The effect of conservation measures on the spatial and temporal variation of rocky fish assemblages in the Arrábida Marine Park.

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TÍTULO DA TESE: O efeito das medidas de protecção na variação espacial e temporal das comunidades de peixes do recife rochoso do Parque Marinho da Arrábida.

Resumo:

As áreas marinhas protegidas (AMPs) têm sido amplamente propostas para cumprir objectivos de conservação e como ferramenta de gestão das pescas. O parque marinho da Arrábida é a primeira AMP em Portugal continental com um plano de gestão, completamente implementado desde 2009. O principal objectivo deste estudo foi a avaliação do efeito das medidas de protecção na comunidade de peixes de recife rochoso e invertebrados comerciais, comparando dados antes-depois e controlo-efeito (reserva *vs.* áreas de pesca) obtidos através de censos visuais subaquáticos e da análise dos padrões de descargas em lota. Em segundo lugar, amostrámos a actividade da pesca antes, durante e depois da implementação do plano de gestão com o objectivo de compreender as preferências dos pescadores pelas zonas de pesca e a sua adaptação às restrições. Também avaliámos o efeito reserva através da análise conjunta das respostas ecológicas e dos padrões de densidade do esforço de pesca. Em terceiro lugar, identificámos as principais variáveis oceanográficas que influenciam a estrutura da comunidade de peixes de recife e modelámos previsões para a estrutura dessa comunidade para os últimos 50 anos, à luz das alterações climáticas.

No geral, os resultados sugerem respostas positivas em biomassa, embora ainda não em números para algumas espécies comerciais, o que não aconteceu para as espécies não comerciais. O efeito reserva é reforçado pelo aumento das descargas em lota de espécies comerciais, apesar do aumento da densidade do esforço de pesca em algumas áreas, especialmente dos covos para captura de polvo. Os pesqueiros são principalmente escolhidos com base na distribuição das espécies comerciais e seus habitats associados, mas a distância ao porto, as condições do mar e a segurança também influenciam as escolhas dos pescadores. Além disso, as diferentes pescarias respondem de forma diferente às medidas de protecção, e dentro de cada arte de pesca, os pescadores mostram estratégias distintas, com alguns a operar numa área mais extensa enquanto outros mantêm territórios preferidos. Os nossos resultados mostram que o vento e a temperatura são as principais variáveis oceanográficas a influenciar a estrutura da comunidade de peixes do recife rochoso, que revela uma tropicalização associada a uma deslocação de espécies para norte nos últimos 50 anos, consistente com os padrões da temperatura. Acreditamos que este estudo apresenta importantes conclusões para a conservação marinha e gestão dos sistemas costeiros.

Palavras-chave: Áreas Marinhas Protegidas, pesca artesanal, peixes de recife rochoso, sistemas temperados, Portugal

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THESIS TITLE: The effect of conservation measures on the spatial and temporal variation of rocky fish assemblages in the Arrábida Marine Park.

Abstract:

Marine protected areas (MPAs) have been widely proposed for conservation purposes and as a tool for fisheries management. The Arrábida Marine Park is the first MPA in continental Portugal having a management plan, fully implemented since 2009. The main objective of this study was to evaluate the effect of protection measures on rocky reef fish assemblages and target invertebrates through before-after and control-effect (no-take *vs.* fished areas) underwater visual surveys and analysis of landings trends. Second, we used surveys before, during and after implementation of the management plan to understand fishers' preferences for fishing grounds and adaptation to the new rules, and evaluated the reserve effect through analysis of both ecological responses and fishing effort density. Third, we identified the main oceanographic drivers influencing the structure of reef fish assemblages and predicted the community structure for the last 50 years, in light of climatic change.

Overall results suggest positive responses in biomass but not yet in numbers of some commercial species, with no effects on non-target species. The reserve effect is reinforced by the increase in landings of commercial species, despite increased fishing effort density in some areas, especially with octopus traps. Fishing grounds are mainly chosen based on the distribution of target species and associated habitats, but distance to port, weather conditions and safety also influence fishers' choices. Moreover, different fisheries respond differently to the protection measures, and within each fishery, individual fishers show distinct strategies, with some operating in a broader area whereas others keep preferred territories. Our results also show that wind stress and temperature are the main oceanographic drivers for rocky reef fish assemblages, with tropicalization of assemblages and polewards movements of species over the last 50 years consistent with temperature trends. We believe this study provides significant lessons for marine conservation and management of coastal systems.

Keywords: Marine Protected Areas, artisanal fisheries, rocky reef fish assemblages, temperate systems, Portugal

Resumo alargado:

As áreas marinhas protegidas (AMPs) têm sido amplamente propostas para cumprir objectivos de conservação e como ferramentas de gestão das pescas. Diversos estudos sugerem o seu potencial para a protecção dos recursos e dos ecossistemas bem como podendo contribuir para a exportação de adultos e juvenis para as zonas adjacentes, beneficiando a pesca. No entanto, os efeitos da protecção dependem do contexto ecológico e socioeconómico local bem como dos objectivos da criação das AMPs.

O parque marinho da Arrábida é a primeira AMP em Portugal continental com um plano de gestão que inclui diferentes medidas de protecção, tendo sido aprovado em 2005 e está completamente implementado desde 2009. Os objectivos da criação desta AMP incluem a conservação dos ecossistemas e das espécies bem como a sustentabilidade da pesca artesanal local, que tem uma grande tradição cultural na zona. Esta área possui uma grande variedade de espécies e habitats e situa-se numa zona de transição biogeográfica entre águas temperadas quentes e frias, o que aumenta a sua biodiversidade, mas também faz com que haja uma variabilidade inter-anual na estrutura da comunidade de espécies em relação às condições climáticas. O principal objectivo deste estudo foi a avaliação do efeito das medidas de protecção na comunidade de peixes de recife rochoso e invertebrados comerciais, comparando dados antes-depois (dentro da reserva) e controlo-efeito (reserva *vs.* áreas de pesca) obtidos através de censos visuais subaquáticos. A caracterização dos habitats também foi realizada para poder detectar possíveis diferenças entre zonas com diferentes medidas de protecção, o que pode influenciar as diferenças ecológicas entre essas zonas. Foram realizados modelos lineares generalizados, *bootstraps*, e análises multivariadas. Foram analisados os padrões de descargas em lota também de forma a detectar os possíveis efeitos da reserva em algumas espécies com interesse comercial. Em segundo lugar, amostrámos a distribuição da actividade da pesca antes, durante e depois da implementação do plano de gestão com o objectivo de compreender as preferências dos pescadores pelas zonas de pesca e a sua adaptação às novas regras, que foram sendo sucessivamente implementadas desde 2005 até 2009. Foram usadas diferentes ferramentas da estatística espacial, algumas ferramentas de estudo de territórios, adaptadas de investigações de comportamento animal e utilização de habitat, bem como modelos aditivos generalizados e modelos com inflação de zeros. No final deste trabalho avaliámos o efeito reserva através da análise conjunta dos padrões de respostas ecológicas e de densidade do esforço de pesca. Um terceiro objectivo consistiu em identificar as principais variáveis oceanográficas que podem influenciar a estrutura da comunidade de peixes de recife

recorrendo a dados de presença/ausência de espécies obtidos através de censos visuais durante 12 anos, a dados de resolução local e regional de várias variáveis oceanográficas e a modelos de árvores de regressão multivariada. Através do melhor modelo descritivo seleccionado, realizámos modelação preditiva para a estrutura dessa comunidade para os últimos 50 anos, à luz das alterações climáticas.

No geral, os resultados sugerem respostas positivas em biomassa, embora ainda não em números para algumas das espécies comerciais, que aparentemente não se devem a diferenças no habitat entre zonas, o que não aconteceu para as espécies não comerciais. O efeito reserva é reforçado pelo aumento das descargas em lota de espécies comerciais, apesar do aumento da densidade do esforço de pesca em algumas áreas, especialmente dos covos para captura de polvo. Os pesqueiros são principalmente escolhidos com base na distribuição das espécies comerciais e seus habitats associados, mas a distância ao porto, as condições do mar e a segurança também influenciam as escolhas dos pescadores. Além disso, as diferentes pescarias respondem de forma diferente às medidas de protecção, e dentro de cada arte de pesca os pescadores mostram estratégias distintas, com alguns a operar numa área mais extensa enquanto outros mantêm territórios preferidos. Alguns pescadores a operar com covos também parecem ser influenciados pela proximidade à área de protecção total. Os nossos resultados mostram que o vento e a temperatura são as principais variáveis oceanográficas a influenciar a estrutura da comunidade de peixes do recife rochoso, que revela uma tropicalização associada a uma deslocação de espécies para norte nos últimos 50 anos, o que é consistente com os padrões da temperatura. Acreditamos que este estudo apresenta importantes conclusões para a conservação marinha e gestão dos sistemas costeiros.

Palavras-chave: Áreas Marinhas Protegidas, pesca artesanal, peixes de recife rochoso, sistemas temperados, Portugal

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CHAPTER 1: GENERAL INTRODUCTION

1.1 The role of marine conservation

Over the past decades, densely populated coastal areas have been strongly affected by high anthropogenic pressures, compromising their ability to maintain ocean health and to provide goods and services. With marine biodiversity and marine ecosystem functions increasingly being lost due to human-induced impacts, the ecological role of no-take areas in mitigating or reducing such impacts is widely recognized (Lotze *et al.*, 2006; Worm *et al.*, 2006; Mora, 2008). These fully protected areas, also called marine reserves, are a special class of Marine Protected Areas (MPAs), where extractive or destructive activities are forbidden (Lubchenco *et al.*, 2003). Therefore, marine reserves are increasingly proposed as a tool for ecosystem-based management, since they provide a spatial refuge for all species and habitats inside their borders and have the potential to promote socio-economic benefits in surrounding areas (Halpern *et al.*, 2010; Fraschetti *et al.*, 2011a; Fraschetti *et al.*, 2011b). However, social acceptance of no-take zones is usually difficult to accomplish and therefore multiple-use MPAs are being increasingly preferred as they encompass a variety of protection measures (including no-take areas) and uses, increasing stakeholders' acceptance and socio-economic benefits while still addressing, depending on a number of factors (e.g. size, enforcement, location, types of impacts and uses), the recovery of exploited populations, habitats and ecosystems (Claudet *et al.*, 2006b; Parsons and Thur, 2007).

Challenges concerning the conservation of marine systems go, however, beyond MPAs. Terrestrial pollution from domestic, industry and agriculture sewages, untreated waste, leakage of oils from marine platforms or tank vessels, dams, destructive fisheries, invasive species, climate change, among others human-induced impacts may severely affect marine systems. Therefore, marine conservation should be integrated in a broader management strategy, in which MPAs have an important role, but that also includes an efficient management of marine and terrestrial activities, a shift towards green energy and into a more local and less consumer society.

While MPAs alone cannot address widespread impacts such as pollution, climate change or overfishing (Halpern *et al.*, 2010), they may contribute to local and regional marine conservation and management and potentially increase ecosystem resilience in the face of global changes, especially if a network of connected MPAs exists (Munday *et al.*, 2008; McLeod *et al.*, 2009).

1.2 The science of marine protected areas

The science of marine reserves (no-take zones) has developed rapidly over the last three decades, with a vast amount of literature showing significant benefits of protection measures (Russ, 2002; Gell and Roberts, 2003; Micheli *et al.*, 2004; Claudet *et al.*, 2008; Lester *et al.*, 2009). The expected effects from the exclusion or reduction of extractive activities within MPAs – the so-called “reserve effect” – are an increase in biodiversity, abundance, size and biomass of marine species, especially those that are highly commercial and fished (Mosquera *et al.*, 2000; Russ, 2002; Tetreault and Ambrose, 2007; Lester *et al.*, 2009). Since protection measures attempt to recover resources by allowing a sufficient fraction of the population to grow, reach maturity and reproduce, the fecundity and subsequent total production are also likely to increase exponentially (Tetreault and Ambrose, 2007; Fraschetti *et al.*, 2011a). Sedentary species are expected to show larger responses to protection, since even small reserves may include most of their home ranges (Kramer and Chapman, 1999), but demersal mobile species forming ‘facultative schools’ also demonstrate positive responses to protection (Claudet *et al.*, 2010).

Depending on a number of factors such as mobility and other fish traits (Botsford *et al.*, 2003; Claudet *et al.*, 2010), density-dependent mechanisms and carrying capacity of protected and adjacent areas (Abesamis and Russ, 2005), as well as connectivity of suitable habitats (Forcada *et al.*, 2008; Vega Fernández *et al.*, 2008), the “reserve effect” may also include biomass export of post-settlers to the surroundings (spillover) (Kramer and Chapman, 1999; Roberts *et al.*, 2001; Russ, 2002), which may create fishing opportunities in adjacent areas. The magnitude and extent of spillover depends on the presence of contiguous suitable habitats further away from the reserve and on the mobility and site fidelity of species (Lowe *et al.*, 2003; Forcada *et al.*, 2009). Despite this, some authors have also suggested that fisheries are more likely to benefit through larval export from reserves to surrounding areas due to an increase in size and fecundity of adults inside the reserve (Russ, 2002; Tetreault and Ambrose, 2007), although these benefits have been much more difficult to detect (Goñi *et al.*, 2010; Pelc *et al.*, 2010).

Indirect effects may also occur and affect nearby areas after some time due to the build-up of top-predators and subsequent trophic cascades inside no-take areas (Micheli *et al.*, 2004; Guidetti and Sala, 2007). In fact, some studies showed considerably different compositions of reef assemblages among protected and fished areas due to distinct predator-prey abundances (Micheli *et al.*, 2005; Guidetti, 2006; Ling *et al.*, 2009).

The exclusion of destructive fishing gears, such as trawls, or physical damage (such as anchoring) can also lead to habitat recovery since those activities have the potential to cause severe habitat degradation. Therefore, the taxonomic and functional diversity of habitat-forming species and of higher levels of the food web may improve with protection measures. This way, the recovery of impacted populations and the subsequent shifts in marine communities contribute to an increased ecological resilience (Ling *et al.*, 2009).

These conservation and fisheries benefits vary in magnitude depending on the design and age of the reserve but also on species and habitat characteristics within and surrounding no-take areas. Although in the meta-analysis of Guidetti and Sala (2007), reserve age was not related to species' responses (only to functional groups), more recent studies found significant differences with areas protected for more than 3 years responding more strongly than younger ones (Halpern and Warner, 2002; Claudet *et al.*, 2006a; Claudet *et al.*, 2008). This is not unexpected given that the recovery of exploited populations takes time and depends on particular environmental and ecological conditions. In fact, size and biomass of individuals probably respond faster than density which, in some cases is dependent on increases in size, fecundity, production and recruitment (White and Caselle, 2008; Borges *et al.*, 2009; Taylor and McIlwain, 2010). Enhanced recruitment is, in turn, also dependent on interannual oceanographic and climatic fluctuations (Perry *et al.*, 2005; Munday *et al.*, 2008). Moreover, the recruitment success is influenced by density-dependent processes which may vary within and across species (Hixon and Webster, 2002; Goodwin *et al.*, 2006).

Although reserve size could be expected to influence the magnitude of effects and the range of species showing positive benefits, some empirical studies did not find a size effect (reviewed in Côté *et al.*, 2001; Halpern, 2003; Guidetti and Sala, 2007), possibly due to the relatively small size of the reserves, to the more sedentary target species studied (Lester *et al.*, 2009), or even to the wide variety of MPAs across different ecosystems. However, a more recent review, studying European MPAs concluded that protection performance is size-dependent (Claudet *et al.*, 2008). Very small reserves may fail to encompass all essential habitats and species' home ranges, leading to only a partial protection of the daily or seasonal range of most targeted species (Kramer and Chapman, 1999). Despite this, even small reserves can cause large increases in individual size, especially for less mobile commercial species (Claudet *et al.*, 2008; Lester *et al.*, 2009). The optimal size depends on the reserve goals and species considered. For example, in some modelling studies, reserves that were larger than a particular size did not add any subsequent increase in fisheries yield, but conservation goals were better achieved by the largest area. In fact, there has been a large debate about whether to establish several small reserves or single large ones (Single Large or Several Small: SLOSS) leading to a number of theoretical studies about this topic (Gerber *et al.*, 2003; Hastings and Botsford, 2003). Empirical evidence about what is the best design is challenging to gather in marine systems but by evaluating tradeoffs among multiple goals in modelling frameworks, optimization may be possible.

Until recently, different responses were hypothesized to occur between tropical and temperate MPAs. Due to the generally greater mobility patterns and to theoretically longer larval durations of temperate species, and thus higher rates of adult movement and larvae dispersal, temperate MPAs were suggested to have less reserve benefits for exploited species than tropical ones or would have to be much larger (Shipp, 2003; Floeter *et al.*, 2004; Kaiser, 2004; Blyth-Skyrme *et al.*, 2006; Laurel and Bradbury, 2006; O'Connor *et al.*, 2007; but see Leis *et al.*, 2013 for tests of some of these assumptions). However, a recent study analyzing several MPAs around the globe found similar if not greater reserve effects in temperate systems, at least for reefs (Lester *et al.*, 2009).

Despite growing evidence of the ecological benefits of no-take areas around the globe, other types of MPAs may not provide similar responses. Some authors found that partially protected areas (PPAs) do not have positive effects, since they do not decrease mortality by fishing and may attract users, especially displaced fishers, to the surroundings of closed areas due to the expected benefits they may supply (Shears *et al.*, 2006; Stelzenmüller *et al.*, 2007; Claudet *et al.*, 2008; Lester and Halpern, 2008). In fact, even a moderate fishing effort may reduce a significant proportion of larger individuals (Di Franco *et al.*, 2009). Despite the greater benefits of no-take areas, PPAs may still show some positive effects when compared to openly fished areas, for some heavily fished species (Floeter *et al.*, 2006; Sciberras *et al.*, 2013). Moreover, no-take areas may generate stronger antagonism from local users, whereas PPAs may lead to a higher socio-economic acceptance, while also reducing economic costs associated with full protection (Lester and Halpern, 2008; Sciberras *et al.*, 2013). However, more research is needed to distinguish among the effects of the huge variability in design and allowed uses of PPAs in order to understand which aspects affect positively both the ecological and socio-economic responses.

An important factor influencing the effectiveness of MPAs is related to the attraction of fishers to the reserve border. Some studies showed that expectations of benefits or perceived spillover lead to an increase of fishing effort adjacent to no-take borders (‘fishing the line’, Kellner *et al.*, 2007). Moreover, even if perceived benefits are not the reason, displaced fishers may prefer to operate as close as possible to former fishing grounds, resulting in increased fishing effort in the reserves surroundings (Lédée *et al.*, 2012). In both situations, spillover may then be compromised by higher densities of fishing effort in particular areas, with its magnitude and extent depending on gear selectivity and the target species ability to avoid such gears. If gears are very selective and their density is too high in the surroundings of reserves, repercussions on target species may extend inside the no-take areas, compromising their conservation goals (Goñi *et al.*, 2008). Spillover is usually interpreted from a gradient of abundance or biomass from the centre of a no-take zone to further away outside its borders (Francini-Filho and Moura, 2008; Goñi *et al.*, 2010). Since such gradient can be an artifact of a large increase of fishing effort in the surroundings of the reserve, spillover should only be assumed if there is evidence of a biomass build up inside the reserve after protection (Wilcox and Pomeroy, 2003; Goñi *et al.*, 2010), and net movements of species across the boundaries of reserves (Abesamis and Russ, 2005; Amargós *et al.*, 2010).

In addition, the appearance of a reserve effect may also be detected if reserves are placed in better habitats or areas with higher diversity or species abundances. Indeed, many reserves were chosen to protect such areas from human activities without assessing the extent of pre-existing differences between reserves and open areas (Roberts, 2000). Although a recent meta-analysis found that generally the reserve effect was not due to reserves being sited in better habitats (Lester *et al.*, 2009), data from inside and outside reserves before and after they were established are crucial to accurately evaluate protection benefits.

Besides the importance of ecological baseline information to distinguish natural habitat variability between no-take and fished areas (Claudet *et al.*, 2006a; Lester *et al.*, 2009; Huntington *et al.*, 2010), socio-economic data also have an important role in assessing the effectiveness of protection at different levels (Leleu *et al.*, 2012). Moreover, data about the distribution of fishing effort before protection is also essential to detect patterns of change within and surrounding MPAs in relation to the reserve effect (Abbott and Hayne, 2012; Campbell *et al.*, 2012).

To better understand the spatial allocation of fishing effort within and around MPAs, studies have assessed the importance of the proximity of fishing activity to MPA borders, water depth, home port or specific habitats (Goñi *et al.*, 2008; Stelzenmüller *et al.*, 2008; Forcada *et al.*, 2010) and they found that the distance to home port and to no-take borders were important for most fisheries. However most of these studies did not include data before protection and therefore it is difficult to have clear understanding of fishers' adaptations when displacement occurs. Only a few and recent empirical studies have analysed fishing effort distribution before and after protection (Murawski *et al.*, 2005; Abbott and Hayne, 2012; Campbell *et al.*, 2012; Lédée *et al.*, 2012), while to the best of our knowledge only one related fisheries displacement to ecological responses of fishing closures (Campbell *et al.*, 2012).

Environmental fluctuations can also influence the response of marine species to protected areas. In fact, regardless of the role of habitat complexity and diversity in shaping species assemblages, oceanographic or climatic variables drive the temporal variability of community structure since the presence and density of species can be strongly related to specific environmental conditions (Pörtner and Peck, 2010; Heath *et al.*, 2012). For example, in the face of steady global warming where more frequent extreme events may occur, species are predicted to move their ranges polewards, affecting existing marine assemblages and trophic interactions (Cheung *et al.*, 2009; Hawkins *et al.*, 2009; Figueira and Booth, 2010; Cheung *et al.*, 2012; Cheung *et al.*, 2013; Wernberg *et al.*, 2013). Additive or synergistic effects possibly arise with ocean warming and other multiple human activities, such as fisheries, leading to non-linear and unpredictable responses of species and ecosystems (Munday *et al.*, 2008; Griffith *et al.*, 2011). In this context, MPAs may contribute to increasing resilience of marine communities, buffering against some of these unpredicted effects (Ling *et al.*, 2009). The potential role of MPAs in mitigating the effect of ocean warming can be enhanced by a network of connected MPAs. Target species changing their distribution limits may achieve protection in connected MPAs (Munday *et al.*, 2008; McLeod *et al.*, 2009), through larval dispersal and/or adult migration if habitats are connected (Gruss *et al.*, 2011).

1.3 The model system

The Arrábida Marine Park (Portugal) also called the Luiz Saldanha Marine Park is an ideal model system to investigate many of the above mentioned topics since different initiatives over time allowed gathering ecological and fishing data before and after the implementation of the MPA management plan in 2005. This marine park is a multiple use-MPA located near important cities such as Lisbon and Setúbal, and has a small touristic fishing town, Sesimbra, at its centre. In this region human uses occur year round as the orientation of the majority of the park is towards the south and the adjacent high cliffs protect its waters from the prevailing north and north-west winds and waves. The main users are local commercial fishers, recreational fishers, divers and tourists taking advantage of scenic views, beaches and maritime recreational activities.

The park is located in an important biogeographic transition area along the Portuguese continental coast, resulting in exceptional levels of biodiversity. The park encompasses the northern and southern limits of warm and cold species, respectively, and also contains highly complex shallow rocky reefs and variety of other habitat types. The marine park was created to accomplish conservation goals as well as to guarantee fisheries benefits for small-scale local artisanal fishers.

The Arrábida Marine Park is adjacent to a terrestrial park (established in 1978), but only in 1998 was the marine portion designated a protected area. However, it remained without a management plan until 2005 when zoning and management measures were finally approved (Figure 1.1).

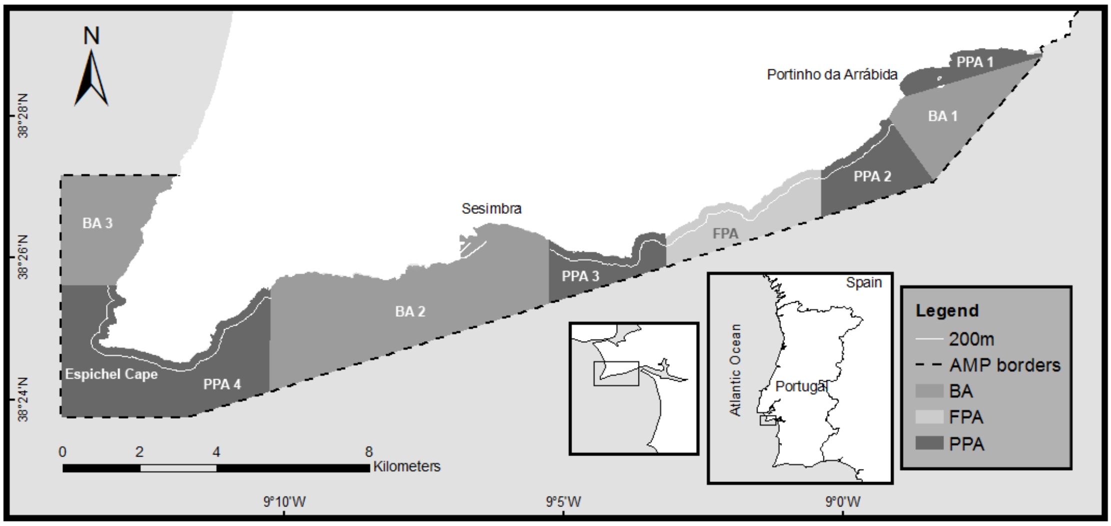

Figure 1.2 - Arrábida Marine Park zoning map. FPA: fully protected area; PPA 1 to 4: partially protected areas; BA 1 to 3: buffer areas.

Zoning (Figure 1.1) includes a fully-protected area (FPA) (4 km^2 in area); four partially-protected areas (PPA) (totalling 21 km^2); and two buffer areas (BA) (totalling 28 km^2). The FPA is a no-take, no-go area (with the exception of research, monitoring and education purposes). The PPAs (except PPA1 where no extractive uses are permitted) allow local commercial fishing with traps and lines but only beyond 200 m from shore and no extractive recreational activities (e.g. angling, spearfishing) are permitted. In the BAs, fishing vessels less than 7 m in length and recreational fishing activities are allowed. Commercial diving for bivalves or other marine organisms, spearfishing, trawling and purse seine nets are forbidden in the entire park. Only commercial fishers from Sesimbra are allowed to operate within the park .These licensed fishers have to report a minimum 100 sales per year to retain the license.

Since this is a region with historical artisanal fishing activity, a transition phase was included in the management plan for fisheries, in which the different zones were gradually implemented during the first four years. In 2006, management measures were enforced in the BAs, the east half of the current FPA began as a PPA and the Portinho PPA (PPA1) was implemented. In 2007, the remaining PPAs were implemented and the west half of the FPA started as a PPA. In 2008, the east half of the (current) FPA changed from PPA to FPA. The west half of the FPA was enforced in the summer of 2009, which ended the transition period.

The licensed fishing gears allowed to operate within the park are nets (trammel and gill nets) catching mainly soles, skates, demersal fish and cuttlefish; octopus traps that also capture some crustaceans during some months, longlines for fish and jigs for cuttlefish and squid. These gears are operated from small vessels, especially jigs and rod and reel angling, which are the main recreational fishing gears.

1.4 Dissertation goals

There are three overarching goals to this dissertation. The first one is the investigation of the ecological effects of protection measures on the rocky reef fish and invertebrate assemblages (**Chapter 2**). Before and after data as well as no-take and fished areas were compared in terms of fish and invertebrates density, size and biomass while accounting for differences in physical and biotic habitat among areas. Trends in landings of the most important commercial species (i.e. those most likely to contribute to a reserve effect) were also assessed from before and after protection using data from commercial fishers allowed to operate within the park.

The second objective is to understand and describe fishers' preferences and fisheries displacement dynamics after the implementation of the management plan (**Chapter 3**). The comparison of the adaptations of licensed fishers in the selection of fishing zones before, during (transition phase) and after the implementation of protection measures allowed an assessment of fishers' behaviour, preferences and constraints while choosing available fishing grounds (**Chapter 3A**). This is one of the first empirical studies evaluating, through direct observations, spatial patterns of artisanal fisheries before, during and after the establishment of protection measures within a temperate MPA. Moreover, the location, home range and site fidelity of the most important gears and of individual fishers before zoning was implemented could help to detect the factors influencing their previous distribution (**Chapter 3B**).

The third goal of this study was to evaluate the most important oceanographic variables affecting interannual fluctuations of the rocky reef fish communities and to try to understand and predict the effect of ocean warming on the structure of such assemblages in this marine park (**Chapter 4**). Fish community structure from the past 12 years was related to oceanographic variables at local and regional scales and predictive modelling for the past 50 years was conducted to recognize past trends and predict future consequences of global climate change (**Chapter 4A**). Furthermore, the detection of the first occurrence of a vagrant tropical species in this temperate MPA located in at a biogeographic transition zone was also described. This record could also contribute to the evidence of the role of such areas as important climatic ‘barometers’ (**Chapter 4B**).

At the end of this dissertation, the main findings from each chapter were integrated and discussed considering the current state of knowledge (**Chapter 5**). To better understand the ecological responses of the most targeted species with apparent reserve effect, we combined the fishing trends with the information gathered about the redistribution of fishing effort density. The future challenges to both rocky fish assemblages and artisanal fisheries on them, were discussed in relation to scenarios of ocean warming in temperate biogeographic transition zones and in the context of connected areas and the existence of a network of MPAs.

1.5 List of manuscripts

Chapter	List of manuscripts
Chapter 2	Horta e Costa, B., Erzini, K., Caselle, J.E., Folhas, H., Gonçalves, E.J. 2013. The reserve effect within a temperate marine protected area in the north-eastern Atlantic (the Arrábida Marine Park, Portugal). *Marine Ecology Progress Series*, **481**: 11-24.
Chapter 3A	Horta e Costa, B., Batista, M. I., Gonçalves, L., Erzini, K., Caselle, J. E., Cabral, H. N. & Gonçalves, E. J. 2013a. Fishers' Behaviour in Response to the Implementation of a Marine Protected Area. *PLoS One*, **8**: e65057.
Chapter 3B	Horta e Costa, B., Gonçalves, L. & Gonçalves, E. J. *submitted*. Vessels' site fidelity and spatio-temporal distribution of artisanal fisheries before the implementation of a temperate multiple-use marine protected area, *submitted to Fisheries Research.*
Chapter 4A	Horta e Costa, B., Assis, J., Franco, G., Caselle, J. E., Erzini, K., Henriques, M. & Gonçalves, E. J. *submitted*. Tropicalization of fish assemblages at temperate biogeographic transition zones. *submitted to MEPS.*
Chapter 4B	Horta e Costa, B. & Gonçalves, E. J. 2013. First occurrence of the Monrovia doctorfish Acanthurus monroviae (Perciformes: Acanthuridae) in European Atlantic waters. *Marine Biodiversity Records*, **6**: e20. Doi: 10.1017/s1755267213000055.

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CHAPTER 2: THE RESERVE EFFECT WITHIN A TEMPERATE MARINE PROTECTED AREA IN THE NORTH-EASTERN ATLANTIC (THE ARRÁBIDA MARINE PARK, PORTUGAL)

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2.1 Abstract

Marine protected areas have been widely studied and their global potential to recover marine resources and ecosystems has been largely confirmed. However, benefits of protection are dependent on local ecological and socio-cultural aspects which are critical to the success of the protection measures. In particular, patterns derived from before-after comparisons are indispensable if one wants to disentangle the effects of protection from those of different physical and ecological characteristics among areas. Here we assessed, using underwater visual surveys, whether biomass and abundance of temperate reef fish assemblages and target invertebrates increased inside a no-take area in the Arrábida Marine Park (Portugal) 3 to 4 after its establishment. Data were compared to a previous study, ten years before protection was effective. This is one of the few studies in temperate systems where before and after reserve effects were investigated. Habitat features were analysed as correlates of change and fishing effort trends were also compared. Control-effect comparisons after reserve establishment show that a positive response to protection was observed for legal-size demersal fish and below legal size target invertebrates. The first signs to protection were found in biomass but not in numbers, since only this variable showed significant higher values inside the reserve in legal-size fish. Non-target groups and below legal size demersal fish were found to have a significant interaction among reserve and habitat complexity indices for either density or biomass, suggesting a lack of a reserve effect. Before-after comparisons revealed non-significant patterns of increase in numbers of target species compared to non-target ones. The most important commercial species for fisheries operating within this Marine Park showed however the largest increase in density after protection was established. Significant higher abundances and proportionally heavier individuals of these species were also found inside the reserve in the control-impact comparisons. These findings are reinforced by an increasing trend on landings which are consistent with the early detection of a reserve effect.

Keywords: Marine Protected Area, before-after data, temperate reef fishes, target species, habitat, artisanal fisheries.

2.2 Introduction

A major goal of conservation and fisheries science is to restore exploited marine resources, habitats, ecosystems and biodiversity that have suffered human induced declines in abundance, genetic and functional diversity and altered food web structure (Claudet *et al.*, 2011b). Marine protected areas (MPAs) have been increasingly promoted as one of the tools for ecosystem-based management of marine systems (Fraschetti *et al.*, 2011) since they restrict or exclude human uses in some areas, and are often aimed at protecting whole communities and ecosystems. When a MPA or a particular zone within a MPA excludes extractive uses (becoming a no-take zone), it is called a marine reserve (Lubchenco *et al.*, 2003). Although some studies failed to show effects in marine reserves due to a range of different causes (Claudet *et al.*, 2011b), the majority of reserves have shown the so-called 'reserve effect' with increasing levels of biomass, density and size inside the reserve (Lester *et al.*, 2009). Effects are expected to be greater when fishing pressure is high before protection (Micheli *et al.*, 2004; Tetreault and Ambrose, 2007; Lester *et al.*, 2009) and the magnitude of those effects has been related to species composition, size, trophic level, mobility, habitat dependence, and commercial value (Pelletier *et al.*, 2008; Claudet *et al.*, 2010; Claudet *et al.*, 2011a; Claudet *et al.*, 2011b). Efficacy of a marine reserve also depends on effective enforcement and compliance by local users (Claudet and Guidetti, 2010), time since protection started (Micheli *et al.*, 2004; Di Franco *et al.*, 2009; Claudet *et al.*, 2010), size of no-take and adjacent buffer areas (Claudet *et al.*, 2008; Claudet and Guidetti, 2010; Claudet *et al.*, 2010), species-habitat interactions (García-Charton *et al.*, 2004; Pérez-Matus and Shima, 2010; Claudet *et al.*, 2011a) and connectivity with adjacent zones (Vega Fernández *et al.*, 2008).

The primary direct effect predicted by closures is the elimination of fisheries-related mortality allowing target individuals to live longer and produce more young. Moreover, when fishing gears impact habitat and are unselective, marine reserves safeguard habitat integrity, and increase fish density and size, also leading to an increase in fecundity and spawning biomass (Lester *et al.*, 2009). These effects might increase commercial stocks and benefits may be exported to adjacent areas through the migration of adults (Kramer and Chapman, 1999; Goñi *et al.*, 2008; Goñi *et al.*, 2010) and spillover of larvae (Pelc *et al.*, 2010). Indirectly, restoring a particular assemblage within a reserve may also affect predator-prey interactions and the dynamics of food webs, as larger target predators that had historically been caught will increase in abundance and size (Halpern, 2003; Claudet *et al.*, 2011b). However, the build-up of top predators is usually a slow process (Russ and Alcala, 2004; Hamilton *et al.*, 2010; Russ and Alcala, 2010).

Marine reserves can, in the long term, become control areas for the evaluation of population and ecosystem effects of fishing and other impacts on the marine environment. Coupling historical data from before the establishment of a reserve with data collected after its implementation, in which variables related to ecological changes in the assemblages (such as density, diversity, size, biomass) are monitored, may help to understand the sources of ecological variability at different scales, as well as the response of different systems to conservation and fisheries measures (Pelletier *et al.*, 2008).

To assess the effectiveness of marine protected areas, multiple designs can be used, but all rely on the comparison of a control site or time to an impacted situation (Osenberg *et al.*, 2011). In fact, misleading estimates of the effect of protection may arise when control-impact designs do not consider intrinsic habitat or other environmental features which may vary among nearby sites. In many situations, MPAs are likely sited in places with a higher ecological value and when assessing protection benefits this should not be disregarded (García-Charton and Pérez-Ruzafa, 1999; Côté *et al.*, 2001).

This study was performed at the Arrábida Marine Park in Portugal, which was designated in 1998. The management plan, approved in 2005, created different protection zones based on the natural values present (Gonçalves *et al.*, 2003), and included a transition phase for fisheries measures with the successive implementation of areas with different protection status until 2009. This marine park is one of the few situations where data collected before and after the implementation of the park can be compared. Our study aims to evaluate the reserve effect in this temperate marine protected area using before-after and control-impact comparisons of rocky reef fish assemblages, including commercially important invertebrate species. For that, we examined abundance and biomass responses to protection, comparing species that are targeted and not targeted by fishing before and after the implementation of the park and inside and outside the no-take areas. We also explored species-habitat interactions and fishing effort trends in order to account for such sources of variability.

2.3 Methods

2.3.1 Study area

The Arrábida Marine Park is a 38 km stretch of coastline (53 km^2) on the west coast of Portugal, adjacent to a terrestrial nature reserve created in 1976 – the Arrábida Nature Park. The marine park includes the rocky shore and adjacent mixed sandy substrata between Cape Espichel (38º27'N, 9º12'W) and Portinho da Arrábida (38º29'N, 8º57'W) (Figure 2.1). This area is utilized year-round for commercial and recreational activities as it faces south and is protected from the prevailing north-northwest winds and waves. Nearby are the cities of Lisboa and Setúbal, the later being an important fishing port located to the east of the park in the estuary of the Sado river. In the middle of the park the small touristic town Sesimbra has a long fishing tradition. This area is a biogeographic and oceanographic transition zone between warm and cold temperate waters and is also near the northern limit of the main north-east Atlantic upwelling events (Wooster *et al.*, 1976), which are stronger in the summer and increase the productivity of coastal waters. The intertidal zone is steep and subtidal rocky reefs are dominated by boulders originated from the erosion of the cliffs and by bedrock with fissures and crevices, creating a complex diversity of macro and microhabitats which supports a high diversity of algae, invertebrates and fish. These features make this area an important hotspot of diversity for this biogeographic region (Henriques *et al.*, 1999; Gonçalves *et al.*, 2003). North and south of the park extensive sandy shores prevail, making this marine park a 'continental island' for coastal species living on rocky reefs.

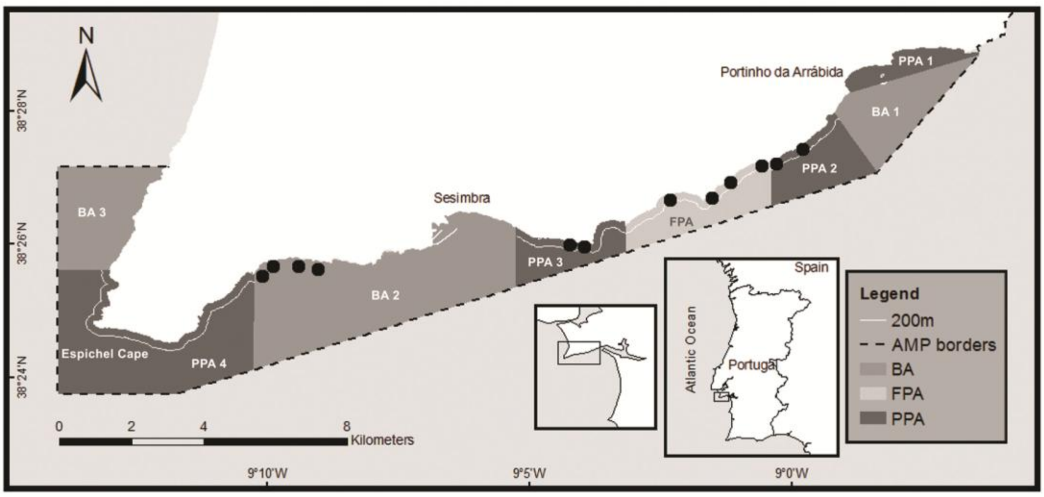

Figure 2.7 - Arrábida Marine Park zoning map. FPA: fully protected area; PPA 1 to 4: partially protected areas; BA 1 to 3: buffer areas. Black dots: survey sites (outside the reserve: four sites in BA2; inside the reserve: four sites in PPAs 2 and 3, and four sites in FPA).

The management plan of the Park was approved in 2005 and different areas with different levels of protection have been designated (Figure 2.1): a fully-protected area (FPA) (4 km^2 in area); four partially-protected areas (PPA) (totalling 21 km^2); and two buffer areas (BA) (totalling 28 km^2). The FPA is a no-take, no-go area (with the exception of research, monitoring and education purposes). The PPAs allow local commercial fishing with traps and lines but only beyond 200 m from shore and no extractive recreational activities (e.g. angling, spearfishing) are permitted. In the BAs, fishing vessels less than 7 m in length and recreational fishing are allowed. In the whole park, commercial diving for bivalves or other marine organisms, spearfishing, trawling and purse seine nets are forbidden. Commercial fishing licenses for the park were allocated only to fishers from Sesimbra.

The park's management plan was implemented with a transitional period for fisheries in which the different zones were gradually implemented during the first four years. In 2006, management measures were enforced in the BAs, the east half of the current FPA began as a PPA and the Portinho PPA (PPA1) was implemented. In 2007, the remaining PPAs were implemented and the west half of the FPA started as a PPA. In 2008, the east half of the (current) FPA changed from PPA to FPA. The west half of the FPA was enforced in the summer of 2009, which ended the transition period.

Since fishing is prohibited from the shoreline out to 200 m in the PPAs and the full extent of rocky reefs are encompassed within this range in most of the park (only on the west tip of the park do the reefs extend beyond this range), in this study we considered both the FPA and PPAs surrounding it as (no-take) reserves (PPA2 + PPA3 = 6.78 km^2 and FPA = 4.32 km^2, total = 11.1 km^2, Figure 2.1) and compared them to fishing areas (BA2 = 16.13 km^2). So, the outside reserve data were collected in the BA whereas the inside reserve aggregates data from the FPA and PPAs. Since no similar habitats are found adjacent to the park, the 'control' area is the BA.

2.3.2 Community survey methodology

During 2009 and 2010 three trained divers performed 65 m strip transects perpendicular to the coastline using underwater visual census in 12 fixed stations (Figure 2.1), containing a continuous rocky reef habitat. Four replicate transects were surveyed in each station in two field seasons each year (spring and autumn), totalling 192 replicates (48 per season). Due to a known seasonal variability in these coastal fish assemblages (Beldade *et al*., 2006; Claudet *et al.*, 2011a) we opted to survey the sampling sites twice a year. These surveys were done after the final stage of the management plan implementation and so this is considered the 'after' period in the analyses. The 'before' data were collected by EJG and colleagues (Gonçalves *et al.*, 2003, Henriques *et al.*, 2007, see data analysis below).

Each diver collected the following information for demersal and cryptobenthic fish and commercially important invertebrates: species, number, and estimated total length (or mean lengths for fish in schools). Demersal species were recorded in the first pass and at the end of the transect the diver turned back, moved two meters to the side and did another pass to record cryptobenthic fish and invertebrates. Several dive tests were conducted in the different stations before sampling initiated and the estimated modal length of rocky reefs was ≈ 65 m among sites. Thus, we used a 65 x 4 m strip transect (2 m on each side of the diver) for demersal fish species and a 65 x 1 m (0.5 m on each side of the diver) for cryptobenthic fish and invertebrate species. Data for both groups (demersal and cryptobenthic) were pooled together and no species were counted on both passes. Initial and ending depths and duration of each census were recorded. Visibility at each site was also recorded with a minimum of 5 m visibility established for the visual surveys. Densities refer to abundance in numbers per square meter. Lengths (L) were transformed to weights (W) using a L-W relationship for each species from published literature (using whenever possible primary references from the closest region or information from Froese and Pauly, 2011) and, if this was not available for a particular species, the closest congener was used as a proxy. Biomass was then calculated by multiplying abundance in number and individual weight.

Small juveniles (< 3cm) were not included to overcome inflated estimates of recent recruits, as is widely adopted in visual census studies (Bellwood and Alcala, 1988). The Mediterranean rainbow wrasse *Coris julis* was also not counted in surveys due to the very high abundance of this species. Unpublished data from the previous study showed that *C. julis* was ubiquitous and very abundant throughout the marine park, representing almost 1/3 of all encounters with abundances around 3 times higher than the second most abundant species. The accuracy of the visual census technique was incompatible with including this species and a similar procedure may be found in Colton and Alevizon (1981). Pelagic species (e.g. mackerel, sardines, bogue) were counted but not included in the analysis due to their high mobility and weak association with the rocky reefs. On the other hand, large commercial benthic fish with cryptic habits (e.g. Mediterranean moray eel, forkbeard, scorpionfish, Lusitanian toadfish) were very hard to detect and clearly underestimated due do their dependence on large refuges during daytime, and therefore these species were also excluded from the analysis.

Habitat data were gathered in 2009 (spring and autumn) and 2010 (spring) at all survey sites. Three transects were conducted with six 1 x 1 m quadrats in each transect, with one quadrat laid each 10 m from the deeper to the shallower zones of the transect. Each quadrat was divided in four 0.5 x 0.5 m areas. Divers collected detailed physical habitat categories that, for the purpose of this paper, were pooled together as shown in Table 2.1.

Table 2.6 - Physical habitat categories of the surveyed sites in Arrábida Marine Park, Portugal.

Physical habitats	Description
Sand	Grains < 0.2cm
Gravel	Grains between 0.2 - 5.0cm
Cobbles/pebbles	Small round rocks between 5.0 - 25.0cm
Boulders	Small (< 30.0cm), medium (30.0cm – 1.0m) and large rocks (> 1.0m), usually originated from the erosion of the high coastal cliffs
Bedrock	Rock adjacent and continuous to the coastal rock
Vertical rock	Vertical wall in an intertidal bedrock or a very large boulder
Holes	Small (opening < 30.0cm), medium (opening 30.0cm – 1m), large (opening > 1.0m)
Caves	Large and deep vertical holes (> 1m)
Overhangs	Oblique spaces below rocks generally boulders: small (opening < 30cm), medium (opening 30cm -1m)
Crevices	Narrow and thin spaces between rocks: small (length < 30cm), medium (length 30cm – 1m)
Fissures	Passages or corridors generally in the bedrock: medium (width < 30cm), large (width > 30cm)

Percent cover was calculated for each category at each site. Biotic habitat included algae and benthic invertebrates which were identified to species whenever possible but, due to the high diversity of species in the area and difficulties with *in situ* identification, algae were aggregated in functional groups (Steneck and Dethier, 1994) identifying whenever possible the most common genus or species (algae groups), and invertebrates were grouped by Phylum (Table 2.2). The intervals of percent biotic cover (the midpoint class was used for the analysis) were: A = <5%; B = [5-15%]; C = [15-25%]; D = [25-50%]; E = [50-75%]; F = [75-100%].

Table 2.7 - Biotic habitat (algae and invertebrates): algae functional groups and corresponding algae groups (algae species, genus or type); invertebrate groups (aggregated by Phylum) surveyed in the Arrábida Marine Park, Portugal.

Biotic Habitat	
Functional groups	**Algae groups**
Corticated macrophytes	*Asparagopsis sp.*, *Codium sp.* (erect or incrusting), *Cystoseira sp.*, *Halopteris sp.*, *Plocamium sp.*, *Rhodymenia pseudopalmata*, *Sargassum sp.*, *Sphaerococcus sp.*
Articulate calcareous algae	Erected coralline
Crustose algae	Encrusting coralline
Corticated foliose algae	*Dictyopteris polypodioides*, *Dictyota dichotoma*, *Padina pavonica*
Filamentous algae	*Falkenbergia rofulanosa*
Foliose algae	*Ulva sp.*
Leathery macrophytes	Laminaria n.id. (Phaeophyceae)
Invertebrates (Phylum)	
Annelida	
Arthropoda (*Balanus* sp.)	
Bryozoa	
Chordata (ascidean n. id.)	
Cnidaria	
Echinodermata (sea stars, sea urchins, sea cucumbers, ophiurideans)	
Echiura (*Bonellia viridis*)	
Mollusca	
Porifera	

2.3.3 Data Analysis

2.3.3.1 'Control – effect' comparison

The response of fish and target invertebrates to protection was compared based on the average response variables biomass (g.m^{-2}) and density (n.m^{-2}) of all sites inside the reserve (In) to the average response of all sites outside (Out), in the after period. In/Out ratios and their standard errors (SE) were calculated for the following groups of species which might respond differently to fisheries effects: non-target cryptobenthic fish, non-target demersal fish, target demersal fish and target invertebrates. For commercial fish, individuals larger than legal size (for those with size limits) were analysed separately from those below legal size. Species without legal size limits were included in the legal size target group. Ratios > 1 indicate higher density or biomass inside the reserve relative to outside, and the opposite is the case for ratios < 1.

To understand the role of protection while accounting for possible habitat differences, two non-collinear variables – roughness and boulder size diversity indices – were used as proxies for structural complexity. Shannon-Wiener's diversity index was calculated for the percent cover of the different sized boulders and cobbles/pebbles (Claudet *et al.*, 2011a). Roughness was estimated as the ratio between the length measured with a leaded cable contouring the bottom profile of the whole extension of the reef and the linear distance measured as the reef length perpendicular to the coast obtained by a geographic information system (GIS) shape file of rocky reefs provided by the marine park authority.

Statistical comparisons were performed using GLMs (generalized linear models) (McCullagh and Nelder, 1989; Dobson, 1990), testing the fixed effect 'reserve' and its interaction with habitat covariates – reserve * roughness + reserve * boulders diversity – for the response variables biomass and density of each group of species. Data from both seasons were pooled together to encompass intra-annual variability and to increase replication and statistical power. Choosing gamma as the exponential family and using a fourth-root transformation, the residuals showed good approximation with normality. Linear models were run to assess the reserve effect on roughness and boulders diversity indices (squared root transformed). After each model, ANOVAs were applied. These analyses were conducted using the open source statistical software R (version 2.12.2, R Core Team, 2012).

Responses of assemblage's biomass to the percent cover of different habitat features were assessed using BEST (BIO-ENV) routine in PRIMER 6.0 (Clarke and Warwick, 1994). This procedure searches for all possible combinations of environmental variables and selects the subset that best explains the multivariate pattern of fish assemblages. Moreover, it calculates a global BEST match permutation test (using 999 permutations) to evaluate significant associations between species groups' assemblages and the environmental variables.

Since habitat transects within each site could not be assigned to each species observations, comparisons were done at the site level using both percent cover and average biomass. Abiotic data (Table 2.1) were previously normalized and the resemblance matrix calculated using the Euclidean distance (procedure for environmental data). For algae and invertebrate groupings (Table 2.2), percent cover data was fourth-root transformed and the resemblance matrix was calculated using the Bray-Curtis similarity index. To account for the protection level, a dummy variable of '1' was assigned to sites inside the reserve and '0' for those outside (Forcada *et al.*, 2008). PERMANOVA with 999 permutations was also used to test for the effect of protection (fixed effect) on each habitat cover type (PRIMER 6.0).

In/Out response ratios and correspondent standard errors for biomass and density were also calculated for the most frequently observed species from each group. To test the significance of the obtained species ratios, the original ratio was compared with 9999 random In/Out ratios (bootstrap procedure) using the same number of In and Out observations but randomly permuting the response vector (biomass or density) at each 9999 replicates. Then, from the bootstrap results, a confidence interval (CI) was calculated and compared to the original species In/Out ratio, which was considered significantly different from random if it fell above or below the correspondent CI (R Core Team, 2012).

2.3.3.2 'Before – after' comparison

Underwater surveys of rocky reef assemblages were done by two divers in the autumn of 1998 and spring of 1999 using the same methodology described above in three sites common to both time periods. This period is referred hereafter as 'before'. Data from the same stations was used with two of the three sites being currently placed in the FPA and the third in an adjacent PPA. Transects in both periods were run perpendicular to the coast and the same groups of species were recorded in each direction (except target invertebrates which were not documented before). Although earlier surveys were based on timed counts and not on fixed transect length, dive tests with researchers from both periods ensured that survey procedures were identical. In the after period, transects noting time and distance were performed by the team members and speeds were kept constant and comparable with the before data set. In addition, in the 'after' period, the initial and ending time of each survey was always registered. Thus, in order to accurately compare both periods, analyses were based on time instead of area since this was the common metric to both data sets, and density was calculated as fish per minute (hereafter designated density.min^{-1}). Finally, the before surveys recorded fish length in categories (small, medium, large) so no comparisons of size structure or biomass were attempted.

Although there is data from before and after as well as in the reserve and fished areas, a before-after-control-impact (BACI) design could not be used to assess changes in these stations in relation to protection since all before data came from stations which were categorised as inside the reserve. Therefore separate before-after and control-impact (for after data) analysis were performed. The response of fish to protection was assessed by comparing the average density.min^{-1} in each site in the before and after periods (with seasons and years within each period pooled together). After/Before ratios and standard errors were calculated for the groups of species referred above (except for target invertebrates which were not surveyed before). For these comparisons, all sizes were used since before data did not provide enough detail to evaluate legal size limits. A GLM analysis was conducted to test the 'period' (i.e. before-after) fixed effect using the same procedures as described above for the GLMs from the 'after' data. Additionally, After/Before ratios and standard errors were also obtained to the most frequently observed species followed by a bootstrap and CI analysis, as explained above.

2.3.3.3 Landings data

To detect possible inter-annual trends for some target species, complementing the observations from visual census, we analysed landings data at the Sesimbra port using the available information for the years 1995 through 2009 since licences to fish within this marine park were all assigned to fishers from Sesimbra. To make sure that fishing effort was comparable among years, we followed landings only from vessels with active licenses for 2010 (n = 73 vessels). Moreover, these local and small vessels (< 7 m) maintained their gear licences and vessel capacity through time, fishing mainly close to port and within the marine park. Annual total landings (kg) by the number of active licensed vessels per year were calculated for each target species to allow inter-annual comparisons. Price per kg (in euros) by species or groups of species was also obtained. Data was provided by the General Directorate of Fisheries and Aquaculture (DGPA). Revenues (in euros) were calculated through landings and price per kg.

2.4 Results

2.4.1 Control – effect comparison

In the shallow rocky reefs of the Arrábida Marine Park, the groups analysed were composed of: 17 species (from 6 families) of non-target cryptobenthic species (NTCF), 14 species (from 5 families) of non-target demersal fish (NTDF), 24 species (from 13 families) of target demersal fish (TDF) and 7 species (from 7 families) of target invertebrates (TI; Table 2.S1).

Ratios of abundance and biomass revealed higher values inside than outside for most species groups (Figure 2.2, Table 2.S1). However, differences were larger for target species, especially for the response variable biomass in legal-size (LS) specimens, suggesting that commercial fish and invertebrates are larger inside the reserve. Although legal-size target invertebrates did not show significant higher values of biomass inside the reserve due to the large variability observed, target demersal fish revealed a very significant positive effect (Table 2.3). Both groups also showed a significant association between bottom roughness and biomass and between boulders diversity and density.

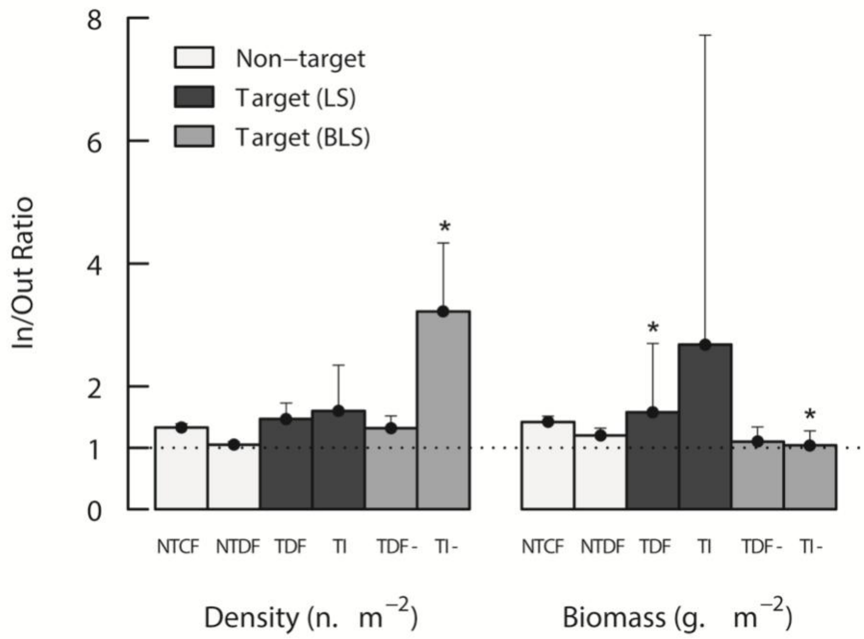

Figure 2.8 - Response In/Out reserve ratio ± SE for density (n.m^{-2}) and biomass (g.m^{-2}) of non-target (white bars), legal-size (black bars) and below legal size (grey bars; indicated with -) target groups; NTCF: Non-target cryptobenthic fish; NTDF: non-target demersal fish; TDF: target demersal fish; TI: target invertebrates. Ratios greater than 1 indicate that response variables are higher inside the reserve. *: significant ratios (GLM results).

On the other hand the two non-target groups did not show positive effects to protection since significant interactions were found between reserve and habitat complexity indices. Non-target demersal fish density varied with the interaction between reserve and both roughness and boulders diversity, whereas non-target cryptobenthic fish biomass and density showed a significant interaction between reserve and boulders diversity.

Table 2.8 - GLM results of reserve and habitat complexity indices effects (roughness and boulders diversity) on biomass ($g.m^{-2}$) and density ($n.m^{-2}$) of the four different species groups: NTCF: Non-target cryptobenthic fish; NTDF: Non-target demersal fish; TDF: Target demersal fish; TI: Target invertebrates; LS: legal size; BLS: below legal size. Significant *post-hoc* comparisons of factor reserve are indicated (highest values): (+) indicates significant reserve effects; significant p-values are shown in bold.

	Control – effect comparison									
	Reserve		Roughness		Boulders diversity		Reserve * Roughness		Reserve * Boulders diversity	
	Biomass	Density	Biomass	Density	Biomass	Density	Biomass	Density	Biomass	Density
NTCF	**<0.001**	**<0.001**	**0.002**	0.32	0.81	0.45	0.27	0.27	**0.003**	**0.001**
NTDF	0.16	0.12	0.36	0.60	0.38	**0.003**	0.66	**0.02**	0.57	**<0.001**
TDF (LS)	**<0.001** (+)	0.30	**0.006**	0.43	0.28	**<0.001**	0.38	0.60	0.96	0.96
TI (LS)	0.21	0.08	**0.012**	0.72	0.11	**0.007**	0.24	0.98	0.08	0.84
TDF (BLS)	0.28	**0.004**	**<0.001**	**0.002**	**0.036**	**0.005**	0.65	0.37	**0.006**	**<0.001**
TI (BLS)	**0.016** (+)	**<0.001** (+)	**0.004**	0.09	0.13	0.61	0.40	0.31	0.58	0.57

As opposite to the legal size target groups, below legal size (BLS) individuals for both demersal fish and invertebrates showed a larger response in density when compared to biomass (Figure 2.2). The density of BLS target demersal fish tended to be higher inside the reserve. However, both density and biomass showed a significant interaction between reserve and boulders diversity (Table 2.3). On the other hand, the density of BLS target invertebrates showed the largest In/Out ratio (Figure 2.2) and a positive significant effect of reserve. Biomass was also affected by reserve and roughness without an interaction of these factors (Table 2.3). Additionally, the relation between protection and habitat complexity indices revealed similar values for roughness ($p = 0.77$) but higher diversity of boulders inside the reserve ($p = 0.027$).

The relationship between habitat features and the different species groups was analysed for the variable biomass using protection level as a dummy variable (Table 2.4). Non-target cryptobenthic fish and legal-size target demersal fish revealed a significant correlation with algae cover whereas non-target demersal fish did not associate with any habitat type. On the other hand target invertebrates (LS) showed a dependence on invertebrates cover and a marginal non-significant relation with algae functional groups. For groups below legal size, no correlations with habitat features were detected. Interestingly, none of the groups showed a significant correlation with physical habitat although this was the only habitat variable with significant differences between reserve and fished locations (p = 0.007).

Table 2.9 - BEST (Bio-Env) results of correlation between habitat features (multivariate data) with a dummy-coded variable to protection and the biomass of the four species groups. NTCF: Non-target cryptobenthic fish; NTDF: Non-target demersal fish; TDF: Target demersal fish; TI: Target invertebrates; LS: legal size; BLS: below legal size. For physical and biotic habitats see Tables 2.1 and 2.2, respectively. Spearman correlation coefficient (Rho) is included; significant p-values are shown in bold.

	Control – effect comparison			
	Algae Groups	Algae Functional Groups	Invertebrates' phylum	Physical habitat
NTCF	**p = 0.01** (Rho=0.48)	**p = 0.01** (Rho=0.32)	p = 0.07 (Rho=0.28)	p = 0.34 (Rho=0.24)
NTDF	p = 0.63 (Rho=0.20)	p = 0.73 (Rho=0.10)	p = 0.43 (Rho=0.19)	p = 0.81 (Rho=0.17)
TDF (LS)	**p = 0.01** (Rho=0.39)	**p = 0.02** (Rho=0.38)	p = 0.09 (Rho=0.29)	p = 0.41 (Rho=0.24)
TI (LS)	p = 0.11 (Rho=0.31)	p = 0.054 (Rho=0.27)	**p = 0.016** (Rho=0.38)	p = 0.08 (Rho=0.41)
TDF (BLS)	p = 0.07 (R=0.28)	p = 0.18 (Rho=0.20)	p = 0.13 (Rho=0.24)	p = 0.60 (Rho=0.18)
TI (BLS)	p = 0.34 (Rho=0.28)	p = 0.44 (Rho=0.2)	p = 0.75 (Rho=0.17)	p = 0.51 (Rho=0.27)

Comparing the In/Out ratios for the most common species detected in the visual surveys, the following patterns in density (Figure 2.3a) and biomass (Figure 2.3b) emerge: i) in general, target species showed a higher variability, especially in biomass; ii) legal-size *Octopus vulgaris* tended to be more abundant and were significantly much larger inside the reserve, and below legal size individuals also showed significantly higher biomass and density values inside the reserve; iii) white seabreams (*Diplodus sargus*) of all sizes were significantly more abundant and larger inside the reserve, as was the target velvet crab (*Necora puber*); iv) the salema (*Sarpa salpa*), was more abundant and larger inside the reserve but showed a large variability and therefore In/Out ratios were not significant; v) the small cryptobenthic triplefin blenny (*Tripterygion delaisi*) was the non-target species with the largest significant reserve effect on both variables; vi) for the legal-size common two-banded seabream (*Diplodus vulgaris*), biomass and density were similar outside and inside, but small specimens were significantly more abundant inside; vii) Mugilidae did not show significant ratios, but nevertheless presented higher abundances outside but larger biomass inside; vii) the comber (*Serranus cabrilla*), which has no legal size limit, was the only species having a significant ratio below one (indicating higher values outside) both in density and biomass; ix) the non-target wrasse *Symphodus roissali* was significantly more abundant outside, but biomass values were similar between protected and fished areas.

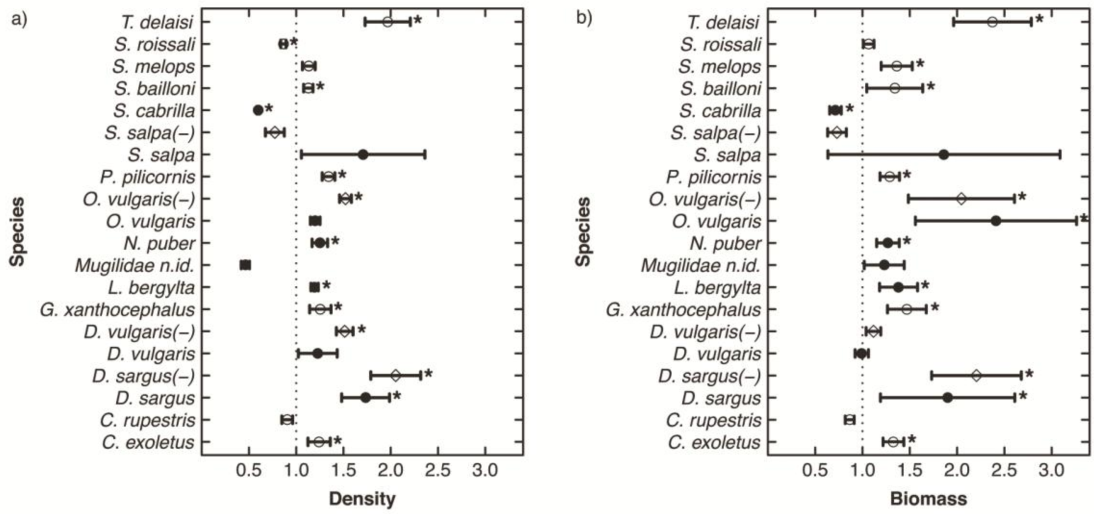

Figure 2.9 - Response In/Out reserve ratio ± SE for: (a) density ($n.m^{-2}$) and (b) biomass ($g.m^{-2}$) of the most frequently observed non-target (open circles), legal-size (black circles) and below legal size (open diamond; indicated with (-)) target species in the Arrábida Marine Park. Ratios greater than 1 indicate that response variables are higher inside reserve. Significant ratios are indicated with *.

2.4.2 Before – after comparison

The data collected in this study were compared to a previous work performed in the same area before the establishment of the marine park (see materials and methods). Target species were more abundant in the after period but no significant differences were found for any group (Figure 2.4).

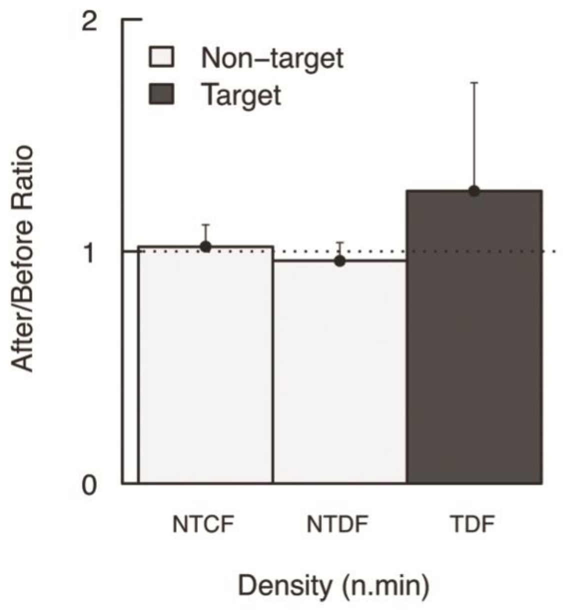

Figure 2.10 - Response ratio After/Before the implementation of the management plan ± SE for density ($n.min^{-1}$) of non-target (white bars) and target (black bar) groups; NTCF: Non-target cryptobenthic fish; NTDF: non-target demersal fish; TDF: target demersal fish. Ratios greater than 1 indicate that response variables are higher inside the reserve.

After/Before density ratios for the most frequently observed species (Figure 2.5) show that the only significant variation was for *Serranus cabrilla*, which was more abundant in the before period. In addition, all other below one ratios (albeit non-significant) were from non-target species. The sparids *Diplodus vulgaris*, *Diplodus sargus* and *Sarpa salpa* showed the largest variability but also the largest increase among periods.

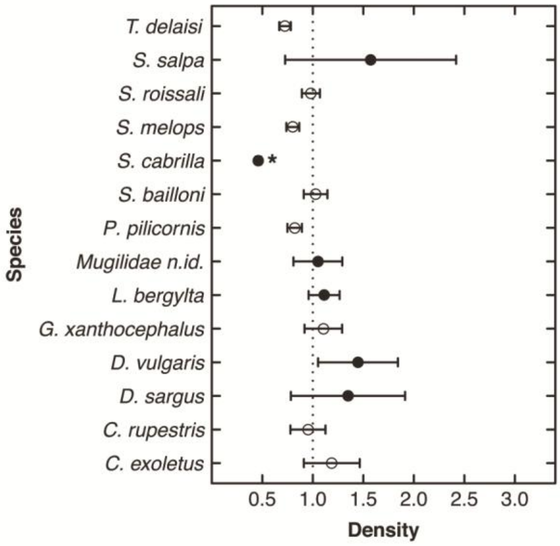

Figure 2.11 - Response ratio After/Before the implementation of the management plan ± SE for density ($n.min^{-1}$) of the most frequently observed non-target (open circles) and target (black circles) species in the Arrábida Marine Park. Ratios greater than 1 indicate that response variables are higher in the after period. Significant ratios are indicated with *.

2.4.3 Landings data

Octopus vulgaris was by far the most captured species from the local fishing fleet in the marine park (Table 2.5). The sparid *Diplodus sargus* was the most valuable species in price per kg. *Sarpa salpa*, mullets (Mugilidae n. id.) and wrasses (Labridae n.id.) are bycatch species with low market value. *Serranus cabrilla* being also a bycatch species with low market value is shown as a valuable species due to data aggregation in the official records with other more valuable subtropical serranids caught elsewhere (West African coast).

Table 2.10 - Mean annual landings (kg) and $price.kg^{-1}$ (euros) of the most important commercial species from vessels with a license to operate in the marine park in 2010. Means were calculated from 1995 to 2009. *D. sargus* price was obtained from *Diplodus* spp. and *S. salpa* price was obtained from Sparidae n. id. category. No price information was available for Mugilidae n. id. or Labridae n. id. (Source: General Directorate of Fisheries and Aquaculture, DGPA).

Species	Landings (kg)	$Price.kg^{-1}$ (euros)	Revenue (euros)
O. vulgaris	105679	4.2	443852
D. sargus	976	6.2	6052
D. vulgaris	576	4	2303
Mugilidae n. id.	306	-	-
S. salpa	188	0.9	169
Labridae n. id.	28	-	-
Serranidae n. id.	4.6	4.9	23

Landing patterns of park licensed vessels (Figure 2.6) showed a steady increase and great dependence of the local fisheries on octopus, with a significant increase in landings immediately before the Park was established (2004-2005) followed by a decrease (2006-2007) compensated with another increase in the most recent years (2008-2009). A decrease in captures of seabreams from 2003 to 2006 is also apparent, followed by an increase in landings until the last year with official statistics (before 2006, *Diplodus sargus* was mixed with other seabreams, but not with *Diplodus vulgaris*, in the category *Diplodus* spp. of which it represents the largest element, explaining the sharp decline of this group from 2007 on).

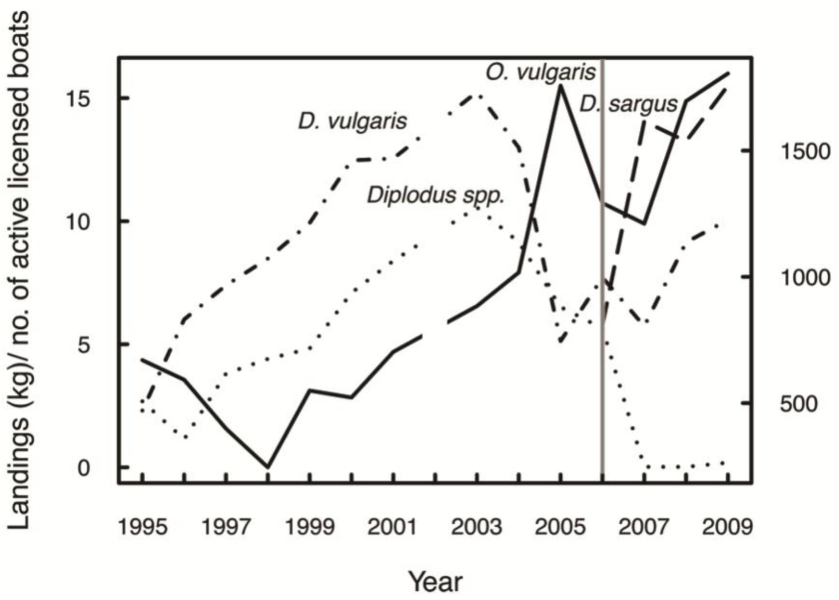

Figure 2.12 - Annual fisheries landings (kg) by the number of active licensed vessels of *Diplodus vulgaris, Diplodus sargus* and *Diplodus spp* (x axis). Landings of *Octopus vulgaris* (kg/no. of active licensed vessels) are in the right y axis. Data is from vessels with a license to fish in the park. No data are available for 2002 (Source: General Directorate of Fisheries and Aquaculture, DGPA). The vertical grey line in 2006 marks the start of the implementation of the management plan approved in August 2005.

2.5 Discussion

Marine protected areas have been widely studied and several reviews confirmed their global potential to recover marine resources and ecosystems (Halpern, 2003; Micheli *et al.*, 2004; Lester *et al.*, 2009; Hamilton *et al.*, 2010). However, a major challenge is disentangling the effects of protection from those of unique habitat characteristics (García-Charton and Pérez-Ruzafa, 1999). In fact, there is a general lack of before data which is particularly evident in temperate systems (but see Claudet *et al.*, 2006; Shears *et al.*, 2006). This is a potential confounding effect in the assessment of MPA effectiveness and the interpretation of positive responses since some differences between reserve and fished areas in, for example, habitat quality, could pre-date MPA implementation. In fact, it has been hypothesised that the location of several marine reserves is associated with particularly diverse habitats or assemblages and therefore the observed differences between fished and reserve areas could be due to the fact that reserves are better sited than fished zones (García-Charton and Pérez-Ruzafa, 1999; Côté *et al.*, 2001). Nevertheless, the observed reserve effect in many MPAs does not seem to be an artifact of reserves being sited in better locations (Lester *et al.*, 2009). The existence of data collected before the establishment of marine reserves is therefore central to disentangle habitat influences in the assemblages' composition from responses to management effects.

The present study addresses some of these shortcomings through a control-impact (inside-outside) and a before-after (inside only) comparison. Although outside data was not available in the before period, the evaluation of the reserve effect in the after period was complemented by a habitat assessment inside and outside the reserve and the analysis of landings trends of the artisanal fishing feet operating in the marine park. The small size of the studied reserve, the phasing-in of the management plan during the first four years of its implementation and the multiple activities impacting this marine region over time, led us to expect small differences, if any, between the reserve and fished areas of the Arrábida Marine Park.

Most species groups showed however higher density and biomass ratios inside the reserve. This raises the hypotheses that: i) the reserve is starting to produce effects which are widespread for all groups of species; ii) the reserve could be a zone with more favourable habitat features for local species; iii) there is an interaction between these two effects (more diverse habitats inside and a reserve effect) that could explain the observed trends. The largest effects were detected in the target species groups, with significant higher biomass of demersal fish and small invertebrates (below legal size) inside the reserve. These differences were not influenced by habitat complexity.

In fact, all groups showed a significant effect of either roughness or boulders diversity in relation to biomass, density or both. However, only small species such as those from the non-target cryptobenthic or below legal size demersal groups revealed a strong interaction between the reserve and boulders diversity in biomass and/or density, indicating a lack of reserve effect. When the response to the reserve was tested for complexity indices, only boulders diversity showed significantly higher values inside the reserve. This predictor did not influence the biomass of target groups. The greater variability of boulders sizes potentially increases the abundance of small refuges, which may affect cryptobenthic species such as gobies and blennies, by definition strongly dependent on habitat (Willis and Anderson, 2003). Additionally, despite the differences in the physical features of the habitat between reserve and fished sites, no correlation was found between any species groups' assemblages and these characteristics. Moreover, there were significant correlations with different biotic habitats for all species groups, but these did not vary with the reserve.

Contrary to what could be expected, target invertebrates did not reveal a significant response to protection, in spite of being the second group with the highest In/Out ratio for density and by far the highest ratio for biomass. The lack of a significant reserve effect in this group is probably related to the high variability associated to this ratio which suggests large differences between samples.

The analyses of the most frequently observed species support our hypothesis that the positive response to protection of target species is a first sign of the reserve effect since both the valuable seabreams (especially *Diplodus sargus*) and *Octopus vulgaris* (the most landed target species) showed proportionally the largest increase in biomass, suggesting larger individuals inside the reserve. It is particularly striking that *Diplodus sargus* showed such a strong response to protection since it is potentially a vagile species with a wide home range (Abecasis *et al.*, 2009; Lino *et al.*, 2009) and low habitat connectivity requirements, allowing them to cross large sandy areas (Vega Fernández *et al.*, 2008). They are therefore potentially vulnerable to fishing when they move out from the reserve. Further studies on the behaviour of this species throughout ontogeny as well as patterns of movement of individuals within the reserve habitats are needed to fully explain these results. Claudet *et al*. (2010) also found however that the effect of protection was as strong for mobile as for sedentary species and that this effect was enhanced for larger species that were not found in obligate schools (which is the case of seabreams). Di Franco *et al.* (2009) also found high densities of large fishes inside reserves which were attributed to a change in behaviour since fish seemed to avoid the reefs when they were intensively fished. Daily and seasonal movements can also be influenced by local social dynamics and fish social status (Afonso *et al.*, 2008) which together with attraction from conspecifics may increase the probability that certain vagile species will remain within the reserve, increasing competition but benefiting from a decrease of mortality from fishing (Claudet *et al.,* 2010).

Prior to the implementation of the marine reserve, the nearshore rocky reefs were intensively exploited by spearfishing and recreational angling due to the prevailing year-round calm seas in the park, its shallow rocky reefs and high habitat complexity (Gonçalves *et al.*, 2003). Recreational fishing has been shown to have large impacts on higher trophic levels and in particular on nearshore shallow ecosystems (Cooke and Cowx, 2004), since even moderate fishing effort performed continuously can remove a significant proportion of larger fishes (Di Franco *et al.*, 2009). This recreational fishery is mainly directed to large sparids and octopus (Rocklin *et al.*, 2011) which are also targeted by commercial fishing with hooks and lines, traps and jigs (Erzini *et al.*, 2008). The exclusion of these fishing pressures from the reserve may explain why these formerly intensively exploited species on the shallow rocky reefs showed the largest responses in biomass among different protection zones.

Comparisons between before and after periods in species patterns, suggest that almost all non-target species reduced or maintained their abundance whereas target species have an opposite trend towards an increase in density after the establishment of the marine park. However, due to the high variability in species responses, these differences were non-significant. Additionally, no sites were sampled in the current fished zones in the before period. For these reasons, these trends should be interpreted with caution. Several studies have demonstrated that time since protection is essential to detect reserve effects (Micheli *et al.*, 2004; Claudet *et al.*, 2008), especially for large and long-lived species since they require time to grow and reproduce.

In the non-target groups, a few species, especially *Tripterygion delaisi* and some wrasses, were more abundant inside than outside but showed a decreasing pattern in the reserve from before to after protection was established. Possible differences in habitat quality (with higher boulders diversity inside) could be an important factor leading to a higher abundance of *T. delaisi* inside and also supporting a higher rate of post-settlement survivorship in wrasses (Pérez-Matus and Shima, 2010). In fact, adult wrasses require a high level of connectivity among similar habitats to be able to migrate between different coastal zones since, unlike sparids, these species do not easily cross extensive areas of sand (Vega Fernández *et al.*, 2008). Another interesting result was that the only target species having a significant higher density and biomass outside the reserve (*Serranus cabrilla*) showed also a significant decrease in density between periods. The lack of a reserve effect for this species was also found in other studies (García-Rubies and Zabala, 1990). Nevertheless, further work is needed to explain its decrease in density.

In spite of the recent implementation of the marine park management plan, comparing areas inside and outside the reserve and the before and after periods showed that target species are responding positively to protection whereas non-target ones are not, and these responses are occurring in biomass but not yet in numbers. The positive response to protection of individuals' size and biomass have been described as early indicators of the reserve effect (Pelletier *et al.*, 2008; Di Franco *et al.*, 2009; Lester *et al.*, 2009), even only after a few years of closure, especially when fishing targets large individuals (Erzini *et al.*, 2006). Indeed, effects were found in other studies short time (2-4 years) after the establishment of the reserve (García-Charton *et al.*, 2004; Micheli *et al.*, 2005; Claudet *et al.*, 2008; Di Franco *et al.*, 2009).The increase in number take a longer time to become detectable as it depends on inter-annual biological and environmental conditions such as variability in recruitment patterns, changes in pre- and post-settlement mortality, larval dynamics and oceanographic features (García-Charton *et al.*, 2004). Additionally, although reserve size and age affects the magnitude of the response (Tetreault and Ambrose, 2007), effects in small reserves (similar in size to the present case) have shown large increases in average individual size (Claudet *et al.*, 2008; Lester *et al.*, 2009; Claudet *et al.*, 2010), particularly for intensively fished species (Micheli *et al.*, 2004).

While commonly lacking in MPA studies, information on spatial and temporal patterns of fishing effort relative to reserve placement and timing may be one of the most critical factors for interpreting patterns of change (Claudet *et al.*, 2008; Di Franco *et al.*, 2009). In fact, it has been shown that positive responses to protection may be influenced by an increase in fishing effort in adjacent fished areas due to displacement (Tetreault and Ambrose, 2007; Claudet and Guidetti, 2010). In this study we followed the same licensed vessels operating with a stable fishing capacity throughout time and interestingly, there was a steep increase in landings immediately before the implementation of management measures, especially for the most captured species (*O. vulgaris*), suggesting that fishers were concerned about the impacts of the impending reserve implementation and the loss of fishing grounds. There are two potential and non-mutually exclusive explanations for this increase. First, fishermen fished harder in the time leading up to reserve implementation to bolster revenues before a perceived loss. Second, reporting of catches increased in order to guarantee a renewal of the park fishing licence, which required a minimum of 100 sales per year. We know of no published cases where fishing pressure increased specifically in response to future reserve implementation but the ramifications to fisheries independent studies of reserves are potentially large. Fisheries data also revealed an increase of landings for commercial species targeted by the local artisanal fishing fleet after the marine park implementation, supporting the positive trend in response to protection, and reinforcing the evidence of a recover in size and possibly in numbers for these species. Both larger octopus and seabreams could perform movements possibly related to some degree of spillover which may explain this increase.

The inclusion of previous baseline data and landings information together with habitat influence and control-impact comparisons, discriminating commercial legal-size individuals from juveniles and small fish provided a stronger case for the detection of reserve effects even after just only 3-4 years since the establishment of protection.

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2.8 Supplementary Information

Table 2S.2 - Mean ± SE biomass and density (and respective minimum and maximum values), frequency of occurrence, number of species and families between inside vs. outside the reserve for each species group analysed: NTCF: Non-target cryptobenthic fish; NTDF: non-target demersal fish; TDF: target demersal fish; TI: target invertebrates; LS: legal size; BLS: below legal size.

Species Group	Mean Biom **In** ± SE (min - max)	Mean Biom **Out** ± SE (min - max)	Mean Dens **In** ± SE (min - max)	Mean Dens **Out** ± SE (min - max)	Freq **In**	Freq **Out**	No. Species	No. Families
NTCF	0.48 ± 0.01 (0.005 - 4.72)	0.34 ± 0.01 (0.006 - 3.83)	0.094 ± 0.002 (0.015 - 0.711)	0.071 ± 0.002 (0.015 - 0.785)	3399	1169	17	6
NTDF	0.3 ± 0.01 (0.001 - 6.87)	0.25 ± 0.01 (0.002 - 4.29)	0.018 ± 0.0004 (0.004 - 0.308)	0.017 ± 0.001 (0.004 - 0.25)	3198	1334	14	5
TDF (LS)	4.3 ± 0.24 (0.001 - 166.9)	2.73 ± 0.4 (0.009 - 181.8)	0.035 ± 0.002 (0.004 - 0.833)	0.024 ± 0.001 (0.004 - 0.25)	2145	811	24	13
TI (LS)	10.48 ± 1.59 (0.011 - 328)	3.9 ± 0.842 (0.023 - 45.7)	0.039 ± 0.0045 (0.004 - 1.231)	0.024 ± 0.0017 (0.004 - 0.077)	498	132	7	7
TDF (BLS)	3.01 ± 0.16 (0 - 66.24)	2.73 ± 0.28 (0.01 - 55.88)	0.063 ± 0.004 (0.004 - 1.386)	0.077 ± 0.006 (0.004 - 1.154)	1224	405	20	12
TI (BLS)	4.35 ± 0.48 (0 - 34.42)	4.16 ± 0.54 (0.01 - 10.59)	0.063 ± 0.005 (0.005 - 0.308)	0.019 ± 0.002 (0.015 - 0.077)	225	60	5	5

CHAPTER 3: FISHERIES DISTRIBUTION AND FISHERS' ADAPTATIONS TO PROTECTION MEASURES

V. Ferreira

CHAPTER 3A: FISHERS' BEHAVIOUR IN RESPONSE TO THE IMPLEMENTATION OF A MARINE PROTECTED AREA

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3A.1 Abstract

Marine Protected Areas (MPAs) have been widely proposed as a fisheries management tool in addition to their conservation purposes. Despite this, few studies have satisfactorily assessed the dynamics of fishers' adaptations to the loss of fishing grounds. Here we used data from before, during and after the implementation of the management plan of a temperate Atlantic multiple-use MPA to examine the factors affecting the spatial and temporal distribution of different gears used by the artisanal fishing fleet. The position of vessels and gear types were obtained by visual surveys and related to spatial features of the marine park. A hotspot analysis was conducted to identify heavily utilized patches for each fishing gear and time period. The contribution of individual vessels to each significant cluster was assessed to better understand fishers' choices. Different fisheries responded differently to the implementation of protection measures, with preferred habitats of target species driving much of the fishers' choices. Within each fishery, individual fishers showed distinct strategies with some operating in a broader area whereas others kept preferred territories. Our findings are based on reliable methods that can easily be applied in coastal multipurpose MPAs to monitor and assess fisheries and fishers responses to different management rules and protection levels. This paper is the first in-depth empirical study where fishers' choices from artisanal fisheries were analysed before, during and after the implementation of a MPA, thereby allowing a clearer understanding of the dynamics of local fisheries and providing significant lessons for marine conservation and management of coastal systems.

Keywords: Marine protected area; artisanal fisheries; fishing effort allocation; spatial modelling; hotspots analysis; fishers' choices

3A.2 Introduction

Besides conservation purposes, marine protected areas (MPAs) have also been suggested as important fisheries management tools (Gell and Roberts, 2003; Claudet *et al.*, 2011; Goñi *et al.*, 2011). The expected effects from the exclusion of extractive activities in marine reserves (no-take) are an increase in abundance, size and fecundity of fished individuals, especially for those most impacted by fisheries (Russ, 2002). This so-called "reserve effect" is expected to translate to biomass export of post-settlers to adjacent areas (spillover) which may, in turn, depend on density-dependent mechanisms and carrying capacity of protected and adjacent areas, as well as connectivity of suitable habitats (Kramer and Chapman, 1999; Russ, 2002). Some authors have also suggested that fisheries are more likely to benefit through larval export from reserves to surrounding areas due to an increase in size and fecundity of adults inside the reserve (Russ, 2002; Tetreault and Ambrose, 2007), but these benefits have been much more difficult to detect (Goñi *et al.*, 2010; Pelc *et al.*, 2010). Further to these direct responses, indirect effects may also occur and affect nearby areas after some time due to the build-up of top-predators and subsequent trophic cascades inside no-take areas (Micheli *et al.*, 2004; Hamilton *et al.*, 2010).

While some of these effects are well documented, their magnitude depends not only on factors such as habitat connectivity, oceanographic characteristics, species life histories, environmental requirements and mobility patterns (Claudet *et al.*, 2010; Hamilton *et al.*, 2010), but also on the enforcement of rules and compliance by local users (Claudet and Guidetti, 2010). Several reviews have focussed on the evaluation of the reserve effect (Micheli *et al.*, 2004; Lester *et al.*, 2009; Claudet *et al.*, 2011), but fewer studies have empirically considered the patchy distribution of species and fishing effort (Murawski *et al.*, 2005; Goñi *et al.*, 2008; Stelzenmüller *et al.*, 2008; Goñi *et al.*, 2010), which might have a large influence on the assessment of fisheries benefits of a MPA. In fact, the loss of fishing grounds and the redistribution of fishing effort in adjacent areas may affect the magnitude of the reserve effect (Claudet and Guidetti, 2010). Hence, it is important to include and understand fishers' behaviour in relation to enforced management rules, habitat preferences of commercial species and other fishers or competing activities.

The concentration of fishing effort near boundaries of no-take areas (i.e. fishing-the-line) is not uncommon and can be interpreted as spillover benefits to adjacent fisheries (Kellner *et al.*, 2007; Goñi *et al.*, 2008). On the other hand, very intense fishing-the-line behaviour may produce a sharp decrease in density adjacent to a reserve boundary (Goñi *et al.*, 2010). This is intrinsically related to gear selectivity since species catchability influences the extent of spillover and the effects inside the reserve (Goñi *et al.*, 2008). Traditional fishing grounds and travel costs may also influence fisheries allocation (Abesamis *et al.*, 2006). Recently, some studies have shown that the distance to borders of no-take areas, water depth and distance to the landing port are the most important factors explaining fisheries aggregations around

MPAs, which can be associated, respectively, with fishery benefits, target species distribution and costs (Wilcox and Pomeroy, 2003; Murawski *et al.*, 2005; Goñi *et al.*, 2008; Stelzenmüller *et al.*, 2008; Forcada *et al.*, 2010). The responses of coastal (Campbell *et al.*, 2012; Lédée *et al.*, 2012) and recreational (De Freitas *et al.*, 2013) fisheries distribution before and some years after rezoning have been reported for tropical MPAs. Fisheries displacement was assessed mainly based on face-to-face interviews, and the direct observations conducted after rezoning in one of the studies showed that fishers were reluctant to self-report spatial infringements (Campbell *et al.*, 2012). Therefore, in spite of work on the redistribution of fishing effort in large-scale trawl fisheries (Murawski *et al.*, 2005; Abbott and Hayne, 2012), there are no empirical studies using direct observations to compare spatial fishing allocations before and after implementation of protection measures in coastal MPAs where artisanal fisheries dominate.

Here we provide the first in-depth assessment of spatial redistribution of fishers in response to MPA implementation. The Arrábida Marine Park is a multiple-use MPA containing a core no-take zone surrounded by several zones with intermediate levels of protection where some human activities are allowed (e.g. small-scale fisheries, diving, tourism and recreational fisheries). In this coastal area artisanal fisheries prevail, where fishers use multiple gears, including trammel and gill nets, traps, longlines and jigs (Batista *et al.*, 2011). This study aims to analyse density patterns of the main fishing gear types by comparing the spatial distribution of vessels and buoys before, during and after implementation of the MPA management plan. Density clusters of individual fishers in preferred fishing grounds were investigated through time to understand fishers' choices and adaptability to the MPA rules.

3A.3 Methods

3A.3.1 Study area

The Arrábida Marine Park (AMP) is a 38 km stretch of coastline (53 km^2) on the west coast of Portugal, adjacent to a terrestrial nature park created in 1976 – the Arrábida Nature Park. The marine park includes the rocky shores and adjacent mixed sandy substrata between north of the Espichel Cape (38º27'N, 9º12'W) and Portinho da Arrábida (38º29'N, 8º57'W) (Figure 3A.1). This area is utilized year-round for commercial and recreational activities as it faces south and is protected from the prevailing north and northwest winds and waves. Nearby are the cities of Lisboa and Setúbal, the latter being an important fishing and commercial port located to the east of the park in the Sado estuary. In the middle of the park there is a small fishing town, Sesimbra, which has a long fishing tradition and is nowadays an important touristic area.

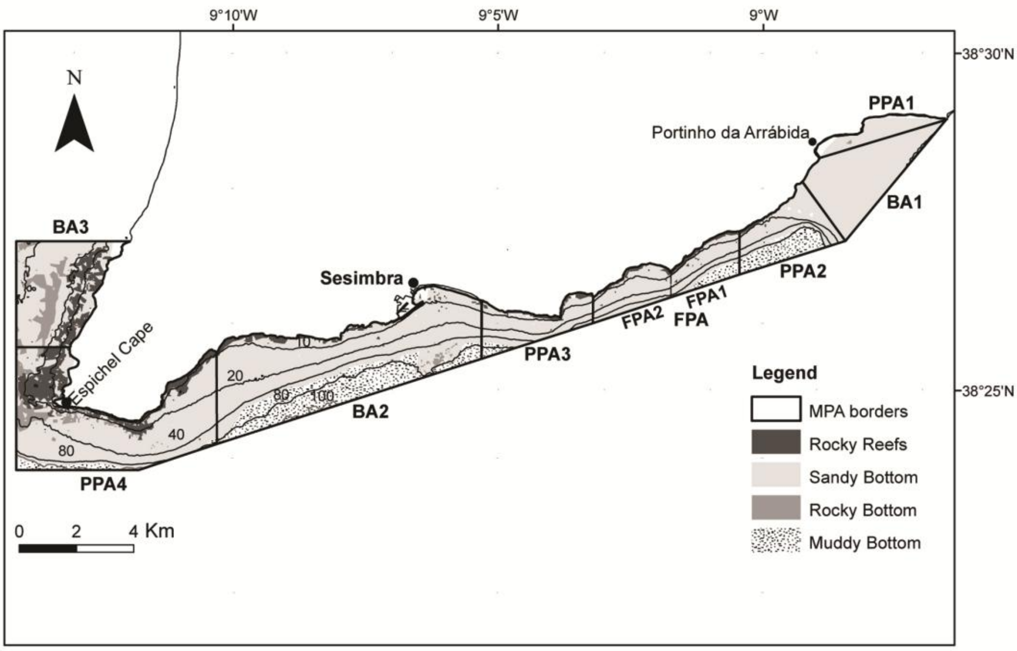

Figure 3A.8 - Map of the Arrábida Marine Park with zoning implemented by the management plan. Zoning: BA – Buffer areas; PPA – Partially-protected areas, FPA – Fully-protected area (divided in FPA1 and FPA2 due to the transitory phase of the management plan implementation – see Methods). Bathymetry and main habitat types are shown (Cunha *et al.*, 2011).

Nearshore, the subtidal shallow rocky reefs are dominated by boulders created by the erosion of the calcareous cliffs and by bedrock with fissures and crevices generating a complex habitat. This habitat is confined to the first 100-150 m from shore except on the west tip of the park where reefs extend beyond this range. Sand is the primary habitat covering the majority of the park from shallow (adjacent to rocky reefs and rocky outcrops) to deeper areas where it is replaced by mud.

The management plan was approved in August 2005 and multiple areas with differing levels of protection have been designated (Figure 3A.1): a fully-protected area (FPA) totalling 4 km^2; four partially-protected areas (PPA) totalling 21 km^2; and three buffer areas (BA) encompassing 28 km^2. Commercial diving for bivalves or other marine organisms, spearfishing, trawling, dredging and purse seining are forbidden in the whole park. These activities were considered to be the ones with the largest impact on coastal marine communities. Commercial fishing licenses for the park were exclusively allocated to fishers from Sesimbra who owned vessels smaller than 7 m in length. The FPA is a no-take, no-go area (except for research, monitoring and education purposes). In the PPAs, artisanal fishing with traps and jigs is allowed, but only beyond 200 m from coast and no extractive recreational activities (i.e. angling) are permitted. In the BAs, licensed fishing vessels and authorised recreational fishing are allowed.

The park's management plan established a transitional period for fisheries, aimed at facilitating the adaptation by fishers to the changes in uses, in which the rules of the different protection zones were gradually implemented during the first four years. In August 2006, management measures were enforced in the BAs, the east half of the FPA (FPA1) began as a PPA and the Portinho PPA (PPA1) was implemented. One year later, the remaining PPAs were implemented and the west half of the FPA (FPA2) started as a PPA. In 2008, the east half of the (current) FPA changed from PPA to FPA. The west half of the FPA was enforced in the summer of 2009, ending the transition period (Portuguese legislation, Council of Ministers Resolution 141/2005) (see Figure 3A.1).

The zoning and rules of the marine park were submitted to a consultation process as required by the Portuguese Law. This process involved NGOs, local authorities, professional fishers associations and other stakeholders. However, there is generally a low level of representation of artisanal fishers using small vessels in the fishers associations. This created problems in understanding the park objectives and accepting the management rules and it is still a focus of mistrust not only between the fishers association and the park authority, but also among fishers themselves. The exclusion of larger vessels from the Park was also very contentious, since the associations represent mainly these fishers. Zones were decided based on the MPA objectives and natural values present, with fishers' perceptions not influencing the zoning scheme. However, the initial plan (before consultation) was greatly changed to address the artisanal fishers' concerns, namely by including nets in the BAs (no nets were to be allowed in the MPA in the initial proposal) and reducing the level of protection in the PPAs with traps and jigs being allowed beyond 200 m from coast (in the initial proposal no fishing activities were considered in the PPAs). Nowadays, fishers with license to operate within the marine park appear to generally support it (Cunha *et al.*, 2011), possibly due to the decrease in fishing effort from competing gears (e.g. dredges) and larger vessels, but also to the exclusion of other competing fishing activities, such as spearfishing. However, most seem to disagree with several measures and enforcement strategies (depending on which type of gear they use), although poaching inside the no-take area is not supported, which suggests recognition of the benefits this area may provide.

3A.3.2 Sampling surveys

Fishing vessels and buoys within the marine park limits were surveyed along transects by boat. During each sampling day (one sample), the location of vessels and buoys (using a Global Positioning System - GPS), fishing gear type and vessels' names were recorded for all vessels and fishing buoys surveyed (the Portuguese legislation requires that fishing buoys at sea have to be identified with a code for fishing gear type and vessel identification). Two transects were performed each day covering the entire marine park (except the area north of the Espichel cape due to frequent rough sea conditions). The first transect, focussed on vessels, started early (6:45 to 7:45 am) in the east part of the Park, near the Portinho da Arrábida bay, and ended at Espichel cape. All buoys were then surveyed on the second transect in the opposite direction. Sampling was carried out inside the Arrábida Marine Park under a permit by the marine park authority (Parque Natural da Arrábida, Instituto da Conservação da Natureza e da Biodiversidade).

Sampling was carried out in five different periods corresponding to the 'before', 'implementation' and 'after' phases of the management plan (see above): 'Before' period – from April to November 2004 (7 samples for buoys and 28 samples for jig vessels); 'Implementation' period refers to Years 1, 2 and 3: Year 1 – from March to August 2007 (15 complete samples: for both buoys and vessels), Year 2 – September 2007 to February 2008 (14 complete samples), Year 3 – November 2008 to August 2009 (16 complete samples);

'After' period – September 2009 to December 2009 (6 complete samples). This classification was used for all analyses. Buoy surveys in the Before period were not uniformly distributed over time, whereas in the Implementation and After periods an average of three and two samples/month were conducted, respectively. The small vessels using jigs were only identified in Year 3 and in the After period.

In the Before period, vessel surveys were shore based, with ten stations established on the high cliffs along the coast covering the entire marine park. Sampling was done early in the morning on a weekly basis, and vessels were georeferenced based on the topographic triangulation method (Davis *et al.,* 1981), using an electronic theodolite (Topcon, model DT – 30) and a GPS. This method has a high level of accuracy in terms of the spatial positioning of objects/features (Singh *et al.*, 2000).

Three fishing gear types were analysed since they were identified as the most important in the study area: traps, trammel and gill nets, and jigs. Other fishing gears were recorded but were observed infrequently (longlines) or occurred only before the management plan was approved (purse seines). Data for vessel location was used for jigs, since this gear is operated manually directly below the fishing vessel, while buoy geographic coordinates were used for stationary gear (traps and nets).

Even though some vessels were seen few times or only in one of the periods (some fish infrequently, others did not maintained their license or were transferred to other ports), others were observed consistently over the course of the study, with some of them fishing with more than one gear. Three vessels were detected fishing with the three gear types, one with jigs and traps and twenty with nets and traps. Fifty four vessels were seen fishing exclusively with traps, fourteen with nets and a hundred and thirteen vessels were fishing only with jigs.

3A.3.3 Data analysis

3A.3.3.1 Generalized Additive Models

For all analyses data were aggregated by periods and the three fishing gear types. The spatial and temporal fishing dynamics in the AMP and possible explanatory variables were analysed combining geographic information system (GIS) techniques and generalized additive models (GAMs). The marine park limits and zoning (source: AMP authority) were superimposed onto a map of habitats and bathymetry (source: Cunha *et al.*, 2011) using a 500 x 500 m grid (0.25 km^2 cells), although some grids were smaller due to the coastal line and legal borders. Densities (counts per area) of the main fishing gears in the park were summarized by grid and for GAMs only fished grids (with recorded fishing activities) were included (Goñi *et al.*, 2008; Stelzenmüller *et al.*, 2008).

Fishing effort allocation was related to the following features using GIS to measure the shortest linear distance (m) from each feature to the mid-point of each grid: distance to Sesimbra port (DistPort), distance to coast (DistCoast), depth, distance to the partially-protected areas (DistPPA), distance to the fully protected area (DistFPA), distance to the 200 m line offshore of the coast (Dist200m) and distance to the ¼ nautical mile line (Dist1/4nm). The variable Dist200m was only related to traps and jigs since it is a limit implemented inside the PPAs, where nets are excluded. Therefore, when these gears were used beyond the 200 m limit but inside the PPAs, the distance to the borders of these areas was negative, to distinguish from fishing gears operating outside these areas. On the other hand, Dist1/4 nm is a national legal limit only for bottom fixed nets (trammel nets and gillnets). The DistPPA1 (PPA1 refers to Portinho bay) was removed from the analyses since only forbidden small drift nets were found fishing there before the management plan started. Throughout the implementation period (Years 1, 2 and 3), DistPPA and DistFPA refer to the respective regime of each protected area in each period.

GAMs were used to explore the density response to the explanatory variables as non-linear relationships were expected and this non-parametric technique does not require linear trends (Zuur *et al.*, 2009). Autocorrelations among spatial features were tested for each period and only variables with no or low levels of correlation were used to conduct these models.

Choosing gamma as the exponential family and using a square root transformation of the response variable resulted in residuals showing a good approximation with normality. Several GAMs were run to test for the best fitted model for each gear type and period. Since some variables were highly correlated we selected those considered to better explain fishers' choices: DistPort, depth, DistPPA and DistFPA. Additionally, alternative models were run to test the influence of the current FPA (DistFPA) during the Before, Year 1 and Year 2 periods (i.e. before full protection was implemented) to evaluate if this was an area pre-selected for its specific characteristics. All explanatory variables included in the models were allowed to be non-linear (using smoothers). Approximate significance of the smooth terms and deviance explained were obtained from each GAM.

For all gears, depth was highly correlated to distance to the coast. For traps and jigs, depth was also highly correlated to Dist200m and for nets to Dist1/4nm. Therefore, significant results for depth should be interpreted with caution as they may also reflect significant effects of those other variables. Additionally, a bottom type was assigned to each grid cell using habitat maps. Bottom type by grid cell was classified into the following categories: sand, mud, rock (isolated rocky outcrops) and reefs (coastal shallow rocky reefs). Variables related to habitat were not included in GAMs due to co-linearity but since bottom type may influence both species and fishers' distribution, a Kruskal–Wallis test was conducted to assess the density of gear types (square root transformed) relative to bottom type in each period. Multiple comparison tests evaluated differences in density for each pair of habitat-types. These analyses were conducted with the R 2.14.1 software (R Core Team, 2012).

3A.3.3.2 Spatial hotspot analyses

Fishing areas were analysed using area pattern statistics (Fortin and Dale, 2005). Specifically, hotspot analysis was performed in order to study the changes in uses in the main locations chosen by fishers across the five periods, for each fishing gear type. Spatial patterns were investigated using GIS modelling techniques with Arcgis 10.0 (ESRI) software. For this, a 250 x 250 m grid covering the marine park was superimposed to the fishing GPS points. Hotspot analysis was conducted separately for each of the main gear types with the geographic positions of each vessel occurring in each grid with the aim to study the patterns of use of fishing grounds by individual fishers. To determine statistically significant hotspots, Getis-Ord Gi* statistic (which gives a Z-score and a p-value) (Ord and Getis, 1995) was calculated for each grid cell. Statistical tests for significant spatial patterns in data (obtained by a Z score, which varied between -1.96 and +1.96), were compared with the null hypothesis of complete spatial randomness (CSR) with a 95% confidence level against the alternative hypothesis that events are spatially clustered or dispersed. The larger the Z score, the more intense is the clustering of high values (i.e. a hotspot) whereas for negative Z scores, the smaller the Z score, the more intense is the clustering of low values (cold spot) (Ord and Getis, 2001). Significant clusters were defined as the aggregation of adjacent grid cells with a Z-score $\geq |1.96|$, consistent with spatial clustering. To understand the vessel composition in each aggregation, the number of vessels and their percent contribution to each significant cluster was calculated by period and fishing gear. However, since the identification of individual vessels using jigs was not always possible in the Before, Year 1 and Year 2periods, the contribution of these vessels was only evaluated for clusters from Year 3 and the After period.

To perform these analyses, the best distance band was chosen based on global Morans I statistics for spatial autocorrelation (Ord and Getis, 2001). This tool provides a Z-score for the entire study area, measuring spatial autocorrelation based on feature locations and attribute values. To calculate Morans I, the 200 meters distance was used as the starting distance with a cut-off at 800 meters. The minimum distance was chosen based on grid size and the maximum observed dispersion of points. The conceptualization of spatial relationships used for the analysis was the zone of indifference. The final global Z-scores were plotted against the Euclidean distance values and when the increase of the distance caused a decrease in the Z-value (peak), that distance was selected as the best distance band to use in the hotspot analysis (Ord and Getis, 2001).

3A.4 Results

3A.4.1 Traps

The selected models for the density of traps by period explained between 16.5% and 53.2% of the total variability (Table 3A.S1). Overall, the distance to port significantly influenced fisher's behaviour in the Before period and Year 1 ($p < 0.05$), whereas depth influenced effort density allocation in all periods (Year 1: $p < 0.05$; Years 2, 3, After: $p < 0.001$), although in the Before period it was marginally non-significant ($p = 0.055$). The distance to PPAs was not significant in all periods but, interestingly, after the two halves of the fully protection zone were implemented (in Year 3 and After periods), the distance to their borders significantly influenced the variance (Year 3: $p < 0.01$; After: $p < 0.001$).

In separate models (not shown), distance to the current FPA was tested for the periods before this protection level was effective (Before, Year 1 and Year 2). This variable did not influence the density of traps in the Before period but significant differences were found in both Year 1 ($p < 0.05$) and Year 2 ($p < 0.001$).

The additive fits of the significant predictor variables from all modelled time periods are shown in Figure 3A.2. During the Before and Year 1 periods the density increased with distance to port showing two peaks, at around 5000 m and 13000 m (Figure 3A.2a, b). Trap density decreased steeply with depth up to approximately 18-20 m, and then increased up to approximately 80-90 m, although there are few observations at those deeper locations (Figure 3A.2c-f) which were mainly situated in front of Sesimbra port where depth increases rapidly offshore (Figure 3A.1). The density of traps also decreased with distance to the fully protected area (Figure 3A.2g, h), but this trend shifted at around 8000 m from the FPA border, where density started to increase.

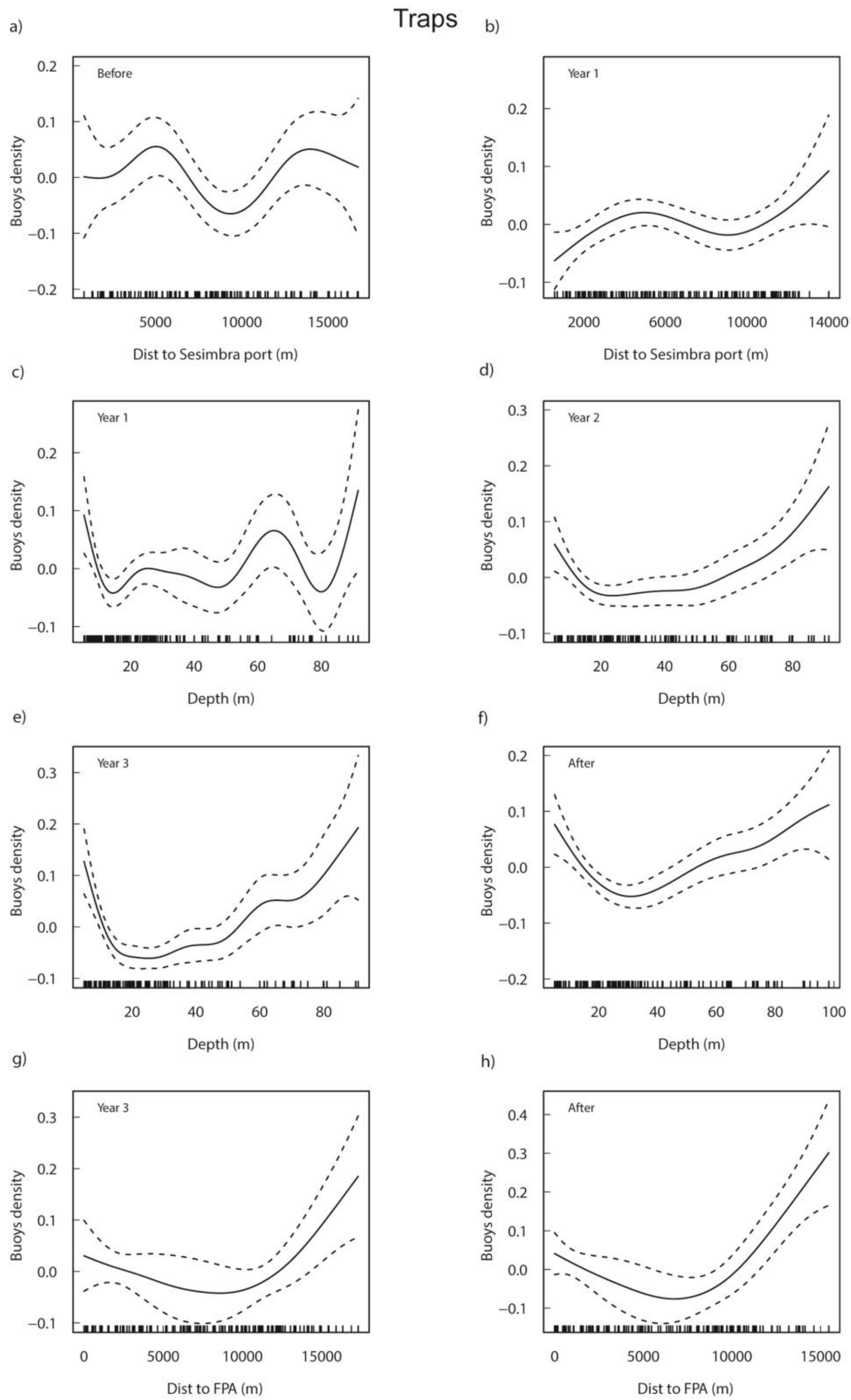

Figure 3A.9 - Additive fits of the significant predictor variables to the density of trap buoys for each period. Distance to Sesimbra port (a, b), depth (c-f), and distance to FPA (g, h) from the significant periods of the selected GAMs (see Table 3A.S1) are shown. Tick marks above the *x*-axis indicate the distribution of observations and the *y*-axis shows the contribution of the smoother to the fitted values. The solid line is the estimated smoothing curve and the dotted lines show the 95% confidence intervals.

Density patterns were not influenced by habitat type in each of the five periods. However, aggregating data from all periods showed a significant effect of habitat on effort density allocation (Kruskal-Wallis chi-square = 16.6, $p < 0.001$). Multiple comparisons revealed significantly ($p < 0.05$) higher density of traps in sand compared to mud and rock, but not compared to reefs.

3A.4.2 Nets

The selected models for the density of nets by period explained between 37.6% and 64.30% of the total variability (Table 3A.S2). Overall, distance to port (except in Year 3; Before, Year 2: $p < 0.001$; Year 1: $p < 0.005$; After: $p < 0.01$) and depth ($p < 0.05$; except in Year 3 and the After period) had an important role in the spatial allocation of nets. Additionally, distance to PPA and to FPA started to have a significant influence in Year 2 and in the After period, respectively ($p < 0.05$). Unlike traps, in the models testing the distance to the current FPA (not shown) for the periods before this protection level was effective, this descriptor was significant for the density of nets before the management plan was implemented ($p < 0.01$) but lost significance after its implementation.

The density of nets increased with the distance to port (Figure 3A.3a-d) although there are few observations beyond 10000 m (Before), 8000 m (Year 1) and 6000 m (Year 2). In the After period, there was a decrease in density between 3000 m and 6000 m. Nets generally decreased with depth up to approximately 20 meters, increasing afterwards (Figure 3A.3e-g). In Year 1 a steep decrease in density was found at around 50 m. Density in relation to distance to PPA increased significantly in Year 2 (and was marginally non-significantly in Year 3) and in the After period, when distance to FPA also increased significantly (Figure 3A.3h-j).

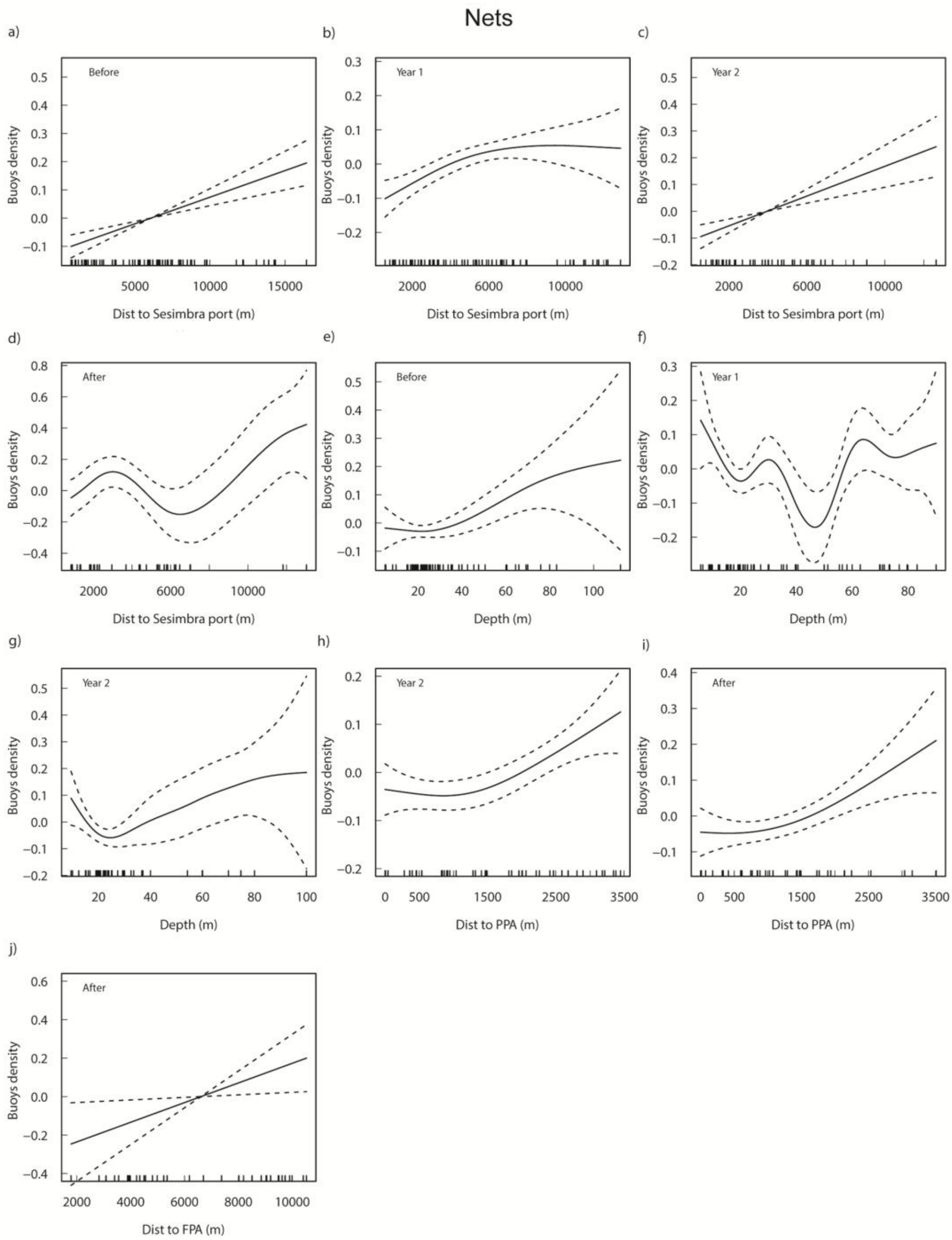

Figure 3A.10 - Additive fits of the significant predictor variables to the density of nets buoys for each period. Distance to Sesimbra port (a-d), depth (e-g) and distance to PPA (h, i) and to FPA (j) from the significant periods of the selected GAMs (see Table 3A.S2) are shown. Tick marks above the *x*-axis indicate the distribution of observations and the *y*-axis shows the contribution of the smoother to the fitted values. The solid line is the estimated smoothing curve and the dotted lines show the 95% confidence intervals.

Habitat type was significantly related to the density of nets only in Year 3 (Kruskal-Wallis chi-square = 9.4, $p < 0.05$), with multiple comparisons showing that density on rock was higher than on mud ($p < 0.05$), with reefs and sand showing intermediate values. When aggregating data from all periods (Kruskal-Wallis chi-square = 12.9, $p < 0.005$) the same pair of habitats differed significantly ($p < 0.05$).

3A.4.3 Jigs

The selected models for the density of vessels fishing with jigs show a very high deviance varying between 56.6% and 84.4% which was much higher than that of the other gear types (Table 3A.S3). Depth was a highly significant factor associated to the density of jigs in all periods ($p < 0.001$; except in Year 1 when it was marginally non-significant). In both Year 3 and the After period, it was the only significant factor in the model. Distance to port was also an important factor both in the Before ($p < 0.05$) and Year 2 ($p < 0.001$) periods. The only period where protection measures significantly influenced the density of jigs was in Year 1 ($p < 0.05$) with a decreasing pattern with the distance to PPA. No significant influence was found on jigs allocation in relation to the distance to FPA. When the distance to the current FPA was tested for the first periods (Before, Year 1 and Year 2) in separated models (not shown) it was also highly significant before protection started ($p < 0.001$) and in Year 2 ($p < 0.005$).

Overall, jig density increased with the distance to port, especially in the first 5000-6000 m (Figure 3A.4a, b). Additionally, in the Before period there was a decrease in density between 7000 m and 10000 m followed by a second increase further away from port. In the following periods, very few vessels were seen beyond 8000 m from port. Depth greatly influenced density (Figure 3A.4c-f), decreasing to up to approximately 18 m with a subsequent increase to approximately 30 m (but where few vessels occurred). Fitted significant models showed a complex response of the density of jigs with the distance to the PPA during Year 1 (Figure 3A.4g), when only the current FPA1 was enforced with a partial protection status.

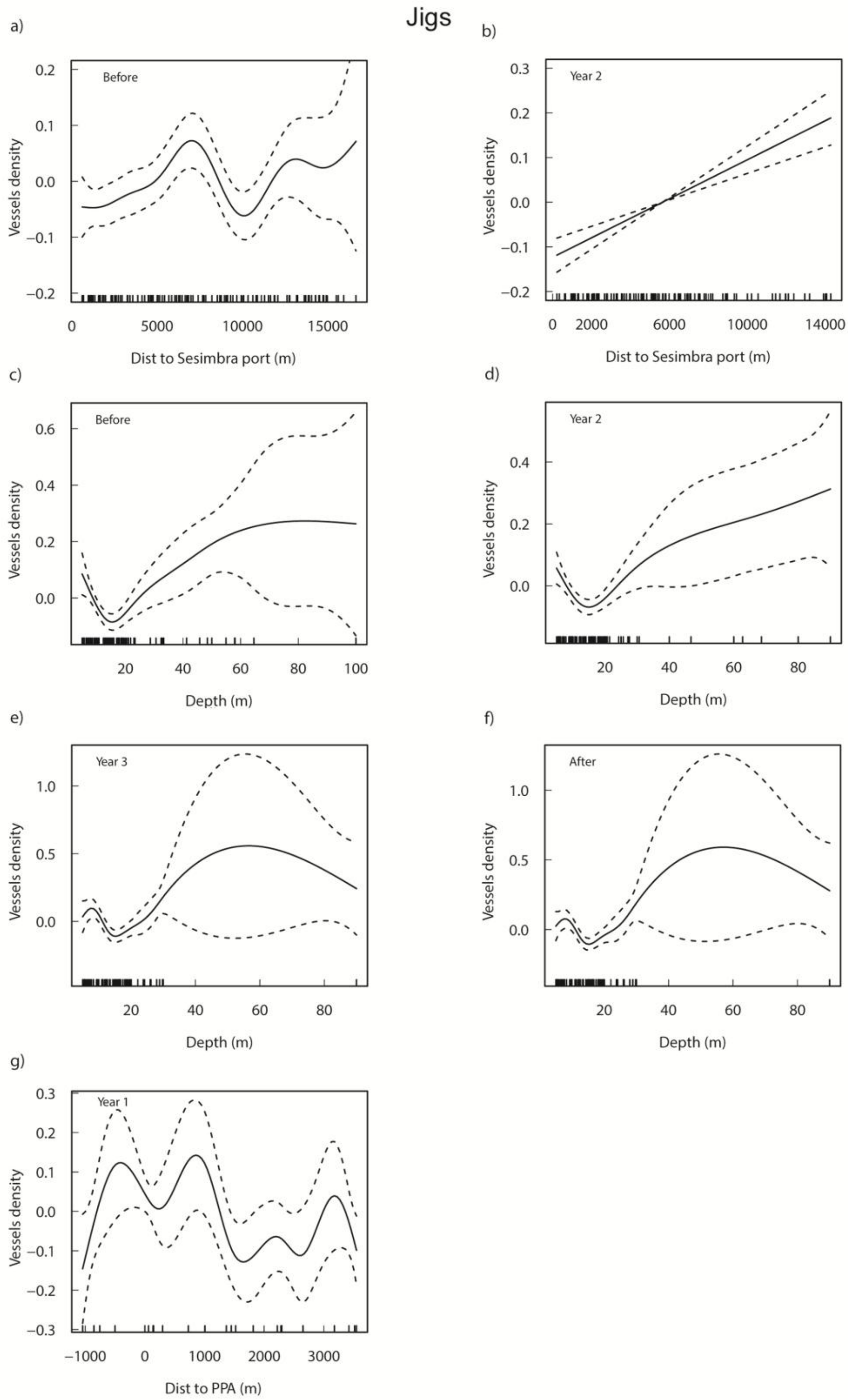

Figure 3A.11 - Additive fits of the significant predictor variables to the density of jig vessels for each period. Distance to Sesimbra port (a, b), depth (c-f), and distance to PPA (g) from the significant periods of the selected GAMs (see Table 3A.S3) are shown. Tick marks above the *x*-axis indicate the distribution of observations and the *y*-axis shows the contribution of the smoother to the fitted values. The solid line is the estimated smoothing curve and the dotted lines show the 95% confidence intervals. Negative distances refer to jig vessels fishing inside the PPA.

Habitat significantly influenced density in the Before ($p < 0.001$), Year 2 ($p < 0.05$) and Year 3 ($p < 0.05$) periods, but was marginally non-significant in Year 1 ($p = 0.06$) and no relation was detected in the After period. Similarly to the other fishing gears, when aggregating all periods, the density of jigs was highly influenced by habitat (Kruskal-Wallis chi-square = 26.92, $p < 0.001$), with significantly ($p < 0.05$) higher values in rocky reefs comparing to sand and mud.

3A.4.4 Hotspot analysis and individual vessels trends

The hotspot analysis revealed the dynamics of significant fishing clusters throughout the different time periods and gear types analysed (Text 3A.S1). Traps followed closely the sequential enforcement of rules through the Implementation years, with some fishing effort displaced from no-fishing areas as shown in the rearrangement of clusters, some of which merged as a result of the MPA rezoning (Figure 3A.5). On the other hand, the cluster closer to the no-take area was divided in two, with vessels surrounding its borders. The same rearrangement of clusters and changes in preferred areas (Text 3A.S1) as a consequence of the management plan implementation were also detected in nets (Figure 3A.6) and jigs (Figure 3A.7), although nets remained relatively stable through time in their main fishing grounds, which were already in fished areas. Jigs showed larger changes, with vessels generally moving towards home port but keeping close to the no-take zone. The contribution of individual vessels to each cluster in each time period was also analysed (Text 3A.S1) for traps (Figure 3A.S1), nets (Figure 3A.S2) and jigs (Figure 3A.S3).

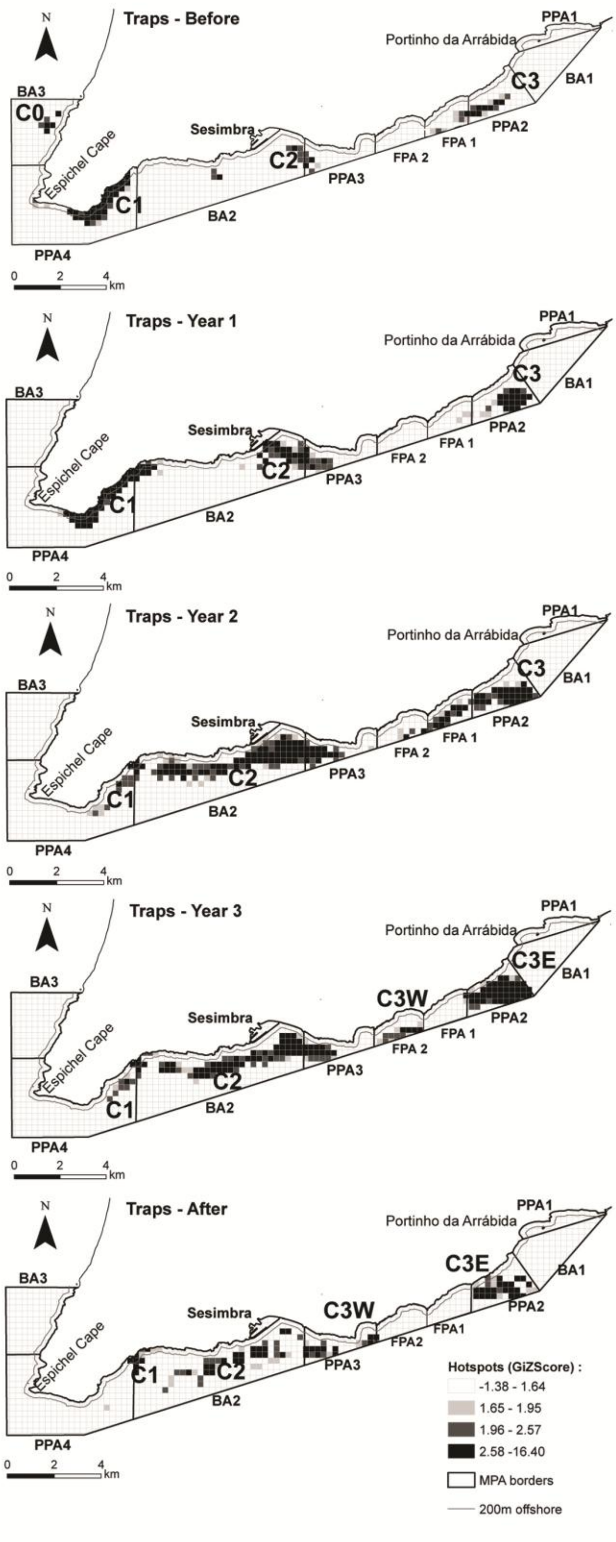

Figure 3A.12 - Maps obtained from the hotspot analyses of trap buoys for each period. The location of significant clusters (GIZScore > 1.96) by period (a – Before, b – Year 1, c – Year 2, d – Year 3, e – After), and the different protection levels at the Arrábida Marine Park are shown: BA – buffer area; PPA – partial protection area; FPA – fully protected area (see Methods for a detailed description of the protection levels in the park and their implementation through time). The 200 m offshore line is also shown.

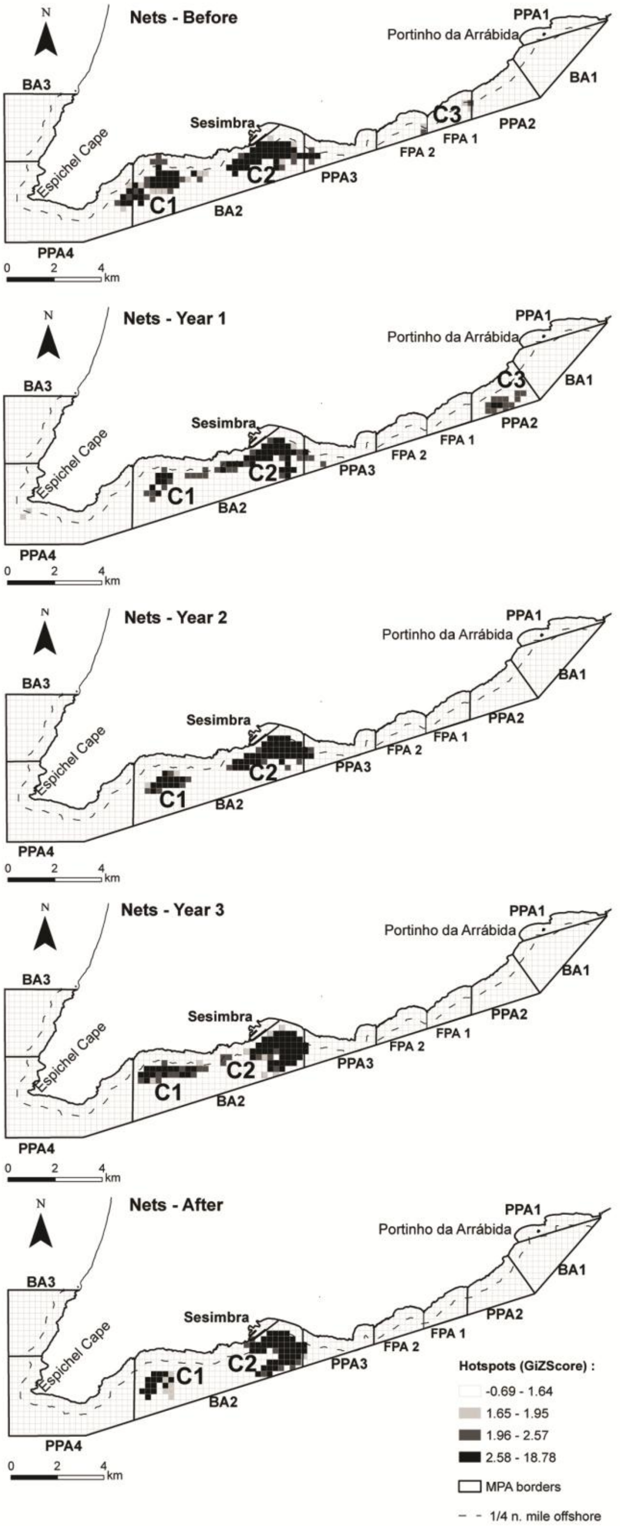

Figure 3A.13 - Maps obtained from the hotspot analyses of nets buoys for each period. The location of significant clusters (GIZScore > 1.96) by period (a – Before, b – Year 1, c – Year 2, d – Year 3, e – After), and the different protection levels at the Arrábida Marine Park are shown: BA – buffer area; PPA – partial protection area; FPA – fully protected area (see Methods for a detailed description of the protection levels in the park and their implementation through time). The national legal limit for nets of the line of ¼ nautical miles offshore is also shown.

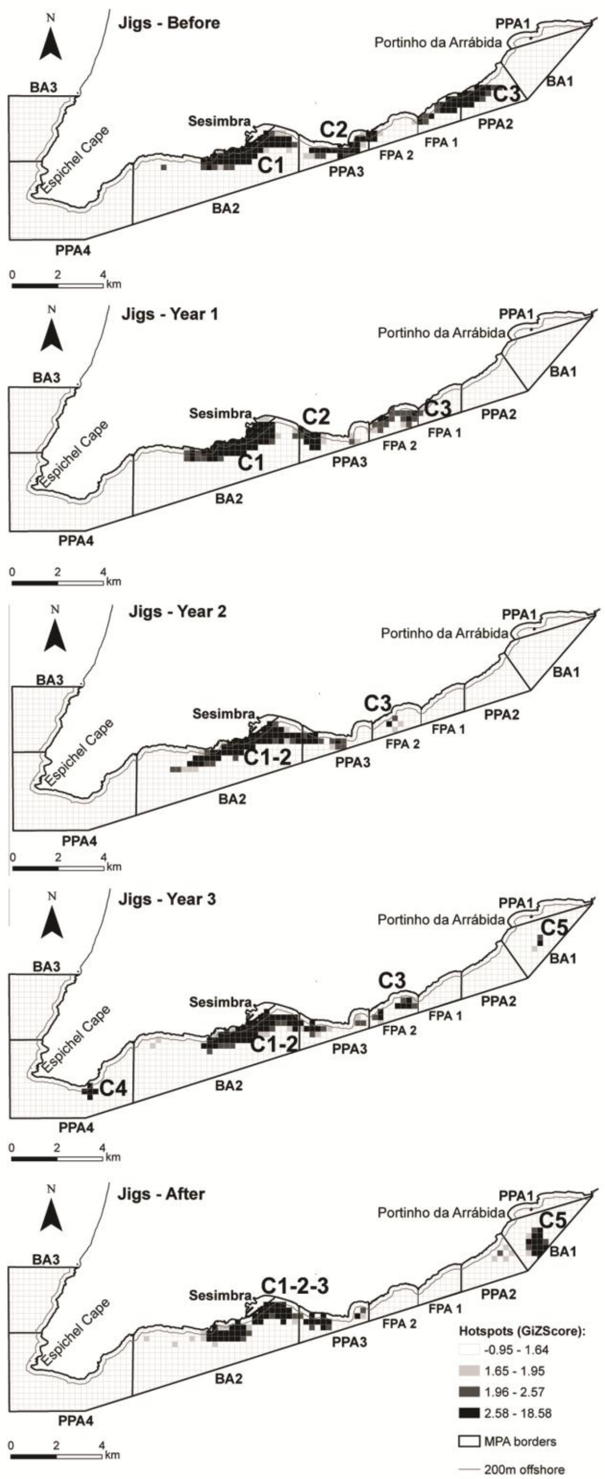

Figure 3A.14 - Maps obtained from the hotspot analyses of jig vessels for each period. The location of significant clusters (GIZScore > 1.96) by period (a – Before, b – Year 1, c – Year 2, d – Year 3, e – After), and the different protection levels at the Arrábida Marine Park are shown: BA – buffer area; PPA – partial protection area; FPA – fully protected area (see Methods for a detailed description of the protection levels in the park and their implementation through time). The 200 m offshore line is also shown.

3A.5 Discussion

Here we found that artisanal fisheries showed fisher- and fisheries-specific adaptations to multiple protection measures in a marine protected area (MPA). These findings suggest that artisanal fisheries from temperate systems have complex dynamics and that accounting for individual fishers' behaviour and preferences in exploiting fishing grounds is crucial to implement more successful and effective multiple-use MPAs (i.e. areas with different zones with different types of rules applied to uses).

Different fisheries responded differently to the implementation of protection measures, with preferred habitats of target species driving fishers' preferences in the selection of fishing grounds. Moreover, within each fishery individual fishers showed distinct strategies, with some operating in a broader area whereas others kept preferred territories, some of them being adjacent to a no-take area. Spatial allocation of fishing grounds was well defined and apparently agreed upon among the most common fishers, supporting the occurrence of traditional routines. One of the possible consequences of effort reallocation inside multiple-use MPAs is an increase of spatial competition for setting fishing gears in buffer areas (Lédée *et al.*, 2012). When fishing effort is very high, the catchability of each gear may be reduced, affecting the expected benefits from protection. Interestingly, when fishers have licences for multiple gears, adapting to management rules may be easier. In fact, in our study traps and jigs faced a smaller reduction of fishing grounds than nets, although jigs may have lost important areas close to shore. Several fishers can opt to operate with various gears with a preference for traps instead of nets, as revealed by the increasing trend in the number of vessels fishing with traps. This suggests fishing with traps was the least affected fishery and that fishers are adapting to other productive alternatives in response to the zoning and rules of the marine park.

Some recent studies addressed the allocation of fisheries before and in response to spatial closures (temperate trawl fisheries: Murawski *et al.*, 2005; Abbott and Hayne, 2012; tropical artisanal fisheries: Campbell *et al.*, 2012; Lédée *et al.*, 2012), although we could not find other empirical cases in the literature where artisanal fishers' distribution were analysed through direct observations before, during and after the implementation of a temperate MPA. Tracking the spatial position of vessels and fishing gears through time and analysing factors affecting the selection of a fishing ground may allow for a clearer understanding of the fishers' choices and adaptations to different situations as well as of the dynamics of small scale artisanal fisheries, which comprise a large percentage of the fishing communities throughout the world.

Jigging for cuttlefish (*Sepia officinalis*) and squid (*Loligo vulgaris*) from small wooden vessels is a traditional artisanal fishing activity in the region. Jigging takes place mainly close to shore and near rocky reefs at depths up to 20 m. Thus, the jigging effort distribution in shallow areas can be attributable to species occurrence, gear characteristics and safety for these small vessels. Jigs were mainly influenced by depth and habitat through time. They were significantly more associated to rocky reefs than to other habitat types. Some previously preferred fishing grounds located inside the reserve may have become off-limits to these fishers since association with nearshore habitats lost significance with time and there was an important effect of the FPA location on vessels' density before its implementation. Consequently, this fishery seems to have been impacted by the zoning as the fishers lost fishing grounds close to shore within the full and partially-protected areas. This may have negative consequences on the acceptance by fishers and on their attitudes towards the marine park (Leleu *et al.*, 2012).

A highly dynamic spatial distribution of jig vessels through time was detected with three main clusters identified. These were typically formed by a high number of vessels, sometimes with a large contribution of occasional fishers. In the After period, the three clusters that were previously scattered throughout the park merged into a single large cluster in front of the port where no restrictions apply to this fishery. The management plan implementation therefore caused some significant changes to the spatial distribution of this type of fishery, operated by small 3-4 m vessels, which take advantage of the very sheltered conditions of this coastline, mainly in front and to the east of their home port. They operate by drifting with the alongshore tides and target cephalopod species which occur in nearshore environments.

Benefits from protection may however have occurred since jig fishers remained in the area beyond 200 m in the PPAs, and near the western border of the FPA, which was the closest to their home port, even during the implementation phase of the fully protected area. This suggests that some fishers were able to profit by staying a little further away from shore, probably intercepting species over sandy bottoms adjacent to the shallow rocky reefs, rather than competing with other commercial and recreational fishers in the buffer zone.

Jig fishers' adaptations suggest they tried to keep as close as possible to their former fishing ground, possibly also benefiting from protection, whereas at the same time their displacement was towards their home port, revealing other additional concerns probably related to security and operating costs. Lédée *et al.* (2012) found that fishers preferentially redistributed to areas already known before protection, suggesting that previous experience and tradition may play an important role in the site-fidelity behaviour, influencing the choice of a fishing location. However, similar to the present study, the authors also report that most of the fishers' displacement was towards their home port, mainly due to the lower costs, leading to an increase in the fishing pressure in areas that had already high density.

There were several factors explaining the spatial and temporal distribution of nets. Distance to port influenced effort density except in Year 3. The two main clusters occurred right in front of and to the west of the port and remained stable through time. A third cluster was detected in the initial periods in the east of the park encompassing part of the fully protected area but disappeared thereafter, with some fishing activities probably moving adjacently to the southern limit of the fully protected area outside the marine park limits. The proximity to both the partial and fully protected areas became important in the After period, with nets being located further away from these areas, which is consistent with the location of the main clusters.

The area in front and to the west of vessels' home port is an important fishing ground where the main clusters consisting of several vessels were detected. Those clusters did not relocate after protection started as they were already in an allowed area. This is an extensive shallow sandy area used by commercial fishers targeting soles, cuttlefish and fish species such as sparids by trammel and gill nets (Batista *et al.*, 2009). When all periods were combined, buoy density was significantly associated with rock suggesting that fishers prefer shallow habitats, especially those with the potential to attract fish such as rocky outcrops and adjacent sand or shallow reefs.

The trap fishery showed preferred sites with clusters close to the home port and on each side of the park. Depth was the strongest influence in trap allocation with higher densities in shallow waters (18-20 m) and at around 70 m (but with fewer vessels), suggesting the possibility that these fishers were targeting different habitats. There were more traps distributed on sand than on other habitats, except for shallow rocky reefs. This is consistent with the behaviour of octopus (*Octopus vulgaris*, Cuvier, 1797), the target species of this gear, which is found in mixed sandy habitats, from the coastline to depths of around 200 m, usually spending the winter in deeper waters and migrating inshore by early spring to breed (Guerra, 1975; Roper *et al.*, 1984).

The spatial dynamics of trap fishers showed a cluster close to their home port, which is advantageous for small vessels that cannot travel far for safety reasons, are limited by sea and weather conditions (Forcada *et al.*, 2010) and where operating costs are a significant burden. Another cluster was found near the most complex reefs of the park (Gonçalves *et al.*, 2003) which are also near the entrance of the Sado estuary, an important spawning and nursery area (Vasconcelos *et al.*, 2010), with vessels extending their activity outside the park limits. Interestingly, the analysis of fishers' choices through time showed that in this cluster (which in the Before period partially occupied the future fully protection area), considerable changes occurred in both the spatial distribution of traps and composition of vessels dominating this area. Although in Year 1 no cluster was found in the fully protection area (FPA), in Year 2 the eastern cluster extended to this area with fishers likely trying to gain access to this fishing ground before it became off-limits. This interpretation is reinforced by the fact that distance to the current FPA was not significant in the Before period, but became an important explanatory variable in the model during Year 3 and After periods (when the FPA was fully implemented), indicating that fishers were attracted to this area possibly due to the expectation of future benefits.

A few (3-5) vessels dominated several of these clusters and their behaviour changed through time. The western cluster became a hotspot dominated by a single vessel which was able to secure this fishing ground, whereas the central cluster was characterized by a larger number of vessels with a more erratic behaviour (i.e. vessels joined other clusters through time). This may be related to high competition in this fishing ground. On the other hand, with the retrieval of one dominant vessel that did not receive a license from the park, the eastern cluster was taken over by two new dominant vessels showing specific territories and dominance in these fishing grounds. These two vessels fished mainly on the borders of the FPA adopting a strategy of "fishing the line" (Kellner *et al.*, 2007). Several reasons can explain this increase in effort at the edge of a no-take area: the reallocation and aggregation of effort because of the reduction of fishing grounds or due to perceived or expected benefits from protection (McClanahan and Mangi, 2000; Goñi *et al.*, 2010).

In spite of the loss of fishing ground as a consequence of MPA designation, the spatial competition between trap fisheries and, namely, nets decreased on important and traditional fishing grounds since nets became only allowed in the buffer areas. Moreover, before the management plan implementation, nearshore reefs were heavily exploited by spearfishing. The exclusion of this type of recreational fishery, which has a large impact on high trophic level species such as large sparids, seabass and octopus (Cooke and Cowx, 2004; Rocklin *et al.*, 2011), likely contributed to the increase of such target species' biomass inside the marine park. In fact, the landings of octopus for vessels licensed to fish in the park have increased since protection started (Horta e Costa *et al.*, 2013).

Other studies in Mediterranean MPAs found that the proximity to the reserve borders significantly affected the spatial distribution of fishing effort (Stelzenmüller *et al.*, 2007; Goñi *et al.*, 2008; Stelzenmüller *et al.*, 2008; Forcada *et al.*, 2010). The loss of fishing grounds and the attraction to the reserves' boundaries when spillover effects are substantial, are important factors explaining the reallocation of fishing effort related to the implementation of MPAs. These effects are however influenced by the spatial distribution of habitats and target species inside and outside the reserve (Forcada *et al.*, 2008). Thus, the proximity to no-take zones may not be involved in the choice of the fishing ground or may be due to the fishers' preference for being closer to their former fishing location (Lédée *et al.*, 2012; Leleu *et al.*, 2012).

Abesamis *et al.* (2006) found that artisanal fishers tended to select traditional fishing grounds which were probably preferred due to their guarantee of higher stability in catches and a higher minimum average income. The experience and familiarity with fishing grounds, one component of traditional and ecological knowledge (Davis *et al.*, 2004; Leleu *et al.*, 2012), may also help to minimize gear loss and enhance catches.

Conclusion

To understand the complexity of impacts (both positive and negative) on fisheries related to marine protected areas, one needs to closely follow the dynamics of fisheries operating nearby. This is particularly challenging for coastal multiple-use MPAs where artisanal fisheries occur. Here we show an effective method for the study of fishing effort allocation and dynamics for artisanal fisheries using different gears by following individual fishers' choices before, during and after the implementation of protection. Our results have relevance to the vast majority of global MPA designs; that is, single, relatively small multiple-use areas utilized by local fishers using multiple gear types. Besides the importance of assessing fishing effort within and around MPAs, this study shows that gear type, habitat features and MPA design influence individual fishers' behaviour and this must be taken into account when planning MPA design and evaluating the effects of marine conservation measures. This type of information is lacking in most studies evaluating the effects of marine protected areas although it is central for an unbiased assessment of biological, social and cultural responses to marine protection.

3A.6 Acknowledgments

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3A.8 Supplementary Information

3A.8.1 Text 3A.S1 - Hotspot analysis and individual vessels trends

Traps

The hotspot analysis of trap distribution revealed three significant clusters common to all periods and a fourth one north of the Espichel cape (Figure 3A.5). The latter (the north cluster – C0) is exclusive to the Before period since this area was only assessed then. The other three are the western cluster (C1), located south-east of the Espichel cape, the central cluster (C2), in front of Sesimbra port, and the eastern cluster (C3) occupying mainly the current PPA2. In Year 2, when all PPAs started and traps were allowed only beyond 200 m from shore (except in the BAs), the western cluster (C1) shifted and merged with the central cluster (C2) which then occupied a large part of BA2. The eastern cluster (C3) occupied the PPA2 in the Before and Year 1 periods and a portion of the FPA1 (not statistically significant in Year 1). In Year 2 this cluster increased in size and extended well into the FPA1 (still a PPA at that time). In Year 3, the FPA1 was enforced with full protection status and this cluster divided in two parts which respectively occupied the FPA2 and PPA2. In the After period, when both FPA1 and FPA2 were enforced, the part of the cluster which occupied the FPA2 in the previous period moved to the eastern border of PPA3 (adjacent to the fully protected area).

The contribution of individual vessels to each cluster in each time period was analysed (Figure 3A.S1). Some important vessels from the Before period shift location or disappeared after that. The number of vessels in the cluster C1 decreased with time, with one vessel (T13) increasing their contribution throughout periods and dominating that hotspot in the After period (94%). The central cluster (C2) contained a large number of vessels changing their relative importance through time with only two vessels (T24 and T41) occurring in all periods (except in the Before period). The eastern cluster (C3) was initially dominated by vessel T18 (85%) which also disappeared from the park after this period (this vessel was not granted a license to fish in the park). Vessels T42 and T44 started to dominate this cluster after Year 1 and were joined by two additional vessels (T34 and T39). Interestingly, in Year 3 this cluster was split in half with the enforcement of the eastern half of the FPA, with vessels T42 (79%) and T44 (19%) dominating the FPA2 area (C3W) (Figure 3A.S1). In the After period, T42 was the only vessel responsible for the significance of this cluster, which was located adjacent to the western limit of the FPA. In the eastern half of the cluster (C3E), all four vessels were equally contributing to this aggregation both in Year 3 and the After period.

Nets

Three significant clusters were obtained from the hotspot analysis in the Before and Year 1 periods, but the eastern cluster (C3) located in FPA1 (Before) and in PPA2 (Year 1) disappeared (Figure 3A.6). The two large clusters located in the central (C2) and western (C1) part of the park remained relatively stable through time. Significant clusters were generally beyond the legal limit of ¼ nautical miles for nets.

The western cluster (C1) had important contributions from vessels that were only present in the park before the management plan implementation (N15, N29, N56, N58). After that, vessel N48 was present in all periods and several other vessels contributed significantly to this cluster (Figure 3A.S2). The main vessels contributing to the central cluster (C2) remained stable over time (N14, N17, N33). Several vessels changed between these clusters both within the same period and between periods (Figure 3A.S2).

Jigs

The hotspot analysis showed varying numbers of significant clusters in the different periods (Figure 3A.7). Three clusters remained relatively stable through time. A western cluster (C1) located very close to shore immediately to the west and in front of Sesimbra port, merged with a central cluster (C2) located to the east of Sesimbra in Year 2, forming thereafter a single cluster. The eastern cluster (C3) was located in FPA1 and PPA2 in the Before period but it moved closer to Sesimbra after that, first to FPA2 and PPA3 in Years 1, 2 and 3, merging with the cluster C1-2 in the After period although it remained adjacent to the FPA border. In Year 3 two new significant clusters formed in the park: one near Espichel cape (C4) and one close to the Portinho da Arrábida bay (C5). However, the former did not remain significant in the After period whereas the latter increased in density.

The analysis of individual vessel contributions to each cluster was only conducted for Year 3 and After periods (when information of individual vessels was collected for this gear type) (Figure 3A.S3). In Year 3, the merged western (C1) and central (C2) clusters contained a high number of vessels (n = 61) detected. The eastern cluster (C3) also contained several vessels (n = 15) but with a lower percent contribution of less frequent (occasional) fishers. The Espichel cluster (C4) contained eight contributing vessels with two main fishers (J69, J104) influencing this distribution, whereas the Portinho cluster (C5) had six vessels that were only seen once and thus their contribution was even. Although the Espichel cluster was only detected in Year 3, this was however a very important area for longlines which are operated by similar vessels and so fishers found jigging could have previously been fishing with longlines (which are only allowed in the buffer area). In the After period, the three merged clusters (C1-2-3) contained fewer vessels (n = 53) than in the previous period but again with a high proportion of occasional fishers. The Portinho cluster had a larger number of vessels (n = 26) although these were not detected in the previous period (Figure 3A.S3).

3A.8.2 Tables 3A.S

Table 3A.S4 - Results of the smoothing terms from the generalized additive models (GAM) testing the density of trap buoys relative to the distance to several spatial features in the Before, implementation (Years 1, 2 and 3) and After periods.

Period	Explanatory variables	edf	F	p-value	Deviance explained (%)
Before	s(DistSes)	5.161	2.361	**p < 0.05**	16.50
	s(Depth)	1	3.764	p = 0.055	
Year 1	s(DistSes)	3.561	3.17	**p < 0.05**	30.20
	s(Depth)	7.868	2.457	**p < 0.05**	
	s(Dist_PPA)	2.804	1.515	n.s.	
Year 2	s(DistSes)	1.852	2.714	p = 0.062	23.00
	s(Depth)	4.872	4.295	**p < 0.001**	
	s(Dist_PPA)	1	0.258	n.s.	
Year 3	s(DistSes)	5.421	2.096	p = 0.053	53.20
	s(Depth)	6.752	7.657	**p < 0.001**	
	s(Dist_PPA)	1	0.692	n.s.	
	s(Dist_FPA)	2.433	4.102	**p < 0.01**	
After	s(DistSes)	5.676	1.24	n.s.	50.10
	s(Depth)	4.166	7.316	**p < 0.001**	
	s(Dist_PPA)	2.502	1.717	n.s.	
	s(Dist_FPA)	2.451	6.567	**p < 0.001**	

Explanatory variables selected are: distance to Sesimbra port (DistSes), depth, distance to PPA (DistPPA), distance to FPA (DistFPA). Estimated degrees of freedom (edf), F-statistics and corresponding p-values are indicated. Significant values are in bold; n.s.: non-significant; marginally non-significant values are also shown.

Table 3A.S5 - Results of the smoothing terms from the generalized additive models (GAM) testing the density of nets buoys relative to the distance to several spatial features in the Before, implementation (Years 1, 2 and 3) and After periods.

Period	Explanatory variables	edf	F	p-value	Deviance explained (%)
Before	s(DistSes)	1	24.215	**$p < 0.001$**	37.60
	s(Depth)	2.371	3.537	**$p < 0.05$**	
Year 1	s(DistSes)	1.878	6.792	**$p < 0.005$**	56.70
	s(Depth)	7.198	2.816	**$p < 0.05$**	
	s(DistPPA)	4.326	1.76	n.s.	
Year 2	s(DistSes)	1	18.398	**$p < 0.001$**	64.30
	s(Depth)	3.982	2.975	**$p < 0.05$**	
	s(DistPPA)	2.112	4.605	**$p < 0.05$**	
Year 3	s(DistSes)	1	1.693	n.s.	60.60
	s(Depth)	4.501	1.526	n.s.	
	s(DistPPA)	3.001	2.581	$p = 0.054$	
	s(DistFPA)	1.683	0.544	n.s.	
After	s(DistSes)	4.566	3.712	**$p < 0.01$**	62.30
	s(Depth)	1	0.447	n.s.	
	s(DistPPA)	1.982	4.624	**$p < 0.05$**	
	s(DistFPA)	1	5.274	**$p < 0.05$**	

Explanatory variables selected are: distance to Sesimbra port (DistSes), depth, distance to PPA (DistPPA) and distance to FPA (DistFPA). Estimated degrees of freedom (edf), F-statistics and corresponding p-values are indicated. Significant values are in bold; n.s.: non-significant; marginally non-significant values are also shown.

Table 3A.S6 - Results of the smoothing terms from the generalized additive models (GAM) testing the density of jig vessels relative to the distance to several spatial features in the Before, implementation (Years 1, 2 and 3) and After periods.

Period	Explanatory variables	edf	F	p-value	Deviance explained (%)
Before	s(DistSes)	7.12	2.663	**$p < 0.05$**	56.60
	s(Depth)	5.091	6.832	**$p < 0.001$**	
Year 1	s(DistSes)	1	1.355	n.s.	84.40
	s(Depth)	4.545	2.238	$p = 0.085$	
	s(Dist_PPA)	8.234	2.637	**$p < 0.05$**	
Year 2	s(DistSes)	1	38.814	**$p < 0.001$**	61.80
	s(Depth)	4.53	6.829	**$p < 0.001$**	
	s(Dist_PPA)	1.343	2.839	$p = 0.075$	
Year 3	s(DistSes)	5.523	1.321	n.s.	65.90
	s(Depth)	6.478	6.058	**$p < 0.001$**	
	s(Dist_PPA)	1.885	1.153	n.s.	
	s(Dist_FPA)	2.502	0.955	n.s.	
After	s(DistSes)	1.281	1.013	n.s.	65.80
	s(Depth)	6.306	5.967	**$p < 0.001$**	
	s(Dist_PPA)	2.585	0.856	n.s.	
	s(Dist_FPA)	6.01	1.651	n.s.	

Explanatory variables selected are: distance to Sesimbra port (DistSes), depth, distance to PPA (DistPPA), distance to FPA (DistFPA). Estimated degrees of freedom (edf), F-statistics and corresponding p-values are indicated. Significant values are in bold; n.s.: non-significant; marginally non-significant values are also shown.

3A.8.3 Figures 3A.S

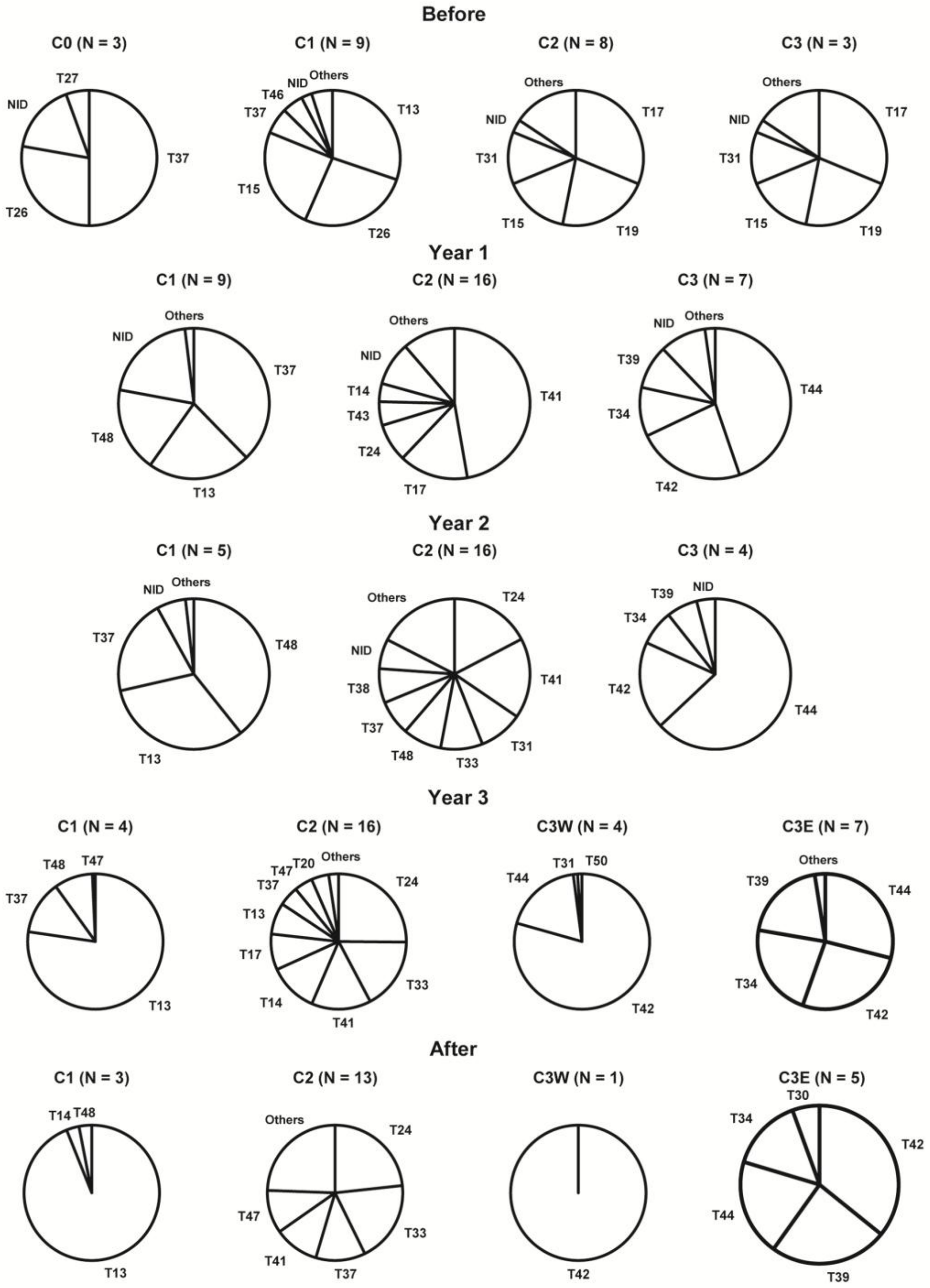

Figure 3A.S4 - Proportion contribution of vessels using traps (T) to each of the significant clusters obtained in the hotspot analysis of trap buoys by period: Before, Year 1, Year 2, Year 3, After. See the location of each cluster in Fig. 3A.5. The number of vessels observed in each cluster is also shown.

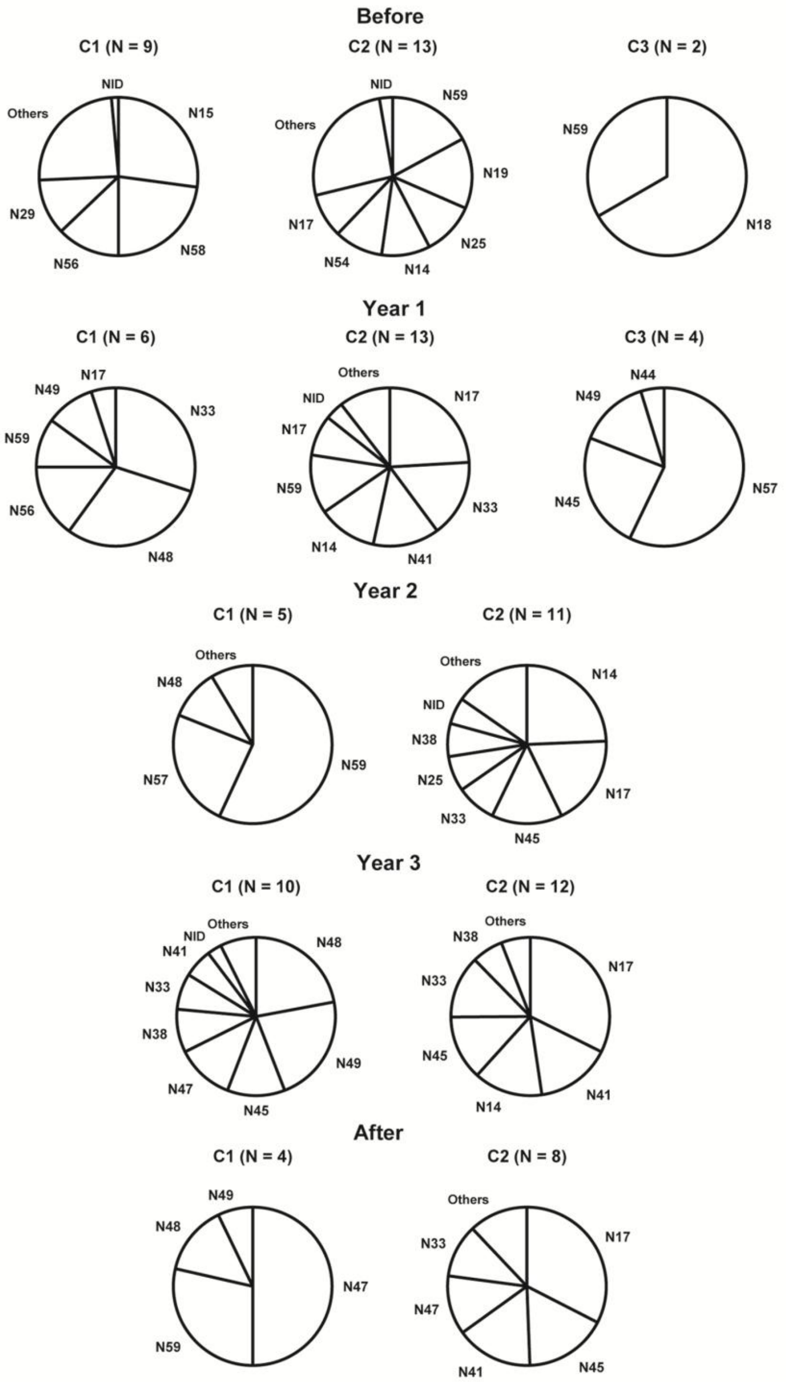

Figure 3A.S5 - Proportion contribution of vessels using nets (N) to each of the significant clusters obtained in the hotspot analysis of nets buoys by period: Before, Year 1, Year 2, Year 3, After. See the location of each cluster in Fig. 3A.6. The number of vessels observed in each cluster is also shown.

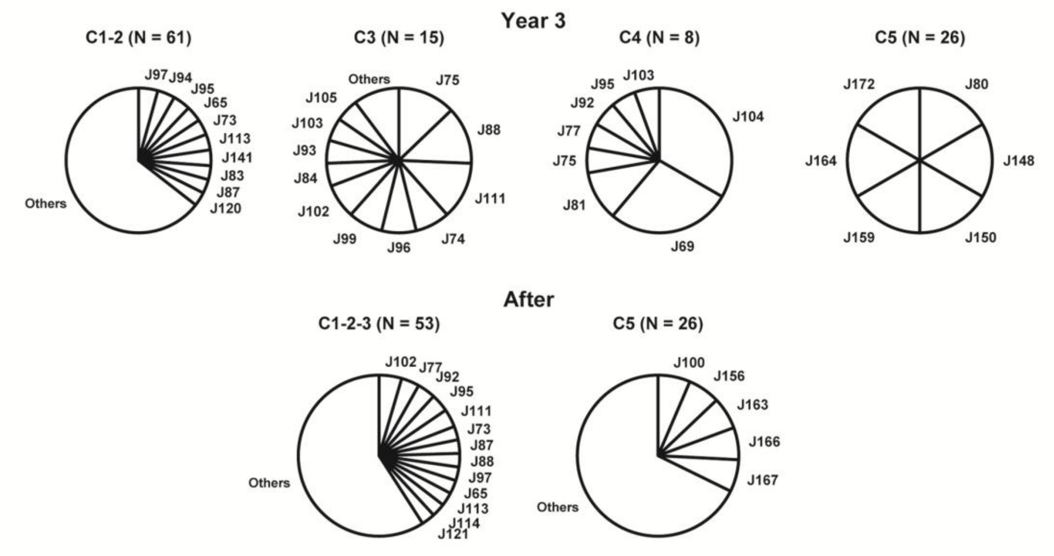

Figure 3A.S6 - Proportion contribution in of jig vessels (J) to each of the significant clusters obtained in the hotspot analysis of jig vessels by period (jigs were only correctly identified in Year 3 and After periods). See the location of each cluster in Fig. 3A.7. The number of vessels observed in each cluster is also shown.

CHAPTER 3B: VESSELS' SITE FIDELITY AND SPATIO-TEMPORAL DISTRIBUTION OF ARTISANAL FISHERIES BEFORE THE IMPLEMENTATION OF A TEMPERATE MULTIPLE-USE MARINE PROTECTED AREA

Submitted to *Fisheries Research*

Horta e Costa, B., Gonçalves, L. & Gonçalves, E. J. *submitted*. Vessels' site fidelity and spatio-temporal distribution of artisanal fisheries before the implementation of a temperate multiple-use marine protected area, *submitted to Fisheries Research.*

3B.1 Abstract

Marine Protected Areas (MPAs) are increasingly proposed as a fisheries management tool besides their conservation purposes. When assessing the ecological, economic, and social-cultural impacts of protection, the dynamics of fisheries and fishers reallocation within and around multiple-use MPAs should be analysed. Despite this, few studies incorporate the baseline information of fisheries distribution, therefore compromising an understanding of fishers' preferences, choices and constrains before losing fishing grounds through the establishment of zoning and protection measures. To fulfil this gap, here we assess the spatial and seasonal fishers' preferences from local artisanal fisheries (nets, traps, jigs and longlines) before the implementation of a MPA management plan (the Arrábida Marine Park, Portugal). Zero inflated modelling, hotspot analysis, vessels distribution range and site fidelity statistics showed that the main drivers of fishing effort allocation are the placement of preferred fishing grounds which are likely related to the distribution of target species and associated habitats. Proximity to port, weather conditions and distance to coast are also important factors influencing, in different ways, these artisanal fisheries. Our findings highlight the complex dynamics of the distribution of artisanal fisheries operating multiple-gears and targeting multiple-species and are likely transferable to several coastal multiple-use MPAs where no baseline data exist. Moreover, the variety of responses and preferences found between gears and fishers before the establishment of zoning are important to understand local fisheries, to contribute to an ecosystem-based management and to improve management decisions.

Keywords: Artisanal fisheries; fishing effort allocation; baseline information; marine protected areas, spatial modelling

3B.2 Introduction

Marine protected areas (MPAs) have been identified as being an important conservation and fisheries management tool (Gell and Roberts, 2003; Claudet *et al.*, 2011) with the potential to function as an ecosystem based management approach (Fraschetti *et al.*, 2011). Multiple-use MPAs have been widely implemented due to their potential to accomplish conservation objectives while allowing human uses and minimizing conflicts by including some degree of protection for commercial species and important habitats as well as promoting the use of local fisheries and a range of recreational activities (Claudet *et al.*, 2006; Lester and Halpern, 2008;

Claudet *et al.*, 2010a; Rocklin *et al.*, 2011). Several studies and reviews indicate the potential of no-take areas (marine reserves), either isolated or embedded inside multiple-use MPAs, to increase density, size, biomass and diversity of species, especially those most targeted by fisheries (Russ, 2002; Micheli *et al.*, 2004). Currently, there is strong evidence showing that in several appropriately designed and well enforced marine reserves there is an export of adults to nearby areas (Stelzenmüller *et al.*, 2007; Goñi *et al.*, 2008; Stobart *et al.*, 2009). However, the strength of this 'reserve effect' depends both on species characteristics and behaviour, such as mobility, commercial value and association to particular habitats (Claudet *et al.*, 2010b), and also on socio-economic factors such as enforcement, compliance and fishers' preferred fishing grounds (Abesamis *et al.*, 2006; Samoilys *et al.*, 2007).

Fishing effort allocation may depend on target species distribution (Murawski *et al.*, 2005) and on fishers' traditional routines and ecological knowledge (Davis *et al.*, 2004). The loss of fishing grounds with the designation of a MPA may affect those fishers' choices as well as the profitability of some fisheries. Moreover, if a high density of fishing effort aggregate near the boundary of a no-take area (a.k.a. "fishing the line", Kellner *et al.*, 2007) and if gear selectivity is high (Goñi *et al.*, 2010), the effectiveness of protection on the spawning biomass of species targeted by fisheries may be compromised. Even if fishers do not aggregate in no-take borders but their effort concentrates and increases in adjacent buffer areas, the fisheries productivity of those areas will likely be affected (Goñi *et al.*, 2008). Therefore, knowledge on fisheries' dynamics, preferences and constrains before the implementation of a MPA allows a better understanding of the benefits and impacts of conservation policies on small scale coastal communities.

Some studies have been conducted in MPAs to investigate how the distance to their borders, depth, and particular habitats influence the spatial allocation of fishing effort (Wilcox and Pomeroy, 2003; Abesamis *et al.*, 2006; Goñi *et al.*, 2008; Stelzenmüller *et al.*, 2008; Forcada *et al.*, 2010). However, those studies lack baseline information on the spatial distribution of fishing effort before the implementation of protection, thus potentially biasing the interpretation of the observed patterns. Recent studies compared fisheries allocation before and after the implementation of protection measures both in temperate large-scale fisheries (Murawski *et al.*, 2005; Abbott and Hayne, 2012) and in tropical artisanal fisheries (Campbell *et al.*, 2012; Lédée *et al.*, 2012), but there is a lack of empirical studies assessing fishery displacement through direct observations in temperate MPAs where artisanal fisheries predominate (but see Horta e Costa *et al.*, 2013).

We use a set of observational and statistical methods to explore spatial data on fisheries allocation before the implementation on a multiple-use MPA (the Arrábida Marine Park, Portugal). This case study fills the existing gap, by addressing the spatial distribution and dynamics of fishing effort and site fidelity of individual vessels before the MPA implementation, i.e. when fishing is not constraint by additional zoning and regulation schemes. This type of information is potentially usefull to managers of coastal fisheries and to a large majority of MPAs where artisanal fisheries are the norm.

3B.3 Methods

3B.3.1 Study Area

The Arrábida Marine Park is a 38 km stretch of coastline (53 km^2) on the west coast of Portugal, adjacent to a terrestrial nature park created in 1976 – the Arrábida Nature Park. This marine park includes the rocky shores and adjacent mixed sandy substrata between north of the Espichel Cape (-138668 N, -95008 E, Portuguese coordinate system ETRS89 – PT-TM06, with Transverse Mercator projection or 38º27'N, 9º12'W wgs84) and Portinho da Arrábida (-131485 N, -73618 E or 38º29'N, 8º57'W) (Figure 3B.1). The shore is steep and bordered by high calcareous cliffs in most of the park. Excluding beach areas, throughout the park the shallow rocky reefs and rocky outcrops are confined to the first 100-150 m from shore except near the Espichel Cape where extend to deeper waters. Extensive sand banks prevail, especially in front and to the west of Sesimbra port and in the Portinho da Arrábida bay, where seagrass meadows used to occur. In fact, sand is the primary habitat covering the majority of the park from shallow (adjacent to rocky reefs and rocky outcrops) to deeper areas where it is replaced by mud near the Park limits. This marine park is utilized year-round for commercial and recreational activities as it faces south and is protected from the prevailing north and northwest winds and waves. Nearby are the cities of Lisbon and Setúbal, the latter being an important fishing and commercial port located to the east of the park in the Sado estuary. In the middle of the park there is a small fishing town, Sesimbra, which has a long fishing tradition and is nowadays an important touristic area.

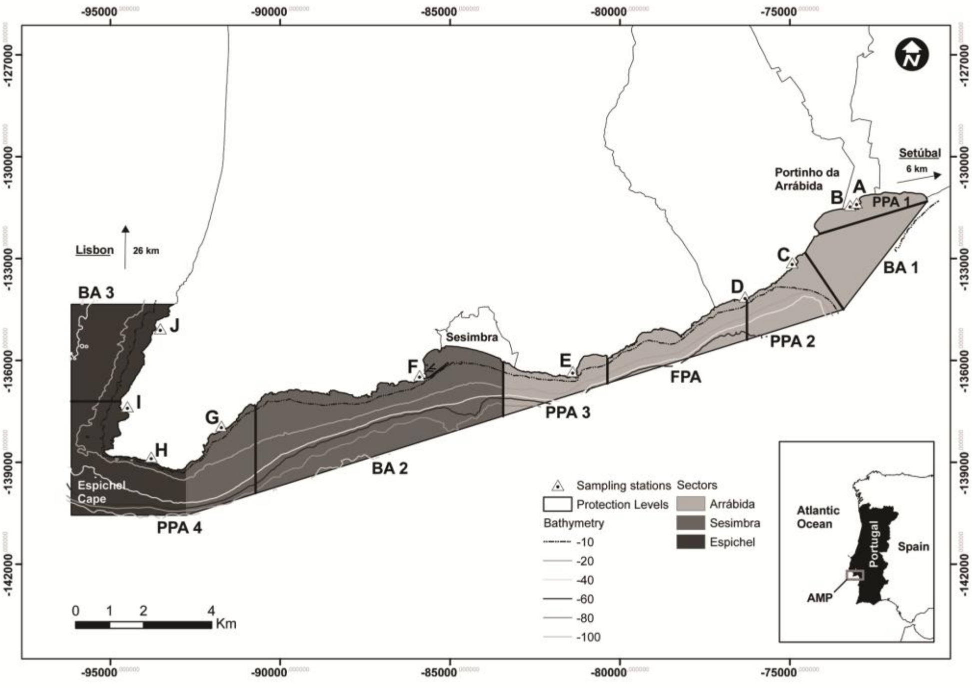

Figure 3B.6 - Map of the location of the Arrábida Marine Park (Portugal). The zoning of the different protection levels (fully protected area – FPA; partially protected areas – PPA; buffer areas – BA) implemented in the management plan (2005) is shown. Sampling stations (A-J), depth contours and park sectors from the observation network along the marine park are also shown.

The marine park was designated in 1998 but the management plan was only approved in 2005 (Portuguese legislation, Council of Ministers Resolution 141/2005). It includes: a fully protected area (FPA) which is a no-take, no-go area (with the exception of research, monitoring and education purposes); four partially protected areas (PPAs) (two surrounding the no-take area) in which non-extractive recreational activities and licensed commercial fishing with traps and jigs are allowed beyond 200 m from coast; and three buffer areas (BA) where recreational activities (including fishing) and licensed commercial fishing are allowed. Vessels larger than 7 m, trawling and dredging, purse-seining, commercial fishing divers and spearfishing are excluded. Licensed fishers allowed to operate within this marine park after the management plan was established are only from Sesimbra. Local artisanal fisheries use multiple gears, including trammel and gill nets, traps, longlines and jigs (Batista *et al.*, 2011). Nets mesh sizes in the park are the same as in other areas. There are two mesh sizes of gill nets 80-99mm, ≥100mm. Trammel nets are only allowed with a mesh size ≥100mm.

Before the management plan approval, this marine park did not have any zoning but commercial fishing divers (collecting clams) and clam dredging were already forbidden throughout the park since its creation (1998). Thus, despite of not having any zoning its borders were defined and recognized. Fishers started to discuss the management plan regulations with the authorities in 2001 and by the time of the present study (2004) the public consultancy was still ongoing. Zoning was proposed initially and suffered few changes throughout time, but regulations were not concluded until 2005. The inclusion of fisheries in the PPAs (traps and jigs) was posterior to 2004, as well as the inclusion of nets in the BAs. This means that fishers contributed to the definition of final regulations. The fishers' consultancy was contentious mainly because vessels above 7 m were to be excluded inside the park borders and fishers operating with smaller vessels were not well represented in the fishers' associations. Overall, during 2004 fishers were not yet aware about the future regulations of the management plan.

3B.3.2 Sampling

During 2004, fishing vessels within the marine park limits were surveyed weekly from March to October, along ten stations located at the top of the cliffs (shore-based sampling). When a vessel operates with multiple gear types, fishers usually use one gear at a time and gear type was distinguished and identified by these visual surveys. Sampling occurred early in the morning when local commercial fisheries operate. These stations enabled 100% visual coverage of the marine park and fishing activity was sampled by randomly surveying from all stations, which usually took four days to complete. The sampling area was divided into three sectors according to habitat structure, depth and wind and wave exposure, as described by Gonçalves *et al.* (2003) (Figure 3B.1).

Sector 1 has shallow sandy areas in the Portinho da Arrábida and complex but narrow nearshore rocky reefs, with different sized-boulders due to the erosion of adjacent vertical cliffs, which also protects the area from the predominant winds and currents. Sector 2 includes nearshore rocky reefs and an extensive sandy area in front and to the east of Sesimbra. Depth increases more rapidly than in Sector 1. Rocky reefs are mainly bedrock with crevices and the exposure to offshore storms is higher than in Sector 1. Sector 3 is the most exposed area of the park since it includes the north of the park and the area surrounding the Espichel Cape. Here rocky reefs are wider and go deeper than in the previous sectors with sandy beaches only present in the northern area.

Fishing activity was georeferenced based on the topographic triangulation method using an electronic theodolite (Topcon, model DT–30) and a geographic positioning system (GPS) (see Text 3B.S1 for a detailed description of the topographic measurements method). Additionally, a sea-based sampling survey was also conducted to assess the distribution and extent of static gears (nets and traps) by recording the GPS position of buoys from each fishing vessel (the Portuguese legislation requires that fishing buoys must be identified concerning gear type and vessel identification). Seven transects were conducted by boat between April and November 2004, covering the entire area, to assess the distribution range of individual vessels. One or two sampling days were conducted between April and July and one more in November, in days with calm conditions and good visibility to detect buoys at sea. In each sampling day, one large transect was done throughout the whole park in a zigzag mode from the park limits to the coast line to detect all fishing buoys present.

3B.3.3 Data Analysis

The spatial distribution of trammel and gill nets, traps, jigs and longlines were analysed as they were the most frequently observed gears in the study area. Other fishing gears were recorded but were observed infrequently (e.g. purse seines, small boat dredges for shellfish) and were not included in the following analyses.

3B.3.3.1 Zero inflated data modelling

The spatial fishing dynamics and possible explanatory variables were analysed by combining geographic information system (GIS) techniques and linear statistical models. The marine park limits were superimposed to a bathymetry map (source: Marine Park authority) and to a 500 x 500 m grid (0.25 km^2 cells). Cells adjacent to the coastline and the marine park borders were adjusted. Fishing effort allocation of the most important fishing gears (nets, traps and jigs) was related to the explanatory variables distance to port (Sesimbra or Setúbal depending on the port of registry of each vessel), distance to coast and depth, using GIS to measure the shortest linear distance (m) from each feature to the mid-point of each grid. Response variables were the density of buoys (nets and traps) or vessels (jigs) by grid area. Longlines were not used in these models since few observations were obtained.

The resulting dataset showed overdispersion due to the presence of excessive zeros (zero inflated) but also due to more variation than expected by the Poisson distribution in the non-zero component. This last type of overdisperion was addressed by using the negative binomial distribution. The source of zeros could be due to either false or true zeros but since the contribution from each of these could not be determined, the two-part model, also named hurdle or zero altered model with negative binomial distribution (ZANB) was chosen (Zuur *et al.*, 2009).

This model deals with the data in two separated components: the non-zero part (count model) which in this case uses the negative binomial distribution with the log link function; and the presence/absence (1/0) part (here defined as zero hurdle model) using the binomial distribution with logit link function. The area was introduced in the model as an offset, allowing dealing with counts instead of a continuous variable (Zuur *et al.*, 2009).

However, these (zero inflated) models assume linearity of the response variable in relation to the predictors and so, previously, those patterns were explored using generalized additive models (GAM). These models use a smoother function for each of the predictors in relation to the response variable, allowing non-linearity of the data (Zuur *et al.*, 2009). Only variables with no or low collinearity were used in each model. Depth and distance to coast were highly correlated in vessels from Sesimbra port and, since depth had a more variable pattern with a non-linear trend, the variable distance to coast was preferred as it did not require any transformation. The variable distance to Sesimbra port was squared rooted transformed.

In the case of vessels from Setúbal port, nets were the only fishing gear with enough observations to be modelled. These vessels were fishing in the Arrábida sector, especially in the area near the Portinho da Arrábida (see Figure 3B.1) with small trammel nets used as drift nets. The same procedure described above was conducted but no strong collinearity was found between predictors so they were all included in the model. In the ZANB model, counts from the variable distance to Setúbal port were not used as they were not significant and changed the response pattern when square rooted transformed. Despite that, the presence/absence component of the distance to Setúbal port was modelled without the need of being transformed.

All models were chosen after they were checked to evaluate their performance by analysing the maximum log-likelihood and the residuals plots. These analyses were conducted in software R 2.14.1 (R Core Team, 2012).

3B.3.3.2 Spatial analysis

The spatial distribution of fishing gears (buoys and vessels) were also analysed using point and area pattern statistics (Fortin and Dale, 2005). To that end, spatial analysis was divided in four steps: (i) Vessels' core areas; (ii) Vessels' distribution range and site fidelity; (iii) Seasonal spatio-temporal distribution of fishing effort; (iv) Hotspot analysis for each gear. Spatial patterns were investigated using geographic information system (GIS) modelling techniques with Arcgis 9.3x (ESRI) software.

i) Core areas for fishing vessels operating with nets and traps in the marine park were determined using the spatial distribution of buoys identified by the sea-based sampling. Standard deviational ellipses (SDE) polygons (Lee and Wong, 2001) covering approximately 68% (1 SDE) of the full spatial extent of those gears were calculated since, the setting of static fishing gears (nets and traps) is usually parallel to the coastline in a geometric shape assumed to be more similar to an ellipse than a rectangle, wider in the centre than at the ends of the distribution. Only vessels occurring at least in half of the sea-based sampling were used to determine the core areas in this analysis. This way, an average distribution from the most common vessels was applied to vessels not detected in the sea-based sampling but frequently observed in the shore-based surveys. The core area considered for each vessel was the average of all their sea-based SDE's.

ii) Vessels site fidelity was determined for nets and traps based on an index of reuse (IOR), adapted from the index modified by Morrissey and Gruber (1993) and used by Rechisky and Wetherbee (2003) for animal distributions. Vessels distribution range was used in replacement to the core areas utilised by these authors, being 95% of the full spatial extension of the buoys distribution (2 SDE) of the sea-based sampling data. This option allows understanding if fishers reused the same fishing grounds, including locations at the extremes of the distribution and not only in the core area. Vessels average distribution range ellipses were superimposed to the geographic positions of the respective vessels obtained regularly by the shore-based sampling (the ellipse was centred in the position recorded). This way, vessels point observations were replaced by an area referring to their average distribution range. For vessels not censused during the sea-based sampling, the distribution range considered was the average distribution range from vessels operating with the same gear type. Only vessels occurring at least in six observations in shore-based sampling were included in this analysis.

The IOR formula (Morrissey and Gruber, 1993), in this study referred to IOR_{95}, is given by:

$IOR_{95} = [OV (A_1 + A_2)] / (A_1 + A_2)$,

where $[OV (A_1 + A_2)]$ corresponds to the overlapped area (OV) between two distribution ranges, and $(A_1 + A_2)$ to the total area of both distribution range spaces (Morrissey and Gruber, 1993). IOR_{95} was performed for all possible sample combinations per vessel, and average IOR_{95} was used for site fidelity determination. According to Morrissey and Gruber (1993) an IOR of 1 indicates complete overlap of maximum activity spaces. If IOR = 0 movements are completely isolated, i.e., maximum activity spaces do not overlap.

The percentage of the total samples where overlap occurred within a vessel distribution range (2 SDE) was also computed for vessels included in the IOR_{95} procedure, dividing the number of pairs of samples in which there was some overlap by the total number of possible pairs of samples. Possible pairs of samples were calculated as the possible combinations among different samples where the vessel was detected.

The Kruskall-Wallis test ($\alpha = 0.05$) was conducted to compare the dimension of the vessels' core areas, distribution ranges and index of reuse (IOR_{95}) within each gear. A Mann-Whitney U test ($\alpha = 0.05$) was used to compare those variables and the percent of samples overlapped between gears (Zar, 1999).

iii) Spatio-temporal distribution of fishing effort density (FED = number of gears per vessel x number of vessels x km^{-2}) was calculated averaging samples (shore-based sampling data) within each season and sector by 1 km^2 square grid area for each fishing gear (nets, traps, jigs and longlines). Mean number of gears per vessel for each gear type (nets = 36; traps = 501; longlines - number of hooks = 507; jigs = 4) was obtained from Batista (2007).

iv) Hotspot analysis was performed in order to study the preferred fishing grounds of each fishing gear during spring, summer and autumn. This temporal analysis was conducted only for nets, traps and jigs, since longlines did not have enough records to include in the model (it was also not possible to model traps during the autumn due to few observations). This analysis was performed using the spatial statistics hot spot analysis tool of ArcGIS which uses the Getis-Ord Gi* algorithm (Ord and Getis, 1995). Statistical tests for significant spatial patterns (obtained by a Z score), were compared with the null hypothesis of complete spatial randomness (CSR) with a 95% and 90% confidence level (Z score is between -1.96 and +1.96 for 95% and between -1.65 and +1.65 for 90%) whereas the alternative hypothesis is that events are spatially clustered or dispersed. Significant clusters were defined as the aggregation of adjacent grid cells with a Z score consistent with the alternative hypothesis ($\geq |1.96|$ for $p < 0.05$ and ($\geq |1.65|$ for $p < 0.1$) of spatial clustering. Different confidence levels were used to allow the detection of significant clusters when few records were obtained. The larger the Z score, the more intense is the clustering of high values (i.e. a hotspot) whereas for negative Z scores, the smaller the Z score, the more intense is the clustering of low values (coldspot) (Ord and Getis, 2001).

To run these analysis, the best distance band was chosen based on global Morans I statistic for spatial autocorrelation (Ord and Getis, 2001). This tool provides a Z score for the entire study area, measuring spatial autocorrelation based on feature locations and attribute values. To perform Morans I statistic, 200 m was used as the starting distance and 1000 m as a cut off. The minimum distance has been chosen based on the grid size and the maximum length of the longest gear (nets). The conceptualization of spatial relationships used for the analysis was the zone of indifference. The final global Z scores were plotted against the Euclidean distance values and when the increase of the distance caused a decrease in the Z value (peak), that distance was selected as the best distance band to use in the hotspot analysis (250 m, 400 m , and 260 m for nets, traps and jigs, respectively).

3B.4 Results

3B.4.1 Zero inflated data modelling

Sesimbra port data

Zero-altered negative binomial model (ZANB) results from Sesimbra vessels fishing with nets (mainly trammel nets but also gillnets) were only dependent on the distance to port (Table 3B.1). This covariate was marginally non-significant for the abundance of this gear (count model) but it was significant for the presence/absence of these vessels (zero hurdle model).

Table 3B.6 - Zero-altered negative binomial model (ZANB) results of the effects of the distance to Sesimbra port (square rooted) and distance to coast in the distribution and abundance of vessels fishing with nets, traps and jigs. Significant values are in bold; n.s.: non-significant; marginally non-significant values $0.05 \leq p \leq 0.07$ are also indicated.

Sesimbra vessels		Count model		Zero hurdle model	
		z value	p-value	z value	p-value
Nets	Not explained	0.457	n.s.	-1.052	n.s.
	Distance to port	-1.899	0.058	-2.257	**< 0.05**
	Distance to coast	-0.251	n.s.	0.34	n.s.
Traps	Not explained	1.46	n.s.	-4.102	**< 0.001**
	Distance to port	-0.385	n.s.	4.701	**< 0.001**
	Distance to coast	-3.792	**< 0.001**	-3.705	**< 0.001**
Jigs	Not explained	1.383	n.s.	0.041	n.s.
	Distance to port	-2.526	**< 0.05**	-1.869	0.062
	Distance to coast	-0.463	n.s.	-2.719	**< 0.01**

Vessels fishing with traps showed that the abundance of this gear was significantly influenced by the distance to coast, but the presence or absence of a vessel operating with traps was highly dependent on both the distance to coast and to port. The occurrence of these vessels had a significant variability not explained by these models (Table 3B.1).

The abundance of vessels fishing with jigs was significantly related to distance to port, but this variable was marginally non-significant to their presence/absence (Table 3B.1). Distance to coast had a larger influence in the occurrence of these vessels.

Setúbal port data

ZANB results from Setúbal (Table 3B.2) revealed that no covariate influences significantly the abundance of vessels fishing with nets (count component). However the variable distance to port was not possible to model. On the other hand, this variable significantly influenced the distribution of vessels operating with nets (in this case is mainly drift nets) in the presence/absence comparison. Both components had however a significant portion of the variability which was not explained by these predictors.

Table 3B.7 - Zero-altered negative binomial model (ZANB) results of the effects of the distance to Setúbal port, distance to coast and depth in the distribution and abundance of vessels fishing with nets (in this case mainly small trammel nets used as drift nets). The variable Distance to Setúbal port was not possible to model in the count component (NA). Significant values are in bold; n.s.: non-significant.

Setúbal vessels	Count model		Zero hurdle model	
Nets	z value	p-value	z value	p-value
Not explained	6.294	**< 0.001**	3.972	**< 0.001**
Distance to port	NA	NA	-10.828	**< 0.001**
Distance to coast	-1.539	n.s.	0.644	n.s
Depth	-0.789	n.s.	-0.962	n.s

3B.4.2 Spatial analysis

Core areas

Four net vessels and five trap vessels were selected for the core areas analysis (Table 3B.3). Within each gear type, core areas did not show significant differences between vessels (Nets: $H = 2.614$; $p = 0.455$; d.f. = 3; Traps: $H = 4.459$; $p = 0.347$; d.f. = 4). However, traps showed significant larger core areas than nets (Traps mean = 1.5 ± 1.2 km^2, Nets mean = 0.8 ± 1.0 km^2; $U = 79$, $Z = –2.811$, $p < 0.05$).

Table 3B.8 - Mean core areas (km^2) ± standard deviation (sd) by vessel and by gear of the most frequent vessels fishing with nets and traps within the marine park. The number of samples (N) used to compute this analysis is also shown for each vessel.

Gear	Vessel	Core area (km^2)			
	Code	N	Mean	±	sd
Nets	N1	4	0.20	±	0.12
	N2	4	2.49	±	4.44
	N3	3	0.18	±	0.19
	N4	3	0.34	±	0.10
	Total	14	0.88	±	2.38
Traps	T1	6	0.67	±	0.57
	T2	4	0.69	±	0.79
	T3	4	2.10	±	2.19
	T4	5	2.55	±	2.17
	T5	6	1.24	±	1.20
	Total	25	1.41	±	1.55

Vessels site fidelity

Five net vessels and seven trap vessels were used in the analysis of the vessels distribution range and site fidelity (Table 3B.4). Within each gear type, significant differences were found between vessels distribution ranges (Nets: $H = 31.65$; $p < 0.05$; d.f. = 4; Traps: $H = 108.23$; $p < 0.05$; d.f. = 6) and IOR_{95} (Nets: $H = 17.65$; $p < 0.05$; d.f. = 4; Traps: $H = 268.66$; $p < 0.05$; d.f. = 6). Moreover, trap vessels showed significant larger distribution ranges than nets (Traps: mean = 1.48 ± 0.96 km^2; Nets: mean = 0.58 ± 0.08 km^2; $U = 921.5$, $Z = -6.32$, $p < 0.05$) (Table 3B.4). On the other hand site fidelity (IOR_{95}) was significantly higher for vessels fishing with nets when compared to traps (Nets: mean = 0.14 ± 0.08; Traps: mean = 0.10 ± 0.21; $U = 94387$, $Z = -2.292$, $p < 0.05$).

Table 3B.9 - Mean vessels distribution range (km^2) and IOR_{95} ± standard deviation (sd) by vessel and by gear of the most frequent vessels fishing with nets and traps within the marine park. The number of samples (distribution range) and pairs of samples (IOR_{95}) used to compute this analysis (N) are also shown for each vessel.

Gear	Vessel	Distribution Range (km^2)				IOR_{95}			
	Code	N	Mean	±	std	N	Mean	±	sd
Nets	N5	8	0,54	±	0,04	28	0,13	±	0,23
	N6	12	0,54	±	0,02	66	0,18	±	0,27
	N7	8	0,54	±	0,04	28	0,14	±	0,25
	N8	7	0,62	±	0,02	22	0,21	±	0,31
	N9	9	0,4	±	0,01	36	0,02	±	0,09
	Total	44	0,52	±	0,08	180	0,14	±	0,25
Traps	T1	16	0,58	±	0,1	136	0,19	±	0,26
	T3	7	1,77	±	0,06	66	0,15	±	0,25
	T4	22	2,77	±	0,56	378	0,06	±	0,16
	T5	15	0,95	±	0,21	120	0,09	±	0,2
	T6	18	0,35	±	0,07	153	0,02	±	0,08
	T7	22	1,23	±	0,2	231	0,07	±	0,16
	T8	18	2,45	±	0,46	69	0,39	±	0,3
	Total	118	1,48	±	0,96	1153	0,1	±	0,21

No significant differences were found in the percentage of pairs of samples in which some degree of overlap occurred between nets and traps (Nets: mean = 44 ± 18.5%; Traps: mean = 32 ± 16.7%; Table 3B.5).

Table 3B.10 - Mean percentage (%) of the total pairs of samples where some degree of vessels distribution range overlap occurred for each vessel and gear. The number of samples where the vessel was detected (N), the total number of possible pairs of different samples (POS) where the vessel was observed and the number of POS in which overlap occurred within vessels distribution ranges (2 SDE) are also shown for the most frequent vessels fishing with nets and traps within the marine park.

Gear	Vessel Code	% overlap	N	Possible POS	2 SDE POS with overlap
Nets	N5	54	8	28	15
	N6	48	12	66	32
	N7	54	8	28	15
	N8	52	7	21	11
	N9	11	9	36	4
	Mean	44			
Traps	T1	63	17	136	85
	T3	32	12	66	21
	T4	22	28	378	82
	T5	36	16	120	43
	T6	8	18	153	12
	T7	35	22	231	81
	T8	30	22	231	69
	Mean	32			

Spatio-temporal distribution of fishing effort and hotspot analysis

Nets fishing effort showed high densities in the Arrábida and Sesimbra sectors throughout the year, and it was very low or absent (autumn) in the Espichel sector (Figure 3B.2). In general, densities were higher in the spring, but the Arrábida sector showed a maximum in the summer (average of 12.2 nets x km^{-2}). The hotspot analysis revealed that the preferred fishing ground was in front of Sesimbra port, with a large significant cluster occurring in all seasons considered (Figure 3B.2). In the spring, and particularly during summer ($p < 0.1$), an aggregation was also detected in Portinho da Arrábida. The spatial distribution of nets was widespread throughout most of the marine park area in the spring, and a non-significant cluster was detected at the centre of the Arrábida sector in the autumn.

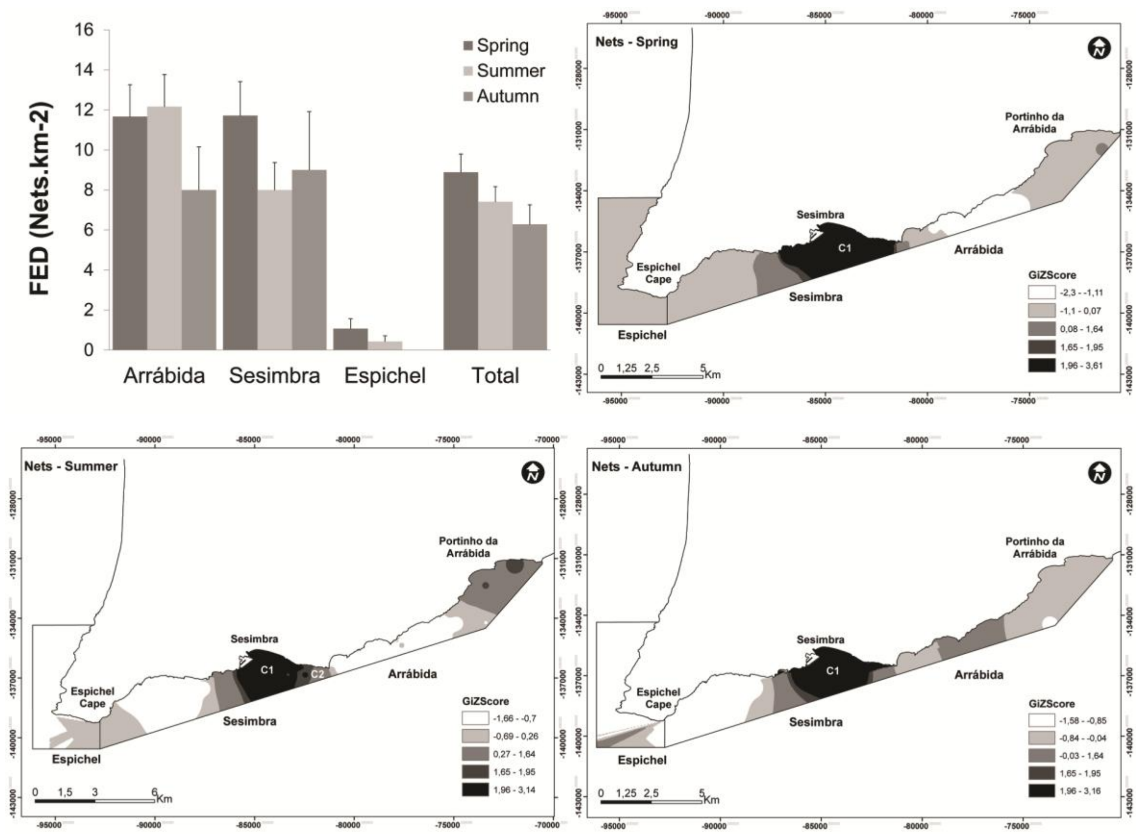

Figure 3B.7 - Average and respective error of the total fishing effort density of nets (FED = nº of nets x nº of vessels using nets x km^{-2}) by season and sector and maps obtained from the hotspot analysis, showing the location of significant clusters (GIZScore > 1.96 or > 1.65 for $p < 0.05$ and $p < 0.1$, respectively) by season: spring; summer; autumn.

Traps were found fishing within the marine park in large densities, reaching an average of 134.2 traps x km^{-2} in the Espichel sector in the spring (Figure 3B.3). This sector was the most densely fished throughout the year, but it was where the reduction in density was larger during the autumn. The hotspot analysis showed that in the spring, a dense (albeit non-significant) aggregation was detected north of the Espichel Cape, whereas in the summer a large significant cluster ($p < 0.05$) was found in the south part of the cape (Figure 3B.3). Other dense aggregation was found in the centre of the Arrábida sector during spring.

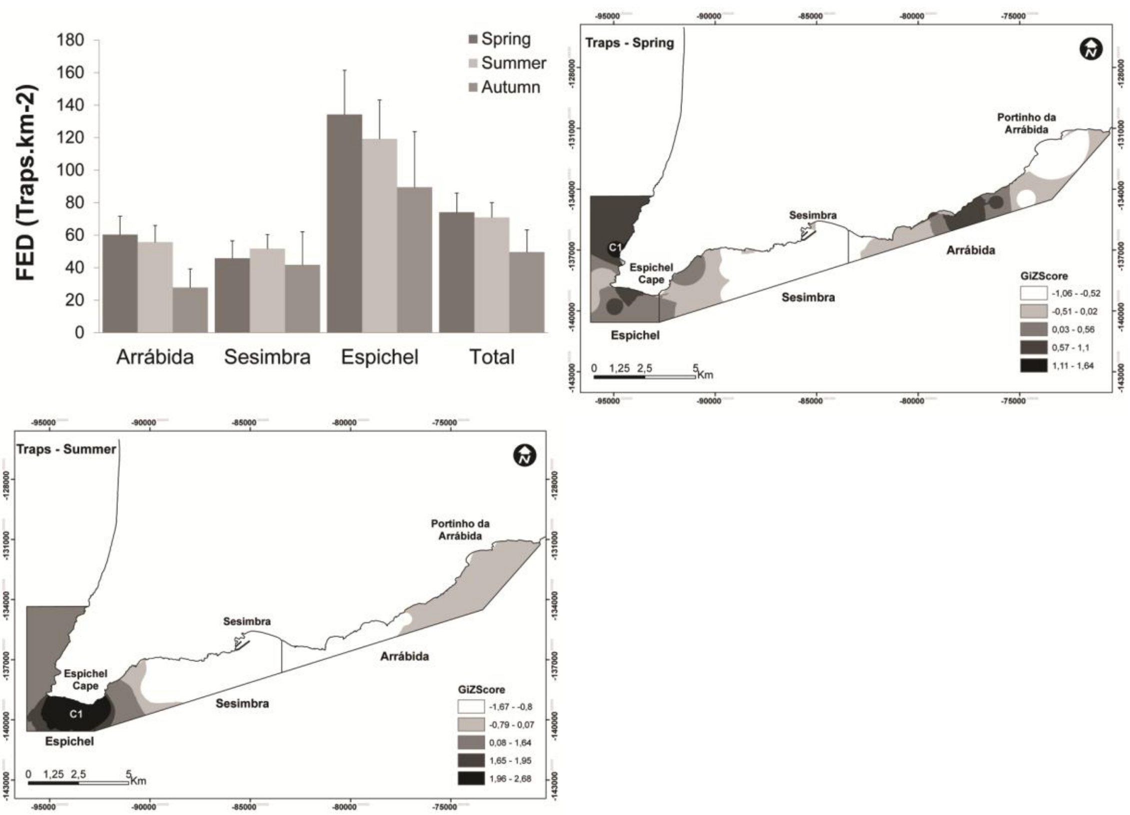

Figure 3B.8 - Average and respective error of the total fishing effort density of traps (FED = nº of traps x nº of vessels using traps x km^{-2}) by season and sector and maps obtained from the hotspot analysis, showing the location of significant clusters (GIZScore > 1.96 or > 1.65 for $p < 0.05$ and $p < 0.1$, respectively) by season: spring; summer.

Jigs were not detected north of the Espichel Cape (Figure 3B.4), but they were seen in the Arrábida and Sesimbra sectors throughout the year. The density of vessels fishing with this gear was higher in the spring at the Arrábida sector (average of 4.2 jigs x km^{-2}), and in the summer at the Sesimbra sector. The hotspot analysis (Figure 3B.4) showed a large significant cluster in front of Sesimbra throughout the year (at $p < 0.05$), which was split into two smaller significant clusters during autumn ($p < 0.1$). Despite of being non-significant, jigs aggregations were also detected in the centre of the Arrábida sector, extending to Portinho da Arrábida in the spring and autumn.

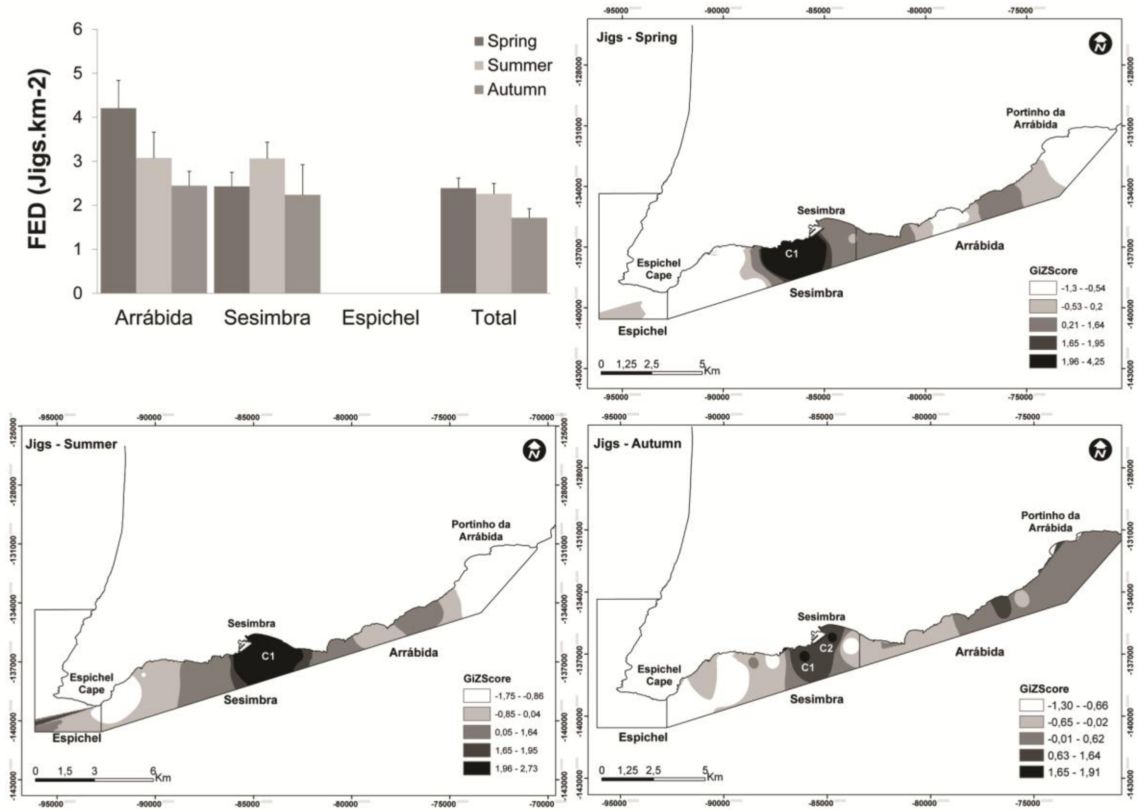

Figure 3B.9 - Average and respective error of the total fishing effort density of jigs (FED = nº of jigs x nº of vessels using jigs x km^{-2}) by season and sector and maps obtained from the hotspot analysis, showing the location of significant clusters (GIZScore > 1.96 or > 1.65 for $p < 0.05$ and $p < 0.1$, respectively) by season: spring; summer; autumn.

Longlines occurred mainly in the Espichel sector, with a higher fishing effort during the summer (average of 69.4 hooks x km^{-2}). The density of this fishery was always higher than in the other sectors, increasing from spring to summer and decreased in the autumn (Figure 3B.5).

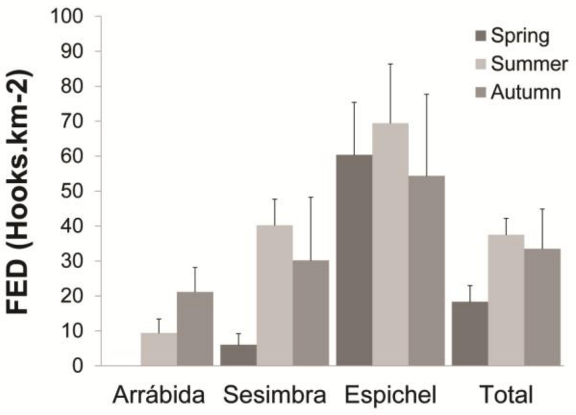

Figure 3B.10 - Average and respective error of the total fishing effort density of longlines (FED = nº of hooks x nº of vessels using longlines x km^{-2}) by season and sector at the Arrábida Marine Park.

3B.5 Discussion

The present case-study is one of the few quantitative surveys of artisanal fishing effort distribution using direct observations and a set of statistical methods to find patterns related to the different preferences in fishers' choices. It shows how artisanal fisheries operate before the implementation of the management plan in a temperate multiple-use Marine Protected Area (MPA). This is one of the few empirical studies where temporal and spatial preferences of local fishers using multiple gears were analyzed before regulations and zoning have been implemented. The baseline information on the dynamics of fishing effort and the main factors influencing fishing allocation and fishers' strategies when selecting fishing grounds, previously to the implementation of this MPA, provides valuable data for both fisheries and MPA managers and practitioners. In fact, the majority of studies addressing fishing effort distribution in relation to MPAs are performed after implementation (Goñi *et al.*, 2008; Stelzenmüller *et al.*, 2008; Forcada *et al.*, 2010; Goñi *et al.*, 2010), with very few studies incorporating data previous to protection. These addressed fishers' adaptations to large trawl closures in temperate systems (Murawski *et al.*, 2005; Abbott and Hayne, 2012) or rezoning of artisanal fisheries in large tropical MPAs (Campbell *et al.*, 2012; Lédée *et al.*, 2012).

Small-scale artisanal fisheries comprise the greatest percentage of fishing communities of coastal multiple-use MPAs (the vast majority of MPAs around the world, UNEP-WCMC, 2008) and the factors influencing fishers' preferences are of major importance to management decisions and to implement ecosystem-based management approaches.

In the present study, distance to port was the most important factor to nets allocation with a significant large aggregation consistently found near port. This fishery is very dependent on weather since nets need to be in good condition to optimize catchability. With rough seas, if fishers do not haul their gears, they may have to spend a hard, time consuming and expensive work in repairing them. Thus, during the autumn the exposure to strong winds and waves near the Espichel Cape is enhanced and consequently fewer boats were observed than in spring and summer months. This fact may influence their occurrence near port, where they can go easily, safer and cheaper. Moreover, the area in front of port has extensive shallow sandy bottoms where target species for trammel and gill nets occur (soles, cuttlefish and several other target species such as sparids and rays) (Batista *et al.*, 2009).

The second largest aggregation of net vessels was detected in the very shallow sandy banks at the east end of the park, during spring and summer. These were mainly drift nets from Setúbal port, fishing illegally, since they can only operate beyond ¼ nautical miles from shore. Distance to port also strongly affected their occurrence in this preferred fishing ground. Furthermore, this area is near the entrance of the Sado estuary, an important spawning and nursery area (Vasconcelos *et al.*, 2010). Drift nets target various species, but in that area they captured mainly cuttlefish, octopus and soles which are valuable commercial species migrating inshore during spring to breed (Roper *et al.*, 1984; Ramos *et al.*, 2010), thus explaining the seasonal variation of effort in this fishing ground.

The small core areas and fishers' distribution ranges of net vessels as well as the high level of site fidelity supports that this fishery has very well defined fishing grounds, occupying the sandy areas of the marine park. This indicates that, besides the importance of the proximity to port, traditional routines and fishers' knowledge about target species and habitats are also decisive in the spatial and seasonal allocation of this fishery. Nevertheless, net vessels showed similar core areas but different distribution ranges among vessels, revealing that, in spite of a high site fidelity, different fishers used different strategies with some vessels dispersing their effort throughout the marine park whereas others selecting restricted fishing grounds.

On the other hand, trap vessels showed a broader average core area and distribution range, which translated into a smaller site fidelity than nets. Although evidencing a smaller reuse of fishing grounds than nets, the percentage of samples among which some overlapping occurred was considerably high, suggesting that fishers moved and explored different zones in a preferred larger area. Each set has a large number of traps, and thus higher densities were detected throughout the year when compared to nets (although the impacts of the two fisheries cannot be compared directly). Trap fishing is one of the most important artisanal fisheries locally and appear to be more resilient to weather than nets, selecting suitable fishing grounds farther away from port, namely exploring the large area of rocky reefs around Espichel Cape (Gonçalves *et al.*, 2003) and avoiding the spatial competition with nets. These factors support the significant clusters detected around the Espichel Cape. Dense aggregations of traps were found both north and south of this cape during spring, and in the summer a large significant aggregation was also detected south of the cape. This shift is probably due to the strong north winds that are more frequent during the summer (Henriques *et al.*, 2007) with the cape offering protection from these adverse weather conditions. Additionally, the hotspot analysis of autumn samples was not possible due to few records of trap vessels, although fishing effort density revealed their presence in the Espichel sector. The second preferred fishing ground was located at the centre of the Arrábida sector, especially in the spring. Seasonal variation in the effort density of this fishery is consistent with the breeding migration in early spring to inshore habitats of the most important target species occurring in traps, *Octopus vulgaris* Cuvier, 1797 (Roper *et al.*, 1984).

A strong relation between the presence of trap vessels and the distance to port was found, but the variable distance to coast was also associated with trap vessels' presence and abundance. This variable has rarely been assessed possibly due to its strong correlation with depth in most systems, being the latter a central predictor of species abundances and, therefore, gear allocation. However, distance to coast is also probably an important factor for small boats in local artisanal fisheries where weather conditions may be challenging and lack of suitable habitat may limit their use more offshore.

Jigging for the cuttlefish *Sepia officinalis*, Linnaeus, 1758 and the squid *Loligo vulgaris* (Lamarck, 1798) from small (3-4m) wooden vessels is a traditional artisanal fishing activity in the region. The nature of this fishery and its vulnerability to weather conditions (Forcada *et al.*, 2010), explains why fishing with jigs was not detected in the more exposed Espichel sector. Moreover, in this region they are also limited to a range of 3 nautical miles from their home port by a local legislation and by logistical constrains to operate in deep areas. Jigs therefore occupied a large portion of the marine park in Sesimbra and Arrábida sectors, where very sheltered conditions prevail (Gonçalves *et al.,* 2003). Distance to port explained the density allocation of vessels fishing with jigs, whereas distance to coast influenced the presence and absence of this fishery. This is consistent with both the spatial distribution of these vessels near port and the way they operate by drifting with the alongshore tides, catching cephalopod species occurring in nearshore environments. In spring, the sharp density increase of vessels in the Arrábida sector indicates that fishers are targeting cuttlefish which migrate inshore to breed and use the nearby estuary as a nursery area.

Finally, longlines were mainly found in the Espichel sector with densities increasing in the summer and being lower in the autumn, suggesting some dependence on good weather conditions on this exposed fishing ground. Since longline vessels are usually very small, they are also limited by weather and safety requirements. This fishery targets however mobile fish such as sparids (Erzini *et al.*, 1998; Erzini *et al.*, 2003) which occur on rocky reefs and are abundant in the vicinity of the Espichel Cape (unpublished data). Therefore, the choice of fishing grounds for longline fishers seems to be largely influenced by the target species distribution and not to proximity to port or shelter conditions.

Different factors are here shown to drive the distribution of artisanal coastal fisheries, explaining their temporal and spatial dynamics. The location of fishing grounds and distribution of target species and adequate habitats are main drivers of fishing effort allocation. However, proximity to port, weather conditions and distance to coast are also important features influencing in different ways these artisanal fisheries. Traps are more resilient to weather and thus are able to avoid densely fished areas near port. On the other hand, nets and jigs, for distinct reasons, usually do not operate far away from their home port. Nets are restricted by gear performance and habitat and jigs by the safety conditions associated to these very small vessels. Fishing with longlines, on the contrary, seem to depend greatly on habitat requirements of target species with these small vessels risking to fish in the most exposed fishing grounds.

Temporal patterns were also striking for several of these fisheries with seasonal movements suggesting an optimization of the target species catches and also a strong influence of exposure to rough sea conditions. Drift nets and jigs occurred mainly on the shallow sand banks of Portinho da Arrábida during spring, when their target species migrate inshore and to the nearby Sado estuary. Traps were also found in considerable densities in the central part of the marine park in the spring, since target species (mainly octopus) breed nearshore, in the surroundings of the rocky reefs of the marine park.

The study of the spatial and temporal dynamics of small scale artisanal fisheries and the factors influencing fishers' preferences are key aspects to incorporate in the management of these fisheries as well as in the implementation of multiple-use marine protected areas. Namely, new management regimes that displace fishers from their fishing grounds need to take into consideration their fishing preferences in order to evaluate the possible impacts of regulations on both the ecological, economic and social (cultural) dimensions. Additionally, fishers' knowledge and perceptions may likely contribute to better interpret those fishing preferences and regulations impacts (McClanahan *et al.*, 2005; Campbell *et al.*, 2012; Lédée *et al.*, 2012; Leleu *et al.*, 2012; De Freitas *et al.*, 2013). However, when comparing face-to-face interviews and direct observations after a rezoning of a tropical MPA, it became clear that fishers were reluctant to self-report spatial infringements (Campbell *et al.*, 2012), a situation certainly widespread throughout the world. Therefore, empirical studies assessing artisanal fisheries distribution through direct observations are a necessary complement to evaluate fishers' choices and behaviour.

This study highlights that artisanal fisheries' dynamics are complex to evaluate, and to relate to biological and ecological reserve effects, since different gears with distinct target species are driven by different strategies. Fishers' choices may also differ within fishing gears and thus affect their adaptations after the loss of fishing grounds. In spite of this complexity, species-specific impacts should be carefully evaluated when understanding protection effects, especially for those species most targeted by fisheries. In fact, understanding fishing effort displacement, which requires assessing fisheries allocation before changes in management rules, is an important step to better interpret the responses of fisheries and the species they target. This information is however lacking from most of the studies trying to evaluate the effectiveness of protection measures associated with marine protected areas or other management regimes in small scale artisanal coastal fisheries.

This is one of the few empirical studies focusing on the before data of artisanal fisheries allocation. Moreover, a novel approach has been applied combining different statistical methods infrequently used in fisheries distribution studies: zero inflated models were preferred to understand which factors influenced the presence and absence of fishing gears besides their abundance (most studies remove un-fished grids); hotspots analysis was selected due to its potential to statistically detect the spatial patchiness of fishing gears (most studies analyze only density patterns); site fidelity and vessels distribution range were also calculated to understand fishers' choices among and within fishing gears (these were adapted from methods studying animal moving patterns). Our findings and the methodology here described to assess the spatio-temporal dynamics of these fisheries can be used in most coastal marine protected areas, especially in similar multiple-species artisanal fisheries. When integrated with the analysis on the main factors influencing the preferences of fishers using different gears, these methods also constitute powerful tools to a better assessment of fisheries data on conservation and management decisions and therefore to implement an ecosystem-based management approach.

3B.6 Acknowledgments

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3B.8 Supplementary Information

3B.8.1 Text 3B.S1 – Description of the Topographic measurements

In order to apply the triangulation method (Davis *et al.*, 1981), georeference control points (with known coordinates and elevation) were selected for each station in the coastline. Additionally, observation stations coordinates were determined with a GPS (MAGELLAN Meridian model).

Before sampling, the theodolite was precisely mounted on its tripod head and placed vertical above the sampling station using a plumb bob. The instrument was then set level using tubular spirit bubbles. With the equipment properly calibrated, the vessels locations were determined through the following steps (see Figure 3B.S1):

1) Determination of the horizontal distance (Hd_1) and route (RA) between the device (e) and the control point (a) from:

$Hd_1^2 = (M_A – M_E)^2 + (P_A – P_E)^2$

$Tg\ R_A = \underline{(M_A – M_E)}$

$(P_A – P_E)$

2) Device height determination (h):
2.1) Zenithal angle determination (z), measured between the device and the top of the control point.
2.2) Determination of the vertical angle (v) formed between the device and the control point from: v = 100 (grads) – z
2.3) Gap (dN) determination between the device and the control point from: dN = sen (v) x Hd_1
2.4) knowing dN and the control point elevation (H), the device elevation was determined from: h = H – dN
3) Determination of the horizontal distance (Hd_2) between the device (E) and the observed vessel (b) through the following steps:
3.1) Zenithal angle determination (z), measured between the device and the vessel (b)
3.2) Determination of the vertical angle (v) formed between the device and the vessel from: v = z -100 (grads)
3.3) Hd_2 determination from: Hd_2 = h/sen (v)
4) Route determination between the device and the vessel (R_{EB})through the following steps:
4.1) Zenithal angle determination (Z_A), measured between the control point and the vessel (b)
4.2) Route calculation from: $R_{EB} = R_A + Z_A$
5) Vessels coordinates (M,P) from: $M_b = M_E + Hd_2$ x sen (R_{EB}) and $P_b = P_e + Hd_2$ x cos (R_{EB})

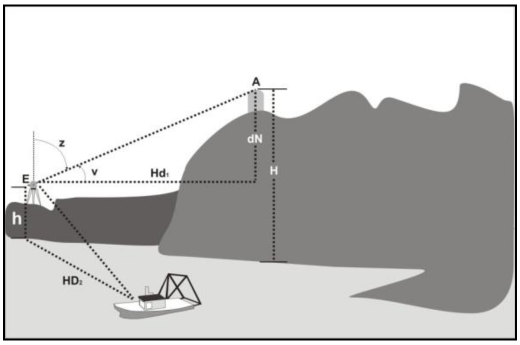

Figure 3B.S2– Schematic representation of the topographic measurements applied in the shore-based surveying method.

CHAPTER 4: CLIMATIC INFLUENCE IN THE ROCKY REEF FISH ASSEMBLAGES OF THE ARRÁBIDA MARINE PARK

CHAPTER 4A: TROPICALIZATION OF FISH ASSEMBLAGES AT TEMPERATE BIOGEOGRAPHIC TRANSITION ZONES

Submitted to *Marine Ecology Progress Series*

Horta e Costa, B., Assis, J., Franco, G., Caselle, J. E., Erzini, K., Henriques, M. & Gonçalves, E. J. *submitted*. Tropicalization of fish assemblages at temperate biogeographic transition zones. *submitted to MEPS*.

4A.1 Abstract

Biogeographic transition zones in marine temperate systems are often hotspots of biodiversity with high levels of resilience to temporal climate shifts due to the cyclic oscillations of oceanographic conditions. However, these environments are vulnerable to a steady warming scenario when cyclical conditions are disrupted. Here we evaluate how changes in local oceanography affect the structure of rocky reef fish assemblages over a period of 50 years in a biogeographic transition zone.

We used visual census data of rocky reef fish species to understand the most important oceanographic variables influencing the assemblages' dynamics. Descriptive and predictive models (multivariate regression trees, MRTs) were compared to observed data. Winter northward wind stress and sea surface temperature (SST) were the most important drivers of changes in community structure. In the MRT only warmer years had indicator species, with warm-temperate or tropical affinities. A fish community 'tropicalization' index was developed in response to both high spatial resolution but short-term environmental variation (1993-2011) and to regional long-term SST (1960-2012). Predictive modelling for the last 50 years revealed that species with tropical affinities are increasing in relation to cold-temperate ones, coinciding with the trend of increasing mean winter SST. Since the mid-eighties, warm-temperate and tropical species are responding rapidly to more frequent warm winters, suggesting that species distributions are shifting polewards. Our results support that cold-species retreat slower than the advance of warm species and possible reasons are discussed. We highlight the importance of transition zones as 'barometers' of climate change.

Keywords: marine biogeographic transition zone, resilience, climate change, tropicalization, fish assemblages, species distribution shifts

4A. 2 Introduction

Global warming effects on marine ecosystems are motivating increasing efforts to conduct research across different ecosystems. Increasing temperature will affect the physiological performance of marine organisms, and may be especially important to ectotherms, which are the vast majority of marine species (Pörtner and Peck, 2010; Heath *et al.*, 2012). Marine fish and other organisms are expected to experience altered growth rates, metabolism, reproductive behaviour and outputs, habitat and food requirements, as well as movement patterns (Lafrance *et al.*, 2005; Perry *et al.*, 2005; Caputi *et al.*, 2010; Pörtner and Peck, 2010), as organisms search for suitable habitats and optimal physiological conditions (Pörtner and Peck, 2010; Heath *et al.*, 2012). Moreover, early life stages such as eggs and larvae often depend strongly on particular environmental features to disperse, survive and find suitable settlement habitat and thus may also be significantly affected by oceanographic changes (Perry *et al.*, 2005; Munday *et al.*, 2008). A primary predicted effect of ocean warming is a shift in species abundance and distribution ranges which may cause dramatic changes in community assemblages and trophic webs (Perry *et al.*, 2005; Cahill *et al.*, 2012). Latitudinal shifts in marine species ranges have been already widely described (Southward *et al.*, 1995; Brander *et al.*, 2003; Perry *et al.*, 2005; Cheung *et al.*, 2009; Hawkins *et al.*, 2009; Figueira and Booth, 2010; Cheung *et al.*, 2012; Nicastro *et al.*, 2013; Wernberg *et al.*, 2013) and shown to affect ecosystems and fisheries (Sumaila *et al.*, 2011; Cheung *et al.*, 2013). Connectivity patterns of marine systems and species' mobility and dispersal mechanisms, result in much larger range among some marine fauna when compared to terrestrial species (Heath *et al.*, 2012). Distinguishing between natural oscillations and any added effects of human-induced warming is a major challenge but critical to understanding climate effects in marine ecosystems.

Marine biogeographic transition zones present higher diversity and potential resilience due to "normal" cyclic oscillations of climatic conditions and species distribution limits (Henriques *et al.*, 1999; Henriques *et al.*, 2007; Bernhardt and Leslie, 2013). These areas can act as important field 'laboratories' to detect and distinguish organism responses to natural climatic fluctuations versus continuous ocean warming. Analyzing past trends of species adaptations and community structural changes in response to natural variability, and in particular more frequent warming periods, may contribute to better understand future consequences of sustained global warming.

Oceanographic properties that have been shown to drive changes in coastal marine communities and species distribution limits are temperature, winds and currents. Wind stress is an important driver of surface currents and upwelling events on the western coasts of the world continents (Relvas *et al.*, 2007; Sánchez *et al.*, 2007). Some systems such as the Iberian Atlantic west coast have seasonal upwelling, in which, during summer, strong southward winds and currents induce Eckman transport eastwards ((Relvas *et al.*, 2007; Sánchez *et al.*, 2007), facilitating the movement of deep, cold and nutrient rich-water into the surface, and leading to a decrease in sea surface temperature (SST) and sea surface height (Relvas and Barton, 2005) as well as an increase in coastal productivity (often tracked by the concentration of chlorophyll a, Cravo *et al.*, 2010). During winter, weaker southward winds and currents do not sustain upwelling events and sometimes counter currents and northward storms prevail (Wooster *et al.*, 1976; Sánchez *et al.*, 2007). Similar dynamics occur on the western coasts of North America, South America (Mendelssohn and Schwing, 2002; Bakun *et al.*, 2010), South Africa (Hutchings *et al.*, 2009) and New Zealand (Chiswell and Schiel, 2001; Blanchette *et al.*, 2009) and have been shown to drive recruitment dynamics, predator-prey relationships and community structure (Iles *et al.*, 2012; Menge and Menge, *In press*).

The importance of oceanographic drivers measured at small spatial scales on marine communities are, in general, less well understood than regional or even global effects (Caselle *et al.*, 2010; Selig *et al.*, 2010; Langlois *et al.*, 2012), perhaps due to the scarcity of fine-scale and long time series data. The west coast of Portugal is an important temperate biogeographic transition zone (Henriques *et al.*, 1999; Boaventura *et al.*, 2002; Lima *et al.*, 2007). Seasonal variability in oceanographic conditions has been well studied (Wooster *et al.*, 1976; Relvas *et al.*, 2007; Sánchez *et al.*, 2007). Ecologically this region marks the northern and southern distribution limits of species with warm and cold affinities, respectively (Henriques *et al.*, 1999; Henriques *et al.*, 2007; Lima *et al.*, 2007). This area is also near the northern limit of the main NE Atlantic upwelling events (Wooster *et al.*, 1976). The importance of the large-scale North Atlantic Oscillation (NAO) index in driving inter-annual variability of this region has also been described (Hurrell, 1995; Henriques *et al.*, 2007).

Here, we move beyond the usual use of average conditions and investigate extremes as well as variance since they may be even stronger drivers of ecosystem-level changes in population structure and in community composition of marine regions (Pörtner and Peck, 2010). Our main goal was to detect the importance of local-scale oceanographic variables and their temporal relationship with regional and large-scale climatic features and hindcast their influence in fish community structure for the last 50 years.

4A.3 Methods

4A.3.1 Study area

The Arrábida Marine Park is a 38 km stretch of coastline (53 km^2) on the west coast of Portugal (Figure 4A.1). The habitats present in this Park support a high diversity of algae, invertebrates and fish, totaling more than 1320 marine species (Henriques *et al.*, 1999, Arrábida Marine Park Authority/ICNF 2012) making this area an important hotspot of biodiversity for this biogeographic region (Henriques *et al.*, 1999; Gonçalves *et al.*, 2003)Henriques et al. 1999, Gonçalves et al. 2003). Subtidal rocky reefs are shallow and narrow (≈100-150 m width) in most of the park (Horta e Costa *et al.*, 2013).

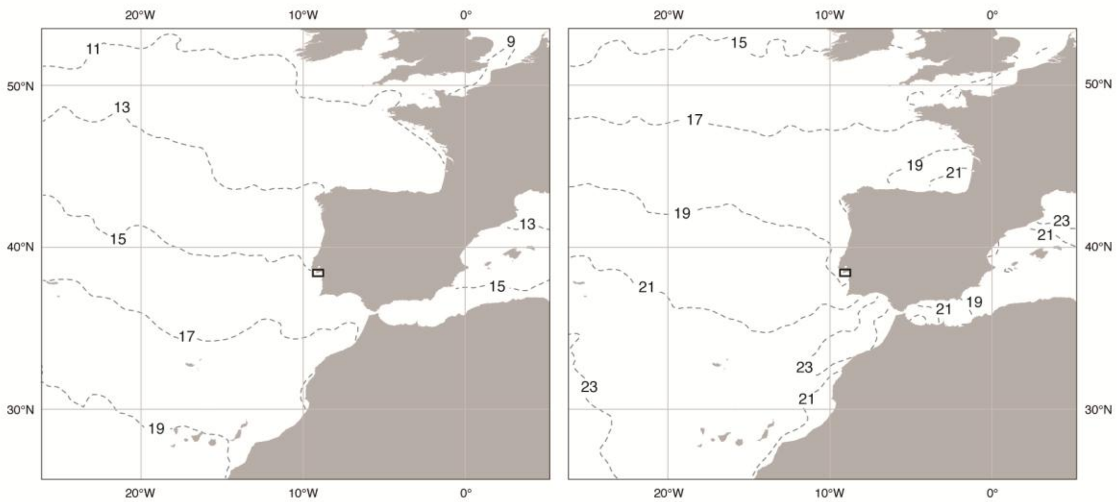

Figure 4A.5 - Location of the study area (Arrábida Marine Park, Portugal). Mean winter (December to April; left panel) and summer (June to September; right panel) sea surface temperature (SST, ºC) for the period 1992-2012 is shown.

4A.3.2 Data collection

Fish assemblages were surveyed using SCUBA from May 1992 to December 2002 (Henriques *et al.*, 2007). We made approximately 30 dives per year with each dive lasting 60 min, beginning 10 m offshore the rocky substrate in the sandy area and ending at the intertidal. Surveys were carried out by two divers each of whom searched all of the available habitat and recorded every species. This procedure was repeated in 2010 (n= 36 dives).

Fish species were grouped by their climatic affinity following Henriques *et al.* (2007). These authors classified fish biogeographic climatic affinities as Tropical (Tr), Warm-temperate

(WT), Temperate (T), Cold-temperate (CT) and Eurythermic (E) based on species distributions. Some species were identified only to higher taxonomic levels and, in this case, the genus or family was considered in the analysis if co-occurring species had the same climatic affinity (Table 4A.S1).

Oceanographic data was obtained for a location at the centre of the marine park with a resolution of 9 km (hereafter called 'high resolution'), based on remote sensing and direct observations (Table 4A.S2): sea surface temperature (SST, Cº), eastward (U) and northward (V) wind stress (WINDst) components ($N.m^{-2}$), significant wave height (SWH, m), sea surface height (SSH, m) and Chlorophyll a (Chla, $mg.m^{-3}$). Two seasons were considered: winter (December to April), and summer (June to September of the previous year) (Henriques *et al.*, 2007). For each oceanographic variable, the annual mean, minimum and maximum values for winter (variables 1 to 3) and for the summer (variables 4 to 6) were calculated. Winter (7) and summer (8) deviations (above or below) from monthly mean ± 1 standard deviation (sd) as well as winter and summer number of days above and below (winter + (9); winter – (10); summer + (11); summer – (12)) long term monthly mean ± 1 sd were calculated.

Additionally, the mean North Atlantic Oscillation (NAO) was also obtained annually for winter (13) and previous summer (14) from NOAA (Table 4A.S2). A dummy variable for seasonal NAO (variables 15 to 16) was also used to refer to positive NAO (1) and negative NAO (0). Strong positive NAO values indicate cold years in southwest Europe and in the Mediterranean, and a negative NAO indicates warm years (Hurrell, 1995).

The small spatial scale data described above were only available for the period 1992-2011 (or 1997-2011 for Chla). To create a long time series for use in hindcasting fish community structure, we used SST and wind stress from ICOADS (1-degree grid resolution) for the period 1960 – 2012 (Table 4A.S2). The NAO data was also obtained for this long-term period.

4A.3. 3 Data analysis

4A.3.3.1 Descriptive models

To understand which oceanographic variables, at the local scale, were influencing the presence and absence of Arrábida rocky reef fish species, multivariate regression trees (MRT) were run using multivariate partitioning (De'Ath, 2002, 'mvpart' package v1.6–0, R Development Core Team, 2012).

A model was run for each type of the 6 explanatory variables (e.g. SST, SSH, WINDstV, etc.) to choose which of the 12 components of the variable (e.g. winter min, mean, etc.) were the best predictors. This step prevented us from generating a huge matrix of collinear variables. NAO was tested separately using the four components described above (variables 13 to 16: seasonal average and seasonal dummy variable).

All non-collinear (i.e. correlations below |0.6|) combinations of variables (1035 of 12985) were run to choose the set of variables that minimize the cross-validated relative error (CVRE, De'Ath, 2002) of the tree. NAO predictors were not included in this 'local' model run. This process was repeated 1000 times to increase confidence in the model choice. The final model was chosen as the one with lowest CVRE (cross-validated relative error). Since different sets of variables had the same CVRE, those were compared based on the relative error. Finally, the most frequently selected model (with the lowest CVRE and relative error) was chosen. The most important variable (mean winter wind stress northward component or V: WINDstV2) was collinear with an important predictor also frequently selected during the process of the model choice (mean winter sea surface temperature: SST2), although with a CVRE one-hundredth part larger than in the previous model.

4A.3.3.2 Indicator species

Indicator species from each tree split and leaf, and discriminant species for each tree node were calculated for the model selected using MVPARTwrap package v0.1-9 (R Development Core Team, 2012). The indicator index is based on the relative abundance and frequency of occurrence of each species, varying from 0 to 1. If a species occurs in one group but is absent from the others, its indicator value (i.v.) is 1, and if a species is absent within a group its i.v. is 0 (De'Ath, 2002). For the present study, only species with indicator values above 0.5 were chosen, indicating the presence of such species in a cluster.

The contribution of individual species (percentage of explained variance) to each split and to the total tree was also computed. Species with zero contributions to any split are the ubiquitous species present throughout years.

4A.3.3.3 Semblance analysis

To understand temporal relationships among the most important predictors, a paired semblance analysis was conducted between the main candidate predictors using the Wavelet Toolbox from Matlab software (MATLAB, 2011). This produces a cross-correlation (|0 - 1|) plot between two time series as a function of both time and wavelength (Cooper and Cowan, 2008). This analysis used the NAO dataset and the high resolution (9 km) and 1-degree datasets of the main predictor and its collinear oceanographic variable for the short-term (1993-2011) and for the long-term (1960-2012) periods. Pearson's correlation was also calculated for the same pairs of explanatory variables.

4A.3.3.4 Predictive models

Since predictive models do not cope with distance matrixes such as those used in the selection of the best descriptive model, the chosen model was repeated without transforming the response variables, since the purpose was to predict the presence/absence of individual species.

Long-term species hindcasting were then modeled using the set of high-resolution (9 km) explanatory variables selected in the best MRT descriptive model for the short-term period (1993-2011). However, since the main driver selected was collinear with other important and commonly used oceanographic variables, two additional models were run: a) a model substituting the main predictor WINDstV2 by the collinear predictor SST2 and b) a model using a combined predictor (created with a principal components analysis of WINDstV2 and SST2 and utilizing the first component (PC1)) as the main predictor.

Long-term hindcasting could not be done with high-resolution oceanographic variables since they were not available prior to 1992. However, since the purpose of this study was to detect fish community responses to local drivers, only available regional (1-degree) oceanographic variables found to be highly correlated with the correspondent high-resolution drivers were considered to reflect local variability and were used as proxies to predict community structure for the long term analysis (1960-2012).

Since the results of the predictive models are a continuous response instead of resulting in a presence or absence for each species, which was the structure of the observed data, a threshold on the continuous response had to be defined. Response values falling above or below a particular value (threshold) are respectively classified as a species presence or absence. Twenty levels of threshold were simulated and the best threshold was chosen based on the simultaneous maximization of the specificity and sensitivity of the predictive model (Manel *et al.*, 2001).

The performance of the short- and long-term predictive models was evaluated by measuring the agreement between the observed and modeled communities from the common years (1993-2002, 2010) through the mean Area Under the receiver-operated characteristic Curve (AUC), as recommended by Allouche *et al.* (2006). Mean AUC was obtained averaging AUC from each year. When a predictive model estimates the presence of a species and that species was observed for that year, the value of AUC is 1; AUC is 0.5 if the model fails. A perfect match is considered with AUC values above 0.8 (Hosmer and Lemeshow, 2000). Additionally, a Wilcoxon rank sum test was performed to the mean AUC to statistically compare the power between the selected short-term high-resolution models.

4A.3.3.5 Community indices

We developed a 'tropicalization' index for our fish species data adapted from Wernberg *et al.* (2013) who measured the proportion of tropical species. In the present study the tropicalization index was calculated as the ratio between the sum of the tropical and the sum of the cold-temperate species. We used these groups since they are most likely to have their northern and southern range limits in this transition zone and were previously shown to contribute to distinctive warm and cold fish assemblages among years (Henriques *et al.*, 2007).

The tropicalization index was calculated using the observed and modeled annual community data for the high-resolution (9 km) datasets of the best predictive models for the short-term period (1993-2011) and for the related 1-degree datasets for the long-term period (1960-2012). However, to account for possible differences of both main predictors, hindcastings of the tropicalization index (for the period 1993-2011) using the two high resolution datasets were also generated by merging the resulting projections of the main model with those of the alternative model (using the collinear predictor), weighing their correspondent AUC (i.e. ensemble modeling; (Araújo and New, 2007). This approach is often more robust than predictions based on the result of a single model. This way, predictions indicate the consensus of the two highest discriminatory sets of predictors. We did not include the model with PC1 for this analysis.

4A.4 Results

During the 12 years of underwater surveys, 95 species (or groups of species) from 35 families were observed and included in the multivariate regression tree (MTR) modeling (Table 4A.S1). From those, 45 species were considered to have Warm-temperate affinities, 25 were Temperate, 12 were Tropical, 11 were Cold-temperate and 4 were Eurythermic.

Descriptive models

From the 12 components x 6 variables tested in the MRTs, mean winter values for the northward component (V) of wind stress (WINDstV2) and sea surface height (SSH2), and minimum winter significant wave height (SWH1) were selected as the most important set of explanatory variables for the rocky reef fish assemblages at the Arrábida Marine Park.

Mean winter North Atlantic Oscillation (NAO) was the component selected when modeling the various NAO metrics alone.

The best MRT had three nodes explaining 44.3% of the total species variance (Figure 4A.2). This tree was headed by mean winter wind stress northward component (WINDstV2), explaining 18.43%, and the second and third nodes were driven by SSH2 and SWH1, explaining 13.34% and 12.53% of species variance, respectively.

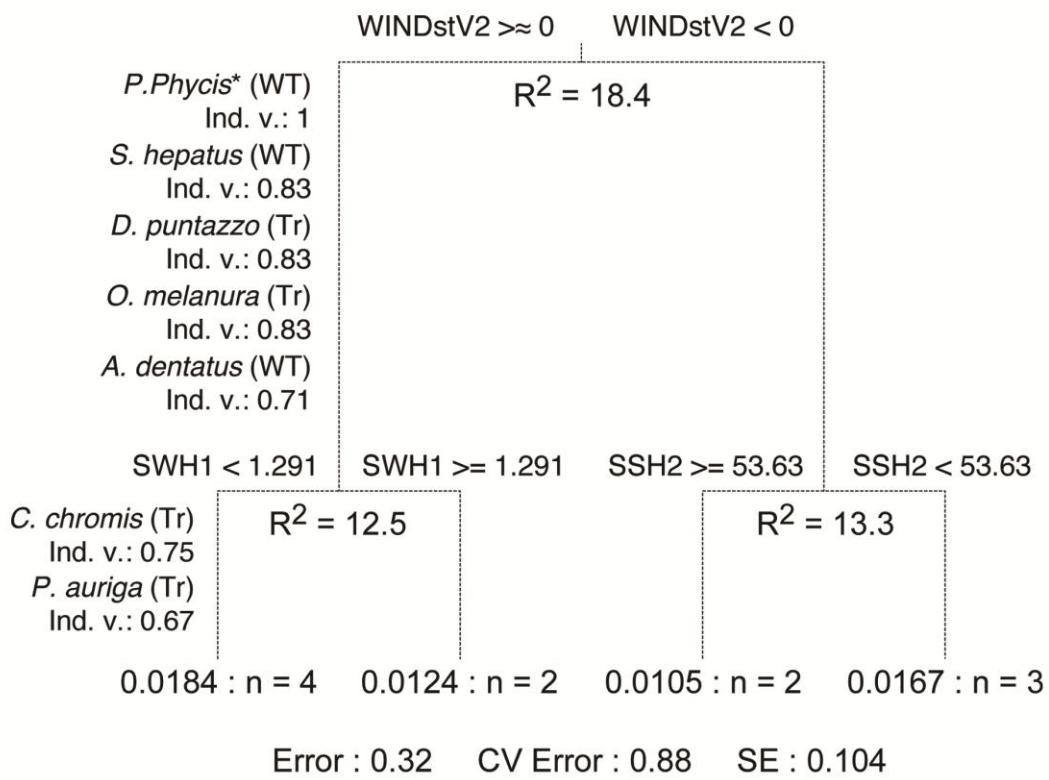

Figure 4A.6 - Best descriptive multivariate regression tree of a 12-year dataset on presence/absence of rocky reef fish from the Arrábida Marine Park. Most important predictor variable was mean winter northward component (V) of wind stress (WINDstV2). Minimum winter significant wave height (SWH1) and mean winter sea surface height (SSH2) were also selected predictors. The tree explained 44.3% of the total species variance. Indicator species for clusters are listed with affinities in parentheses and variance explained (* refers to node discriminant species).

Although individual species contribution to the variance explained by each split were relatively low (up to 2.4%), a large number of species contributed to differences between years. In the selected model, 58 species contributed to some (> 0) of the variance explained by the tree (Table 4A.S3). The two splits arising from the wind stress northward (V) component node showed that fish species similarities were higher within positive or negative mean winter values, since the threshold was very close to zero.

Indicator species

Five indicator species (3 warm-temperate (WT) and 2 tropical (Tr)) were obtained for the clusters created by this node, with all being located on the left side, which refers to positive northward wind stress (Figure 4A.2). *Phycis phycis* (WT) was the discriminant species of this node. In the final partition, two species with tropical affinities were indicators of the left leaf referring to the lower wave height and to positive northward winds.

Predictive models

The most important predictor, mean winter wind stress northward component (WINDstV2), was highly collinear with mean winter sea surface temperature (WINDstV2 and SST2: Pearson's r: 0.78, p-value < 0.001). Thus, we ran two models with each as the main driver.

The mean AUC revealed a perfect match (AUC > 0.8; Hosmer and Lemeshow, 2000) between observed and modelled fish community data for the model headed by the wind stress V (mean AUC = 0.92 ± 0.03) and for the alternative model headed by SST (mean AUC = 0.93 ± 0.03). The performance of both models was not statistically different (W = 59.5, p-value = 0.97). The performance of the predictions of a model using the PC1 (obtained by a PCA between WINDstV2 and SST2) as an explanatory variable had a much lower AUC value than each separate model (mean AUC = 0.81) and thus was not used in further analyses.

Since wind stress values for 1-degree resolution were not well correlated to wind stress values at a local scale (Pearson's r: 0.52, p-value = 0.098), this variable was not used for the long-term analysis. On the other hand, the two SST datasets were strongly correlated (Pearson's r: 0.94, p-value < 0.0001) revealing that regional SST reflects its local variability. Since local mean winter WINDstV and SST were also significantly collinear, the 1-degree SST2 was used as the best driver for the long-term community modeling. The validation of the long-term predictions using 1-degree SST showed a high AUC (0.9 ± 0.07) also indicating a strong power of prediction.

Predictors and community indices

Mean winter wind stress V and SST (9 km resolution) trends observed for the period 1993-2011 were strongly correlated (Figure 4A.3a, b), showing positive semblance values in most years (Figure 4A.3c). These two explanatory variables showed short amplitudes, especially for wind stress (wind stress V: [-0.05, 0.03] $N.m^{-2}$; SST: [14.21, 15.84] ºC).

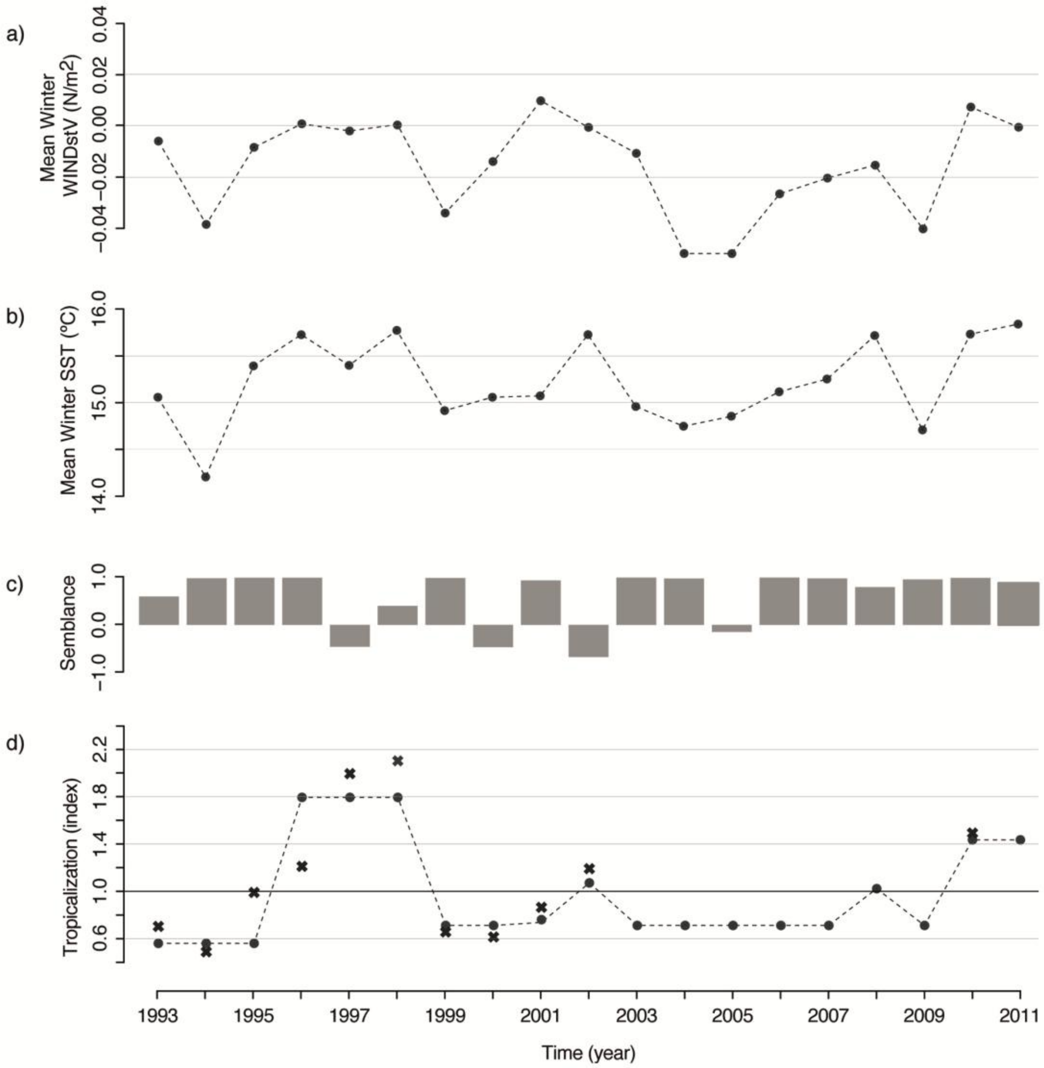

Figure 4A.7 - Trends of high resolution (9 km) mean winter a) northward component (V) of wind stress (WINDstV2) and b) SST (SST2) for the period 1993-2011; c) the correspondent semblance analysis (temporal wavelet relation) between these two predictors, and d) the tropicalization index (t. i.) for the observed (black crosses) and modelled communities (grey filled circles; connected by the dashed line) obtained from the averaged tropicalization results of the two predictive models (headed by WINDstV2 and SST2).

Mean winter North Atlantic Oscillation (NAO) showed a similar temporal pattern to high resolution wind stress V and SST datasets for period 1993-2011, although in opposite directions, but they were marginally not significantly correlated (winter WINDstV-NAO: r: -0.61; p-value = 0.05; winter SST-NAO: r: -0.60, p-value = 0.05; Figure 4A.S1a, b).

Despite slight differences due to the variability between mean winter wind stress and SST, the index of tropicalization calculated for both models together (ensemble procedure) showed similar trends to those found for separated models (Figure 4A.3, S2a, b). The tropicalization index obtained from predictive models was very similar to the index found from observed communities (Figure 4A.3d).

In the ensemble (averaging) procedure the highest value of the tropicalization index (t.i.; Figure 4A.3d) was found during 1996-1998 (t.i. = 1.8), then it decreased during the following two years and increased again in 2001 and even more in 2002 (t.i. = 0.8; 1.1), in which species with tropical affinities exceed cold-temperate ones (> 1); another annual increase was observed during 2008 (t.i. = 1), and the second largest peak was reached during 2010 and 2011 (t.i. = 1.4). The maximum value of the index was achieved during warmer years (Figure 4A.3b), corresponding also to years with high values of mean winter northward wind stress (Figure 4A.3a). During 2005, when the discrepancy of those two variables was larger and the semblance relation was weaker, the ensemble tropicalization index, which averages the effect of the wind and SST, resulted in intermediate values.

Long-term patterns of regional SST2 showed relative stability in inter-annual variability, oscillating between warm and colder periods (SST2 values between 13.89 and 16.07ºC), although the overall trend suggests an increase of mean winter SST in the last 50 years (Figure 4A.4a). The variability between warm and colder periods appeared to become larger and more abrupt since mid-eighties, with the increase of SST occurring during longer periods.

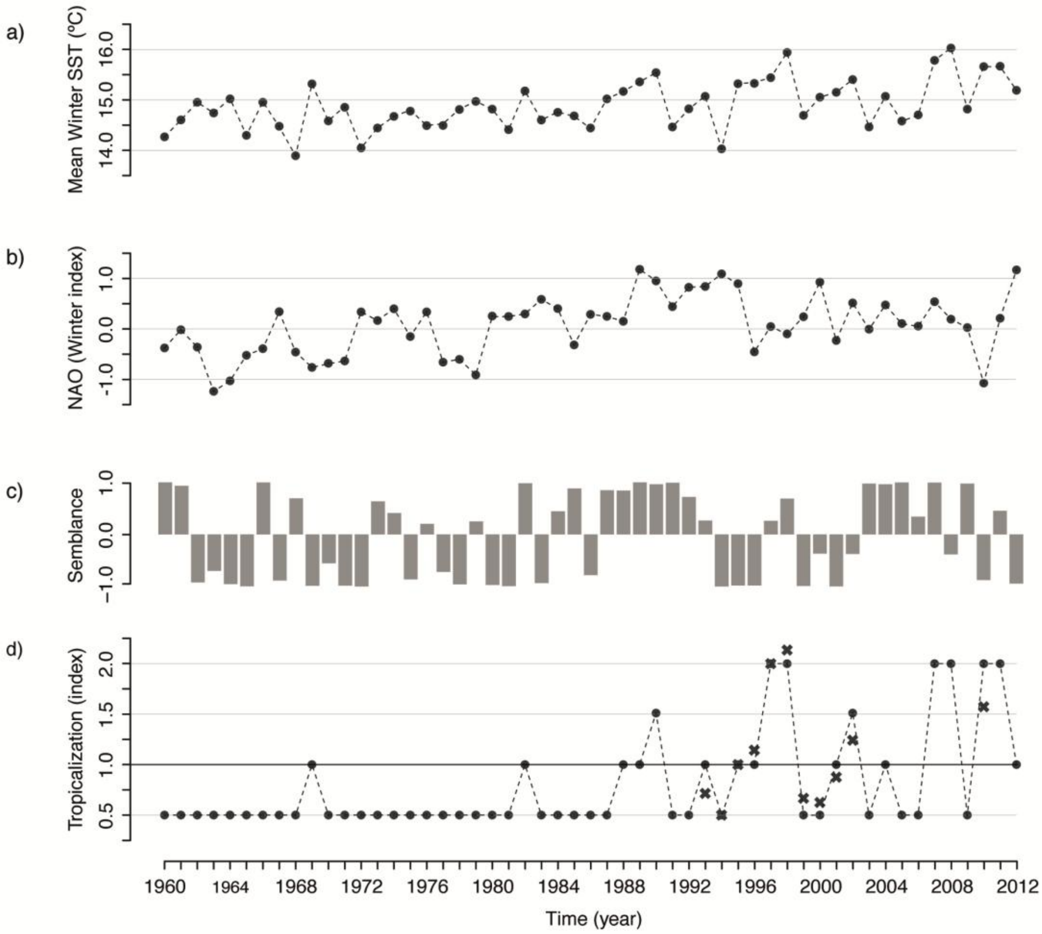

Figure 4A.8 - Trends of a) 1-degree resolution mean winter SST (SST2), b) large-scale North Atlantic Oscillation (NAO) for the period 1960-2012, c) the correspondent semblance analysis (temporal wavelet relation) between these two predictors, and d) the tropicalization index (t. i.) for the observed (black crosses) and modelled communities (grey filled circles; connected by the dashed line) obtained from the predictive model using long-term SST2 as explanatory variable.

Long-term patterns of NAO also showed inter-annual variability (values range between -1.32 and 1.18) with a period of strong positive NAO in the beginning of the nineties (Figure 4A.4b). Although the correlation between NAO and 1-degree SST was not significant for this long-term period (Pearson’s r: 0.065, p-value = 0.65; Figure 4A.4c), semblance analysis suggest high temporal correlation although in varying directions throughout time.

The long-term prediction of fish community structure revealed more frequent large values of the tropicalization index since the mid-1980s, especially since the mid-1990s when the largest index (t.i. = 2) was recorded during some years for the first time in fifty years (Figure 4A.4d).

4A. 5 Discussion

Our study suggests that at a the temperate transition zone on the Portuguese west shore a 'tropicalization' of community structure is occurring due to more frequent warming events brought about by the ocean warming trend over the last 50 years, the same period showing an accelerated warming worldwide (Rosenzweig *et al.*, 2008). These conclusions are supported by the strong influence of local oceanographic variables in the community shifts, which have been showing consistent patterns with warming.

In a previous study, Henriques *et al.* (2007) showed that: i) rocky reef fish assemblages changed among years with contrasting climatic features and ii) winter conditions are the most important drivers of variability in community structure. This was demonstrated using both large- and regional-scale climatic and ocean variables in Henriques *et al.* (2007) and now with local-scale variables (this study). The winter North Atlantic Oscillation (NAO) showed considerable influence on inter-annual variability (Henriques *et al.*, 2007) being also the main large-scale predictor of changes in fish communities. However, we found that high-resolution (local-scale) oceanographic variables such as the northward component of wind stress, sea surface temperature (SST), sea surface height and significant wave height better explain community patterns. Along-coast winds, which drive coastal upwelling events, are influenced both by local processes and large-scale changes, in particular the location and intensity of subtropical anticyclones which affect NAO patterns (Miranda *et al.*, 2013). Despite this, the relationship between winter NAO and high-resolution northward component of wind stress and SST was weak, suggesting that local patterns of change in oceanographic variables have a strong influence in community variations.

Fish community composition differed between years in association with wind stress. Multivariate regression trees showed a considerable portion of species variability (44.3%), with indicator species of the clusters having warm-temperate or tropical affinities and being associated to years with warmer conditions (left side of the tree). On the other hand, no indicator species associated with winters with strong southward wind stress and thus colder temperatures.

The rocky fish assemblage in this temperate transition zone is currently composed mainly by warm-temperate species. This assemblage, however, might have been more influenced by cold-temperate winters in the past decades, in which warm-temperate species may have had their northward dispersion limited for several years. This is consistent with the fact that cold-temperate species do not show an association with the largest changes in fish assemblage composition. These results point to the importance of local- and large-scale oceanographic variables in the dispersion and dynamics of marine assemblages (Sánchez *et al.*, 2007; Selig *et al.*, 2010).

How does the taxonomic makeup of fish communities change so rapidly? Hawkins *et al.* (2009) found that warmer years were characterized by an increase of warmer species albeit the persistence of cold-temperate ones, possibly due to their higher competitive ability and occasionally massive recruitment during spring blooms. However, after several warm years with consecutively poor recruitment events, cold-temperate species are likely to retreat rapidly (Svensson *et al.*, 2006). Santos *et al.* (2001, 2005, 2007) found for the Portuguese coast that winters with strong southward winds and weak but consistent winter upwelling events lead to poor recruitment of sardines in the following months due to a large offshore transport of eggs and larvae during their spawning season (winter). Additionally, Henriques *et al.* (2007) suggested that recruitment strength and survival were key to rapid changes between warm and cold year's fish assemblages, both studies corroborating the importance of recruitment success in the local fish assemblage structure of the following months.

A complementary hypothesis is that recruits and young individuals, which are usually the main drivers for changing distribution limits of marine species (Figueira and Booth, 2010), may shift more rapidly into new, warmer environments than older and settled adults would retreat due to suboptimal thermal conditions. This idea is supported by the much larger dispersion ranges of early stages than of adults in demersal fish (Gruss *et al.*, 2011). Even if reproduction is inhibited by temperatures below threshold levels, existing adults will likely persist until they die (naturally or by other causes) as long as conditions are adequate for survival (Pörtner and Peck, 2010). In fact, this is what is expected by a continuous but relatively slow warming. Local extinctions may occur due to unsuitable climatic conditions (Malcolm *et al.*, 2002) but will possibly take longer and occur less frequently than the appearance of new immigrants.

Similar to most major upwelling regions of the world, strong southward winds in summer favour offshore Eckman transport in our study area, leading to upwelling events and cold coastal waters with increased productivity (Cravo *et al.*, 2010). Cold-temperate species spawning in the spring probably experience suitable conditions during recruitment, due to summer and autumn upwelling events and juveniles and young adults could move to deeper areas in a response to thermal stress in warm years. The ability to alter depth ranges as a response to changing temperatures has been shown in some studies (Lafrance *et al.*, 2005; Perry *et al.*, 2005; Caputi *et al.*, 2010), and may be an explanation for how cold-temperate species persist during warm winters. Depth shifts may also be more common as a response to persistent climate change in marine ecosystems than previously thought. Similarly, terrestrial plants are also predicted to respond to warming by moving polewards and upwards in mountains (Jump *et al.*, 2009; Randin *et al.*, 2009).

SST patterns and the related modelled community for the last 50 years suggest that since the mid-1980s warm winters increased in frequency, leading to an increased proportion of tropical species in the fish assemblage. Tittensor *et al.* (2010) found that SST was the only environmental predictor related to marine diversity across several taxa, although habitat availability and historical factors also influenced coastal species. Previous studies for North Europe described a cold event during 1962-1963 followed by cooler conditions until the mid-1980s after which warming conditions prevailed (Crisp, 1964; Hawkins *et al.*, 2009). This cold period probably affected southern Europe climate conditions and may have influenced the expansion and persistence of the southern distribution limits of cold-temperate species, contributing to the low tropicalization index found for that period. Other studies for the north of Europe described stable biogeographic range limits for several species until the mid-twentieth century (Southward and Crisp, 1956). Studies conducted in our study area in the past showed very abundant canopy-forming brown algae (*Laminaria ochroleuca*, *Sacchoriza polishides, Fucus vesiculosus*) which may have provided complex structuring habitats to fish communities (Palminha, 1958; Saldanha, 1974; Santos, 1993). The loss of kelp and fucoid beds and their important habitat function in recent years is probably related to an increase in temperature (Nicastro *et al.*, 2013); Assis *et al.*, unpublished data), possibly affecting their recruitment and resilience (Wernberg *et al.*, 2010). Both warming conditions and a reduction in habitat complexity and structural function may have acted synergistically to contribute to changes in fish assemblage's structure found from the mid-1980s onwards.

Regional wind stress was not a good proxy for local patterns in our study, preventing reliable projections using 1-degree data. However, high-resolution northward component of wind stress was strongly related to local SST, which is correlated to long-term regional SST, suggesting that sea water temperature is a valid proxy for both predictors. Therefore, with current and future global and regional warming scenarios for the oceans (Lemos and Pires, 2004; Somot *et al.*, 2006; Bakun *et al.*, 2010; Miranda *et al.*, 2013), it is probable that more frequent strong northward winds and currents associated with storm events will occur, intensifying their role on the tropicalization of this region and contributing to considerably altered fish assemblages in the near future (Hawkins *et al.*, 2009; Heath *et al.*, 2012). Storms arriving from the south may promote nearshore retention of eggs and larvae in opposition to upwelling events (Henriques *et al.*, 2007), and facilitate the transport of tropical species into the area. An increase in warming winters will allow tropical fish recruits to overwinter and persist in their marginal distributions, gradually extending their distributions polewards (Perry *et al.*, 2005; Cheung *et al.*, 2009; Hawkins *et al.*, 2009; Figueira and Booth, 2010; Cheung *et al.*, 2012; Heath *et al.*, 2012). More frequent extreme climatic events nested within longer-term climatic trends can accelerate shifts in species ranges, and favour the establishment of warmer species due to the persistence of suitable conditions (Jentsch *et al.*, 2007; Cheung *et al.*, 2009; Garrabou *et al.*, 2009; Wernberg *et al.*, 2013).

Temperate transition areas are often considered 'hotspots' of biodiversity since they are typically characterized by complex and diverse habitats and contain species adapted to heterogeneous inter- and intra-annual oceanographic conditions. In these areas, northern and southern distribution ranges of warm and cold species change with the temporal cyclic fluctuations (Henriques *et al.*, 1999). Thus, the most likely pattern for a future persistent warming scenario is with the advance of tropical species polewards and the simultaneous, but slower, gradual retreating of cold water species (Hawkins *et al.*, 2009). Interestingly, most of the species with tropical affinities found in this study have historical local commercial value (e.g. Baldaque da Silva, 1891), evidence of their past presence in the area. Despite the rare occurrences of adults of tropical species far from their distribution limits (Horta e Costa and

Gonçalves, 2013), the tropicalization of this temperate fish community is occurring with species that have historically existed in the region and not with invasive or vagrant species. This illustrates the important ‘barometer’ role of this area to study the effects of climate change (Horta e Costa and Gonçalves, 2013). In fact, a rapid response of fish communities to climate change is more probable in areas of continuous and/or contiguous habitats which facilitate high dispersal rates (Hiddink and ter Hofstede, 2008). Our findings also highlight the potentially higher resilience of temperate transition areas to climatic shifts when compared to typical tropical or cold regions where high rates of extinctions and invasions are very likely (Cheung *et al.*, 2009; Sumaila *et al.*, 2011).

Despite contradictory projections about the intensification or reduction of upwelling with global warming (Bakun *et al.*, 2010), Miranda *et al.* (2013) predicted a future increase in upwelling events, especially in the northern limit of the Iberian west coast (near Cape Finisterre), where the mean effect may extend hundreds of kilometers. This could mitigate the effect of local warming (Miranda *et al.*, 2013) and facilitate the persistence of cold-temperate species. If this is true and upwelling events increase in a warmer ocean, temperate transition zones affected could have high levels of resilience to climate change. However, although species from these areas show some plasticity to deal with variable oceanographic conditions, a disruption on the historical cyclic fluctuations could gradually change communities until very different assemblages and trophic interactions remain (Perry *et al.*, 2005; Cheung *et al.*, 2012; Heath *et al.*, 2012). Furthermore, synergistic and additive effects on marine communities are likely to occur between climate change and other human-induced activities such as overfishing (Griffith *et al.*, 2011). Such impacts may contribute to a decrease in ecosystem resistance and accelerate the disturbance of community structure and species interactions (Ling *et al.*, 2009). Intensively fished populations were found to be the most susceptible to ocean acidification, revealing that stressed populations show higher vulnerability to climate change (Griffith *et al.*, 2011). Therefore, networks of marine protected areas are suggested to increase the resilience of ecosystems in relation to future warming scenarios, while reducing the impact of fishing and other human uses (Ling *et al.*, 2009; McLeod *et al.*, 2009), possibly mitigating non-linear and unpredictable responses of species and ecosystems in this changing world (Munday *et al.*, 2008).

The coastal communities and artisanal fishers, such as the ones of the studied area, are usually strongly dependent on local resources (Batista *et al.*, 2009; Horta e Costa *et al.*, 2013) and thus the tropicalization of the rocky reef fish communities could lead to large biological and socio-economic impacts in the near future (Sumaila *et al.*, 2011; Cheung *et al.*, 2012; Cheung *et al.*, 2013). To reduce uncertainties in future projections it is crucial to have an improved understanding of past responses (Pandolfi *et al.*, 2011). If the tropicalization of transition areas is becoming more frequent worldwide (Hawkins *et al.*, 2009; Cheung *et al.*, 2012; Cheung *et al.*, 2013; Wernberg *et al.*, 2013) and if the rate of human-induced impacts does not decrease, community changes could be very large in a coming future, disrupting ecosystems and leading to tropical species dominating previous temperate zones.

4A. 6 Acknowledgements

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4A.8 Supplementary Information

4A.8.1 Tables 4A.S

Table 4A.S4 - Rocky reef fish species and corresponding Families observed in the underwater surveys and included in the multivariate regression models. The biogeographic group affinity is indicated: CT – Cold-temperate; T – Temperate; WT – Warm-temperate; Tr – Tropical; E – Eurythermic.

Species	Family	Biogeographic group
Ciliata mustela	Phycidae	CT
Gaidropsarus vulgaris	Phycidae	CT
Pollachius pollachius	Gadidae	CT
Entelurus aequoreus	Syngnathidae	CT
Nerophis lumbriciformis	Syngnathidae	CT
Taurulus bubalis	Cottidae	CT
Centrolabrus exoletus	Labridae	CT
Labrus bergylta	Labridae	CT
Symphodus melops	Labridae	CT
Gobiusculus flavescens	Gobiidae	CT
Zeugopterus punctatus	Scophthalmidae	CT
Zeus faber	Zeidae	E
Trigloporus lastoviza	Triglidae	E
Boops boops	Sparidae	E
Balistes capriscus	Balistidae	E
Conger conger	Congridae	T
Sardina pilchardus	Clupeidae	T
Gaidropsarus mediterraneus	Phycidae	T
Trisopterus luscus	Gadidae	T
Mugilidae n.id.[a]	Mugilidae	T
Atherina sp.[a]	Atherinidae	T
Belone belone	Belonidae	T
Syngnathus acus	Syngnathidae	T
Syngnathus typhle	Syngnathidae	T
Dicentrarchus labrax	Moronidae	T
Trachurus trachurus	Carangidae	T
Pagellus acarne	Sparidae	T
Spondyliosoma cantharus	Sparidae	T
Mullus surmuletus	Mullidae	T
Coris julis	Labridae	T
Ctenolabrus rupestris	Labridae	T
Labrus mixtus	Labridae	T
Symphodus bailloni	Labridae	T
Lipophrys pholis	Blenniidae	T
Diplecogaster bimaculata	Gobiesocidae	T
Callionymus sp.[a]	Callionymidae	T
Gobius niger	Gobiidae	T
Pomatoschistus sp.[c]	Gobiidae	T

Thorogobius ephippiatus	Gobiidae	T
Scomber colias	Scombridae	T
Myliobatis aquila	Myliobatidae	Tr
Halobatrachus didactylus	Batrachoididae	Tr
Serranus cabrilla	Serranidae	Tr
Epinephelus marginatus	Serranidae	Tr
Diplodus puntazzo	Sparidae	Tr
Diplodus sargus	Sparidae	Tr
Oblada melanura	Sparidae	Tr
Pagrus auriga	Sparidae	Tr
Sarpa salpa	Sparidae	Tr
Chromis chromis	Pomacentridae	Tr
Parablennius pilicornis	Blenniidae	Tr
Sphoeroides marmoratus	Tetraodontidae	Tr
Raja undulata	Rajidae	WT
Muraena helena	Muraenidae	WT
Phycis phycis	Phycidae	WT
Hippocampus hippocampus	Syngnathidae	WT
Hippocampus guttulatus	Syngnathidae	WT
Scorpaena notata	Scorpaenidae	WT
Scorpaena porcus	Scorpaenidae	WT
Aspitrigla obscura	Triglidae	WT
Serranus atricauda	Serranidae	WT
Serranus hepatus	Serranidae	WT
Trachurus sp.[a]	Carangidae	WT
Diplodus annularis	Sparidae	WT
Diplodus bellottii	Sparidae	WT
Diplodus cervinus	Sparidae	WT
Diplodus vulgaris	Sparidae	WT
Pagellus erythrinus	Sparidae	WT
Pagrus pagrus	Sparidae	WT
Sparus aurata	Sparidae	WT
Spicara maena	Centracanthidae	WT
Symphodus cinereus	Labridae	WT
Symphodus mediterraneus	Labridae	WT
Symphodus ocellatus	Labridae	WT
Symphodus roissali	Labridae	WT
Symphodus rostratus	Labridae	WT
Tripterygion delaisi	Tripterygiidae	WT
Coryphoblennius galerita	Blenniidae	WT

Lipophrys canevae	Blenniidae	WT
Paralipophrys trigloides	Blenniidae	WT
Parablennius gattorugine	Blenniidae	WT
Parablennius incognitus	Blenniidae	WT
Parablennius rouxi	Blenniidae	WT
Parablennius sanguinolentus	Blenniidae	WT
Clinitrachus argentatus	Clinidae	WT
Apletodon dentatus	Gobiesocidae	WT
Lepadogaster candollii	Gobiesocidae	WT
Lepadogaster lepadogaster [b]	Gobiesocidae	WT
Gobius xanthocephalus	Gobiidae	WT
Gobius cobitis	Gobiidae	WT
Gobius cruentatus	Gobiidae	WT
Gobius paganellus	Gobiidae	WT
Gobius gasteveni	Gobiidae	WT
Zeugopterus regius	Scophthalmidae	WT
Pomatomus saltator	Pomatomidae	WT

[a] *Trachurus picturatus* and *T. mediterraneus* was considered *Trachurus* sp. since they were not always distinguished and have the same biogeographic group. Similarly, *Atherina presbyter* and *A. boyeri* were considered *Atherina* sp., species from the family Mugilidae were aggregated to the family level (since the three most abundant species are from the same biogeographic group) and finally, *Callionymus lyra* and *C. reticulatus* were identified as Callionymus sp.

[b] As in Henriques *et al*. (2007), *Lepadogaster purpurea* was considered as *L. lepadogaster* since they were considered the same species until 2002.

[c] Pomatoschistus sp. is most probably *Pomatoschistus pictus* but it was not possible to distinguish the species at all the times.

Table 4A.S5 - Environmental predictors used for modelling purposes. Predictors name (SST - Sea Surface Temperature; WINDstV - Daily Wind Stress - northward component (V); WINDstU - Daily Wind Stress - eastward component (U); SSH - Sea Surface Height; SWH - Significant Wave Height; Chla - Chlorophyll a; NAO - North Atlantic Oscillation), data source, temporal range, original resolution, predictor type (RS: Remote sensing; DO: Direct observation), units and derived metric (Winter - DJFMA ; Summer - JJAS). Seasonal NAO dummy variables were also tested: 1 when NAO was positive and 0 when it was negative.

Predictors	Source	Temporal range	Resolution	Type	Units
SST	OSTIA; Donlon et al. 2011	1992 - 2011	0.05º	RS	ºC
SST	ICOADS 1 Degree; Woodruff et al. 2011	1960-2010	1º	RS, DO	ºC
WINDstV	Blended Sea Surface Winds; NOAA; Zhang et al. 2006	1992 - 2011	0.25º	RS	$N.m^{-2}$
WINDstU	Blended Sea Surface Winds; NOAA; Zhang *et al*. 2006	1992 - 2011	0.25º	RS	$N.m^{-2}$
WINDstV	ICOADS 1 Degree; Woodruff et al. 2011	1960 - 2010	1º	RS, DO	$N.m^{-2}$
SSH	AVISO; Schaeffer et al. 2012	1992 - 2011	0.33º	RS	m
SWH	AVISO; Schaeffer et al. 2012	1992 - 2011	0.33º	RS	m
Chla	MODIS AQUA; Huot et al. 2005	1997 - 2011	0.05º	RS	$mg.m^{-3}$
NAO	NOAA:http://www.cpc.ncep.noaa.gov/products/precip/CWlink/pna/nao.shtml	1960 - 2010	Large scale	RS, DO	Index

Derived metrics for predictors SST, wind stress (V and U), SSH, SWH and Chla:
1 - Minimum Winter; 2 - Mean Winter; 3 - Maximum Winter; 4 - Minimum Summer; 5 - Mean Summer; 6 - Maximum Summer; 7 -Winter deviation from the monthly mean; 8 - Summer deviation from the monthly mean; 9 - Number of Winter days above long term monthly mean; 10 - Number of Summer days above long term monthly mean; 11 - Number of Winter days bellow long term monthly mean; 12 - Number of Summer days bellow long term monthly mean.
Derived metrics for predictor NAO: 13 - Mean Winter; 14 - Mean Summer; 15 - Dummy variable mean Winter (+/-); 16 - Dummy variable mean Summer (+/-)

Table 4A.S6 - Species variance for the best multivariate regression tree of the presence/absence data of rocky reef fish assemblage of the Arrábida Marine Park. The total species variance is partitioned by species, the whole tree, and the three splits of the tree. Selected predictors of the three principal splits obtained by the best model selected were the mean winter northward component (V) of wind stress (WINDstV2) and sea surface height (SSH2) and minimum winter significant wave height (SWH1).

Species	WINDstV2 < -0.004	SSH2 < 53.63	SWH < 1.29	Tree total	Species total
Ciliata mustela	0.13	0.07	1.04	1.25	1.90
Gaidropsarus vulgaris	0.07	0.07	0.00	0.14	0.79
Pollachius pollachius	0.59	0.00	0.00	0.59	1.90
Entelurus aequoreus	0.45	0.29	0.46	1.20	2.22
Nerophis lumbriciformis	0.17	1.16	0.46	1.79	2.37
Taurulus bubalis	0.38	0.00	0.46	0.84	1.42
Centrolabrus exoletus	0.00	0.00	0.00	0.00	0.00
Labrus bergylta	0.00	0.00	0.00	0.00	0.00
Symphodus melops	0.00	0.00	0.00	0.00	0.00
Gobiusculus flavescens	0.00	0.00	0.00	0.00	0.00
Zeugopterus punctatus	0.52	0.07	0.26	0.85	2.37
Zeus faber	0.00	0.29	0.26	0.55	1.42
Trigloporus lastoviza	0.02	0.00	1.04	1.07	2.37
Boops boops	0.00	0.00	0.00	0.00	0.00
Balistes capriscus	0.04	0.07	0.12	0.23	1.90
Conger conger	0.01	0.29	0.03	0.33	2.22
Sardina pilchardus	0.21	0.65	0.12	0.98	2.22
Gaidropsarus mediterraneus	0.13	0.07	0.46	0.67	1.90
Trisopterus luscus	0.07	0.29	0.00	0.36	0.79
Mugilidae n.id.	0.00	0.00	0.00	0.00	0.00
Atherina sp.	0.00	0.00	0.00	0.00	0.00
Belone belone	0.45	0.29	0.03	0.76	2.22
Syngnathus acus	0.00	0.00	0.00	0.00	0.00
Syngnathus typhle	0.85	0.00	0.03	0.88	1.90
Dicentrarchus labrax	0.04	0.29	0.12	0.45	1.90
Trachurus trachurus	1.05	0.07	0.00	1.13	2.22
Pagellus acarne	0.01	1.16	0.03	1.20	2.22
Spondyliosoma cantharus	0.00	0.00	0.00	0.00	0.00
Mullus surmuletus	0.00	0.00	0.00	0.00	0.00
Coris julis	0.00	0.00	0.00	0.00	0.00
Ctenolabrus rupestris	0.00	0.00	0.00	0.00	0.00
Labrus mixtus	0.00	0.00	0.00	0.00	0.00
Symphodus bailloni	0.00	0.00	0.00	0.00	0.00
Lipophrys pholis	0.00	0.00	0.00	0.00	0.00
Diplecogaster bimaculata	0.04	0.07	0.12	0.23	1.90
Callionymus sp.	0.00	0.00	0.00	0.00	0.00
Gobius niger	0.00	0.00	0.00	0.00	0.00
Pomatoschistus sp.	0.00	0.00	0.00	0.00	0.00

4A.8.2 Figures 4A.S

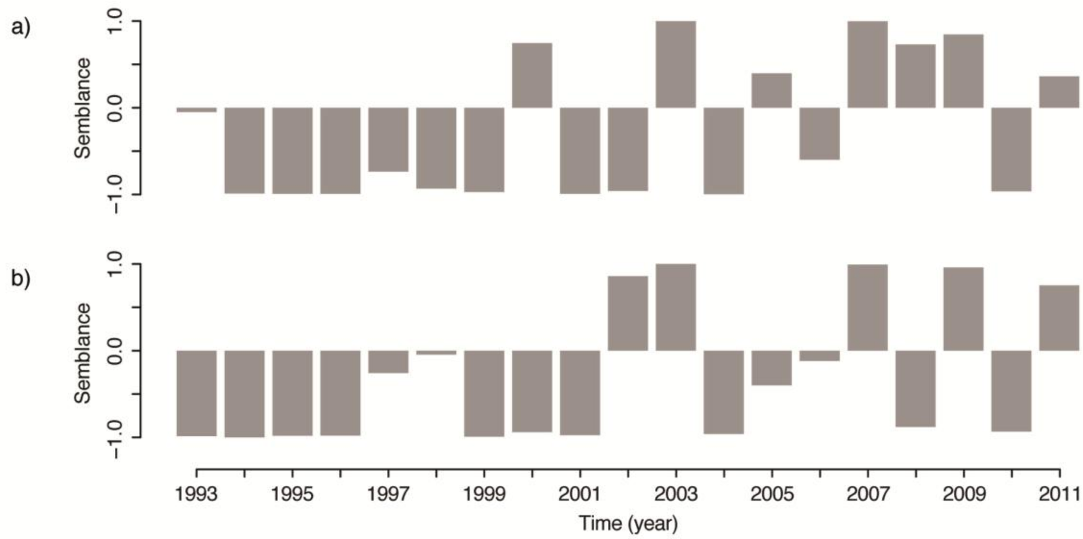

Figure 4A.S3 - Semblance analysis (temporal wavelet relation) between high-resolution (9 km) a) mean winter northward component (V) of wind stress (WINDstV2) or b) mean winter sea surface temperature (SST2) and North Atlantic Oscillation (NAO) for the period 1993-2011.

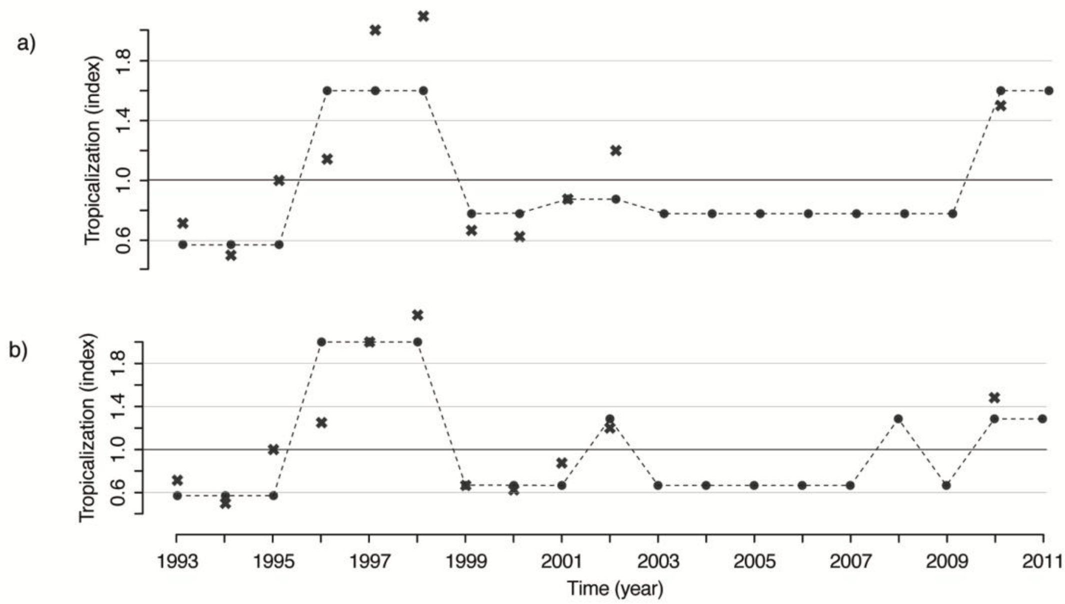

Figure 4A.S4 - Tropicalization index (t. i.) for the observed (black crosses) and modelled communities (grey filled circles; connected by the dashed line) obtained from the a) main predictive model headed by the driver mean winter northward component (V) of wind stress (WINDstV2) and b) alternative model headed by the driver mean winter sea surface temperature (SST2).

CHAPTER 4B: FIRST OCCURRENCE OF THE MONROVIA DOCTORFISH *ACANTHURUS MONROVIAE* [PERCIFORMES: ACANTHURIDAE] IN EUROPEAN ATLANTIC WATERS

Published in *Marine Biodiversity Records*

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4B. 1 Abstract

Acanthurus monroviae is reported for the first time in western European waters. One adult specimen of this species was observed and photographed in the Arrábida Marine Park (Portugal, 38.43º N 9.07º W) on December 2007.

Short term temperature changes associated with warmer winters may favour the occurrence of vagrants' individuals of tropical species in this temperate biogeographic transition zone. This area is considered an important 'barometer' for studying the effects of climatic warming with possibly permanent expansions of the geographical ranges of these species.

Keywords: Tropical surgeonfish, range expansion, temperature, NAO, temperate biogeographic transition zone, climatic events

4B.2 Introduction

The tropical eastern Atlantic surgeonfish *Acanthurus monroviae* (Steindachner, 1876) is native along the coast of Africa (Morocco to Angola) and the São Tomé, Cape Verde and Canary Archipelagos. However, some vagrants of this species have been occasionally seen outside their normal geographical range in the Mediterranean (Crespo *et al.*, 1987; Golani and Sonin, 1996; Hemida *et al.*, 2004; Ben Souiss *et al.*, 2011) with a few specimens detected as far out as the Brazilian coast (Joyeux *et al.*, 2001; Luiz *et al.*, 2004)Joyeux *et al.*, 2001; Luiz-Júnior *et al.*, 2004) and St Paul's Rocks in the equatorial Atlantic (Ferreira *et al.*, 2009).

Here, we describe the occurrence of a specimen of *A. monroviae* at the Portuguese western coast in 2007 and discuss the significance of this observation in relation to the life-history characteristics of this species and the biogeography of this part of the western European shores.

4B.3 Methods

One adult specimen of *Acanthurus monroviae* was observed and photographed by divers at the Arrábida Marine Park at a site called "Pedra do Leão" (38.43º N; 9.07º W), on 30 December 2007 at 1100 hours and at 12 m depth. Water temperature was around 15º C and the estimated length was 25–30 cm.

Sea surface temperature (SST) was obtained for the period 1985-2009 (5-day average; 4 km grid; Pathfinder AVHRR SST; source: NOAA http://www.nodc.noaa.gov/sog/pathfinder4km/userguide.html - last accessed 18.02.10) and the long term average was calculated to produce a color map of the Iberian Peninsula. Optimum interpolation SST for the region (37.9º–39º N; -10.1º –-8.5º W) for the last 20 years was used as a proxy of the sea temperature in the marine park (source: NOAA http://www.emc.ncep.noaa.gov/research/cmb/sst_analysis/ - last accessed 31.10.12) as it was highly correlated with our *in situ* temperature data loggers located at 5 m depth during 2008 and 2009 (38.45º N; 9.02º W). The North Atlantic Oscillation (NAO) Index (source: NOAA http://www.cpc.ncep.noaa.gov/products/precip/CWlink/pna/nao.shtml - last accessed 31.10.12) was also obtained for the same period and both data were analysed separately by season: winter (December to April) and summer (June to September).

4B.4 Results

The specimen of *Acanthurus monroviae* (Monrovia doctorfish) was registered and photographed at the Arrábida Marine Park on 30 December 2007 at 12 m depth (Figure 4B.1A–C). The fish was feeding on the rocky reefs together with a mixed school of common two-banded seabreams (*Diplodus vulgaris*) and white seabreams (*D. sargus*).

This marine park is located at an important biogeographic transitional zone between warm and cold temperate waters, as observed by the long term average of sea surface temperature (SST) (Figure 4B.2). Average optimum interpolation SST analysis showed that the year before the sighting (2006) was the warmest summer of the last 20 years, with a steady SST increase being observed since 2002 (Figure 4B.3). Winter temperature also showed an increase from 2005 to 2008 (Figure 4B.3). The annual NAO average was similar among seasons between 2001 and 2005 but after that, higher values were found in winter than in summer NAO (Figure 4B.4). During 2006, the average summer NAO was negative and the winter NAO was close to zero. During 2007 a small peak was found in both NAO averages (Figure 4B.4). *In situ* data loggers from 2008 and 2009, located at 5 m depth, revealed an annual temperature range between 11.5 ºC and 21.5 ºC within this marine park with December temperatures varying between 12.9 ºC and 16.9 ºC.

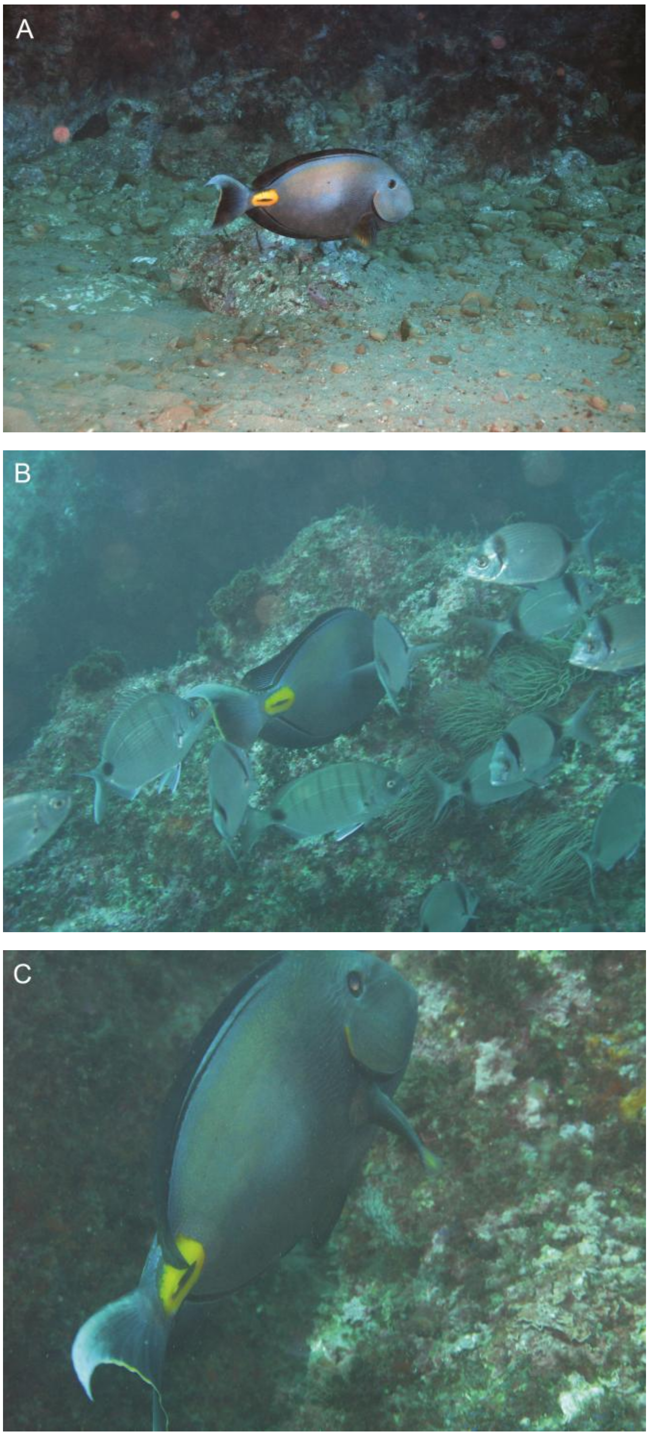

Figure 4B.5 - *Acanthurus monroviae* (Monrovia doctorfish) observed on 30 December 2007 at the Arrábida Marine Park, Portugal (12 m depth). Photography credits: (A) – Carlos Monteiro and (B, C) – Cláudio Dias.

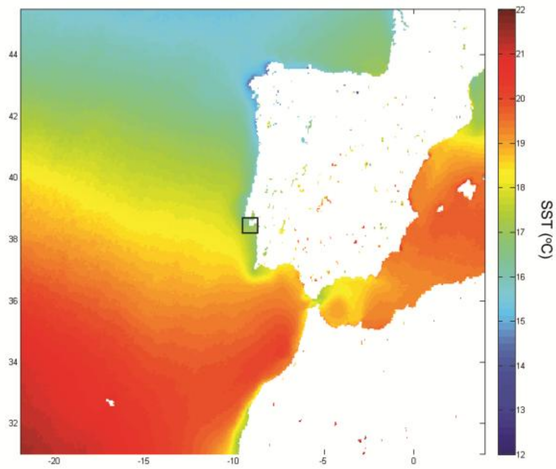

Figure 4B.6 - Long-term average SST (ºC) in the Iberian Peninsula. Mean 5-day composite Pathfinder AVHRR SST, 1985-2009 (4 km). The location of the Arrábida Marine Park is shown.

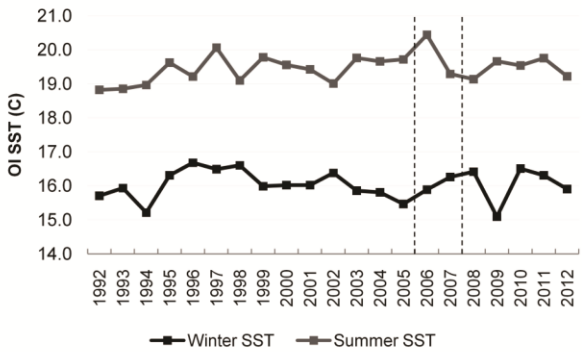

Figure 4B.7 - Average seasonal OI (optimum interpolation) SST by year (1992-2012) from the study region (source: NOAA) with the years 2006/2007 highlighted.

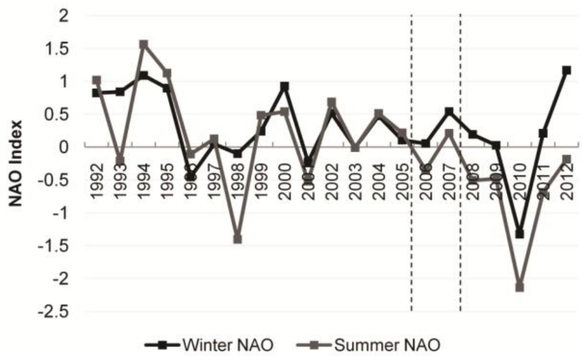

Figure 4B.8 - Average seasonal NAO Index by year (1992-2012), with the years 2006/2007 highlighted (source: NOAA).

4B.5 Discussion

Fish assemblages at the Arrábida Marine Park show a high variability among years, especially in species of tropical, warm-temperate and cold-temperate affinities. These changes in the coastal fish communities have been related to the intensity and duration of the winter NAO index (Henriques *et al.*, 2007). This is a biogeographic transition zone between warm and cold temperate waters and is also near the northern limit of the main north-east Atlantic upwelling events (Wooster *et al.*, 1976). Summer SST in 2006 was the highest of the last 20 years and the NAO index showed both winter and summer warm conditions. The long-term average SST map also shows that the marine park is in a transition region between cold and warm waters, suggesting this is probably the northernmost location where most species of tropical affinity could eventually reach following a short-term increase in water temperature.

The first occurrence of *Acanthurus monroviae* (Monrovia doctorfish) in the Mediterranean was in Málaga (off the Spanish south coast) on March 1981 (Crespo *et al.*, 1987). Further records of vagrant individuals were documented in Israel in 1995 (Golani and Sonin, 1996) and Algeria in 2001 (Hemida *et al.*, 2004), being the present record the first one for the Atlantic coast of Europe. One puzzling question remains unanswered which is: what are the mechanisms by which these vagrant individuals reach such far locations as adults? In fact, the above mentioned examples, as well as the present case, all report sightings of adult fish and, at least at the Arrábida Marine Park, intensive sampling of nearshore communities was being performed. It is therefore highly unlikely that the reported individuals reached these areas as a larvae or juvenile.

The mechanisms by which this species colonises areas outside its natural home range remain unknown. Other species of surgeonfish are known to have extended periods of pelagic larval duration as, for example, 45-70 days in *Acanthurus bahianus*, *A. chirurgus* and *A. coeruleus* (Rocha *et al.*, 2002) with the species *A. triostegus* being able to delay metamorphosis and extend their PLD to 44-60 (McCormick, 1999). This could offer a plausible explanation to how vagrant individuals of this species are found so far away from their natural distribution areas. However, since vagrant individuals in all the above mentioned cases are adults, other mechanisms must be at play.

The effects of stochastic climatic events on ocean currents (Joyeux *et al.*, 2001) and some degree of habitat connectivity and/or viable dispersal mechanisms such as transport by currents in association with drifting materials (natural or artificial), may favour the direct dispersal of adults to areas outside their natural home range. Recently, Luiz *et al.* (2012) showed that rafting is a key mechanism for species crossing the mid-Atlantic barrier in the tropics, providing strong evidence that reef fish can overcome large oceanic distances by associating with drifting material. Additionally, winters with strong south winds and waves bring warm temperatures nearshore to the western Portuguese shore and are known to be associated with an increase in the occurrence of species with warm temperate and tropical affinities in this region (Henriques *et al.*, 2007). One month after the detection of this surgeonfish (31 January 2008) fishers from Sesimbra village (a small fishing port inside the Arrábida Marine Park), caught one specimen of another tropical species, *Fistularia petimba* (red cornetfish) (local fishers, personal communication), adding these occurrences to a list of species with warm water affinity detected in this region. This makes the western Portuguese coast, in particular the region around the Arrábida Marine Park an important 'barometer' for studying the effects of more frequent environmental changes and a potential warmer ocean, with possibly permanent expansions of the geographical ranges of these species.

4B. 7 Ackowledgements

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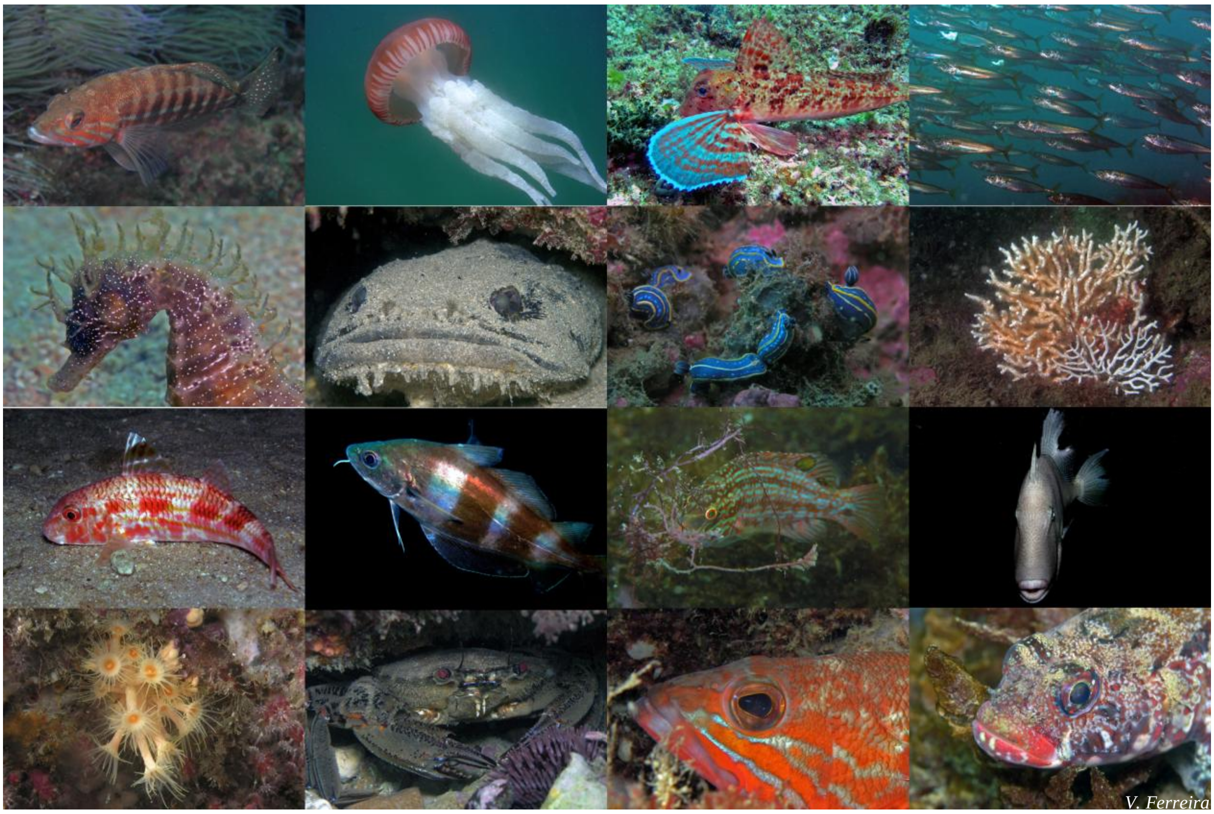
CHAPTER 5: GENERAL DISCUSSION



5.1 Importance of the Arrábida Marine Park

The present study reinforces the ecological importance of the Arrábida Marine Park. Previous studies showed the high diversity of habitats, biotopes and species, with 1320 species recorded so far (Henriques *et al.*, 1999; Almada *et al.*, 2000; Arrábida Marine Park Authority/ICNF 2012). When compared to other published studies of areas with similar latitudes in the north-east Atlantic and Mediterranean, the biodiversity level of this region is considerably higher (Henriques *et al.*, 1999). In fact, this temperate system supports a highly diverse set of habitats and assemblages with several vulnerable and key species such as rays, sharks, sea horses, gorgonians, sponges, kelps as well as a number of fish and invertebrate species targeted by local commercial fisheries (hereafter also called ‘target’ species) (Palminha, 1958; Saldanha, 1974; Santos, 1993; Henriques *et al.*, 1999; Almada *et al.*, 2000; Gonçalves *et al.*, 2003; Rodrigues, 2008; Cunha *et al.*, 2011).

Strong seasonal upwelling favours the increase in nutrients and a high primary productivity, supporting complex food webs, with fish and invertebrates that are the target of local fisheries (Wooster *et al.*, 1976; Relvas *et al.*, 2007; Sánchez *et al.*, 2007; Cravo *et al.*, 2010). Moreover, the park is situated in a biogeographic transition zone between warm and cold waters, also contributing to the high diversity and ecological importance of this area (Henriques *et al.*, 1999; Henriques *et al.*, 2007).

Those characteristics make this an important area for conservation purposes. Importantly, the area also attracts fishers and other users, who can benefit from the diversity of this area year round due to its protection from the prevailing north and northwest wind and waves, conferred by the high cliffs and the south facing direction of most of the park. Locally, artisanal fishers come mainly from Sesimbra, the small town with a strong historical fishing culture in the middle of the park. This increases the importance of the marine park for preserving both the ecosystem and human activities, especially local fisheries.

On the other hand, the unique features of this area make the conservation goals of the marine park more challenging to achieve, since conservation measures are more difficult successfully implement in areas with stronger human uses (Claudet *et al.*, 2006b; Lester and Halpern, 2008). Since local users had historical and cultural traditions in the region, the designation of a multiple-use Marine Protected Area (MPA) with zones with different protection levels was the management strategy adopted for the Arrábida Marine Park. With multiple use MPAs, conservation and fisheries benefits can occur simultaneously, together with a greater ecological and social-economic resilience if adequately designed, implemented and enforced (Floeter *et al.*, 2006; Lester and Halpern, 2008; Sciberras *et al.*, 2013). However, to promote local acceptance and commitment to its goals, it is crucial to evaluate those benefits and to disseminate such knowledge to users and managers. Therefore, one of the main contributions and relevance of the present study was the assessment of the initial ecological responses of the protection measures of this MPA to the rocky reef fish and target (i.e. commercial) invertebrate assemblages. Those ecological responses may then translate into fisheries benefits in a near future. However, this marine park is too young to expect a large fishery benefit and thus here we focused on assessing the ecological responses which are known to characterise the early reserve effects in marine protected areas. These results are important to disseminate to users and managers in order to empower them with the information needed for assessing the efficacy of the management plan to both conservation objectives and artisanal fisheries sustainability.

5.2 Summary of results

5.2.1 Recent reserve effect

Even though protection measures and zoning in the Arrábida Marine Park were only designated and fully established very recently, we found, based on multiple sources of data that some commercial species began to recover in biomass inside the reserve (no-take zone) when compared to fished areas. Although the management plan was approved in mid-2005, a transition period was established for fisheries that resulted in areas with stronger protection measures being progressively implemented between 2006 and 2009. Partially protected areas (PPAs) were all enforced after mid-2007 while the fully protected area (FPA) was completely established only after mid-2009. Since PPAs are essentially no-take areas for the nearshore reefs (i.e. the first 200 m from shore) we may assume that protection started in mid-2007, at least for less mobile species that are dependent on the shallow rocky reefs. In fact, rapid improvement in ecological responses were detected after 3-4 years in the nearshore reefs for the several commercial species and this study showed that such changes were most likely related to protection measures (Chapter 2). Other studies found similar early responses to protection (Halpern and Warner, 2002; Claudet *et al.*, 2006a), showing the importance of fishing closures in areas with ecological importance and intense human activities. The significant difference in biomass of the most targeted species between the reserve and fished areas and the increase in landings (weight) in recent years reinforce other studies where biomass was the most rapid and visible indicator of the reserve effect (Pelletier *et al.*, 2008; Di Franco *et al.*, 2009; Lester *et al.*, 2009).

In fact, we concluded that some of the most targeted species are gaining a greater opportunity to survive and grow in the protected rocky reefs, which may likely lead to greater fecundity of adults. Since some self-recruitment may be occurring locally (Borges *et al.*, 2009), the significant increase in density of some target species is also expected to take place in the coming years. Similar responses occurred in older protected areas around the globe until the carrying capacity of the reefs was attained (Lizaso *et al.*, 2000; Russ and Alcala, 2004; 2010). Even though significant density gains have not yet been detected, increasing abundance patterns were observed between before and after protection, especially for the most targeted species.

5.2.2 Adequacy of zoning

Although details of the zoning (i.e. the location of different protection levels) were decided based on previous studies of the complexity of habitats, the diversity of species and the abundance of recruits and juveniles (Almada *et al.*, 2000; Gonçalves *et al.*, 2003), we found that biomass response of targeted species above legal size was independent of habitat features, whereas both the biomass and density of non-target cryptobenthic species were associated with the diversity of different boulder sizes (Chapter 2).

As stated before, we defined the fully protected area (FPA) and the surrounding partially protected areas (PPAs) as our reserve, since in the PPAs, fishing with traps and jigs are only allowed beyond the 200 m offshore and thus the nearshore reefs are a continuous protected habitat. Although the total no-take area of the reserve is less than a dozen square kilometres, the initial positive responses in biomass of the most targeted species only a few years after protection seem to result from the management measures implemented in this MPA (Chapter 2). The size and design of the protection measures for the Marine Park are, therefore, apparently showing the first signs of ecological responses to protection, at least for the surveyed rocky reefs.

In fact, unpublished data about the home range of the most valuable fish surveyed in the rocky reefs, the white seabream (*Diplodus sargus*), suggest this species stays within the reserve most of the time. Preliminary analyses show that even the largest individuals moving greater distances during the reproduction season were rarely detected outside the protected reefs. In fact they showed an average home range of 0.65 km^2 which is contained within the reserve area (Abecasis et al. *unpublished data*). In the coastal area of the Algarve, this species was reported to travel a maximum of 5 km between two surveyed sites, with an average of 2.4 km (Abecasis *et al.*, 2013). *Diplodus sargus* was the most valuable fish assessed in the nearshore reefs by our visual surveys and had one of the largest positive responses in biomass and density, related to protection (Chapter 2).

Since large adults of this species stay within the no-take area, they can grow larger and reproduce, possibly explaining the rapid response of the biomass build-up of this species. Similar patterns are expected for other species, such as some other seabreams, that have analogous behaviour. Claudet *et al.* (2010) also showed reserve responses for demersal target fish species with facultative schooling behaviour (including *Diplodus spp.*). Additionally, they found positive responses for mobile species with wide home ranges, concluding that they may also benefit as much as sedentary species.

5.2.3 Fishery benefits

When the density of commercial species increases inside a reserve and those species have some mobility they are expected to move to fished areas, contributing to fisheries outside the protected area (i.e. the ‘spillover’ effect; Russ, 2002; Gell and Roberts, 2003). Goals of the present thesis did not include the assessment of the carrying capacity of the protected reefs, mechanisms of density-dependence, recruitment dynamics, species home ranges or assessment of potential gradients of density with the distance to the reserve borders and thus possible spillover effects were not assessed in the present work. Moreover, due to the very early stage of implementation of this marine park, these effects are not expected to be clearly detected. However, to better understand the ecological responses to protection we did evaluate fisheries landings patterns. The increasing trend in landings observed for the most valuable species after the park was established and only for vessels allowed to operate within the park, suggests that at the least there was no decrease in revenue for these fishers. This conclusion is reinforced by combining those trends with the build-up of biomass of such species observed within the reserve through underwater visual surveys, when compared to fished areas and to before data (Chapter 2).

Although our data suggest some benefits for fishers allowed to operate within the park, since landings and biomass apparently increased, we cannot be sure about the influence of the fishing effort in these patterns, unless we combine both sources of data. To the best of our knowledge, few studies have conducted such a comparison (Campbell *et al.*, 2012). The absence of these comparisons could possibly bias the detection of reserve benefits (Lester *et al.*, 2009), as differences between protected and fished areas may also arise if fishing effort increases in the surroundings of reserves, leading to a decrease in species density and biomass compared to trends detected inside.

5.2.4 Fishing gears

We assessed fishing effort allocation preferences and fishery displacement from before, during and after different protection measures were implemented (Chapter 3A). After the management plan was established, traps apparently lost one important fishing ground near the Espichel Cape, and some nearshore areas close to the reserve, whereas nets maintained their apparent preferred fishing grounds, which were already in the sandy areas near Sesimbra. However, some nets were operated in the south-eastern part of the current reserve, so they had to move after protection measures were implemented, possibly to outside the marine park borders but also reallocating to close to Sesimbra port, increasing the fishing effort in this buffer area. Although traps apparently lost important areas, our results suggest that among gear types, the octopus trap probably benefited most from the management plan since it has fewer restrictions than nets. Nets are only allowed in buffer areas (BAs) beyond 1/4nm offshore , whereas traps are only prohibited in the FPA. Local artisanal fisheries are characterized by operating multiple gears and targeting multiple species. We found that complex dynamics are involved in artisanal fisheries allocation but we also discuss that fishers with licenses for multiple gears may have stronger resilience to the loss of fishing grounds. Fishers with licenses for both nets and traps prefer to operate with traps more frequently than before, revealing an adaptation strategy in response to fishing closures, made possible only by the multiplicity of gears typical of artisanal fisheries. Since the selectivity and the target species of each gear differs, this adaptation pattern raises several questions: is the resource targeted by the newly preferred gear suffering higher levels of fishing pressure? How is the resource responding to this?

5.3 Combining and interpreting ecological and fisheries data

5.3.1 Octopus

The main target species of the trap fishery is octopus (*Octopus vulgaris*, Cuvier, 1797), accounting for 91% of the total catches (in biomass) of this gear for fishers operating at the Arrábida Marine Park (Alves, 2008). Considering only catches for sale, octopus account for 99% of the biomass of trap catches (Alves, 2008), justifying the designation of octopus traps.

Strong evidence of a reserve effect is not expected in highly mobile species, especially after only few years of protection. Although primarily territorial and strongly habitat and refuge associated, *Octopus vulgaris* performs large seasonal inshore-offshore migrations related to breeding and has a pelagic paralarva stage that is strongly influenced by environmental factors (Guerra, 1975; Roper *et al.*, 1984) and thus was not expected to have a strong response to a small nearshore MPA. However, our findings suggest that inside the reserve octopuses probably benefit from the opportunity to grow, thereby increasing the probability of successfully reproducing and generating offspring Juveniles are more abundant and larger inside the reserve and adults are significantly larger inside when compared to fished areas (Chapter 2).

Above all, the abundance of octopus is mainly dependent on oceanographic features which influence the paralarvae stage and its recruitment variability and success (Thiaw *et al.*, 2011). So, while protection measures in the nearshore can play an important role in enhancing reproduction and recruitment success, interannual variability of octopus is mostly influenced by environmental factors. Therefore, adaptive traditional management measures and an increasing knowledge about octopus are also required besides fishing closures. Since octopus spawning biomass may vary greatly between years due to environmental conditions, it is important to identify the main drivers affecting the interannual variability of recruitment. This knowledge could help guide predictions of octopus abundance, allowing rapid adaptation of management strategies to prevent overfishing in years of poor recruitment.

Octopus is a short-life span species with a high growth rate and thus it has the potential for high turnover and rapid renewal of the stock (Caddy and Rodhouse, 1998; Ezzeddine and El Abed, 2004). These characteristics make this species more resilient to high fishing pressure and to negative climatic events than longer-lived species. Despite this, if the fishing effort is continuously above the renewal limits of the species and if poor environmental conditions remain for some time, this resource could temporally collapse as has occurred with other cephalopods (Caddy and Rodhouse, 1998).

Even if the local octopus stock is being replaced and renewed under current environmental and fishing conditions, the increase in spawning biomass in protected areas during the reproduction season will probably contribute to a higher resilience of this resource. This is especially important because the trap fishery has recently increased effort in response to the management plan measures on other gear types and increasing market demand for octopus.

Here we show that fishing effort allocation (the total number of trap buoys) sharply increased from 2004 to 2009, due largely to the patterns observed in zone BA2 (Figure 5.1). This was expected due i) to the loss of fishing grounds in the FPA and PPAs, ii) to the large distance of the other BAs to the home port to and iii) to the number of vessels that operate more frequently with traps after the management plan. Even in the PPA2 and PPA3 the number of trap buoys increased temporally (Figure 5.1), suggesting that even beyond 200 m offshore, traps possibly benefit from protection when octopus are moving. The analyses of the temporal trends of LPUE (landings per unit effort: kg/number of buoys) revealed a steep decrease in octopus LPUE observed from before the park to 2007 (Year 1) when all PPAs were already enforced (Figure 5.2). After this, the total number of buoys continued increasing sharply but the LPUE remained relatively stable (Figure 5.1, 5.2). These results have several interpretations: i) the increase of the number of buoys from before to after the management plan approval can be related to legal aspects and to a stronger vigilance after, since several fishers previously avoided signalling the position and extent of their traps sets; ii) the decrease in the LPUE from before the park to 2007 (Year 1) might be associated to a consequent disproportionate increase in buoy density related to the described phenomenon in i); iii) Even if the number of signalling buoys increased after the management plan, the sharp increase between 2007 (Year 1) and 2009 (After the park) is probably mostly driven by more buoys inside the park (proxy for more traps), since most fishers were already aware about their obligation to use buoys since 2005-2006 when park rangers were more actively informing fishers. Also fishers complying with the rules (of correctly signalling their buoys) could have a higher probability of maintaining their licenses and that might have influenced their behaviour.

The relative stability of the LPUE since 2007 (Year 1), even with the abrupt increase in the number of buoys within the park, could be related to an increase in octopus biomass. It is too early to be able to confirm if this pattern is related to protection measures, but having fishing closures in spawning areas could possibly lead to positive responses in the future. Both our fishing effort (Figure 5.1) and census surveys when compared to total landings (Figure 5.3) and LPUE (Figure 5.2), point to an increase in biomass for this species after the establishment of the marine park, even with higher trap density. However, due to the high interannual variability of this short-lived species and its dependence on oceanographic drivers, our results should be interpreted with caution.

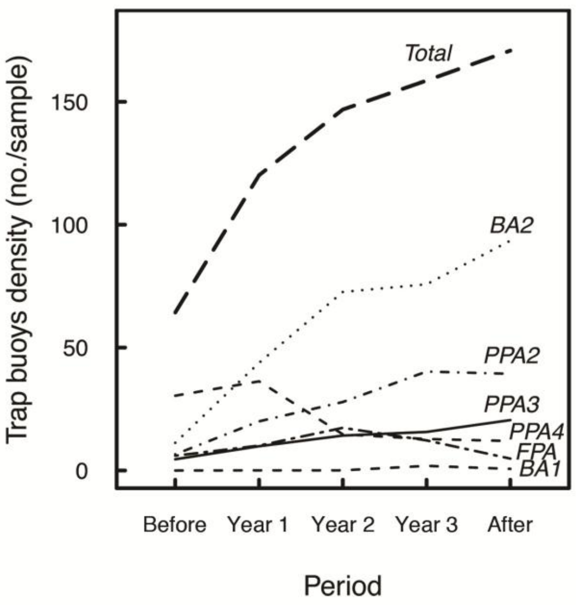


Figure 5.5 - Density of trap buoys by protection area and combining all sampled zones at the Arrábida Marine Park througouth the different periods of the implementation of the management plan: Before, Year 1, Year 2, Year 3 and After (see Chapter 3A for details).

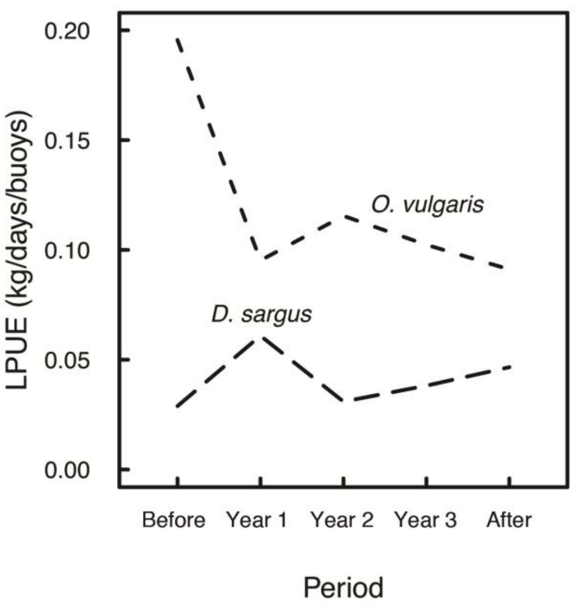


Figure 5.6 - Landings per unit effort (LPUE) of *Octopus vulgaris* and *Diplodus sargus* througouth the different periods of the implementation of the management plan: Before, Year 1, Year 2, Year 3 and After. Unit effort refers to the number of days fishing and the number of buoys (traps and nets for *O. vulgaris* and *D. sargus*, respectively) detected in each fisheries surveying (see Chapter 3A for details).

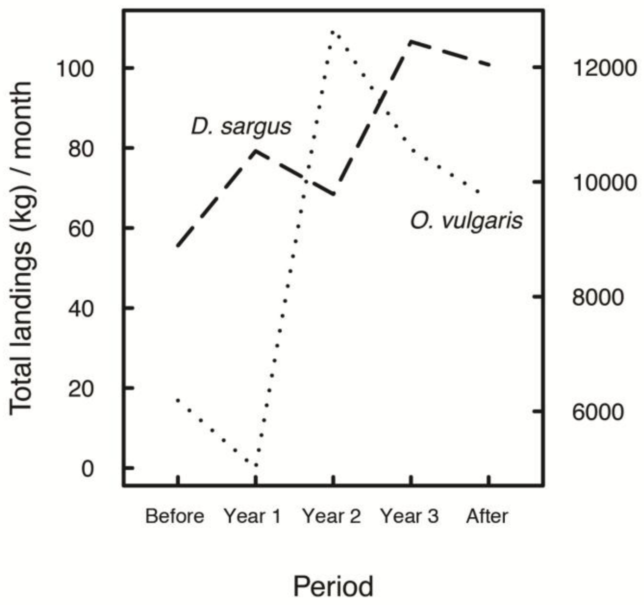


Figure 5.7 - Total fisheries landings of *Octopus vulgaris* and *Diplodus sargus* througouth the different periods of the implementation of the management plan: : Before, Year 1, Year 2, Year 3 and After (source: General Directorate of Fisheries and Aquaculture; see Chapter 2 and 3A for details). Landings are in kg/number of months included. Landings of *D. sargus* are in the left y-axis whereas landings of *O. vulgaris* are in the right y-axis.

5.3.2 White seabream

Fish account for most (64%) of the catches in biomass of trammel and gill nets, followed by molluscs (25%). They also dominate longline catches (Alves, 2008). From this group the white seabream (*Diplodus sargus*) is an important representative of demersal species from rocky reefs and is the most valuable fish from our surveys. Other target cryptobenthic species occurring in the nearshore reefs may have higher commercial value but are secretive and difficult to accurately assess by diurnal underwater visual surveys and thus were not included in our analyses. *Diplodus sargus*, other large sparids and *Octopus vulgaris* were the main targeted species of spearfishing (Rocklin *et al.*, 2011), a common recreational fishery before the management plan implementation.

In our fisheries distribution surveys, we did not detect longline vessels since they operate at night and return to home port at sunrise and also since the few vessels using this gear can only operate in the BAs. We also did not detect spearfishers as they were forbidden since 2005. Therefore, here we tested the influence of the effort density of the only static gear we could relate to seabreams after the management plan was implemented (nets).

Contrary to results from traps, the trend of net buoy density within the park decreased from the before period (2004) to afterwards, but again it was mostly due to patterns observed in the BA2 (Figure 5.4). The decrease in the number of buoys (proxy for nets) even where nets are allowed, and in an area encompassing the main fishing grounds of this fishing gear (Chapter 3A), is likely related to a shift in fishers' preferences to other gears, such as traps. Moreover, in the BA2 the competition for space to set a static fishing gear has become very high. In the remaining areas (PPAs and FPA), there was also a decreasing pattern, which was expected since nets were excluded. The BA1, the buffer area further east, was the less used throughout the periods, probably due to the large distance to port and because before the management plan was approved, nets found in this area were mainly drift nets from Setúbal. So, fishers from Sesimbra possibly avoided the sandy areas of BA1 because they preferred to be as close as possible to the home port and/or to their former traditional fishing grounds where they have a greater ecological knowledge of target species and habitats (Chapter 3A and 3B).

The white seabream increased in total landings (Figure 5.3), and in LPUE only during 2007 (Year 1; Figure 5.2). The LPUE patterns appear related to the variability in the buoy density between periods, but since *D. sargus* is also frequently captured by other gears, such as longlines, the total landings per month from the licensed fishers of the park (without grouping by gears) may give us more useful information concerning patterns of white seabream abundance (Figure 5.3).

Moreover, major changes from before to after the management plan approval are probably more related to the exclusion of spearfishing. This is a selective fishery that, as with other recreational fisheries, has been shown to have large impacts on the biomass of target species, since even moderate fishing effort performed continuously can remove a significant proportion of larger fishes (Cooke and Cowx, 2004; Di Franco *et al.*, 2009), which may be also true for longline fisheries.

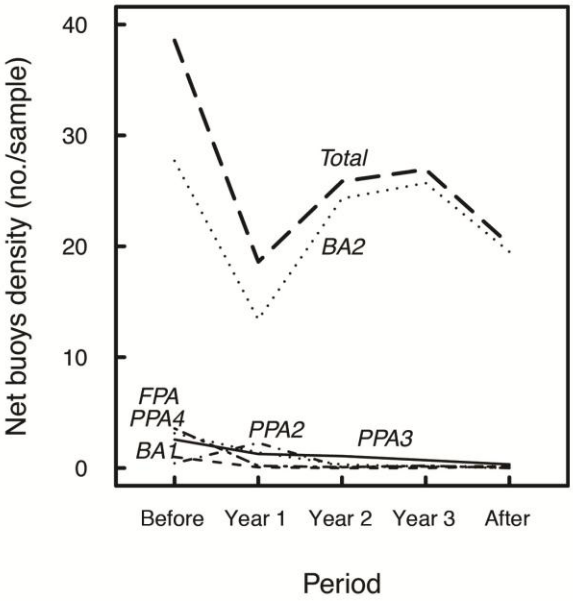


Figure 5.8 - Density of nets buoys by protection area and combining all sampled zones at the Arrábida Marine Park througouth the different periods of the implementation of the management plan: Before, Year 1, Year 2, Year 3 and After (see Chapter 3A for details).

Diplodus sargus is apparently one of the species that has benefited most from protection (Chapter 2), since tagged white seabreams seem to be able to stay year round within the area of the protected reefs. The white seabream responded in biomass and density between protected and fished areas, and although not significant there was an increase in density in the reserve area from before to after protection. Moreover, this species showed an increasing trend in landings, suggesting a positive effect of protection. Our findings combining multiple sources of data support the study of Claudet *et al.* (2010) which suggests that seabreams are species with potentially rapid and significant responses in protected areas.

The combination of fishing effort trends, landings patterns and direct observations of biomass and its relation to habitat features are very rare. Our findings constitute one of the few studies combining multiple data types, reinforcing the early reserve effect detected in the recently established and small no-take area at the Arrábida Marine Park.

5.4 Climatic influence in the Arrábida Marine Park

The Arrábida Marine Park is located at an important biogeographic transition zone with cyclical climatic fluctuations. It is strongly influenced by the North Atlantic Oscillation

(NAO) affecting coastal winds, temperature and current patterns (Henriques *et al*., 2007). Seasonal upwelling events are driven by strong north wind stress and are responsible for local productivity and peaks in recruitment, due to the rise of cold, nutrient-rich deep water (Relvas *et al.*, 2007; Sánchez *et al.*, 2007; Cravo *et al.*, 2010).

We found that despite the importance of the large-scale NAO to interannual climatic oscillations (Henriques *et al.*, 2007), regional and local drivers such as mean winter sea surface temperature and the northward component of the wind stress, strongly influence the rocky fish community structure (Chapter 4A). Interannual variability in the occurrence of species is related to their climatic affinity and to environmental fluctuations. Years with stronger south storms and weaker north winds and upwelling events, have higher SST and northward currents that can facilitate the dispersal, movement and retention of more tropical species in the area of the marine park. In contrast, cold years with strong upwelling driving winds have higher occurrence of cold-temperate species (Henriques *et al.*, 2007). This rapid and strong climatic influence on the fish assemblages is strongly related to the fact that this is a biogeographic transition zone where cold and warm species have their southern and northern limits, so they are able to occupy their distribution range edges in years with suitable environmental conditions.

5.4.1 Climatic influence in a continental island

The predominant alongshore current in western Portugal, the “Canary current”, usually carries cold-species propagules from the north (Bischof *et al.*, 2003). However, years with consistent upwelling events during spring blooms may lead to considerable offshore transport of eggs and larvae (Santos *et al.*, 2001) and to high natural mortality. Despite this, in the Arrábida Marine Park, warm water associated species appear to be more limited than cold water associated ones, explaining their faster response to recent warmer years (Chapter 4A). This is probably related to the previous long period (1989-1995) of cold years and positive winter NAO (Henriques *et al.*, 2007; Chapter 4A). Furthermore, the Espichel Cape and the southward orientation of the marine park possibly function as a barrier to dispersing eggs and larvae carried by south currents. Also, extensive sandy beaches surround this region to the north and south, making this area a ‘continental island’ for rocky species (Chapter 4B). These features may contribute to retain recruits and adults moving north since this area provides complex rocky settlement habitats along several kms.

5.4.2 'Barometer' of climate change

Our models of fish communities and environmental ocean conditions hindcast for the last 50 years suggest a progressive 'tropicalization' of the rocky fish community due to ocean warming, since in the last two decades values of the ratio between tropical and cold-temperate species greater than one were more frequent, revealing that there is an increasing importance of tropical species in these coastal assemblages (Chapter 4A). These results support other evidence that species are shifting polewards (Perry *et al.*, 2005; Cheung *et al.*, 2009; Hawkins *et al.*, 2009; Cheung *et al.*, 2012; Wernberg *et al.*, 2013). Our findings also indicate that cold species are retreating more slowly than the advance of tropical species (Hawkins *et al.*, 2009), since in our statistical models we only found indicator species in warmer years and with tropical or warm-temperate affinities (Chapter 4A).

The tropicalization of fish assemblages is occurring with increases in species which have northern distribution edges and/or historical presence in the region (Baldaque da Silva, 1891; Henriques *et al.*, 2007). However, recently, vagrants from distant tropical species have been recorded for the first time in the region (Chapter 4B) pointing to possible larger distribution shifts induced by consecutive warmer years and more frequent extreme events.

The rapid responses of the rocky fish assemblages to warming, even for species theoretically more adapted to cyclical climatic oscillations and thus more resilient to oceanographic changes, suggest that this area can be considered a 'barometer' for climate change. In fact, temperate fish assemblages from transition areas vary among years depending on oceanographic and climatic conditions, but important additive changes (e.g. continuous tropicalization) possibly occur only when there is a disruption of the natural fluctuations for some time, which is attained by steady ocean warming. Thus, transition zones, such as Arrábida, where responses of more resilient assemblages are detected and can be related to warming, are important to assess persistent climatic changes.

5.5 MPAs and fisheries in a warming ocean

Future scenarios of ocean warming and our recent projections showing the tropicalization of the rocky fish assemblages in the last decades, suggest further changes will occur in the future. Nine of the twelve species with tropical affinities included in the tropicalization index have commercial value, whereas only three of the eleven cold-temperate species are targeted by local fisheries. Thus, since a higher proportion of tropical species are commercial, when compared to cold-temperate ones, the fish community response detected in our study comprises an increasing occurrence of commercial species. A possible shift towards a system with other commercial species might lead to future changes in fishers' target preferred species and to a tropicalization of fishing catches (Sumaila *et al.*, 2011; Cheung *et al.*, 2012; Cheung *et al.*, 2013). In fact, coastal artisanal fisheries are strongly dependent on local commercial species, and these predicted changes will possibly lead to large biological and socio-economic impacts in the near future. How fish community structure and fishers will adapt to different sources of impacts is uncertain.

MPAs are suggested to increase the resilience of ecosystems in relation to a warming future, reducing the impact of fishing and other human uses (Ling *et al.*, 2009; McLeod *et al.*, 2009). In the same way, MPAs (in particular marine reserves) also allow an unbiased assessment and understanding of the warming impacts in species assemblages and in ecosystem functions (Cheung *et al.*, 2011). However, single MPAs cannot protect commercial species shifting their distribution ranges due to persistent major oceanographic changes. Thus, networks of MPAs must be designed and planned to cope with this challenge and possibly mitigate non-linear and unpredictable responses of species and ecosystems in this changing world (Munday *et al.*, 2008).

5.6 Conclusions and future directions

The unique characteristics of this marine park such as its habitat complexity and species diversity in addition to the regional isolation of its rocky reefs and to its geographic position within an important temperate transition zone and upwelling region, all contribute to the ecological role and ecological and socio-economic relevance of the Arrábida Marine Park. The protection of such an important ecosystem may increase the socio-ecological resilience to future challenges and uncertainties related to climate change and fisheries. Therefore a deeper knowledge of the area and a continuous monitoring of species and habitats should be part of present and future goals.

If this area maintains spawning adults due to suitable habitats and ecosystem diversity and complexity, it is likely that recent protection measures will also contribute to the improvement of the 'source' function of this marine park. On the other hand, if pelagic larvae are retained or reach this area by mechanisms of dispersal, and if settlement is successful in the local rocky reefs, this area may also represent a 'sink' for several rocky species (Henriques and Almada, 1998; Borges *et al.*, 2009).

Since these reefs may provide functions of source and sink and they are isolated from similar reefs in a range of several kms, they may act as an important step in the dispersal and connectivity patterns of rocky species moving north or south. The potentially important role of this area in the connectivity with other coastal zones should be further studied.

We have highlighted the importance of this area in understanding distributional changes of species in relation to climate and shorter-term oceanographic features. This work sets the scene for further work on connectivity.

In fact, in this dissertation we show that a tropicalization of species assemblages and a shift in species ranges already started to happen. Moreover, we conclude that species are responding to protection and are exposed to fishing mortality outside the reserve. Also, we indicate that small-scale fisheries are strongly dependent on this area and on local resources.

Therefore, since this MPA is vulnerable to climate and oceanographic variability and to fishing pressure, it is important to predict what is going to happen in the future to the most targeted species under different climatic and fisheries scenarios. If this MPA aim at protecting such species and ecosystems, as well as coastal communities' food security, it is important to understand if protection can be efficient in the future.

Indeed, marine protected areas cannot cope for environmental and human-induced challenges alone. Complementary management strategies and a network of marine protected areas should be planned and enforced. Some authors refer that only connected MPAs may function as a true network, by ensuring protection of species and habitats in a set of connected sites instead of including only a group of single reserves (Grorud-Colvert *et al.*, 2011). These networks are defined by a group of reserves connected by the dispersal of larvae and/or movement of juveniles and adults (Planes *et al.*, 2009), but should also achieve conservation, management and regional goals (Grorud-Colvert *et al.*, 2011).

A network of MPAs has to be established in Portugal (to accomplish international agreements), and recently some MPAs have been designated throughout the country extending terrestrial reserves into the coastal areas, but this has happened without planned or designed management plans. Therefore, it is important to know if some of the species produced or growing in the Arrábida Marine Park, especially the most targeted species, reach other protected areas, where the spawning biomass of those species might be also secured.

For the above mention reasons, the role of the Arrábida Marine Park in the context of a regional connectivity matrix should be further explored, by means of the distance to other suitable habitats, local and regional current patterns and species dispersal ranges. The recruitment dynamics and the temporal and spatial variability of larvae are being studied locally, but it would be very important to further study the extent of export and retention between this marine park and surrounding areas. It is crucial to understand if this area is dependent on others, since if this is true (which is likely due to the small area considered), commercial species with their source elsewhere should be managed carefully or those spawning areas should be detected and protected. Otherwise, protecting only the sink would not prevent overfishing such species.

Further studies must also be carried out in the marine park aiming to address questions about density-dependence mechanisms and carrying-capacity, since the success of the recruitment and growth of local species are likely to lead to an increase in density and biomass in the next years until the habitats capacity is attained. After that, species interactions may promote larger movements of individuals into adjacent areas and fisheries benefits could be more evident.

That is why continuous monitoring of species and fisheries are also required. The intra and inter-specific relations should also be the aim of further studies since unpredicted trophic interactions may occur with some species, and possibly some functional groups, decreasing their proportion and role in the community structure, likely affecting local ecosystems.

Species site fidelity and home ranges should be further understood as well, to allow for more accurate conclusions when assessing conservation and fisheries benefits. Additionally, species ontogenetic movements are crucial to understand, and this information is still lacking for several species. Detecting areas of spawning, recruitment and growth help to better protect a portion of all essential habitats, especially for the most targeted and vulnerable species, whose growth and reproduction might be compromised.

Combining ecological and fisheries data should be also a short-term objective. Here we combined such data for the most targeted species suggesting positive responses to protection after only a few years of management measures, but a more in depth and continuous analysis must be carried out to assess patterns after more years of protection and also for soft bottoms. One key aspect is the effectiveness of surveillance and enforcement measures since we had recent worrying reports of an increase of poaching and illegal activities inside the reserve due diminishing surveillance activities both by the marine park authority and the maritime police.

All these aspects could help to support some of our findings and to increase the knowledge of such an important MPA and their relation to adjacent areas and thus we strongly recommend them to be considered.

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