.E., Rodwell, L.D., Searle, S., Bell, A., 2013. Identifying the issues and options for managing the social impacts of Marine Protected Areas on a small fishing community. Fish. Res. 146, 51–58.
Rice, J., Moksness, E., Attwood, C., Brown, S.K., Dahle, G., Gjerde, K.M., Grefsrud, E.S., Kenchington, R., Ring Kleiven, A., McConney, P., Ngoile, M.A.K., Næsje, T.F., Olsen, E., Olsen, E.M., Sanders, J., Sharma, C., Vestergaard, O., Westlund, L., 2012. The role of MPAs in reconciling fisheries management with conservation of biological diversity. Ocean Coast. Manag. 69, 217–230.
Roberts, C.M., Bohnsack, J.A., Gell, F., Hawkins, J.P., Goodridge, R., 2001. Effects of marine reserves on adjacent fisheries. Science 294, 1920–1923.
Rocklin, D., Tomasini, J.A., Culioli, J.M., Pelletier, D., Mouillot, D., 2011. Spearfishing regulation benefits artisanal fisheries: the ReGS indicator and its application to a multiple-use Mediterranean marine protected area. PloS ONE 6, e23820.
Roman, S., Jacobson, N., Cadrin, S.X., 2011. Assessing the reliability of fisher self-sampling programs. N. Am. J. Fish. Manag. 31, 165–175.
Ruckelshaus, M., Klinger, T., Knowlton, N., Demaster, D.R., 2008. Marine ecosystem-based management in practice: scientific, and governance challenges. Bioscience 58, 53–63.
Statsoft, 2013. Statistica 12. http://www.statsoft.com/
Tanner, S.E., Reis-Santos, P., Vasconcelos, R.P., Thorrold, S.R., Cabral, H.N., 2013. Population connectivity of *Solea solea* and *Solea senegalensis* over time. J. Sea Res. 76, 82–88.
Tsagarakis, K., Palialexis, A., Vassilopoulou, V., 2013. Mediterranean fishery discards: review of the existing knowledge. ICES J. Mar. Sci. 71, 1219–1234.
Tzanatos, E., Castro, J., Forcada, A., Matic-Skoko, S., Gaspar, M., Koutsikopoulos, C., 2013. A metier-sustainability-index (MSI25) to evaluate fisheries components: assessment of cases from data-poor fisheries from southern Europe. ICES J. Mar. Sci. 70, 78–98.
Vasconcelos, R.P., Reis-Santos, P., Costa, M.J., Cabral, H.N., 2011. Connectivity between estuaries and marine environment: Integrating metrics to assess estuarine nursery function. Ecol. Indic. 11, 1123–1133.
Walters, C.J., Hilborn, R., Parrish, R., 2007. An equilibrium model for predicting the efficacy of marine protected areas in coastal environments. Can. J. Fish. Aquat. Sci. 64, 1009–1018.