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Planning and management of Marine Protected Areas: methodological approaches to cope with data scarcity

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O futuro da Humanidade será tão melhor quanto melhor cuidarmos dos ecossistemas.

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Abstract

Changes in ecosystems structure and function due to high impacts of human pressure on oceans have led to the increasing numbers of Marine Protected Areas (AMP) as tools for conservation and fisheries management. Habitat recovery, increases in density, size and biomass of organisms within MPA and beyond their limits have often been found. However, MPA effectiveness is compromised by the interaction of several factors, such as the inadequate conduction of planning and management processes and the lack of appropriate scientific datasets to support decisions. In order to contribute to MPA effectiveness, several approaches to cope with data scarcity and low accuracy were developed and then applied to different case studies. Data analyses were mainly based in published studies and official datasets. Geographic information systems and multivariate statistical analyses were the main methods applied. Cumulative effects of human activities on the Portuguese coast were assessed and showed that implemented MPA are usually under high human impacts from different sources (e.g. fisheries, land-based activities). Small-scale fisheries are often the main economic activity occurring in MPA and a new monitoring method combining data on spatial distribution of fishing effort, on-board observations and official landings records revealed to be an effective approach to the assessment of MPA impacts on small scale fisheries. A framework based on fish life history and habitats used was developed and showed this approach can indicate areas that would potentially increase the effectiveness of protection measures. Finally, the status of marine conservation in SW Europe and the factors mostly contributing to their effectiveness were assessed and showed that high MPA effectiveness is usually related with strong stakeholders support, with suitable goals, management and enforcement. Overall, this thesis results highlighted that the investment in strategies aiming at maximizing MPA performance is urgent and crucial in a context of increasing MPA coverage.

Keywords: Sustainable management of marine resources; marine protected areas; small-scale fisheries; fish's life history; anthropogenic pressures.

Resumo

Os impactos humanos nos ecossistemas marinhos têm provocado alterações na sua estrutura e função, levando ao aumento do número de Áreas Marinhas Protegidas (AMP) como ferramentas de conservação e gestão de pescas. A recuperação de habitats e o aumento de densidade, tamanho e biomassa dos organismos dentro e fora dos limites das AMP são resultados frequentes. A eficiência das AMP é frequentemente comprometida pela conjugação de fatores como a implementação inadequada dos processos de planeamento e gestão e a falta de informação científica apropriada para suporte à tomada de decisões. Desenvolveram-se diferentes abordagens, aplicadas a vários casos de estudo, pretendendo contribuir para o sucesso das AMP quando a informação disponível não é a ideal. Este trabalho baseou-se principalmente em dados publicados ou de fontes oficiais. As metodologias de análise foram Sistemas de informação geográfica e estatísticas multivariadas. O estudo dos efeitos cumulativos das atividades humanas na costa portuguesa revelou que as AMP estão sob elevadas pressões provenientes de diferentes fontes (e.g. pescas, fontes terrestres). Desenvolveu-se um método de monitorização de pescas combinando esforço de pesca, observações a bordo e desembarques oficiais que revelou ser apropriado para uma monitorização eficaz das AMP. Utilizou-se uma metodologia inovadora que integra informação sobre os habitats utilizados durante o ciclo de vida das espécies e habitats protegidos e concluiu-se que esta permite maior eficácia na escolha das zonas a proteger, contribuindo para a eficiência das AMP. Por fim, as características das AMP do SW europeu foram analisadas e foram identificados os fatores que contribuíram para a eficiência das AMP existentes (i.e. apoio por parte dos utilizadores da AMP e maior adequabilidade dos objetivos, da gestão e de fiscalização). No geral, esta tese mostrou que no contexto atual é urgente investir em estratégias que maximizem o bom desempenho e a extensão da área abrangida por AMP.

Palavras-chave: Gestão sustentável dos recursos marinhos; áreas marinhas protegidas, pesca artesanal, ciclo de vida dos peixes, pressões antropogénicas

Resumo alargado

Muitos dos ecossistemas marinhos encontram-se sob forte pressão humana, o que tem levado ao aumento das preocupações com a sustentabilidade da exploração dos recursos do Oceano, tornando inquestionável a necessidade de estabelecer estratégias para a conservação e recuperação de habitats e biodiversidade marinhas. A criação de Áreas Marinhas Protegidas (AMP) é um dos mecanismos mais utilizados na proteção do Oceano e têm sido globalmente consideradas como ferramentas adequadas quer para a conservação dos ecossistemas marinhos, quer para a gestão sustentável das pescas. O ambiente multidimensional (socioeconómico, cultural, político) onde as AMP se enquadram e os múltiplos objetivos que são estabelecidos dificultam muitas vezes o seu sucesso. A implementação eficaz de AMP está assim dependente do desenvolvimento adequado de todos os processos, desde o planeamento até à implementação e gestão. A boa condução destes processos implica à partida o envolvimento que equipas multidisciplinares e a participação ativa dos principais utilizadores da área em questão de forma a integrar decisões que reúnam consenso generalizado, uma vez que tem sido demonstrado em diversos estudos que a omissão das vertentes socioeconómica e cultural nestes processos leva, em geral, à contestação da AMP e ao incumprimento das regras implementadas, dificultando ou inviabilizando o sucesso destas zonas. A tomada de decisões e a adequada condução dos processos de implementação de AMP está ainda relacionada com a qualidade e adequabilidade da informação científica que lhes serve de suporte (e.g. características e distribuição espacial e temporal de habitats e espécies, caracterização socioeconómica das comunidades locais, dados da pesca). No entanto, na maioria dos casos, a informação científica de suporte não é a ideal, está dispersa por diversas fontes, não abrange as escalas temporais e espaciais necessárias, não engloba os vários grupos biológicos e não existe para muitas das questões socioeconómicas, entre outras lacunas. Assim, o objetivo geral desta tese é desenvolver metodologias que permitam utilizar de forma eficiente a informação disponível de modo a maximizar a sua utilidade nos processos envolvidos nas fases de planeamento e gestão de AMP e assim contribuir para o aumento dos níveis de sucesso destas áreas.

Esta tese é constituída por seis capítulos que correspondem a uma introdução geral sobre o tema da tese, quatro artigos científicos publicados ou em revisão em revistas internacionais de arbitragem científica indexadas no Science Citation Index, que englobam abordagens mais específicas sobre aspetos do tema central da tese, e, um capítulo final em que são apresentadas as principais conclusões e perspetivas futuras.

No capítulo 1, introdução geral, são apresentadas algumas considerações sobre as atividades humanas que afetam o meio marinho e os seus principais efeitos nos ecossistemas marinhas; são apresentadas algumas das ferramentas de conservação, destacando-se as AMP; é feita uma breve apresentação que enquadra a sequência de eventos e documentos internacionais que têm marcado o desenvolvimento da ciência em torno das AMP a nível global e, por fim, são apresentadas algumas considerações sobre como deve decorrer todo o processo de planeamento, implementação e gestão das AMP de modo a que estas sejam potencialmente eficientes.

No capítulo 2 é desenvolvido um método de caracterização das atividades humanas ao longo da costa portuguesa (mar territorial). Numa primeira fase foram identificadas e mapeadas as principais atividades que ocorrem ao longo da costa (por exemplo pescas e náutica de recreio), ou que nela têm impacto direto (por exemplo pressão proveniente dos estuários), recorrendo a um Sistema de informação geográfica. Após a caracterização individual de cada atividade foram calculados valores de pressão cumulativa para diferentes zonas da área de estudo, nomeadamente para as AMP. Os níveis de pressão cumulativa mais elevados ocorreram em zonas mais próximas da costa (profundidade inferior a 30m), onde se localizam também as AMP estudadas. A maioria das AMP está localizada em zonas próximas dos maiores centros urbanos do país e as pressões que ocorrem nas zonas adjacentes são também elevadas. Nas AMP consideradas, as atividades mais relevantes foram as pescas e as pressões resultantes da influência das águas de transição (estuários). O índice utilizado constitui uma ferramenta de base nos processos de planeamento e gestão do meio marinho, nomeadamente nas AMP, uma vez que permite integrar informação sobre a localização e intensidade das atividades humanas existentes e assim minimizar as situações de incompatibilidade das atividades humanas com a conservação do meio marinho.

No capítulo 3 é feita uma avaliação dos efeitos da implementação da AMP da Arrábida (Portugal) nas pescas locais através da combinação de três tipos de informação: distribuição espacial do esforço de pesca na AMP, dados de capturas obtidos a bordo de algumas embarcações locais e dados oficiais de desembarques das embarcações licenciadas para pescar na AMP. As três fontes de informação foram analisadas conjuntamente para o período de 2004 a 2010, considerando o mesmo grupo de embarcações. A análise conjunta da informação mostrou-nos que os resultados obtidos através da análise isolada de dados de desembarques em lota apenas permite a identificação de padrões gerais das capturas, uma vez que há grandes discrepâncias entre as capturas e os desembarques oficiais. Em termos gerais, verificou-se um aumento do esforço de pesca com covos, acompanhado pelo aumento dos desembarques de polvo. A utilização de redes diminuiu com a implementação da AMP e as capturas de algumas espécies mais associadas a esta arte de pesca (por exemplo, linguados) mostraram tendências mais constantes ao longo do tempo. Os desembarques globais (em biomassa) aumentaram ao longo do período em análise. Apesar da discrepância entre capturas reais e desembarques, a integração dos três tipos de informação referidos revelou ser um método apropriado na caracterização da evolução da dinâmica das pescas em resposta à implementação da AMP. O método utilizado pode ser muito útil no futuro, se implementado num período de tempo mais alargado, com recolha dos três tipos de informação em simultâneo e incluindo aumento do esforço de amostragem a bordo de embarcações de pesca. Num contexto em que os recursos humanos e financeiros são normalmente escassos, o método utilizado poderá ser útil também na determinação de fatores de correção aos valores de desembarques oficiais, permitindo a sua utilização futura como indicador da evolução das pescarias em AMP ou noutros contextos de gestão de pescas.

No capítulo 4 desenvolveu-se um quadro conceptual de avaliação do potencial de uma AMP para ser eficiente na proteção da biodiversidade. O método integrou informação sobre a relação entre espécies e habitats, o ciclo de vida das espécies de peixes que ocorrem numa AMP e os seus grupos funcionais e a pressão a que os habitats estão sujeitos. O método parte do pressuposto que para proteger a biodiversidade existente, todos os habitats necessários ao desenvolvimento do ciclo de vida das espécies deveriam estar protegidos e sob baixos níveis de impacto e a sua finalidade é assim identificar as maiores falhas na proteção de determinados habitats e propor medidas que permitam a adequação da AMP aos objetivos de conservação da biodiversidade. O quadro conceptual foi também aplicado à AMP da Arrábida de modo a exemplificar os seus potenciais resultados práticos. Esta AMP demonstrou englobar habitats adequados ao ciclo de vida da maioria das espécies de peixes que nela ocorrem, no entanto, alguns habitats necessários como zonas de viveiro ou desova para algumas espécies de grande interesse comercial não estão englobados na AMP. Estes resultados são discutidos no âmbito da gestão adaptativa das áreas já implementadas. O quadro desenvolvido é útil sobretudo como um método de suporte à tomada de decisões precoces no que diz respeito à implementação ou adequação das medidas de gestão de uma AMP uma vez que permite uma rápida identificação de medidas prioritárias para o cumprimento dos objetivos da AMP ou reconhece a necessidade de adaptar esses objetivos.

No capítulo 5 compilou-se informação sobre as características gerais das AMP existentes (idade, área total, área de total interdição, objetivos, localização e gestão) no SW europeu (Portugal, Espanha e França) de modo a avaliar o estado geral de conservação destas áreas em relação ao exigido em diversos documentos internacionais. Para além desta caracterização geral foram feitos questionários aos gestores das AMP consideradas, pedindo-lhes uma caracterização e uma perceção da adequabilidade dos processos de planeamento, gestão, monitorização, governança e fiscalização das suas AMP. As respostas obtidas foram analisadas de modo a identificar os fatores que mais contribuem para maiores níveis de sucesso global das AMP. Verificou-se que a maioria das AMP da área de estudo têm dimensões reduzidas (cerca de 50% das áreas tem menos de 20 km^2), localizam-se em zonas costeiras e têm normalmente uma multiplicidade de objetivos, desde a conservação de espécies e habitats até ao desenvolvimento de diversas atividades económicas (e.g. pesca profissional, naútica de recreio, pesca de recreio, mergulho). Apenas 9% das AMP tem mais de 1000 k m^2 e a sua distribuição na área de estudo é bastante desigual (maioritariamente nos Açores e França). Globalmente, 46% das AMP e 59% da área coberta foram designadas nos últimos cinco anos, no entanto, o número de áreas de total exclusão de atividades foi bastante mais reduzida. O suporte das populações locais, a adequabilidade dos objetivos, da gestão e de fiscalização sobressaíram como os fatores que mais contribuíram para uma maior eficiência das AMP estudadas. A integração dos resultados permitiu concluir que apesar dos indicadores serem positivos, existe ainda uma falta de coerência, representatividade e eficácia das AMP implementadas. Assim, sublinha-se neste capítulo a necessidade de ampliar a área atualmente coberta por AMP, tendo contudo em atenção a integração das novas áreas e medidas de gestão com as atualmente existentes, de modo maximizar a representatividade e coerência das redes de AMP. Estas redes deviam incluir áreas, potencialmente de dimensão reduzida, que conferissem maiores níveis de proteção a zonas “chave” para a conservação dos ecossistemas marinhas. A necessidade de melhorar os programas de monitorização e adequar as medidas de fiscalização foi também evidente.

No último capítulo, apresentou-se um resumo das principais conclusões obtidas nesta tese e tecem-se comentários sobre futuras abordagens, nomeadamente sobre a necessidade de recolher informação mais detalhada que permita definir com maior exatidão as áreas que efetivamente podem contribuir mais para a proteção dos ecossistemas marinhos permitindo otimizar as relações de custo-benefício inerentes à implementação e gestão de AMP. A implementação de métodos padrão de monitorização das AMP, contínuos no tempo, que possam ser integrados e permitam no futuro uma avaliação global dos sistemas de AMP implementados e identificação das maiores necessidades de conservação e a adaptabilidade das medidas de gestão.

List of papers

This thesis is comprised by the papers listed below, each corresponding to a chapter (Chapters 2 to 5). The author of this thesis is the first author in all papers and was responsible for conception and design of the work, field surveys, sample collection and processing, data analysis and manuscript writing of all the papers. Remaining authors collaborated in some or several of these procedures. All papers published were included with the publishers' agreement.

CHAPTER 2

Assessment of cumulative human pressures on a coastal area: integrating information for MPA planning and management

Marisa I. Batista, Sofia Henriques, Miguel P. Pais, Henrique N. Cabral

Published in Ocean & Coastal Management 102: 248-257 (2014)

CHAPTER 3:

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

Marisa I. Batista, Bárbara Horta e Costa, Leonel Gonçalves, Miguel Henriques, Karim Erzini, Jennifer E Caselle, Emanuel J. Gonçalves, Henrique N. Cabral

In review in Fisheries Research

CHAPTER 4:

A framework to the rapid assessment of MPA effectiveness based on life history of fish species

Marisa I. Batista, Sofia Henriques, Miguel P. Pais, Henrique N. Cabral

In review in Aquatic Conservation: Marine and Freshwater Ecosystems

CHAPTER 5:

An overview of SW Europe Marine Protected Areas: factors contributing to their effectiveness

Marisa I. Batista, Henrique N. Cabral

In review in Ocean & Coastal Management

CHAPTER 1

General Introduction

General Introduction

Threats on marine ecosystems and services they provide

Marine ecosystems are among the most productive ecosystems on earth, providing numerous goods and services, from gas regulation, nutrients cycling, energy source and food production to the underpinning of several economic activities and social and cultural services to human communities (Costanza et al., 1997). Marine and coastal ecosystems services were once believed to be inexhaustible. However, the past decades have proved that marine ecosystems and resources are limited, vulnerable and becoming increasingly degraded (Toropova et al., 2010). The continuous rise of human population densities, namely along world's coasts (Martínez et al., 2007) and the rapid advances in technology are strongly contributing to alarming levels of anthropogenic impacts on marine ecosystems currently observed. Halpern et al. (2008) found that worldwide no marine area is unaffected by human influence and that a large fraction (41%) is strongly affected by multiple drivers. In that global study, the highest impacts were found on hard and soft continental shelves and rocky reefs while shallow soft-bottom and pelagic deep-water ecosystems had the lowest impacts.

The impacts affecting marine ecosystems worldwide originate from several sources related with anthropogenic activities. For instance, sea pollutants that usually derive from human settlements (urban wastes and sewage sludge), resources uses and exploitation (oil spills, alien species, dirty fishing such as bottom trawling), agricultural activities (fertilizers, pesticides and agrochemicals, sediments), industrial developments (heavy metals and trace elements, industrial wastes), aquaculture (alien species, sediments, organic inputs), shipping (noise, introduction of alien species) and touristic uses (diving, recreational boating) (Islam and Tanaka, 2004; Williams, 1996). The impacts on oceans are visible in complex and fundamental alterations to marine ecosystems (and consequently in all ecosystems as they are intrinsically connected), such as quasi-extinctions and global decrease in biodiversity (fishes, seabirds, invertebrates), decline in open-ocean and coastal fisheries stocks, degradation and destruction of natural habitats (e.g. nursery areas), populations decline, interruptions in life cycles, malformations, poor reproductive success, changes in water chemistry, environmental changes (e.g. water temperature increase, ocean acidification) (Ban and Alder, 2008; Islam and Tanaka, 2004; Lotze et al., 2006; Palumbi et al., 2008; Rice et al., 2012; Ruckelshaus et al., 2008).

This plethora of negative effects from human activities on oceans have led to the development of several approaches aiming to oceans conservation. The 1970s marked a decade of general recognition of the insufficiency of management of marine resources and habitats, which led to a growing interest in approaches to ensure the continuing viability of marine ecosystems (Toropova et al., 2010). Some of the strategies implemented rely on single-species based approaches that aim to protect a species in a given area, namely with fisheries management purposes (e.g. seasonal fishing closures, catch-control measures, fishing effort limitation) (Vinther et al., 2004) or with the development of lists of endangered species such as the International Union for Conservation of Nature (IUCN) Red List of threatened species and the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) although the focus of these lists on marine species is generally very weak. But the inefficiency of single species approaches is recognized (Garcia and Cochrane, 2005) and ecosystem-based approaches (i.e. considering entire ecosystems and their components, such as habitats and food webs) are nowadays accepted as the most adequate to the conservation and management of marine ecosystems (Frid et al., 2005; Katsanevakis et al., 2011). Marine Protected Areas play a major role in the context of ecosystem-based management approaches (Halpern et al., 2010).

The science of Marine Protected Areas and legal framework

Marine Protected Areas (MPA) are currently considered key elements to the achievement of conservation and sustainable marine management targets since they allow an ecosystem-based approach and are able to protect habitats and biodiversity, as well as enhance fisheries stocks and benefit fisheries incomes. According to the IUCN, a MPA “is a clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values” (Dudley, 2008).

Despite their recognized importance, the science and development of MPA is relatively recent and has increased rapidly in the past three decades (Fenberg et al., 2012; Jones, 2001; Lubchenco et al., 2003). Concerns regarding the protection of marine environments only became widely accepted during the course of the 1950's and early 1960's (Kelleher and Kenchington, 1992). One of the major steps forward in this science was the "First international conference on marine parks and reserves", in 1975, where increasing pressures upon marine environments were noted and the establishment of a well monitored system of marine protected areas representative of the world's marine ecosystems were called for deal with the degradation of ocean ecosystems (IUCN, 1976). The first integrated approaches for selecting, promoting and managing MPA were also developed in this conference. However, the first IUCN guidelines for establishing MPA were only published in 1984 and were mostly dedicated to coral reefs and mangroves in developing countries (Salm and Clark, 1984). Only in 1991 were broader guidelines published by IUCN and these have been updated in 1999 (Kelleher and Kenchington, 1992; Kelleher, 1999). In 2002, The United Nations world summit on sustainable development (WSSD) and the Conference of the Parties for the Convention on Biological Diversity (CBD), set important targets related to oceans conservation, namely "the establishment of marine protected areas consistent with international law and based on scientific information, including representative networks by 2012 and time/area closures for the protection of nursery grounds and periods, proper coastal land use and watershed planning and the integration of marine and coastal areas management into key sectors" and concretely "to achieve 10% coverage of ecologically representative and effectively managed MPAs by 2012". This target was revised and updated by CBD in 2010 to "achieve at least 10% of coastal and marine areas (…) through effectively and equitably managed, ecologically representative and well connected systems of protected areas (…) by 2020". At the European level, in 2003, the Convention for the protection of the marine environment of the North-East Atlantic (OSPAR Convention) recommended the implementation of a coherent and representative network of MPA by 2010. However, this target was recently revised and actually aims to ensure that regarding the network of MPA in North-East Atlantic "by 2012 it is ecologically coherent and includes sites representative of all biogeographic regions in the OSPAR maritime area (…) and "by 2016 it is well managed (…)" (see OSPAR recommendation 2003/3 and 2010/2). The NATURA 2000 network of protected sites is a pillar of European action to halt biodiversity loss. It consists of Special

Protection Areas (SPA) and Special Areas of Conservation (SAC) selected according to the Birds and Habitats Directives respectively (EEC, 1979, 1992).The extension of Natura 2000 network to the sea was held during the 2000 decade and aimed to complete NATURA 2000 network at sea by 2008. But in what concerns marine ecosystems conservation, marine species and habitats listed in these directives are very reduced (five habitats and eighteen marine species). Finally, the European Marine Strategy Framework Directive (MSFD; EU, 2008) (MSFD) aims to achieve the "good environmental status" of member states' marine waters by 2020, through several mechanisms, namely the establishment of coherent networks of MPA.

However, despite these progresses and efforts, the coverage of MPA remains very low and far from the international targets established. According to Spalding et al. (2012), MPA registered in the World Database of Marine Protected Areas in 2012 covered over 8.3 million km^2, representing 2.3% of the global ocean area, 7.9% of continental shelf and equivalent areas (areas less than 200m deep), and 1.79% of off-shelf areas (mostly within jurisdictional waters). Despite this low number, it represents a high increase rate when compared to previous assessments in very recent years (Spalding et al., 2008; Wood et al., 2008). In addition, though all coastal realms and provinces have MPA (Spalding et al., 2007), the percentage of coverage is not equilibrated and most of them show low percentages of area covered by MPA (Spalding et al., 2012).

Marine Protected Areas: planning, implementation and management

MPA have been established for a wide range of purposes focusing processes and ecosystems functioning (Abdulla et al., 2009; Botsford et al., 2003; Gerber et al., 2003; Kelleher, 1999). The main goals of MPA include to: prevent and reverse the widespread declines in exploited marine populations, preserve and restore marine habitats, maintain ecosystem services, restore fisheries stocks, manage other economic activities, minimize conflicts among resource users and decrease poverty. Nevertheless this multiplicity of objectives is often hard to achieve simultaneously. The achievement of objectives depends on several factors, from the design characteristics of the MPA itself to the compliance by local communities. Marine Protected Areas need to be designed and managed effectively, taking into consideration the socio-economic needs of their surrounding communities (Agardy et al., 2011; Jones and Carpenter, 2009). The high degree of linkage between land and adjoining sea, and the inter-connectivity of oceans, require that MPA be integrated into management regimes that deal with all human activities that affect marine life. Thus MPA should be integrated with other policies for land use and use of the sea (Kelleher, 1999). Natural factors such as unusual climatologically events can have a stake in the success of protection measures, but MPA success is also highly dependent of the suitable conduction of all processes, from planning to implementation and management (Figure 1.1).

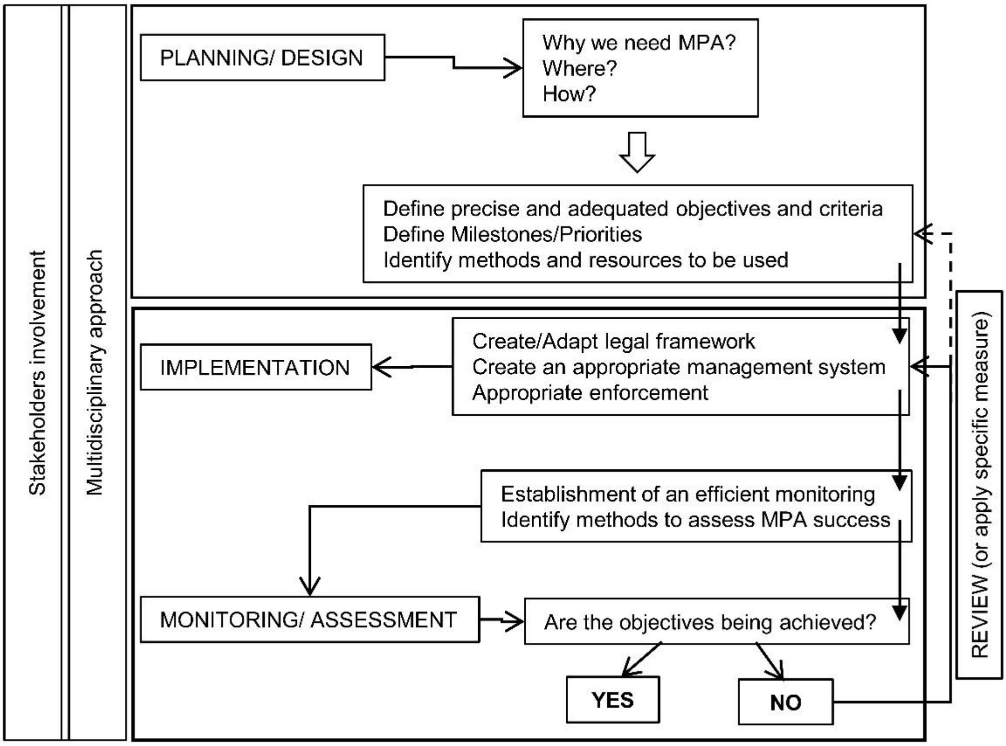

Figure 1.1. Schematic approach showing principal steps needed to obtain successful MPA

An MPA must have clearly defined objectives that take into account the overall reality of the area, namely natural values to protect, but also social and cultural feature (Halpern, 2003). Once the general goals are established, more specific goals and relative priorities should be set, whilst simultaneous accounting for the available resources to allocate for an efficient MPA planning (e.g. define zones with different levels of protection, define MPA size, and collect scientific data) (Figure 1.1). For instance, depending on its objectives, a MPA can have different levels of protection, various design and zoning strategies. MPA can vary from areas where human activities are totally excluded (no-take areas or marine reserves) to areas where several activities are allowed although under specific measures. Furthermore, an individual MPA can include several areas with different levels of protection. MPA size also depends on the aims established for a given area, for instance larger areas are usually selected as most appropriate for fisheries management goals than for biodiversity conservation (Botsford et al., 2003; Kelleher, 1999; White et al., 2010).

The process of MPA implementation should be adapted depending on the MPA characteristics defined during the planning phase. First, legal frameworks need to be created or adapted according to the conservation needs identified, management boards and strategies need to be defined and an adequate enforcement needs to be established (Figure 1.1). MPA implementation is usually a complex process because to achieve stakeholders compliance, science-based information needs to be integrated with social and economic needs. If MPA are not well implemented, future revisions of the process will be costly and time consuming (Charles and Wilson, 2009; Rees et al., 2013). Furthermore, it is well documented that the concrete integration of local communities (stakeholders, e.g. fishers) into MPA management systems is the most adequate and successful way to achieve higher MPA effectiveness (Dudley, 2008; Rodriguez-Martinez, 2008).

Finally, an adequate monitoring programme needs also to be implemented to regularly check MPA performance, i.e. if the initial objectives are being achieved, and to assess management effectiveness. Based on the monitoring results, previous steps can be revised or adapted (e.g. improve enforcement, adapt the objectives) following an adaptive management approach (Ban et al., 2012; Grafton and Kompas, 2005; Walters and Hilborn, 1978)) (Figure 1.1). The assessment of the major reasons or processes contributing for MPA success/failure is thus of the utmost importance. Monitoring plans also depend on MPA objectives and can be based on several approaches depending on the available resources (e.g. budget, technicians, time). Several monitoring/ assessment approaches rely on the local users perceptions, others are based on more complete scientific approaches (e.g. long-term sampling of fisheries captures, marine communities metrics) (Claudet et al., 2006; Pomeroy et al., 2005)

In addition, the participation of stakeholders in all these processes and the involvement of multidisciplinary teams are essential to the success of MPA implementation (Figure 1.1).

General aims and thesis outline

This thesis consists of a collection of scientific articles that aim to develop methodological approaches to cope with the common lack of adequate scientific data for MPA planning and management, and thus provide scientific knowledge that can contribute to the increase in global rates of MPA effectiveness. The thesis includes four scientific papers published, in review or submitted in peer reviewed international journals included in Science Citation Index, each corresponding to a chapter.

Chapter 2 conducts an assessment of cumulative anthropogenic pressures along the Portuguese coast, namely in MPA and their boundaries. The distribution of human activities along the coast, namely in MPA, represents baseline information to integrate with ecological data to support the efficient planning and management of MPA since it allows the recognition of areas where impacts need to be minimized and also areas where conflicts with users would be higher. The developed approach provides a standard framework to support MPA planning and management and contributes with relevant information for effective marine management and ecosystem conservation worldwide.

The effects of MPA implementation on local fisheries are assessed in chapter 3, using a fisheries monitoring method for MPA that combines spatial distribution of fishing effort, onboard observations and official landings records at appropriate scales. Given that fisheries are among the economic activities most affected by MPA and that long-term reliable data on fishing captures and fishing effort are often not available, this chapter develops a method where landings data can be calibrated by sampling data on captures and effort, aiming at its application in MPA monitoring at long time frames.

In chapter 4 the life history of fish species occurring in a MPA are used to support the construction of a framework to perform a rapid assessment of MPA effectiveness. The common lack of appropriate scientific datasets to support planning and management decisions are overcome by this framework where species life history are used to indicate areas where conservation would be more efficient and areas/habitats that should be protected to maximize MPA effectiveness. The framework is underlined by the concept of active adaptive management.

In chapter 5, an overview of MPA characteristics in SW Europe and of the factors most contributing to their effectiveness is performed. The general characteristics of MPA from Portugal, Spain and France were studied to assess the conservation state of this study area. The suitability of planning, management, monitoring, governance and enforcement of these MPA were also analysed to define patterns that contribute to higher rates of MPA effectiveness.

This thesis contributes with tools that maximize the utility of several types of information in supporting MPA implementation. Whilst these tools were not developed for this purpose, they are commonly the only available information to integrate in MPA processes, which adds to their pertinence of the approach. This thesis contributes to the best use of existent knowledge into the implementation of MPA. Final considerations on the main findings gathered from the overall work and some recommendations on the main issues to address towards achieving higher MPA effectiveness are pointed out in chapter 6.

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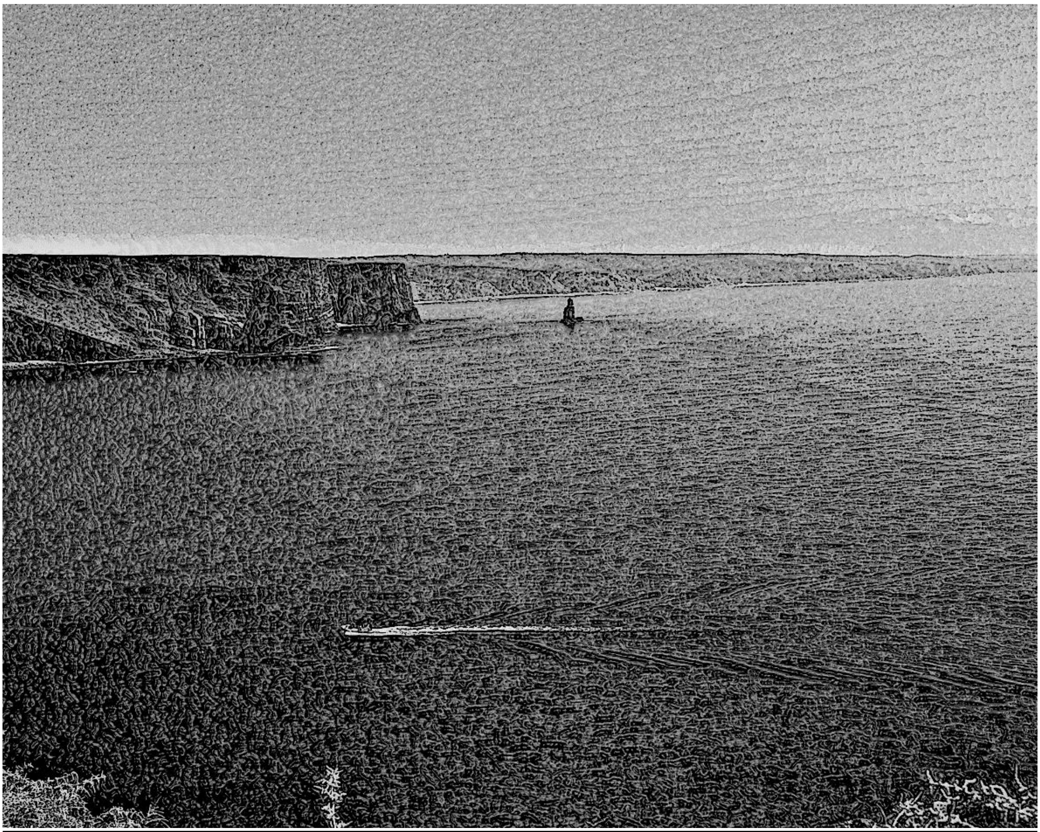
CHAPTER 2



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Assessment of cumulative human pressures on a coastal area: integrating information for MPA planning and management

Abstract

As recently reinforced in the EU Marine Strategy Framework Directive (MSFD), knowledge on the location and intensity of human impacts on marine ecosystems is critical for effective marine management and conservation. Human interaction with ecosystems has to be accounted for in order to efficiently implement marine management strategies. In the present study, the main human activities occurring along the mainland Portuguese coast were identified and mapped. The cumulative impact of these activities was calculated in order to assess impacts in different zones, namely in Marine Protected Areas (MPA) and their boundaries. Higher impact values were obtained near the coast, where all the analysed MPAs are located. Furthermore, most MPAs are sorrounded by areas with very high impacts, near the largest urban settlements and the most industrialized coastal sections. These results are the first assessment of cumulative human pressures in this study area as a whole (and with this level of resolution) and might be of great usefulness to overcome the current challenges of sustainable management in marine ecosystems. Knowledge provided by this study strengthens the need for a more integrative approach to design and manage MPAs and can be useful to support the requirements of the MSFD. The approach here developed is also a powerful tool to apply in several contexts of sustainable marine management and can be developed in any geographic area.

Keywords: Cumulative impact assessment; Marine Protected Areas; Environmental management; Geographic Information System; Marine Strategy Framework Directive

INTRODUCTION

Human activities are having a major impact on ecosystems worldwide (Baillie et al., 2004; Hails, 2008; Halpern et al., 2008; Micheli et al., 2013). While this has long been recognized in terrestrial ecosystems, concerns regarding the need to protect the marine environment only became widely accepted in the 1950's and 1960's, when strong declines in catches of various fisheries occurred worldwide (Toropova et al., 2010). Prior to this, the general idea was that marine resources were inexhaustible.

Sea pollutants are usually derived from human settlements, resource use and exploitation, agricultural activities, industrial developments, aquaculture, shipping, touristic uses, among others (Islam and Tanaka, 2004; Williams, 1996). These numerous activities and their pollutants cause severe impacts in marine ecosystems such as decreases in species diversity, population declines, degradation and destruction of natural habitats as well as changes in water chemistry and temperature (Islam and Tanaka, 2004). Even if there are no areas in the ocean unaffected by human impacts (Halpern et al., 2008), it has been shown that open oceans are in good condition when compared to coastal areas (Ban et al., 2010; Halpern et al., 2008; Kelleher, 1999; McIntyre, 1995). Open oceans receive contaminant inputs mainly from the atmosphere and sea transport, while coastal zones around the world are vulnerable to a much larger array of human activities.

The effects of human activities on the oceans have been well documented. However, in most of the cases these studies evaluate the effects of individual activities (e.g. impacts of fisheries (Batista et al., 2009; Jennings and Kaiser, 1998; Swartz et al., 2010) or aquaculture (Forchino et al., 2011; Sarà et al., 2011) in different contexts and lack an holistic perspective. As marine ecosystems are usually under the influence of multiple anthropogenic stressors, the combination and interaction of impacts from various sources over space and time (i.e. cumulative impacts) must to be considered (MacDonald, 2000). Unfortunately, interactions among multiple stressors cannot be easily modelled because they generate net impacts that either exceed (i.e. synergism) or fall below (i.e. antagonism) the addition of individual effects (Folt et al., 1999). Discussions focusing on the type of interactions and how they can be modelled are common, with all types of interactions found among natural systems and synergisms assumed as the most frequent interactions (Crain et al., 2008; Darling and Côté, 2008; Myers, 1995; Sala and Knowlton, 2006). Notwithstanding, there are authors that argue that additive impacts are the most common type of interactions that occur (Cada and Hunsaker, 1990; see also MacDonald, 2000). Apart from these discussions, additive models have been recently used in the field of marine spatial planning and are considered a valuable approach for management and conservation of marine areas (Ban and Alder, 2008; Ban et al., 2010; Coll et al., 2012; Halpern et al., 2007; Halpern et al., 2008; Halpern et al., 2009; Korpinen et al., 2012; Selkoe et al., 2009; Stelzenmüller et al., 2010b).

Developing adequate and efficient conservation measures to respond to the high impacts generally affecting marine ecosystems is imperative. In this context, marine protected areas (MPA) are increasingly viewed as an important management tool to reduce, prevent and/or reverse ongoing declines in marine biodiversity (Agardy, 1994; Pauly et al., 2002; Roberts et al., 2005) and have been pointed out as essential tools in several documents and scientific studies (e.g. EU Marine Strategy Framework Directive (MSFD); Code of Conduct for Responsible Fisheries (FAO); Spalding et al. (2008); Wood et al. (2008)). In fact, the MSFD underlines the need for MPA networks that adequately covering the diversity of the constituent Ecosystems (EU, 2008). Despite their rapid expansion in the last decade MPA still represent less than 3% of the total ocean (Abdulla et al., 2013 and references therein). Additionally, fewer than 10% of MPA are achieving their management goals (Wood et al., 2008). There are a multitude of reasons for these failures but in general we can consider that many are due to inefficient implementation and management processes (Fenberg et al., 2012; Halpern, 2003). Socioeconomic conflicts are a typical problem arising from the customary restrictions on several human activities established within MPA (e.g. fishery closures, prohibition of recreational activities), which may result in illegal behaviours and a consequent MPA inefficiency. In this sense, one of the major challenges affecting MPA success is the overlap between human activities, socioeconomic interests and natural values. The need to protect often derives from the need to minimize human activities in important marine areas in order to avoid irrecoverable ecosystems. Thus, developing and implementing adequate tools for the selection, implementation and management of MPA from the beginning is critical to their success.

Knowledge on pressure sources and impacts on ecosystems is important not only for a better understanding of the ecosystem responses to pressures but also to formulate effective prevention or management measures (Islam and Tanaka, 2004). For example, the MSFD highlights the need to undertake *a priori* analyses of the pressures and impacts on the marine waters of member states, before the implementation of programmes of measures (see EU (2008) for details).

Given that the already implemented MPAs are clearly insufficient to fulfil conservation targets, namely the requirements of the MSFD, there is a recognised need for more MPAs or the re-dimensioning of existing ones in mainland Portuguese marine waters. In this context, the present study aims to characterise cumulative anthropogenic pressures along the whole extent of the territorial marine waters of Mainland Portugal. This baseline information is of major importance to support an efficient implementation of conservation strategies including the MPA proposals. In addition to its relevance for this particular study area, the developed approach provides a standard framework to support MPA planning and management in other spatial contexts and contributes with relevant information for effective marine management and ecosystem conservation at larger spatial scales.

METHODS

Study area

The area considered in this study was the territorial sea of mainland Portugal, i.e. the area comprised between the coast line (942 km long) and twelve nautical miles offshore (Figure 2.1). The continental shelf of the west coast is relatively narrow, varying between 5 to 60 km wide, and is wider in the north coast (35 to 60 km) than in the southwest and south coasts (5 to 28 km) (Cunha, 2001; Dias, 1987; Pinheiro et al., 1996).

The western coast of Portugal is a high energy shelf environment exposed to NW swell from the North Atlantic, whereas the southern shelf sector has a lower energy regime with dominant SW–S and SE swell (Mil-Homens et al., 2007). Due to upwelling events during the summer, biological productivity along the Portuguese coast is high, particularly in the west coast (e.g. Cunha, 2001; Fiúza et al., 1982; Santos et al., 2011; Wooster et al., 1976).

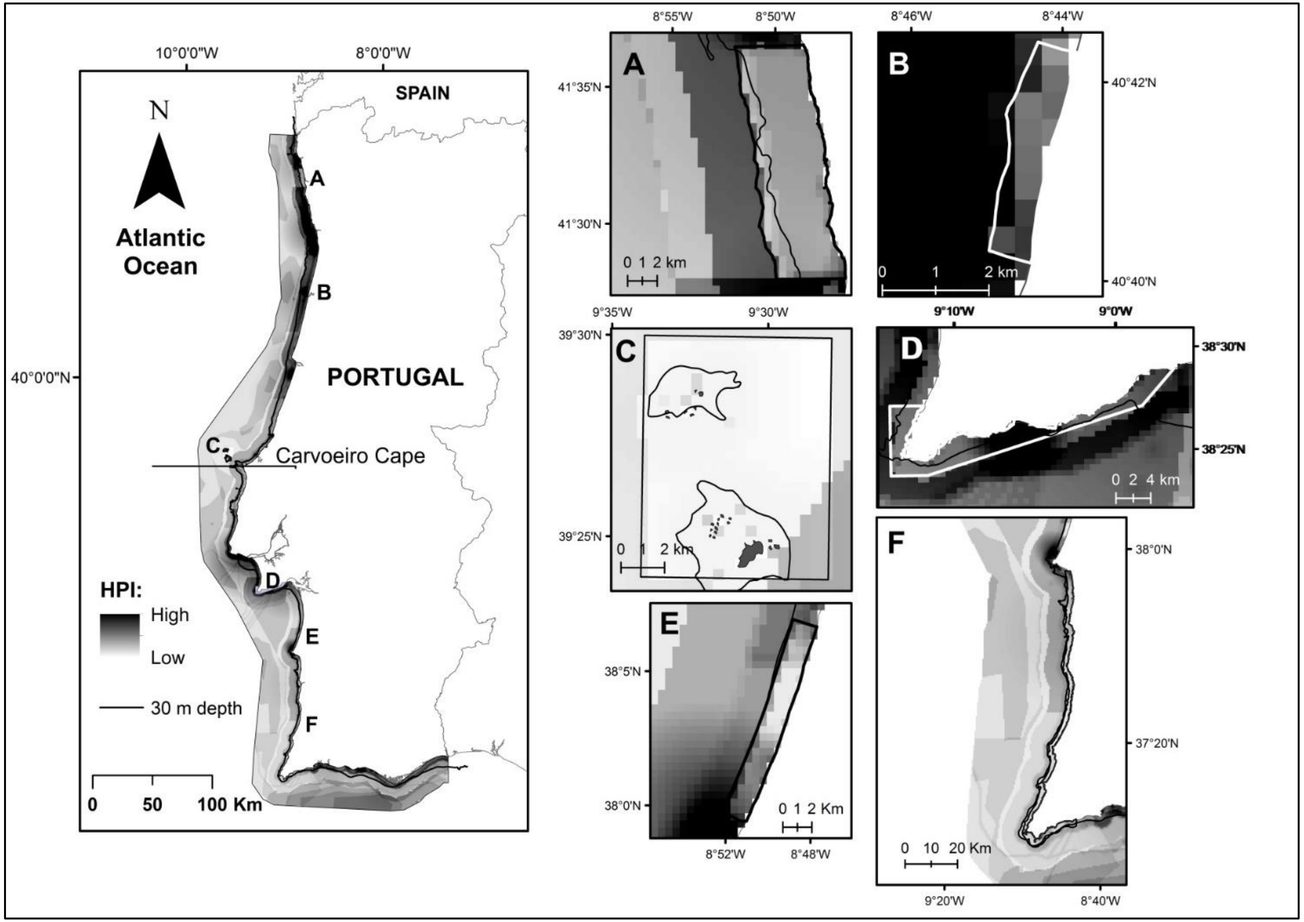

Figure 2.1. Human pressure index values obtained for the study area. The image at the left represents the global results for the study area and figures A to F show in detail the results obtained for Marine Protected Areas. A – Litoral Norte; B – São Jacinto; C - Berlengas; D – Arrábida; E - Santo André and Sancha and F- Southwest Alentejo and Costa Vicentina

In terms of sediment composition, deeper areas of the northwestern and central sectors and most of its southwestern sector are covered by fine and very fine sands. Coarse deposits are found in the inner and middle shelf of the northernmost sector and immediately south of the Nazaré and Setúbal canyons. Extensive mud patches are present in the southern shelf due to its lower energy environment, whilst in the remaining coast they are restricted to areas adjacent to major river mouths (Martins et al., 2012).

In mainland Portugal there are six MPA: Litoral Norte, São Jacinto, Berlengas, Arrábida, Santo André and Sancha (hereafter referred as S. André/Sancha) and Sudoeste Alentejano and Costa Vicentina (hereafter referred as SWACV) (Figure 2.1, Table 2.1).

Table 2.1. Marine Protected Areas (MPA) of the study area. Classification status, designation and management plan publication dates, total protected area (km^2) and main goals. Dates between parentheses correspond to alterations in MPA limits.

MPA	International designation	IUCN Category	Designation	Management plan publication	Total area (km^2)	Main goals
Litoral Norte	Nature Park	V	2005	2008	76.5	Protect marine biodiversity; Sustainable exploitation of marine resources
São Jacinto	Nature Reserve	IV	2004	2005	2.5	Protect habitats and associated fauna, flora and landscape
Berlengas	Nature Reserve World Biosphere Reserve	IV	1981 (1998)	2008	95.6	Protect marine biodiversity; Promote sustainable nature tourism and fishing with traditional selective fishing gears
Arrábida	Nature Park	V	1998	2005	52.9	Protect marine biodiversity and habitats restoration; Promote nature tourism and sustainable development of traditional fisheries
Santo André and Sancha	Nature Reserve	IV	2000 (2004)	2007	21.4	Protect marine biodiversity and improve ecological status of exploited species
Sudoeste Alentejano and Costa Vicentina	Nature Park	V	1995	2011	289.9	Protect marine biodiversity and improve ecological status of exploited species

Data collection and Human Pressure Index (HPI)

In this study, we used human activities to represent human-derived pressures or stress factors in the marine environment. Due to the complexity and lack of information to precisely classify the impacts of each pressure source in ecosystems, we considered each pressure as having a potentially negative impact on the marine environment.

The main human activities occurring along the Portuguese coast, or having direct impact on it, were identified (Table 2.2). Due to the lack of appropriate or quantifiable information, recreational fishing, illegal activities, hand harvesting, dumping and offshore ship traffic were not included in this study though their occurrence along the Portuguese coast is recognized. Expected natural pressures (e.g. ocean acidification, increase in sea temperature) were also not included in the analyses.

The areas and locations of human activities were mapped using ArcGIS 10.1 software. Each activity was mapped in an individual vector layer. All layers were combined to calculate the Human Pressure Index (HPI) in order to account for the differences of scale and features in relation with this study area. HPI calculation and overall methodology were adapted from previous work by (Halpern et al., 2008) and have been previously successfully applied by the authors in a smaller area (Henriques et al., 2014):

$$HPI = \sum_{i=1}^{n} A_i * w_i$$

where A_i is the intensity of the human pressure at location *i* and w_i is the weight attributed to that human activity at the same location *i*. *n* is the number of human activities considered (*n* = 18). An HPI value was calculated for each grid parcel (a 500 m grid size was chosen because it represents a good compromise between resolution and information availability) using the module “Environmental Risk Surface (ERS)” of the package “Protected area tools v4” for ArcGIS 10 (Schill and Raber, 2009). Created risk surfaces are therefore raster files with 500 m wide pixels, and each pixel value is the HPI obtained for that location. The range of values obtained represent the relative importance of pressures among grid parcels where the higher HPI value obtained corresponds to the higher pressure level identified in the study area.

Table 2.2. List of threat categories and sub-categories used to calculate the Human Pressure Index (HPI). Metrics, influence distance (m), impact frequency (1−4, where 1 is the lowest frequency), magnitude (1−5, where 1 is the lowest magnitude) and weight (1−3, where 1 is the lowest) for each sub-category are indicated and data sources are presented.

Threat category	Threat sub-category (shapefiles)	Metrics used for intensity categorization	Range of influence (m)	Impact Frequency	Impact magnitude	Weight	Data Sources		
							Spatial location	Intensity categorization	Range of influence
Offshore Aquaculture	Fish and shellfish aquaculture	Annual production (kg/km^2/year)	1000	4	1	2	Aquaculture companies	Aquaculture companies	Sarà et al. (2011) and references therein; Forchino et al. (2011)
Commercial Fisheries	Habitat-destructive - trawling	Hauls per zone	500	3	4	3	VMS data Legislation	VMS data (DGRM, 2000-2005)	
	Non-habitat-destructive - purse seines	Landings (kg/km2/year)	100	3	2	2	Legislation Interviews Wise (2005, 2007)	DGRM (2005-2010)	
	Multi-gear (lines, nets, traps). Local vessels (Total lenght < 9m)	Landings (kg/km2/year)	100	3	3	2	Legislation Interviews Cabral et al. (2007) Batista et al. (2009)		
	Multi-gear (lines, nets, traps). Coastal vessels (Total lenght > 9m)	Landings (kg/km2/year)	100	3	3	2	Erzini et al. (2003) Santos et al. (2003)		
Subtidal artificial reefs	Artificial reefs	Presence/ Absence	1000	4	2	2	Nautical cartography	-	Ban et al. (2010) and references therein
Dredging activities	Dredging deposition	Amount of dredged materials deposited (m^3) - from 2000 to 2010	1500	3	3	2	Reports from national ports authorities (IPTM; port administration)	Reports from national ports authorities (DGRM; port administration)	Ban et al. (2010) and references therein

Table 2.2. (Continued)

Threat category	Threat sub-category (shapefiles)	Metrics used for intensity categorization	Range of influence (m)	Impact Frequency	Impact magnitude	Weight	Data Sources		
							Spatial location	Intensity categorization	Range of influence
Commercial port activities	Ship traffic	Number of entrances and exits per year per port - 2005-2008 average	1000	4	2	2	Nautical Cartography; areas adjacent to commercial ports	National statistics institute (INE, 2005-2008)	Ban et al. (2010) and references therein
Reacreational activities	SCUBA diving	Presence/ Absence	100	3	1	1	Interviews to diving enterprises;	-	Ban et al. (2010) and references therein
	Recreational motor boating	Number of anchor places in the adjacent marinas	1000	3	1	1	Interviews to recreational associations; personal observation; national cartography	National ports authorities (DGRM , 2010)	Ban et al. (2010) and references therein
	Marinas	Number of anchor places	1000	4	2	2	Nautical cartography; marina's administration; POEM (2010)	National ports authorities (DGRM, 2010)	Estimated by authors

Table 2.2. (Continued)

Threat category	Threat sub-category (shapefiles)	Metrics used for intensity categorization	Range of influence (m)	Impact Frequency	Impact magnitude	Weight	Data Sources: Spatial location	Data Sources: Intensity categorization	Data Sources: Range of influence
Pressures from transitional waters	Larger estuaries	Population, N and P concentrations in the watershed (averaged intensity level) - 2007-2009	30000	4	3	3	National water management authorities (INAG); National Cartography	Watershed Management plans (INAG, 2007-2009)	Estimated by authors
	Small estuaries and coastal lagoons	Ecological quality - as requested for EU-WFD 2004-2006	2000	4	3	3		INAG (2007-2009)	
Direct human impact/ Population	Benthic structures (pipelines, communication structures)	Presence/ Absence	750	4	1	2	Nautical Cartography	-	Estimated by authors
	Beaches (leisure activities, e.g. swiming, kayaking, kitesurf)	Beach typologies (classification from coastal management plans) - 1998-2007	500	3	1	1	National water management authorities (INAG); National Cartography	Coastal management plans	
	Coast-line artificialization (peers, docks, harbours and other constructions)	Presence/ Absence	1000	4	1	2	National water management authorities (INAG); National Cartography	-	Ban et al. (2010) and references therein; DiGiacomo et al. (2004)
	Sewage (urban nature)	Annual sewage discards to the sea (m^3/ year)	5000	4	3	3	National water management authorities (INAG); sewage management entities	INAG (2005-2009)	Ban et al. (2010) and references therein
Industry	Industrial infrastructures (I) and thermal plants (TP)	Annual discards to the sea (m^3/ year)	7500	4	4 - I 2 - TP	3 - I 2 - TP	National water management authorities (INAG);	INAG (2005-2009)	Estimated by authors

For HPI calculation, values of (1) Intensity (relative intensity among spatial distribution), (2) Weight (relative importance of an activity in different locations) and (3) Influence distance (one value per layer, equal for all locations) were assigned to each human activity considered and added to the layer attribute tables. (1) Intensity was calculated based in measurable parameters among different locations. The best data available for the period 2000−2010 was used to fulfil this purpose and a five-level scale was used to assign an intensity class to each location (e.g. see Cabral et al. (2007), Wise et al. (2005); (2007), Erzini et al. (2003), Santos et al. (2003) for commercial fisheries; but see table 2.2 for details). For some activities (e.g. SCUBA diving), for which an intensity gradient did not make sense, or when appropriate data were not available, an intermediate intensity level was applied to all locations. (2) The Weight parameter reflects the relative importance of an activity in different locations and was obtained by averaging the values for *frequency* and *magnitude* attributed to each of the activities. *Frequency* values were assigned following a scale from 1 to 4 (adapted from Halpern et al. (2007)*;* see also table 2.3) and *magnitude* values were attributed by expert judgment, following a scale from 1 to 5 (see table 2.3 for details). Finally, obtained weight values were standardized in a 1−3 scale (1 being the smallest weight). Overall, this parameter reflects the relative "potential for environmental damage" among the various activities. (3) For each activity, an influence distance was assigned based on the fact that each human activity exerts an influence over the areas adjacent to their main sources of impact. Influence distances were based on a literature review and, in some cases, these ranges were estimated by the authors (based on the compilation of available information, informal expert consultation and personal experience; see Table 2.2 for further details). A linear decay function was utilized in this parameter in order to simulate the decrease in intensity with increasing distance from the source.

Data analyses

Average HPI values were calculated for the whole study area, for each Marine Protected Area (MPA) and by habitat type. In this study, due to the limitations of the available data, habitat type was defined as the combination of sediment type (mud, sand and gravel and rock), depth range (<30 m; >30 m) and latitude (North and South of the Carvoeiro cape, due to proven biogeographic differences between these regions – see figure 2.1 and Cunha (2001)). Sediment information was available for 90,1% of the study area. HPI means and standard deviations for each zone (i.e. MPA and habitat types) were calculated (using the *zonal statistics tool* of *spatial analyst extension* of ArcGis 10.1 software, ESRI) and compared in order to understand how human activities affect different areas. The relative importance of each individual pressure was also analysed. All geographic data were obtained from official cartography of the Portuguese Hydrographic Institute.

Finally, the 18 individual pressures considered were analysed through a Principal Component Analysis (PCA), using Statistica 11 software. PCA aimed to identify patterns and relationships between human pressures and the different areas considered (MPA and habitat type).

Table 2.3. Classification scales and criteria for frequency (1−4, where 1 is the lowest frequency) and magnitude (1−5, where 1 is the lowest expected impact magnitude) to human activity threat sub-categories.

Parameters	Level	Description
Frequency	1	Rare - occur less than once a year, but affects nature.
	2	Occasional - Frequent but irregular.
	3	Annual or regular - Frequent and regular or seasonal.
	4	Persistent - More or less constant year-round, lasting through multiple years or decades.
Magnitude	1	Weak impact, affects only one ecosystem component.
	2	Weak impact, affects one ecosystem component; medium impact and affects few ecosystem components.
	3	Weak impact, affects several ecosystem components; medium impact, affects one ecosystem components; high impact, affects few ecosystem components.
	4	Medium impact, affects several ecosystem components; high impact, affects few ecosystem components.
	5	High impact, affects several ecosystem components.

RESULTS

Mapping human activities along the study area (Figure 2.1) revealed that, although there are differences in intensity and number of overlapping activities, there is no single area free of human pressure. The maximum HPI value, i.e. the maximum level of pressure obtained, was 73 though the average for the entire study area was much lower (9.18).

HPI achieved much higher values in areas near the coast (usually shallower than 30m), as shown in Figure 2.2. In general, northern areas had higher HPI values, deeper areas had similar HPI in average and there were no relevant differences among sediment types. In shallower areas, the average HPI was slightly lower on rocky substrates, followed by sand and gravel. Higher HPI values were found for shallower muddy areas, mainly to the north of Carvoeiro cape. Average HPI calculated inside MPA was higher than the one obtained for the whole study area or to areas deeper than 30 m. However, when compared to shallower zones, HPI values were generally lower in MPA (Figure 2.2). Nevertheless, it is clear that most MPA are sorrounded by areas with very high HPI, as shown in Figure 2.1, namelly areas near the largest urban and industrial settlements (Porto, Aveiro, Figueira da Foz, Lisboa, Setúbal and Sines).

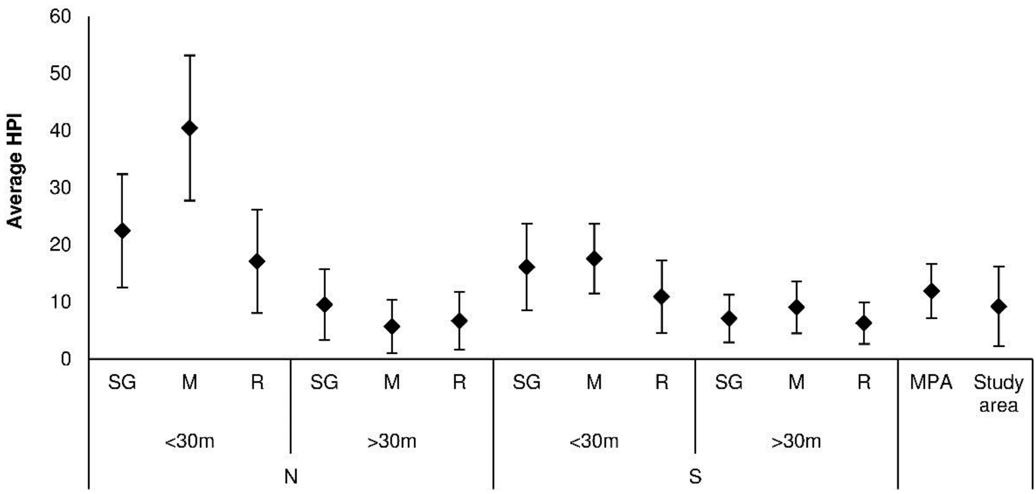

Figure 2.2. Mean Human Pressure Index (HPI) values and standard deviations obtained for each zone. SG – sandy and gravel sediments; M – muddy sediments; R – rocky sediments; N – Areas to the north of Carvoeiro Cape; S - Areas to the south of Carvoeiro Cape; < 30 m – areas shallower than 30 m; > 30 m – areas deeper than 30 m.

An analysis of the contribution of individual activities to the global (cumulative) HPI values showed that fisheries accounted for 74% of the global HPI (32% for bottom trawl, 25% for multigear fisheries and 17% for seine fisheries) and pressures from transitional waters contributed to 18% of the global HPI. Among the remaining considered activities, industrial loads and sewage discards were the most important (8% of the global HPI).

The assessment of human pressures in the different MPA (Figure 2.3) showed that higher HPI values were obtained for São Jacinto (mean HPI = 23) and Arrábida (mean HPI = 20), followed by S. André/ Sancha (mean HPI = 11), Litoral Norte (mean HPI = 11) and SWACV (mean HPI = 6) (see Figure 2.1 for locations). Berlengas (mean HPI = 1) obtained the lowest HPI value among MPA. The values obtained for MPA follow the tendencies observed in areas shallower than 30 m, where HPI was higher near larger urban and/or industrial areas.

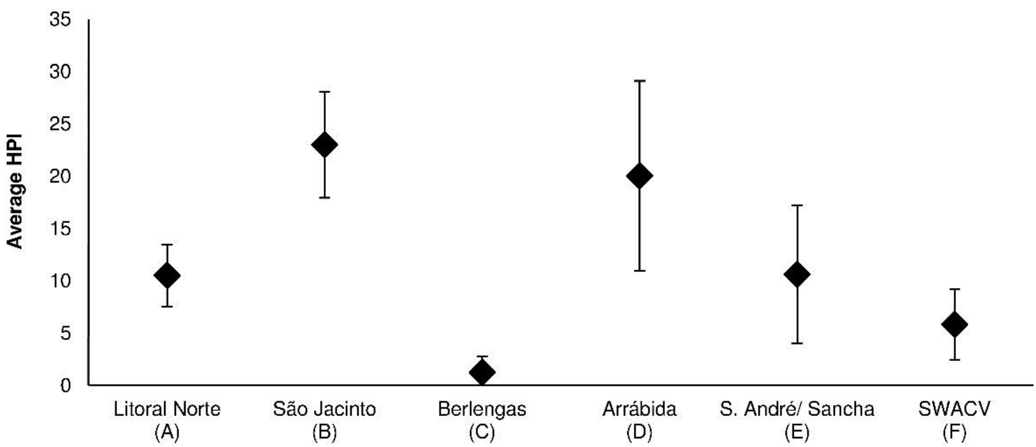

Figure 2.3. Mean Human Pressure Index (HPI) values and standard deviations obtained for each Marine Protected Area (for MPA identification see figure 2.1).

The relative importance of individual pressures affecting MPA are shown in figure 2.4. Activities with major importance for the final HPI within MPA are similar to the ones affecting the study area as a whole, although with differences in their relative importance. Within MPA, the contribution of fisheries did not exceed 50% of the HPI, and these were mostly multi-gear fisheries (albeit purse seine also occurred). The remaining contribution came mainly through inputs from transitional waters. In Litoral Norte, transitional waters represented 55% of the global HPI in this MPA. Sewage loads also had an important contribution to the HPI of most MPA, while industrial discards were relevant in the southern MPA. In Berlengas the HPI was very low and the contribution of tourism activities the most representative pressure source, with a relative importance of 55% to the its HPI. Fisheries and sewage also had a role in the obtained HPI value for the Berlengas MPA.

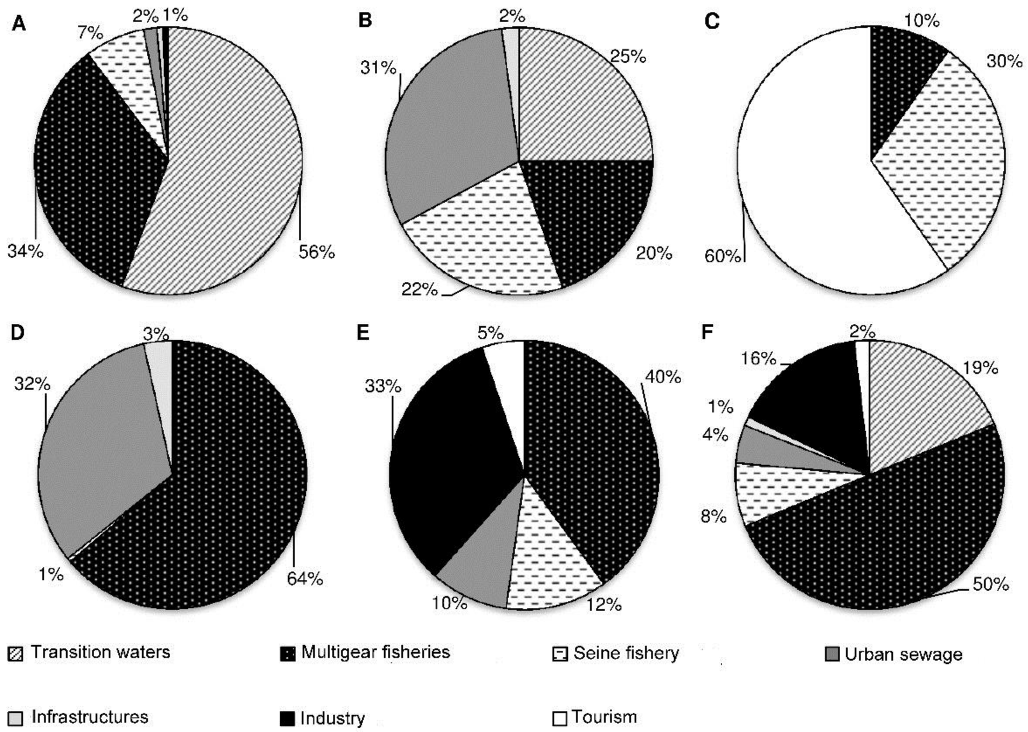

Figure 2.4. Contribution of individual groups of human activities to the final HPI value within MPA. **A** – Litoral Norte**; B** – São Jacinto**; C** – Berlengas**; D** – Arrábida**; E** – S. André/ Sancha**; F** – *SWACV*.

The PCA complemented and supported observed patterns in HPI, with 53.42% of the total variance explained by the first two principal components (Figures 2.5a and 2.5b). This analysis showed that shallower zones (< 30 m deep) were clearly separated from the deeper areas. Most of the MPA were clustered together in between these two groups with the exception of Arrábida (A). The cumulative effects of diving, beach activities, sewage and pipelines seem to influence Arrábida in a particular way, which differentiated it from the other zones (Figure 2.5a and 2.5b). Local fishing and coastline artificiality were also important in this MPA. Trawl fishing has an important role in the observed HPI in areas deeper than 30 m. The PCA diagram also showed that most of the human pressures considered seems to be of particular importance in shallower zones.

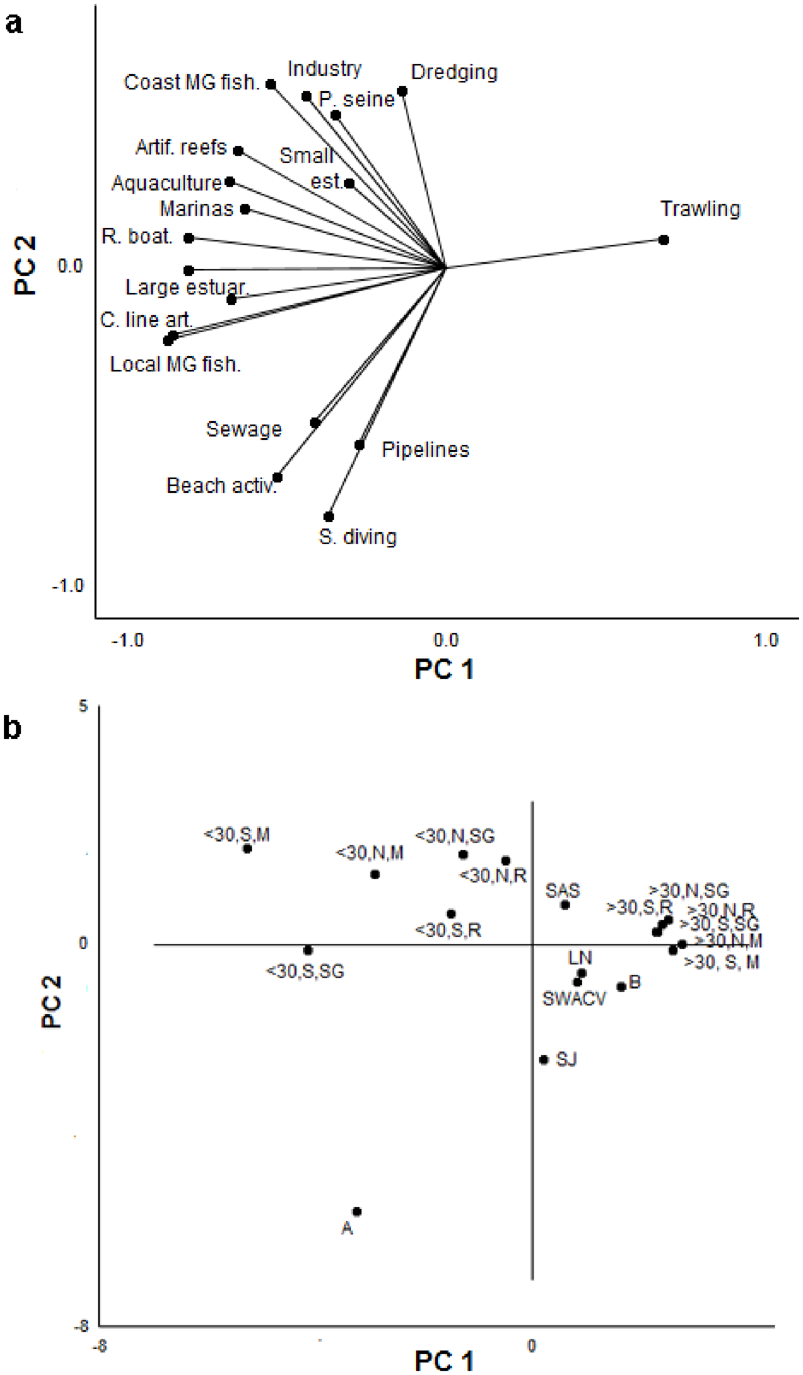

Figure 2.5. Principal components analysis (PCA), based on the assessment of human activities in habitats and marine protected areas (MPA). a) Relationship between the intensity of human activities and the first two principal components (activities are listed in table 2.2) . b) Habitats (abbreviations are explained in figure 2.3) and MPA groups obtained in the PCA analysis. LN- Litoral Norte; SJ – São Jacinto; B – Berlengas; A – Arrábida; SAS – Santo André and Sancha; SWACV – Sudoeste Alentejano and Costa Vicentina.

DISCUSSION

Recognizing the magnitude of impacts acting upon the oceans has led to a general increase in awareness towards the need to protect marine ecosystems and ensure the sustainable use of resources (e.g. CBD, 1999; EU, 2008; Halpern et al., 2008; OSPAR, 2010-3). Although this problem has been scarcely addressed in most countries, several efforts to implement appropriate legislation are being made (e.g. EU, 2008; EU, 2013). However, to accomplish the aims of these documents there is a crucial need to obtain more and/or better information, namely on human impacts spatial distribution and their effects on marine ecosystems (Benn et al., 2010; Henriques et al., 2013). The better the knowledge base and the quality and representativeness of data available to scientists and managers, the more effective marine management will be (Pais et al., 2012). There is a growing need to base marine management on a wide range of factors and also consider their potential interactions (e.g. spatial distribution of human activities, ecosystems overlapped by those activities, ecosystem resilience, predictable influence of natural pressures, socioeconomic issues, species life cycles and connectivity data) (Ban et al., 2010; Christie et al., 2003; Kelleher, 1999; Pomeroy et al., 2005; Reis-Santos et al., 2013; Stelzenmüller et al., 2008; Tanner et al., 2013). In this sense, the present results provide new basal information on spatial distribution of human activities in the study area and their overlap with the established MPA; the integration of the potential effects of the overlapped human activities via the HPI calculation; and the assessment of cumulative pressures affecting each habitat (including inside and around MPA). The obtained results can also be integrated with accurate habitats data as well as species distributions (not assessed in this study) and thus directly contributing to marine management, namely MPA planning and management. This integration can be performed, for example, trough site selection software such as Marxan (Ball and Possingham, 2000) allowing managers to define conservation scenarios that take into consideration the consequences of human exclusion from a given area, and choose those for which the balance between overall negative impacts to economic activities (e.g. commercial fisheries, recreational activities) and overall conservation benefits of their exclusion from a given area is best. This way, the level of compliance with protection areas is expected to be higher and contributing to their higher efficiency. In addition, the identification of areas where levels of impacts are higher than those deemed reasonable to maintain ecosystems integrity (namely in implemented MPA) allows a faster implementation or enhancement of conservation measures. Thus, present results are a valuable contribution to the ongoing processes on marine management and MPA implementation in Portugal (e.g. EU, 2008 National Strategy for the Portuguese Marine Environment). The main outputs (pressure maps) from the present approach are quickly understood by managers and decision-makers, their relevance increased by integrating ecological data, and can easily be recalculated if more data is available. Furthermore, the developed tool can easily be adapted to new geographic or marine management contexts.

Although the previous global approach by Halpern et al. (2008) included this study area, the present study provides the quantification of the extent and intensity of human activities at a finer scale. This higher resolution can better contribute to overcome current challenges of sustainable management in marine ecosystems (Coll et al., 2012), particularly in respect to MPA implementation, which usually requires the analyses at smaller scales. For example, information on the relative importance of adjacent areas to small scale fisheries are of major importance since the location of a no-take area can vary by only a few kilometres while greatly minimizing the economic impacts on fisheries.

Approaches like the one presented here have many advantages but also some constraints that have to be taken into account when interpreting the results. One of the main issues is the fact that the method only considers additive effects, though synergistic and antagonistic processes are known to sometimes occur. Furthermore, the method does not take into account historical impacts on marine ecosystems, as well as the ecosystems' health prior to the present state. Additionally, due to the lack of available data or their low resolution, some potential impact sources (namely, recreational fishing, illegal activities, hand harvesting, dumping and offshore ship traffic) were not included in this model, and data on habitats had a low level of detail. Among the excluded activities, recreational fishing, hand harvesting and illegal activities related to fisheries should mostly impact areas where artisanal fisheries were more intense since these areas are expected to be more productive and more easily accessible (e.g. due to favourable sea conditions). Therefore, accounting for those activities would probably increase the HPI of shallower areas for which higher HPI were already obtained in the present assessment. Dumping and offshore ship traffic are harder to predict without data since these activities can occur along the study area with irregular patterns, and the intensity of impacts is highly dependent on the type of substances dumped or spilled. However, it is expected that these activities achieve higher magnitude in offshore areas, due to increased traffic control and more efficient enforcement near shore.

In addition, it is important to stress that the results obtained for the HPI are relative to the study area, which means that the highest HPI value obtained can still be under a lower level of impact when compared to other geographical areas. Despite these constrains, the presented approach is a powerful tool to estimate cumulative human impacts in a given area. The method is appropriate to assess the relative importance of human uses among different zones within a study area as the accuracy and spatial resolution of the data used for calculating the HPI is the same.

As a result of the low level of spatial resolution data for habitat characterization, habitat sensitivity was left out of the index calculation (as firstly defined by Halpern et al. (2008). However, the weight factor (see section 2 for details) represents a classification according to frequency and magnitude of potential impacts which is an adequate approach to consider when the quality of data available is poor.

The results obtained in the present assessment showed that nearshore areas were under higher cumulative impact scores when compared to offshore zones. Generally, nearshore zones are under increased direct pressures (e.g. fisheries, shipping activities) and also several indirect land-based activities (e.g. urban sewage, agriculture-related nutrient input). This general conclusion was observed in many studies (e.g. Ban et al., 2010; Halpern et al., 2008; Stelzenmüller et al., 2010a). Although areas farther from shore have lower scores in general, pressures in offshore areas have high destructive potential (e.g. high impact fisheries, intensive maritime traffic, spilling of hazardous substances) which, in combination with the poor data available, can be of great concern for governments and scientists (IUCN, 2004; Korpinen et al., 2012) since HPI values would be higher than the estimated. The unknown magnitude of illegal activities can also represent a risk that is scarcely assessed (e.g. Ainsworth and Pitcher, 2005). Furthermore, some of these offshore areas can encompass valuable and sensible ecosystems that may be suffering irreversible impacts (Korpinen et al., 2012). In the present study, the deepest areas were mainly affected by bottom trawl fisheries which uses highly destructive gear and generally implies long recovery periods, when recovery is even achievable (Hiddink et al., 2006; Lambert et al., 2011).

Similar to findings of other authors worldwide, relatively high HPI were obtained for some MPA (Ban et al., 2010; Coll et al., 2012). In the study area, these relatively high values were not entirely surprising since all MPA are extensions of terrestrial protected areas (i.e. generally highly influenced by land-based activities), and most of them are located near estuaries and highly populated urban areas (except Berlengas which is a continental island). In fact, although particular activities such as fishing have less impact within the MPA than in the surrounding areas, other pressures, mostly ones related with land based activities and human settlements, are not regulated within most MPA (Mora and Sale, 2011). Furthermore, a MPA can undergo increasing pressure from some activities, such as scuba diving, leisure boating and tourism. Even though they have lower impacts on ecosystems, it is important to consider carrying capacity issues and establish measures that limit this occupation (Needham et al., 2011) and minimize stakeholder conflicts (Ballantine and Langlois, 2008; Scholz et al., 2004). In the authors' view, scores obtained within MPA have to be seen with caution since some of the excluded activities (e.g. recreational fisheries, illegal practices), can highly increase the estimated HPI values. In addition, the high HPI values found in surrounding areas should be seen with concern, as they are likely to affect the fulfilment of MPA goals (i.e. high human pressures in surrounding areas can minimize the expected effects of protection).

All studied MPA contain undeniable ecological value however, there was a general lack of research to establish their adequate locations, dimensions and goals. A national strategy to protect ecosystem functions, habitat integrity and the survival of vulnerable or commercially important species (e.g. Botsford et al., 2003; Sala et al., 2002) would have been a much more appropriate approach, rather than an out-dated approach of scattered individual areas and isolated goals. Coherent MPA networks (e.g. including both offshore and coastal areas in order to enclose inter-dependent ecosystems), is needed for a more efficient marine management.

Regarding conservation planning efficiency and MPA establishment, there are some principles that should be generally followed. Primarily, a conservation strategy with clear goals should be defined, along with precise and adequate criteria. This must be followed by the estimation of the resources needed, which have to be adjusted in face of the existing financial constraints and human resources. It is important to keep in mind that the simultaneous consideration and integration of biodiversity conservation targets, the distribution of sources of impact, and the attitudes and perspectives of local stakeholders is imperative for the success of MPA processes (Fraschetti et al., 2009). Although there is a large number of publications focusing on efficient MPA planning (e.g. Botsford et al., 2003; Kelleher, 1999; Sala et al., 2002), recent studies reveal a worrying number of inefficient MPA (Pomeroy et al., 2005; Wood et al., 2008). When the spatial extent and interaction among stressors and their influence on ecosystems is poorly addressed, it is important to adopt a precautionary management approach, in order to ensure an efficient protection of ecosystems and, consequently, the sustainability of key services they provide. Some precautionary approaches are the protection of representative habitats, allocation of larger areas than the apparently needed (e.g. spatial / temporal closures) and improvement on the knowledge regarding spatial distribution, intensity and frequency of human activities.

The present study underlines the importance of implementing efficient approaches to marine management, a task in which MPA are powerful tools. The presented method contributes to the improvement of planning and management of MPAs, since it allows for a high resolution, spatially explicit characterization of human activities affecting marine resources. This approach can and should be extended to other areas and applied to other management contexts, such as the MSFD. In fact, this information is extremely valuable to adequately design programmes of measures targeting the activities responsible for environmental degradation, on the road to the achievement of a “good environmental status” for European marine waters.

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CHAPTER 3

Batista MI, Horta e Costa B, Gonçalves L, Henriques M, Erzini K, Caselle JE, Gonçalves EJ, Cabral HN. Assessment of catches, landings and fishing effort as useful tools for MPA management. In review in *Fisheries Research*.

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

ABSTRACT

Marine protected areas (MPAs) have been widely suggested as a tool to achieve both fisheries management and conservation goals. These multiple objectives are hard to achieve simultaneously due to potential conflicts between conservation and economic objectives. MPA implementation often includes some type of restrictive measures to fisheries (e.g. no-take areas, vessel size restrictions, gear exclusion) that in the short-term may have negative effects on local fishers' communities (particularly when implemented in the absence of other policies to make the burden less on fishermen). Thus, monitoring fisheries catches before, during and after MPA implementation is essential to document changes in fisheries activities and to evaluate the impact of MPAs in fishers' communities. Remarkably, compared to standard fisheries independent biological surveys, these values are rarely measured at appropriate spatial scales following MPA implementation. Here, the effects of MPA implementation on local fisheries is assessed at Arrábida Marine Park (Portugal), using a fisheries monitoring method that combines spatial distribution of fishing effort, onboard observations and official landings records at appropriate scales. Fisheries spatial distribution, fishing effort, onboard data collection and official landings registered for the same vessels were analysed between 2004 and 2010. We tested the applicability and reliability of using landings statistics alone (i.e. when no sampling data are available) and found that this data only allows the identification of general patterns. However, when landings information is known to be unreliable, the combination of the additional data sources proved to be an effective method to evaluate fisheries dynamics in response to MPA implementation. Additionally, as resources for monitoring socio-ecological responses to MPAs are frequently scarce, the suitability of using landings data calibrated with information from both vessels and gear distribution and onboard validation of effort is explored.

Keywords: Cumulative impact assessment; Marine Protected Areas; Environmental management; Geographic Information System; Marine Strategy Framework Directive

INTRODUCTION

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

Ecosystem-Based Management (EBM) is recognized as the most appropriate approach to maintain ecosystem structure and function (Claudet, 2011; Frid et al., 2005; Jennings and Kaiser, 1998; Murawski, 2000). However, its implementation is challenging, since ecosystems are complex and dynamic, often requiring an enormous amount of data and multidisciplinary approaches to assess them which, most of the times, is incompatible with human and economic resources available for management (Claudet, 2011). In this context, Marine Protected Areas (MPAs) became a mainstream management tool for a variety of management problems, with goals such as the conservation of exploited stocks, preservation of biodiversity and enhancement of fisheries yields and some societal goals (Costanza et al., 1998; Field et al., 2006; Gerber et al., 2003; Halpern, 2003; Murawski, 2007; Roberts et al., 2001). MPAs have been found to contribute to the protection of essential habitats (e.g. spawning and nursery grounds) (Gell and Roberts, 2003; Nagelkerken et al., 2012), the reduction of fishing mortality (Grüss et al., 2011; Mesnildrey et al., 2013), benefits for local fisheries by contributing to restoration of fish stocks and improvement of catches in adjacent areas through spillover effects (Goñi et al., 2008; Goñi et al., 2010; Halpern et al., 2010; Harmelin-Vivien et al., 2008; Roberts et al., 2001) and through larval export (Harrison et al., 2012).

MPAs are often located in coastal areas where human activities (both extractive and non-extractive) are intense and ecosystem functions and services are thus under risk. Therefore, to accomplish conservation goals, MPAs usually include some type of protection measures, namely to fisheries (e.g. no-take areas, gear exclusion, catch and/or effort control), that at least in the short-term may have negative effects on local fishing communities, particularly when not implemented together with other policies to make the burden less on fishermen (Batista et al., 2011; Lester and Halpern, 2008; Mascia et al., 2010; Rees et al., 2013). Hence, MPA implementation is frequently accompanied by controversies and fishers' opposition (Christie, 2004; Rice et al., 2012) which can put at risk the MPAs success at achieving their goals (Pollnac et al., 2001). Compliance by users, including fishers, their acceptance of MPA rules, their cooperation in monitoring and management processes and adequate enforcement are critical issues to MPA success (Claudet and Guidetti, 2010; Guidetti et al., 2008). When MPA benefits in well-designed MPAs are recognized by fishers, they are more likely to accept protection measures, and MPAs are more likely to 'succeed' (McClanahan et al., 2005; McClanahan et al., 2014). However, measurement of 'success' can be challenging due to the difficulty in establishing adequate monitoring that accounts for temporal and spatial variability in the variety of possible responses. Measuring success is further compromised by a widespread lack of data from the period before MPA establishment (Horta e Costa et al., 2013c). Monitoring fisheries catches before, during and after MPA implementation is essential not only to understand the ecological responses of marine populations but also to evaluate the socio-economic impacts an MPA might have on fisher's communities. However, programs to monitor socio-economic responses to MPAs are rare compared with those directed to ecological responses, despite their recognized interdependence. In fact, monitoring fisheries catches is costly to implement and to maintain during long-time frames, resulting in a general lack of scientific knowledge on MPA impacts (both positive and negative) on fisheries (Mesnildrey et al., 2013). However, avoiding the deterioration of socio-economic conditions is critical for MPA success since they will possibly promote illegal behaviours and contribute to MPA inefficiency. Furthermore, adequate use of monitoring information may contribute to the efficient adaptation of management measures in order to promote the fulfilment of MPA goals, including the minimization of social and economic negative impacts and the improvement of benefits to local fishers' communities. The implementation of volunteer monitoring projects where fishers actively participate are became common around the world and can play a role in achieving higher rates of MPA acceptance and success (Danielsen et al., 2005; Lloret et al., 2012)

The aim of the present study was to design, implement and test the efficiency of a fisheries monitoring method at a temperate MPA (Arrábida Marine Park (AMP), Portugal), that combines measures of fine-scale spatial distribution of fishing effort, onboard sampling and official landings. This approach was used to evaluate the impact of protection measures on local fisheries, by assessing how landings and revenues changed with MPA implementation. The applicability and reliability of landings statistics alone (i.e. when no sampling data are available) was also assessed. Our approach is one of the few case studies showing how artisanal fisheries respond to MPAs and is a useful tool for managing local fisheries in response to MPA implementation. The methods explored here can be applied to most coastal MPAs worldwide and also be adapted for other types of coastal monitoring and management programs.

METHODS

Study area

The present study was conducted in a coastal multiple-use MPA, the Arrábida Marine Park (AMP), adjacent to a terrestrial nature park (Figure 3.1). The marine park extends along a 38 km stretch of coastline (53 km^2). Most of the park faces south being protected from the prevailing north and northwest winds and waves, allowing human activities throughout the year. The nearshore subtidal habitats are dominated by complex rocky reefs resulting from the erosion of coastal calcareous cliffs. 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. These features make this area an important hotspot of biodiversity, with more than 1100 species identified (Henriques et al., 1999; Horta e Costa et al., 2013a). In the middle of the park there is a small fishing town, Sesimbra, which has a long fishing tradition and is presently an important tourism area as well.

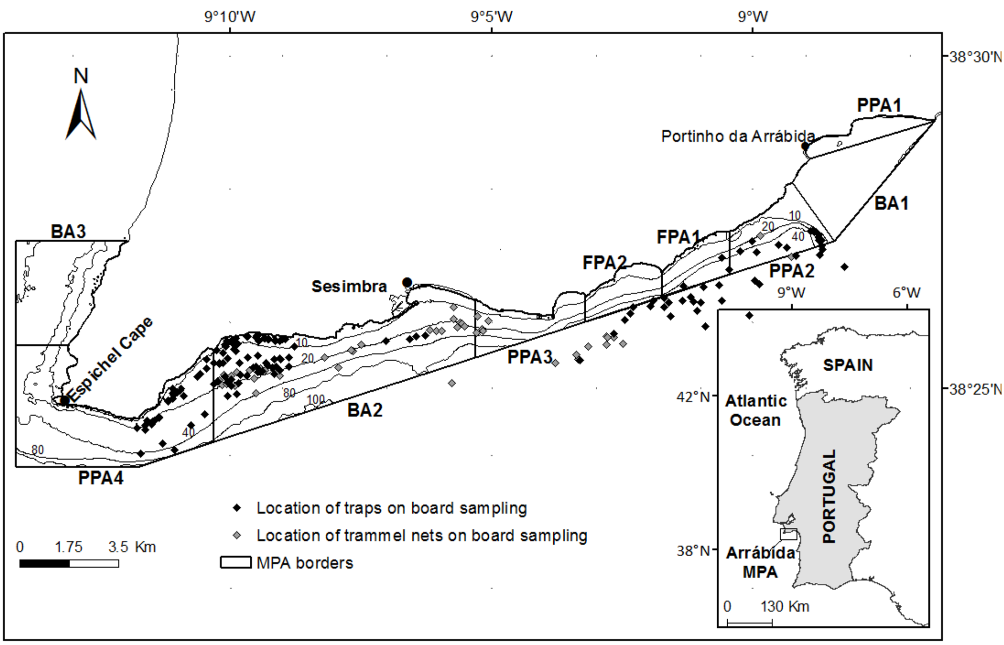

Figure 3.1. Map of the Arrábida Marine Park (AMP) with location, bathymetry lines and zoning implemented by the management plan. Zoning: PPA – Partially-protected area; BA - Buffer area; FPA – Fully-protected area. Location of the onboard surveys for traps (black squares) and trammel nets (grey squares) are also shown.

Although the AMP was created in 1998, the management plan was only approved in 2005. The plan defined eight zones subjected to three levels of protection: a fully-protected area (FPA: 4 km2), four partially-protected areas (PPAs: 21 km2) and three buffer areas (BAs: 28 km2). The AMP objectives have a wide scope: preserve marine biodiversity, recover habitats, promote scientific research, environmental awareness and education, promote nature oriented tourism and sustainable development, and promote economic and cultural regional activities, such as the traditional “hook and line” (i.e. small-scale handlines, longlines, jigs) fishery. The Arrábida MPA management plan defines limits and protection measures to various activities, namely to protect the local small-scale fisheries which have a high socio-economic importance in the area. Thus, trawling, dredging, purse-seining, discards and spearfishing are forbidden in the entire MPA and only vessels smaller than 7 meters are allowed to operate in the park. Furthermore, fishing licenses to commercial fisheries are only allotted to vessels from the fishing port inside the park (Sesimbra) and are renewed annually only if active. In partially-protected areas, only traps and jigs are allowed and only beyond 200m offshore. In the fully-protected area, human presence is generally not allowed. Since this is a traditional fishing region, the different protection measures were implemented sequentially during a four year transition period, with all the BAs, the PPA1 and half of the FPA (with PPA regulations) established in mid-2006, and all the PPAs and the second half of the FPA (with PPAs regulations) in mid-2007. The first half of the FPA started with FPA regulations in mid-2008 and full implementation of the management plan was achieved in mid-2009 with the second half of the FPA (see Horta e Costa et al., 2013a for details).

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

Data collection

Fishing effort assessment

The buoys of the static gears (traps and nets) were surveyed and identified within the marine park limits in all areas south of the Espichel cape, through zigzag transects by boat (Horta e Costa et al., 2013a; Horta e Costa et al., 2013c). During each sampling day, the entire study area was sampled and the location of all buoys was marked using a Global Positioning System – GPS. Fishing gear type and vessels' names were recorded for all fishing buoys surveyed (Portuguese legislation requires that fishing buoys at sea are identified with a code for fishing gear type and vessel identification).

Sampling was carried out in five different periods corresponding to the 'before', 'implementation' (three periods: year 1, 2 and 3) and 'after' phases of the management plan. Each of the five distinct periods had different protection measures (Table 3.1).

We focussed on traps and trammel nets fisheries for analyses since they were identified as the most important and abundant in the study area (along with jigs), and due to their static features, allowed us to accurately assess their position by buoys surveys.

Table 3.1. Fishing effort sampling phases and periods, number of samples collected and indication of protection zones implemented by period (cells in grey).

Sampling phases	Sampling periods	Number of samples	MPA zones implemented				
			FPA1	FPA2	PPA1	PPA 2,3,4	BAs
Before	Apr - Nov '04	7					
Implementation	Year 1 (Mar – Aug '07)	15	a				
	Year 2 (Sep '07 – Feb '08)	14	a	a			
	Year 3 (Nov '08 – Aug '09)	16		a			
After	Sep to Dec '09	6					

a - Implemented with PPA regulations

Official landings data

Fisheries landing data were obtained from the Portuguese "Direção Geral de Recursos Naturais, Segurança e Serviços Marítimos". Requested datasets had daily landings (biomass) per species, for all vessels fishing in the MPA for the years 2004 to 2010. This database also included average price per kilogram and vessels characteristics (age, length, HP, fishing licenses). Vessels accounting for 97% of total buoys (identified through the surveys performed for the fishing effort assessment) were selected as the group continuously fishing in the MPA in each period. Although these vessels also fished in the boundary areas, the majority of their catches were most probably from within MPA limits and thus it was considered that their landings were a good proxy for landings from the MPA. This group of vessels was the group considered in all the analyses of landings. Furthermore, interviews of local fishers were performed in order to clarify the interpretation of landings data.

Five species were excluded from the analysis due to known inconsistencies in the landings records (bogue *Boops boops* (Linnaeus, 1758), green crab, *Carcinus maenas* (Linnaeus, 1758), Atlantic horse mackerel *Trachurus trachurus* (Linnaeus, 1758), blue jack mackerel *Trachurus picturatus* (Bowdich, 1825) and Atlantic chub mackerel *Scomber colias* Gmelin, 1789). In addition, some species in the landings data appear grouped. So, *Solea* spp. corresponds to *S. solea* and *S. senegalensis*, and *Raja* spp. to undulate ray, *Raja undulata* Lacepède, 1802, blond ray, *Raja brachyura* Lafont, 1871, thornback ray*, Raja clavata* Linnaeus, 1758, spotted ray, *Raja montagui* Fowler, 1910, sandy ray, *Leucoraja circularis* (Couch, 1838), cuckoo ray, *Leurcoraja naevus* (Müller and Henle, 1841) and other species of genus *Raja*.

Fisheries onboard sampling

Sampling was carried out onboard of commercial vessels fishing primarily with trammel nets and traps in the MPA and nearby its limits. It should be noted that these vessels also fished with other gears at particular times or for short periods. Vessels operating with jigs were also common in the AMP (Horta e Costa et al., 2013a; Horta e Costa et al., 2013c), however, their small size did not allow onboard observations and thus they were not included in the present study. Surveys were performed onboard of five vessels, between May 2007 and April 2008 with a total of 35 fishing trips (16 for traps – 3 in *Year 1* and 13 in *Year 2* - and 19 for trammel nets– all samples from *Year 2*). Vessels were chosen considering the fishing area (identified through the fishing effort assessment surveys), in order to distribute sampling along the entire study area and also fishers' willingness to participate in the study (Figure 3.1). Sampling was also regularly distributed in time, in order to account for possible seasonal fluctuations in catches. All vessels and fishing gears sampled had similar design characteristics.

Generally, fishing trips were conducted separately for nets and traps (except for five surveys in which fishers hauled both gears in the same fishing trip). In each survey, two researchers accompanied one full-day fishing trip; each trip lasted ca. 6 to 8 hours. Nets' length, number of traps or nets per set, haul location, depth, fishing time (total immersion time) and bait (for traps) were recorded for each set. Catches were separated by fishers into specimens for selling, for their own consumption or for discarding (discards were separated only because of study sampling procedures but they are usually promptly thrown back into the sea). All retained individuals were identified, measured (total length to the nearest mm) and weighed (with a dynamometer, to the nearest 5 g). Individuals discarded alive (mostly small octopus and skates, *Raja* spp.) were also carefully measured and weighed onboard. The remaining (dead) discards were preserved on ice and brought to the laboratory to be identified, measured and weighed. A total of 5521 traps (grouped in a total of 102 sets) and 930 nets (grouped in a total of 48 sets) were surveyed.

Data analysis

Fishing effort assessment

In order to assess fishing effort, buoy density (buoys.km^{-2}) was calculated for all the AMP and by protection zone, for the five periods considered together and separately for each fishing gear (traps and trammel nets) using ArcGIS 10.0 (ESRI, 2011). Data from the fisheries onboard surveys were used to calculate the average number of traps and the average length of nets per set. Significant differences in gear density among periods and among zones per period were tested with ANOVA (for parametric data) and Kruskal-Wallis (for non-parametric data) statistics, using Statistica 12 software (Statsoft, 2013). Tukey and Dunn post hoc tests were done, respectively for tests of protection zones.

Official landings

In order to evaluate landings within the timeframe of this study, we estimated the relative importance of species landed by selected vessels per time period. To avoid potential seasonal effects in landings data, we considered data from an entire year for these analyses (Before period –2004; Year 1 – from September 2006 to August 2007; Year 2 – September 2007 to August 2008; Year 3 – September 2008 to August 2009;

After – September 2009 to August 2010). This aggregation allowed us to compare a full year's landings under the same protection measures. Averaged monthly landings and revenues per vessel were also calculated.

In order to analyse the trends of landings per species across the study periods, landings per unit of effort (LPUE, $kg.vessel^{-1}.day^{-1}$) for each period were calculated for the group of species representing 95% of total landings (only data for the months sampled in the fishing effort assessment surveys were considered). Total revenues per unit effort (RPUE, $€.vessel^{-1}.day^{-1}$) were also estimated, based on landings data.

In order to analyse the relation between official landings and sampled fishing effort, we calculated a second measure of LPUE, landings per gear unit (LPGU), that accounted for the average amount of fishing gears set (obtained during the fishing effort assessment surveys; $kg.\ day^{-1}.\ trap^{-1}$; $kg.\ day^{-1}.\ 1000m$ of $nets^{-1}$). LPGU was calculated for *O. vulgaris* (for traps) and *Solea* spp. and *S. officinalis* (for nets) per period. For each period, landings observed during the same days of the fishing effort surveys were included and non-identified buoys (i.e. buoys where vessels' identification were not visible) were excluded. A preliminary assessment of seasonal effects on landings was performed (seasons were set based on landings trends and species biology) with no significant differences among seasons found (ANOVA, $p\text{-value} < 0.05$). The only exception was *S. officinalis*, from June to October, when landings of this species were almost null and thus excluded from the analyses of LPGU. ANOVA and Kruskal-Wallis tests were performed to analyse statistical significance of differences found in LPGU between periods.

Fisheries catches from onboard sampling

Average, standard deviation and mode of the number of traps per set, total length of each net, number of sets per day, depth and fishing time (soak time) were calculated. Data obtained during onboard fisheries surveys were analysed in order to calculate total catches, catches for fish to be sold, catches for self-consumption and discards per species (in biomass), separately for each fishing gear. The primary reasons for discards (i.e. no commercial value, undersized individuals or damaged condition) were also assessed.

Comparisons between official landings and catches from onboard sampling

Unreported catches (catches retained but not sold at fish auction or catches discarded) were estimated comparing data from onboard sampling with landings from the same day, for the same vessels using the most important species (*O. vulgaris* in the trap fishery and *Solea* spp., European hake *Merluccius merluccius* (Linnaeus, 1758) and *Raja* spp. in the trammel net fishery). Species LPUE (from official landings data; LPUE, kg.vessel^{-1}.day^{-1}), when available, was compared with catch per unit effort (CPUE; data of captures from onboard observations, kg.vessel^{-1}.day^{-1}) obtained in fisheries surveys, by vessel for the same species and for the same days.

Relationship of effort and season on catches

To understand which factors influence the total catch of commercial species in the AMP, generalized linear modelling (GLM) was run relating the response variable total catches (from the onboard sampling) to the fixed covariates season or month (categorical), the number of traps or nets' total length and average water depth (numerical). Separate models were computed for the most caught species by each fishing gear: *O. vulgaris* for traps fishery; and *S. officinalis*, *Solea spp.*, *M. merluccius* and Rajidae for the trammel net fishery. Different seasonal groupings or months were tested in order to represent the influence of seasons on the different species analysed. Choosing gamma as the exponential family and using log as a link function resulted in residuals showing a good approximation with normality in all models. ANOVAs were conducted on each model to test the significance of the covariates tested. Since these GLM models assume linearity of the response variable in relation to the predictors, we used generalized additive models (GAM; package mgcv 1.7-27, R software) and multivariable fractional polynomials (MFP; package mfp1.4.9, R software) to confirm linearity and to test for the best transformation of the response variable to achieve linearity, respectively. Both approaches confirmed the linearity of the dataset. Another assumption of GLMs is the independence between samples. The onboard sampling is fishery-dependent data since it is obtained by commercial fishers (and registered by an onboard observer). Different crews from different vessels may have distinct fishing strategies which possibly influence total catches. Therefore, including the effect of individual vessels in the model as a random variable is a common approach to deal with this lack of independence (Coelho, 2007). Generalized linear mixed models (GLMM) are extensions of GLM and combine two important statistical features commonly used in biological studies namely, linear mixed models, which incorporate random effects, and generalized linear models, which handle non-normal data (Bolker et al., 2009). GLMMs were run using the model characteristics selected above for the GLMs but including vessels as a random effect (package lme4 1.0-5, R software). The best model obtained for each situation was compared with the respective model testing only the random variable.

For each model the values of the AIC - Akaike Information Criterion (Akaike, 1974), and R^2 - Nagelkerke coefficient of determination (Nagelkerke, 1991) were calculated and used for model comparison and selection in terms of goodness-of-fit. The R^2 was modified for the GLMMs following the method described by Nakagawa et al. (2013). The obtained marginal R^2 (i.e. variance explained by fixed factors) was compared to the conditional R^2 (i.e. variance explained by fixed and random factors) for each GLMM tested. When the random effect of the vessel had approximately zero in variance, and the conditional R^2 was equal to the marginal R^2, we assumed the variances of the different vessels were similar, that is, the vessel term did not add extra explanatory power to the model. In that situation the model selected was the GLM instead of GLMM.

The best models obtained were GLMM for octopus in traps and cuttlefish in nets, and GLM for fish in nets. The GLMM model for octopus related total catches (without discards) with total number of sampled traps, season and vessel as the random variable. The GLMM model for cuttlefish related the total catches with the nets total length and season, with vessels as the random variable. The GLM model for fish related the total catches with the nets total length and months. All these analyses were conducted in the R 3.02 software (R Core R, Development Core Team 2012).

RESULTS

Fishing effort assessment

The analyses of fishing effort within the AMP and by protection zone across time periods showed different patterns for each of the fishing gears studied. Generally, trap density increased over time (Figure 3.2a). However, the specific pattern of increase differed among protection zones (Figure 3.2a). In PPA2, PPA3 and BA2 trap density increased continuously whereas in the FPA trap density increased between Before and Year 2 when the FPA was functioning as a partially-protected area (where traps are allowed beyond 200m from shore) and then decreased after year 2 when the FPA was fully implemented as a no-take zone. PPA4 showed a marked decrease after Year 1 when this area became a partially-protection zone. These variations across time periods, per protection zone, were statistically significant (p - value < 0.05). There were also significant differences in the overall trap density between periods and zones (p < 0.05). Post-hoc tests showed that trap density was significantly lower in Before and in Year 1 than in subsequent periods and that differences among zones (considering all periods together) were also statistically significant (p <0.05), except for the groups FPA, BA2 and PPA4 (lower density) and PPA2, PPA3 (higher densities).

Patterns of net density in the trammel net fishery were almost the opposite of those of the trap fishery, with a steep decrease in trammel nets over the study period (Figure 3.2b). There was a significant decrease between Before and Year 1 (p - value < 0.05). Fishing effort per zone showed a similar trend, with the exception of increases in the density of nets in PPA2 between Before and Year 1 and in BA2 between Year 1 and Year 2. BA2 had the highest density of nets in all periods, and was significantly different from the remaining zones (p - value < 0.05). PPA3 and FPA were the second and third most important zones for this fishing gear during the Before period, losing their importance to the following periods. Statistical tests performed on nets density for each zone separately only detected significant differences in PPA4 (p - value < 0.05), with significantly higher densities in Year 2, comparing to the following periods.

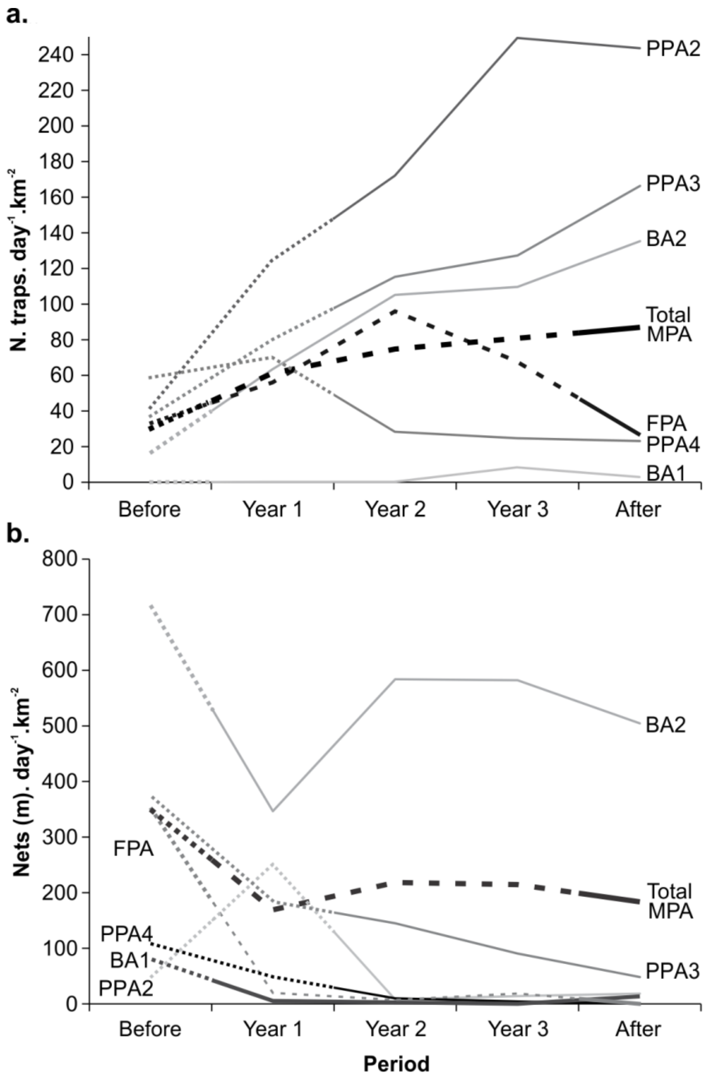

Figure 3.2. Density of (a.) traps and (b.) nets by protection zones at the Arrábida Marine Park throughout the different periods of the implementation of the management plan: Before, Year 1, Year 2, Year 3 and After. Dotted lines – no conservation measures are implemented; dashed lines – conservation measures of FPA are partially implemented; solid lines – Conservation measures are fully implemented.

Official landings

Total landings of vessels operating in the park or in adjacent fishing grounds showed an increasing pattern both in weight and overall revenues through time (Supplementary data A.1). The number of vessels continuously fishing with nets and traps in the MPA decreased between Before (22 vessels) and After (18 vessels) periods. Total revenues per unit of effort (RPUE = €.vessel^{-1}.day^{-1}), estimated through landings data, increased over the studied time frame (Supplementary data A.1).

Although there was some variability in the relative importance of species landed, the average monthly landings per vessel (in biomass) generally mirrored the pattern of total landings, showing an increase over the study period (Figure 3.3). A total of approximately 160 taxa were landed but 95% of total landings were comprised of only 10 taxa (Figure 3.3, Supplementary data A.1). *Octopus vulgaris* was the most landed species in all periods, accounting for between 71.7% (Year 1) and 86.5% (Year 3) of total landings. Species targeted by trammel net fishery (*S. officinalis*, *Raja* spp. and *Solea* spp.) showed in general a decreasing pattern in total catches between Before and After periods (Figure 3.3). Although variable, trends observed for LPUE were generally similar to those of the global biomass landed per species (Figures 3.4a, b, c, d).

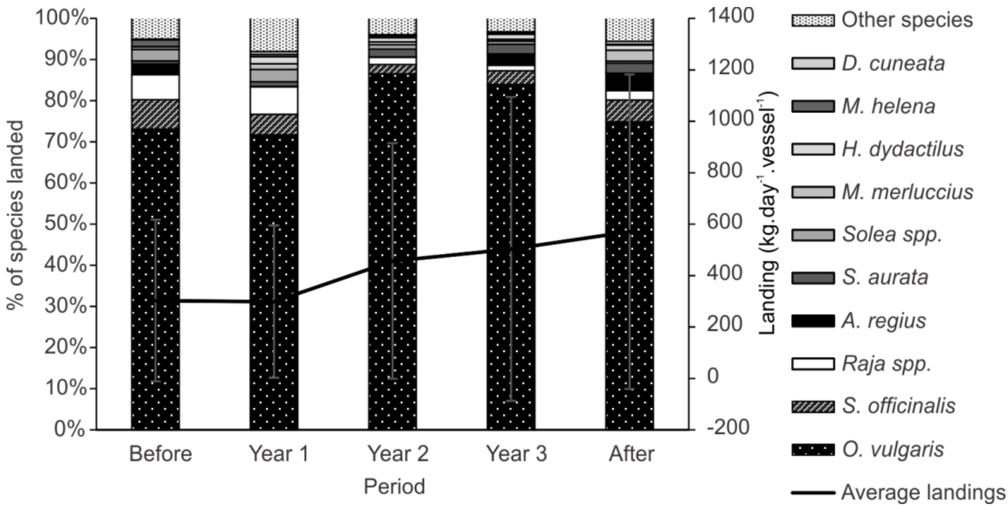

Figure 3.3. Relative importance (%) of species in the overall landings at the Sesimbra port throughout the different periods of the implementation of the management plan: Before, Year 1, Year 2, Year 3 and After. Averaged monthly biomass (kg) landed per vessel and respective standard deviation are also represented.

The 10 most landed taxa (except *H. dydactilus* and *Muraena helena*) plus the European seabass, *Dicentrarchus labrax* (Linnaeus, 1758), were also responsible for 95% of total revenues (Supplementary data A.1). *Octopus vulgaris* was the most important for total revenues, followed by gilthead seabream *Sparus aurata* Linnaeus, 1758, *Solea* spp. and *S. officinallis*, respectively (Supplementary data A.1).

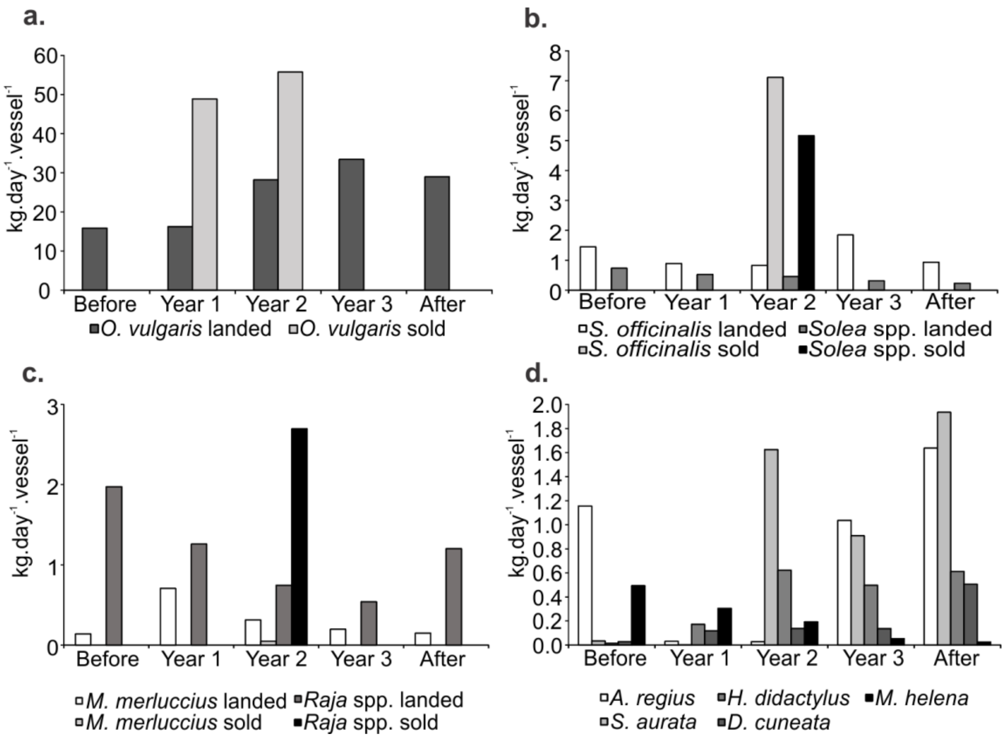

Figure 3.4. Landings per unit of effort (LPUE; kg. day^{-1}. vessel^{-1}) at the Sesimbra port for the most landed species throughout the different periods of the implementation of the management plan: Before, Year 1, Year 2, Year 3 and After. For the most captured species during onboard observations (only available for Year 1 to species from trammel nets fishery and for Years 1 and 2 for *O. vulgaris*), captures separated by fishers to sell are shown (CPUE: kg.day^{-1}.vessel^{-1}). (a) *O. vulgaris* LPUE and CPUE; (b) *S. officinalis*, *Solea* spp. LPUE and CPUE; (c) *Raja* spp., *M. merluccius* LPUE and CPUE and (d) *A. regius*, *S. aurata*, *H. didactylus*, *D. cuneata* and *M. helena* LPUE.

Results from the analyses of LPGU (kg. day^{-1}. trap^{-1}, kg. day^{-1}. 1000m of nets^{-1}) per period are shown in Figure 3.5. *Octopus vulgaris* LPGU increased over the study period with a significant difference between the Before and After periods (p-value < 0.05). *Solea* spp. did not show significant differences among periods. *Sepia officinalis* LPGU were relatively constant between Before and Year 3 showing a maximum in the After period although no significant differences in LPGU among periods were detected.

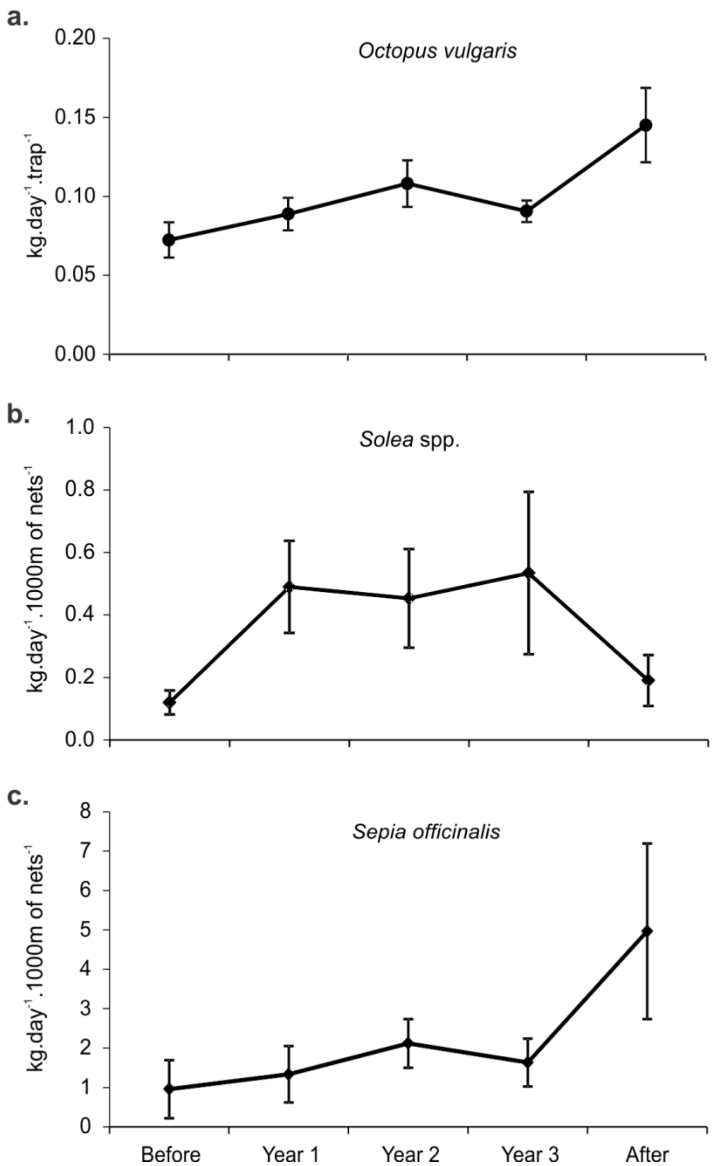

Figure 3.5. *Octopus vulgaris* (a), *Solea* spp. (b) and *Sepia officinalis* (c) average landings per gear unit (LPGU), (a) kg.day^{-1}. trap^{-1}; (b, c) kg.day^{-1}. 1000m of nets^{-1}, and standard error at the Sesimbra port throughout the different periods of the implementation of the management plan: Before, Year 1, Year 2, Year 3 and After.

Fisheries catches

Trap fishery

Traps are made of an iron or steel frame with a hard plastic net stretched around it (mesh size: 3cm, trap dimensions: 50 cm x 30 cm x 20 cm). Each trap has a bait holder and a funnel shaped opening. *Scomber colias* and *C. maenas* were the most commonly used baits. Sometimes *B. boops*, *T. trachurus* and sardine *Sardina pilchardus* (Walbaum, 1792) were also added to *S. colias* as bait. All bait species were dead, except *C. maenas*. Traps were set from 24h to several consecutive days, attached to a mainline (Table 3.2).

Table 3.2. Technical characteristics of fishing gears sampled in onboard surveys. Average, standard deviation and mode of depth, number of traps or nets total length, soak time and number of set hauled per day.

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

During trap sampling 41 taxa (36 taxa identified at species level and the remaining classified in higher taxonomic levels) were identified. 78% of the total biomass caught was sold, 13% was discarded and the remaining catches were retained by fishers for personal consumption. The target species *O. vulgaris* represented 91% of total catches. *Halobatrachus didactylus* (Bloch & Schneider, 1801), Holothuroidea and *Scorpaena notata* Rafinesque, 1810 were the following most-caught taxa, although with low relative importance. Moreover, several economically valuable species were caught, however these accounted for low biomasses (e.g. *S. officinalis*, *Necora puber* (Linnaeus, 1767)). 93% of *O. vulgaris* biomass caught was retained (8.5% of which was for fishers own consumption) and the remaining 7% were discarded mostly alive as underweight (Supplementary data A.2). Interestingly, discards of undersized individuals were much higher in nearshore zones (within 200 m offshore) than offshore (ca. 8 individuals. 100 traps^{-1} and 0.4 individuals. 100 traps^{-1}, respectively).

Trammel net fishery

Trammel nets were composed of 3 panels, made of polyethylene, with an inner panel with 100mm stretched mesh (the minimum allowed by Portuguese legislation) with ca.

50 meshes high and two outer panels with 600mm stretched mesh with 3–4 meshes high (Table 3.2).

Approximately 80 taxa were identified in the trammel net fishery but 80% of total biomass caught was comprised of only 15 species (Supplementary data A.3). From the total biomass caught, 44% was for sale, 34% was discarded and 22% was for fishers own consumption. Target species were *Solea* spp. (mainly *S. senegalensis* and *S. solea*) and *S. officinalis* which represented 30% of total biomass caught (6.9%, 4.5% and 19.1%, respectively) and 65% of total sales (in biomass). Several other species showed relatively high retention of catches (e.g. *Raja spp.*, *M. merluccius*, grey triggerfish, *Balistes capriscus* Gmelin, 1789, *O. vulgaris*). Among those species, *Raja* spp. accounted for high relative importance among total captures for sale (13.5% of the total biomass sold) and *B. capriscus* and *M. merluccius* were most likely to be retained for self-consumption (20.1% and 13.9% of total catches for personal consumption, respectively). *Scomber colias* and various invertebrates were the most discarded species (*S. colias*: 42.7% of total discards). In general, catches directed to fishers' self-consumption had small market value (due to species characteristics, individuals' size or to the small number of individuals of a given species caught). Unmarketable value or damaged conditions were the major reasons for discards.

Comparisons between official landings and catches from onboard sampling

Comparison of trends from official landings data and those from onboard sampling showed that targeted and some high value bycatch (e.g. *Raja* spp.) species occupied the first positions in the ranks of most landed and most caught species (in biomass). However, among the most landed species there were some with relatively lower onboard catches than those recorded in official landings (e.g. *M. merluccius, M. helena,* wedge sole *Dicologlossa cuneata* (Moreau, 1881)). Although having high relative importance among landings, meagre *Argyrosomus regius* (Asso, 1801) and *S. aurata* were never observed during onboard surveys since these species are mainly caught during specific short periods by longlines (some vessels fish with longlines sporadically or within short time-frames each year) and this would not be adequately sampled in our onboard program.

The analyses of official landings from the trap fishery on the same days of onboard sampling and for the same vessels showed that for approximately 68.8% of the fishing trips observed there were no registered landings (but in every sampled trip there were catches separated onboard for sale). An average of 72.2% of total octopus caught (without discards) per day was not reported in fish auctions and thus was not accounted for in official statistics. Furthermore, besides octopus, among all catches recorded during onboard observations, only black seabream, *Spondyliosoma cantharus* (Linnaeus, 1758) and *S. officinalis* (ca. 20% and 64% of total catches were landed at fish auction, respectively) were reported in official landings. CPUE ($kg.vessel^{-1}.day^{-1}$) calculated for *O. vulgaris* (separated onboard for sale by the fishers) were on average more than twice as high as LPUE ($kg.vessel^{-1}.day^{-1}$).

For trammel net fishery, for approximately 16% of fishing trips in which there were catches registered onboard, no landings were registered. The highest levels of unreported catches were for *Solea* spp., *S. officinalis* and *Raja* spp. (45.9%, 36.6% and 75.3%, respectively). For *M. merluccius* there were no reported landings, although 15% of total catches were separated onboard for sale (67% of total catches were for own consumption and 18% was discarded). There were records of landings for 13 more taxa, but several inconsistencies were identified: i) landings recorded species that were not caught during onboard observations (e.g. 127 kg of *T. trachurus* in two different days; 137 kg of *S. colias* in one day; ii) species caught were landed under a wider taxonomic category or incorrect species designation (e.g. *Lepidorhombus boscii* was landed under a category that can include the genus *Citharus* and *Lepidorhombus;* sometimes species from genus *Raja* were landed with incorrect species designation or only as *Raja* spp.); iii) some species (non-target species) were recorded with lower biomass than observed onboard and with lower frequency than caught (e.g. *B. capriscus*, tub gurnard *Chelidonichthys lucerna* (Linnaeus, 1758)). CPUE of target species (*Solea* spp., *S. officinalis*) in trammel nets were strongly higher than LPUE (Figure 3.4b).

Relationship of effort and season on catches

The best model selected for the total catches of *O. vulgaris* (GLMM) targeted by traps fishery explained 72.10% of total variability (Table 3.3). Both fishing effort and season (fixed factors) significantly influenced *O. vulgaris* total catches but 7.11% of variability was explained by random factors (vessels). This species showed a general increasing pattern of catches with the increase in number of traps, until a limit where total catches no longer rise in spite of an increase in the number of traps. Octopus catches were much higher in late winter/ spring than in summer (Table 3.3, Figure 3.6a)

Two distinct models were selected to explain catches from trammel nets fishery, one using *S. officinalis* and the other using the most-caught species lumped. The best model obtained for *S. officinalis* (GLMM), explained 82.80% of total variability (Table 3.3; Figure 3.6b). There was no clear relationship between fishing effort (fixed factor) and the total amount of *S. officinalis* caught, as total catches only increased with the increase of net length in the first few meters, until a point where an additional increase in net length does not result in higher catches. Despite this there are numerous zero catches, probably related to season. In fact, season (fixed factor) had a strong influence on catches, with higher catches observed in autumn/winter (S1) and spring seasons (S2) compared to catches from summer/autumn (S3). Random factors such as individual vessel also influenced *S. officinalis* captures (Table 3.3). The model obtained for the analyses of the most caught fishes in trammel nets (target and relevant bycatch species) explained a small percentage of total variance (10.75%). Only month was significant with higher catches generally observed during cold months (Table 3.3, Figure 3.6c).

Table 3.3. Results of the best models obtained - generalized linear (GLM) and mixed (GLMM) models - testing captures from onboard samples, from traps and trammel nets fisheries, in relation to the best set of explanatory variables selected: fishing effort - number of traps – Ntraps - or trammel nets – Nnets (continuous variable); season or month (*O. vulgaris*: February, March, April; S2: August, September, December; *S. officinalis*: S1: November, December, January, February; S2: March, April; S3: August, September (b); Fish species: month). Species considered were *Octopus vulgaris*, *Sepia officinalis* and the trammel nets most targeted fish species (*Solea senegalensis*, *Solea solea*, *Merluccius merluccius*, *Raja clavata*, *Raja undulata*, *Raja alba*, *Raja montagui* and *Raja brachyura*). The % of variance explained is given by the R^2. In the GLMMs the random variable is the vessel (random effects). Estimated degrees of freedom (edf), statistical tests Chisq (for GLMM), F-statistics (for GLM) and corresponding p-values are indicated. Significant values are in bold. Marginal (i.e. variance explained by fixed factors) and conditional (i.e. variance explained by fixed and random factors) variances are shown for GLMMs.

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

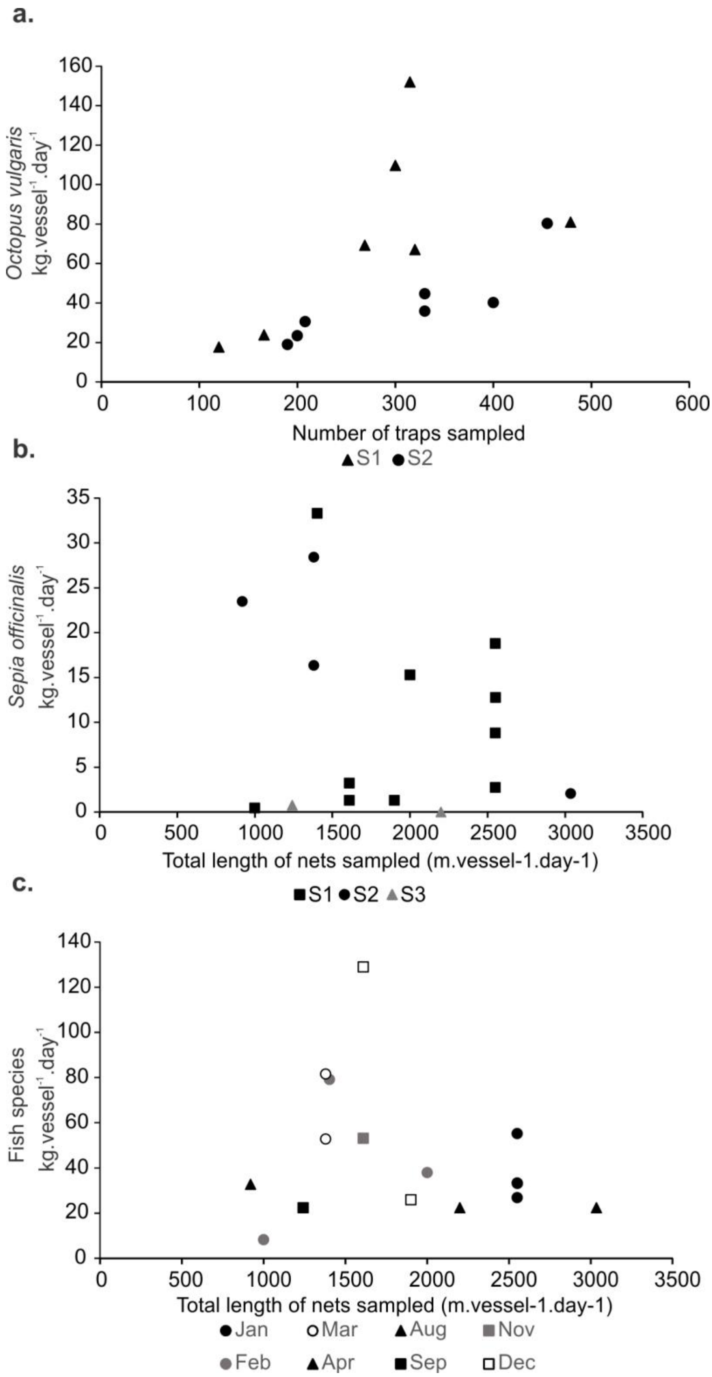

Figure 3.6. Relationship of fishing effort (number of traps. vessel^{-1}.day^{-1} or total length (m) of nets sampled.vessel^{-1}.day^{-1}) and season or month on *Octopus vulgaris* (a), *Sepia officinalis* (b) and a group of fish species (c) catches observed during onboard sampling of vessels fishing in Arrábida Marine Park with traps and trammel nets. Seasons considered were S1: February, March, April; S2: August, September, December (a) and S1: November, December, January, February; S2: March, April; S3: August, September (b).

Discussion

Fishery redistribution and reporting

Conservation measures associated with the implementation of MPAs frequently have impacts on fishing communities and create a need for fishers to adapt (Mascia et al., 2010; Rees et al., 2013). Unfortunately, the range of potential effects on fishers are rarely assessed, especially for small-scale or artisanal fisheries. Therefore, the assessment of catches (and consequently revenues) are essential to evaluate impacts (both positive and negative) of MPAs to local fisheries, the success of fishers' adaptation and the potential effects of protection on fished species. The combination of methods used in this study assessing and monitoring fisheries catches in different phases of a MPA establishment is, to our knowledge, a new approach with strong potential for future application in many other similar systems.

Fishing catch and effort assessments of artisanal fisheries are essential to calibrate official reported commercial landings. When interpreting observed trends obtained from fisheries statistics and monitoring changes in the spatial distribution of fishing effort related to the implementation of marine protected areas, a direct assessment of fishing effort and locations allowed us to identify which vessels fish within the MPA, thus minimizing the inclusion in the analyses of landings from vessels operating primarily in locations outside the park.

Fishing effort analyses revealed that, as expected, MPA implementation had the consequence that some fishing grounds became inaccessible to particular fisheries. In the case of Arrábida, approximately half of the park is off-limits to very nearshore traps and jigs (up to 200m from shore) and also to trammel nets fishery, as discussed in a recent work showing spatial adaptations of fisheries in response to this marine park implementation (Horta e Costa et al., 2013a). In that study, authors analysed the dynamics inherent to spatial re-allocation of fishing effort and the re-adjustment of preferred fishing grounds. They found that, besides general fisheries adaptations to the spatial limitations, individual fishers showed particular strategies with "individual" fishing grounds well defined and apparently a high agreement on individual "territories" among the most frequent fishers. Here, changes in fishing effort (both spatial and temporal) showed a general increase in the number of traps within AMP suggesting that this fishing gear supported most of the fishers' adaptation to MPA regulations replacing trammel nets that expectedly decreased with MPA establishment (since they are not allowed in FPA and PPA). However, nets density also decreased even in the area where they are still allowed, suggesting that MPA rules were not the only factor leading to the decreased use of trammel nets. A re-direction of fishing effort from finfish (caught mostly by nets) to cephalopods (caught mostly by traps) has also been observed at the national level (Moreno et al., 2014; Pilar-Fonseca et al., 2014) and seems to explain the observed trends. Moreno et al. (2014) suggested that this re-direction of fishing effort is largely due to the decline in national demersal finfish stocks. On the other hand, maybe the area where nets were allowed after MPA implementation are very small to support worth fishing and thus fishers changed from nets to traps.

It's important to note that in small-scale coastal artisanal fisheries occurring inside marine protected areas, such as the one studied here, several factors may influence shifts in the use of fishing gears, namely: (i) increases or decreases in the abundance of target species due to environmental conditions, reserve effects and/or excessive fishery pressure (e.g. our observed increase in octopus and decrease in trammel nets target species such *Solea* spp., *S. officinalis*), (ii) market driven factors (e.g. increase in market demand and higher prices for octopus); (iii) loss of fishing grounds. Independently of the specific factors and the levels of their interactions, the multigear and multispecies nature of these local artisanal fisheries seems to be an advantage in response to spatial limitations since other options, such as moving to other fishing grounds far from home port are less viable due to technical limitations (e.g. vessels small size and low power) (Lédée et al., 2012).

The differences observed between reported landings and onboard catches revealed a reality frequently identified in artisanal fisheries worldwide, that is the high prevalence of unreported catches and allowed the estimation of levels of unreported catches (Batista et al., 2009; Coll et al., 2014; Pauly et al., 2014). If we assume that official landing statistics will always be inaccurate, then only by creating means to estimate realistic levels of bycatch and the level of unreported catches may be able to improve fisheries management of these important artisanal fisheries (Ainsworth and Pitcher, 2005; Alverson et al., 1994). Accurate estimation is also key for effective MPA management and monitoring. In this study, we worked directly with fishers, and their collaboration was highly valuable, even essential for accurate sampling procedures and reliable interpretation of landings data. This strongly supports the findings of many previous works that fishers acceptance of MPA can highly influence its success (e.g. Guidetti and Claudet, 2010).

Landings data reliability of these artisanal fisheries proved to be weak. However, landings and catches of target species had similar relative importance despite high levels of unreported catches. Besides target species, other species were found to be economically important for local fisheries, namely skates. However, for these species landings data was even less reliable than for target species, but they are very important for overall revenues of local fisheries, as detected by the onboard samples.

The importance of non-target species to the total revenues of fishers suggests that future studies should assess the role of these species in the context of fisheries restrictions due to the implementation of MPAs (Guenette et al., 2014; Libralato et al., 2010; Valls et al., 2012). Understanding environmental conditions, species life history characteristics and even fishers' behaviour would be important factors to integrate in evaluating the effects of marine protected areas in coastal artisanal fisheries. To integrate the contribution of each factor, longer and more complete data series on catches (as well as data on fishing effort) are essential.

Official landings did not account for discarded species or those that were typically retained for fishers own consumption, and therefore do not reflect the real catches of the fishery as has been widely discussed in the literature (Ainsworth and Pitcher, 2005; Alverson et al., 1994; Batista et al., 2009). Some species (e.g. *M. merluccius*, *H. dydactilus*) were important both in landings and in onboard catches but were mostly discarded or kept for fishers own consumption, while others were important in landings but were not observed in onboard catches (e.g. *S. aurata*, *A. regius*, *D. cuneata*). There are a variety of possible explanations which may prove relevant to similar coastal MPAs found worldwide: (i) landings could be from catches made by other larger vessels (fishing with gill and trammel nets or longlines) and recorded as being from MPA vessels (e.g. *M. merluccius*, *D. cuneata*) due to several reasons (e.g. the need to report at least 100 days of catches to renew the licence); (ii) species could be caught and landed by vessels fishing in the MPA but using non-licenced or other less used fishing gears, since vessels have licenses for several fishing gears (e.g.

longlines), although these were not regularly used; (iii) landed species were not correctly identified (e.g. *D. cuneata* due to morphological similarity with soles). All these factors may explain the differences found between landings and onboard observations. Official landings were also a poor proxy of fishing effort since no landings are reported in most fishing days. Thus onboard observations and number or length of fishing gears set per day would be more appropriate measures of fishing effort for monitoring these coastal fisheries.

Individual species responses

The increase in LPGU of octopus, although not statistically significant, suggest that protection measures may have contributed to some recovery in abundance and/or size of this species within the AMP. Horta e Costa et al. (2013b) found more octopus in highly protected zones (FPA and PPAs) than in control zones, suggesting that protection can benefit the fishery outside these areas if spillover events occur. Also, overall catches of small individuals probably diminished with MPA establishment due to exclusion of fishing within 200 m from shore in PPAs and also to the exclusion of spearfishing and other limitations on recreational fisheries inside the park. In fact, these limitations have been shown to benefit small-scale commercial fisheries in multiple use MPAs (Cooke and Cowx, 2004; Rocklin et al., 2011). Nevertheless, since octopus recruitment and year class strength are known to be strongly influenced by environmental factors, namely salinity and water temperature (Lourenço et al., 2012; Moreno et al., 2014), it is expected that those will be the main factors affecting population abundance in a long time frame. In addition, the increase of trap fishing effort in some PPAs and BAs bordering the no-take reserve may offset any potential protection effects on octopus populations through “dispersion imbalance” phenomenon. Dispersal imbalance effects occur when there is movement out of sites near the MPA boundary, which is not balanced by movements into the site because of lack of source animals from outside the MPA boundary, e.g. due to high fishing effort outside MPA boundaries (Abesamis and Russ, 2005; Walters et al., 2007). This effect can be affecting many other species since trammel net fishing effort also increased in BAs and potentially outside MPA borders and this fishing gear has low species selectivity (see e.g. Erzini et al., 2006). On the other hand, positive effects of excluding net fishing from large zones of MPAs can be overcome by negative effects of “fishing the line” (Babcock et al., 2010; Stelzenmüller et al., 2008).

S. officinalis also showed an increasing trend in LPGU in the After period which could be a sign of positive effects of the reserve. However, the small size of the no-take area together with the short life cycle and rapid growth of this species (Dunn, 1999; Neves et al., 2009) make it unlikely that protection alone played a major role for the trends observed in this species. This assumption is supported by the recent results from Abecasis et al. (2013) who found there were no significant differences in cuttlefish biomass before and after the reserve implementation and that this species has low site fidelity (shown through biotelemetry). The protected area may however contribute to a decrease in mortality (by fishing) of mature individuals before breeding since individuals use this coastal area before entering the nearby Sado estuary (Abecasis et al., 2013; Neves et al., 2009). These authors also suggested that larger individuals can spawn in the adjacent coastal areas which could improve the positive effects of the reserve, as also suggested by Abecasis et al. (2013). Again, as found for octopus, environmental conditions are likely to contribute more strongly (and over a longer time-frame) to recruitment success and population trends, since the growth of juveniles is extremely dependent on environmental conditions (Dunn, 1999; Koueta and Boucaud-Camou, 2003).

Solea spp. LPGU showed similar values from Before to After periods with no significant differences through time. In any case, benefits for the sole fishery due to protection effects, if occurring, are unlikely to be detected through landings data *per se* since high levels of unreported catches obscure any possible sign of protection effects. The lower fishing pressure on these exploited species is likely to allow high adult biomass and abundance in the "no-nets" area (PPA and FPA). For instance Claudet et al. (2010) in a study considering several temperate reserves of Southern Europe found significant increases in exploited fish species density inside reserves. In addition, these authors showed responses increased with time since protection and with the size of the no-take zone (Babcock et al., 2010; Claudet et al., 2008). However, in a recent study by Abecasis et al. (2014) no significant changes in mean abundance or biomass of *S. senegalensis* attributable to MPA protection levels were found. Hence, predictions about the contribution of the study area to an increase in the surrounding fishery are premature but the reserves' small size (see e.g. Claudet et al., 2008; Palumbi, 2003) and the fact that key parts of these species life-cycle did not occur inside the park make it unlikely that this MPA will be the main factor contributing to the sustainability of some of these species. For instance, the quality and availability of known nursery grounds in the nearby estuaries may be an important key component to add for conservation efforts of species such as soles, cuttlefish and some finfish (Tanner et al., 2013; Vasconcelos et al., 2011).

Conclusions

Here we show that vessels fishing inside a MPA did not suffer a decline in revenues as would many expected due to the implementation of fisheries restrictions inside the park (Batista et al., 2011; Lester and Halpern, 2008). Fishers apparently compensated for losses in some target species (such as cuttlefish and soles) with increases in the use of other gears (such as traps and the consequent increase in octopus catches). However, caution is needed since it is not clear if the current levels of resource exploitation are sustainable and also if other factors might influence misinterpretations of landings trends as discussed by Horta e Costa et al. (2013b). In this MPA, the way protection measures may benefit some target species of these coastal fisheries is still an open question, although early reserve effects have been shown for finfish and some invertebrates (Horta e Costa et al., 2013b). Also, expected effects associated with spillover or larval export processes (Gell and Roberts, 2003) are still unknown. The small size of the no-take zone, the high fishing effort observed in the area, the recent nature of this park and problems in enforcing the MPA rules all can contribute to the lack of a clear signal in the recovery of these coastal fisheries.

The short duration of the onboard sampling made it difficult to accurately assess catches over the entire sampling period. However, the levels of unreported catches and the large inconsistencies detected in official landings when compared with sampling-based data are alarming and need to be addressed as they are likely to be a common feature to many coastal MPAs (Lescrauwaet et al., 2013). Longer data series of onboard observations would allow a higher confidence in the correction factors estimated since several features could influence the level of inconsistency in official statistics (e.g. economic, cultural, seasonal, fishers behaviours). Thus, landings data could be a useful and cost-effective tool for MPA monitoring but only when calibrated with onboard sampling of catches collected for extended periods by combining these data sources and deriving accurate correction factors. Longer sampling datasets would also allow a better fit for the models here applied, allowing the use of fishing effort (number of traps and net's length) to estimate global catches of target species. Efforts to include fishers in MPA monitoring and increasing their awareness to the importance of reliable datasets could be advantageous since it would potentially provide medium/ high reliable data and/ or complementary information (Leleu et al., 2014; Roman et al., 2011) with reduced costs.

The approach described here of combining fishing effort, onboard data collection and official landings proved to be an effective tool for monitoring small-scale artisanal fisheries in MPAs. However in most cases official landings are the only datasets available. In order to be able to use these data reliably one needs to assume that biases are consistent over time (e.g. the relative importance of unreported catches among years are viewed as constant), allowing deriving general patterns and trends in putative catches and revenues. Also, knowledge of fishers' behaviours and perceptions and on the socio-ecological system under study are essential. Understanding these general trends may help in the early detection of unsustainable exploitation practices allowing managers to implement more efficient or preventive monitoring and management measures. The inclusion of local fishers in monitoring processes can also be of great importance for the interpretation of landing results since local knowledge, evaluation of unreported catches and spatio-temporal changes in fishers' behaviour influence the outcome and consistency of those landings data.

Landings data for coastal artisanal fisheries worldwide are very scarce, unreliable and biased. In particular, when using these data for monitoring or evaluating the effectiveness of marine protected areas, methodological approaches which allow disentangling the different bias effects in these data are essential. Here we provide a method for increased reliability in studying these types of fisheries which are prevalent worldwide in coastal systems.

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Supplementary data A.1. Total biomass (kg) and value (€) of the species most landed in Sesimbra port in Before (official landings from 2004), Year 1 (from September 2006 to August 2007), Year 2 (September 2007 to August 2008), Year 3 (September 2008 to August 2009) and After (September 2009 to August 2010) periods**.** Relative importance of biomass and value per species are presented between parentheses. Number of vessels selected and averaged LPUE (Kg.vessel^{-1}.day^{-1}) and RPUE (€.vessel^{-1}.day^{-1}) are also indicated for each period.

	Before (22 vessels)		**Year 1 (23 vessels)**		**Year 2 (18 vessels)**		**Year 3 (16 vessels)**		**After (18 vessels)**	
Species	**Kg (%)**	**€ (%)**	**Kg (%)**	**€ (%)**	**Kg (%)**	**€ (%)**	**Kg (%)**	**€ (%)**	**Kg (%)**	**€ (%)**
	55809.1	283712.1	55723.1	259782.9	84932.6	383723.2	80690.6	326370.2	90094.4	326911.3
Octopus vulgaris	(73.1)	(71.3)	(71.7)	(66.6)	(86.5)	(83.0)	(83.9)	(75.7)	(73.4)	(65.4)
	5448.5	21211.2	3880.6	17487.9	2275.7	11343.9	3216.6	14453.8	7586.0	29639.5
Sepia officinalis	(7.1)	(5.3)	(5.0)	(4.5)	(2.3)	(2.5)	(3.3)	(3.4)	(6.2)	(5.9)
	4650.6	15177.8	5181.5	16883.8	1699.7	5047.4	1277.3	4594.2	3255.3	10696.2
Raja spp.	(6.1)	(3.8)	(6.7)	(4.3)	(1.7)	(1.1)	(1.3)	(1.1)	(2.7)	(2.1)
Argyrosomus	1949.2	18151.4		854.1		818.7	2681.7	18405.2	5059.4	47303.4
regius	(2.6)	(4.6)	99.6 (0.1)	(0.2)	89 (0.1)	(0.2)	(2.8)	(4.3)	(4.1)	(9.5)
	595.5	6813.1	867.4	10630.0	1888.5	23591.6	2269.6	34425.3	2987.5	34445.4
Sparus aurata	(0.8)	(1.7)	(1.1)	(2.7)	(1.9)	(5.1)	(2.4)	(8.0)	(2.4)	(6.9)
	2127.2	28902.0	2328.9	39763.1	1008.1	16073.5	706.1	10286.5	625.9	8202.7
Solea spp.	(2.8)	(7.3)	(3.0)	(10.2)	(1.1)	(3.5)	(0.7)	(2.4)	(0.5)	(1.6)
Merluccius	468.7	2140.6	1070.8	6122.3	730.1	3366.8	474.4	1373.2	3253.9	7343.7
merluccius	(0.6)	(0.5)	(1.4)	(1.6)	(0.7)	(0.7)	(0.5)	(0.3)	(2.7)	(1.5)
Halobatrachus		49.5	1328.7	567.6	971.8	552.2	1158.5	821.8	1527.8	793.0
didactylus	58.8 (0.1)	(0.01)	(1.7)	(0.1)	(1.0)	(0.1)	(1.2)	(0.2)	(1.2)	(0.2)
	1251.8	2141.7	431.7	942.6	431.6	1138.6	215.1	577.4	238.6	756.7
Muraena helena	(1.6)	(0.5)	(0.6)	(0.2)	(0.4)	(0.2)	(0.2)	(0.1)	(0.2)	(0.2)

Dicologlossa cuneata	153.9 (0.2)	902.8 (0.2)	542.4 (0.7)	3460.2 (0.9)	308.3 (0.3)	3111.8 (0.7)	391.4 (0.4)	2667.6 (0.6)	907.9 (0.7)	6664.9 (1.3)
Dicentrarchus labrax	238.9 (0.3)	2741.5 (0.7)	549.7 (0.7)	6165.5 (1.6)	63.6 (0.1)	757.2 (0.2)	333.4 (0.3)	4373.3 (1.0)	556 (0.5)	6377.4 (1.3)
Other species	3573.5 (4.7)	15976.0 (4.0)	5714.3 (7.4)	27386.0 (7.0)	3788.5 (3.9)	12653.0 (2.7)	2746.1 (2.9)	12842.5 (3.0)	6577.9 (5.4)	21013.8 (4.2)
Total	76325.7	397919.7	77718.7	390045.9	98187.5	462178.0	96160.8	431190.6	122670.6	500147.8
LPUE ($Kg.vessel^{-1}.day^{-1}$)/ RPUE ($€.vessel^{-1}.day^{-1}$)	28.96	150.96	29.61	148.85	40.65	191.44	42.08	188.70	50.92	207.62

Supplementary data A.2. Species representing 99% of biomass caught during traps fishery onboard sampling. Relative importance in total captures and percentage of each species captures by finality (own consumption, discards, sale) are shown.

Species	**% of total captures**	**Captures finality (%)**		
		Own consumption	**Discards**	**Sale**
Octopus vulgaris	90.65	7.59	7.15	85.26
Halobatrachus didactylus	3.85	0.00	100.00	0.00
Scorpaena notata	0.97	0.00	100.00	0.00
Holoturithae	0.84	0.00	100.00	0.00
Spondyliosoma cantharus	0.65	92.55	0.00	7.45
Necora puber	0.54	54.47	5.41	40.11
Sepia officinalis	0.49	19.16	0.00	80.84
Marthasterias glacialis	0.24	0.00	100.00	0.00
Balistes capriscus	0.20	46.90	53.10	0.00
Maja squinado	0.19	0.00	41.28	58.72
Conger conger	0.19	0.00	0.00	100.00
Astropecten spp.	0.16	0.00	100.00	0.00
Other species	1.03	36.11	62.95	0.94
TOTAL	100	8.34	13.41	78.25

Supplementary data A.3. Species representing 80% of total biomass caught during trammel nets fishery onboard sampling. Relative importance in total captures and percentage of each species captures by finality (own consumption, discards, sale) are shown.

Species	% of total captures	Captures finality (%)		
		Own consumption	Discards	Sale
Sepia officinalis	19.12	10.69	0.00	89.31
Scomber colias	14.66	1.49	98.51	0.00
Solea senegalensis	6.92	0.00	0.26	99.74
Merluccius merluccius	6.72	67.08	18.33	14.59
Parastichopus regalis	4.55	0.00	100.00	0.00
Solea solea	4.49	14.45	0.65	84.89
Balistes capriscus	3.84	81.08	14.02	4.90
Raja undulata	3.73	6.52	1.48	92.00
Octopus vulgaris	3.31	23.65	0.89	75.46
Solea lascaris	2.99	52.65	3.51	43.84
Echinoida	2.63	0.00	100.00	0.00
Lepidorhombus boscii	2.60	24.95	9.70	65.35
Raja brachyura	2.02	9.51	4.47	86.02
Astropecten spp.	1.49	0.00	100.00	0.00
Halobatrachus didactylus	1.46	58.69	19.38	21.94
Other species	19.54	38.81	41.46	19.73
TOTAL	100	22.39	33.81	43.80

CHAPTER 4

Batista MI, Henriques S, Pais MP, Cabral HN. A framework for a rapid assessment of MPA effectiveness based on life history of fishes. In review in *Aquatic Conservation: Marine and Freshwater Ecosystems*

A framework for a rapid assessment of MPA effectiveness based on life history of fishes

ABSTRACT

Changes in ecosystems structure and function due to the high impacts human pressures on oceans have led to the increasing numbers of Marine Protected Areas (MPA), since MPA are widely accepted as adequate tools to protect, maintain, and restore ocean ecosystems. Increases in density, size and biomass of organisms within protected areas have often been found. However, their worldwide effectiveness are compromised by the interaction of several factors, such as inadequate conduction of processes (e.g. due to political reasons) and the common lack of appropriate scientific datasets to support planning and management decisions. Here a framework for the rapid assessment of potential for effectiveness and identification of major gaps in already implemented MPA were developed. The framework is based on three components: species-habitat association, species' life history and functional groups and pressures affecting the MPA. The potential of MPA to support life history of fish species were assessed in order to identify the main actions to be taken regarding effective management. The theoretical framework was applied to Arrábida MPA (Portugal) in order to exemplify its practical results. This MPA showed a potential to support the life cycle of most of the species although some key life phases (spawning and nursery) are not expected to be efficiently protected by the actual limits/ level of protection of MPA, namely for some of the most important target species. Results are discussed in the view of adaptive management. This framework is particularly useful as an alternative or a complementary support to early decisions for MPA management and the rapid identification of priority actions needed to ensure the accomplishment of initial objectives or the suitability of their adaptation.

Keywords: Conservation planning, Environmental adaptive management, Marine Protected Areas, Biodiversity, Fish life history, Functional guilds

INTRODUCTION

The worldwide evidence of marine ecosystem degradation have led to the implementation of increasing numbers of Marine Protected Areas (Agardy, 1994; Halpern et al., 2008b; Lubchenco et al., 2003; Wood et al., 2008). Changes in marine ecosystem structure and function (e.g. changes in species diversity, population abundance, size structure, habitat structure, trophic dynamics, biogeochemistry, biological interactions) can influence the overall goods and services provided by marine ecosystems. Marine Protected Areas are viewed as adequate tools to prevent and reverse the widespread declines in exploited marine populations and to protect, maintain, and restore ocean ecosystems since they allow for multidisciplinary approaches (social, economic, cultural and environmental) and focus on processes and ecosystems functioning (Claudet, 2011; Lubchenco et al., 2003; Palumbi et al., 2008). The assessment of MPA worldwide have shown that reserve protection can result in significant increases in density, biomass, organism size and species richness of the communities within reserve boundaries (Gell and Roberts, 2003; Halpern and Warner, 2002; Lester et al., 2009). However, only 2.9% of the world's oceans have any form of protected status and only 0.01% of the global protected area is fully protected from extractive uses (Abdulla et al., 2013 and references therein). Although MPAs might play an important role in biodiversity conservation and fisheries enhancement, their implementation, management and evaluation processes are often inadequate, leading to an overall low effectiveness rate (Wood et al., 2008). One of the primary factors contributing to MPA effectiveness is their adequate planning and several studies have been published in order to guide these processes (FAO, 2011; Francour et al., 2001; Kelleher, 1999; Roberts et al., 2003a; Roberts et al., 2003b). It is essential to define concrete and clear goals for a given MPA at an early stage and keep them in mind throughout the design, management and evaluation processes (Halpern, 2003). Although protected areas always affect the entire ecosystem, a reserve aiming for the enhancement of a fish stock (i.e. fisheries sustainability) needs to be placed, sized and designed taking into account target species' biology, such as spawning and nursery areas, species mobility or migration patterns and ensure that the supply of surrounding areas (trough migration or spillover) is efficient (Botsford et al., 2003; Gell and Roberts, 2003; Halpern, 2003). On the other hand, strictly conservation reserves should focus more on the maintenance of diversity and abundance of organisms within the reserve itself (Botsford et al., 2003; Hastings and Botsford, 2003). Once well-defined goals exist, location, size, shape and zoning of the protected area are among the principal factors to define (Margules and Pressey, 2000; Roberts et al., 2003a). In general, to attain fisheries sustainability goals, larger protected areas (or systems of various protected areas) are needed, when compared to conservation goals (Hastings and Botsford, 2003).

However, both fisheries and conservation goals can be attained within the same reserve since marine reserves provide a refuge in space to all species occurring within its limits (Halpern, 2003; Hastings and Botsford, 2003; Rice et al., 2012). Thus, to be effective, reserves must protect species with different life histories and ecological characteristics (Palumbi, 2004). Expected responses to protection may depend from a combination of several factors, such as differences in design (e.g. size, location) or age of reserves (Claudet et al., 2008; Halpern and Warner, 2002; Halpern, 2003), intensity of exploitation to which species are subject outside the reserve and prior to its establishment, their larval, juvenile, and adult dispersal ability and differences in their life history characteristics (Claudet et al., 2006; Claudet et al., 2010; Lester et al., 2009; Micheli et al., 2004). In this context the effects of protection on fish species functional guilds have been addressed in few studies and revealed some common trends. For example, responses to marine reserve establishment have been stronger positive for: exploited species (Claudet et al., 2010; Henriques et al., 2013b; Micheli et al., 2004; Mosquera et al., 2000); species with larger body sizes (Mosquera et al., 2000); species with benthic eggs, ovoviviparity, and small body size (i.e. species likely to have limited dispersal in the larval, juvenile, or adult stages) (Fisher and Frank, 2002); species from higher trophic levels (Micheli et al., 2004); fishes with small to medium home ranges (Claudet et al., 2010). Conversely, unexploited species have often showed no or negative responses to protection (Claudet et al., 2010; Micheli et al., 2004; Mosquera et al., 2000).

Despite all the scientific research to find more efficient approaches to MPA implementation, decisions on the design and location of most existing reserves have largely been the result of political or social processes and thus these are often scattered, disconnected and frequently not based on any application of ecological or environmental principles. This contributes to high levels of MPA inefficiency (Francour et al., 2001; Fraschetti et al., 2005; Gray, 1999; Halpern, 2003; Roff and Evans, 2002). If a MPA is not adequately implemented, its effectiveness may be compromised (e.g. on fisheries, biodiversity conservation, nature tourism enhancement), contributing to the rise of stakeholder movements (e.g. fishers) opposing the MPA. In this situation, noncompliance and illegal behaviors will increase with time and probably lead to total inefficiency (Christie, 2004; Fenberg et al., 2012; Rice et al., 2012) and consequently to a waste of valuable resources used in implementation.

Once a MPA is created, it is expectedly easier to obtain financial and technical resources for its management than in the planning phase. These difficulties can be minimised with an adaptive management approach (McCarthy and Possingham, 2007; Walters and Hilborn, 1978), where managers begin by analyzing the adequacy of several design options and identify the major gaps regarding reserve design. Then they can verify what tools are available to better address and adapt MPA characteristics (e.g. size, zonation, regulations) in order to successfully achieve the proposed goals (Claudet et al., 2008). Adaptive management is essential to enhance the efficiency of reserves because, although the initial design of a reserve may be suboptimal, key variables can be changed later depending on the information gained from monitoring and evaluation (Grafton and Kompas, 2005). Besides assessment, active adaptive management requires also a mechanism to ensure feedback from that assessment into policy development (McCarthy and Possingham, 2007).

Management decisions should be supported by adequate data on several biological components of the protected area, namely habitat distribution, species composition, species life history characteristics and also the level at which threats are affecting biological components. However, adequate scientific data such as long time series (before and after implementation) are rare (Francour et al., 2001; Fraschetti et al., 2005; Ojeda-Martinez et al., 2006) or only available after several years of MPA implementation. For this reason, even after MPA establishment, managers can often count on little more than species richness and snapshots of community status in a limited time-frame to support their initial decisions and, on this scenario, the development of methodological approaches to make an *a priori* crude assessment of the adequacy of the MPA are extremely valuable since it is not practical to delay conservation actions until better data can be collected (Mosquera et al., 2000).

In this study, a framework for rapid assessment of effectiveness and identification of major gaps in an already implemented MPA was developed. The framework is based on three components: (1) species-habitat association, (2) species' life history and functional groups and (3) pressures affecting the MPA. It can be entirely based on published literature and information is integrated in a geographic information system (GIS). The theoretical framework was applied to the Arrábida MPA (Portugal) in order to exemplify its practical application and, ultimately, its applicability to other biological groups and management contexts was discussed.

METHODS

The proposed framework was applied to a Portuguese Marine Protected Area (MPA), Arrábida (Figure 4.1), in order to exemplify and test its applicability. The Arrábida MPA is a coastal multiple-use protected area created in 1998 as an extension to a terrestrial nature park. This MPA extends along a 38 km stretch of coastline (53 km^2) in the west coast of Portugal and its management plan was only approved in 2005. This coast is in a transitional zone where many species with warm and cold affinities reach their southern and northern limits of distribution, respectively (Henriques et al., 2007; Lima et al., 2007; Pereira et al., 2006). It is thus an important hotspot of marine biodiversity in this biogeographic region, where more than 1300 species have been identified (Henriques et al., 1999; Horta e Costa et al., 2014). The management plan defined eight zones subjected to three levels of protection: a fully-protected area (FPA: 4 km^2), four partially-protected areas (PPA: 21 km^2) and three buffer areas (BA: 28 km^2). The objectives of this MPA have a wide scope: preserve marine biodiversity, recover habitats, promote scientific research, encourage environmental awareness and education, control and regulate urban effluent emissions, promote nature-oriented tourism and sustainable development, and promote economic and cultural regional activities, such as the traditional “lines and hooks” (i.e. longlines, jigs) fishery. The Arrábida MPA management plan imposed limits and protection measures to various activities, mostly fisheries related (for details on MPA regulation see e.g. Horta e Costa et al., 2013b and references therein).

The framework has six tasks: data collection, definition of habitat categories, classification of species into functional groups, habitat characterization, set of management priorities and, finally, identification of available frameworks to support priority actions (Figure 4.2). For this case study only fish species were considered.

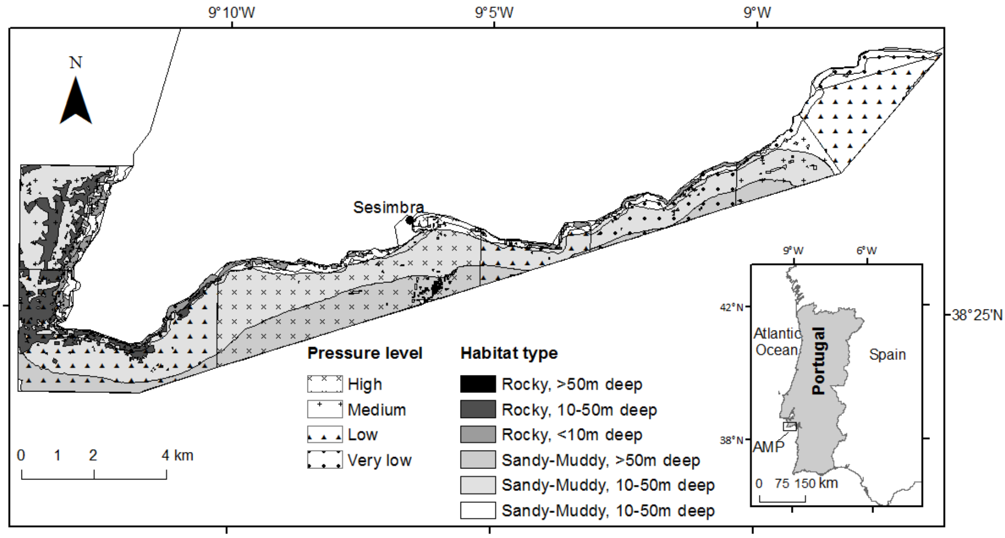

Figure 4.1. Map of the Arrábida Marine Protected Area, Portugal. Habitat type and pressure levels along the different level of protection zones are shown (BA – Buffer areas; PPA – Partially-protected areas; FPA – Fully-protected areas). The line parallel to the coast line are placed to indicate the distance of 200 m from the coast line (area where extractive activities are forbidden in PPA).

Task 1: Data collection - Geographic information, inventory of species and human pressures

For the purpose of the present framework, data collection is based on publicly available data, namely peer reviewed articles, academic theses and technical reports. Three main types of information are required: geographic information (e.g. coast line, limits of the study area, bathymetry, sediment type and benthic habitats), an inventory of the species occurring in the study area and a quantification of the main human pressures potentially affecting the studied communities.

Geographic information

Geographic information is the basis for habitat delimitation and is needed to implement a GIS for the study area. For the purpose of the present framework, at least bathymetry and sediment type maps should be available. However, benthic habitat maps with a higher level of detail are desirable since they allow for more accurate results. This type of information is generally available from research institutions and government agencies.

For the Arrábida case study, bathymetry and coast line information were obtained from the Portuguese Hydrographic Institute and benthic habitat cartography was provided by the Arrábida MPA authority/ ICNF (Cunha et al., 2011).

Species inventory

In order to compile a list of all species occurring in the study area, an exhaustive collection of research publications on the biodiversity of the study area should be performed. Data collected are then included in a database where the organisms studied, sampling characteristics (methods, periodicity, sample size, target habitats) are registered. Once this database is complete, the taxonomic groups (e.g. algae, crustaceans, cephalopods, fish) for which enough information exists need to be identified. In order for a group to be eligible, the data collected must ensure that: (1) all the habitats and seasons are covered and (2) a sufficient number of replicates were performed (a subjective concept, but can be based on a pre-established threshold). In summary, the combination of data from all collected studies has to ensure that most species from a given group occurring in the study area are identified.

Based on these criteria, a database was built for the Arrábida MPA case study. This database includes only studies focusing on fish species, since this was the only group with enough information available (Table 4.1). The sampling methods used in the published studies were underwater visual census for rocky substrates and the rock-sand interface and trammel nets for soft substrates. Studies on fish larvae were also included, as they contain very important data regarding species composition on early life stages inside the MPA.

Cumulative human pressures

Human pressures affecting the study area need to be identified and selected according to their potential impact on the habitats and species considered. Once selected, the spatial extent and intensity of each pressure identified need to be assigned in order to calculate the different levels of pressure affecting the study area. The assignment of levels of pressure to the study area zones can be implemented differently depending on available information. The application of a cumulative pressure index would be desired as showed by recent studies in marine areas (e.g. Ban and Alder, 2008; Ban et al., 2010;

Batista et al., In press; Halpern et al., 2008a; Halpern et al., 2008b; Halpern et al., 2009; Micheli et al., 2013; Stelzenmüller et al., 2010).

For this case study, the methodology used in a recent study that estimated cumulative human pressure levels along the Portuguese coast was used (Batista et al., In press). However, these published results were not directly applied, since more accurate data on fishing effort intensity and distribution as well as recreational activities privilege locations were available for a smaller spatial scale (Cabral et al., 2008; Horta e Costa et al., 2013b).

Due to the uncertainty related to the influence distance of diffuse pressure sources (estuarine pollution and sewage outfalls) and the relatively small dimension of the study area, all zones were considered equally affected by these sources and that this influence was not enough to affect habitat structure. Tourism related activities (beach sports, diving and recreational navigation) were considered, but their influence was diluted by the larger weight attributed to commercial fisheries. Finally, the values of cumulative pressure obtained were standardized into a four level scale, where 1 is the lowest pressure and 4 is the highest pressure level.

Task 2: Definition of habitat categories

After data collection, available information for habitat characterization (e.g. bathymetry and sediment type) is combined and areas are divided into categories within the study area. This task has high relevance since many of the following analyses are performed based on habitat categories. These habitat categories have therefore to find the best compromise between the level of detail from habitat information and the level of current knowledge regarding its use by species. For example, if knowledge on habitats used during a species' life history has low detail (e.g. large depth intervals, low detail on sediment type) or species occupy several levels of habitat categories available, those categories need to be merged until an acceptable scale is reached. Nevertheless, categories that are too broad or ambiguous should be avoided. It is important to note that, since some species change habitat preference during their life cycle, different habitat categories will likely need to be assigned to different life stages (adult life (non-breeding periods), spawning periods and nursery). For the study area, data on bathymetry and benthic habitats were combined in order to define habitat categories.

Task 3: ***Classification of species into functional guilds***

More than looking at species richness, accounting for the functional roles of species within a system is very important in marine conservation, namely for monitoring and assessment purposes (Claudet et al., 2010; Henriques et al., 2008; Henriques et al., 2013b; Micheli et al., 2004; Pais et al., 2012; Pais et al., 2014). Given this, each species listed in the previous task has to be assigned into different functional guilds. Since biological characteristics, ecological functions, sensibility to environmental disturbance and capacity of recovery are different between organisms, the classification of species into functional guilds needs to be adapted depending on the taxa selected, i.e. depending on their role in the ecosystem. In this task, a literature review on the most adequate functional guilds to use should be conducted. For instance, Bremner et al. (2003), Bremner et al. (2006) and Frid et al. (2008) worked on benthic marine communities, while other authors applied functional guild approaches to fish species (Henriques et al., 2008; 2013a; 2014; Pais et al., 2014).

Fish species inventoried for the studied area were classified into 14 functional guild categories, namely their position in water column, migration type, trophic group, trophic level, life span, generation time, body size, mobility, biogeographic affinity, “type” of larvae, reproductive guild, commercial value, level of exploitation and resilience to fishing pressure (see Table 4.2). In addition, the main habitats used by species for three life stages, namely adult life (non-breeding periods), reproduction (spawning) and nursery were also set. Classification of species into functional guilds and guilds definitions was mainly based on previous studies by Henriques et al. (2008), Henriques et al. (2013a) and Henriques et al. (2014). For the new guilds (not included in the referred studies) and for the identification of principal habitats per life history phase, information available in the scientific literature and FishBase database (Froese and Pauly, 2014) was used. Definition of categories for body size were adapted from (Claudet et al., 2010) and classification of species according to their exploitation level (unexploited, exploited with medium-high commercial value and exploited as bycatch) were based in previous studies characterizing fisheries captures in the study area (Alves, 2008; Batista et al., 2009). The assignment of habitat categories followed the categories defined in Task 2 and each species could be assigned into more than one depth interval or sediment type (e.g. species that occur in all depth intervals and sediments were integrated into all categories). For spawning and nursery habitats, the “deep” category was limited to 50-100 m deep. Qualitative abundance levels were assigned to rocky reef species based on densities obtained by Henriques et al. (2013b) and previous qualitative abundance levels assigned by (Henriques et al., 1999). For soft substrate species, catches per unit of effort obtained in reference studies were used (Table 4.1). The remaining functional guilds followed FishBase classifications (Froese and Pauly, 2014). Functional guilds attributed to each species are shown in Table 4.B1.

Task 4: Habitat characterization: functional guilds and pressure level per habitat

This task is an integration of the information collected in order to assess the importance of each habitat for functional guilds (and species) and to understand the extent to which pressures can influence habitats and their structural functions in the ecosystem. Firstly, each species and respective functional classifications were assigned to its habitat (separately for each life stage considered) and secondly, pressure information was superimposed to habitats using a geographic information system (GIS). Categorical functional groups were transformed to a numerical scale, where 1 was attributed to the lowest level, in order to obtain averaged values for each functional group, per habitat. Furthermore, percentage of: species with small body size, species with medium body size, non-migratory species, territorial and sedentary species, species with medium mobility, rare and uncommon species, high value species, species with medium-high commercial value, bycatch species, species with very low and low resilience and species with non-planktonic larvae were calculated per habitat type and added as additional metrics to the analyses.

Both the averaged functional guild levels and the additional metrics were added to the attribute table of the habitats shapefile, which was then intersected with the pressures shapefile (vectorial information). A sampling grid of 50 x 50 m was then applied over the previous shapefiles, and values for pressure and averaged guild levels and additional metrics of 500 grid units per habitat were extracted. Finally, unconstrained Principal Coordinates Analysis (PCO; Anderson et al., 2008) was applied to these datasets in order to understand patterns of functional guild distribution per habitat type and the influence that human pressures may have on this habitats-guild relationship. PCO were performed separately for the three life stages considered (adults, spawning and nursery).

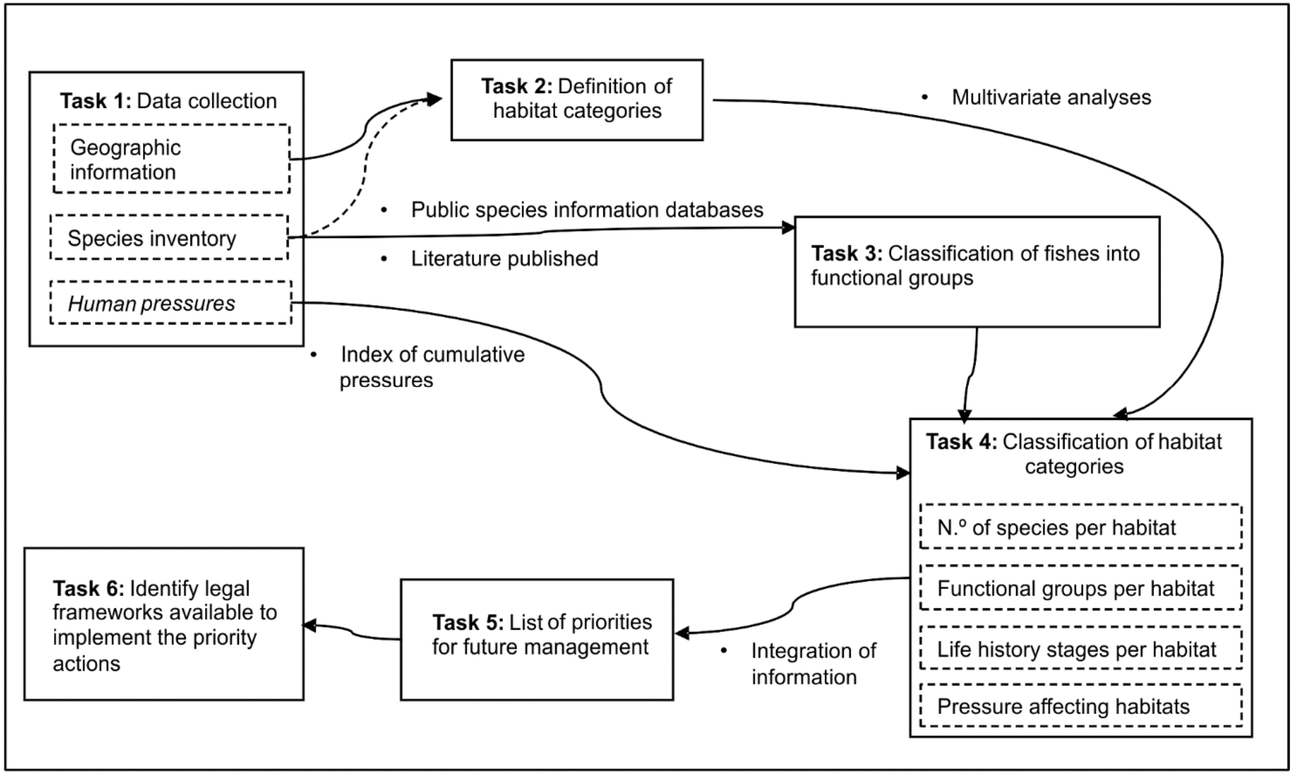

Figure 4.2 Scheme of the framework developed for the rapid assessment of the potential for effectiveness and for the identification of major gaps of already implemented MPA. Boxes represent the main task of the framework and arrows indicate the flow between information. The main methods used to fulfill each task are placed near the arrows.

Table 4.1 References of research publication used to inventory fish species occurring in Arrábida MPA. A resume of sampling characteristics are also shown including short descriptions of method, habitat, number of replicates and periods sampled.

Reference	Sampling characteristics			
	Method	**Habitats**	**Replication**	**Period**
Henriques et al. (1999)	Underwater visual census (UVC)	From the surface to the limit of the rocky substrate, cryptobenthic habitats, interface rocky-sand	26 dives yr^{-1}. Overall MPA.	Monthly, from May 1992 to December 1998
Beldade et al. (2006)	Bottom trawls with a plankton net	Rocky nearshore	3 days $week^{-1}$, 3 dives day^{-1}. 81 samples. Localized.	July 2002
Borges et al. (2006)	Sub-superficial trawls and bottom trawls with plankton net	Rocky nearshore	9 to 12 days yr^{-1}, 4 dives day^{-1}, 30-54 samples yr^{-1}. Localized.	Monthly/ seasonally 1999-2003,
Borges et al. (2007)	Sub-superficial trawls with plankton net	Rocky nearshore, pelagic	9 days, 400 samples. Localized.	Monthly, May-October 2000
Alves (2008)	Trammel nets commercial fishing	Soft substrates (sand and mud)	20 days, 37 sets x 15200 m. Several sites covering most of the MPA.	Seasonally, August 2007 to April 2008
Sousa (2011)	Trammel nets fishing sampling	Soft substrates (sand and mud) – 12 to 45m	43 days, 107 sets x 500m. Most of the MPA. Several sites covering different levels of protection.	Seasonally, December 2007 to April 2010
Henriques et al. (2013b)	Underwater visual census (UVC)	Rocky substrate, cryptobenthic habitats, interface rocky-sand	36 days yr^{-1}, 2 dives day^{-1}, 72 dives yr^{-1}. Several sites covering different levels of protection.	Monthly, May 2010 to February 2011
Horta e Costa et al. (2014)	Underwater visual census (UVC)	From the surface to the limit of the rocky substrate, Cryptobenthic habitats, interface rocky-sand	30 dives yr^{-1}, beginning in the sandy area 10 m beyond the rocky substrate and ending at the intertidal. Several sites covering the overall MPA.	Monthly, from May 1992 to December 2002; Monthly, 2010

Table 4.2 Number of species per habitat category and functional guilds considered. Categories included per functional guild are also shown.

Functional guilds	Rocky substrate (R)			Sandy and Muddy substrates (SS)		
	Deep (>50 m)	Mid (10-50m)	Shall (<10 m)	Deep (>50 m)	Mid (10-50m)	Shall (<10 m)
Position in water column						
Bathydemersal	0	0	0	1	0	0
Benthopelagic	16	15	11	16	14	12
Demersal	24	40	53	50	62	51
Pelagic	13	15	9	14	16	10
Reef-associated	5	16	13	3	3	2
Migration type						
Amphidromous	0	0	2	0	0	3
Anadromous	1	1	1	1	1	1
Catadromous	0	3	1	0	3	2
Non-migratory	29	49	56	55	58	43
Oceanodromous	28	33	26	28	33	26
Trophic group						
Herbivores	1	1	1	1	1	1
Invertebrate feeders	10	22	27	21	22	19
Macrocarnivores	32	35	25	45	46	35
Omnivores	9	20	25	9	18	13
Piscivores	1	1	1	2	2	1
Zooplanctonivores	5	7	7	6	6	6
Trophic level						
2-3	4	8	14	5	9	9
3-4	41	65	62	62	70	54
>4	13	13	10	17	16	12
Life span (years)						
< 5	5	13	22	5	7	13
5-10	16	27	29	28	33	24
10-20	19	25	19	30	33	22
20-30	6	10	9	9	12	9
> 30	10	9	6	8	6	3
Unknown	2	2	1	4	4	4
Generation time (years)						
< 2	7	15	24	9	13	19
2-4	20	34	32	37	42	28
4-6	11	14	12	14	15	9
4-10	11	14	14	13	15	12
> 10	7	7	3	7	6	3
ni	2	2	1	4	4	4

Table 4.2 (continued)

Functional guilds	**Rocky substrate (R)**			**Sandy and Muddy substrates (SS)**		
	Deep (>50 m)	**Mid (10-50m)**	**Shall (<10 m)**	**Deep (>50 m)**	**Mid (10-50m)**	**Shall (<10 m)**
Body size						
Large (>50 cm)	31	35	23	40	42	28
Medium (26-50 cm)	19	26	23	29	34	26
Small (<25 cm)	8	25	40	15	19	21
Mobility						
High mobility	30	36	28	29	34	29
Medium mobility	23	38	28	48	51	31
Sedentary	3	5	8	4	5	9
Territorial	2	7	22	3	5	6
Biogeographic affinity						
Cold-temperate	5	13	13	5	7	7
Eurythermic	7	8	5	7	7	5
Temperate	24	29	25	39	41	32
Tropical	6	9	10	11	12	12
Warm-temperate	16	27	33	22	28	19
Larvae						
Unknown	22	32	32	35	37	24
Attached to parental body	0	0	2	0	0	2
In brood pouch	1	1	2	1	1	1
In close association with substrate	2	3	5	2	2	4
Planktonic	33	50	45	46	55	44
Reproductive guild						
Unknown	23	27	25	25	26	21
Oviparous brooders	2	2	6	2	2	5
Oviparous guarders	1	13	23	0	0	4
Oviparous nonguarders-generalist	28	39	27	49	59	37
Oviparous nonguarders-hiders	2	3	3	2	2	2
Viviparous/ ovoviviparous	2	2	2	6	6	6
Commercial Value						
Nule or very low	16	35	52	25	30	33
Medium	16	23	17	22	26	20
High	26	28	17	37	39	22
Level of exploitation						
Exploited, bycatch	13	21	16	24	25	15
Exploited, medium-high commercial value	39	47	31	51	56	37
Unexploited	6	18	39	9	14	23

Table 4.2 (continued)

Functional guilds	Rocky substrate (R)			Sandy and Muddy substrates (SS)		
	Deep (>50 m)	**Mid (10-50m)**	**Shall (<10 m)**	**Deep (>50 m)**	**Mid (10-50m)**	**Shall (<10 m)**
Resilience to fishing pressure						
High	2	8	20	7	9	10
Low	11	14	6	16	17	5
Medium	41	60	56	55	63	54
Very low	4	4	4	6	6	6
Qualitative abundance						
Very common	5	11	11	8	11	9
Common	17	26	24	25	29	22
Uncommon	15	23	19	22	25	18
Rare	21	26	32	29	30	26
Adult life (non-breeding periods) habitats	58	86	86	84	95	75
Reproduction habitats (spawning)	8	60	56	22	71	30
Nursery habitat	3	44	77	11	64	47

Task 5: Guidelines to future management of the MPA: identify problems, possible solutions and list priority actions

Conclusions derived from the previous tasks should be synthetized in order to identify the main problems affecting MPA effectiveness. Problems were identified based on the objectives established for the MPA (except for those not directly dependent from the MPA design and location). For each problem, a list of potential solutions and essential actions need to be drawn. With this final list of problems and their solutions, managers will be able to plan the future directions of a MPA. For instance, at this point managers have a much clearer idea about the level of protection per habitat type, the most important habitats for each life stage and they can extract information regarding species or life stages that are not being adequately protected by the MPA. The ultimate target is to establish a list of priorities to guide the future direction of MPA management. Results can range from total inadequacy (objectives will not fulfilled) to total adequacy of a MPA (MPA is adequate, considering its goals).

Task 6: Identification of frameworks available to support the implementation of priority actions

This task consists of finding the frameworks available to support the implementation of the priority actions identified in task 5. These frameworks can be of different natures: legal (if new regulations are needed for the study area), economical (if the priorities depend on available budget to implement survey procedures, technical measures to reduce impacts or to improve enforcement actions), social (if there is a need to improve public awareness and compliance towards the MPA) and scientific (if priorities need input and support from scientific expertise). Considering that these fall out of the main scope and objectives of this case study, tasks 5 and 6 were only superficially addressed.

RESULTS

The results obtained for the present case study are grouped by task for convenience.

Tasks 1 to 3:

Considering the case study, a list of 157 fish species, from 97 genus and 53 families were collected from 8 publicly available studies (Table 4.1). Regarding qualitative abundance, 53 species were rare, 40 were uncommon, 46 were common and 18 were very common in the study area. Near 20% of rare and uncommon species have trophic levels higher than 4, while only 3% of very common and common species belong to this higher trophic level. Most of the species listed are macrocarnivores or invertebrate feeders and only one species is herbivore (*Sarpa salpa*). All species and their classification into functional guilds are provided in Table 4.B.1.

For the purpose of the present study, habitat categories were defined trough the combination of bathymetry and sediment type (from benthic habitat maps). Given this, six habitats categories were delimited: rocky substrates shallower than 10 m (R-shall), between 10 and 50 m (R-mid) and deeper than 50 m (R-deep) and soft substrates (sandy and muddy substrates) shallower than 10 m (SS-shall), between 10 and 50 m (SS-mid) and deeper than 50 m (SS-deep). Figure 4.1 shows the spatial extent of habitats and the different pressure levels to which they are exposed within the study area. Regarding pressure levels per habitat type, mid and deep categories were widely affected by medium/high pressure levels, while shallower habitats had more than 50% of their extent under very low or low pressure levels (Figure 4.3). Considering averaged pressures per habitat (Table 4.3), the highest values were obtained for deep habitat classes, while the lowest pressure levels correspond to shallow habitats. In general, pressure levels followed the logic of MPA zoning, with higher pressures affecting areas of lower protection (buffer areas, BA) and lower pressure levels in the no-take zone (FPA). Nevertheless, pressures affecting the coastal strip of PPA (within 200 m from shore) were very low, similar to the observed within the no-take zone, while the remaining PPA extent was affected by intermediate pressure levels (longlines and traps are allowed only farther than 200 m from shore). The highest pressure levels were identified within the BA and PPA (except the 200 m coastal strip), in areas close to the Sesimbra port.

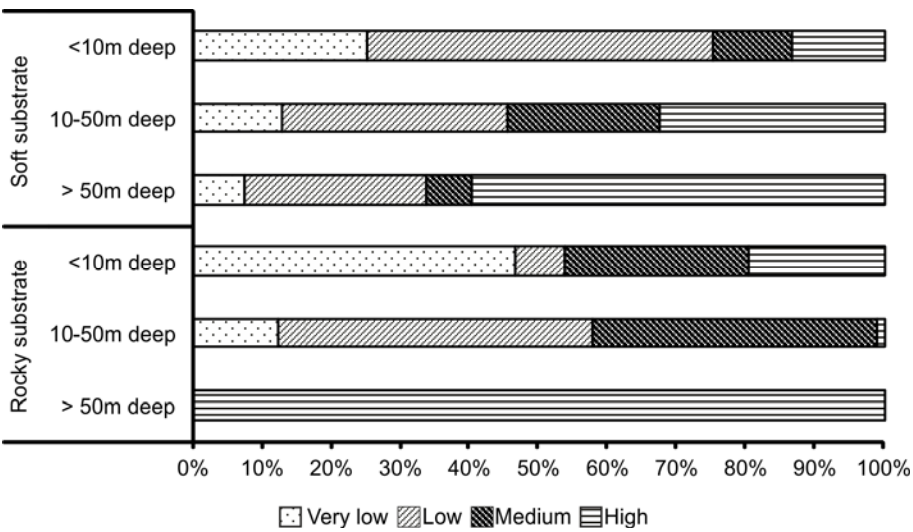

Figure 4.3 Proportion of the Arrábida MPA under very low, low medium or high pressure levels per habitat category.

Rocky substrates represent only 12.50% of the MPA, almost all from mid and shallow categories (Table 4.3, Figure 4.3). Despite their relatively small extent, R-shallow plays a major role as a nursery ground (77 species) and reproduction habitat (56 species) for fish within the MPA. Mid-depth habitats had the largest extent of all habitat categories considered (24.3 km^2 for SS and 4.12 km^2 for R), supporting a higher species richness for adult and reproduction stages and being also important as nursery for several species. Deep habitats showed the lowest species richness, namely for reproduction and nursery stages. SS-shall also had a relatively lower importance for all life stages (Table 4.3, Figure 4.4).

Table 4.3 Habitat characteristics and number of species per life history phase for each habitat category considered in Arrábida MPA case study. Average pressure (from 1 – low pressure - to 4), area occupied by each habitat and their relative importance are shown. Number of species and Number of species per km^2 potentially using habitats during their adult (Ad), spawning (Sp and nursery (Nu) phases are also shown.

Habitat category	Habitat characteristics			Number of species per life history phase					
	Average Pressure	Area (km^2)	Relative importance (%)	Ad	Sp	Nur	Ad/km^2	Sp/km^2	Nur/km^2
Rocky substrate > 50m deep	4.00	0.11	0.22	58	8	3	-	-	-
Rocky substrate 10-50m deep	2.30	4.12	8.12	86	60	44	20.89	14.57	10.69
Rocky substrate <10m deep	2.19	2.10	4.15	86	56	77	20.89	13.60	18.70
Soft substrates > 50m deep	3.18	9.81	19.34	84	22	11	20.40	5.34	2.67
Soft substrates 10-50m deep	2.74	24.30	47.91	95	71	64	23.07	17.24	15.54
Soft substrates <10m deep	2.12	10.27	20.26	75	30	47	18.21	7.29	11.41

From a general point of view, this MPA contains potential habitats for the entire life cycle of 120 of the 157 species listed, among which nearly 57% are rare or uncommon in the MPA. For 8 of the species listed, MPA habitats supported only adult and spawning stages (AS), while for another 8 species spawning is not likely to occur in the MPA. Two Soleidae (*Solea solea* and *Solea senegalensis*) and four Mugilidae (*Chelon labrosus*, *Liza aurata, Liza ramada* and *Liza saliens*) have their nursery areas inside estuaries while *Pollachius pollachius* and *Scyliorhinus canicula* have their potential nurseries on rocky substrates deeper than the area covered by the MPA. Spawning habitats not present in the MPA (for the species listed) were also estuaries (for *Argyrosomus regius*) and deeper substrates (for *Macroramphosus scolopax*, *Merluccius merluccius, Raja miraletus*, *Synapturichthys kleinii, Pagellus bogaraveo*, *Conger conger* and *Muraena helena*). In addition, both spawning and nursery habitats for 15 species were likely to occur outside the MPA (*Alosa fallax*, *Atherina boyeri*, *Lophius budegassa*, *Micromesistius poutassou*, *Mugil cephalus*, *Phycis phycis*, *Pomatoschistus microps*, *Pteromylaeus bovinus, Rostroraja alba*, *Sarda sarda*, *Serranus atricauda*, *Sphoeroides marmoratus*, *Trachurus mediterraneus*, *Trachurus picturatus* and *Trachurus trachurus*). Spawning habitats were unavailable in the literature for six species and nursery habitat information was missing for one species. Only one viviparous species occured (*Carcharhinus plumbeus*).

Shallower rocky substrate (R-shall) was the habitat supporting the highest number of habitat-specific species, mostly from families Blenniidae and Gobiesocidae (i.e. species for which this is the only suitable habitat for most of their life cycle). Additionally, this habitat was also highly important for spawning and nursery phases of Gobiidae and Syngnathidae. Shallow and mid rocky habitats support the three life stages of most *Serranidae* and *Labridae* species.

Some species, most with important commercial value, depend specifically on SS-mid on their spawning stage (e.g. *S. senegalensis*, *S. solea*, *S. rhombus*, *R. undulata* and *S. lusitanica*). This habitat was also important as a nursery ground for *R.undulata*, *R. montagui*, *S. rhombus*, *B. luteum* and *S. lusitanica*.

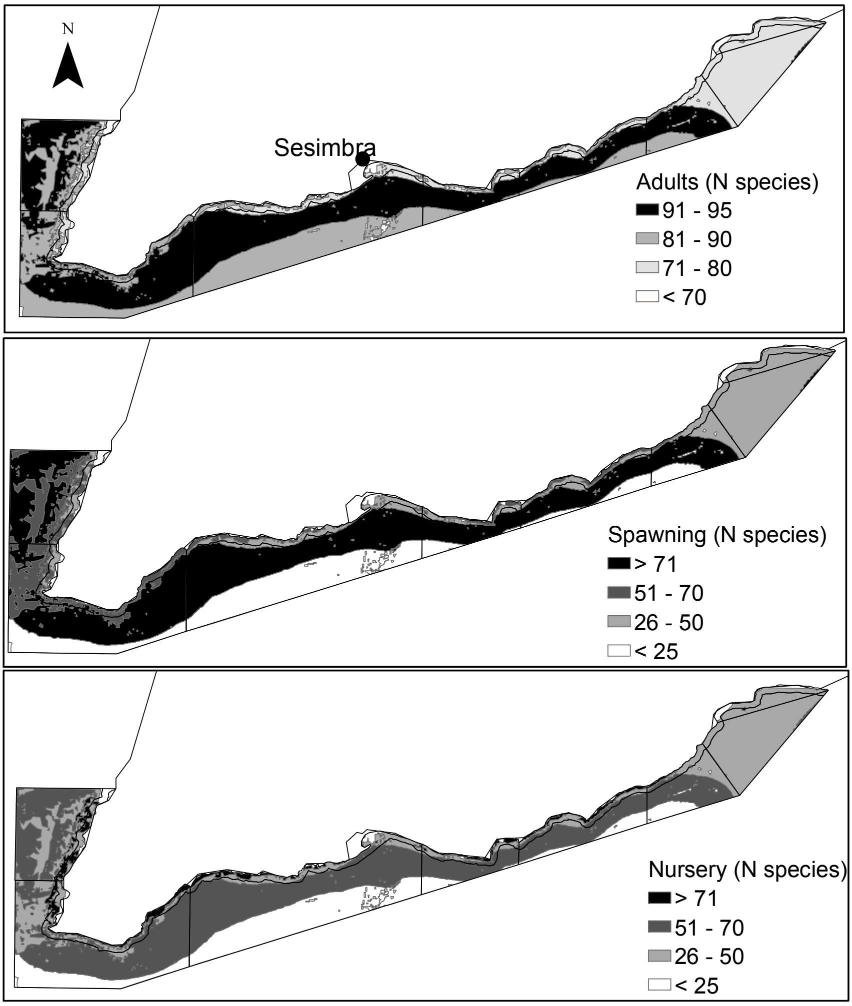

Figure 4.4 Spatial distribution of different levels of species richness in Arrábida MPA, considering the typical habitats used in three different phases of species life history: Adult life (a), Spawning (b) and Nursery (c).

Task 4:

The PCO performed to analyse habitat preferences for the adult (non-breeding) stages showed that species composition (and consequently functional characteristics) were firstly differentiated by substrate type (rocky and soft substrates), being more similar between SS-mid and SS-deep and between R-mid and R-deep (Figure 4.5). Nursery habitats, species with small body length, more resilient species, territorial and sedentary species and species with non-planktonic larvae had a stronger connection with R-shall habitats, while almost all the other variables considered, namely the guilds associated with commercially targeted species (e.g. higher commercial value, higher percentage of exploited species, percentage of species with medium body size), are associated with mid and deep habitats. It is also important to highlight that highly vulnerable species, with longer generation time, were also more associated with deeper habitat categories. Lower pressure levels were related to shallower habitats while higher pressures affected mostly mid and deep habitat categories (Figure 4.5).

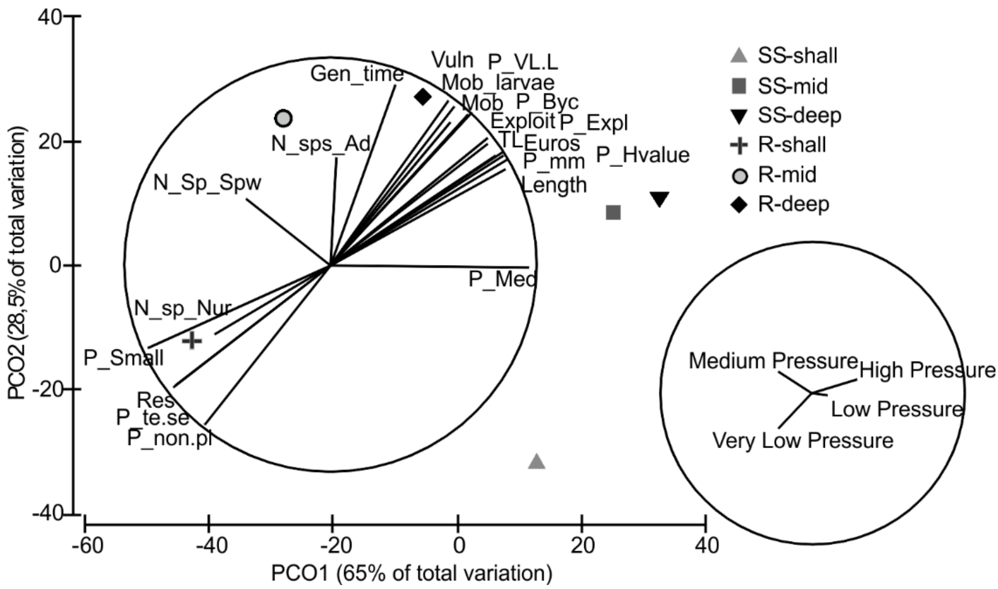

Figure 4.5 Principal Coordinates Analysis (PCO) showing the relationship of functional groups per habitat supporting adult phase (circles represent vector correlations of 1). Also shown in the bottom right is the equivalent ordination plot with the position of pressure levels. Only functional groups with correlation higher than 0.5 are shown: N_sp_Nur- Number of species occurring during nursery phase; N_sp_Spw - Number of species occurring during spawning phase; N_sps_Ad - Number of species occurring during adult phase; P_small – Percentage of species with small body size; P_Med – Percentage of species with medium body size; Res – Number of high resilient species; P_te.se – Percentage of territorial and sedentary species; P_non.pl – Percentage of species with non-plantonic larvae; Vuln – Species vulnerability; Mob_larvae – Level of larvae mobility; Mob – Species mobility; P_VL.L – Percentage of species with low and very low resilience to fisheries; P_Byc – Percentage of bycatch species (with economic value); Exploit – level of species exploitation; P_expl – Percentage of exploited species; TL – trophic level; Euros – Species economic value; P_Hvalue – percentage of high value species; P_mm – percentage of species with medium mobility; Length – Species length.

The main patterns found by the PCO regarding spawning and nursery stages showed some general common patterns (Figures 4.6 and 4.7). The habitats were grouped differently when compared to adult stages, with three distinct “clusters”: R-shall and R-mid, SS-mid and SS-shall and SS-deep and R-deep, showing that species are more depth-specific during these phases. However, variables characteristic of each cluster were in general similar to the observed for adult stages (Figures 4.6, 4.7). During these two stages, deep habitats appear to acquire more importance to rare and uncommon species, species that are usually bycatch and species occupying higher trophic levels. Pressure levels were also similarly placed in the multivariate space both in spawning and nursery stages and highlight the trends previously described for adults.

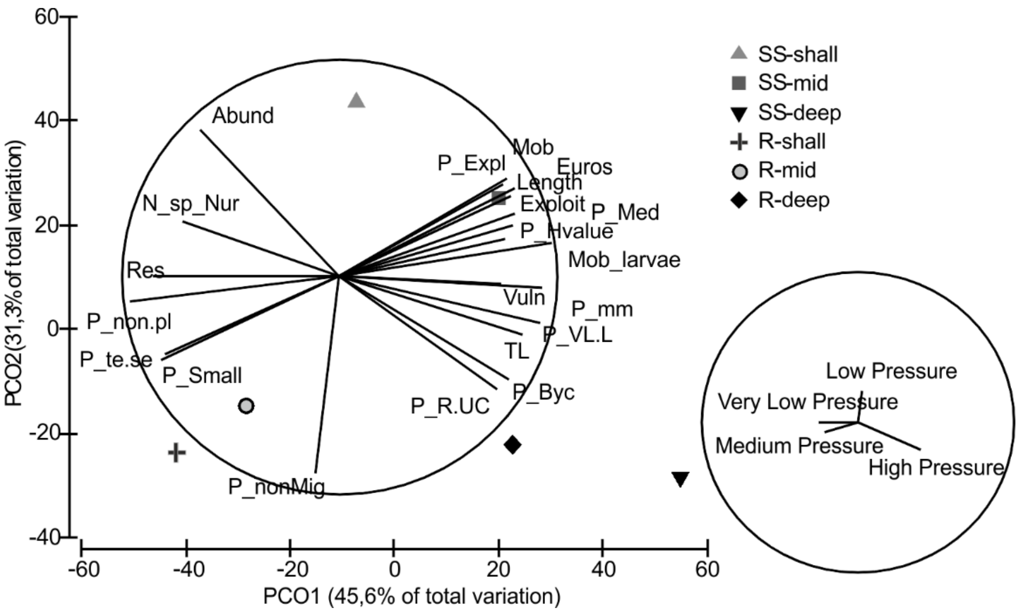

Figure 4.6. Principal Coordinates Analysis (PCO) showing the relationship of functional groups per habitat supporting spawning phase (circles represent vector correlations of 1). Also shown in the bottom right is the equivalent ordination plot with the position of pressure levels. Only functional groups with correlation higher than 0.5 are shown: Abund – Species qualitative abundance; N_sp_Nur- Number of species occurring during nursery phase; P_small – Percentage of species with small body size; P_Med – Percentage of species with medium body size; Res – Number of high resilient species; P_te.se – Percentage of territorial and sedentary species; P_non.pl – Percentage of species with non-plantonic larvae; Vuln – Species vulnerability; Mob_larvae – Level of larvae mobility; Mob – Species mobility; P_VL.L – Percentage of species with low and very low resilience to fisheries; P_Byc – Percentage of bycatch species (with economic value); Exploit – level of species exploitation; P_expl – Percentage of exploited species; TL – trophic level; Euros – Species economic value; P_Hvalue – percentage of high value species; P_mm – percentage of species with medium mobility; Length – Species length; P_R.UC – Percentage of rare and uncommon species.

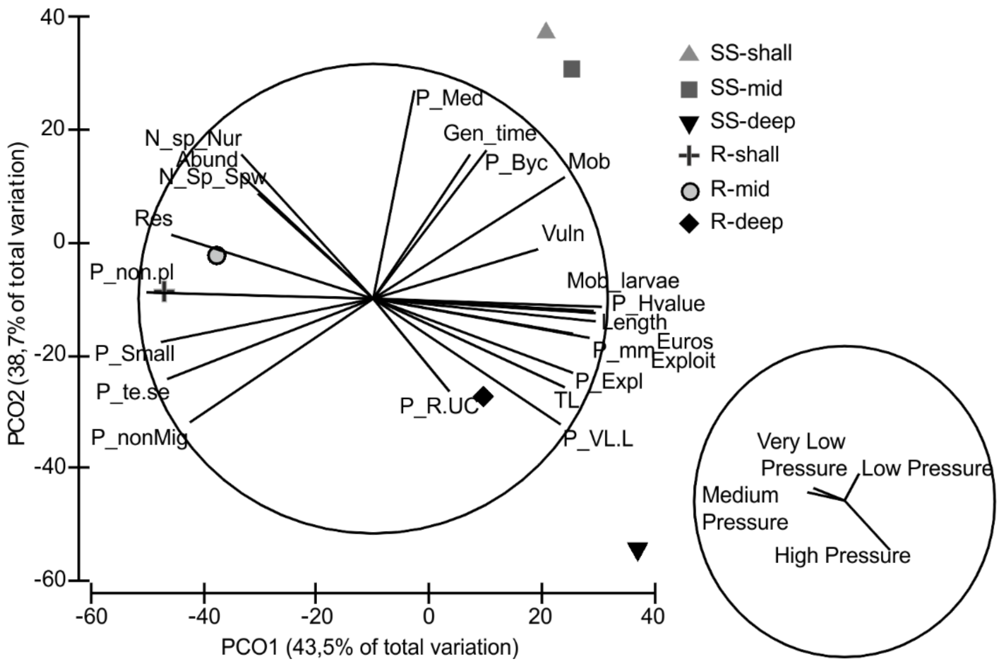

Figure 7. Principal Coordinates Analysis (PCO) showing the relationship of functional groups per habitat supporting nursery phase (circles represent vector correlations of 1). Also shown in the bottom right is the equivalent ordination plot with the position of pressure levels. Only functional groups with correlation higher than 0.5 are shown: Abund – Species qualitative abundance; N_sp_Nur- Number of species occurring during nursery phase; N_sp_Spw - Number of species occurring during spawning phase; P_small – Percentage of species with small body size; P_Med – Percentage of species with medium body size; Res – Number of high resilient species; P_te.se – Percentage of territorial and sedentary species; P_non.pl – Percentage of species with non-plantonic larvae; Vuln – Species vulnerability; Mob_larvae – Level of larvae mobility; Mob – Species mobility; P_VL.L – Percentage of species with low and very low resilience to fisheries; P_Byc – Percentage of bycatch species (with economic value); Exploit – level of species exploitation; P_expl – Percentage of exploited species; TL – trophic level; Euros – Species economic value; P_Hvalue – percentage of high value species; P_mm – percentage of species with medium mobility; Length – Species length; P_R.UC – Percentage of rare and uncommon species; Gen_time – Species generation time.

Tasks 5 and 6:

Based on the results of previous tasks and regarding the exemplification of the practical applicability of the present framework, positive results, potential problems and possible solutions and priority actions are synthesized in table 4.4.

Regarding task 6, only general considerations are possible, since the specific frameworks and opportunities are highly dependent on factors only perceptible by managers at a given time (e.g. political will, budget available, human resources).

Table 4.4. Integration of the results obtained from the implementation of the framework (task 5). Positive results and problems identified for the Arrábida Marine Park, according to their objectives are described. Suggestions for possible solutions and actions are also presented.

MPA objectives	Positive results	Problems identified	Possible solutions and actions
Preserve marine biodiversity	Very low - low pressure levels in shallower habitats Low-medium pressure levels in mid habitats MPA includes suitable habitats for whole life history phases of 120 species Rocky shallow habitats (<10m deep) account with the entire life history phases of 39 species (plus 7 species with broader use of habitats)	High pressure levels in deep habitats Life history phases of 31 fish species were not totally embraced by MPA habitats	Minimization of pressures in deep habitats Increase MPA limits in order to include deeper habitats Provide protection of nearby estuarine areas Analyse connectivity between habitats Account with larval dispersal
Sustainable development through the promotion of economic and cultural regional activities, such as the traditional “lines and hooks” fishery (i.e. artisanal fisheries with longlines and jigs)	Life cycle of most important target species (e.g. Sparidae) have suitable habitats inside MPA. Shallow habitats (low pressure levels) are important for nursery of some commercial species (mostly *Sparidae*) Shallow and mid habitats are important for spawning of commercial species	Habitats suitable for nursery and spawning of some target species are not available within MPA limits (e.g. for spawning of *A. regius* and *C. conger*) Exploited invertebrates, e.g. *O. vulgaris*, *Sepia officinalis* and *Loligo vulgaris* (important target species of jigs fishery) were not included in this study	Collect data on fisheries catches and fishing effort Evaluate if spillover/ migration events occur Assess relation between fishing métiers Local fishers consulting Assess the impact of outside MPA pressures Increase MPA limits in order to include deeper habitats Provide protection of nearby estuarine areas

*AMP objectives not related with the present framework are not shown (e.g. sustainable nature tourism)

Nevertheless, it seems like the case study MPA has the potential to protect at some level many of the species occurring in the area, mainly near-shore reef species. For the remaining species, clear goals must be established to understand the best approach to follow (e.g. increase enforcement, increase spatial extension) and thus the management actions required.

DISCUSSION

Despite their worldwide implementation, MPAs have often seen their effectiveness compromised by multiple factors, namely inadequate planning and inefficient management and enforcement (Agardy et al., 2011). The basis of low MPA effectiveness, regardless of the implementation phase, is often the lack of quality scientific information to support planning actions and decisions (Fraschetti et al., 2002; Mosquera et al., 2000), together with the inadequate conduction of processes (e.g. by implement co-management or participatory management, therefore minimizing negative social impacts) (Agardy et al., 2011). However, collecting these data requires long term sampling and adequate spatial replication, which is a costly approach, requiring high expertise in different fields. In addition, even when well-designed surveys are implemented, their results are usually available after several years of MPA implementation and comparisons with a "before" scenario are rare due to lack of data (Fenberg et al., 2012; Horta e Costa et al., 2013a). In this context, the approach hereby presented is mostly important as an alternative or a complementary decision-support tool for MPA management at an early stage, when no regular assessment plans are implemented and when planning was poorly based in scientific information. The method integrates the best ecological data published for a given area with the best scientific knowledge on species life history in order to quantify the potential effectiveness of a given MPA. This is mainly important for highly impacted areas where scientific knowledge is sparse and disconnected, despite the recognized biological importance of the area, which is a frequent reality. For instance, in Europe many "Special Areas of Conservation (SACs)" on the marine environment (Natura 2000 network) have been designated under the Habitats Directive (EU Council Directive 92/43/EEC), although, besides general guidelines, no effective management or assessment practices are in place for the majority of them (Pullin et al., 2009). Despite known limitations (e.g. lack of capability to account for the complex nature of ecological interactions and identify spatial and temporal variations), the use of published data can be very useful for coastal planning and management, when accompanied by cautious interpretations and a careful assessment of data uncertainty (Pais et al., 2012). This type of framework can provide general directions for management but decisions leading to higher socio-economic impacts (e.g. large-scale fishing closures) should of course be based on more accurate research.

The framework here developed does not substitute the need to have well-designed surveys as soon as possible, but allows managers to make the best possible management decisions given what is known at the time, while taking into account a wide perspective on species life history parameters. It also contributes to avoid situations of long term “paper reserves” and sustained inefficiency of a MPA, a problem highlighted by several authors (Fenberg et al., 2012; Guidetti et al., 2008; Spalding et al., 2008; Wood et al., 2008). In fact, several authors defend that “paper reserves” and inefficient reserves can be even worse than the complete absence of protection, since they often create social friction without producing any benefits, thus transmitting the wrong message to coastal communities and stakeholders (Fenberg et al., 2012; Guidetti et al., 2008).

The present approach is particularly useful in the context of adaptive management, since management decisions balance the requirements of management with the need to learn about the system being managed (McCarthy and Possingham, 2007). Adaptive management has been frequently considered appropriate in the field of marine management (Grafton and Kompas, 2005; McCarthy and Possingham, 2007; McCook et al., 2010) and successful cases, such as the Great Barrier Reef MPA, have been described (e.g. McCook et al., 2010). Nevertheless, most of these studies rely on passive adaptive management (learning from past successes and failures), while active adaptive management would be the most effective approach (deliberate experimentation and carefully designed monitoring to measure and improve management effectiveness), despite all the intrinsic limitations of the latter, as discussed by Ban et al. (2012). In this sense, and depending on several context-specific situations (e.g. economical resources), the present approach can be used not only to help managers make early decisions, but also to define which are the main challenges and uncertainties facing MPA effectiveness and thus focus efforts and resources on the priorities, namely in the implementation of surveys and experiments.

In the case study of the Arrábida MPA, since one of the objectives is to “promote economic and cultural regional activities, such as the traditional lines and hooks (i.e. longlines, jigs) fishery”, an experiment including local fishers and a controlled fishing effort could be implemented in some of the main fishing areas lost by fishers (Horta e Costa et al., 2013b) in order to evaluate alternative protective measures for partially protected areas, such as seasonal openings. Results of this type of experiment could contribute to positively adapt management measures and also to improve the compliance of stakeholders, contributing to the effectiveness of the MPA, as the inclusion of stakeholders during planning phases is key to minimize future conflicts and social problems (Dudley, 2008). Furthermore, this type of co-management practices have proven to be efficient in several MPA around the globe (see e.g. Guidetti and Claudet, 2010). In fact, the consultation of local stakeholders during the last tasks of the proposed framework would be advantageous, since their knowledge and concerns could greatly improve the assessment of priorities for management and the adjustment of limits and regulations. If stakeholders are involved in MPA management and identify the potential benefits of protected areas, their inclusion as data collection agents could also be an effective and low-cost method of obtaining useful information. Volunteer monitoring projects are becoming common around the world and can play a role in achieving higher rates of MPA effectiveness (see e.g. Danielsen et al., 2005; Léopold et al., 2009; Lloret et al., 2012; Lopes et al., 2013; Obura, 2001).

The combination of species richness and species life history, through the identification of key habitats used during life history phases, spatial distribution of available habitats and their overlap with different pressure levels in the study area revealed to be a step forward in the assessment of potential effectiveness of a MPA. The main underlying assumption of the proposed framework is that protecting key habitats for adults (e.g. feeding, refuge), reproduction (e.g. spawning) and nursery leads to a greater effectiveness of a given MPA in terms of species conservation. This is of course a simplified view of ecosystem functioning, but nonetheless revealed high potential to be used as basal knowledge. Results showed that, for some of the most important target species, no suitable nursery habitats are protected, namely estuarine grounds (e.g. for *S. senegalensis*, *S. solea*), which means that the potential protection provided for adult and spawning phases is compromised, since species lack protected nursery grounds

(Tanner et al., 2013; Vasconcelos et al., 2011). In the long run, this can lead to failure of both of the MPA objectives, biodiversity conservation and the enhancement of local traditional fisheries. Furthermore, if habitat for a given life history phase is present but mostly under high anthropogenic pressure, or if it is present only in small extensions, it is unlikely that its function will be fulfilled. For instance, a species with deep rocky spawning habitats and nurseries outside the MPA are unlikely to be effectively protected by the Arrábida MPA, since the habitat suitable for spawning has a very low extension and is under high pressure levels. If the goal is to protect overall biodiversity, it is likely that deep rocky habitats need higher protection in order to increase the potential of the MPA to protect species requiring such habitats.

The consideration of a species' function in the ecosystem is also an important feature of the present approach, since the characteristics of individual species are linked with community-level responses and have revealed to be very useful to understand changes due to human-induced pressures, such as fishing (Henriques et al., 2013a; Henriques et al., 2014). Several authors have criticized the use of species richness and abundance *per se* for conservation planning, suggesting that assessing the functional roles played by species is key in the assessment of ecosystem integrity (Frid et al., 2008; Halpern and Floeter, 2008; Jennings et al., 1999; Parravicini et al., in press). The present approach identifies the functional roles that are supported by each habitat, as well as their vulnerability (i.e. level of pressure affecting habitats), rather than simply the number of species occurring. In addition, predicting the potential response of a given functional guild to stress can be applied to an area independently of species or location, since functional guild abundances tend to be more resistant to natural variations and vary in a more consistent manner in the face of similar pressure sources (Elliott et al., 2007; Jennings et al., 1999).

There is a widespread agreement that the use of functional guilds to assess the health of biological communities, rather than taxonomic-based methods alone, has deep implications in the overall understanding of ecosystem functioning, as they ultimately represent species adaptations to the environment (Bremner, 2008; Henriques et al., 2014). In this context, knowing the effects that MPAs could have on the structure and function of communities strengthens the need for an approach such as the one presented, together with more traditional ones, in the planning phase of MPAs, to ensure that their boundaries include all areas and protection levels needed to safeguard the recovery and maintenance of a healthy system. It is important to highlight, however, that it is extremely necessary that this transposition is carefully implemented, since the complexity of ecosystem interactions is hard to predict and several factors can influence community responses to protection (Claudet et al., 2010; Micheli et al., 2004).

Besides ecological health, stakeholder compliance, level of enforcement and fishing effort around the MPA boundaries are some of the factors likely to influence the response of biological communities to protection (Dudley, 2008; Fenberg et al., 2012; Guidetti et al., 2008; Guidetti and Claudet, 2010; Rice et al., 2012; Stelzenmüller et al., 2008). For instance, results obtained for the study area showed that high trophic level species are rare, which would be expected in exploited systems (Henriques et al., 2013a; Henriques et al., 2013b). The increase in abundance of predators or even the presence of new top predator species in the study area can be expected, as documented for other protected areas (Micheli et al., 2004). Results also showed a great potential for protection of territorial species in shallow habitats, with low mobility, low larval dispersal ability and low generation times (e.g. Blenniidae, Gobiesocidae, Gobiidae). However, some studies have shown that the expected benefits for such type of functional guilds were not observed, probably due to the enhancement of predator abundance in protected areas (Micheli et al., 2004; Willis and Andersen, 2003). Another finding of the present analyses was the suitability of mid and deep habitats to support nursery and spawning of some of the most vulnerable species (with longer generation times), although these habitats are under medium to high pressure levels, which could contribute to the inefficiency of the MPA to this stages and thus compromise the biodiversity conservation objective.

As referred to previously, the study area was chosen to exemplify the applicability of the theoretical framework. Thus, replace or criticize the present management practices implemented on the Arrábida MPA were not aims of the present study. Despite the undeniable utility of the results obtained, the study area already has several monitoring actions implemented and a number of research projects have been collecting biological and fisheries data accessible for management (Abecasis, 2013; Abecasis et al., 2014; Henriques et al., 2013a; Horta e Costa et al., 2013b). The initial problem with the lack of management and enforcement (from 1998 to 2005) is being gradually overcome in recent years, which has led to a MPA that is now fully transferred from paper to practice. In fact, the general findings obtained with this framework are in accordance with the results of the abovementioned studies (e.g. the importance of neighboring estuaries for some species).

Given the above considerations, it is important to realize the limitations of this framework in order to take advantage of its application. Otherwise, the implementation of such "ecosystem simplification" approaches is dangerous. The present approach does not allow for a fine-scale evaluation of MPA effectiveness, but helps in the establishment of general directions and priorities for early management decisions in cases where scientific research is scarce and/or MPA planning was not adequate. Future implementations of this framework would benefit from the inclusion of abundance data and more accurate fish species distribution models, as well as information on other taxonomic groups (e.g. Invertebrates, algae). Furthermore, the more accurate the knowledge on species life history (e.g. data on spawning sites for the study area) and habitat characteristics, the higher the efficiency of the framework. Besides being "low-cost", this framework is also adaptable to the planning phases of new MPA (or MPA networks) for which no research data are available, through the combination of local knowledge (e.g. location of habitats and species lists) with global knowledge on the characteristics of species and habitats.

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Table 4.B.1. Categories per functional guilds of species listed for Arrábida Marine Park. Species were characterized according to their position on water column (P- pelagic, D- demersal, BP- benthopelagic, BD- bathydemersal, RA- reef associated); migration type (A- anadromous, C- catadromous, O- oceanodromous, NM- non migrantory, A- amphidromous); Trophic group (inv- invertebrate feeders, ma- macrocarnivores, pi- piscivores, om- omnivores, zoo- zooplanktonivores, he- herbivorous); trophic level, life span, generation time, body size (L- large, M- medium, S- small); mobility (hm- high mobility, mm- medium mobility, te- territorial, se- sedentary); biogeographic affinity (T- temperate, WT- warm temperate, CT- cold temperate, Tr- tropical, Eu- eurythermic); "type" of larvae (P-planktonic, AS- in close association with substrate, BP- in brood pouch, APB - attached to parental body); Reproductive guild (ON-G- oviparous, non guarders, generalists, ON-H- oviparous, non guarders, hiders, OG- oviparous guarders, V/O- viviparous/ ovoviviparous, open water/substratum egg scatterers column, OB- oviparous brooders), commercial value (€- nule or very low, €€ - medium, €€€ - high), level of exploitation (E- exploited, medium-high commercial value, B- exploited, discarded, U- unexploited); resilience to fishing pressure (VL- very low, M- medium, H- high), qualitative abundance (L-low, M-medium, H-high) and habitats suitable for adults, spawning and nursery phases (R- rocky substrates, SS-soft substrates, S- shall (<10m), M- medium (10-50m), D- deep (>50m), out (>100m, outside MPA)).

Species	Pos.	Mig.	TG	TL	LS	GT	BS	Mob	Biog	Lv	Rep	CV	LE	Res	Ab	Adults	Spawning	Nursery
Alosa fallax	P	A	zoo	3.6	13.0	4.5	L	hm	T		ON- G	€€	E	M	R	B	R	E
Aphia minuta	P	O	ma	3.1	1.7	0.8	S	te	T	P		€	U	H	R	SS-B	SS-S-M-D-E	SS-S-M-D-E
Apletodon dentatus	D	NM	inv	3.1	5.7	2.7	S	te	WT	P		€	U	H	R	R-S	R-VS	R-VS
Argyrosomus regius	BP	O	ma	4.3	32.6	11.2	L	hm	WT			€€€	E	L	R	B	E	R-SS-VS-S-E
Arnoglossus imperialis	D	NM	ma	3.8	9.7	2.7	S	mm	Tr	P	ON- G	€	B	M	C	SS-B	SS-S-M	SS-VS-S-E
Arnoglossus laterna	D	NM	ma	3.6	20.0	7.7	M	mm	WT	P	ON- G	€	B	M	R	SS-B	SS-S-M	SS-VS-S-E
Arnoglossus thori	D	NM	ma	3.3	5.5	1.7	S	mm	WT	P	ON- G	€	B	M	UC	SS-B	SS-S-M	SS-VS-S-E
Atherina boyeri	D	A	ma	2.3	17.5	6.7	S	hm	T	AS	ON- G	€	U	M	R	B-S	E	E
Atherina presbyter	P	O	ma	3.7	4.0	1.2	S	hm	T	P	ON- G	€	U	H	UC	B-S-M	R-SS-VS-E	R-SS-VS-E
Balistes capriscus	RA	NM	inv	3.6	6.6	2.2	L	hm	Eu		OG	€€	E	H	C	R-M	SS-VS-S-M	R-SS-VS-S

Table 4.B.1. (continued)

Species	Pos.	Mig.	TG	TL	LS	GT	BS	Mob	Biog	Lv	Rep	CV	LE	Res	Ab	Adults	Spawning	Nursery
Belone belone	P	O	pi	4.2	8.5	3.1	L	hm	T	P	ON- G	€	U	M	R	B	R-SS-VS-S	R-SS-VS-S-E
Boops boops	D	O	om	3.0	15.7	4.4	M	hm	Eu	P	ON- G	€€	B	M	VC	B	R-SS-VS-S	R-SS-VS-S-E
Bothus podas	D	NM	ma	3.4	16.8	4.9	M	mm	Tr	P	ON- G	€€	B	L	C	SS-B	SS-VS-S	SS-VS-S-E
Buglossidium luteum	D	NM	inv	3.3	5.2	2.1	S	mm	WT	P	ON- G	€€	B	M	R	SS-M	SS-M	SS-M
Callionymus lyra	D	NM	inv	3.3	6.6	2.1	M	mm	T	P	ON- G	€	B	M	C	SS-B	SS-S-M	SS-VS-S-E
Callionymus reticulatus	D	NM	inv	3.3	3.3	1.3	S	mm	T	P	ON- G	€	B	M	UC	SS-B	SS-VS-S-M	SS-VS-S-E
Capros aper	D	NM	inv	3.1		ni	M	mm	T	P		€	B	M	R	B-D	R-SS-D-VD	R-SS-VS-S
Carcharhinus plumbeus	RA	O	ma	4.5	50.9	17.9	L	hm	Eu		V/O	€€	U	VL	R	B-B	viviparous	SS-VS-S
Centrolabrus exoletus	RA	NM	inv	3.5	3.9	1.6	S	mm	CT	P	OG	€	B	M	C	R	R-VS	R-VS
Chelidonichthys cuculus	D	NM	ma	3.9	6.9	2.0	M	mm	T	P	ON- G	€	B	M	UC	B-M-D	SS-S-M-D	SS-VS-S-M
Chelidonichthys lucerna	D	NM	ma	3.7	16.8	5.3	L	mm	T	P		€€€	E	L	C	B-M-D	SS-S-M-D	SS-VS-S-M
Chelidonichthys obscurus	D	NM	ma	3.4	8.4	2.4	M	mm	WT			€€	B	M	VC	B-M-D	SS-S-M-D	SS-VS-S-M
Chelon labrosus	D	C	om	2.6	23.7	7.2	L	mm	T	P	ON- G	€€	E	M	C	B-S-M	SS-VS-S	E
Chromis chromis	RA	NM	zoo	3.0	11.2	4.4	S	mm	Tr	AS	OG	€	U	M	R	R-S-M	R-SS-VS-S	R-VS-S
Ciliata mustela	D	O	inv	3.5	3.7	1.1	S	hm	CT	P	ON- G	€	U	H	R	B-S-M	R-VS	R-VS-E

Table4.B.1. (continued)

Species	Pos.	Mig.	TG	TL	LS	GT	BS	Mob	Biog	Lv	Rep	CV	LE	Res	Ab	Adults	Spawning	Nursery
Citharus linguatula	D	NM	ma	4.0	11.0	3.0	M	mm	T			€€	E	M	UC	SS-B	SS-VS-S-M	SS-VS-S-E
Clinitrachus argentatus	D	NM	inv	3.5		ni	S	se	WT			€	U	H	R	B-S	R-VS	R-VS
Conger conger	D	O	ma	4.3	42.6	15.0	L	hm	T	P		€€€	E	VL	C	B	R-SS-VD	R-SS-VS-S-E
Coris julis	RA	NM	inv	3.2	26.4	7.2	M	mm	T	P	ON- G	€	B	M	VC	B	R-VS-S	R-SS-S-M
Coryphoblennius galerita	D	NM	om	2.2	3.8	1.6	S	te	WT		OG	€	U	H	C	R-S	R-VS	R-VS
Ctenolabrus rupestris	RA	NM	inv	3.4	9.1	3.5	S	mm	T	P	ON- G	€	U	M	VC	R-S-M	R-VS	R-VS
Dasyatis pastinaca	D	NM	ma	4.1		ni	L	mm	T		V/O	€€	E	VL	UC	SS-B	SS-S-M-D	SS-VS-S
Dicentrarchus labrax	D	O	ma	3.8	17.9	5.5	L	hm	T	P	ON- G	€€€	E	M	UC	B-S-M	R-SS-S-M	VS-S-E
Dicologlossa cuneata	D	NM	inv	3.3	6.0	1.8	M	mm	T		ON- G	€€€	E	H	C	SS-B	SS-VS-S	SS-VS-S-E
Diplecogaster bimaculata bimaculata	D	NM	inv	3.3	8.3	3.7	S	te	T	P	OG	€	U	M	R	R-S-M	R-VS	R-VS
Diplodus annularis	BP	NM	om	3.4	17.6	6.5	S	mm	WT	P	ON- G	€€€	E	M	UC	B	R-SS-VS-S-M	R-VS-E
Diplodus bellottii	BP	NM	om	3.6	10.5	2.8	M	mm	WT		ON- G	€€€	E	M	UC	B-M-D	R-SS-S-M	R-VS-E
Diplodus cervinus cervinus	RA	O	om	3.0	35.8	10.7	L	hm	WT		ON- G	€€€	E	L	C	B-M-D	R-SS-S-M	R-VS-E
Diplodus puntazzo	BP	O	om	2.9	8.0	2.4	L	hm	Tr	P		€€€	E	M	UC	B	R-SS-VS-S-M	R-VS-E

Table 4.B.1. (continued)

Species	Pos.	Mig.	TG	TL	LS	GT	BS	Mob	Biog	Lv	Rep	CV	LE	Res	Ab	Adults	Spawning	Nursery
Diplodus sargus sargus	D	O	om	3.0	28.6	8.5	M	hm	Tr	P	ON- G	€€€	E	M	VC	B-S-M	R-SS-VS-S	R-VS-E
Diplodus vulgaris	BP	O	om	3.2	7.9	2.2	M	hm	WT	P	ON- G	€€€	E	M	VC	B	R-SS-VS-S	R-VS-E
Echiichthys vipera	D	NM	ma	4.4		ni	S	se	T	P	ON- G	€	U	H	C	SS-B	SS-VS-S	SS-VS-E
Engraulis encrasicolus	P	O	zoo	3.1	2.4	0.8	M	hm	T	P	ON- G	€€	E	M	UC	B	SS-M-D-VD	SS-VS-E
Entelurus aequoreus	D	NM	inv	3.5	8.0	2.4	L	mm	CT	BP	OB	€	U	M	R	B	R-SS-VS-S	R-SS-VS-E
Epinephelus marginatus	RA	NM	ma	3.7	31.0	10.0	L	te	WT		ON- G	€€€	E	L	R	R-B	R-S-M-D	R-VS-S
Gaidropsarus mediterraneus	D	O	om	3.4	4.7	1.5	M	hm	T	P	ON- G	€	B	H	R	R-B	R-VS-S	R-VS
Gaidropsarus vulgaris	D	NM	ma	3.3	6.0	1.8	L	mm	CT		ON- G	€	B	M	R	B	R-VS-S	R-VS
Gobius cobitis	D	O	om	3.0	13.0	3.5	M	te	WT			€	U	M	UC	B-S-M	R-VS	R-VS
Gobius cruentatus	D	O	om	3.1	8.3	3.2	S	te	WT			€	U	M	C	B-M	R-VS	R-VS
Gobius gasteveni	D	NM	om	3.1	5.6	2.2	S	te	WT			€	U	M	R	SS-M-D	R-VS	R-VS
Gobius niger	D	NM	om	3.2	14.7	5.6	S	te	T	P	ON- H	€	U	M	UC	B	R-VS	R-VS
Gobius paganellus	D	A	om	3.3	3.8	1.2	S	te	WT	AS	OG	€	U	M	C	B-S	R-VS	R-VS
Gobius xanthocephalus	D	NM	om	3.1	4.7	1.9	S	te	WT			€	U	H	VC	B-S	R-VS	R-VS
Gobiusculus flavescens	D	NM	zoo	3.2	2.9	1.3	S	mm	CT	P	ON- H	€	U	H	C	R-S-M	R-VS	R-VS
Gymnammodytes semisquamatus	D	NM	zoo	2.7	10.1	2.8	M	mm	CT	P		€	U	M	R	SS-B	SS-S-M	SS-VS-S

Table 4.B.1. (continued)

Species	Pos.	Mig.	TG	TL	LS	GT	BS	Mob	Biog	Lv	Rep	CV	LE	Res	Ab	Adults	Spawning	Nursery
Halobatrachus didactylus	D	NM	ma	4.0	21.9	6.5	M	se	WT			€€	E	L	VC	B-M	R-VS	R-SS-VS-S-E
Hippocampus guttulatus	D	NM	zoo	3.5	4.9	1.6	M	se	WT	AS	OB	€	U	M	R	B-S	R-VS	R-VS
Hippocampus hippocampus	D	NM	zoo	3.2	3.1	1.2	S	se	WT	AS	OB	€	U	H	R	B	R-VS	R-VS
Hyperoplus lanceolatus	D	O	ma	4.2	7.1	2.4	M	hm	CT	P		€	U	M	R	B-S-M	SS-VS-S	SS-VS-S
Labrus bergylta	RA	NM	inv	3.1	28.6	8.6	L	mm	CT	P	OG	€€	E	L	C	R-S-M	R-VS-S	R-VS-S
Labrus mixtus	RA	NM	inv	3.9	30.6	8.8	M	mm	T		OG	€€	E	L	UC	R-B	R-VS-S	R-VS-S
Lepadogaster candolii	D	NM	inv	2.8	10.2	4.3	S	te	WT	P		€	U	M	C	R-S	R-VS	R-VS
Lepadogaster lepadogaster	D	NM	inv	3.3	8.9	3.9	S	te	WT	P		€	U	M	C	R-S	R-VS	R-VS
Lepadogaster purpurea	D	NM	inv	3.3	10.2	4.3	S	te	T			€	U	M	UC	R-S	R-VS	R-VS
Lepidorhombus boscii	D	NM	ma	3.7	10.7	3.0	M	mm	T	P		€€€	E	M	R	SS-B	SS-S-M-D-VD	SS-VS-S-E
Microlipophrys canevae	D	NM	om	2.1	3.1	1.3	S	te	WT		OG	€	U	H	R	R-S	R-VS	R-VS
Lipophrys pholis	D	NM	om	3.1	9.4	3.6	M	te	T	P	OG	€	U	H	C	R-S	R-VS	R-VS
Lipophrys trigloides	D	NM	om	3.5	5.3	2.1	S	te	WT		OG	€	U	H	C	R-S	R-VS	R-VS
Liza aurata	P	C	om	2.5	14.4	4.4	L	hm	T	P	ON- G	€€	E	M	C	B-M	SS-VS-S	E
Liza ramada	P	C	om	2.2	9.2	2.7	L	hm	T	P	ON- G	€€	E	L	C	B-M	SS-VS-S	E
Liza saliens	P	NM	om	3.0	14.2	4.1	M	mm	WT	P	ON- G	€€	E	M	R	B-M	SS-VS-S	E
Lophius budegassa	bathyD	NM	ma	4.5	35.9	11.4	L	se	T		ON- G	€€€	E	M	UC	SS-D	SS-VD	SS-VD

Table 4.B.1. (continued)

Species	Pos.	Mig.	TG	TL	LS	GT	BS	Mob	Biog	Lv	Rep	CV	LE	Res	Ab	Adults	Spawning	Nursery
Macroramphosus scolopax	P	NM	inv	3.5	6.1	2.3	S	mm	Eu	P		€	B	M	R	B-M-D	SS-VD	SS-D-VD
Merluccius merluccius	D	NM	ma	4.4	22.8	7.1	L	mm	T	P	ON- G	€€€	E	L	VC	SS-M-D	SS-VD	SS-D-VD
Microchirus azevia	D	NM	inv	3.2	12.9	3.7	M	mm	T		ON- G	€€€	E	M	VC	SS-B	SS-M-D-VD	SS-VS-S-E
Microchirus boscanion	D	NM	inv	3.2	5.6	2.1	S	mm	Tr		ON- G	€	B	H	R	SS-B	SS-M-D-VD	SS-VS-S-E
Microchirus ocellatus	D	NM	inv	3.2	5.6	2.1	S	mm	WT		ON- G	€	B	H	R	SS-M-D	SS-M-D-VD	SS-VS-S-E
Microchirus variegatus	D	NM	inv	3.3	7.5	2.2	M	mm	T	P	ON- G	€€€	B	M	R	SS-M-D	SS-M-D-VD	SS-VS-S
Micromesistius poutassou	P	O	ma	4.0	7.8	2.4	M	hm	T	P		€€	E	M	R	B-D	SS-VD	SS-VD
Mola mola	P	O	om	4.0		ni	L	hm	Eu		ON- G	€	B	M	UC	B-M-D	R-SS-M-D-VD	R-SS-VS-S-M
Monochirus hispidus	D	NM	inv	3.2	7.4	2.9	S	mm	Tr		ON- G	€€	B	H	R	SS-M-D	SS-VS-S-E	SS-VS-S-E
Mugil cephalus	BP	C	ma	2.1	8.4	2.6	L	hm	Tr	P	ON- G	€€	E	M	C	SS-S	R-SS-VD	E-Rivers
Mullus barbatus barbatus	D	NM	inv	3.2	13.2	3.6	M	mm	T	P		€€€	E	M	UC	SS-B	SS-D-VD	SS-M-D
Mullus surmuletus	D	O	inv	3.4	6.6	2.4	M	hm	T	P		€€€	E	M	C	SS-B	R-SS-S-M	R-SS-M
Muraena helena	RA	NM	ma	4.2		ni	L	se	WT	P	ON- G	€€€	E	L	UC	R-M	R-VD	R-VS-S
Mustelus mustelus	D	NM	ma	3.8	24.1	8.0	L	hm	T		V/O	€€€	E	VL	C	B	R-SS-M	R-SS-M
Myliobatis aquila	BP	NM	ma	3.6	13.8	4.7	L	mm	Tr		V/O	€€	B	VL	C	SS-B	SS-E-VS-S-M-D	SS-E-VS-S-M-D

Table 4.B.1. (continued)

Species	Pos.	Mig.	TG	TL	LS	GT	BS	Mob	Biog	Lv	Rep	CV	LE	Res	Ab	Adults	Spawning	Nursery
Nerophis lumbriciformis	D	NM	inv	4.0	6.9	2.7	S	se	CT	BP	OB	€	U	M	UC	R-S	R-VS	R-VS
Oblada melanura	BP	O	om	3.0	9.5	2.8	M	hm	Tr	P	ON- G	€€	E	M	C	R-S-M	R-SS-VS-S	R-VS
Pagellus acarne	BP	O	ma	3.5	13.5	3.8	M	hm	T	P	ON- G	€€€	E	M	C	B-M-D		R-SS-VS-S
Pagellus bogaraveo	BP	NM	ma	3.7	31.4	9.4	L	mm	T	P	ON- G	€€€	E	L	R	B-D	R-SS-VD	R-SS-VS-S
Pagellus erythrinus	BP	NM	ma	3.4	17.3	4.9	M	hm	T		ON- G	€€€	E	M	UC	B	R-SS-S-M-D	SS-VS-S
Pagrus auriga	BP	O	inv	3.4	31.9	9.9	L	hm	Tr		ON- G	€€€	E	VL	R	R-B		SS-VS-S
Pagrus pagrus	BP	O	ma	3.7	29.9	9.2	L	hm	WT	P	ON- G	€€€	E	M	C	B	R-SS-M	SS-VS-S
Parablennius gattorugine	D	NM	om	2.9	3.8	1.1	M	te	WT	P	OG	€	U	H	C	R-S	R-VS-S	R-VS-S
Parablennius incognitus	D	NM	om	2.4	2.9	1.3	S	te	WT		OG	€	U	H	R	R-S	R-VS-S	R-VS-S
Parablennius pilicornis	D	NM	om	3.2	6.2	2.4	S	te	Tr		OG	€	U	H	VC	R-S	R-VS	R-VS
Parablennius rouxi	D	NM	om	2.6	3.3	1.4	S	te	WT		OG	€	U	H	R	B-S	R-VS	R-VS
Parablennius ruber	D	NM	om	2.9	4.5	1.8	S	te	T		OG	€	U	H	UC	R-S	R-VS	R-VS
Parablennius sanguinolentus	D	NM	om	2.1	7.8	2.9	S	te	WT		OG	€	U	H	R	R-S	R-VS	R-VS
Phycis phycis	BP	NM	inv	4.3	15.1	4.6	L	mm	WT	P	ON- G	€€€	E	M	UC	B-D	R-VD	R-VD
Pollachius pollachius	BP	O	inv	4.2	15.0	15.4	L	hm	CT			€€€	E	M	UC	R-M-D	R-D-VD	R-VD

Table 4.B.1. (continued)

Species	Pos.	Mig.	TG	TL	LS	GT	BS	Mob	Biog	Lv	Rep	CV	LE	Res	Ab	Adults	Spawning	Nursery
Pomatoschistus marmoratus	D	NM	inv	3.4	3.0	1.3	S	se	WT		OG	€	U	M	UC	SS-S	R-SS-VS-S	R-SS-VS-S
Pomatoschistus microps	D	A	inv	3.3	9.4	3.8	S	hm	T	P	OG	€	U	M	R	SS-S	E	E
Pomatoschistus pictus	D	NM	inv	3.1	2.9	1.3	S	se	T	P		€	U	M	VC	SS-S	R-SS-VS-S	R-SS-VS-S
Pteromylaeus bovinus	BP	NM	ma	3.8	24.2	8.5	L	mm	Tr		V/O	€€	B	VL	R	SS-B	Out	Out
Raja brachyura	D	NM	ma	4.0	15.2	5.8	L	mm	T		ON- G	€€€	E	L	UC	B-M-D	SS-S-M	SS-S-M
Raja clavata	D	NM	ma	3.8	32.1	10.7	L	mm	T		ON- G	€€€	E	L	C	B-M-D	SS-S-M	SS-S-M
Raja miraletus	D	NM	ma	3.8	14.9	5.0	L	mm	WT		ON- G	€€€	E	L	UC	SS-M-D	SS-VD	SS-M-D
Raja montagui	D	NM	inv	3.7	19.4	6.6	L	mm	T		ON- G	€€€	E	L	UC	SS-M-D	SS-D-VD	SS-S-M
Raja undulata	D	NM	ma	3.5	26.2	8.4	L	mm	WT		ON- G	€€€	E	L	UC	SS-M-D	SS-M	SS-M
Rostroraja alba	D	NM	ma	4.4	41.4	14.4	L	mm	T		ON- G	€€€	E	L	UC	B-M-D	Out	Out
Sarda sarda	P	O	ma	4.5	4.1	1.6	L	hm	Eu	P	ON- G	€€€	E	M	R	B	Out	Out
Sardina pilchardus	P	O	zoo	3.1	10.9	4.0	M	hm	T	P	ON- G	€€€	E	M	C	B	SS-M	SS-VS-S
Sarpa salpa	BP	O	he	2.0	12.5	3.6	L	mm	Tr	P	ON- G	€€	E	M	VC	B	R-SS-VS-S-M	E-R-VS
Scomber colias	P	O	ma	3.9	5.2	3.2	L	hm	Tr			€€	E	M	C	B	SS-S-M	SS-S-M
Scomber scombrus	P	O	ma	3.7	10.9	3.7	L	hm	CT	P	ON- G	€€	E	M	C	B	SS-S-M-D-VD	SS-S-M-D-VD
Scophthalmus maximus	D	O	ma	4.0	11.0	3.8	L	hm	T	P		€€€	E	M	UC	B-M-D	E-SS-S	E-SS-S

Table 4.B.1. (continued)

Species	Pos.	Mig.	TG	TL	LS	GT	BS	Mob	Biog	Lv	Rep	CV	LE	Res	Ab	Adults	Spawning	Nursery
Scophthalmus rhombus	D	O	ma	3.8	5.7	1.8	L	hm	WT	P		€€€	E	M	UC	SS-S-M	SS-S	SS-S
Scorpaena notata	D	NM	ma	3.5	15.7	4.2	S	se	WT	P		€	B	M	C	R-B	R-VS-S-M	R-VS-S-M
Scorpaena porcus	D	NM	ma	3.9	26.6	7.6	M	se	WT	P		€€	E	M	C	R-B	R-VS-S	R-VS-S
Scyliorhinus canicula	D	NM	ma	3.7	24.3	7.8	L	mm	T		ON- G	€€	E	L	C	SS-B	R-S-M	R-VD
Serranus atricauda	D	NM	ma	4.3	25.9	7.6	M	mm	WT			€€€	E	L	R	R-B	VD	VD
Serranus cabrilla	D	NM	ma	3.4	25.5	7.2	M	mm	Tr	P		€	E	M	C	B	R-VS-S	R-VS-S
Serranus hepatus	D	NM	ma	3.5	7.8	3.0	S	mm	WT			€	E	M	UC	B	R-SS-S-M-D	R-SS-S-M-D
Solea lascaris	D	NM	inv	3.2	6.3	2.0	M	mm	T	P	ON- G	€€€	E	M	VC	SS-S-M	SS-VS-S	SS-VS-S
Solea senegalensis	D	NM	inv	3.1	19.1	5.8	L	mm	WT		ON- G	€€€	E	L	VC	SS-M-D	SS-M	E-S
Solea solea	D	O	inv	3.1	10.8	3.2	L	hm	T	P	ON- G	€€€	E	M	C	SS-M-D	SS-M	E-VS-S
Sparus aurata	D	NM	om	3.3	7.7	2.4	L	mm	WT	P	ON- G	€€€	E	M	UC	B-S-M	R-SS-M	E-R-SS-VS-S
Sphoeroides marmoratus	D	NM	inv	3.2	4.9	1.8	S	mm	Tr			€	U	M	R	R-B	Out	Out
Spicara maena	P	NM	zoo	4.2	9.0	3.4	M	mm	WT	AS		€	B	M	R	B-M-D	SS-S-M	SS-S-M
Spondyliosoma cantharus	BP	O	om	3.3	16.0	4.7	L	hm	T	P	ON- H	€€	E	M	C	B		E-R-SS-S
Symphodus bailloni	RA	NM	inv	3.3	8.8	3.3	S	mm	T		OG	€	B	M	C	R-S-M	R-VS-S	R-VS-S
Symphodus cinereus	D	NM	inv	3.3	6.2	2.5	S	mm	WT	P	OG	€	B	M	UC	R-S-M	R-VS-S	R-VS-S

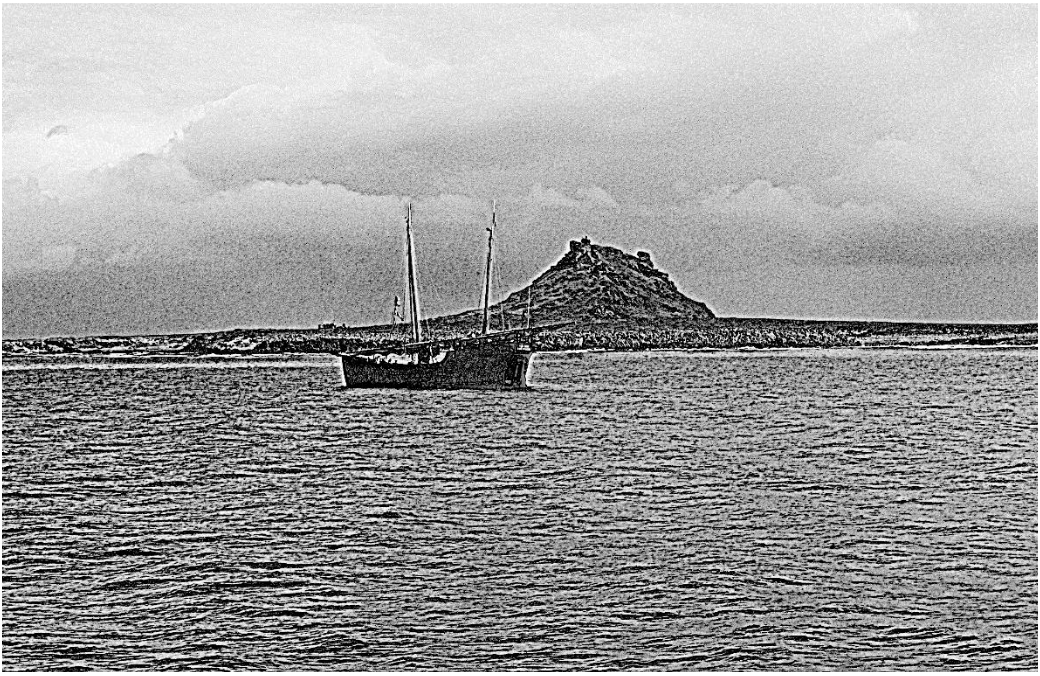
Table 4.B.1. (continued)

Species	Pos.	Mig.	TG	TL	LS	GT	BS	Mob	Biog	Lv	Rep	CV	LE	Res	Ab	Adults	Spawning	Nursery
Symphodus mediterraneus	D	NM	inv	3.1	4.5	1.3	S	mm	WT		OG	€	B	M	R	R-S-M	R-VS-S	R-VS-S
Symphodus melops	RA	NM	inv	3.2	9.1	2.5	M	mm	CT	P	OG	€	B	M	VC	R-S-M	R-VS-S	R-VS-S
Symphodus ocellatus	RA	NM	inv	3.3	2.9	1.2	S	mm	WT		OG	€	B	M	R	R-S-M	R-VS-S	R-VS-S
Symphodus roissali	RA	NM	inv	3.5	8.1	3.1	S	mm	WT	P	OG	€	B	M	VC	R-S-M	R-VS-S	R-VS-S
Symphodus rostratus	RA	NM	inv	3.5	3.9	1.6	S	mm	WT		OG	€	B	M	UC	R-S-M	R-VS-S	R-VS-S
Synaptura lusitanica lusitanica	D	NM	inv	3.8	10.9	3.2	M	mm	Tr		ON- G	€€€	E	M	R	SS-B	SS-S-M	SS-S-M
Syngnathus acus	D	NM	inv	3.4	9.5	2.8	M	se	T	APB	OB	€	U	M	UC	B-S	R-VS-S	R-VS-S
Syngnathus typhle	D	NM	inv	4.3	5.1	1.8	M	se	T	APB	OB	€	U	M	R	B-S	R-VS-S	R-VS-S
Synapturichthys kleinii	D	NM	inv	3.6	8.1	2.6	M	mm	WT		ON- G	€€€	E	M	R	SS-M-D	SS-VD	SS-M-D
Taurulus bubalis	D	NM	ma	3.6	11.2	3.2	S	mm	CT	P		€	U	M	R	R-S-M	R-VS-S	R-VS-S
Thorogobius ephippiatus	D	NM	om	3.0	12.1	4.8	S	te	T			€	U	H	UC	R-S-M	R-VS-S	R-VS-S
Torpedo torpedo	D	NM	ma	4.5	25.9	7.5	L	mm	WT		V/O	€€	E	L	C	SS-B	SS-M-D	SS-M-D
Trachinus draco	D	NM	ma	4.2		ni	L	se	T	P	ON- G	€	B	M	C	SS-S-M	SS-VS-S	SS-VS-S
Trachurus mediterraneus	P	O	ma	3.6	12.6	3.7	L	hm	WT	P		€€	E	M	R	B	SS Out	SS-Out
Trachurus picturatus	BP	O	ma	3.3	8.4	2.6	L	hm	WT			€€	E	M	UC	B-D	SS Out	SS-Out
Trachurus trachurus	P	O	ma	3.6	18.3	5.5	L	hm	T	P		€€€	E	M	C	B-D	SS-VD	SS-Out

Table 4.B.1. (continued)

Species	Pos.	Mig.	TG	TL	LS	GT	BS	Mob	Biog	Lv	Rep	CV	LE	Res	Ab	Adults	Spawning	Nursery
Trigloporus lastoviza	D	NM	inv	3.4	7.6	2.4	M	mm	Eu			€€	B	M	C	B	R-SS-S-M	R-SS-S-M
Tripterygion delaisi	D	NM	inv	3.4	2.9	1.2	S	te	WT			€	U	H	VC	R-S-M	R-VS	R-VS
Trisopterus luscus	BP	O	ma	3.7	13.5	3.9	M	hm	T	P	ON- G	€€€	E	M	C	B-M-D	R-SS-VS-S-M	R-VS-S
Uranoscopus scaber	D	NM	pi	4.4	8.0	2.4	M	se	WT	P	ON- G	€	B	M	R	SS-M-D	SS-	B
Zeugopterus punctatus	D	NM	ma	4.0	9.1	2.5	M	mm	CT	P		€€	B	M	UC	R-S-M	R-	R-VS-S
Zeugopterus regius	D	NM	ma	3.4	6.1	2.3	S	mm	CT	P		€€	B	M	R	B	R-SS	
Zeus faber	BP	O	ma	4.5	9.6	3.4	L	hm	Eu	P	ON- G	€€€	E	L	R	B	R-SS-S-M-D	R-SS-S-M-D

CHAPTER 5



Batista MI, Cabral HN. An overview of SW Europe Marine Protected Areas: factors contributing to their effectiveness. In review in *Ocean & Coastal Management*

An overview of SW Europe Marine Protected Areas: factors contributing to their effectiveness

Abstract

Marine Protected Areas (MPA) are considered key elements to the achievement of conservation and sustainable marine management targets. Though recently the number of MPA has increased rapidly worldwide, the area of ocean under some type of MPA classification is far behind international targets (e.g. Convention on Biological Diversity) considered essential for world oceans conservation. Furthermore, coherence, representativeness and effectiveness of existing MPA are largely unknown or even weakly defined. In this study, general characteristics of MPA from Portugal, Spain and France were collected and used to assess conservation progress in this geographic area. In addition, MPA managers answered to an online questionnaire on processes inherent to each MPA, namely on the characteristics and suitability of planning, management, monitoring, governance and enforcement. Responses obtained were used to calculate the overall level of MPA effectiveness, and multivariate analyses were used to identify factors that most contribute to differences in effectiveness. Most MPA are adjacent to coast, have small areas (near 50% have less than 20 km^2) and were established with multiple goals concerning species conservation and sustainable development of economic activities (e.g. fisheries). Only 9% of the MPA are larger than 1000 km^2 and are unequally distributed among the study area. Overall, 46% of MPA and 59% of the area covered were established during the last five years, while only 3 of the 35 no-take areas (22% in area) were implemented during this period. High effectiveness of MPA is related with high levels of stakeholders support, with suitable goals, management and enforcement. Global effectiveness of MPA are lacking in the geographic area considered. Results highlighted the need to improve MPA coverage taking into account other already existing MPA to improve coherence and representativeness of networks, new no-take areas should be implemented in key conservation sites and management strategies (e.g. enforcement and monitoring) should be strengthened. These findings are applicable to the study area but can also be adapted for applications worldwide. The investment in strategies aiming at maximizing MPA performance are probably as important as the increase of MPA coverage.

Keywords: Environmental management, Convention for Biological Diversity, OSPAR, MSFD, MPA effectiveness, stakeholders

Introduction

Past, current and expected future impacts of human activities on oceans (e.g. resources overexploitation, habitat degradation) have led to an increased concern of governments and societies with the implementation of measures aiming at the preservation of marine ecosystems services on a long-term perspective, such as the establishment of Marine Protected Areas (MPA) (Halpern et al., 2008). MPA are considered key elements to the achievement of conservation and sustainable marine management targets, and according to the International Union for Conservation of Nature (IUCN), a MPA is *"a clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values"* (Dudley, 2008). The number of MPA has increased rapidly in recent years: 1.8% of areas within economic exclusive zones (EEZ) were covered by MPA in 2008, while in 2010 this percentage raised to 2.9% (Spalding et al., 2010; Spalding et al., 2008). For instance, in the Mediterranean Sea, the number and area of MPA almost doubled between 2008 and 2012 (Gabrié C. et al., 2012). The spatial extent of MPA varies widely across marine ecoregions and biogeographic provinces, and also habitats - with most MPA concentrated in intertidal or near-coastal waters (Gabrié C. et al., 2012; Spalding et al., 2008; Wood et al., 2008). Although still far from the international targets that have indicated, these numbers seems to be in line with the conservation targets set by several international and regional guidelines. The most remarkable of these policy initiatives are: the Convention on Biological Diversity (CBD) that created an international commitment to conserve *"at least 10% of coastal and marine areas (…) through effectively and equitably managed, ecologically representative and well connected systems of protected areas (…) by 2020"* (target revised and updated in 2010); and, at the European level, the Convention for the protection of the marine environment of the North-East Atlantic ("OSPAR Convention") that aimed to ensure that the network of MPA in North-East Atlantic "*by 2012 it is ecologically coherent and includes sites representative of all biogeographic regions in the OSPAR maritime area (…)* and "*by 2016 it is well managed (…)*" (see OSPAR recommendation 2003/3 and 2010/2) and the European Marine Strategy Framework Directive (MSFD) that aims to achieve the "good environmental status" of member states' marine waters by 2020, through for example the establishment of coherent networks of MPA.

MPA are established by institutions and governments for a wide range of purposes, including protecting biodiversity and habitats, maintaining ecosystem services, restoring fisheries stocks, managing other economic activities, minimizing conflicts among resource users and decrease poverty (Abdulla et al., 2009; Botsford et al., 2003; Gerber et al., 2003; Kelleher, 1999). Nevertheless this multiplicity of objectives is often hard to achieve simultaneously and depends on several factors, from the design characteristics of the MPA itself to the compliance by local communities. Key factors underpinning the success of a MPA are often the stakeholder's compliance and participation in decision processes (Claudet and Guidetti, 2010; Guidetti and Claudet, 2010; Jones, 2001; Roberts et al., 2003), the adequate conduction of processes involved in the establishment of a MPA from planning to monitoring, and the implementation of adequate enforcement (Agardy et al., 2011; Fenberg et al., 2012; Fernandes et al., 2005). Though many studies have addressed the factors that mostly contribute to MPA success, results are not always in concordance, showing that uncertainty is also present in these processes (McCook et al., 2009). Claudet et al. (2008), for example, detected positive effects of size and age of no-take areas (i.e. fully protected from extractive uses) in the biomass of exploited species, while Halpern (2003) showed that the relative impacts of no-take zones, such as proportional differences in density or biomass, are independent of reserve size and Edgar et al. (2014) found that the conservation benefits for reef fish communities of worldwide MPA increased exponentially with the accumulation of five key features: presence of no-take zones, good enforcement, age (>10 years), size (>100 km^2), and isolation. Moreover, it is becoming widely recognized that effective marine conservation and management at ecosystem scale requires extensive networks/ systems of no-take MPA (e.g. Fernandes et al., 2005; Lubchenco et al., 2003; McCook et al., 2010; Russ et al., 2008). IUCN emphasizes that protected areas should not be seen as isolated entities and suggested that the long-term success of *in-situ* conservation requires that the global system of protected areas comprise a representative sample of each of the world's different ecosystems (Dudley, 2008). By accumulating the benefits of multiple MPA, networks can have even more benefits than the sum of its individual parts, through synergistic effects (Gaines et al., 2010). Some studies estimate that individual reserves must be at least as large as the average dispersal distance for a species (Botsford et al., 2001; Lockwood et al., 2002). Larval and adult movements typically are long enough to require that protected areas be at least tens, and perhaps hundreds, of kilometres wide (Baskett et al., 2007; Palumbi, 2004), which adds to the potential value of implementing networks of MPA.

Despite the promising current rate of increase in MPA coverage issues such as the coherence, representativeness, management performance or global effectiveness of MPA still need clarification considering large spatial scales (Abdulla et al., 2009; Ardron, 2008; Gabrié C. et al., 2012; Jones and Carpenter, 2009; McCook et al., 2010; Roberts et al., 2003). This study aims to assess the features underlying MPA effectiveness, by focusing on MPA in southwest Europe. MPA characteristics and managers' perceptions about the suitability of MPA processes (i.e. planning, management and monitoring) are combined to identify the main factors contributing to maximize MPA effectiveness. Finally, the main actions needed to achieve international marine conservation targets in view of these key factors are highlighted.

METHODS

Study area and data collection

The present study focused on Marine Protected Areas (MPA) of southwest Europe: Portugal (including Azores and Madeira archipelagos), Spain (including Canarias) and France (Figure 5.1).

The MedPan database for the Mediterranean Sea (www.mapamed.org) and MAIA database for the North Atlantic Ocean (www.maia-network.org) were chosen to identify MPA as they were the most complete and updated databases for the study area (after an exhaustive search at MPA online databases and an analysis of data accuracy and level of detail, by comparing different sources, and comparing the information in databases with legislation). For each identified MPA (n=134) the following aspects were characterized: age, total area, no-take area, goals, location and governance characteristics (i.e. based only on governmental bodies or with effective participation of stakeholders). This was done through an extensive search in governmental and institutional sites, MPA management bodies, peer-reviewed papers and legislation. Overlapped areas were excluded from the analyses (i.e. some areas have various designations for the same perimeter and thus the same perimeter is listed several times in the databases). Furthermore, areas classified only as *Natura 2000* sites were not included in this analyses.

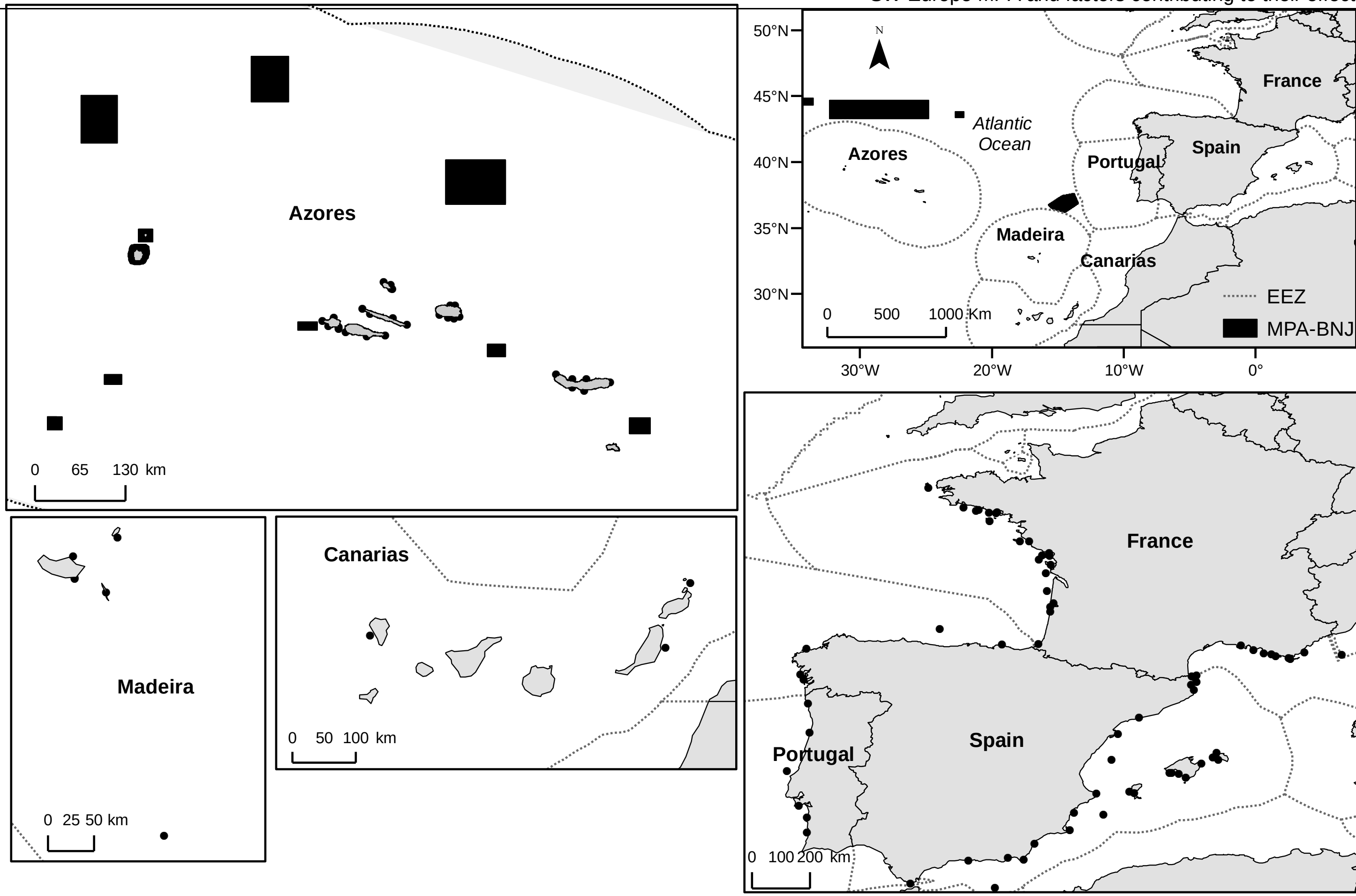


Figure 5.1. Map of southwest Europe showing Marine Protected Areas included in the analyses. The six geographic areas considered are shown: Azores (Portugal-A, top left); Madeira (Portugal-Ma, bottom left); Canarias (Spain-Ca, bottom centre); mainland Portugal, Spain and France (bottom right) and areas beyond national jurisdiction (BNJ, top right). Dotted lines are the boundaries of economic exclusive zones (EEZ). MPA locations are represented in black but their size are not represented except for a few exceptions (black rectangles in the Azores and in areas BNJ).

An online questionnaire was performed to MPA managers to collect detailed information on processes inherent to each identified MPA. The questionnaire comprised six sections, regarding MPA basic characterisation and processes inherent to MPA implementation: (1) MPA characterization (e.g. size, IUCN category, depth range, habitats present), (2) planning process (e.g. designation criteria, objectives, level of stakeholders participation and support, level of compliance, level of scientific information available), (3) management plan (e.g. management measures implemented, activities restricted, level of stakeholders participation and support, level of science based information), (4) monitoring (e.g. monitoring actions, budget for monitoring), (5) governance (e.g. governance model, stakeholders involvement, human resources) and (6) enforcement (e.g. entities with enforcement responsibilities, resources available). Questions on managers' perception about the level of suitability (from 1 to 5) of designation processes, MPA goals, management plan, global management and enforcement were also included in the questionnaire. The online questionnaires were constructed using *Lime Survey* software.

The questionnaires fully answered (hereafter named subset of MPA; n=10) were considered for more detailed analyses to investigate patterns leading to higher levels of MPA effectiveness.

Data analyses

For each studied geographic area, average and standard deviation of MPA total area, no-take area, age, relative size of no-take area comparing to the total area, relative area of MPA comparing to the respective Economic Exclusive Zone area (ZEE, source: *Maritime Boundaries Geodatabase*, http://www.marineregions.org/eez.php) were calculated. In addition we also determined percentage of MPA mainly aiming at conservation, percentages of coastal MPA, of MPA with management plan, and of ZEE covered by MPA. These variables were analysed trough an unconstrained Principal Coordinates Analysis (PCO) (Anderson et al., 2008) to identify patterns among MPA of the various geographic areas included in the study area. Geographic areas considered were mainland Portugal, mainland Spain, mainland France, Azores (Portugal), Madeira (Portugal) and Canarias (Spain).

For the subset of MPA with fully completed questionnaires, a global measure of MPA effectiveness was determined as the mean of suitability levels of: designation processes, MPA goals, management plan, global management and enforcement obtained from managers' responses to the online questionnaires. This level of effectiveness (low, medium and good) is based on the assumption that maximum effectiveness is achieved when all processes are totally adequate. In addition, the subset of MPA were also classified (in three levels, where 1 was the lowest) regarding: stakeholders' support, stakeholders' objection, and restrictions implemented. The existence of a monitoring plan (or not) was also included in the analyses as a presence/absence variable. These variables were analysed trough an unconstrained PCO (Anderson et al., 2008) to identify variables that contribute to high overall MPA effectiveness.

PCO were performed based on Euclidean distances among all pairs of samples with all variables previously normalized to place them on a comparable measurement scale. PRIMER 6 with PERMANOVA+ software package was used to PCO analyses.

RESULTS

A total of 134 MPA were listed and analysed in the present study, covering 227207 km^2 (Table 5.1). MPA located beyond national jurisdiction (BNJ) near Azores Economic Exclusive Zone (EEZ) (*Rainbow Hydrothermal Vent Field*, *Altair Seamount*, *Antialtair Seamount*, *Mid-Atlantic Ridge north of the Azores (MARNA)* and *Josephine Seamount Complex*) accounted with 44.2% of the total area covered by MPA in the study area and *Pelagos Sanctuary* (jurisdiction divided between France, Italy and Monaco) covered 38.5%. Among the remaining area covered by MPA (17.3%), 57.2% is French, 30.4% is Portuguese and 12.3% is Spanish. Nevertheless, near 93.4% of Portuguese MPA total area is from Azores Archipelago. Globally, MPA of the study area covered 1.28% of the Economic Exclusive Zones (EEZ, only the portion of countries' EEZ included in the study area were considered, and MPA BNJ were not included), with French leading with near 6.6% of its mainland EEZ protected (*Pelagos Sanctuary* was not included). The Portuguese EEZ of Azores followed France with near 1.17% of its EEZ covered by MPA (only 0.7% of overall Portuguese EEZ has covered by MPA) and the remaining geographic areas considered fall behind 1% of EEZ covered by MPA.

Table 5.1. Characteristics of marine protected areas per geographic area of southwest Europe. Number of MPA, number of no-take areas, mean age (years), total area **(km^2)**, average MPA area **(km^2)**, average no-take area **(km^2)**, average of MPA area comparing with EEZ area (%), number of MPA mainly aiming at conservation (%), number of coastal MPA (%), number of MPA with management plan (%) and area of EEZ covered by MPA (%) are presented (N = number).

Geographic area	**N**	**N No-take**	**Mean Age - years (SD)**	**Total area of MPA (km^2)**	**Average MPA area - km^2 (SD)**	**Average area of no-take - km^2 (SD)**	**Average of MPA/ EEZ area (%)**	**N of MPA for conserv. (%)**	**N of coastal MPA (%)**	**N of manag. plan (%)**	**EEZ covered by MPA (%)**
France (mainland)	39	5	13.77 (14.44)	110007.82	2820.71 (14001.05)	10.87 (19.59)	0.82 (4.02)	43.59	100.00	38.46	6.55
Portugal (mainland)	6	2	13.17 (3.13)	535.91	89.32 (94.90)	2.15 (1.85)	0.03 (0.03)	0.00	100.00	100.00	0.17
Portugal Azores	41	1	4.49 (6.22)	11175.04	272.56 (818.41)		0.004 (0.01)	21.95	85.37	4.88	1.17
Portugal (Beyond National Jurisdiction)	5	0	1.60 (0.49)	100400.24	20080.05 (36778.46)			20.00	0.00	20.00	
Portugal Madeira	5	2	21.60 (12.24)	253.80	50.76 (43.70)	76.01 (18.31)	0.01 (0.01)	60.00	100.00	100.00	0.06
Spain (mainland)	35	22	14.37 (7.60)	4120.98	117.74 (387.79)	5.91 (7.60)	0.02 (0.07)	17.14	97.14	97.14	0.74
Spain Canarias	3	3	19.67 (4.50)	712.78	237.60 (330.08)	4.55 (4.41)	0.05 (0.07)	33.33	100.00	100.00	0.16
Global	134	35	11.03 (11.03)	227206.58	1695.57 (10991.98)	9.84 (19.31)	0.26 (2.24)	27.61	91.04	49.25	1.28

MPA age, total and no-take areas

In terms of age, Madeira (Portugal) and Canarias (Spain) have the older MPA while Azores (Portugal) and nearby MPA BNJ have the youngest (Table 5.1). However, the older MPA in the study was *Port-Cros* (France), at 49 years old completed in 2012. *Ilhas Selvagens* was the oldest MPA in Portugal (41 years old in 2012) while *Bahia de Palma* and *Islas Chafrinas* (30 years old in 2012) are the oldest MPA in Spain. Among the youngest MPA are 12 French MPA (set in 2012), 15 MPA in Azores and 2 MPA beyond Portuguese national jurisdiction (set in 2011). Overall, 46% of MPA and 59% of the area covered are established during the last five years, while only 3 of the 35 no-take areas (22% in area) were implemented during this period.

Regarding size, near 48.5% have less than 20 km^2 while only 27.6% have more than 100Km^2 (from which 32.4% are larger than 1000 km^2). Larger MPA were established recently, in general, and are mostly located in Azores, near Azores but BNJ and in France (Table 5.1). The largest MPA are *MARNA* (93568 km^2) and *Pelagos Sanctuary* (87500 km^2). Considering only areas totally under countries jurisdiction MPA with largest surface area are *Pertuis Charentais – Rochebonne* (8176 km^2, France), *Sedlo Seamount* (4013 km^2, Azores) and *Golf du Lion* (4009 km^2, France). In Spain, *El Cachucho* is the largest MPA with approximately 2350 km^2. Conversely, mainland Portugal, mainland Spain and Madeira archipelago (Portugal) are the geographic regions with smaller MPA implemented. Larger no-take areas are found in France (*Callanques,* 50km^2) and Madeira (*Selvagens Islands*, 94.72 km^2). Spain was the geographic area with higher number of no-take zones though they are relatively small (Table 5.1, higher no-take zone is *Columbretes Islands*, 31.12 km^2).

Type of MPA and Management

MPA studied are generally coastal, with the main exceptions observed in Azores and near Azores in areas BNJ (Table 5.1). Most of the MPA have management plans in place with youngest MPA representing the main exceptions (most of the Azores MPA and MPA near Azores but BNJ and French MPA).

Generally the studied MPA are multi-use and have multiple goals, and the percentage of MPA mainly aiming at conservation are located in Madeira (Portugal), Canaries

(Spain) and France. Nevertheless, among all MPA studied only one fourth (n=35) is totally or partially no-take. In fact, there are only two areas totally no-take, *Selvagens Islands* (Madeira, Portugal) and *Caldeirinhas* (Azores, 0.1km^2).

Global patterns

Patterns revealed in the PCO analyses corroborate the results described above but there were no clear patterns linked to geographic area, showing that each studied area encompasses different types of MPA (Figure 5.2). The first PCO axis seems to separate larger from smaller MPA (MPA area was standardized by the corresponding ZEE area), with most of French and Azores (Portugal) MPA separated from the remaining MPA. The group mostly constituted by French and Azores MPA was mainly composed by MPA without no-take zones, younger and without management plans. Furthermore, most Spanish MPA were pooled together, and were quite homogeneous, characterised by having no-take area, management plans and multiple goals.

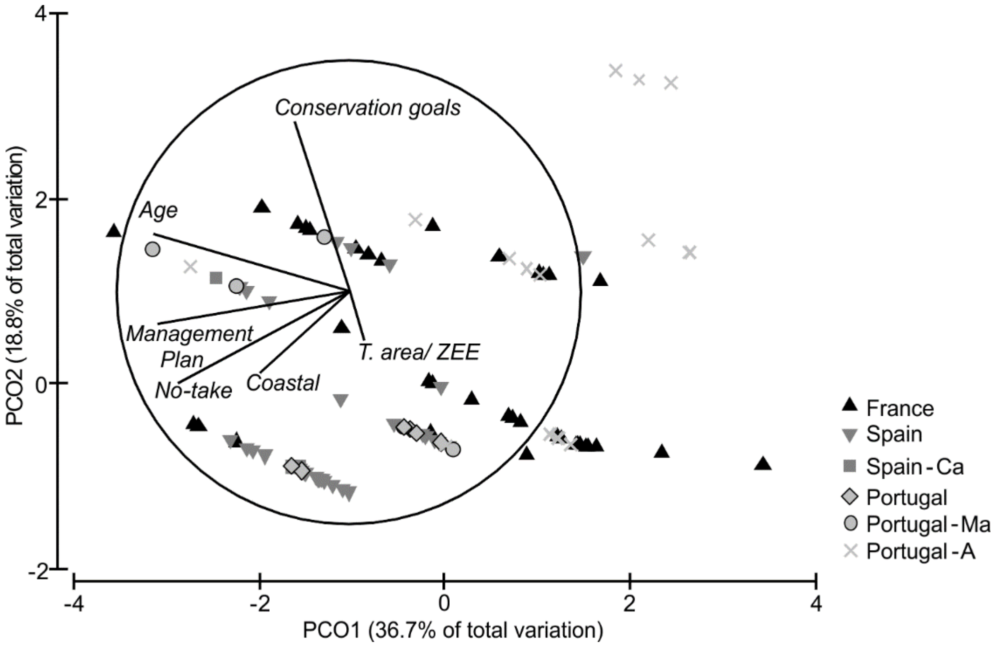


Figure 5.2. Ordination plots of Principal Coordinates Analysis (PCO) comparing MPA characteristics of Southern European countries (circle represent vector correlations of 1). MPA characteristics included in the analyses were: conservation goals (i.e. areas for which the priority is conservation), Age, no-take (i.e. MPA with no-take zones), management plan (i.e. MPA with management plan), coastal (i.e. MPA adjacent to coastline) and T. area/ ZEE (i.e. relative size of MPA comparing to ZEE area).

Effectiveness patterns based in managers' perceptions

Complete responses to questionnaires were obtained for a small proportion of MPA (n=10) though they represent three of the six geographic areas considered: Madeira (Portugal), Portugal (mainland) and Spain (mainland) as shown in table 5.2. For this subset of MPA, high effectiveness level is commonly related with some of the analysed variables (Figure 5.3). The first PCO axis separates MPA with medium from those with good levels of effectiveness while the second axes separated MPA with good effectiveness in two different groups. High levels of stakeholder's objections were common to MPA with medium effectiveness although a good suitability of management seems to occur in these areas. MPA with good effectiveness occur associated with high levels of stakeholder's participation even if restrictions to fisheries are in place. Higher effectiveness levels seems also to occur when designation criteria and MPA goals are adequate, management and enforcement are highly suitable and stakeholders support is high (Figure 5.3).

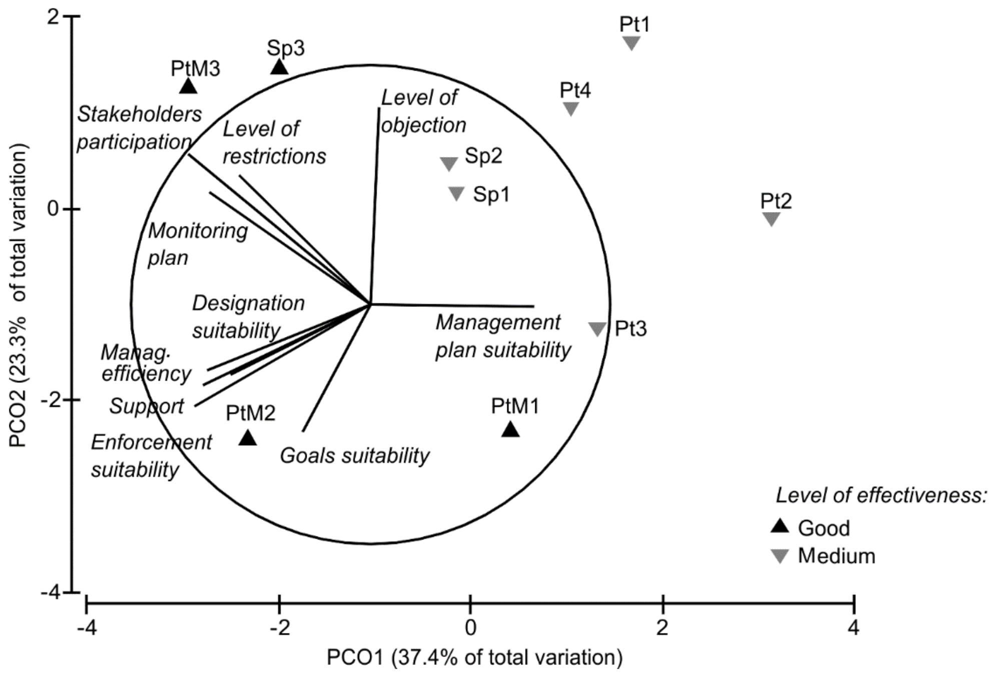


Figure 5.3. Ordination plots of Principal Coordinates Analysis (PCO) comparing MPA effectiveness levels and based on the suitability of processes inherent to MPA establishment (circle represent vector correlations of 1). Levels of management plan, MPA goals, enforcement and designation process suitability and levels of stakeholders' support, stakeholders' objection, restrictions implemented and management efficiency were considered as factors influencing MPA effectiveness. MPA codes are identified in table 5.2.

Table 5.2. Subset of marine protected areas included in the analyses of MPA effectiveness, i.e. MPA for which questionnaires were fully completed by managers. IUCN category (Ia is the high level of conservation targets, V is the lowest level), designation date, area of no-take zones (km^2), total area (km^2) and depth range (m) are presented. Priority designation criteria and priority objectives of the MPA identified in the questionnaires are shown.

Geographic area	MPA	Code	IUCN cat.	Designation	No-take (Km^2)	Total area (Km^2)	Depth range (m)	Priority designation criteria	Priority objectives
Portugal mainland	Arrábida	Pt1	V	1998	4	53	0-100	Biogeographic significance Ecological importance Scientific importance Economic Importance National or international importance	Habitats and species conservation Sustainable tourism Sustainable management of artisanal fisheries
Portugal mainland	Litoral Norte	Pt2	V	2005	0	74.91	0-53	Biogeographic significance Ease of implementation	Habitats and species conservation Conservation of cultural values Sustainable management of economic activities
Portugal mainland	Berlengas	Pt3	V	1998	0	94.56	0-500	Biogeographic significance Ecological importance National or international importance	Habitats and species conservation Conservation of cultural values Sustainable management of fisheries
Portugal mainland	Sudoeste Alentejano and Costa Vicentina	Pt4	V	2005	0.3	290	0-30	Biogeographic significance Ecological importance Scientific importance Socio-economic Importance National or international importance	Habitats and species conservation Conservation of cultural values Sustainable management of economic activities
Portugal Madeira	Selvagens	PtM1	Ia	1971	94.32	94.32	0-200	Degree of naturalness Biogeographic significance Ecological importance Scientific importance	Habitats and species conservation Sustainable tourism Conservation of cultural values

Table 5.2. (continued)

Geographic area	MPA	Code	IUCN cat.	Designation	NT area (Km^2)	Total area (Km^2)	Depth range (m)	Priority designation criteria	Priority goals
Portugal Madeira	Desertas	PtM2	Ia	1990	57.7	111.9	0-100	Degree of naturalness Biogeographic significance Ecological importance Scientific importance	Habitats and species conservation Sustainable tourism Conservation of cultural values Sustainable management of fisheries
Portugal Madeira	Garajau	PtM3	Ib	1986	0	3.76	0-50	Degree of naturalness Biogeographic significance Ecological importance Scientific importance Ease of implementation	Habitats and species conservationConservation of cultural valuesSustainable management of economic activities
Spain mainland	Bahia de Palma	Sp18	IV	2000	2.82	23.94	0-30	Degree of naturalness Ease of implementation	Habitats and species conservation Conservation of cultural values Sustainable management of economic activities
Spain mainland	Illa del Toro	Sp2	IV	2004	0	1.36	0-60	Degree of naturalness Ecological importance Economic Importance Scientific importance Ease of implementation	Habitats and species conservation Conservation of cultural values Sustainable management of economic activities
Spain mainland	Cabo de Palos - Islas Hormigas	Sp3	IV	1995	2.67	19.98	5-100	Degree of naturalness Biogeographic significance Ecological importance Scientific importance National or international importance	Habitats and species conservation Conservation of cultural values Sustainable management of economic activities

Discussion

The global increase of awareness of the potential of Marine Protected Areas (MPA) to contribute to the sustainable management of marine ecosystems is undeniable. The number of MPA recently set across countries in the study area (southwest Europe) is notorious and follows the global trends of rapid increase in MPA coverage during recent years (Spalding et al., 2010) and in particular due to the implementation of a few large MPA. Several factors, such as politics and socio-economics, seem to be interacting in a positive direction contributing to this trend (Grafton et al., 2011). The abundance of documents arguing the importance of MPA for oceans conservation and resources management (e.g. CBD, MSFD), the increase in scientific studies showing positive results of MPA (e.g. Claudet et al., 2010; Lester et al., 2009; Stewart et al., 2009) and the development of integrative methods and software that optimize the results and support management decisions (e.g. Marxan) are all factors that contribute to the high rate of MPA implementation, and, expectedly, to increase MPA effectiveness. Furthermore, it seems that the opinion of the general public favors MPA, at least for MPA that include socio-economic benefits (i.e. contribute to the development of particular economic activities, such as sustainable tourism and fisheries). In fact it has argued by several authors that marine conservation will only be effective in the long-term if human wellbeing is also accounted for in conservation strategies (e.g. Coulthard et al., 2011; Halpern et al., 2010; Rees et al., 2013). The results outlined in this study corroborates these authors. Despite recent advances there is still a generalized lack of MPA coverage and high levels of MPA inefficiency among designated areas, as well as low coherence and representativeness of existing MPA (e.g. Ardron, 2008; Jones and Carpenter, 2009; Spalding et al., 2010). The present study is thus an important contribution to fulfill the identified needs concerning MPA implementation in the study area, as defined by international conservation targets such as CBD, by quantifying current coverage by MPA. In addition, factors most contributing to MPA success in the study area are identified, which can applied to increase MPA success worldwide.

Among SW Europe countries, Portugal (namely Azores and areas BNJ) and France stand out with the highest recent dynamics regarding MPA implementation (several large MPA were implemented in 2011 and 2012, namely in high seas). Nevertheless, effective management measures seem to still be in progress for most of them. The implementation of larger MPA has been highlighted as more effective than smaller areas (Botsford et al., 2001; Botsford et al., 2003; Roberts et al., 2003) particularly since the size should be large enough to provide protection throughout the life cycles of the species (e.g. Grüss et al., 2011). For instance, large marine reserves can more effectively protect highly mobile species (e.g. whales, sharks, tunas) or species with high larval dispersal as discussed by Palumbi (2003). However, large MPA, as some of the included in this study (areas BNJ near Azores and Madeira), are hard to manage and enforce, and their effectiveness can be highly compromised given the data poor context and the lack of past experiences to draw upon (Leenhardt et al., 2013). Furthermore, if management measures implemented in these larger areas are not well accepted by stakeholders, their expected benefits can be highly compromised, for instance due to poaching events (Samoilys et al., 2007; Sethi and Hilborn, 2008). Few of the MPA studied here seem to fulfill the combination of characteristics identified by Edgar et al. (2014) as the key factors for MPA success (i.e. presence of no-take zones, good enforcement, age (>10 years), large size, and isolation). For instance MPA BNJ are large and isolated but do not have no-take areas and good enforcement is unlikely to occur, while Spanish MPA usually have no-take areas, can be potentially well enforced but are generally small and not isolated. Despite the recent increase in area covered by MPA, our results show that MPA coverage in the study area is far behind CBD targets and there is a lack of coherence among the MPA implemented (i.e. there isn't a coherent network of MPA). It is possible that some of the existing areas were established taking into consideration other previously existing and that they are therefore complementary, but this seems to occur for a minority of the cases.

Most of the MPA within the study area are small, individual and coastal and the same holds true for most MPA worldwide (Lester et al., 2009; Spalding et al., 2008). No-take areas represent a very small proportion of the total area protected, which should be a factor to revise in the future. While on the one hand no-take areas are more controversial than multiple use MPA (e.g. due to the total exclusion of human activities and to the high levels of poaching and stakeholders non-compliance), on the other hand they can promote greater benefits both for conservation of habitats and species as well as for restoration of fish stocks and consequent fisheries enhancement (Lester et al., 2009; Stewart et al., 2009). Thus, the implementation of coherent networks of several small no take MPA are viewed by many authors as an effective and most feasible strategy for both conservation and fisheries management objectives (Costello and Polasky, 2008; Gaines et al., 2010; McCook et al., 2010). This type of coherent “small no-take areas” integrated in larger MPA networks where large multi-use MPA (representative of habitats, namely in high seas) are also integrated seems to be a very promissory strategy to improve global MPA effectiveness. In view of this three main guidelines can be suggested for actions concerning MPA in the near future: 1) the implementation of new MPA should take into account other already existing MPA to improve coherence and representativeness of networks (it is important to evaluate areas for the establishment of large MPA); 2) the identification of smaller key areas for no-take regimes (in new or existing MPA); and 3) improvement of management strategies (e.g. through improve enforcement and monitoring processes) for MPA already implemented, namely for the most recent. These guidelines are not only applicable to the study area but can also be adapted for applications worldwide.

Although the study area (SW Europe) includes only three countries it comprises different geographic areas and variability of MPA characteristics, namely from older MPA to very recent MPA, large to small MPA or coastal to remote MPA; this variability allowed the assessment of patterns which can be broadly applied. Several factors can influence the success of a MPA, namely the suitability of processes inherent to MPA implementation (e.g. planning, management, monitoring, enforcement) as discussed by Agardy et al. (2011). The assessment of the major reasons or processes contributing for MPA success/failure is thus of the utmost importance. These should be taken into account in implementation of new MPA or for adaptation of management processes in existing MPA to fulfill gaps and improve effectiveness (McCook et al., 2010).

Several methods, often based on indicators, have been used to evaluate MPA performance but their accuracy and practical usefulness is highly dependent on long-term monitoring programs (Batista et al., 2011; García-Charton et al., 2008; Ojeda-Martínez et al., 2009; Pelletier et al., 2008; Pomeroy et al., 2005). However, MPA with long-term monitoring plans are scarce and indicators and assessment of MPA performance are often based in questionnaires (Leverington et al., 2010), as implemented in the present study. The implementation of questionnaire-based methods relying on the collaboration of trustworthy people (e.g. MPA managers, local stakeholders) that have a recognized knowledge of a given MPA and that closely accompanied all MPA processes may help in the evaluation of MPA and contribute to the improvement of management strategies and MPA performance. For instance, in the present study questionnaires placed MPA in good or medium effectiveness level. The absence of MPA with a low effectiveness level is likely explained by the methodological nature of this study since complete responses to the questionnaire intrinsically imply that an MPA has a management system and also that managers have a wide knowledge on MPA process, which would be more unlikely found in an MPA with low effectiveness. In addition, some responses may be influenced by managers' optimistic views and by the subjectivity inherent to this type of perception-based approaches, since they depend on individual experiences and relative comparisons. Although results obtained could not represent the global reality of MPA, factors contributing for higher MPA success in the present study can be considered in future management strategies and are in line with findings from previous studies. Here, higher effectiveness levels were common in areas where stakeholders involvement in MPA processes were higher even when restrictions to fisheries occur, which strengthens the importance of this factor for MPA success. Several studies reported the importance of stakeholders involvement for MPA success (e.g. Mangi and Austen, 2008; Rodriguez-Martinez, 2008; Springer, 2006). However, published studies usually analyse the influence of single or few factors in MPA performance or address individual MPA (e.g. Guidetti et al., 2008; Kritzer, 2004; Samoilys et al., 2007), while here several factors were integrated and several MPA were included in the analyses providing more robust and integrative conclusions. Despite this acknowledged importance of stakeholders participation for MPA performance, it is also related with a combination of other factors, namely good enforcement, the existence of monitoring plans and the suitability of management and objectives as discussed above.

Despite the low sample size for the analysis of MPA effectiveness (i.e. low number of questionnaires fully completed), the obtained results strengthen the importance of including studies on economic viability and social impact in MPA processes (Grafton et al., 2011) aiming at attaining high levels of stakeholders compliance and involvement. The present approach would be further enriched by the inclusion of more MPA within and beyond the study area. This would allow to better investigate the patterns found and to propose more accurate strategies aiming at large scale coherence and effectiveness of MPA networks. Ultimately, the investment in strategies aiming at maximizing MPA performance are probably as important as the increase of MPA coverage.

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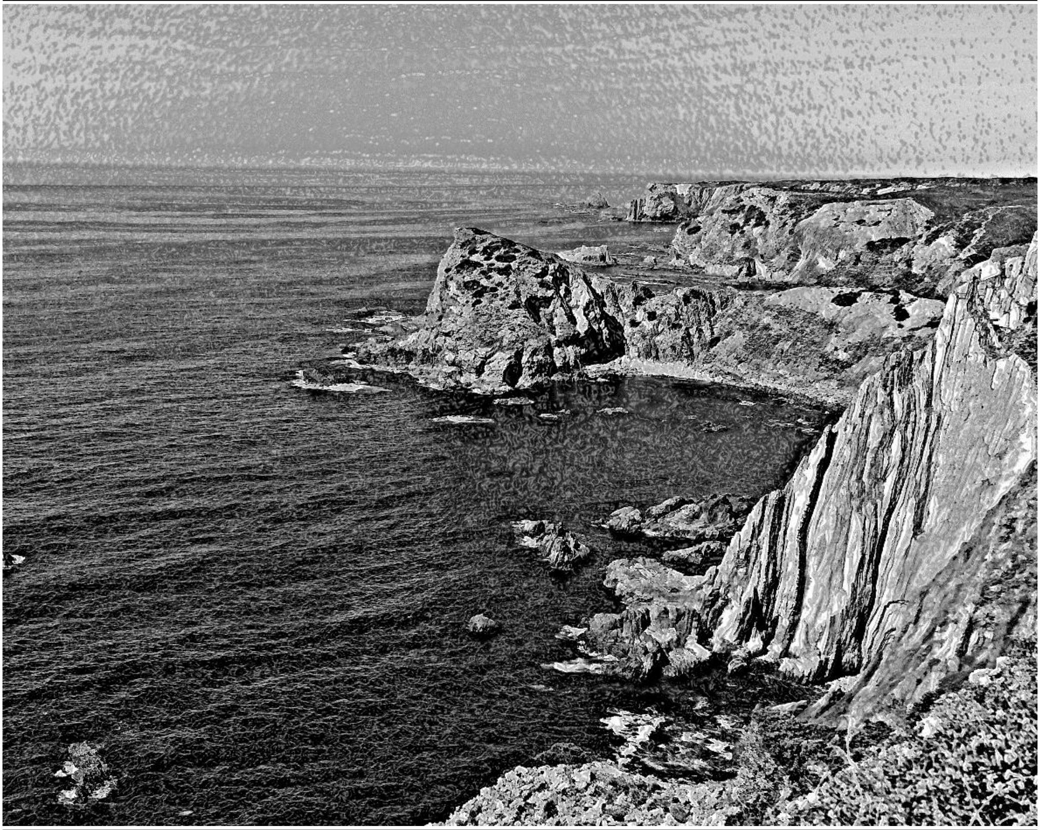
CHAPTER 6



Final remarks and future perspectives

Final remarks and future perspectives

Overall this thesis aimed to develop approaches that could maximize the suitability of existing data in the establishment of Marine Protected Areas (MPA) since the data sets available to support management decisions on marine conservation are usually not as adequate as would be desirable (Francour et al., 2001; Fraschetti et al., 2005). In this context, this thesis used or developed approaches based on the perspective of "use the best data available to produce the best science possible". That resulted in a series of tools that can be used by managers in several contexts of MPA planning, implementation and management. Data used were mostly from official institutions, peer reviewed publications and grey literature. In this context, methods that allow a good integration of information from different data sources, such as those included in the geographic information systems, multivariate statistics analyses and databases management techniques revealed to be a powerful combination. Furthermore, the consultancy of experts in several issues (from species biology to social sciences), with high experience in the topics investigated, namely MPA managers and fishers, showed to be a highly valuable complementary source of information to support conclusions and suggestions performed throughout the study. In addition, the use of data available from past projects, although not directed towards MPA, allows the optimization of costs inherent to its implementation. In a perspective of optimizing the cost-benefits balance, MPA processes should be based on methods that primarily enhance data and resources available and also allow the inclusion of new information directed to complement the existing datasets and thus fill the main gaps identified. Nevertheless, these processes do not encompass data from the period before MPA implementation which is often lacking. In this type of approaches the equilibrium between the accuracy and the level of information provided by the conclusions drawn is challenging. However, main findings and conclusions of this thesis were well supported by published literature. In addition, the inherent uncertainty was addressed and discussed in all the manuscripts.

The present thesis contributed with several methodologies that can be applied both in the planning of new MPA and in management and assessment of existing ones. All approaches were developed in the view of adaptive management (Ban et al., 2012; Grafton and Kompas, 2005; Walters and Hilborn, 1978) and thus have a dynamic nature and allow the inclusion of new data in order to improve the accuracy and reliability of results and thus the better support of future adaptability of management decisions. Overall, the methods and approaches proposed do not substitute the need to have well-designed surveys as soon as possible, but allows managers to make the best possible management decisions given the resources and information available. Furthermore, most of the main outputs from the approaches applied can be easily understood by managers and decision-makers (e.g. maps showing combined information), thus improving the potential for applicability of the results obtained, which is nowadays a major challenge facing the science of MPA.

Four essential aspects to the improvement of MPA effectiveness worldwide were discussed in a different chapter (from 2 to 5, respectively in this thesis): the first was a method to assess the spatial distribution of human activities and cumulative human pressures on a given spatial scale was developed. This method contributes to the efficient integration of data on human activities distribution and impact with ecological features which allows the identification of areas where the balance between overall negative impacts to economic activities (e.g. commercial fisheries, recreational activities) and overall conservation benefits of their exclusion from a given area is best. By minimizing conflicts, this approach can significantly contribute towards a high compliance of stakeholders with MPA and consequently to their major effectiveness (Christie, 2004; Rice et al., 2012). This approach also allows the identification of areas where levels of impacts are alarming, contributing to the faster implementation or enhancement of conservation measures. The second was a monitoring approach combining fishing effort, onboard data collection and official landings proved to be an effective tool for monitoring small-scale artisanal fisheries in MPA. Accurate data on the captures and fisheries dynamics evolution in a MPA contributes to the early detection of ecological and socio-economic problems allowing the rapid implementation of adequate management decisions to face these problems. Additionally, as resources for monitoring socio-ecological responses to MPAs are frequently scarce, the suitability of using landings data calibrated with information from both vessels and gear distribution and onboard validation of effort was explored. Since small scale fisheries are the economic activity most widely affected by MPA, this type of approaches are essential for MPA effectiveness assessments worldwide. Thirdly, a framework for the rapid assessment of potential for effectiveness and identification of major gaps in already implemented MPA was developed. The combination of species-habitat association, species' life history and functional groups and pressures affecting the MPA showed to be particularly useful as an alternative or a complementary support to early decisions for MPA management and the swift identification of priority actions needed to ensure the accomplishment of initial objectives or the suitability of their adaptation regarding effective management of a given MPA. And finally, the fourth was an analyses of overall MPA occurring in SW Europe showed that representativeness, coherence and global effectiveness of existing MPA are lacking. Given the global difficulty in implementing adequate methods to assess MPA effectiveness, managers' perceptions were integrated in a method where the high effectiveness of MPA showed to be related with high levels of stakeholders support, with suitable goals, management and enforcement.

Finally, the work developed showed that, though there are recent improvements in the implementation of MPA, we are still far from achieving the targets internationally considered as essential for an efficient protection and conservation of marine ecosystems worldwide. Global political and scientific based efforts are resulting in an improvement of MPA coverage worldwide. However effectiveness rates should be improved. Ultimately, this thesis underpinned the need to implement approaches that clearly identify conservation needs and established frameworks that could fill the major gaps found in order to achieve high rates of MPA effectiveness.

Future actions should focus the need to implement structured data collection programs that could contribute to the accurate establishment of new MPA that increase the coherence of already established areas. New areas should be implemented based on a global network perspective where the established areas are complementary. In the view of sustainable management, the establishment of new areas should be accurate and include stakeholders participation in order to minimize conflicts and achieve high effectiveness rates since the early stages of MPA.

Considering new and existing MPA there is a critical need to implement monitoring strategies that can inform scientists and managers about the MPA and MPA networks performance. These strategies should be standardized between geographic areas in order to allow global evaluation and cross validation of results obtained within and among areas. The inclusion of stakeholders volunteer participation into these monitoring plans would probably be beneficial since strategies totally based on scientific research would need heavy investments, only available in a minority of the contexts. Furthermore, the inclusion of stakeholders into MPA processes would contribute to their higher compliance and awareness of MPA benefits.

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POPH PROGRAMA OPERACIONAL POTENCIAL HUMANO

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