Universidade do Algarve

Faculdade de Ciências e Tecnologia

Multispecies spatial dynamics under different protection levels:

an evaluation of the effects and optimal design of the Luiz Saldanha Marine Park

Tese para a obtenção do grau de doutoramento em Ciências do Mar na especialidade de Ecologia Marinha / Thesis for the degree in Doctor of Philosophy in Marine Sciences, specialty in Marine Ecology

David Maria Aguiar Abecasis

Orientadores / Supervisors:

Professor Karim Erzini
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Declaração de autoria do trabalho

Declaro ser o autor deste trabalho, que é original e inédito. Os autores e trabalhos consultados estão devidamente citados no texto e constam da listagem de referências incluída.

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Abstract

The main objective of this study was to evaluate the potential benefits of the Luiz Saldanha Marine Park (LSMP) to three of the most important species for the local small scale artisanal fisheries: the white seabream, the Senegalese sole and the cuttlefish. Since this marine protected area (MPA) was implemented before the beginning of this study we also investigated if any changes had already occurred in the abundance or biomass of these species inside the MPA.

High site fidelity was observed in white seabream with all individuals tagged with V9 acoustic transmitters being detected over 50% of the time (average of 85%). Although some Senegalese sole showed high site fidelity the majority of the individuals tagged were more transient. Cuttlefish demonstrated no site fidelity to the study area, spending a maximum of 39 days inside the monitored area. The home range areas (95% KUD) ranged between 0.43 and 1.28 km^2 (average of 0.65 km^2) for white seabream and between 0.54 and 2.22 km^2 (average of 1.19 km^2) for Senegalese sole. The results obtained for cuttlefish were not coherent given the low site fidelity presented by this species. White seabream demonstrated a clear preference for rocky bottoms habitats while Senegalese sole prefer sandy bottoms, especially medium grain size sand.

The results of experimental fishing trials revealed no significant differences in the abundance or biomass of Senegalese sole or cuttlefish between the periods before and after the implementation of the LSMP.

The species distribution models obtained with Maxent indicate that most of the LSMP presents suitable habitats for cuttlefish but only a small proportion is suitable for the Senegalese sole and white seabream. The Senegalese sole suitable habitats are mainly found in sandy areas facing south whereas the white seabream suitable habitats are restricted to the narrow rocky areas that border the coastline.

Our results suggest that the existing marine reserve may offer appropriate protection for Senegalese sole and white seabream. However, given the low site fidelity and the large movements of cuttlefish the LSMP does not provide adequate protection for this species.

According to the results of the different scenarios obtained with Marxan the design of the LSMP could be improved in order to offer more protection to habitats that are not sufficiently protected with the current design.

Keywords: Marine protected areas, acoustic telemetry, experimental fishing, species distribution models, Maxent, Marxan.

Resumo

Nas últimas décadas, a pressão antropogénica sobre os ecossistemas marinhos tem aumentado em magnitude e diversidade, causando uma degradação drástica destes ambientes. Este facto tem levantado sérias preocupações e levou a apelos para a implementação de medidas que visam proteger, preservar e restaurar os ecossistemas marinhos. Uma das medidas que tem recebido maior atenção e apoio por parte da comunidade científica nos últimos 30 anos é a implementação de áreas marinhas protegidas.

As áreas marinhas protegidas podem apresentar uma grande variedade de desenhos, gestão e extensão da proteção, dependendo das características biológicas, físicas e sociais do lugar onde são implementadas e os objectivos que pretendem alcançar. No entanto, de acordo com a IUCN (International Union for Conservation of Nature) uma área marinha protegida é "um espaço geográfico claramente definido, reconhecido, dedicado e gerido, através de meios legais ou outros, para alcançar a conservação a longo prazo da natureza incluindo os serviços ecossistémicos associados e os valores culturais." De um modo geral, a sua aplicação visa atingir um ou ambos dos seguintes objetivos: gestão dos recursos marinhos e conservação da biodiversidade e habitats .

Na costa da Arrábida, situada na área metropolitana de Lisboa e próxima do estuário do Sado, foi estabelecida, em 1998, uma área marinha protegida designada Parque Marinho Luiz Saldanha (PMLS). Esta costa é muito procurada por vários sectores relacionados com o mar, desde actividades náuticas de lazer a actividades como a pesca e a apanha. Constitui uma zona de transição, onde muitas espécies com afinidades de água fria e água quente atingem os seus limites norte e sul da distribuição, respectivamente. Com mais de 1.200 espécies registadas esta zona é considerada um "hotspot" europeu da biodiversidade marinha. Para além disso, o estuário do Sado é conhecido como uma importante área de postura e/ou berçário para várias espécies de peixes e cefalópodes. Devido ao conflito entre as actividades humanas e os valores naturais desta área, o PMLS foi estabelecido com a intenção de

proteger a elevada biodiversidade e alguns dos habitats, bem como promover a sustentabilidade da pesca artesanal local.

O principal objetivo deste estudo foi avaliar os potenciais benefícios do PMLS para três das espécies mais importantes para a pesca artesanal local: o sargo *Diplodus sargus*, o linguado senegalês *Solea senegalensis* e o choco *Sepia officinalis*. Também se investigaram possíveis alterações na abundância e/ou biomassa destas espécies no PMLS ao longo do período de duração deste estudo.

Foi utilizada telemetria acústica passiva para determinar padrões de movimento, o tamanho das áreas de actividade e a fidelidade ao local das espécies em estudo. Vinte receptores acústicos, que cobriram uma área de aproximadamente 2 km^2, detectaram a presença de indivíduos previamente marcados com transmissores acústicos (7 chocos, 20 sargos e 17 linguados). Para identificar possíveis alterações na abundância ou biomassa das espécies estudadas foi utilizada a pesca experimental, realizada 2 vezes por ano nos meses de Primavera e Outono. A pesca experimental foi efectuada com recurso à arte do tresmalho, seguindo as especificações utilizadas pela frota artesanal local. Foram analisados um total de 106 lances de pesca (de 500m) realizados entre 2007 e 2011.

Uma elevada fidelidade ao local foi observada para o sargo, com todos os indivíduos marcados com transmissores acústicos V9 a serem detectados mais de 50% do tempo (média de 85%). Embora alguns linguados tenham demonstrado uma elevada fidelidade ao local de estudo, a maioria dos indivíduos marcados foi mais transiente. Pelo contrário, os chocos não demonstraram fidelidade ao local com apenas um exemplar a ser detectado por um período máximo de 39 dias. Os restantes exemplares marcados foram detectados menos de 15 dias. Para além disso, nenhum choco voltou a ser detectado após ter abandonado a área monitorizada. A recaptura dum choco no interior do estuário do Sado, a mais de 15 km do local de marcação evidencia as migrações para o estuário.

As áreas de actividade (95% KUD) variaram entre 0,43 e 1,28 km^2 (média de 0,65 km^2) para os sargos e entre 0,54 e 2,22 km^2 (média de 1,19 km^2) para os linguados. Os resultados obtidos para o choco não são relevantes dada a baixa fidelidade da espécie ao local.

Os sargos demonstraram uma clara preferência por fundos rochosos enquanto os linguados exibiram uma preferência por fundos arenosos, especialmente os constituídos por areia média.

Os resultados da pesca experimental não revelaram diferenças significativas na abundância ou biomassa de linguados ou chocos entre os períodos antes e após a implementação do PMLS.

A combinação destes resultados sugere que a actual área marinha protegida pode oferecer proteção adequada para o sargo e potencialmente também para o linguado. No entanto, dada a baixa fidelidade ao local e os grandes movimentos do choco, o PMLS não oferece uma proteção adequada para esta espécie.

Os dados de telemetria foram posteriormente utilizados em conjunto com as variáveis "habitat", "profundidade", "aspecto", "inclinação", "curvatura" e "distância a fundo rochoso", de modo a determinar modelos de distribuição para cada uma das espécies utilizando o método de máxima entropia (Maxent). Estes modelos de distribuição sugerem que grande parte do habitat do PMLS é adequado para chocos, mas apenas uma pequena parte do PMLS apresenta habitats adequados para os sargos e os linguados. No caso dos linguados, estes habitats estão praticamente restringidos na parte voltada a Sul do PMLS. O habitat adequado para os sargos é composto por uma estreita franja que contorna os recifes rochosos praticamente ao longo de toda a linha de costa do PMLS.

Posteriormente, utilizou-se o Marxan para testar diferentes soluções para o design do PMLS. De acordo com os resultados dos diferentes cenários obtidos com Marxan, o desenho do PMLS poderia ser melhorado de modo a oferecer mais proteção a alguns dos habitats que, com o presente desenho, apenas estão parcialmente protegidos. No entanto, a localização da zona "no-take" está situada no local considerado como óptimo.

Palavras-chave: áreas marinhas protegidas, telemetria acústica, pesca experimental, modelos de distribuição das espécies, Maxent, Marxan.

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Abecasis D, Horta e Costa B, Afonso P, Gonçalves E, Erzini K (in preparation) Small home ranges of a commercial fish (*Diplodus sargus*) explain early reserve effects

Abecasis D, Afonso P, Erzini K (in preparation) Marine protected areas: the need for multispecies site fidelity and movement patterns information

Abecasis D, Afonso P, Erzini K (submitted) Combining multispecies home range and distribution models to evaluate the optimal design of MPAs

Abecasis D, Afonso P, Erzini K (submitted) Towards an adaptive management of the Luiz Saldanha Marine Park: influence of different conservation targets and costs on the design of no-take areas

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Chapter 1

General introduction

Marine protected areas

Over the last decades, it has become common knowledge that human impacts can cause drastic degradation of marine ecosystems and that these impacts have been increasing in magnitude, diversity and rates of change (Jackson et al. 2001, Lotze et al. 2006, Palumbi et al. 2008). Worldwide, scientists have documented the devastating effects of overexploitation (Pauly et al. 2005), habitat loss (Short & Wyllie-Echeverria 1996, Gray 1997), pollution (Jackson et al. 2001, Shahidul Islam & Tanaka 2004), invasive species (Molnar et al. 2008) and the negative consequences of increasing coastal development (Bulleri & Chapman 2010) and tourism (Hall 2001). According to the FAO's latest report, around 57% of the worlds fish stocks are fully exploited and more than 29% are overexploited (FAO 2012). These factors have raised serious concerns and led to calls for the implementation of measures that aim to protect, preserve and restore marine ecosystems (Lubchenco et al. 2003).

One measure that has received much attention and support from the scientific community over the last thirty years is the implementation of marine protected areas (MPAs) (Roberts & Polunin 1991, Guénette et al. 1998, Pauly et al. 1998, Kaiser 2011). Although they are not a new approach to conservation, MPA research and implementation have escalated since the 1990's (Roberts & Polunin 1991, Agardy et al. 2003, Bogaert et al. 2009, McCay & Jones 2011).

The IUCN (International Union for Conservation of Nature) defines MPAs as "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". MPAs present a wide range of design, management and extent of protection, depending on the biological, physical and social characteristics of the place where they are being implemented and the goals they are intended to achieve. Most authors suggest their implementation to achieve biodiversity and habitat conservation, marine resource management, or both (Roberts & Polunin 1993, Agardy et al. 2003, Pauly et al. 2005, Claudet et al. 2008, Russ & Alcala 2011).

Biodiversity and habitat conservation includes an array of specific objectives such as protecting rare and vulnerable habitats and species, restoring ecological functions, encouraging research and education, maintaining aesthetic values and traditional uses and promoting sustainable tourism and the area's nonconsumptive values (Jones 2001). MPAs focusing on marine resource management have been strongly recommended over the last decades as an alternative and/or complement to traditional fisheries management methods (Roberts & Polunin 1991, Pauly et al. 1998, Murawski et al. 2000, Pauly et al. 2005, Roberts et al. 2005, Kaiser 2011). The ecological benefits that MPAs may provide to fisheries include increases in stock abundance, age/size composition and spawning stock biomass, the enhancement of recruitment in fished areas, the restoration of healthy trophic levels and spillover to adjacent areas (Edgar & Barrett 1999, Murawski et al. 2000, Barrett et al. 2007, Claudet et al. 2008). In fact, recent studies have demonstrated the effectiveness of MPAs in the management of fisheries in adjacent areas (e.g. Abesamis et al. 2006, Stobart et al. 2009, Goñi et al. 2010, Vandeperre et al. 2011) and in the recovery of habitats (Leleu et al. 2012). Hence, one of the advantages of MPAs is that, unlike the traditional 'single species' approach, MPAs offer an ecosystem based approach to conservation and fisheries management (Lubchenco et al. 2003, Crowder & Norse 2008).

While there is widespread recognition of the potential of MPAs to achieve conservation and fisheries management goals, the proper functioning of MPAs is frequently impaired by multiple knowledge gaps about key ecological aspects underlying the functioning of MPAs, and their potential benefits.

Hence, it is critical to increase our understanding of the dynamics of MPAs and their performance as fisheries management tools. In particular, it is important to fully understand species movement patterns and habitat use, and the amount of protection offered by MPAs. Filling these gaps will allow a better evaluation of the effectiveness of MPAs and the forecasting of their optimal designs.

Species movement patterns and habitat use

MPAs rely on spillover (export of adults/juveniles) and/or recruitment subsidy (export of larvae) in order to increase the yield of nearby fisheries (Bohnsack 1993, Sale et al. 2005). Studies on fish movement patterns have increased in the past decades and, in some occasions, successfully demonstrated the spillover effect (Abesamis et al. 2006, Goñi et al. 2008, Stobart et al. 2009). Ideally, this information should be taken in consideration when defining the location and extent of MPAs and even later during the management process. Yet, this information is rarely available before the implementation of MPAs and therefore not integrated into the optimal design.

Together with information on fish movement patterns, knowledge of species habitat use is crucial for the design and adaptive management of MPAs, so as to guarantee not only the existence of spillover but also that the size of the reserve unit is large enough to provide protection during large periods of their life cycle (Glazer & Delgado 2006, Grüss et al. 2011). The existence of preferred habitats should also be investigated to make sure MPAs provide the adequate habitats for the species involved.

In most cases there is no empirical information on the size of suitable habitats where species are effectively protected from local fisheries, even several years after the implementation of MPAs. Acoustic telemetry is one of the most widely used methods to track marine species, as it provides long-term, fine scale spatio-temporal data on individual movement and home range. However, little has been done yet and there is no consensus on how to translate such individual data - the typical output of telemetry studies - into the more relevant population scale projection when evaluating the effectiveness of protection provided from existing MPAs or forecasting their optimal designs. Even more striking is the general lack of information about the protection offered by MPAs to target species in terms of exposure to nearby fisheries.

MPA assessment

The assessment of MPAs has come a long way from the mostly qualitative early studies with poorly replicated designs and inappropriate statistical methods. More recently, studies carried out with appropriate experimental designs that include replicated controls have become common (Claudet & Guidetti 2010). Nevertheless, some authors argue that, in most cases, MPA evaluation has been undertaken without the use of appropriate sampling designs that enable scientists to unambiguously detect changes in species abundance and/or biomass (Fraschetti et al. 2002, Guidetti 2002, Claudet & Guidetti 2010).

Amongst the different statistical approaches, the before-after control-impact (BACI) and the beyond BACI designs (Underwood 1991, 1994, Smith 2002) are considered some of the best to infer on MPAs effects (Fraschetti et al. 2002, Guidetti 2002). These methods and some of their variations, such as after control impact, have been used regularly in recent studies on the effects of MPA (e.g. Cole et al. 2011, Moland et al. 2011, Bertocci et al. 2012, Horta e Costa et al. 2013).

Regardless of the existence of positive effects after the implementation of an MPA, there are some aspects that should be taken in consideration and objectively discussed, such as the temporal scale of sampling and the enforcement/compliance level of the management rules. The very high natural variability of marine ecosystems and the difficulties to fully understand it make it even more difficult to discern stochastic effects from those of the protection given by MPAs. Therefore, the temporal scale of sampling should, ideally, be large enough to accommodate the natural fluctuations in species' abundance, as several factors beyond fishing mortality may have a large effect on species' reproductive success (Guidetti 2002). The level of compliance of the management rules may also have an important role. When there is no compliance then no protection effects are to be expected since there is no effective change in fishing pressure. As an example, Lipej et al. (2003) suggested that the lack of differences in fish diversity and abundance between protected and unprotected areas in Slovenia is due to the lack of enforcement.

Acoustic telemetry is one of the most commonly used tools to investigate fish home range areas, site fidelity and movement patterns (Abecasis & Erzini 2008, Abecasis et al. 2009, March et al. 2010, Mason & Lowe 2010, Afonso et al. 2011, Abecasis et al. 2013). The use of passive acoustic telemetry, where acoustic receivers are placed in the marine environment to detect individuals tagged with acoustic transmitters, allows long term studies on animal movements in the wild without influencing their behavior (Heupel et al. 2006, Koeck et al. 2013b).

Species distribution models (SDM) are extremely valuable tools for the implementation and management of MPAs (Leathwick et al. 2008, Carvalho et al. 2010). There are several approaches to obtain these models including generalized linear or additive models (GLMs or GAMs), multivariate adaptive regression splines (MARS), boosted regression trees and maximum entropy modeling (Maxent) (Elith & Leathwick 2009, Newbold 2009). The low data requirements and the ease of integration with GIS analysis together with its superior performance have made Maxent one of the most widely used software for SDM (Elith et al. 2006, Elith & Leathwick 2009).

The use of software that assists in the implementation phase of MPAs, especially in the location, size and design process is becoming more frequent (Loos 2006, Green et al. 2009, Allnutt et al. 2012, Ban et al. 2013). In most cases these tools provide managers with options based on a wide variety of input data which can include physical, biological and socio-economical information. Among the most used conservation planning software is Marxan. It uses simulated annealing to create near optimal solutions for reserve systems that achieve determined conservation targets while minimizing the cost (e.g. fisheries effort or catch, socio-economic costs) of including these areas in no-take areas. Marxan is the most widely used software due to its ability to include different types of data and the easy connection with geographic information systems (GIS).

Study Area - The Luiz Saldanha Marine Park

The Arrábida coast, located in the metropolitan area of Lisbon and near the Sado estuary, is in high demand by various marine-related sectors, from nautical leisure activities to fishing and shellfish harvesting. This area is in a transitional biogeographic zone where many species of cold and warm-water affinities reach their southern and northern limits of distribution, respectively (Gonçalves et al. 2003). With more than 1200 registered species it is a European hotspot of marine biodiversity (Saldanha 1974, Henriques et al. 1999, Gonçalves et al. 2003). Moreover, the nearby Sado estuary is known as an important nursery area for several fish and cephalopod species (Serrano 1992, Cabral et al. 2007, Neves et al. 2009, Vasconcelos et al. 2010, Vinagre et al. 2010).

The conflict between human activities and the natural values of this area has led to the need to implement regulations. For this reason, the Luiz Saldanha Marine Park (LSMP) was established in 1998, covering approximately 53 km^2 and stretching along 38km of coastline (Figure 1.1). The LSMP is located in an area of high fishing importance that encompasses the fishing port of Sesimbra and is close to the fishing port of Setúbal, upstream in the Sado river estuary. The main objectives of this MPA are to protect the coastal biodiversity of this particular area and to promote the sustainability of local artisanal fisheries and fishers' livelihoods (Gonçalves et al. 2003).

The LSMP is divided into three different protection levels (Figure 1.1): no-take areas where all human activities are prohibited; partially protected (or buffer) areas where only the use of octopus traps, jigging and handlines is allowed; and complementary protection areas where the use of all traditional fishing gears is allowed for fishing vessels smaller than 7m. In addition, all fishing vessels need a specific permit to operate inside the LSMP, spearfishing is prohibited in the entire MPA and recreational angling is only allowed within the complementary areas.

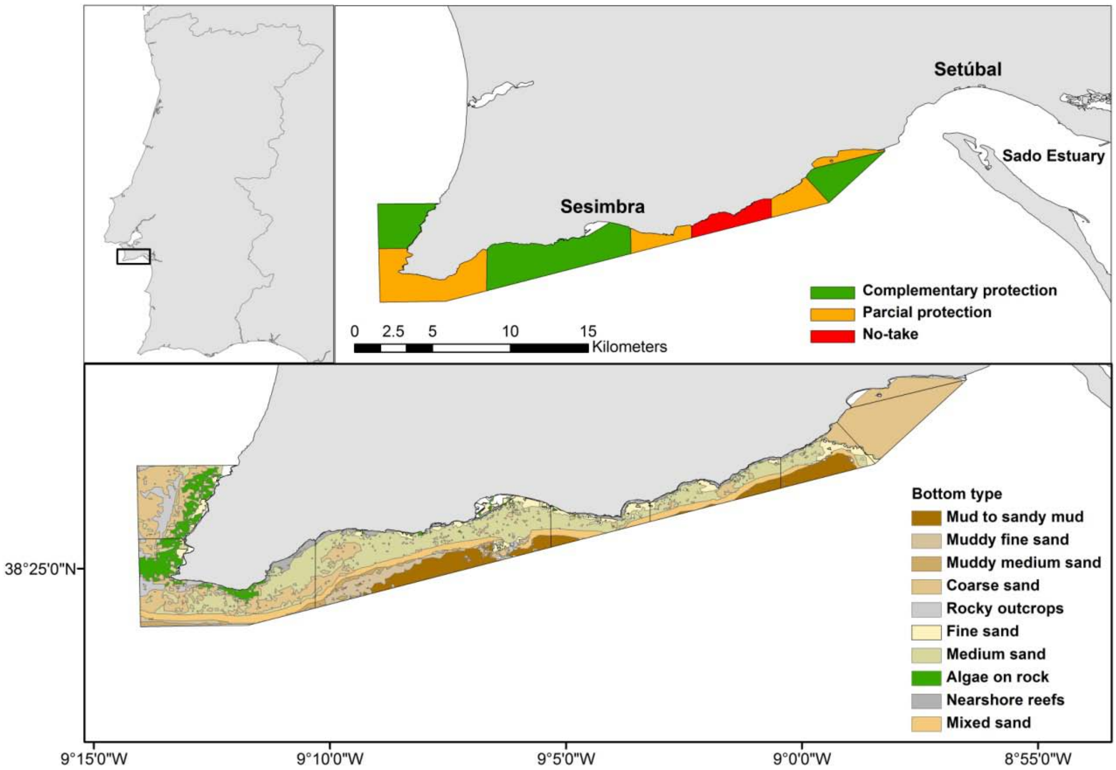

Figure 1.1. Map with the location of the study area.

While the LSMP was designated in 1998, its final planning design was only achieved in 2005 (Figure 1.2). The LSMP's regulations were gradually introduced: in August 2006, the partially protected area of Portinho da Arrábida and the eastern half of the no-take area were set as a partially protected area; in August 2007 the remaining partially protected areas were established and the western half of the no-take area was set as a partially protected area; by August 2008 the eastern half of the total protection area became a no-take zone; finally in August 2009 the whole total protection area became regulated as no-take (Figure 1.2).

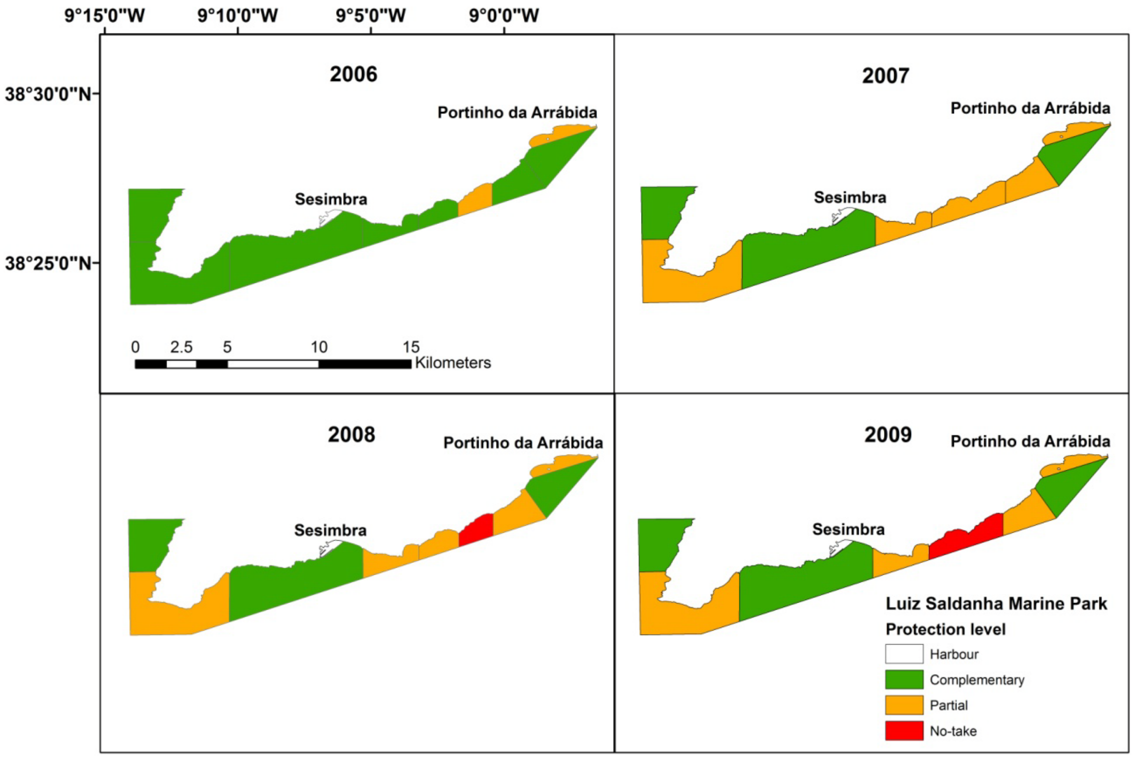

Figure 1.2. Evolution of the regulations in the Luiz Saldanha Marine Park

Objectives

The main objectives of this study are: a) to evaluate the potential of the recently established Luiz Saldanha Marine Park as a management tool for the local populations of commercially and ecologically important species b) to determine how this potential varies with contrasting life history characteristics 3) to assess the extent by which the amount of protection offered by the MPA and the suitability of its current design influence the local management of the three selected species.

We used a combination of passive acoustic telemetry, experimental fishing, species distribution models and conservation planning models to achieve these goals and, in particular, to answer the following questions:

1 - What is the size of cuttlefish, Senegalese sole and white seabream home range areas?

2 - Do these species present site fidelity to the marine reserve?

3 - Do they favour any particular habitat within the study area?

4 - Is it possible to already detect the effects of the marine reserve implementation on the abundance and/or biomass of Senegalese sole or cuttlefish?

5 - Is the actual design of the LSMP appropriate for the protection/management of the three selected species?

6 – Will changes in the size and location of no-take areas improve the efficiency of the LSMP?

Study species

Marine reserves should ensure protection for a wide range of species that have different life history traits and different economic and ecological values, and also that occupy different trophic levels and environments (Palumbi 2004, Afonso 2007, Claudet et al. 2010). For this reason, this study focuses on three species with contrasting life-history, ecological traits, economic values and resilience: cuttlefish *Sepia officinalis* (Linnaeus, 1758), Senegalese sole, *Solea senegalensis* Kaup 1858; and white seabream *Diplodus sargus* (Linnaeus, 1758). Given their different characteristics we believe that this study is relatively representative in that it allows the perception of MPA benefits for a wide range of species. Additionally, despite the large number of studies on MPA efficiency, the vast majority of these studies focus on reef fish species whereas very few have investigated cephalopods or flatfish (Lester et al. 2009, Horta e Costa et al. 2013), as in the case.

The cuttlefish

The cuttlefish is a nekto-benthonic cephalopod that occurs over a wide variety of bottoms (sand, mud and sea grass beds) between the coastline and approximately 200 m depth, although it is more abundant in the upper 100m (Guerra 2006). It is found from Northern England to the northwestern coast of Africa including the Mediterranean (Guerra 2006). It is a semelparous (reproducing only once in a lifetime) fast growing species with a lifetime between 1 and 2 years (Le Goff & Daguzan 1991b). These characteristics suggest that this species has high resilience to fishing pressure (Musick 1999). The first year breeders seem to be more common in the south of their geographical distribution, whereas second year breeders constitute the vast majority of the population around its northern distribution limit (Guerra & Castro 1988, Gauvrit et al. 1997).

The first tagging experiments with cuttlefish were carried out in the 1980's by Ezzeddine-Najai (1997) and by Le Goff & Daguzan (1991a) using plastic fanion tags. In a tag-recapture study carried in the Gulf of Tunis, all recaptured animals migrated towards the shallower coastal areas where the main habitats were sand/mud bottoms and sea grass beds (Ezzeddine-Najai 1997). During this study, the minimum distance travelled by cuttlefish was 4 km in 2 days and the maximum 25 km in 21days (Ezzeddine-Najai 1997). The maximum period between tagging and recapture was 89 days with the cuttlefish covering almost 13.8km (Ezzeddine-Najai 1997). These known migrations enable cuttlefish populations to exploit the temporal and spatial variability of productive systems and fluctuating populations of prey (Rodhouse & Nigmatullin 1996). Royer et al. (2006) suggested that coastal zone management alone is not sufficient to ensure the sustainable exploitation of cuttlefish due to its high mobility. In fact, several authors suggest that MPAs should be used in conjunction with traditional management measures such as gear restrictions, seasonal closures, or catch quotas especially when species present high mobility (Allison et al. 1998, Shipp 2003, Hilborn et al. 2004).

The Senegalese sole

The Senegalese sole is a coastal benthonic fish species that occurs in sandy or muddy bottoms in depths up to 65m. It is common between the Gulf of Biscay and the coasts of Senegal but, according to Desaunay et al. (2006), its distribution is expanding north in the NE Atlantic. There are no studies describing the dispersal or movement patterns of this species, and the knowledge of its ecology and biology is relatively scarce, with most studies focusing on juveniles (e.g. Cabral 2000a, Cabral 2000b, Vinagre et al. 2006, Vinagre et al. 2008). According to Fishbase (Froese & Pauly 2013) this species presents low resilience to fishing pressure.

The Senegalese sole is one of the most important flatfish resources for fisheries in Portugal, yet it has not been properly assessed (Teixeira & Cabral 2009). Nevertheless, Teixeira & Cabral (2009) concluded that, even though there are no evaluations of flatfish stock status in Portugal, there is probably overexploitation of these fisheries resources. In fact, a decreasing trend in the landings per unit effort (LPUE) has been observed for the period between 1992 and 2005 (Teixeira & Cabral 2009).

The sole fishery is one of the most important for the local artisanal fisheries of Sesimbra and Setúbal, the two most important fishing ports located near the LSMP (Batista et al. 2009). In addition, the nearby Sado estuary is an important nursery area for the Senegalese sole (Cabral 2000a, Vasconcelos et al. 2008), as otolith microchemistry shows that many of the fish caught in coastal waters originate from the Sado nursery (Vasconcelos et al. 2011, Tanner et al. 2013).

The white seabream

The white seabream is a demersal fish species usually found at depths less than 50 m on a variety of sea bottoms, including rocky and sandy bottoms as well as seagrass beds. It is one of the most important coastal fish resources in southern European countries, where it is mainly targeted by recreational fishermen and small scale fisheries (FAO 2012). Its age, growth, reproduction and diet have been widely studied (e.g. Gordoa & Moli 1997, Morato et al. 2003, Figueiredo et al. 2005, Leitão et al. 2007, Mouine et al. 2007, Abecasis et al. 2008). Based on the biological parameters obtained by Abecasis et al. (2008) and Morato et al. (2003) this species presents a medium resilience to fishing pressure (Musick 1999). In Portugal, where it is a main target species for recreational rod and line anglers (Rangel & Erzini 2007, Veiga et al. 2010), commercial landings have been declining since the late 1980's (Directorate General of Fisheries and Aquaculture).

A wide variety of studies have focused on the movements, site fidelity and activity patterns throughout the life-history of the white seabream (Abecasis et al. 2009, D'Anna et al. 2011, Di Franco et al. 2012, Abecasis et al. 2013, Koeck et al. 2013). These studies show that the scale of the dispersion pattern of the white sea bream decreases with age: larval dispersion occurs at the scale of 100-200 km; post-settlement dispersion takes place over distances under 30 km, although a portion of 22% to 50% of settlers remains in the same area (Di Franco et al. 2012); and adult fish present high site fidelity with home ranges between 0.65 and 3.93 km^2 (Abecasis et al. 2013).

The movement patterns, site fidelity and habitat use of the white seabream have been previously described for juveniles in a coastal lagoon (Abecasis et al. 2009) and for adults around artificial and natural reefs (Lino et al. 2009, D'Anna et al. 2011, Abecasis et al. 2013, Koeck et al. 2013a). The discrepancy in patterns of activity and habitat use of adult white seabream between some of these studies emphasizes the need for local studies. Nevertheless this information suggests that, due to their high site fidelity as adults and to the dispersal as larvae and post-settlers, white seabream may benefit from the implementation of MPAs. In fact, previous studies on the effectiveness of MPAs as a fishery management tool suggest that marine reserves may provide long term benefits to adjacent local fisheries through the increase of white seabream production (Bennett & Attwood 1991, Lloret & Planes 2003, Tuya et al. 2006, Horta e Costa et al. 2013). This was also observed for other sparids elsewhere (Bennett & Attwood 1991, Willis et al. 2003)

Thesis structure

This thesis comprises 8 chapters. In the first chapter we present an introduction to the main topic and an overview of the study area and selected study species. Chapters 2 through 4 focus on the home range, site fidelity and habitat use of each study species through the use of passive acoustic telemetry. Chapter 5 evaluates the effects of the LSMP on the abundance and biomass of Senegalese sole and cuttlefish, by analyzing data from experimental fishing trials. In chapter 6 we present distribution models for each of the study species estimated through Maxent and determine the size of suitable areas that provide full protection as well as their exposure to local fisheries. In chapter 7 we analyze different protection scenarios using Marxan to identify the best location, size and design of no-take areas in the LSMP. Finally, in chapter 8 we synthesize the main findings of this study and present general conclusions and suggestions for future studies.

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Chapter 2: Movements and site fidelity of *Sepia officinalis* in the Luiz Saldanha Marine Park

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Chapter 2

Movements and site fidelity of *Sepia officinalis* in the Luiz Saldanha Marine Park

Abstract

Knowledge on species site fidelity and movement patterns is key for evaluating the potential of marine reserves as fishery management tools. The cuttlefish is one of the most important resources for the artisanal fisheries operating in the vicinity of the recently established Luiz Saldanha Marine Park. Site fidelity and movement patterns of cuttlefish in the Luiz Saldanha Marine Park were investigated using passive acoustic telemetry.

The results show that cuttlefish have no site fidelity to the study area. The maximum amount of time a cuttlefish spent inside the monitoring area was 39 days with all other individuals remaining less than 15 days. Movements over 15 km towards the nearby estuary were observed with the direction of the movement being independent of the tide direction. Although not conclusive, the results suggest that cuttlefish are more active during the night time.

The fact that cuttlefish have low site fidelity inside the reserve and large movements across and beyond the study area suggest that small coastal marine reserves such as the LSMP are not effective in providing long term protection to cuttlefish populations and, probably, those of other short-lived, highly mobile cephalopods.

Introduction

Understanding spatial and temporal movement patterns of marine species is key to ensure their adequate management. This is especially true in the context of marine protected areas (MPAs). As tools for the conservation and management of biodiversity and fisheries, MPAs are increasingly seen as a way of overcoming uncertainty in fisheries management based on conventional measures (e.g. effort control, size and gear restrictions). Yet, MPA design itself has also rested upon substantial uncertainty, mainly because information on the species habitat use and connectivity of their (sub)populations is generally lacking.

Several authors recognize the need to incorporate specific information such as activity patterns, residency, habitat use and connectivity of (sub)populations into the design and planning phases of MPA establishment (e.g. Botsford et al. 2003, Glazer & Delgado 2006). This is considered essential to ensure an appropriate MPA design that maximizes its beneficial mechanism operating inside and outside the protected areas (Grüss et al. 2011). However, in most cases, this information is either unavailable or not taken into consideration during the design and planning of MPAs. Nevertheless, even in contexts where MPAs have already been implemented, this information can be extremely useful as a tool for adaptive management (Grafton & Kompas 2005, Pomeroy et al. 2005). This study aims to evaluate the site fidelity, activity patterns and home range of the cuttlefish *Sepia officinalis* in the recently established Luiz Saldanha Marine Park (LSMP).

The cuttlefish is one of the main target species of the small-scale fisheries that operate in the LSMP and the nearby Sado estuary (Serrano 1992, Batista et al. 2009). Previous studies on the distribution of cuttlefish for this region suggest a seasonal migration between the two main habitats occupied by the species in the area, as cuttlefish abundance is higher within the estuary during spring and summer but increases in nearby coastal waters, where the MPA is located, during autumn and winter (Batista et al., 2009; Neves et al., 2009). Seasonal migrations of cuttlefish between shallow and deeper waters have also been described for other regions (e.g. Wang et al. 2003, Guerra 2006). These migrations can be due to ontogenic related habitat shifts and/or the onset of the spawning season, which only occurs once during the lifetime of the cuttlefish.

Neves et al. (2009) found that, inside the Sado estuary, the cuttlefish' spawning period ranges from February to June, whereas Serrano (1992) found mature individuals in coastal areas all year round. Both authors suggested a size-related spawning habitat selection, in which smaller individuals would prefer to spawn inside the estuary while larger individuals, which are rarely caught inside the estuary, spawn in adjacent coastal waters (Neves et al. 2009). It is therefore possible that such larger and more fecund cuttlefish may be more site attached, in which case an MPA could provide effective protection and have a direct benefit for the reproductive productivity of the local population. The information on the spatial and temporal movement patterns of cuttlefish that this study aims to unveil will demonstrate the potential of using MPAs to effectively protect cuttlefish populations. It will also provide valuable insight as to the suitability of the LSMP in protecting local cuttlefish populations.

Materials and Methods

Study area

This study took place in the Luiz Saldanha Marine Park (LSMP), which is located in the Setúbal Peninsula (Portugal). The LSMP covers an area of approximately 53 km^2 and stretches over 38 km of coastline (Figure 2.1). It includes a narrow stretch of rocky reef bottom down to depths of about 15 m and a larger stretch of soft substrates (sand and mud) down to the 100 m bathymetric.

Three different types of protection zones can be found in the LSMP: no-take, partial protection and complementary protection. The no-take zone comprises 4.3 km^2 where no extractive or recreational activities are allowed. Octopus traps and jigs are allowed within the four partial protection zones, which cover a total of 21 km^2. Finally, within the three complementary protection zones, totalling 28 km^2, traditional fishing gears for vessels smaller than 7m are allowed, along with recreational angling. In addition, all fishing vessels need a permit to operate inside the LSMP and spearfishing is prohibited in all zones.

Acoustic telemetry

Trammel nets, similar to those used by local fishermen, were used to capture cuttlefish, during the November 2010 experimental fishing campaign (see Chapter 5 for more details). Seven cuttlefish were measured and tagged with an acoustic transmitter (Vemco, Canada), following an adaptation of the method used by Aitken et al. (2005). A small (9mm long and 5mm wide) section of stainless steel tube was attached to each end of the acoustic transmitter, using epoxy resin. The transmitters were then screwed into the cuttlebone through the stainless steel tubes. After the attachment of the transmitter, cuttlefish were released at the site of capture. The whole procedure took less than 1 minute.

Two different transmitter types were used: V7 (7 mm in diameter and 22.5 mm in length), with random emission intervals between 30 and 90 seconds and an estimated lifetime of 95 days, and V9 (9 mm in diameter and 29 mm in length) with random emission intervals between 15 and 45 seconds and an estimated lifetime of 151 days. Preliminary tests showed detection ranges of approximately 300 m for V9 transmitters and 100 m for V7 transmitters.

The presence of tagged cuttlefish was monitored through the use of 18 acoustic receivers (Vemco VR2 and VR2W) moored in two lined arrays parallel to the coastline (Figure 2.1). The inner line comprised nine receivers deployed at depths between 8 and 14 m, thus monitoring both rocky and sandy habitats. The outer line comprised nine receivers deployed at depths between 17 and 21 m, mainly monitoring muddy bottoms. The monitoring period lasted from October 27th 2010 to September 15th 2011, outlasting the duration of the acoustic transmitters by more than twice their expected lifetime. Six receivers from the outer line and two receivers from the inner line were lost after the data download that took place on November 17 2010, most likely due to illegal fishing activities. The loss of these receivers was noticed during the following download, which took place in April 2011, and forced a change in the array design thereafter, with the deployment of four new receivers on May 9th 2011 (Figure 2.1).

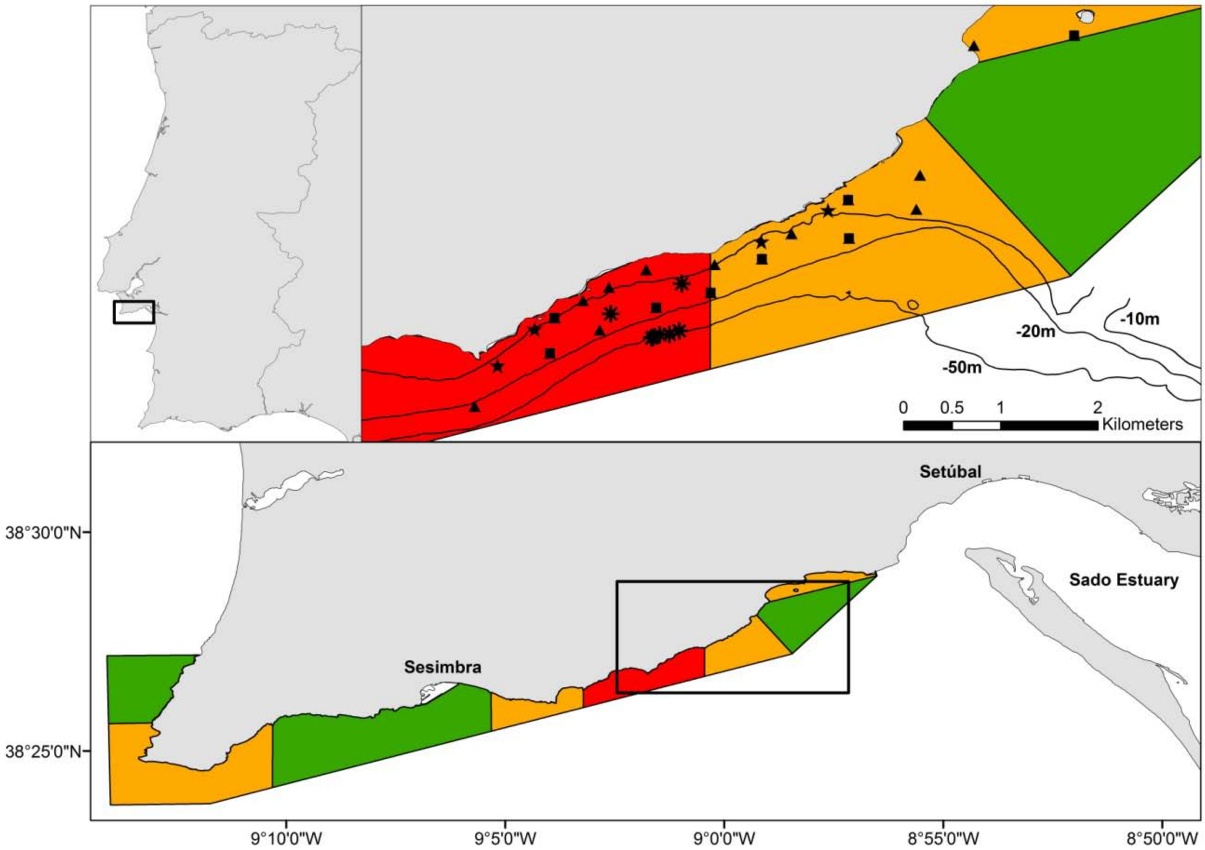

Figure 2.1. Map of the study area with the location of the acoustic receivers shown in the top right panel. Triangles show the initial location of the receivers, squares symbolize receivers that were lost after 17/11/2010 and stars the receivers re-deployed on 9/5/2011. The asterisks mark the locations where cuttlefish were captured and released.

Data analysis

A VUE database (Vemco, Canada) was used to manage acoustic detections. Single detections occurring in periods of more than 24h were considered spurious and were removed from further analysis (Afonso et al. 2009). Acoustic telemetry data was first analysed using Eonfusion software (Myriax, Australia) to visualize movement patterns of tagged cuttlefish fish inside the monitored area.

To evaluate site fidelity two indexes were estimated: a residency index (I_R; Abecasis & Erzini 2008) and a detection efficiency index (I_D). I_R was estimated by dividing the total number of days with detections (Dd) by the number of days between first and last detections (Di). I_D corresponds to the value of Dd divided by the estimated battery life (B), that is, the estimated period for which the transmitter would be heard of if it stayed within receiver range.

Activity patterns throughout the day were evaluated by binning detections into assigned periods (day or night), according to the sunrise and sunset time obtained from the Astronomical Applications Department of U.S. Naval Observatory (http://aa.usno.navy.mil). Chi-square analyses were performed for each fish to test if there was a period of the day with a number of detections significantly different from the expected (Abecasis & Erzini 2008).

To assess home range areas, minimum convex polygons (MCP) were calculated for each cuttlefish based on the location of the receivers. The MCP is the minimum area that encompasses all detections, and is an indicator of the dispersion throughout the monitored area (Kernohan et al. 2001). In this study, given the limited habitat coverage of the array, estimates of MCP were only considered as proxies of the actual home range areas.

Centre of activity (COA) positions were estimated for each cuttlefish for 30 minute periods using the method described by Simpfendorfer et al. (2002). This method uses presence data from multiple receivers and converts them into position estimates, based on the weighed means of the number of detections at each receiver during a particular time period. These COA positions were then used to estimate the total distance (TD) travelled and the direction of the movements. Mean travelled distance per day (MTDD) was obtained by dividing TD by Dd. To determine the influence of the tidal currents in the direction of the movements of cuttlefish we compared the direction of successive COA positions against the direction of the tidal current during that period.

Results

Five of the seven acoustically tagged cuttlefish were successfully detected by the receiver array up to 37 days after tagging (Table 2.1). Additionally, one of the undetected cuttlefish (# 1) was recaptured by a recreational fisherman 76 days after tagging, approximately 15.5 km away from the release location inside the Sado estuary. This cuttlefish was in good condition and its transmitter was well attached.

Table 2.1. Summary data of the cuttlefish tagged with acoustic transmitters. ML is the mantle length in cm.

Specimen ID	ML (cm)	tagging date	last detection	tag type	# detections
Cuttlefish 1	15	27-10-2010	#	V7 30-90	0
Cuttlefish 2	23	27-10-2010	09-11-2010	V9 15-45	6199
Cuttlefish 3	17	28-10-2010	-	V7 30-90	0
Cuttlefish 4	15	28-10-2010	30-10-2010	V7 30-90	4
Cuttlefish 5	14	28-10-2010	04-11-2010	V7 30-90	9
Cuttlefish 6	15.5	28-10-2010	05-12-2010	V9 15-45	13617
Cuttlefish 7	19	28-10-2010	11-11-2010	V9 15-45	16828

Residency values ranged between 0.33 and 1 (I_R median=0.95), yet the detection efficiency was substantially lower, suggesting that the tagged cuttlefish spent the majority of the transmitter lifetime outside receiver range. Moreover, this also suggests that abandonment of the monitored area occurred after a few days of detection, which is evidence of low site fidelity (Table 2.2).

Table 2.2. Results of the residency index (I_R), detection index (I_D), minimum convex polygon (MCP), minimum travelled distance per day (MTDD) and chi-square analysis (χ^2) for differences between night and day detections for each tagged cuttlefish.

Specimen ID	I_R	I_D	MCP (km^2)	MTDD ($km.day^{-1}$)	χ^2	p	Favoured period
Cuttlefish 1	-	-	-	-	-	-	-
Cuttlefish 2	1.00	0.09	0.56	2.34	1.86	0.17	no difference
Cuttlefish 3	-	-	-	-	-	-	-
Cuttlefish 4	0.33	0.01	-	-	-	-	-
Cuttlefish 5	0.50	0.04	-	-	-	-	-
Cuttlefish 6	0.95	0.25	1.26	2.32	191.66	<0.001	night
Cuttlefish 7	1.00	0.10	0.26	1.86	185.71	<0.001	night

MCP areas could only be estimated for three individuals (#2, #6 and #7), as the remaining individuals (#4 and #5) were only detected by one receiver. Short-term 'home range' values ranged between 0.26 and 1.26 km^2 (Table 2.2). Cuttlefish travelled a minimum mean distance of approximately 2 km per day (Table 2.2). Approximately half of these movements were contrary to the prevailing tidal current direction (48.6% for #2, 50.5% for #6 and 52.1% for #7). The chi-square analysis revealed that two analyzed cuttlefish presented a significantly higher number of detections during the night, whereas the third cuttlefish showed no significant difference between the expected and the observed number of day and night-time detections (Table 2.2). The analysis of the total number of detections of all cuttlefish throughout the 24h of the day reveals an activity pattern with slightly higher number of detections during the night and a relatively lower but more stable number of detections during daytime (Figure 2.2).

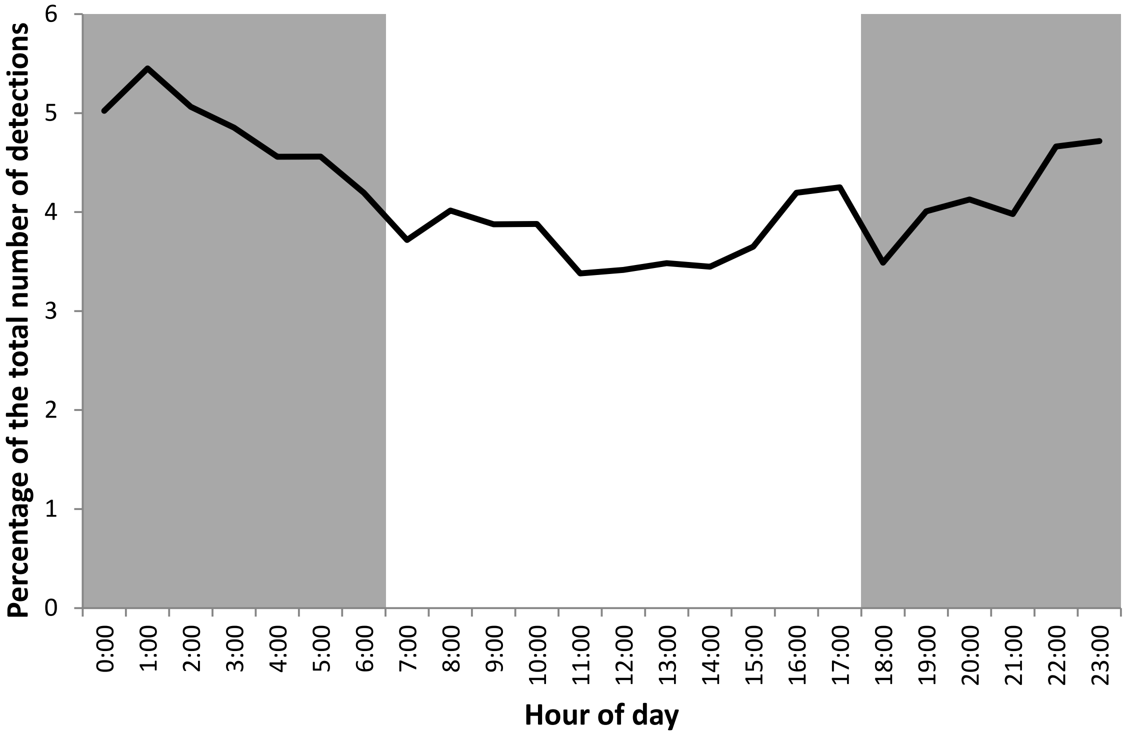

Figure 2.2. Percentage of the total number of detections of tagged cuttlefish by the 24 hours of the day. The dark area represents the night-time and the white area stands for daytime.

Discussion

Acoustic tracking of cuttlefish was first applied in the giant cuttlefish *Sepia apama* by Jackson et al. (2005). These authors found that placing the acoustic transmitter above the cuttlebone was more efficient than placing it inside the mantle. This study used a similar method which proved to be adequate, as the recaptured cuttlefish demonstrated.

Even though the acoustic receiver array spanned a distance of 5.5 km, the average number of detection days was very low. Moreover, no cuttlefish was ever re-detected after leaving the monitored area. The short term individual home range areas obtained in this study ranged between 0.26 and 1.26 km^2 and, although they are 3.5 to 16.5

times smaller than the total protection area, these results are based on a small time period between 14 and 37 days and with limited coverage by the receivers.

The loss of several receivers during the experiment does not seem to affect the results of this study, seeing that most of the tagged cuttlefish detections had stopped before that. The only exception was cuttlefish #6, whose detections still occurred after the loss of some of the receivers.

The recapture of one cuttlefish 76 days after release inside the nearby estuary (15.5 km away) provides direct evidence of migrations between the coastal areas and the estuary, as previously suggested (Batista et al. 2009, Neves et al. 2009). In fact, cuttlefish seems to be a very vagile species with no site fidelity, given that all the individuals permanently abandoned the monitored area after just a few days. This suggests that relatively small MPAs such as the LSMP probably do not provide enough protection to this cuttlefish species.

The direction of cuttlefish movements, which were up to a minimum of 2km per day, appears to be independent of the tide direction. As for the activity pattern, a higher number of detections during the night have also been reported for *Sepia apama* (Payne et al. 2010) suggesting that enhanced night-time activity may be common amongst cuttlefish species. These results show that cuttlefish have low site fidelity inside the reserve, and movements substantially larger than the size of the no-take area and even the marine Park.

In order to offer adequate protection, MPAs should include all or, at least, the large majority of the individuals' long term home range (Kramer & Chapman 1999, Kellner et al. 2008, Moffitt et al. 2009, Afonso et al. 2011), thus ensuring the protection of these individuals from fishing for a considerable amount of their life time. However, protecting mobile species throughout their lifetime has proved challenging, with several authors acknowledging that, for such species, MPAs do not offer effective protection (e.g. Kramer & Chapman 1999, Nowlis & Roberts 1999, Gerber et al. 2003, Afonso et al. 2009), especially considering the small size of most coastal MPAs. This seems to be the case of cuttlefish, which are known to perform seasonal migrations between shallow and deeper waters (Wang et al. 2003, Guerra 2006).

Modest size MPAs, such as the majority of NE Atlantic and Mediterranean MPAs (Claudet et al. 2010), are likely to fail in effectively protecting cuttlefish populations throughout their life cycle. Yet, given that this species lays eggs in the substrate, there is a strong possibility that MPAs may play an important role in protecting the reproductive output of cuttlefish populations, as long as they encompass (most of) the spawning grounds of cuttlefish.

We recommend that long term monitoring studies be carried out to better understand the effects of the LSMP on cuttlefish populations, along with wider scale studies on the spatial ecology and migration. A wider array of acoustic receivers covering deeper waters and the nearby Sado estuary could help clarify the migrations of cuttlefish across its natural habitats and, in particular, the location of their spawning grounds.

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Chapter 3: Home range, site fidelity and habitat use of *Solea senegalensis* in the Luiz Saldanha Marine Park

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Chapter 3

Home range, site fidelity and habitat use of *Solea senegalensis* in the Luiz Saldanha Marine Park

Abstract

Information on the site fidelity and home range areas of adult Senegalese sole, one of the most important resources for the artisanal fishery operating in the Portuguese coast, is inexistent. We conducted long term (up to 293 days) passive acoustic telemetry studies in order to determine the site fidelity and movement patterns of this species in a recently established marine protected area (Luiz Saldanha Marine Park, Portugal). The results revealed that most Senegalese sole spent a large part of the time, between first and last detections (average residency index = 69%), inside a relatively small area (average 95% KUD = 1.2 km^2; LSMP no-take area = 4.3 km^2), during which they clearly preferred sandy bottoms, the most common habitat inside the MPA. Results also demonstrated that Senegalese sole do regular excursions beyond MPA boundaries, eventually disappearing from the MPA. The results of this study suggest that small coastal MPAs providing adequate habitat may protect individuals of this species, while allowing for moderate levels of adult spillover from the MPA to neighbouring areas.

Introduction

Marine protected areas (MPAs) have been widely used as a fishery and biodiversity management tool (Russ & Alcala 2011). MPAs where extractive human activities are reduced or excluded can provide a refuge for overexploited populations. In the long term, they can also act as sources of emigrant larvae or post-recruit spillover that will replenish adjacent (fished) areas (Russ 2002). By continuously protecting individual fish within its boundaries, MPAs are expected to hold larger and older fish, which will reach higher reproductive potential and eventually produce offspring with better survival rates than those from younger fish, thereby benefiting local populations through enhanced recruitment (Birkeland & Dayton 2005).

Spillover can take place when the movements of individual fish across the boundaries of an MPA result in a net emigration of fish to the outside areas, eventually increasing fisheries yields. This net spillover seems to be density-dependent, as several empirical studies demonstrate that it typically occurs in MPAs that hold higher densities than their neighbouring fished areas (e.g. Goñi et al. 2008, La Mesa et al. 2011, Russ & Alcala 2011).

It follows that understanding the habitat use and movement patterns of individual fish is central to establishing the appropriate size, shape, location and separation of MPAs, so that they effectively promote the reserve effect and spillover. Moreover, this knowledge can also provide critical guidelines and input for *a posteriori* adaptive management of MPAs (Gerber et al. 2005, Grafton & Kompas 2005, Pomeroy et al. 2005), especially when there was little or no relevant ecological data available in the first place.

This study aims to shed light on the spatial-temporal movement patterns of a coastal fish species, the Senegalese sole *Solea senegalensis* Kaup 1858, within the context of a recently established MPA, the Luiz Saldanha Marine Park (LSMP). The Senegalese sole is a coastal fish species that inhabits sandy and muddy bottoms at depths down to 100m. Its geographic distribution comprises the NE Atlantic from the Gulf of Biscay to the coasts of Senegal, yet it appears to be expanding north (Desaunay et al. (2006). Knowledge of the ecology and biology of the Senegalese sole is relatively scarce, with most studies focusing on its juvenile phase (e.g. Cabral 2000a, Cabral 2000b, Vinagre et al. 2006, Vinagre et al. 2008). In particular, there are no published studies describing the dispersal or movement patterns of this species.

The Senegalese sole is one of the most important species for the artisanal fisheries that operate along the Portuguese coast, including those of the Setúbal peninsula, where the Luiz Saldanha Marine Park (LSMP) was recently implemented (Batista et al. 2009). The LSMP is located next to the Sado estuary, which is thought to be an important nursery for many coastal fishes including Senegalese sole (Cabral 2000a, Vasconcelos et al. 2008). Indeed, studies using otolith microchemistry showed that many Senegalese soles caught in coastal waters originated from the Sado nursery (Vasconcelos et al. 2011, Tanner et al. 2013).

While these evidences suggest spatial migrations between the estuary and adjacent coastal areas, these are still poorly understood. This study uses passive acoustic telemetry to evaluate the size of individual home range areas, site fidelity and patterns of habitat use of adult Senegalese sole, inside the LSMP. This information will shed light on the suitability of the LSMP design for protecting local populations of Senegalese sole.

Materials and Methods

Study area

The LSMP, located in the Setúbal Peninsula (Portugal), covers an area of approximately 53 km^2 and stretches over 38 km of coastline (Figure 3.1). It includes a narrow stretch of rocky reef habitats down to 15 m deep and a larger stretch of soft substrates (sand and mud) habitats down to 100 m. The LSMP is located in an area of high importance to commercial fishing activities, as it encompasses the fishing port of Sesimbra and is close to the nearby fishing port of Setúbal, located upstream in the Sado river estuary (Figure 3.1).

This MPA was designated in 1998, yet its final planning design was only finalized in 2005 and full implementation of regulations was only accomplished in 2009. The LSMP design includes one no-take zone of 4.3 km^2 where all extractive activities are forbidden, four partial protection zones totalling 21 km^2 where only commercial fishing using octopus traps and jigs is allowed), and three complementary protection zones totalling 28 km^2 where recreational angling and commercial fishing boats less than 7m long are allowed to operate using traditional fishing gear (Figure 3.1a). Additional regulations include the ban on spearfishing in the entire LSMP area and a tight control over fishing access, as all commercial fishing vessels require a permit to operate within the LSMP limits.

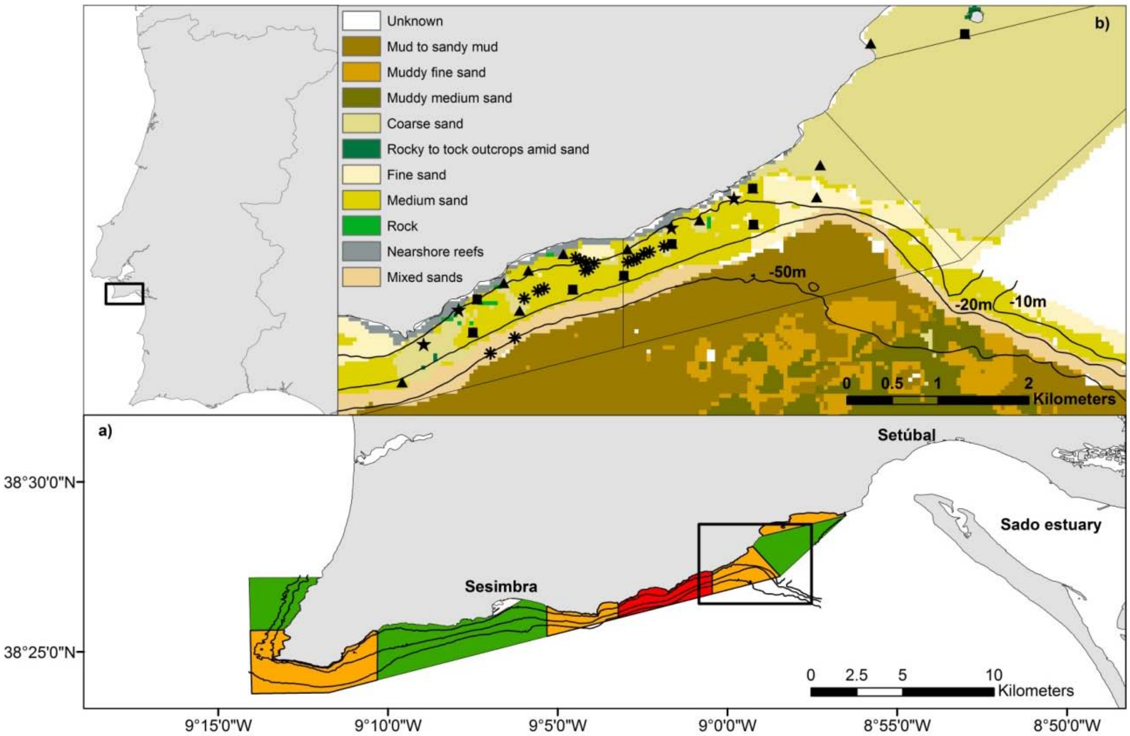

Figure 3.1. Study area with the location of the acoustic receivers. Triangles show the initial location of the receivers, squares symbolize receivers that were lost after 17/11/2010 and stars stand for new receivers deployed on 9/5/2011. The asterisks identify Senegalese sole capture and release locations. The red area corresponds to the no-take area, the orange area represents the partial protection area and the green area shows the complementary protection area.

Tagging and monitoring

Seventeen Senegalese sole over 30cm TL were captured, measured and tagged during the experimental trammel net fishing campaign of October 2010 (see Chapter 5 for more details). Eleven were captured in the no-take area with the remaining six captured in the partially protected area located east of the no-take area (Figure 3.1b). According to Teixeira & Cabral (2010) all tagged fish should correspond to mature individuals between 3 and 8 years old.

Acoustic transmitters (Vemco V9-2L) were externally attached on the eyed side underneath the dorsal fin (median region). The acoustic transmitters were previously glued, together with two nylon sutures 1 cm apart, to a smooth rubber plate. The sutures were then passed through the muscle underneath the pterigiophores of the dorsal fin and another rubber plate was placed on the opposite side (blind side) where threads were knotted (Figure 3.2). This tagging procedure was similar to that used by Bégout Anras et al. (2003) for *Solea solea*. After the attachment of the acoustic transmitter, fish were released at the site of capture.

Figure 3.2. Pictures of a Senegalese sole with an acoustic transmitter attached.

Eighteen acoustic receivers (Vemco VR2 and VR2W) monitored the presence of tagged Senegalese sole. These receivers were moored in an array that comprised two lines of receivers parallel to the coastline that covered the no-take area and one neighbouring partial protection zone (Figure 3.1b). These lines, which had nine receivers each, were deployed so as to cover depths from 8 and 14 m (inner line), and 17 to 21 m (outer line). The monitoring period lasted from October 2010 until January 2012, outlasting the expected lifetime of the acoustic tags (282 days). Six receivers from the outer line and two receivers from the inner line were lost following the first data download that took place on November 17 2010, most probably due to illegal fishing activity. The loss of these receivers was noticed during the second data download that was carried out in April 2011, leading to the deployment of four new receivers in slightly different locations in May 2011 (Figure 3.1b). Tag performance tests were carried out before the release of tagged animals. These tests showed detection ranges of approximately 300 m and no significant differences between the number of observed and expected detections during day and night (χ^2= 1.47 p=0.22).

Data analysis

We used a VUE database (Vemco, Canada) to manage acoustic detections. Single detections occurring in periods of more than 24h were considered spurious and were removed from further analysis (Afonso et al. 2009). Acoustic telemetry data was first analysed using Eonfusion software (Myriax, Australia) to visualize movement patterns of tagged Senegalese soles inside the monitored area.

Home range areas were estimated using two different methods: minimum convex polygons (MCP) and kernel utilization distributions (KUD). MCPs were estimated as the minimum area that encompassed all detections and represent a measure of dispersion over the monitored area used by an animal (Kernohan et al. 2001). The KUDs were estimated based on centre of activity positions (COA). These COA were estimated for each fish for 30 minute periods using the method described by Simpfendorfer et al. (2002). This method uses presence data from multiple receivers and converts them into position estimates based on weighed means of the number of detections at each receiver during a particular time period. A 50% KUD area was used as measure of the individual's core activity area, and a 95% KUD area as the individual's home range area (Afonso et al. 2008). Both MCPs and KUDs were estimated using Hawth's Analysis Tools extension for ArcGIS (Beyer 2004). A smoothing factor of 250 and 25m grid cells were used for KUD estimation. Correlations between fish total length and home ranges areas (MCP, 50% KUD and 95% KUD) were assessed using the Pearson correlation coefficient (Pearson's *r*).

To measure site fidelity, two indices were estimated for each individual: a residence index (I_R), corresponding to the total number of days a fish was detected (Dd) divided by the number of days between the date of release and the last detection (Di) (Afonso et al. 2008), and a weighted residence index (I_{WR}) (Lino 2012), which accounts for the number of days the fish was detected (Dd) as a proportion of the total number of monitoring days (Dt), and is weighted by the interval in days between first and last detection (Di) as a proportion of the total number of monitoring days (Dt),

$$I_{wr} = \frac{D_d}{D_t} \times \frac{D_i}{D_t}$$

This index was used to account for the long monitoring periods when compared with the expected tag lifetime. Therefore, the total number of monitoring days (Dt) was replaced by the tag expected lifetime. Both indices vary between 0 (no residency) and 1 (full time resident).

Whenever there were more than 22 days with detections and more than 2500 detections in total Fast Fourier Transformations (FFT) were applied to the hourly number of fish detections in all receivers to detect diel activity patterns (Abecasis et al. 2013). The FFT decomposes a sequence of values into components of different frequencies. The frequencies of dominant cyclical patterns are then identifiable as peaks within a frequency power spectrum (Chatfield 2004). Activity patterns throughout the day were evaluated by binning detections into assigned periods (day or night), according to the time of sunrise and sunset time as identified by the Astronomical Applications Department of U.S. Naval Observatory (http://aa.usno.navy.mil). A chi-square analysis was performed for each fish to test if there is a period of the day with a number of detections significantly different from the expected (Abecasis & Erzini 2008).

To test for habitat selection, each COA was assigned as pertaining to one of the five available habitat types (fine sand, medium sand, coarse sand, rock and nearshore reefs). We then used chi-squares tests to assess if fish used habitats differently than expected based on habitat availability (Rogers & White 2007). First, we calculated the S statistic

$$S = \sum_{j=1}^{J} \sum_{i=1}^{I} \frac{\left(u_{ij} - p_i u_{+j}\right)^2}{p_i u_{+j}}$$

where p_i is the proportion of available habitat i and u_{ij}, for $1 \le j \le J$ and $1 \le i \le I$, the number of detections for animal *j* in habitat *i*. Under the hypothesis of random habitat use by all animals this test follows the chi-square distribution with *J*(*I*-1) degrees of freedom. Additionally, the hypothesis of equal habitat selection for all individuals was tested by a chi-square test on the contingency table containing the u_{ij} values, such as

$$S_c = \sum_{j=1}^{J} \sum_{i=1}^{I} \frac{\left(u_{ij} - u_{i+} u_{+j}\right)^2}{u_{i+} u_{+j}}$$

The result was compared with a chi-square distribution with (*I*-1) (*J*-1) degrees of freedom. Moreover, we estimated selection ratios ($\omega_{ij} = \frac{u_{ij}}{u_{+j} p_i}$) to measure individual habitat selection (Rogers & White 2007). The selection ratio (ω) is a measure that refers to the selection of the relocations within the study area (Manly et al. 2002). Values above 1 indicate preference, values below 1 indicate avoidance. An eigenanalysis of selection ratios was performed to understand the individual heterogeneity of habitat selection (Calenge & Dufour 2006). All habitat selection analyses were performed using the adehabitatHS package for R (Calenge 2006).

Results

The number of days in which individual tagged Senegalese sole were detected ranged from 5 to 213 days, corresponding to total periods of detection ranging from 5 to 293 days (Figure 3.3), and to a total number of detections per fish that range from 282 to 168133 (Table 3.1).

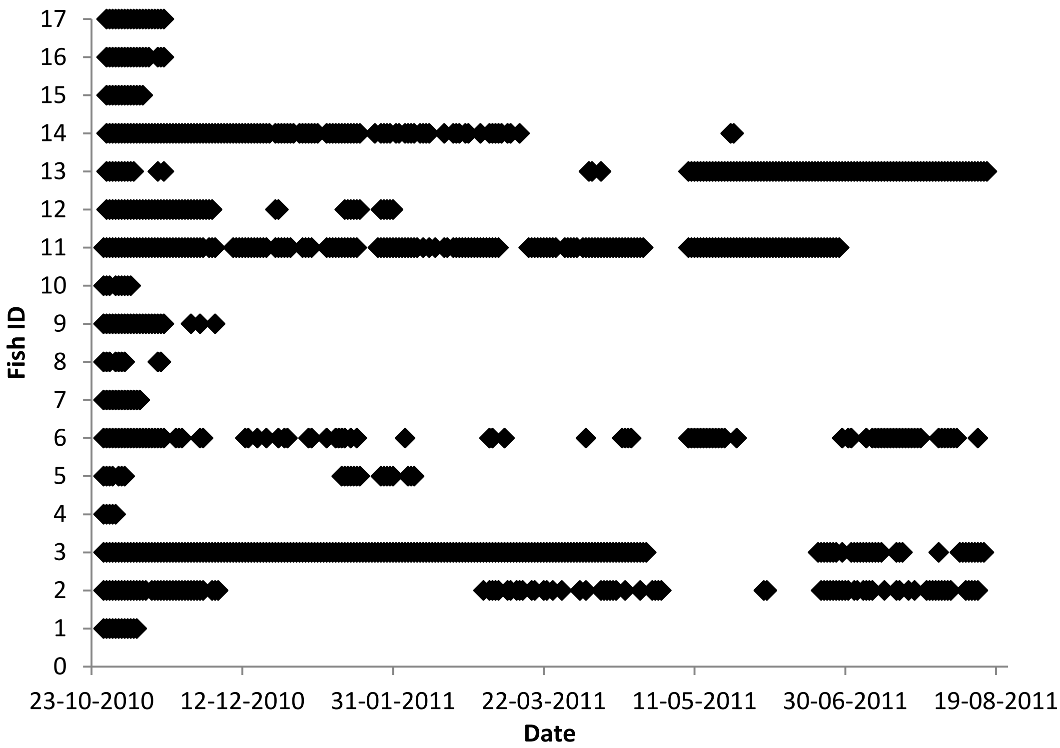

Figure 3.3. Calendar plot of daily detections of Senegalese sole tagged with acoustic transmitters.

About 70% of tagged Senegalese sole presented a I_R above 0.5, which suggests that these fish spent more than half of the detection period within the monitored area (Table 3.1). Conversely, only 2 (<12%) tagged fish had I_{WR} values above 0.5, indicating that very few individuals were detected for over half the days of expected tag lifetime (Table 3.1). Such low I_{WR} also suggests that various fish had long detection periods but only some days with detections in between. It is also noteworthy that, 52% of individuals were no longer detected after just a few days of consecutive detections (Figure 3.3). A significant correlation was found between fish total length and I_R (r=-0.501, p=0.04), indicating that larger fish spent less time in the monitored area.

Table 3.1. Summary data for tagged Senegalese sole. TL– total length; Dd – days with detections; Di – days between 1st and last detection; IR – residency index; IWR – weighted residency index; MCP – minimum convex polygon in km2; 50 % kernel utilization distribution in km^2 corresponding to home range area.[1]

ID	TL (cm)	day tagged	Dd	Di	I_R	IWR	Detections	MCP (km^2)	KUD 50% (km^2)	KUD 95% (km^2)
1	38.8	27-10-2010	12	12	1.00	0.00	6640	0.58	0.39	1.80
2	40.6	27-10-2010	103	291	0.35	0.38	15501	1.24	0.20	1.53
3	30.2	27-10-2010	213	293	0.73	0.78	168133	0.32	0.14	0.65
4	32.5	27-10-2010	5	5	1.00	0.00	2592	1.09	0.28	1.52
5	39.8	27-10-2010	22	104	0.21	0.03	3092	1.45	0.55	2.22
6	41.1	27-10-2010	94	291	0.32	0.34	11462	0.39	0.30	0.96
7	34.3	27-10-2010	13	13	1.00	0.00	3594	1.55	0.23	1.40
8	36.6	27-10-2010	9	20	0.45	0.00	345	0.50	0.25	1.40
9	32	27-10-2010	24	38	0.63	0.01	4544	0.96	0.28	1.29
10	30.6	27-10-2010	9	10	0.90	0.00	282	0.09	0.18	0.80
11	31.4	27-10-2010	192	245	0.78	0.59	38854	0.77	0.33	1.28

[1] Note: No MCP was estimated for fish #17 because it was only detected by 2 receivers. Fish #11 was recaptured by a professional fisherman using trammel nets on 28/08/2011 about 12km E of the capture position. Fish #13 was recaptured on a posterior fishing campaign (19/10/2011) in approximately the same location where it was previously caught.

12	38.1	28-10-2010	49	96	0.51	0.06	8474	0.29	0.15	0.64
13	36.6	28-10-2010	115	293	0.39	0.42	36631	0.24	0.14	0.54
14	40	28-10-2010	113	209	0.54	0.30	45765	0.33	0.15	0.55
15	38	28-10-2010	13	13	1.00	0.00	4386	0.57	0.28	0.94
16	40	28-10-2010	18	20	0.90	0.00	5410	1.10	0.43	2.01
17	37.3	28-10-2010	20	20	1.00	0.01	10764	-	0.15	0.63

Overall, a higher number of detections was found during daytime (Figure 3.4). This pattern of detections was seen in all Senegalese soles, except for individuals #6 and #15, for which there were no significant differences between day and night-time detections (Table 3.2). Eight Senegalese soles presented enough detections to run the FFT analysis. Of these individuals, four presented a cyclical pattern of activity with a strong 24h peak (Table 3.2), whereas no distinct peaks were identified for the remaining four.

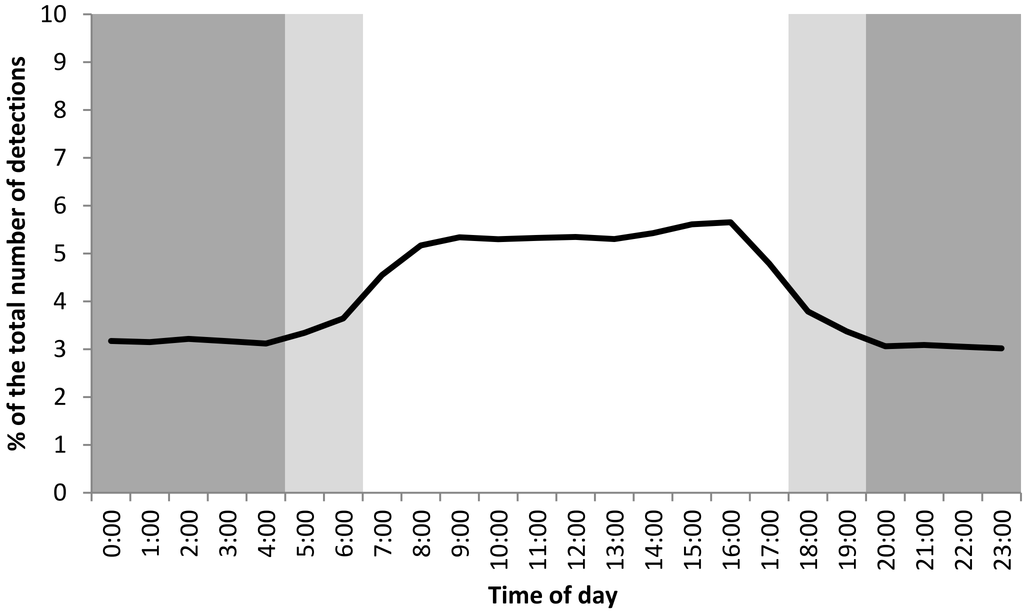

Figure 3.4. Percentage of the total number of detections of tagged Senegalese sole by the 24 hours of the day. Dark areas symbolize night-time and light grey areas stand for dawn and dusk hours.

Home range areas spanned between 0.09 and 1.55 km^2 for MCP; between 0.14 and 0.55 km^2 for 50% KUD and between 0.54 and 2.22 km^2 for 95% KUD (Table 3.1). Home range (95% KUD) and core activity (50% KUD) areas were stable throughout the study period for nine fish whose detections spanned for over 30 days (Figure 3.5 and Figure 3.6). No significant correlations were observed between fish length and MCP (r=0.124, p>0.05), 50% KUD (r=0.273, p>0.05) or 95% KUD (r=0.223, p>0.05).

Table 3.2. Results of the Fast Fourier Transformation (FFT) and chi-square ($\chi2$) tests for tagged Senegalese sole. Bold values in FFT column indicates highest peak.

ID	FFT peaks	$\chi2$	p-value	period of day
1	-	429.07	<0.001	day
2	no clear peaks	288.36	<0.001	day
3	**24**;48;72	5353.70	<0.001	day
4	-	98.85	<0.001	day
5	-	206.70	<0.001	day
6	no peaks	1.82	0.18	no difference
7	-	379.11	<0.001	day
8	-	57.88	<0.001	day
9	12;**24**;72	1282.78	<0.001	day
10	-	84.90	<0.001	day
11	24	16010.86	<0.001	day
12	no peaks	1393.96	<0.001	day
13	no peaks	2847.21	<0.001	day
14	24	4501.71	<0.001	day
15	-	0.74	0.39	no difference
16	-	350.73	<0.001	day
17	-	311.68	<0.001	day

The chi-square test for habitat selection was highly significant (S=43810.78, df=68, P<0.001). Therefore, the null hypothesis was rejected, implying that fish were not distributed proportionately to the habitat type available. Furthermore, this habitat selection pattern was not identical for all individuals (Sc= 29287.92, df=64, p<0.001; Figure 3.7). The eigenanalysis of selection ratios produced two factors that mostly explained the S statistic (98.3%). The nearshore reefs and rock habitats were hardly ever used by any fish, while medium sand habitat was favoured by nearly all fish (Figure 3.7). The habitat selection ratios (ω) confirm these results by demonstrating a general, slight preference for medium sand habitats, with all individuals avoiding the rock and nearshore reefs (Table 3.3).

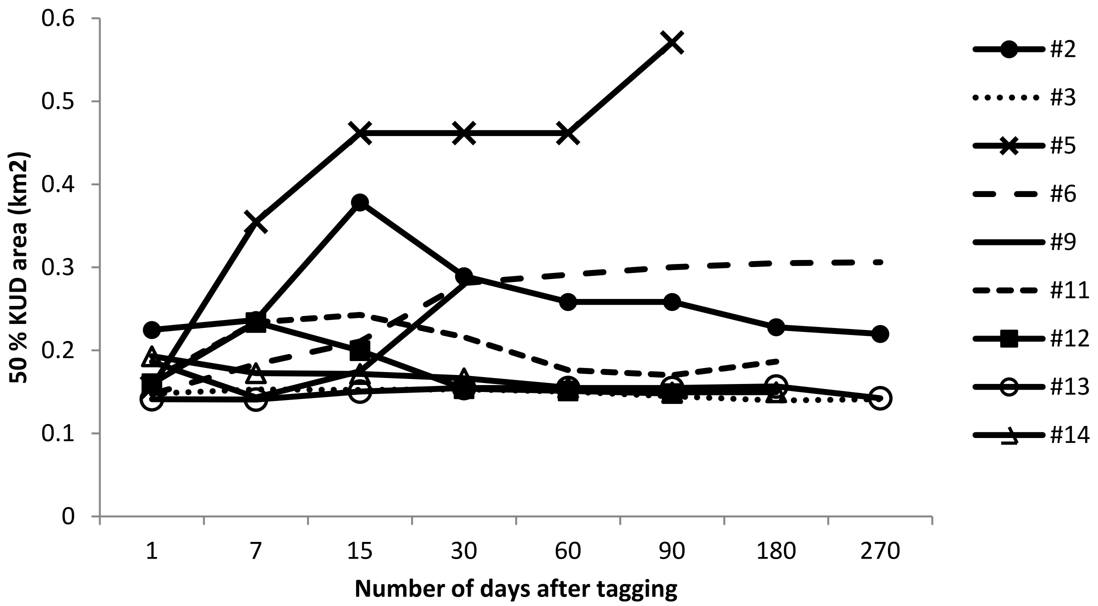

Figure 3.5. Core area (50%) kernel utilization distribution areas for nine Senegalese sole whose detections spanned for over 30 days between first and last detection.

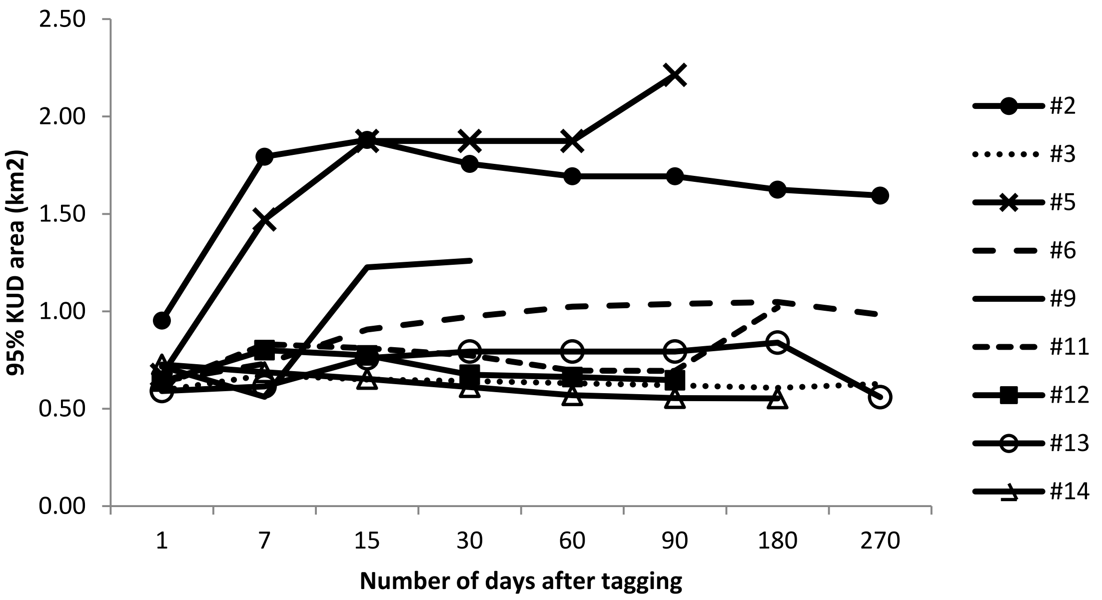

Figure 3.6. Home range 95% kernel utilization distribution areas for nine Senegalese sole whose detections spanned for over 30 days between first and last detection.

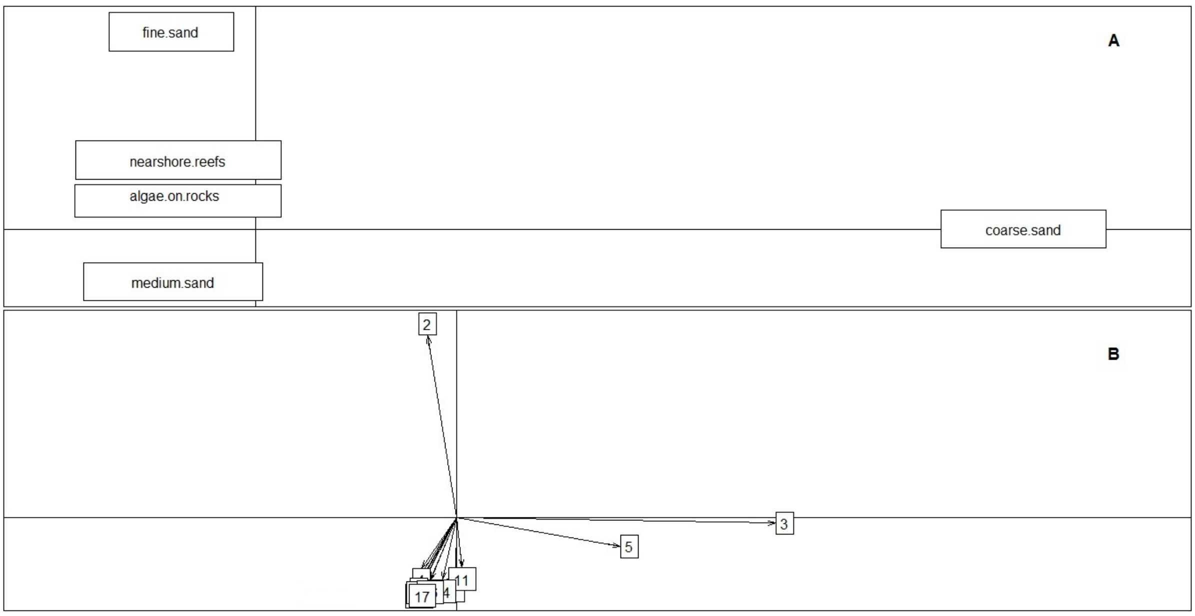

Figure 3.7. Results of the eigenanalysis of selection ratios of habitat selection by 17 Senegalese sole on five different habitats. A) Habitat type loadings on the first two factorial axes. B) Animal scores on the first factorial plane.

Table 3.3. Manly's selection ratio values for each Senegalese sole tagged with an acoustic transmitter. Note: Values in parentheses indicate the number of detections in each habitat; the availability of the five habitat types in the monitored area is displayed at the bottom of the table.

ID	Coarse sand	Fine sand	Medium sand	Rock	Nearshore reefs
1	0.96 (32)	0 (0)	1.35 (312)	0 (0)	0 (0)
2	0.24 (34)	5.11 (1135)	0.3 (294)	0 (0)	0 (0)
3	9.43 (7450)	0 (0)	0.13 (711)	0 (0)	0 (0)
4	0.06 (1)	0.34 (9)	1.4 (164)	0 (0)	0 (0)
5	5.34 (180)	0.08 (4)	0.7 (164)	0 (0)	0 (0)
6	0.07 (8)	0 (0)	1.47 (1154)	0 (0)	0.03 (2)
7	0.3 (10)	0.06 (3)	1.43 (334)	0 (0)	0.04 (1)
8	0 (0)	0.15 (2)	1.45 (85)	0 (0)	0 (0)
9	0.03 (2)	0.06 (5)	1.46 (583)	0 (0)	0.08 (3)
10	0 (0)	0 (0)	1.49 (95)	0 (0)	0 (0)
11	1.13 (428)	0.18 (104)	1.27 (3326)	0 (0)	0.13 (33)
12	0 (0)	0 (0)	1.47 (1049)	0 (0)	0.12 (8)
13	0 (1)	0 (0)	1.48 (3877)	0 (0)	0 (0)
14	0.61 (168)	0 (2)	1.39 (2649)	0.34 (13)	0 (0)
15	0.02 (1)	0.03 (2)	1.45 (420)	0 (0)	0.22 (6)
16	0.3 (15)	0.05 (4)	1.43 (499)	0 (0)	0 (0)
17	0.09 (7)	0 (0)	1.47 (821)	0 (0)	0 (0)
Availability (%)	9.69	15.17	67.33	1.37	6.43

Discussion

The FFT analysis revealed the strong diel activity pattern of the Senegalese sole, a common characteristic of most marine species that can be related with endogenous rhythms, abiotic factors such as daylight intensity, height of tides, moon phase and temperature, or even biotic factors such as predator and prey abundance (Cole 1957, Andrews et al. 2009). In contrast, the closest flatfish species present in the study area – the common sole *Solea solea* (Cabral et al. 2007), has been shown to be more active during night-time. This different behavior may assure that interspecific competition is avoided or might be related with fish size, since the common sole individuals tagged in previous studies were smaller than the Senegalese sole individuals tagged in this study. The lower number of night-time detections may be caused by fish burrowing in the sand. This would reduce the transmitters' range, in a similar way to what was observed in other studies (Abecasis et al. 2013).

Our results seem to indicate that some individuals exhibit high site fidelity while the majority is more transient. The low I_R values for individuals whose detection period spanned over 100 days suggest that movements out of the monitored area occur on a frequent basis and sometimes for prolonged periods. As these periods occurred in-between detections, these individuals abandoned the monitored area and later returned to it. In contrast, individuals that were detected for fewer days spent the majority of the time inside the monitored area. Thus, low I_{WR} values can be divided in two main groups: 1) fish that were detected over short periods of time and never detected again (e.g. # 4, # 10 and # 15) and 2) fish with a long period of time between first and last detections but only some days with detections in between (e.g. # 2, # 3 and # 13).

The first group is most likely composed of fish that performed movements leading to spillover (permanent relocation of home range area outside the MPA) to adjacent areas where they can eventually be captured. This was confirmedly what happened to fish # 11. Another possibility is that (some) fish suffered post release mortality. However, two tagged soles (#11 and #13) were recaptured almost one year after tagging in apparent good condition, suggesting that post release mortality is low. Except for some scale loss around the point of contact these fish were in good condition showing no sign of infection and the acoustic transmitters were still properly attached.

The second group is composed of fish that moved outside the monitored area but ultimately came back, an event described as leakage, where temporary movements beyond the MPA boundaries are part of the home range movements (Pérez-Ruzafa et al. 2008). Although the monitored array only covered part of the LSMP the fact that 2 individuals were captured outside this MPA strengthens this hypothesis. Another possibility is that the monitored area represents only a part of a larger home range that is seasonally visited. However, if this is the case there seems to be no common temporal trend among fish in such movements.

Overall, our results indicate that larger fish have less site fidelity to the monitored area and, therefore, should use areas outside the detection range more often. Larger fish may move over larger areas due to a combination of enhanced exploring capabilities and a size-related increase of energetic needs demanding more captured prey. Rogers (1994) has shown that the geographic range of the common sole (*Solea solea*) increases with its abundance so that individuals may search for new feeding grounds, a behavioural response known as density dependent movements (Kellner et al. 2008). Another hypothesis could be the existence of a social/sex component similar to what has been reported for other flatfish species that exhibit complex haremic territorial systems. In several Bothidae species adult males defend territories that include several smaller female territories (Konstantinou & Shen 1995, Carvalho et al. 2003).

The preference of adult Senegalese sole for medium sandy habitats found in our study contrasts with the preference of juveniles for mud sediments as reported for the nearby Sado estuary (Cabral 2000a). This change in habitat preferences is somewhat expected given the ontogenic shifts and the change in prey preference with age (Garcia-Franquesa et al. 1996, Cabral 2000b).

Conclusions

According to our results, the 'impact' area of the LSMP (no-take and partial protection areas) is large enough to provide protection to individual Senegalese sole, as it represents approximately eleven times the largest individual long-term home range. Moreover, a large part of the no-take and partial protection areas holds the preferred habitat of adult Senegalese sole. In addition, spillover and leakage events, which are one of the supposed benefits of MPAs to local fisheries, seem common (Pérez-Ruzafa et al. 2008, Russ & Alcala 2011). On the other hand, the size of the 'impact' area may not be enough to offer protection to more than a few individuals due to density-dependent movements or if this species presents a social system similar to that present in several Bothidae species. Therefore, future studies focusing on the social and reproductive behaviour of this species are needed in order to better clarify our findings.

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Chapter 4: Home range, site fidelity and activity patterns of *Diplodus sargus* in the Luiz Saldanha Marine Park

Part of this chapter will be included in the paper: Abecasis D, Horta e Costa B, Afonso P, Gonçalves E, Erzini K (In preparation) Small home ranges of a commercial fish (*Diplodus sargus*) explain early reserve effects.

Chapter 4

Home range, site fidelity and activity patterns of *Diplodus sargus* in the Luiz Saldanha Marine Park

Abstract

Passive acoustic telemetry was used to analyze the activity patterns and site fidelity of white seabream (n=20), inside a marine protected area (Luiz Saldanha Marine Park, Portugal). The number of days between first and last detections ranged between 2 and 293 with most individuals presenting high site fidelity. The results show a cyclical 24 hour pattern of activity with most individuals being more active during the day with an increase in the number of detections after sunrise and a decrease just before sunset. A positive correlation was found between fish total length and the maximum range. However, the maximum range observed was 5 km and the average was 2.36 km indicating that even large fish occupy relatively small areas. The results of this study suggest that white seabream might benefit from the implementation of small coastal MPAs such as the Luiz Saldanha Marine Park.

Introduction

Understanding the activity patterns and residency of marine species is a fundamental aspect towards their proper management. With the onset of marine protected areas (MPAs) as a tool for conservation and management of biodiversity and fisheries this information has become even more necessary. Several authors acknowledge that information on species movements, site fidelity and habitat use should be taken in consideration during the design phase of MPAs (e.g. Botsford et al. 2003, Glazer &

Delgado 2006). However, in most cases, this information is either unavailable or not taken in consideration during the design of MPAs. This information is extremely useful even after the implementation of MPAs as a tool for adaptive management (Grafton & Kompas 2005, Pomeroy et al. 2005).

During the recent implementation of the Luiz Saldanha Marine Park (LSMP), a small coastal MPA with different protection levels, located in the vicinity of two large fishing harbours (Setúbal and Sesimbra), no information on the activity patterns and residency of some of the most important species for the local small scale artisanal fisheries was available. The main goals of this MPA are to protect the local high biodiversity and also to promote the sustainability of local artisanal fisheries.

The white seabream, *Diplodus sargus* (Linnaeus, 1758), is one of the most important coastal fish resources in southern European countries and one of the main target species for recreational anglers (Veiga et al. 2010). According to Horta e Costa et al. (2013) the white seabream is one of the most landed reef fish species in the port of Sesimbra while also attaining one of the highest values per kg.

The movement and activity patterns, habitat use and site fidelity have been widely studied in adults and juveniles of white seabream using tag-recapture and acoustic transmitters (Abecasis et al. 2009, Lino et al. 2009, D'Anna et al. 2011, Abecasis et al. 2013, Koeck et al. 2013a). With the exception of juveniles that migrate from an estuarine environment to coastal areas (Abecasis et al. 2009) white seabream demonstrate high site fidelity (D'Anna et al. 2011, Abecasis et al. 2013). The size of the average home range areas of adult white seabream ranged between 0.11 km^2 in the Mediterranean and 1.88 km^2 in the South of Portugal (D'Anna et al. 2011, Abecasis et al. 2013). Different activity patterns have also been reported for this species. The study by D'Anna et al. (2011) revealed that white seabream were more active during the night while the study by Abecasis et al. (2013) showed that white seabream were more active during the day. The observed diversity in activity patterns and on the size of home range areas enforces the need for local studies particularly in MPAs (Abecasis et al. 2012).

The objectives of this study were to determine the site fidelity, activity patterns and home range areas for white seabream in the LSMP. This information is then used to discuss the usefulness of the LSMP towards the management of the local white seabream population.

Materials and methods

Study area

This study took place in the LSMP, a 53 km^2 marine reserve that stretches over 38 km of coastline. With only a narrow rocky reef from the coastline down to 15m deep the vast majority of its bottom is composed of soft substrates (Figure 4.1). This MPA comprises 3 different protection levels: one no-take zone of approximately 4.2 km^2 where all human activities except research and monitoring are banned; four partially protected areas where only octopus traps and jigs are allowed at more than 200m from the shoreline and three complementary protection areas where traditional fishing gears are allowed for vessels smaller than 7m. Recreational angling is only permitted in complementary areas and spearfishing is prohibited in the entire LSMP. The monitored area consisted of approximately 4.8 km of coastline covered by an array of 11 eleven acoustic receivers (VR2 and VR2W, Vemco) placed in a line parallel to the coast (Figure 4.1).

Tagging and monitoring

Fish were captured using hook and line baited with shrimp. Twenty white seabream were fitted with an acoustic transmitter. All transmitters had an emission rate between 30-90 seconds with exception of 4 transmitters which had a 15-45 seconds interval. Two different transmitter sizes were used, V7 and V9 (Vemco) with an expected lifetime of 95 and 282 days respectively. The transmitters with the higher rate of emission (V9; 15-45s) had an expected lifetime of 151 days. Acoustic transmitters were introduced in the coelomic cavity through a 1cm incision made in the ventral line between the insertion of the pectoral fins and the anus following the procedures described in Abecasis & Erzini (2008). After introducing the transmitter, the incision was closed using cyanoacrilate tissue adhesive (Vetseal, BBraun). Preliminary tests showed detection ranges of approximately 300 m for V9 transmitters and 100 m for V7 transmitters. Fish were released in the same location where they were captured as soon as they recovered from surgery (approximately 15 minutes). Tagged fish ranged between 20 and 37 cm in total length (TL) which correspond to fish between 3 and 13 years old (Abecasis et al. 2008).

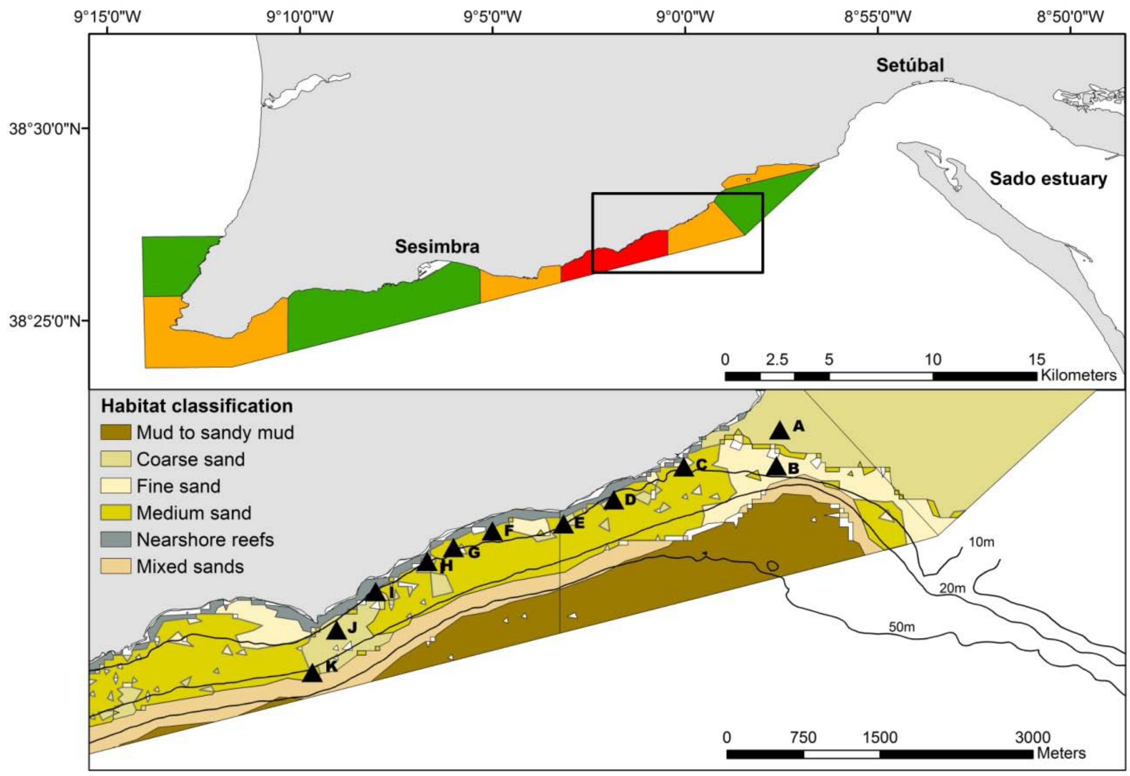

Figure 4.1. Map of the study area with the location of the acoustic receivers.

Monitoring of tagged fish was made through an array of 11 acoustic receivers located parallel to the coastline around 10m deep. Although the monitoring period lasted from 1 May 2011 until 2 July 2013, receiver H was nonoperational from 24 December 2012. On average, the receivers were cleaned from incrusting algae and fauna, and data was downloaded every 3 months.

Data analysis

Detections of tagged white seabream were managed using the VUE database (Vemco). The site fidelity was analysed by using two different indices, the residency index (I_R) proposed by Afonso et al. (2008) and the weighted residency index (I_{WR}) proposed by Lino (2012). The I_R was estimated by dividing the total number of days a fish was detected by the number of days between the date of release and the last detection. This index was estimated for each receiver and for the entire array. The I_{WR} takes in consideration the duration of the transmitter expected lifetime (or study duration) and follows the formula

$$I_{wr} = \frac{D_d}{D_t} \times \frac{D_i}{D_t}$$

where Dd is the total number of days a fish was detected, Di is the number of days between first and last detections and Dt is the expected duration of the acoustic transmitter or study duration whatever is shorter. Both these indices vary between 0 (no residency) and 1 (full time resident).

Home range areas were calculated using the kernel utilization distribution (KUD). The KUD is a two dimensional probabilistic function that estimates the area of probability of finding a fish (Worton 1989). The KUDs were estimated based on centre of activity positions (COA). These COA were estimated for each fish for 30 minute periods using the method described by Simpfendorfer et al. (2002). This method uses presence data from multiple receivers and converts them to position estimates based on weighed means of the number of detections at each receiver during a particular time period. A 50% KUD was used as the core activity area and a 95% KUD as the home range area (Abecasis et al. 2013). We used 25 x 25 m cell grids and a smoothing factor (h) of 250 to estimate KUD with Hawth's analysis tools for ArcGIS. Because acoustic receivers were displayed parallel to the coastline the maximum range length between the acoustic receivers where fish were detected was also estimated in order to better understand the dispersion along the coast. The Pearson correlation coefficient was used to examine the relationship between fish total length and 50% and 95% KUD, I_R and I_{WR}.

To investigate possible diel patterns, detections were separated by day/night time for each individual fish. Sunrise and sunset times obtained from the United States Naval Observatory (http://aa.usno.navy.mil/data/docs/RS_OneYear.php) were used to determine the boundaries of the diel intervals. A chi-square test was then used to test for differences in the number of detections between day and night time (Abecasis & Erzini 2008).

Fast Fourier Transformations (FFT) was applied to the hourly number of detections of each fish to examine diel detection patterns. Only individuals that were detected more than 22 days and had more than 2500 detections in total were analyzed. The FFT decomposes a sequence of values into components of different frequencies. The frequencies of dominant cyclical patterns are then identifiable as peaks within a frequency power spectrum (Chatfield 2004).

Results

The detection period of tagged white seabream, between first and last detection, ranged between 2 and 293 days with most fish being detected close to or even more than the expected lifetime of the transmitter (Figure 4.2). The high values of the I_R and I_{WR} observed for most individuals indicate that white seabream have strong site fidelity to the monitored area (Table 4.1). The fact that for most fish the highest I_R is observed for the receiver closest to the tagging location reinforces the high site fidelity (Table 4.2). The dispersion along the coast was also short with most fish covering a stretch of coastline around 2 km long. Only two fish (# 5 and 8) were detected in ten or more receivers which correspond to a stretch of coastline around 5 km. The 95% KUD ranged between 0.43 and 1.56 km^2 (0.77 km^2 average) whereas the 50% KUD ranged between 0.13 and 0.41 km^2 (0.18 km^2 average) (Table 4.1). The number of detections was significantly higher during the day for all fish with exception of fish #16 and 19 which were detected more frequently during the night-time (Table 4.1). The FFT analysis revealed a 24h activity pattern for all analyzed fish except #15 that showed no pattern.

Additionally, fish #5, 7, 12, 13, 16 and 18 presented additional activity peaks at 48h and fish #8 and 18 also at 72h.

The Pearson correlation revealed a positive correlation between TL and maximum range (Pearson's r = 0.48, p=0.044) and between fish TL and I_{WR} (Pearson's r = 0.45, p=0.048). No significant correlations were found between TL and 50% KUD, 95% KUD or I_R.

Discussion

Although several studies have already been published on the movement patterns, home range and site fidelity of white seabream (Abecasis et al. 2009, D'Anna et al. 2011, Abecasis et al. 2013) the differences observed between them support the need for local studies. Knowledge of local populations' habits and needs are especially relevant for the establishment of marine reserves and their adaptive management (Grafton & Kompas 2005, Kaiser 2011). Therefore, the results of this study provide not only information on the species movements but are also particularly useful for the adaptive management of the LSMP.

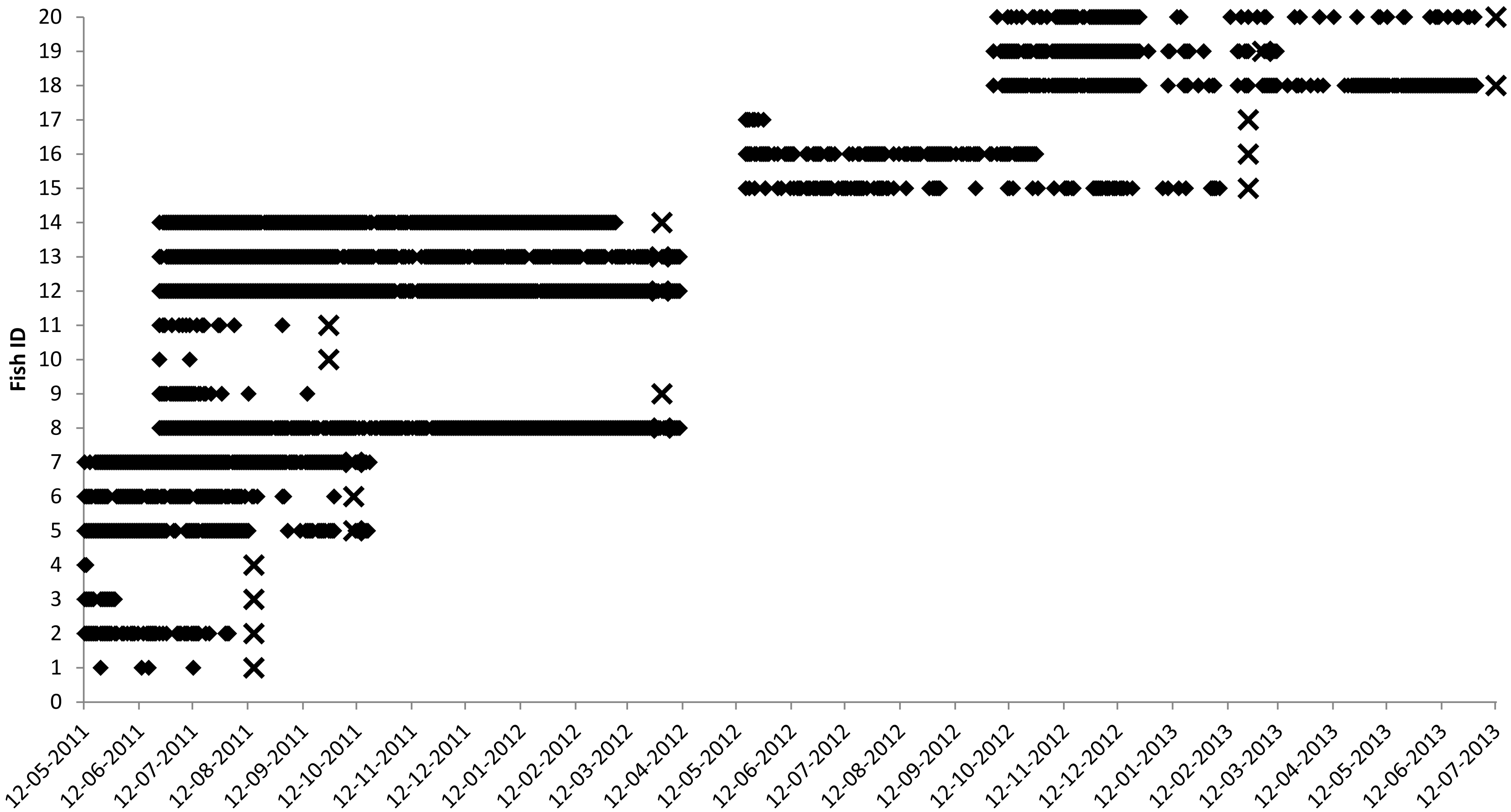

Figure 4.2. Calendar plot showing the detections of white seabream fitted with acoustic transmitters in the Luiz Saldanha Marine Park. The "x" denotes the end of the estimated lifetime of the tag.

Table 4.1. Summary data for tagged white seabream. TL – total length in cm; Dd – days with detections; Di – days between 1st and last detection; Chi-square – period of the with a significantly higher number of detections; KUD 50 % - 50 % kernel utilization distribution in km^2 corresponding to core utilization area; KUD 95% - 95 % kernel utilization distribution corresponding to home range area; I_R – residency index; I_{WR} – weighted residency index.

ID	TL	Tagging	Dd	Di	Tag lifetime	Emission rate	Chi-square	Detections	Maximum range (km)	KUD 95%	KUD 50%	I_R	I_{WR}
1	23	12-05-2011	4	62	95	30-90	-	4	-	-	-	0.06	0.03
2	21	12-05-2011	50	82	95	30-90	day	476	3.2	-	-	0.61	0.45
3	20	12-05-2011	15	18	95	30-90	day	1015	-	-	-	0.83	0.03
4	23	12-05-2011	2	2	95	30-90	-	20	2.1	-	-	1.00	0.00
5	37	12-05-2011	108	160	151	15-45	day	32600	5.0	1.28	0.21	0.68	0.76
6	29	12-05-2011	81	141	151	15-45	day	12281	1.6	0.59	0.18	0.57	0.50
7	26	12-05-2011	149	161	151	15-45	day	20832	1.6	0.43	0.13	0.93	1.05
8	23	23-06-2011	276	293	282	30-90	day	15659	4.5	0.99	0.25	0.94	1.02
9	29	23-06-2011	28	36	282	30-90	day	5315	1.6	0.50	0.14	0.78	0.01
10	22	23-06-2011	2	18	95	30-90	-	3	0.8	-	-	0.11	0.00
11	20	23-06-2011	15	70	95	30-90	-	25	1.1	-	-	0.21	0.12
12	24	23-06-2011	286	293	282	30-90	day	17634	2.1	0.45	0.13	0.98	1.05
13	30	23-06-2011	253	293	282	30-90	day	10624	2.1	0.52	0.13	0.86	0.93

14	29	23-06-2011	250	257	282	30-90	day	16719	2.7	0.46	0.13	0.97	0.81
15	24	17-05-2012	90	267	282	30-90	day	2824	1.5	0.57	0.14	0.34	0.30
16	22	17-05-2012	104	164	282	30-90	night	4302	1.6	0.50	0.13	0.63	0.21
17	24	17-05-2012	7	11	282	30-90	day	558	1.4	0.67	0.17	0.64	0.00
18	29	03-10-2012	173	272	282	30-90	day	18171	3.7	1.56	0.41	0.64	0.64
19	27	03-10-2012	96	160	151	15-45	night	14025	3.7	1.45	0.26	0.60	0.67
20	26	03-10-2012	82	271	282	30-90	day	2041	2.1	0.76	0.17	0.30	0.30

Table 4.2. Individual residency index (IR) for each tagged white seabream and each acoustic receiver. Bold values represent the receiver closest to the tagging location.

ID	TL (cm)	A	B	C	D	E	F	G	H	I	J	K
1	23									0.06		
2	21			0.02	0.01	0.01	0.01	0.04	0.26	0.44		
3	20								0.83			
4	23				0.50	0.50			0.50			
5	37	0.06	0.09	0.17	0.20	0.66	0.28	0.13	0.05	0.04	0.03	0.01
6	29			0.04	0.47	0.52						
7	26			0.02	0.93	0.48						
8	23	0.00	0.00	0.05	0.07	0.80	0.83	0.14	0.01	0.01	0.01	
9	29					0.18	0.29	0.24	0.01			
10	22						0.06	0.06				
11	20						0.01	0.17	0.03			
12	24						0.49	0.97	0.03	0.01	0.02	
13	30						0.26	0.85	0.11	0.01	0.01	
14	29			0.02	0.02	0.02	0.15	0.73	0.92			
15	24								0.28	0.80	0.03	

16	22						0.04	0.47	0.97	0.06	
17	24					0.86	0.71	0.29			
18	29			0.39	0.40	0.40	0.51	0.62	0.42	0.16	0.05
19	27			0.41	0.40	0.28	0.36	0.36	0.68	0.03	0.02
20	26						0.13	0.17	0.65	0.15	0.04

The tagging method used in this study had been previously used in other seabream acoustic telemetry studies with good results (Abecasis & Erzini 2008, Abecasis et al. 2009, 2012, Abecasis et al. 2013). Moreover, a recent study by Koeck et al. (2013b) revealed that the implantation of an acoustic transmitter in the abdominal cavity of white seabream does not influence their survival or behaviour.

The observed 24h diel pattern of activity with significantly more detections during the day is in accordance with the observations in the South of Portugal (Abecasis et al. 2013) and with the results of white seabream inhabiting natural reefs (Koeck et al. 2013a). These results are consistent with the findings of Figueiredo et al. (2005) that show a higher feeding activity during the day. On the contrary, it contrasts with the results of white seabream inhabiting artificial reefs where this species was found more active during the night (D'Anna et al. 2011, Koeck et al. 2013a). These contrasting results suggest that white seabream can easily adapt their diel behaviour pattern according to the habitat and its specifications.

The average size of the home range areas found in this study is less than half of the average size found by Abecasis et al. (2013). This could be due to habitat differences since the study of Abecasis et al. (2013) took place in a set of artificial and natural reefs surrounded by sandy bottoms whereas this study took place in a stretch of rocky coastline that extends for over 20 km providing adequate food resources and plenty of refuges. Moreover, this study took place in a marine reserve whereas in the study area of Abecasis et al. (2013) took place in an area were both recreational and commercial fishing are allowed. A recent study on the sparid *Pagrus auratus* (Parsons et al. 2010) has shown differences on home range size between individuals from the marine reserve and individuals from outside the reserve. Parsons et al. (2010) suggest that individuals with small home ranges centered within the reserve will not often cross the reserve boundaries and therefore will be less likely to be captured by fishing. This would lead to a different selection pressure favouring individuals with smaller home range areas. Given the low age of the LSMP it is unlikely that this effect is already taking place however, future studies should investigate this possibility.

When compared with the home range areas observed by D'Anna et al. (2011) our values were considerably higher. However, the monitoring period of our study was much longer (maximum of 48 vs 293 days of monitoring) and our fish were also bigger.

The fact that no significant correlations were found between fish TL and 95% or 50% KUD might indicate that adult white seabream do not use larger areas as they grow. Yet, larger individuals appear to roam over larger stretches of coastline as indicated by the positive correlation found between fish TL and the maximum range length.

Given that white seabream show high site fidelity and home range areas which are about 5.5 times smaller than the no-take area, it is feasible that the LSMP can have an important role in this species' local management. The results of the experimental fishing trials (Cunha et al. 2011) and of this study suggest that white seabream movements do not extend far away from rocky bottoms. In fact, the previous study by Abecasis et al. (2013) has also shown the white seabream preference for rocky bottoms even though movements to sandy bottoms were also observed. The exclusion of fishing nets in first 200m from the coastline in the partially protected areas confers additional protection areas for white seabream given the narrow width of the rocky reefs. Consequently, the total amount of coastline where white seabream is protected from fishermen is around 20km which is 8.5 times the average maximum range length observed for white seabream.

The results of this study support the observations of Horta e Costa et al. (2013) which suggest that white seabream might already be benefiting from the implementation of the LSMP.

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Chapter 5: Evaluation of the effects of the Luiz Saldanha Marine Park on the local abundance of *Sepia officinalis* and *Solea senegalensis* using experimental fishing

Part of this chapter was published in the paper: Abecasis D, Afonso P, O'Dor K, Erzini K (2013) Small MPAs do not protect cuttlefish. Fish Res 147:196-201

Part of this chapter was accepted for publication in the journal Fisheries Management and Ecology: Abecasis D, Afonso P, Erzini K (submitted) Can small MPAs protect a coastal flatfish (*Solea senegalensis*)?

Chapter 5

Evaluation of the effects of the Luiz Saldanha Marine Park on the local abundance of *Sepia officinalis* and *Solea senegalensis* using experimental fishing

Abstract

With fisheries management moving towards an ecosystem based approach marine protected areas (MPAs) have become a critical tool. Monitoring of MPAs is crucial to understand their effectiveness but also to their adaptive management. Experimental fishing surveys were carried in order to test the effect of the implementation of the Luiz Saldanha Marine Park (LSMP) on the abundance and biomass of Senegalese sole and cuttlefish. A total of 106 fishing sets were carried between 2007 and 2011, using 500 m of trammel nets. The results of the beyond BACI analysis failed to detect any significant difference in either abundance or biomass of both species, between protected and unprotected areas before and after the implementation of LSMP. The possible causes for the lack of a significant effect are discussed.

Introduction

Marine protected areas (MPAs) have been widely implemented as a tool to protect biodiversity and/or manage fisheries (Roberts et al. 2005, Botsford et al. 2009). In most cases, especially those concerning fisheries management, MPAs are still seen mostly as a precautionary measure against local fishery collapse, since empirical evidence about their success is still scarce (Clark 1996, Botsford et al. 1997, Charton & Ruzafa 1999, Goñi et al. 2010, Vandeperre et al. 2011).

MPAs are seen as a way of overcoming uncertainty in fisheries management based on conventional measures (e.g. effort control, size and gear restrictions). Yet, the design of MPAs itself has also rested upon substantial uncertainty, mainly because information on the species habitat use and connectivity of their (sub)populations is generally lacking. This knowledge is crucial to an appropriate MPA design if one is to maximize the benefits inside and outside the protected areas (Grüss et al. 2011). Areas where extractive human activities are reduced or banned can provide a refuge for overexploited populations. In the long term, they can also act as sources of emigrant larvae or post-recruit spillover that will replenish adjacent (fished) areas (Russ 2002). By continuously protecting individual fish within its boundaries, MPAs should hold larger and older fish, which will reach higher reproductive potential and eventually produce offspring with better survival rates than those from younger fish, thereby benefiting local populations through enhanced recruitment (Birkeland & Dayton 2005).

Spillover can take place when the movements of individual fish across the boundaries of an MPA result in a net emigration of fish to the outside areas, eventually increasing fisheries yields. Because those MPAs will typically hold higher densities than neighbouring fished areas, density dependence should promote this net spillover. This was recently demonstrated in several empirical studies (e.g. Goñi et al. 2008, La Mesa et al. 2011, Russ & Alcala 2011).

The Arrábida coastal environment (Portugal) harbours over 1200 registered marine species and is considered a hotspot of marine biodiversity in the northeastern Atlantic (Gonçalves et al. 2003, Cunha et al. 2011). In 1998 an MPA - the Luiz Saldanha Marine Park (LSMP) - was designated to protect this biodiversity and also to promote the sustainability of local artisanal fisheries and fishers' livelihoods.

The cuttlefish, *Sepia officinalis* (Linnaeus, 1758), and the Senegalese sole, *Solea senegalensis* Kaup 1858, are the main targets of the artisanal fisheries based off the ports of Setúbal and Sesimbra, located near the LSMP (Serrano 1992, Batista et al. 2009). Despite the variability in catches observed along the year, with higher catches of cuttlefish and the lowest for Senegalese sole during the autumn and winter months, these two species represent the highest revenues (Batista et al. 2009).

The objective of this study was to evaluate the impact of the LSMP on the abundance and biomass of the local populations of cuttlefish and Senegalese sole.

Materials and methods

Study area

The LSMP is located in the Setúbal Peninsula (Portugal) covering an area of approximately 53 km^2 and stretching over 38 km of coastline (Figure 5.1). It includes a narrow rocky reef down to 15 m deep and wider soft substrates (sand and mud) down to 100 m. This MPA includes one no-take zone of 4.3 km^2, four partial protection zones (only octopus traps and jigs allowed) totalling 21 km^2 and three complementary protection zones (traditional fishing gears allowed for vessels smaller than 7m) totalling 28 km^2. Although designated in 1998, the final planning design of this MPA was achieved in 2005, with full implementation of the management measures only in 2009. The use of fishing nets was banned from the no-take zone and the partial protection areas in August 2007, with the implementation of the full no-take area only taking place in 2009. In addition, all fishing vessels need a permit to operate inside the LSMP. Recreational angling is only permitted in complementary areas and spearfishing is prohibited in the entire LSMP.

Experimental fishing

In order to estimate the abundance and biomass of cuttlefish and Senegalese sole in the LSMP, a total of 106 experimental trammel net fishing sets were carried out. The experimental net was made of monofilament and consisted of an inner mesh panel of 100mm stretched mesh and two outer panels of 600mm stretched mesh. The inner net monofilament mesh is of 0.30mm in diameter while that of the outer net is 0.50mm. The net was constructed to have 50 inner and 3 outer meshes in height, with a total height of 1.60m. Each net had a total of 50m in length and each set consisted of 10 nets.

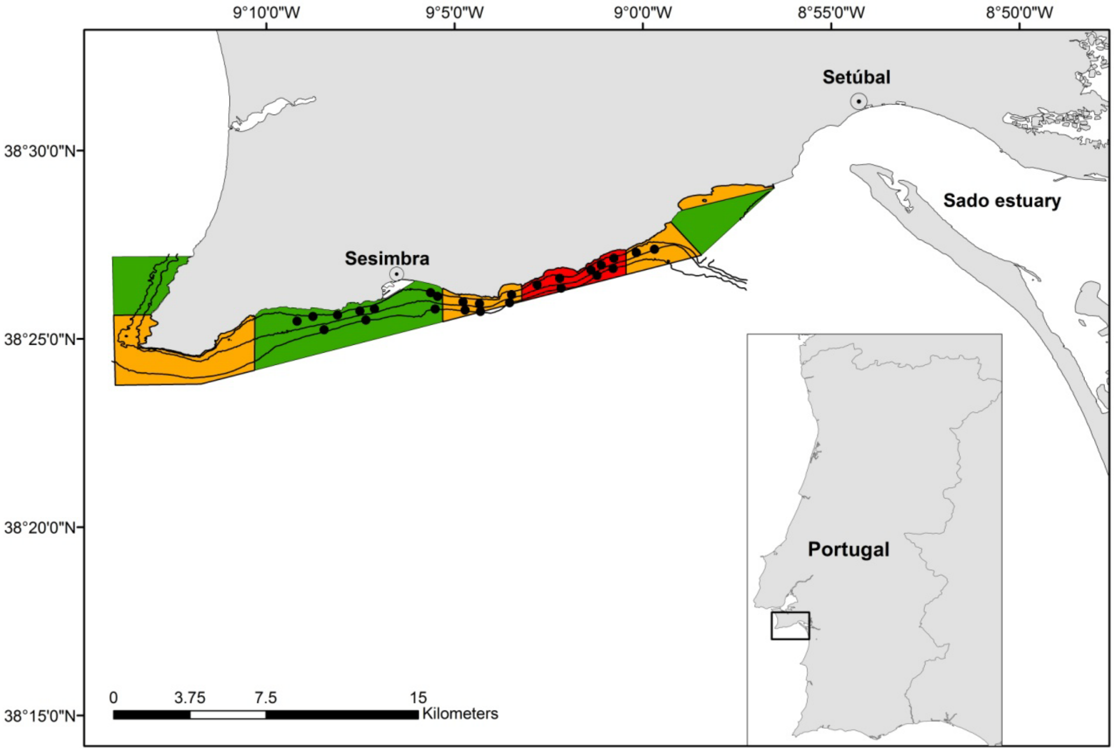

Figure 5.1. Map of the study area. The black dots indicate the location of the experimental fishing sets. The red area corresponds to the no-take area, the orange areas represent the partially protected areas and the green areas stand for complementary protection areas.

These fishing experiments took place in all protection levels: no-take area, partial protection and the complementary protection. Sampling took place three times during the first year after the ban of fishing nets in the no-take and partial protection areas (December 2007, May 2008 and October of 2008) and three times three years after the ban (November 2010, May 2011 and October of 2011). Fishing trials took place in two bottom types, sandy bottoms (12-20 m) and muddy bottoms (35-45 m) using 500m nets per set. Nets were set after sunrise and hauled 20 to 24 h later. The catch was sorted onboard to the species level and fish were measured to the nearest millimetre. Specimens were released alive whenever possible.

Data analysis

Catches of trammel net sets were standardized as catch per unit effort (CPUE) in number (n) and weight (kg) per 1000 m of net and per 24 h. Cuttlefish weight was obtained by using the weight/length relationship $W = 0.325xML^{2.64}$, where W is the weight in g and ML the mantle length in cm (Serrano 1992). Individual Senegalese sole weight was obtained by the weight/length relationship $W = 5.29E\text{-}06xTL^{3.104}$, where W is the weight in g and TL the total length in mm (Gonçalves et al. 1997). Abundance and biomass were log (x+1) transformed after tested for homogeneity of variance (Levene's test).

In order to determine the effects of protection on the local population of Senegalese sole and cuttlefish, differences between the areas where each species was protected from fisheries, hereafter designated as 'impact' and areas where species were accessible to fisheries designated as 'control', were examined using ANOVA following a beyond BACI experimental design (Underwood 1992, 1994). A three-way model was used to examine patterns on the abundance and biomass. The 'location' factor, which includes the 'impact' and 'control' areas, was set as a random factor. The factor 'before/after' was set as a fixed factor with two levels. The 2007 and 2008 experimental fishing campaigns were regarded as 'before' the ban of fishing nets since they occurred during the first year of ban. In fact, for cuttlefish this period was before the onset of protection, since fishing gears such as jigs were still allowed in all areas. The factor 'time of sampling' was set as a random factor nested in the 'before/after' factor. Three replicates for each time x location were used. This design allowed us to test whether there was an interaction between locations through time (Underwood 1992, 1994).

Since Senegalese sole is captured using fishing nets, the areas were these were banned (no-take and partially protected areas) were regarded as the putative 'impact' zone while the complementary protection zone was used as 'control'. For cuttlefish the no-take area corresponded to the 'impact' while the partial and complementary protection zones correspond to 'control' because fishing gears that target cuttlefish such as jigs are still allowed in the partially protected areas.

Results

Large variations were observed in the CPUE of cuttlefish, both in abundance and biomass (Figure 5.2 and Figure 5.3). However, the beyond BACI analysis showed that neither the number of individuals nor the biomass of cuttlefish differed between the no-take area and the controls (Table 5.1). There was also no difference in the temporal pattern from one control location to other. More importantly, there was no significant difference between before and after the implementation of the reserve in either the no-take or the control areas (Table 5.1).

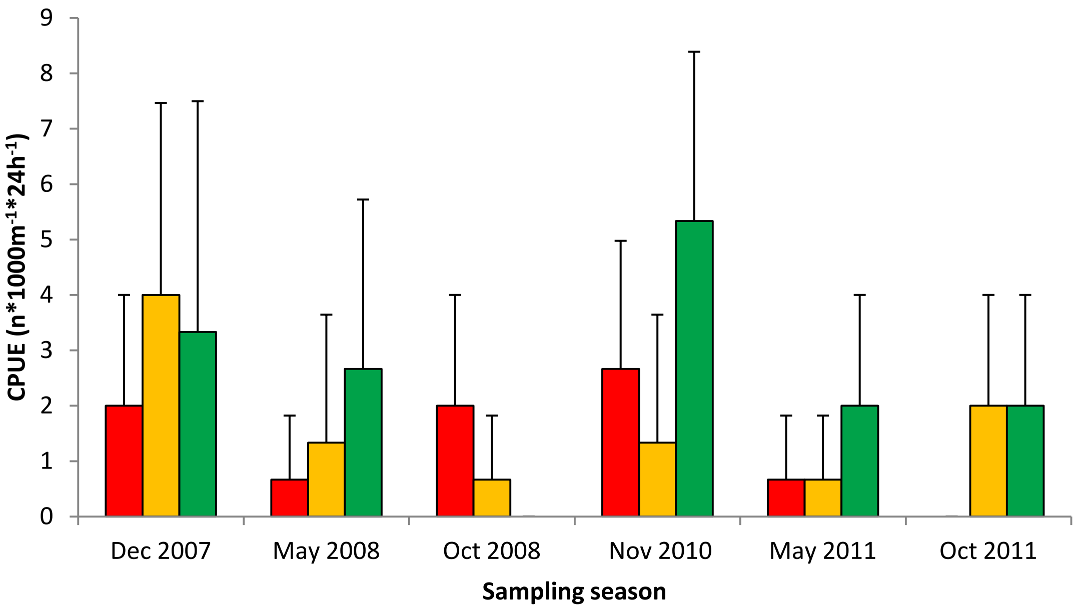

Figure 5.2. Mean CPUE (+SE) in number of individuals of cuttlefish for each of the experimental fishing campaigns. Note: red bars represent the no-take zone; orange bars symbolize the partial protection areas and the green bars indicate the complementary protection.

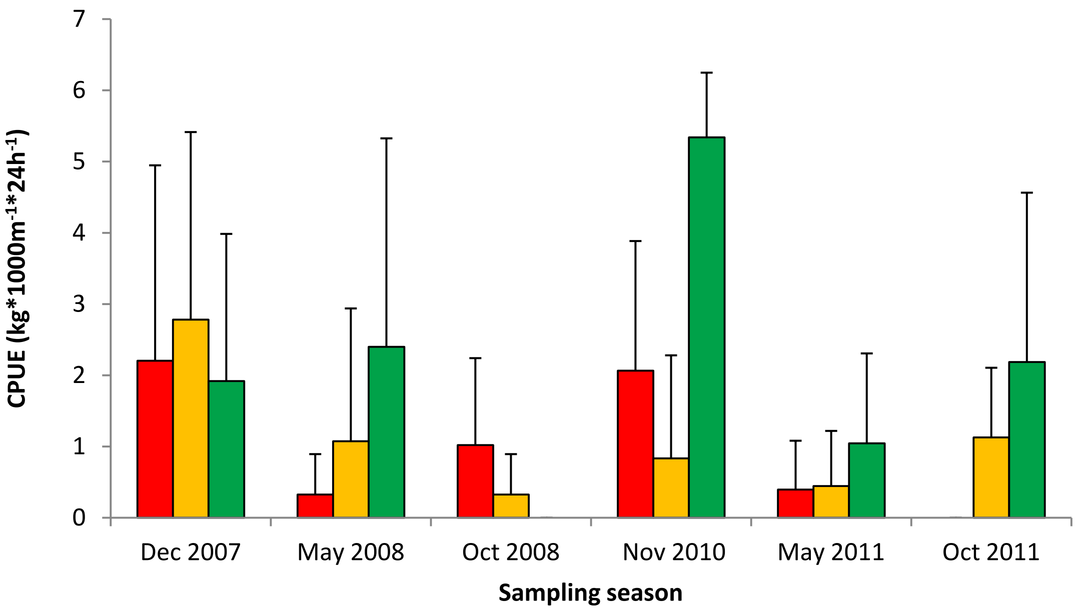

Figure 5.3. Mean CPUE (+SE) in weight (kg) of cuttlefish for each of the experimental fishing campaigns. Note: red bars represent the no-take zone; orange bars symbolize the partial protection areas and the green bars indicate the complementary protection.

For Senegalese sole only the data from fishing trials on shallower sandy bottoms was analysed with the beyond BACI design since there were no captures on muddy bottoms in over 60% of the trials. A decline both in abundance and biomass was observed for Senegalese between the two sampling periods (Figure 5.4 and Figure 5.5). The results of the beyond BACI analysis show that temporal trends in mean abundance and biomass of Senegalese sole were similar among the three locations 'before' the fishing net ban in the no-take and partial protection areas (Table 5.2: T(Bef)xC and T(Bef)xI were non-significant). Additionally, temporal trends in mean abundance and biomass of Senegalese sole did not differ significantly between the two putatively impacted zones (no-take and partial protection), as well as between these and that of the control area from before to after the ban (Table 5.2: BxC and BxI were non-significant). Therefore, no significant change was detected in mean abundance or biomass of Senegalese sole attributable to the ban of fishing nets.

Table 5.1. Summary of the asymmetrical analysis of variance (ANOVA) of the abundance (log (n+1)) and biomass (log (kg+1)) of cuttlefish at the control (complementary and partially protected) and impact areas (no-take) sampled 3 times during the first year after the fishing nets were banned and 3 times 3 years later. No significant differences were found for any of the F values for α=0.05.

			Abundance (Log(n+1))		Biomass (Log(kg+1))	
Source of variation		df	MS	F	MS	F
Before/After	=B	1	0.00		0.04	
Among Times (Before/After)	=T(B)	4	1.09		1.17	
Among Locations	=L	2	0.56		0.74	
Total protection vs. Controls	=I	1	0.57		0.57	
Among Controls	=C	1	0.56		0.92	
B x L		2	0.73		0.71	
B x I		1	0.46	0.79	0.34	0.84
B x C		1	1.00	1.72	1.07	2.62
T(B) X L		8	0.53		0.41	
T(Bef) X L		4	0.59		0.37	
T(Bef) X I		2	0.83	1.46	0.46	1.08
T(Bef) X C		2	0.36	0.56	0.27	0.57
T(Aft) X L		4	0.47		0.45	
T(Aft) X I		2	0.38	0.65	0.25	0.61
T(Aft) X C		2	0.55	0.95	0.64	1.57
Residual		36	0.58		0.41	
F-ratios						
B x I vs B x C				0.46		0.32
T(aft) x I vs T(aft) x C				0.68		0.39
T(aft) x I vs T(bef) x I				0.45		0.57
T(aft) x C vs T(bef) x C				1.70		2.77
T(bef) x I vs T(bef) x C				2.61		1.90

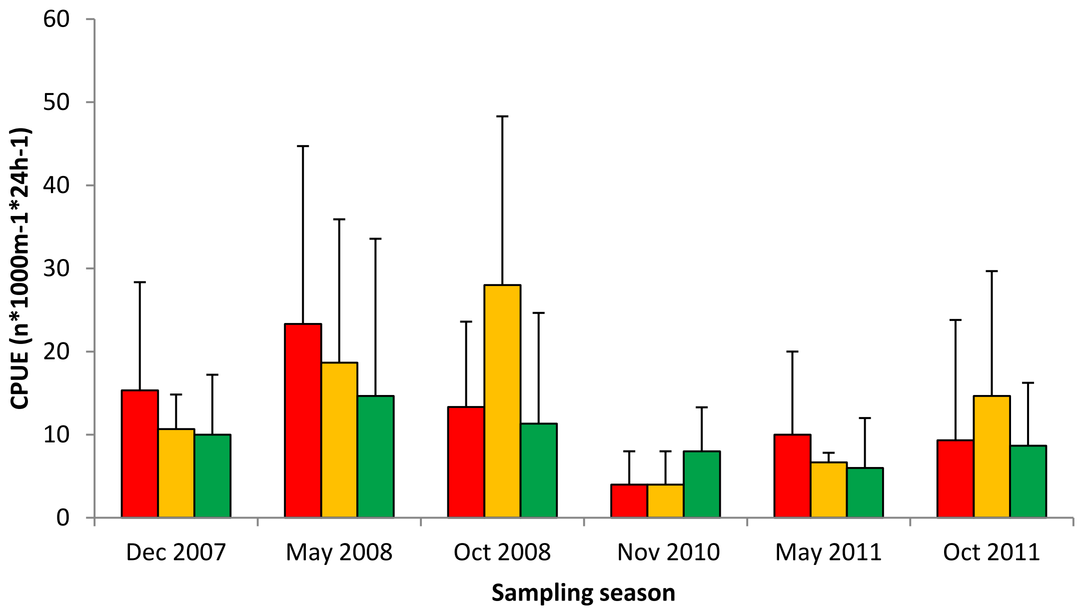

Figure 5.4. Mean CPUE (+SE) in number of individuals of Senegalese sole for each of the experimental fishing campaigns. Note: red bars represent the no-take zone; orange bars symbolize the partial protection areas and the green bars indicate the complementary protection.

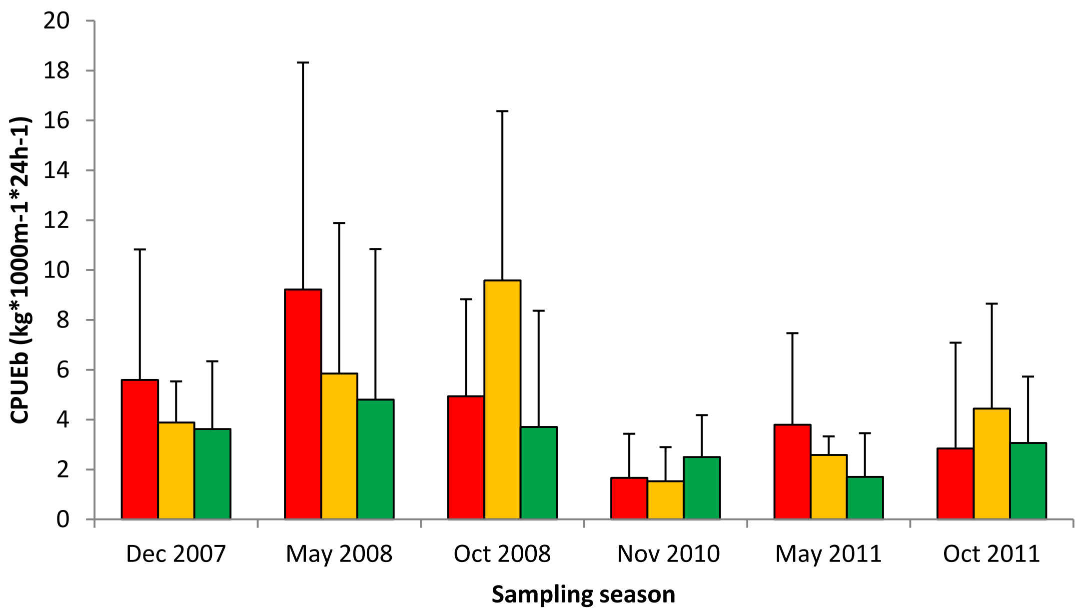

Figure 5.5. Mean CPUE (+SE) in weight (kg) of Senegalese sole for each of the experimental fishing campaigns. Note: red bars represent the no-take zone; orange bars symbolize the partial protection areas and the green bars indicate the complementary protection.

Table 5.2. Summary of the asymmetrical analysis of variance (ANOVA) of the abundance (log (n+1)) and biomass (log (kg+1)) of Senegalese sole at the control (complementary) and impact areas (no-take and partial areas) sampled 3 times during the first year after the fishing nets were banned and 3 times 3 years later. No significant differences were found for any of the F values for α=0.05.

			Abundance (Log(n+1))		Biomass (Log(kg+1))	
Source of variation		df	MS	F	MS	F
Before/After	=B	1	3.97		2.16	
Among Times (Before/After)	=T(B)	4	0.12		0.07	
Among Locations	=L	2	0.32		0.21	
No-take vs. Controls	=I	1	0.02		0.02	
Among Controls	=C	1	0.61		0.40	
B x L		2	0.38		0.14	
B x I		1	0.23	0.18	0.09	0.17
B x C		1	0.54	0.43	0.18	0.33
T(B) X L		8	0.24		0.13	
T(Bef) X L		4	0.22		0.16	
T(Bef) X I		2	0.10	0.08	0.10	0.19
T(Bef) X C		2	0.34	0.27	0.21	0.40
T(Aft) X L		4	0.26		0.11	
T(Aft) X I		2	0.13	0.10	0.10	0.19
T(Aft) X C		2	0.40	0.31	0.12	0.22
Residual		36	1.26		0.54	
F-ratios						
B x I vs B x C				0.43		0.52
T(aft) x I vs T(aft) x C				0.33		0.90
T(aft) x I vs T(bef) x I				1.36		1.00
T(aft) x C vs T(bef) x C				1.18		0.55
T(bef) x I vs T(bef) x C				0.29		0.49

Discussion

The high variation observed in the cuttlefish CPUE among sampling periods is, most probably, a consequence of stochastic variability on reproductive success and recruitment. Large interannual variation in cuttlefish recruitment has been observed in the English Channel (Royer et al. 2006) and is described for several cephalopod stocks (Royer et al. 2002, Young et al. 2004). In fact, large fluctuations in recruitment success are to be expected in semelparous species, such as the cuttlefish, since the success of their reproduction is highly dependent on favorable environmental conditions at the time of their reproduction (Pierce et al. 2008). Sea temperature and currents are two of the variables that are known to affect cuttlefish recruitment and abundance (Wang et al. 2003). Henriques et al. (2007) have shown that SST and other factors such as wind and current patterns, which are greatly influenced by the North Atlantic Oscillation (NAO), strongly influence the fish assemblages of the LSMP.

No evidence of an effect on Senegalese sole or cuttlefish abundance or biomass due to protection status of the LSMP was found. Although it is possible that our sampling design failed to show putative effects, such designs are considered robust enough and therefore we are confident with regard to the no-effect conclusion. In fact, the use of a beyond-BACI approach is considered one of the most robust and valuable methods to examine the potential effects of MPAs and has been advocated by several authors in order to provide unequivocal empirical evidences of such effects (e.g. Fraschetti et al. 2002, Guidetti 2002, Russ 2002). According to Underwood (1992), with the use of this procedure the effect of protection can be identified and distinguished from the stochastic variability of natural populations. Therefore, the fact that no significant interactions were found between the periods 'before' and 'after' the implementation of the LSMP regulations indicate that, if there was indeed an effect of its implementation on the abundance or biomass of the local populations of cuttlefish and Senegalese sole, then this effect was too small to be detected.

Despite the similar results for both species the reasons for the apparent lack of effect of the LSMP on the abundance and biomass are most probably different. In the case of cuttlefish, the results of a previous acoustic telemetry study appear to provide good evidence to explain the lack of reserve effect (Abecasis et al. 2013). The study showed a lack of site fidelity and the existence of large movements which preclude the reserve effect of the LSMP since it cannot offer long term protection (Abecasis et al. 2013). On the other hand, the results of the acoustic telemetry study of Senegalese sole suggest that the LSMP could be providing enough protection to this species given the high site fidelity and the size of the home range (Chapter 3). Yet, the results of the beyond BACI analysis failed to detect any significant interaction between the impact/control areas and the period before/after the ban of fishing nets in the abundance and biomass of Senegalese sole. A very parsimonious explanation for this lack of effect is the relatively young age of the LSMP. The use of fishing nets in the no-take and partially protected areas was only banned in 2007. It has been proven that the age of marine reserves has a positive impact on commercial fish species abundance and biomass (Roberts et al. 2001, Alcala et al. 2005). Another possibility is that the small size of the reserve precludes such an effect if the establishment of new individuals inside the reserve is limited by an already saturated habitat, especially if adult habitat use is mediated by a territorial system and if juveniles face high competition for access to high quality habitat, thus promoting density-dependent emigration. Such a pattern would, however, promote spillover but not the build up of biomass and, consequently, of increased reproductive output and larval supply from the MPA.

Clearly, a conclusive evaluation will require continued monitoring for a period of years concomitant with that necessary for benefits to be seen for most species elsewhere.

Future studies should also be carried in the areas adjacent to the LSMP in order to test the effects of the implementation of the LSMP in these areas where Senegalese sole and cuttlefish are two of the most important fish resources. Additionally, the analysis of the landing records of the nearby fishing ports of Sesimbra and Setúbal could be interesting. However, it must be taken into account not only that this data is mostly incomplete due to illegal and unreported catches but also that there is no differentiation of sole species in the official landings statistics (Batista et al. 2009, Teixeira & Cabral 2009).

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Chapter 6: Combining multispecies home range and distribution models to evaluate the optimal design of MPAs

This chapter is under review in paper: Abecasis D, Afonso P, Erzini K (submitted) Combining multispecies home range and distribution models to evaluate the optimal design of MPAs.

Chapter 6

Combining multispecies home range and distribution models to evaluate the optimal design of MPAs

Abstract

Marine protected areas (MPAs) are today's most important tools for the spatial management and conservation of marine species. Yet, the true protection that they provide to individual fish is unknown, leading to some of the uncertainty associated with MPA effectiveness. Importantly, there has been very little progress on how to translate the individual scale of movement and home range data to the more relevant population scale when evaluating the effectiveness of protection from MPAs or forecasting their optimal designs. In this study, conducted in a small and recently established MPA (Luiz Saldanha Marine Park, Portugal), we combined the results of individual home range estimation and population distribution models for three species of commercial importance and contrasting life histories to infer 1) the size of suitable areas where they would be fully protected, and 2) the vulnerability to fishing mortality of each species throughout the MPA. The results demonstrate that the protection offered by MPAs varies substantially, depending on the species home range and on the extent of the suitable protected area. This study highlights the need to consider a multi-species approach and provides an explicit framework to upscale from individual telemetry data to the needed population scale that can be of wide applicability for studies evaluating the optimal design of MPAs.

Introduction

Marine protected areas (MPAs) have become a key spatial management and conservation tool for coastal nations worldwide but their effectiveness is largely uncertain in most cases, if not all (Kaiser 2011). In fact, the size of the areas where the different species are effectively protected and the amount of time they are available to the fishery is typically unknown.

The adequate design and management of MPAs is highly dependent on the quality of the baseline ecological information. Of particular relevance is the knowledge of the species' site fidelity, distribution and habitat use (Glazer and Delgado 2006; Grüss et al. 2011; Le Quesne and Codling 2009; Schmiing et al. 2013). This data can not only help determine the initial location and correct size of MPAs based on the species habitat requirements, but also provide relevant information for the adaptive management of already implemented MPAs.

Recent studies have presented quantitative models to assess the efficiency of MPAs (Le Quesne and Codling 2009; Moffitt et al. 2009; Walters et al. 2007). However, these models do not take in consideration that no-take areas do not, in most cases, consist of 100% of suitable habitats. It is therefore possible that a no-take area, several times larger than the species home range does not offer adequate protection.

Acoustic telemetry is one of the most widely used methods to track marine species, as it provides long-term, fine scale spatio-temporal data on individual movement and home range (e.g. Abecasis et al. 2012; Abecasis et al. 2013b). However, very little has been done and there is no consensus on how to translate such individual data - the typical output of telemetry studies - into the more relevant population scale projection when evaluating the effectiveness of protection provided from existing MPAs or forecasting their optimal designs.

This study provides a framework to upscale the telemetry data from the individual to the population level. This was achieved by combining information of species home range areas with species distribution models so as to calculate the effective protection provided to three species with contrasting life histories by a small coastal MPA, the Luiz Saldanha Marine Park (LSMP, Portugal). In particular, this study focused on analysing the vulnerability to fishing of the three species – cuttlefish (*Sepia officinallis*), Senegalese sole (*Solea senegalensis*) and white seabream (*Diplodus sargus*) - and on estimating the size of suitable areas where these species are in fact protected from local fisheries. Arguably, an MPA design based on the requirements of only three species is unlikely to ensure the full protection of all local marine species. Nevertheless, the contrasting life histories of these three species, all of which are also of key commercial importance for the region, ensure the wide spectra needed to demonstrate the wider applicability of this framework towards this MPA and elsewhere. This study is also innovative in combining typical finfish with cephalopods and flatfishes, seldom used in MPA studies (Horta e Costa et al. 2013; Lester et al. 2009)

Species distribution models (SDMs) have become an important tool for studies in biogeography, ecology, species management, conservation biology and climate change (Bean et al. 2012; Elith and Leathwick 2009b; Guisan and Thuiller 2005; Guisan and Zimmermann 2000). These statistical methods associate species data (presence, presence/absence or abundance) with mapped environmental predictor variables and/or geographical information so as to provide information on the presence of species across the entire area of interest (Guisan and Zimmermann 2000).

Recent developments in the field of SDMs have produced multiple methods (Elith and Graham 2009; Elith et al. 2006) which are now commonly used to predict species distribution, including generalized linear or additive models (GLMs or GAMs), multivariate adaptive regression splines (MARS), boosted regression trees (BRT) and maximum entropy modeling (Maxent) (Elith and Leathwick 2009a; Newbold 2009). The low data requirements and the ease of integration with GIS analysis have made Maxent one of the most widely used software for SDM (Elith et al. 2006, Elith & Leathwick 2009). Different comparative studies using a wide range of data demonstrated that Maxent is consistently among the best performing methods (Elith et al. 2006; Hernandez et al. 2006; Navarro-Cerrillo et al. 2011). Maxent is a machine learning method that predicts potentially suitable environmental conditions for the species through the use of presence records and a set of environmental variables, continuous and/or categorical, that are likely to influence the species fitness and long-term persistence (Phillips et al. 2006; Phillips and Dudík 2008).

The main objectives of this study were 1) to determine the amount of suitable habitats where three of the most commercially important fish species are effectively protected 2) to determine the vulnerability of these species to fishing throughout the LSMP.

Materials and Methods

Study area

This study took place in the Luiz Saldanha Marine Park (LSMP), which was established in 1998 yet only fully implemented in 2009. Located on the Portuguese western coast, this MPA covers an area of approximately 53 km^2 stretching over 38 km of coastline. It includes a narrow stretch of rocky reef habitats down to a depth of 15 m and a wider stretch of soft substrates (sand and mud) down to 100 m. The LSMP regulations specify different zones and limitations to extractive activities. Commercial fisheries have different limitations within the different zones: all fisheries are excluded from a no-take zone of about 4.3 km^2; octopus traps and jigs are allowed within the four partial protection zones totalling 21 km^2; and commercial fishing boats less than 7m long are allowed to operate using traditional fishing gear within the three complementary protection zones totalling 28 km^2. Spearfishing is prohibited within the entire area of the LSMP, whereas recreational angling is only allowed within the three complementary protection zones. With these regulations, the cuttlefish is only fully protected from its targeted fishery (trammel nets and jigs) within the no-take zone, whereas the white seabream and the Senegalese sole are fully protected from their targeted fisheries (longlines and nets, respectively) within both the no-take zone and partial protection zone.

Study species

This study focused on three species: the sparid *Diplodus sargus* (white seabream), the flatfish *Solea senegalensis* (Senegalese sole) and the cephalopod *Sepia officinalis* (cuttlefish). The three species are very distinctive from each other as they present contrasting ecological traits and life-histories, but share high economical value across southern Europe. In the LSMP area, both the cuttlefish and the Senegalese sole are targeted by the local small-scale commercial vessels that operate with trammel nets and gillnets (Batista et al. 2009), whereas the white seabream is mainly captured by artisanal longlines and recreational fishing (Veiga et al. 2010). Their habitat preferences are also very distinct: the Senegalese sole is a benthonic species that occupies soft substrates, the white seabream is a demersal species that prefers hard substrates such as rocky reefs, but also forages on soft substrates and the cuttlefish is a nekton-benthonic species that makes use of both types of substrates. All three species have very different life-histories, even though they all use estuaries as nursery areas. Cuttlefish are semelparous species with a maximum life time of about two years (Le Goff and Daguzan 1991), whereas the Senegalese sole and the white seabream are iteroparous species that can reach 8 and 18 years old respectively (Abecasis et al. 2008, Teixeira and Cabral 2010). By focusing on species that present such different biological, ecological and economic characteristics, this study should allow us to shed light on the benefits and performance of this MPA for a wider range of species.

Species distribution modelling

To model species distribution, we used the Maxent software version 3.3.3k (available from http://www.cs.princeton.edu/~schapire/maxent/) with the maximum number of iterations set to 5000. Based on the ecological knowledge of the three species and the availability of environmental data for the area, we selected the following variables as explanatory variables in the model: 'habitat', 'bathymetry', 'curvature', 'slope', 'aspect' and 'distance to rocky bottom'. The variables 'curvature', 'slope' and 'aspect' represent the surface curvature, the rate of maximum change in depth from each cell and the direction that the slope is facing (North, South, East, West), respectively. The variables 'habitat' and 'aspect' were set as categorical variables, whereas the remaining variables were set as continuous. Information on 'habitat' was collected by the

Portuguese Sea and Atmosphere Institute (IPMA), using acoustic and video surveys, during the BIOMARES project (Cunha et al. 2011). This data was presented in raster format with a cell size of approximately 40m x 40m. The variable 'bathymetry' was estimated by combining data from a recent bathymetric survey carried out by IPMA with information collected by the project BIOMARES. All the data used to estimate 'bathymetry' were included in the development of a raster file using the 'inverse distance weighting' (IDW) raster interpolation tool available in the 3D Analyst Tools package for ArcGIS 9.3. This final raster had a cell size of approximately 39m x 39m and was the basis for estimating the variables 'slope', 'aspect' (slope direction) and 'curvature' using the Raster Surface tools in the 3D Analyst Tools package for ArcGIS 9.3. In addition, we estimated the variable 'distance to rocky bottom' by extracting the areas containing rocky substrate using the 'extract by attributes' tool and then creating a new raster file with the Euclidean distance to the nearest rocky bottom. All the raster files produced for the different variables were converted to the same projection, cell size and geographic limits. We used the Pearson correlation coefficient (*r*) to test for correlations between variables, given that high correlations between variables should be avoided in SDM as they could lead to over fitting of the model.

We used presence data from previous acoustic telemetry studies on these three species (Abecasis et al. 2013a; unpublished data) as training data for the SDMs. A sampling bias file with the extension of the acoustically monitored area was used to remove the sampling distribution bias (Phillips et al. 2009). Data from experimental trammel net monitoring surveys was used as independent test data for cuttlefish and Senegalese sole (Abecasis et al. 2013a; unpublished data). For the cuttlefish, however, given that its acoustic telemetry data presented a short temporal extent (November to December), we only considered the trammel net surveys carried out during autumn, which correspond to approximately the same time frame. As for the white seabream we obtained test data from underwater visual observations given that this species is rarely caught by the trammel nets (for more details see Horta e Costa et al. 2013). We ran models with regularization multipliers of 0.5, 1, 2, 2.5 and 3, and compared them using the small sample size corrected Akaike Information Criterion (AICc), estimated using ENMTools (Warren and Seifert 2011), as recommended by Rodda et al. (2011).

The regularization multiplier parameter affects how closely fitted the output distribution is – a value smaller than the default of 1.0 will result in a closer fit to the given presence records, while a larger regularization multiplier will give a more spread out, less localized prediction. The AICc approach weights model fit with the number of included variables to provide a relative score for each model. This score was then used to rank the models so as to determine the best resembling model based upon a combination of explanatory power and parsimony criteria (Burnham et al. 2011).

After selecting the most adequate regularization multiplier we tried different feature approaches by running models using hinge only, linear plus quadratic and auto features. Following the comparison of the different models with the AICc approach, we proceeded with the jackknife test of variable importance to see if any of the variables could be removed without sacrificing the model performance. The jackknife test calculates the drop in performance as each variable is removed from the fully specified model. We started with all variables and then proceeded by removing each variable one by one based on the drop of the regularized training gain.

The area under the receiver operating characteristic curve (AUC) was used for model evaluation (Elith 2002). Although Lobo et al. (2008) considered that AUC was not appropriate for model comparison, Elith et al. (2011) have found it suitable to test for the model's predictive performance. The AUC statistic ranges between 0 and 1, with 1 representing a perfect model and 0.5 a model no different from random. To test the significance of the SDMs, we tested the AUC value against a null distribution of expected AUC values based on random sampling (Raes and ter Steege 2007). For each species, 99 random null models were created and the 95 % confidence interval was calculated. Since the presence locations were biased, the randomly drawn points were selected from the acoustically monitored area to avoid higher chances of significantly deviating from the null model.

Because Maxent produces continuous models, thresholds were adopted to make a distinction between suitable and unsuitable habitat areas. Two thresholds were applied, the lowest presence threshold (LPT) and the maximum sensitivity plus specificity threshold (MSST). The LPT, also known as minimum training presence, is the lowest prediction value returned by Maxent for a location with observed presence of the species and is one of the most commonly used thresholds (Bean et al. 2012; Pearson et al. 2007; Thorn et al. 2009). The MSST is equivalent to finding a point in the receiver operating characteristic curve whose tangent slope equals 1 (Cantor et al. 1999). This threshold is one of the various sensitivity-specificity methods, comparable with each other in performance, and has been shown to achieve better results than LPT (Bean et al. 2012; Hernandez et al. 2006; Liu et al. 2005). The performance of the binary models was measured using the true skill statistic (TSS). The TSS is independent of prevalence and its results are highly correlated with the AUC statistic (Allouche et al. 2006). TSS varies between -1 and 1, where values below 0 represent models that perform no better than random and values close to 1 represent perfect agreement.

We combined information provided by the SDMs with home range information so as to determine the effective protection provided by the LSMP to the three study species. The minimum, average and maximum length of home range for each species was estimated from information collected by previous telemetry studies conducted in the area (Abecasis et al, 2013a; unpublished data). From the SDMs, we calculated the size of the suitable areas where species were fully protected (no-take zone for cuttlefish and no-take plus partial protection zones for Senegalese sole and white seabream). Vulnerability to fishing (V_X) was estimated for each discrete point along the coast of the LSMP for an individual with its home range centered there (Moffitt et al. 2009),

$$Vx = \frac{1}{H} \sum_{i=-(\frac{H}{2})}^{i=+(\frac{H}{2})} c_x + i$$

where *H* is the home range length and *c* is the coastline defined as

$$cx \begin{cases} 0 & \text{reserve} \\ 1 & \text{non reserve} \end{cases}$$

Results

Species distribution models

The Pearson correlation test did not reveal any strong significant correlation between the continuous variables: 'slope', 'bathymetry', 'curvature' and 'distance to rock' (r^2=0 to 0.07), with only a medium negative correlation between 'bathymetry' and 'distance to rock' (r^2 = - 0.38). Therefore, all variables were initially included in the Maxent models.

Cuttlefish

The cuttlefish distribution model with the regularization parameter of 3 presented the highest AUC value. Nevertheless, the AICc revealed that the model using the regularization parameter of 1 was the most adequate from a parsimonious perspective (Table 6.1). The cuttlefish distribution model using auto features performed better than the models using only hinge features or linear plus quadratic features (Table 6.1). The AUC value for the models with different predictor variables was higher for the model containing the variables 'bathymetry', 'distance to rock', 'aspect' and 'slope' (Table 6.1). However, the AICc analysis suggests that the best performance was achieved when using all variables except 'slope' (Table 6.1).

The jackknife test revealed that 'bathymetry' was the variable that contributed the most to the model, given that removing this variable resulted in the largest reduction of the regularized training gain. The percent contribution of each variable is shown in Table 6.2.

Table 6.1. Sample size corrected Akaike information criterion (AICc) and area under the receiver operating characteristic curve (AUC) results for the cuttlefish distribution models. Variables: A – 'aspect'; B – 'bathymetry'; C – 'curvature'; D – 'distance to rock'; H – 'habitat' and S – 'slope'.

Regularization multiplier	AICc	Test AUC	Number of parameters
0.5	2229.326	0.648	65
1	2089.050	0.757	27
2	2106.396	0.766	21
2.5	2118.178	0.770	21
3	2121.860	0.773	19
Features (regularization multiplier = 1)			
Auto	2089.050	0.757	27
Hinge only	2186.138	0.790	36
Linear and quadratic	2140.644	0.785	14
Variables (auto features and regularization multiplier = 1)			
B; H; D; A; C and S	2089.050	0.757	27
B; H; D; A and C	2087.975	0.787	27
B; H; D and A	2121.283	0.789	21
B; H and D	2158.772	0.788	23
B and H	2246.013	0.765	15
B	2274.929	0.764	11

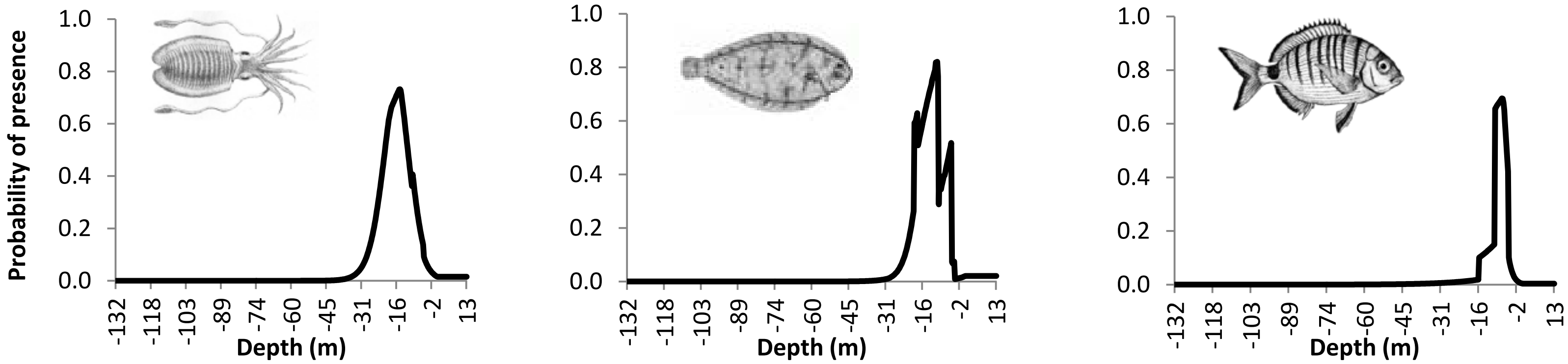
The relationship between 'bathymetry' and 'presence probability' resembles a bell shaped curve that peaks around a depth of 15m (Figure 6.1). The response curve of the relationship between 'presence probability' and 'distance to rocky bottoms' suggests that, at least during the months of November and December, the probability of cuttlefish being present in areas further than 450m away from rocky bottoms is very low (Figure 6.1). Medium sand (category 7) and algae on rock (category 8) were the habitats that presented the highest probability of cuttlefish presence (Figure 6.1).

Table 6.2. Percent contribution of each variable for the cuttlefish, Senegalese sole and white seabream distribution models. – indicates variables that were not used in the final distribution model.

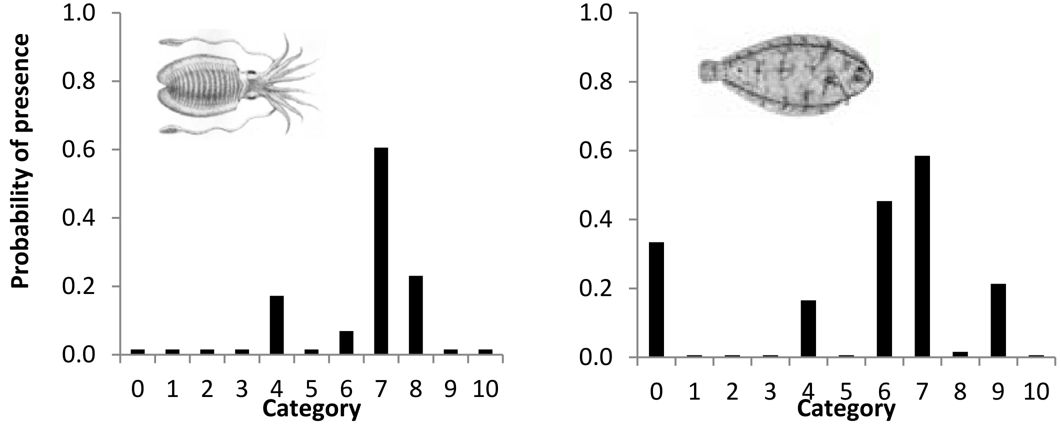
Variable (%)	Cuttlefish	Senegalese sole	White seabream
Habitat	30.0	30.8	-
Distance to rocky bottoms	29.4	-	23.3
Bathymetry	20.9	51.7	58.3
Aspect	15.1	7.0	7.4
Curvature	4.6	1.4	-
Slope	-	9.1	11.0

Figure 6.2 presents the final presence probability map for cuttlefish in the LSMP during the months of November and December (AUC = 0.963; 95% C.I AUC of the biased corrected null model = 0.962). The binary map of suitable and unsuitable areas based on the LPT achieved a TSS of 0.376 (Figure 6.2) while the map using the MSST achieved a TSS of 0.146 (Figure 6.2).

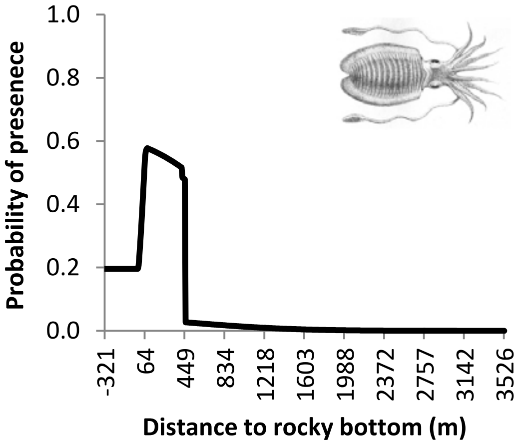
Depth

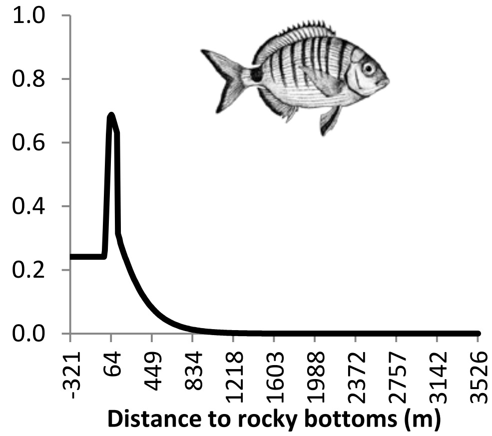


Habitat

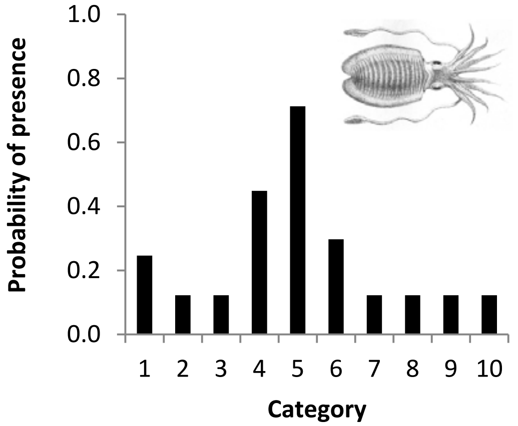


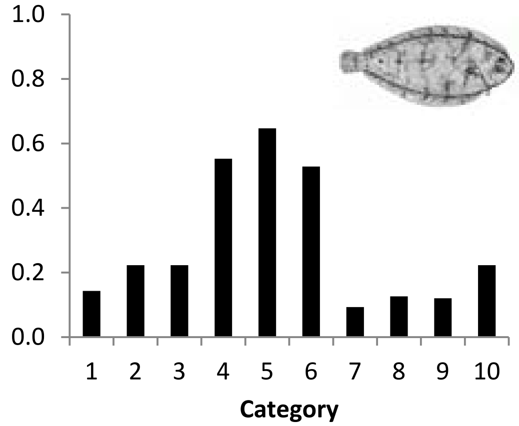
Distance to rocky bottoms

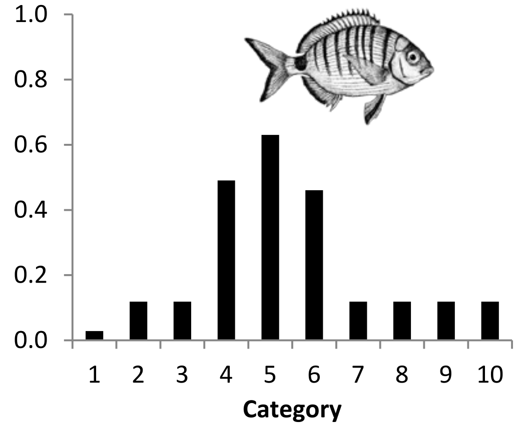




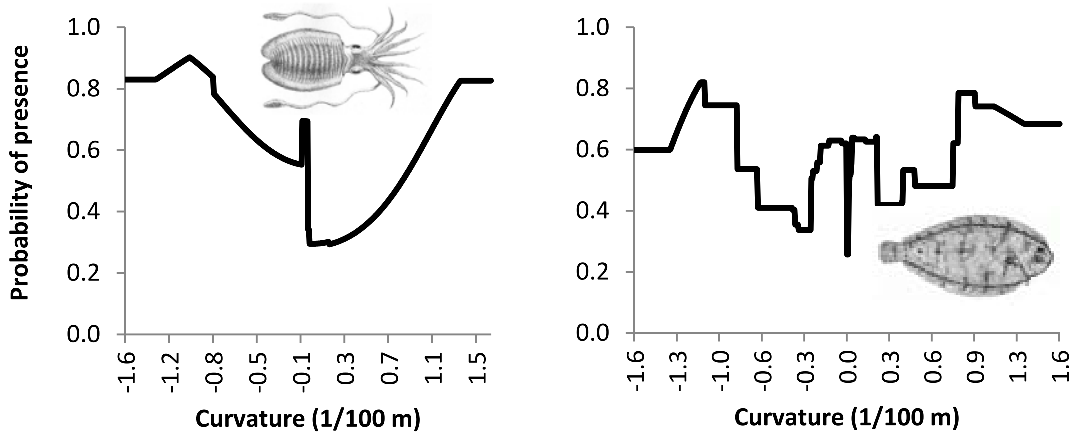
Aspect







Curvature



Slope

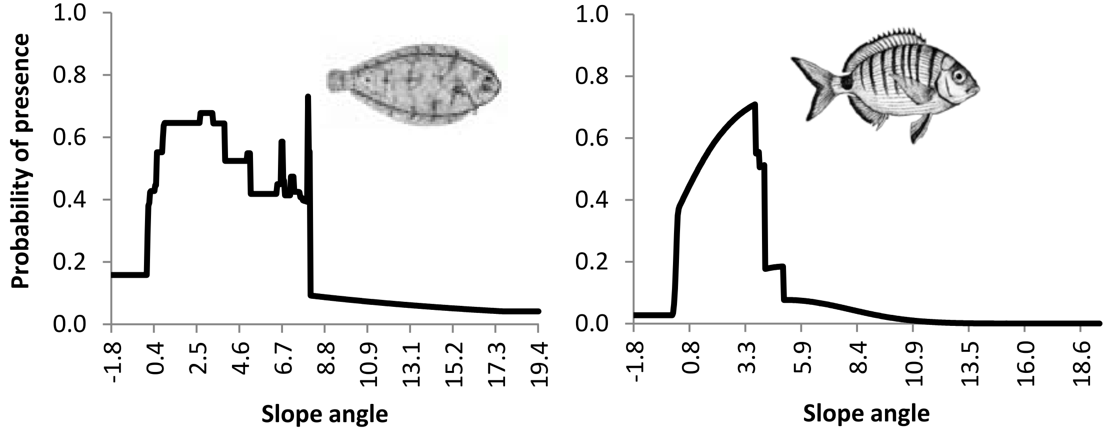


Figure 6.1. Response curves of the different variables for cuttlefish, Senegalese sole and white seabream distribution models. Categories: Habitat: 0 – ‘unknown’, 1 – ‘mud to sandy mud’, 2 – ‘muddy fine sand’, 3 – ‘muddy medium sand’, 4 – ‘coarse sand’, 5 – ‘rocky outcrops’, 6 – ‘fine sand’, 7 – ‘medium sand’, 8 – ‘algae on rock’, 9 – ‘nearshore reefs’, 10 – ‘mixed sands’. Aspect: 1 – ‘’flat’, 2 – ‘North’, 3 – ‘Northeast’, 4 – ‘East’, 5 – ‘Southeast’, 6 – ‘South’, 7 – ‘Southwest’, 8 – ‘West’, 9 – ‘Northwest’, 10 – ‘North’.

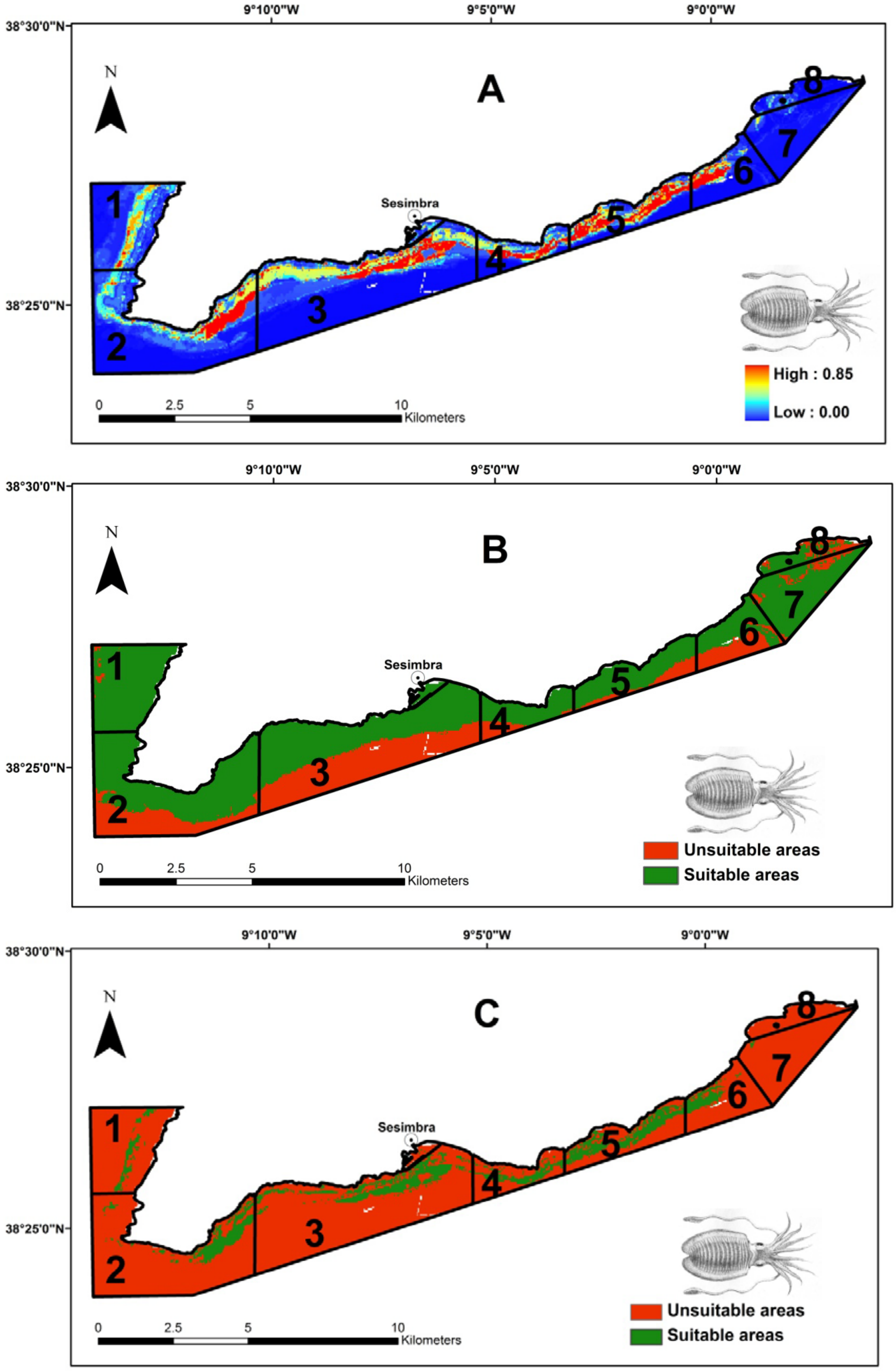


Figure 6.2. Presence probability map (A) and habitat suitability map using the lowest presence threshold (B) and the maximum sensitivity plus specificity threshold (C) of cuttlefish in the Luiz Saldanha Marine Park during the months of November and December. Green areas represent suitable habitats. Zones 1, 3 and 7 are complementary protection areas. Zones 2, 4, 6 and 8 are partially protected areas. Zone 5 is the no-take area.

Senegalese sole

The Senegalese sole distribution model with a regularization parameter of 0.5 achieved the best AUC and AICc values (Table 6.3). Although the model with hinge only features achieved the best AUC the AICc revealed that the model using auto features was the more adequate. According to the AICc the best model was achieved when considering the variables 'bathymetry', 'habitat', 'aspect', 'slope' and 'curvature'. The jackknife test revealed that 'bathymetry' was the variable that contributed the most to the model. The removal of this variable resulted in the largest reduction of the regularized training gain, indicating that bathymetry is the variable with the most useful information and also the one that appears to have the most information that is absent in the other variables. The distribution of Senegalese sole is highly dependent on habitat and depth, which account for more than 80% of the SDM (Table 6.2).

Figure 6.1 presents the probability of presence of Senegalese sole in the study area according to the different variables. The highest probability of presence occurred between the bathymetries of 5 m and 25 m, sea bottoms facing East, South-East and South, fine sands and medium sands habitats, and flatter sea bottoms in general with slope angles between 0.3 and 5.

The map with the final model of the presence probability of Senegalese sole in the LSMP area (AUC=0.951; 95% C.I AUC of the biased corrected null model = 0.944) shows that the highest presence probabilities were found within the no-take zone and adjacent partial protection zones (Figure 6.3). Figure 6.3 also includes the binary maps with suitable and unsuitable areas for Senegalese sole according to the final Maxent model using the LPT, which reached a TSS of 0.308, and using the MSST, which reached a TSS of 0.423.

Table 6.3. Sample size corrected Akaike information criterion (AICc) and area under the receiver operating characteristic curve (AUC) results for Senegalese sole distribution models. Variables: A – 'aspect'; B – 'bathymetry'; C – 'curvature'; H – 'habitat'; and S – 'slope'.

Regularization multiplier	AICc	Test AUC	# Parameters
0.5	45721.64	0.775	99
1	46198.62	0.769	56
2	46947.42	0.760	36
2.5	47245.70	0.762	39
3	47524.66	0.763	35
Features (regularization multiplier= 0.5)			
Auto	45721.64	0.775	99
Hinge only	47367.21	0.778	94
Linear and quadratic	49866.32	0.775	19
Variables (auto features and regularization multiplier= 0.5)			
B; H; A; S and C	45721.64	0.775	99
B; H; A and S	46065.45	0.781	65
B; A and S	47622.46	0.778	65
B and S	48538.18	0.760	63
B	47837.46	0.784	76

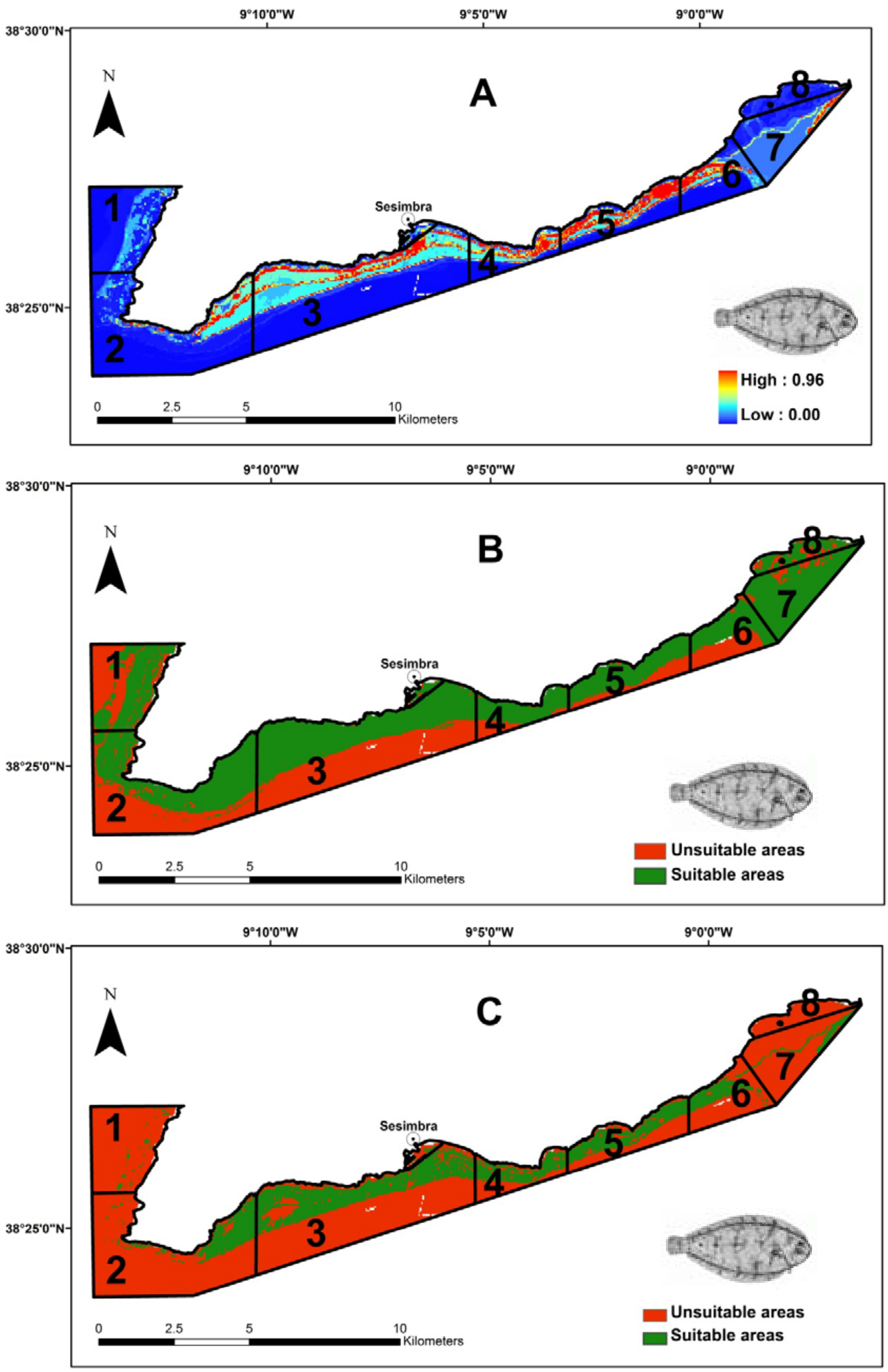


Figure 6.3. Presence probability map (A) and habitat suitability map using the lowest presence threshold (B) and the maximum sensitivity plus specificity threshold (C) of Senegalese sole in the Luiz Saldanha Marine Park. Green areas represent suitable habitats. Zones 1, 3 and 7 are complementary protection areas. Zones 2, 4, 6 and 8 are partially protected areas. Zone 5 is the no-take area.

White seabream

Although the best training AUC results were obtained for the model that used a regularization parameter of 0.5, from a parsimonious point of view the most adequate model was achieved when using a regularization parameter of 2 (Table 6.4). When the features used were changed, the best model, in terms of AICc was achieved when using the auto features option (Table 6.4). According to the AICc results, the best model was achieved when only the variables 'bathymetry', 'distance to rock', 'aspect' and 'slope' were used (Table 6.4).

As for the previous species, 'bathymetry' was the variable that contributed the most to the model, according to the jackknife analysis of variable importance. Besides providing the most useful information, this variable seems to present information that is absent for other variables. Table 6.2 shows the contribution of each variable to the white seabream distribution model.

According to the final distribution model obtained for white seabream the highest probability of presence occurs between the depths of 5 and 10m and when the distance to rocky bottoms is less than 120m (Figure 6.1). The map with the final model of the presence probability of white seabream in the LSMP area (AUC=0.981; 95% C.I AUC of the biased corrected null model = 0.959) demonstrates that the areas with highest probabilities are located near rocky shore areas throughout the entire MPA (Figure 6.4). The map of suitable and unsuitable areas for white seabream using the LPT reached a TSS of 0.494, and the binary map using MSST achieved a lower TSS of 0.260 (Figure 6.4).

Table 6.4. Sample size corrected Akaike information criterion (AICc) and area under the receiver operating characteristic curve (AUC) results for white seabream distribution models. Variables: A – ‘aspect’; B – ‘bathymetry’; C – ‘curvature’; D – ‘distance to rock’; H – ‘habitat’; and S – ‘slope’.

Regularization multiplier	AICc	Test AUC	# Parameters
0.5	1917.178	0.788	59
1	1860.726	0.820	42
2	1859.498	0.854	25
2.5	1876.307	0.865	24
3	1882.392	0.877	21
Features (regularization multiplier = 2)			
Auto	1859.498	0.854	25
Hinge only	1937.731	0.780	32
Linear and quadratic	1970.980	0.941	14
Variables (auto features and regularization multiplier = 2)			
B; H; D; A; C and S	1859.498	0.854	25
B; D; A; C and S	1854.470	0.851	23
B; D; A and S	1853.764	0.842	20
B; D and A	1855.285	0.840	16
B and D	1901.842	0.823	13
B	1980.530	0.814	8

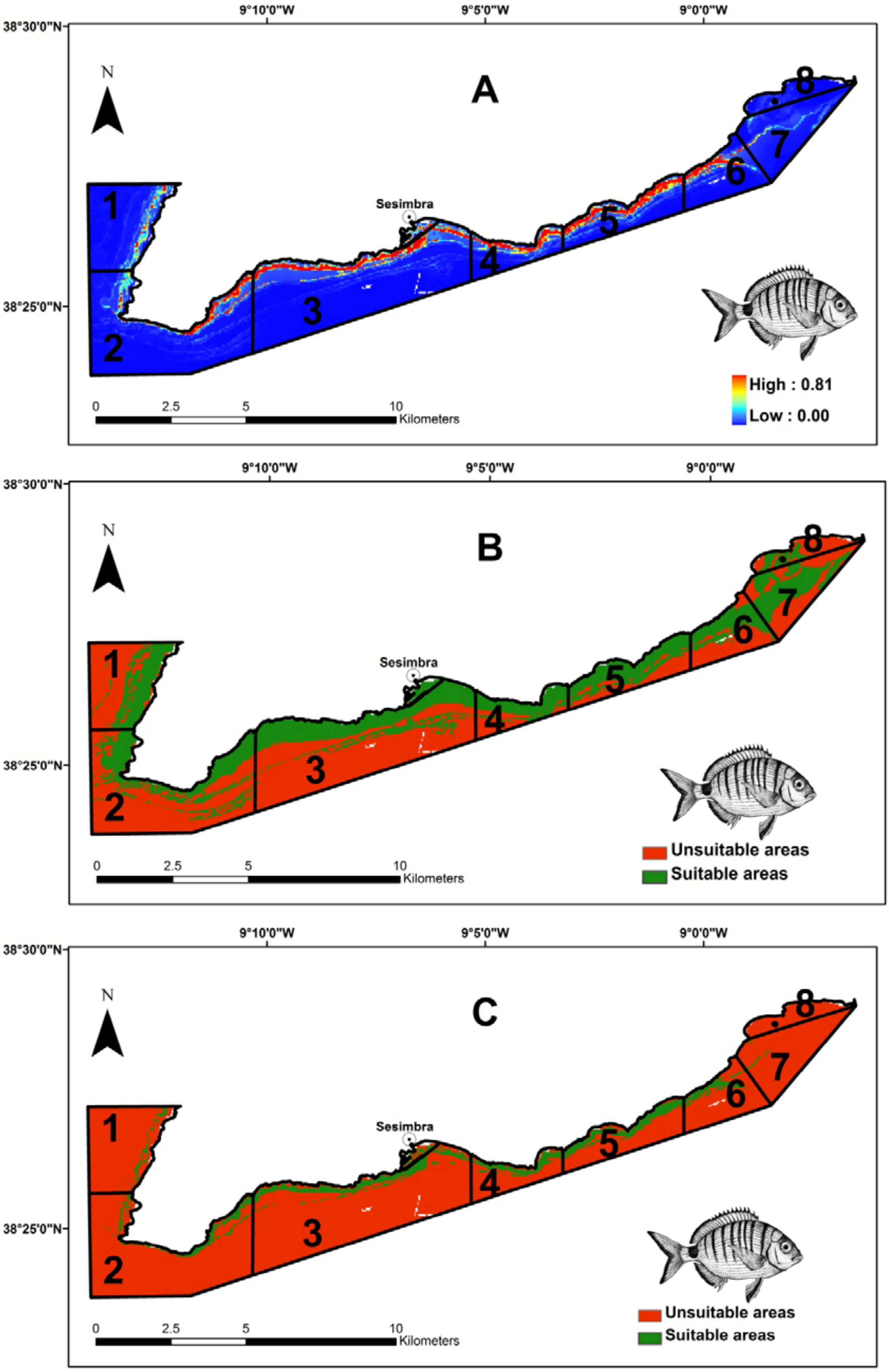


Figure 6.4. Presence probability map (A) and habitat suitability map using the lowest presence threshold (B) and the maximum sensitivity plus specificity threshold (C) of white seabream in the Luiz Saldanha Marine Park. Green areas represent suitable habitats. Zones 1, 3 and 7 are complementary protection areas. Zones 2, 4, 6 and 8 are partially protected areas. Zone 5 is the no-take area.

Vulnerability to fishing

The regulated zones where both the Senegalese sole and the white seabream are protected from fishing are larger than 25 km^2 in total (no-take plus partially protected areas). For cuttlefish, the area where it is fully protected from fisheries is the no-take area which corresponds to approximately 4.2 km^2. Although cuttlefish are also protected in the first 200m from the coastline in partially protected areas these areas were not considered given their small size. However, only a small proportion of these protected areas corresponds to suitable habitats for these species (Table 6.5). The vulnerability to fishing, considering an individual's home range centered in the middle of the no-take area, was 0.0 for every species and home range considered (Figure 6.5). Regarding the western partial protection area the vulnerability to fishing was 0.0 for white seabream and Senegalese sole except when considering the maximum home range for Senegalese sole where it reached a minimum of 0.05 (Figure 6.5). Table 6.6 shows the percentage of suitable habitat in each protection area for one, two and three of the studied species. The overlap of the studied species suitable areas can be seen on Figure 6.6.

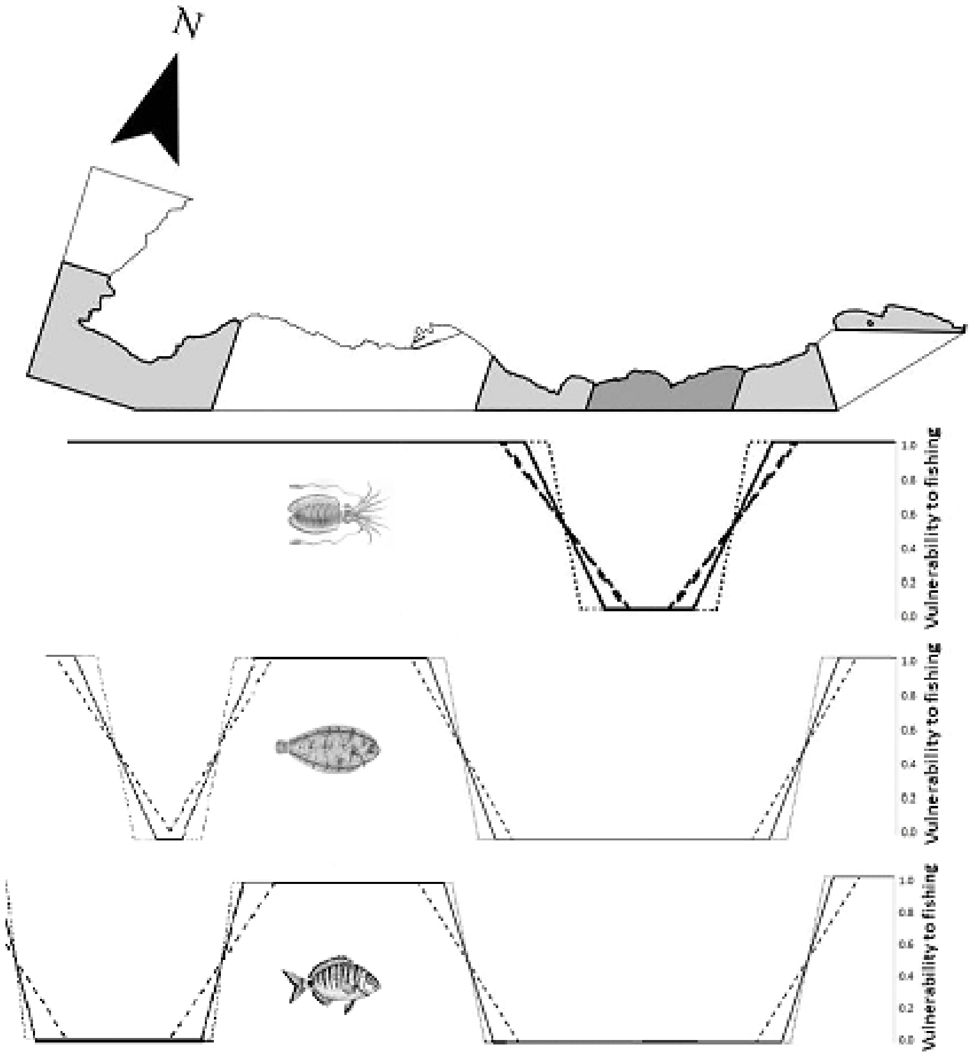


Figure 6.5. Vulnerability to fishing for cuttlefish, Senegalese sole and white seabream in the Luiz Saldanha Marine Park. The dotted line indicates vulnerability estimates considering the minimum home range, the black indicates vulnerability considering the average home range and the dashed line indicates vulnerability considering the maximum home range.

Table 6.5. Average home range area (Avg HR), size of full protection areas (FPA), full protection suitable areas (FPSA), and minimum, average and maximum home range length for cuttlefish, Senegalese sole and white seabream. A – indicates habitat suitability maps based on the lowest presence threshold (LPT). B – indicates habitat suitability maps based on maximum sensitivity plus specificity threshold (MSST).

	S. officinalis A	*S. officinalis* B	*S. senegalensis* A	*S. senegalensis* B	*D. sargus* A	*D. sargus* B
Avg HR (km^2; KUD 95%)	1.26[2]	1.26[2]	1.19	1.19	0.65	0.65
FPA (km^2)	4.2	4.19	25.3	25.3	25.3	25.3
FPSA (km^2)	3.2	1.5	14.7	6.4	10.4	2.9
Minimum HR length (km)	0.9	0.9	0.9	0.9	0.9	0.9
Avg HR length (km)	2.26	2.26	1.89	1.89	1.39	1.39
Maximum HR length (km)	3.5	3.5	2.8	2.8	3.4	3.4

[2] home range areas based on minimum convex polygon. For Senegalese sole and white seabream full protection area includes the no-take area and the partially protected areas (areas 2, 4, 5 and 6 of Figure 2). For cuttlefish the full protection area only includes the no-take area (area 5 of Figure 2).

Table 6.6. Percentage of suitable habitat for 0, 1, 2 and 3 of the study species (cuttlefish, Senegalese sole and white seabream) in the whole MPA and in each protection level.

Species	Entire MPA	No-take area	Partial protection	Complementary protection
0	29.47 %	22.94 %	31.54 %	29.03 %
1	44.29 %	26.19 %	44.64 %	46.77 %
2	18.85 %	28.27 %	16.76 %	19.03 %
3	7.39 %	22.60 %	7.07 %	5.17 %

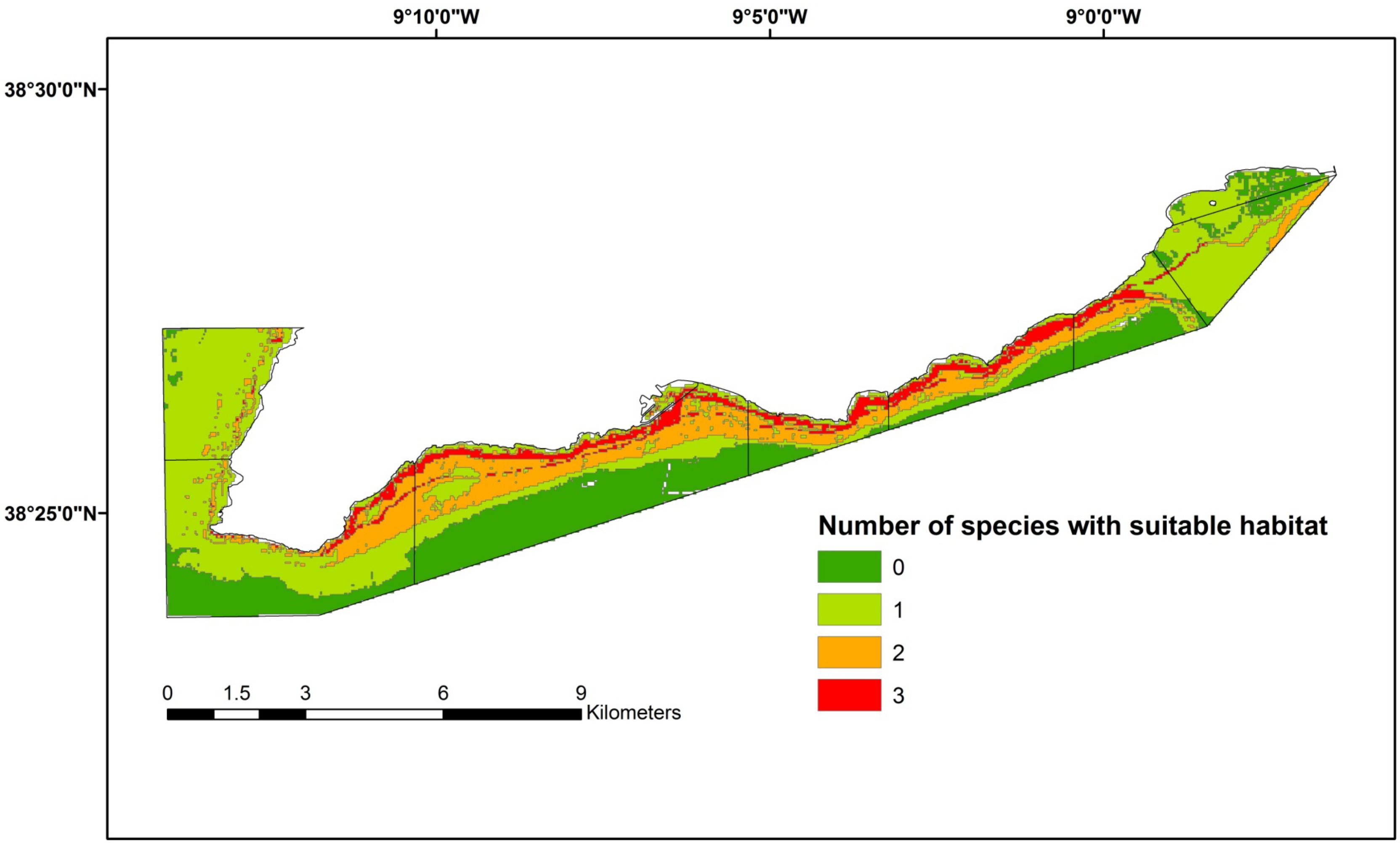


Figure 6.6. Overlap of the studied species suitable areas in the Luiz Saldanha Marine Park.

Discussion

The results of the home range studies together with the strong adequacy of the models produced, allow us to draw important conclusions with regard the habitat suitability of the LSMP for cuttlefish, Senegalese sole and white seabream, and the extent of protection offered by the MPA to these three species.

Model adequacy

The values of TSS and AUC for the different SDMs are evidence that the final models obtained through Maxent are adequate and likely useful instruments for the evaluation of the protection offered by MPA and the exposure to fishing. It could be argued that other potentially relevant input variables (e.g. hydrodynamics and prey distribution and abundance) could also prove useful to improve the predictive power of the spatial distribution models of these species. However, information on such variables was either absent or unavailable at adequate spatial scales for this area. Also, given the number of individual geographical positions obtained from the acoustic telemetry studies, only a small number of variables could be used to develop models that would be parsimonious and yet biologically relevant (Elith and Leathwick 2009a). A model using static environmental variables is more parsimonious because it only requires that the distribution of animals is the result of time independent environmental cues (Aarts et al. 2008).

Importantly, the six variables that were selected to run the SDMs reflect various environmental factors known to influence marine species distributions. The variable ‘habitat’ is widely used because marine species are known to prefer specific and sometimes distinct habitats throughout their life cycle. The variable ‘bathymetry’ is widely used as an indirect proxy for several proximal factors such as temperature, salinity, light and pressure (Elith and Leathwick 2009b). The variable ‘aspect’ was selected as a proxy for hydrodynamic variables since, in this specific case, bottoms oriented to the S and W quadrants are more influenced by strong winds and high seas whereas those facing N and E quadrants are more sheltered. The variables ‘slope’ and ‘curvature’ were also taken in consideration because these have been associated as predictor variables for several marine species (Leathwick et al. 2008; Owens et al. 2012; Schmiing et al. 2013). The variable ‘distance to rocky bottoms’ could be interpreted differently depending on the studied species. The white seabream, for instance, is known to prefer rocky bottom habitats (Abecasis et al. 2013b) and therefore ‘distance to rocky bottoms’ is likely to simply stand for distance to its preferred habitat. In the particular case of the cuttlefish, however, it can be interpreted as distance to spawning grounds, given that this species prefers soft substrate but attaches its eggs to hard substrates like seaweeds, shells and debris, and such substrates are more frequently found on rocky bottoms. In fact, egg clutches were frequently observed in the acoustic receivers’ mooring structures, particularly in those located in vast sandy areas furthest away from rocky bottoms (Abecasis et al. 2013a), where other hard substrates are absent or rare. These observations support the hypothesis that cuttlefish uses habitats closer to rocky bottoms during the spawning season because it is easier to find adequate substrates to attach their eggs. It also explains why the variables ‘distance to rocky bottom’ and ‘bathymetry’ were the most important ones for the final cuttlefish’s SDM, especially taking in consideration that data collection took place during the migration/spawning months of November and December.

The observed Pearson correlation values were not sufficiently high to require the removal of any predictor variable, a step that some authors recommend when high correlations are observed (Gregr 2011; Leathwick et al. 2005; Rodda et al. 2011), even though Maxent is known to be more stable than other regression methods when using correlated variables (Elith et al. 2011; Phillips et al. 2006).

The response curves obtained for the predictor variables were, in most cases, very complex and thus reduced the number of false negatives. This is especially relevant because a low number of false negatives is highly desirable in the particular case of conservation spatial planning because false negatives could lead potentially suitable areas to be left out of management plans (Araújo and Guisan 2006).

The SDMs for Senegalese sole and white seabream were estimated with presence locations obtained throughout almost the entire year. Therefore, it is likely that the suitability and presence probability maps represent an accurate picture for habitat selection of adults of both species in the LSMP. However, this was not the case for cuttlefish, for which presence data was only obtained for a shorter period of time (Abecasis et al. 2013a). Considering that cuttlefish is a migratory species that inhabits a wide range of habitats, it is highly probable that the distribution model obtained underestimates the true distribution for this species during its adult phase. Instead, the information provided by the model should be interpreted as an SDM for the cuttlefish's spawning period because presence data were obtained from adults during this period (Abecasis et al. 2013a).

Habitat suitability

The suitability maps revealed that the LSMP area facing south contains the highest amount of suitable habitats for all three species. The fact that this area is sheltered from the dominant North winds and ocean swell has been put forward as one of the reasons for its high biodiversity (Gonçalves et al. 2003).

Despite the good TSS values associated with the obtained SDMs, the results for white seabream might be slightly biased given that the areas defined as suitable when using the MSST expanded further away from rocky bottoms than anticipated, considering the results of experimental fishing trials. Some bias related with less accurate positions used as training data may have occurred and, as a result, several sandy bottoms areas relatively distant from rocky bottoms scored high probability of presence. Additionally, the method used to obtain the fishes' fine scale positions has limited capability to distinguish between a true position over rocky bottom from an assumed position over sandy bottoms. This limitation results from the fact that the acoustic receivers were located in line with each other, parallel to the coast, and on sandy bottoms. Consequently, nearly all locations were associated with sandy bottoms when, in fact, fish were likely roving over the nearby rocky bottoms within the detection range. This possibility is supported by the inclusion of 'distance to rocky bottoms' as the second most important variable for the white seabream distribution model, confirming previous studies that demonstrated this species' preference for rocky bottoms, even though excursions to sandy bottoms may be frequent (Abecasis et al. 2013b). Regarding the Senegalese sole, the SDM model suggest that fine and medium sand are the habitats with the highest probabilities of presence, which is consistent with the results of habitat selection studies (unpublished data).

Depth was the variable that most contributed to the distribution model of white seabream and Senegalese sole and the third most important for the cuttlefish model. According to our models, the depth interval in which the white seabream and the Senegalese sole were more common is consistent with the results obtained during the experimental fishing (unpublished data). As for cuttlefish, the model suggested a presence probability limited to the interval between 0 and 40 meters deep. However, this might not reflect the true bathymetric distribution of this species, which is known to occur at depths up to 200m (Guerra 2006), particularly because the area monitored during the acoustic telemetry campaigns was confined to shallower habitats given the limited number of receivers available. Moreover, the presence locations were obtained during a short period of time which overlapped the spawning season, during which cuttlefish has been reported to migrate into shallower waters (Ezzeddine-Najai 1997; Gauvrit et al. 1997; Wang et al. 2003).

Protection

Our results demonstrate that the LSMP offers different levels of protection, depending on species. This is not only the result of the different regulations applied to each of the LSMP's zones (e.g. the fishery targeting cuttlefish is only forbidden within the no-take zone, whereas the fisheries targeting Senegalese sole and white seabream are forbidden in both the no-take and partially protected zones) but also a consequence of the different movement patterns and home range areas required by each species.

Overall, the LSMP appears to provide full protection from fisheries to resident individuals of white seabream and Senegalese sole that have their home range centered anywhere in the no-take area or in central areas of partial protection zones. In fact, the results of a recent study suggest that white seabream may already be benefiting from the implementation of the LSMP (Horta e Costa et al. 2013). As for the Senegalese sole, the LSMP seems to play an important role in the protection of local stocks, given the large size of suitable areas for this species located within areas where the species is fully protected. However, the effects of protection to this species are yet to be detected (unpublished data).

Cuttlefish, on the other hand, appears to benefit from less protection as our results indicate higher vulnerability to fishing throughout the LSMP. Previous studies suggest that this species presents low site fidelity and undertakes large migrations (Abecasis et al. 2013a), which is consistent with the short periods of time in which tagged cuttlefish remained within the study area. Therefore, despite the protection provided by the no-take area to cuttlefish, this result might be misleading since this species presents no site fidelity.

It should be taken in consideration that this study focuses on adult individuals only. Important factors such as larval dispersal and recruitment should be taken in consideration when overall MPA efficiency is assessed. Nevertheless, this framework provides important information regarding the protection of commercially important fish species.

Conclusions

This study demonstrates that long-term passive acoustic telemetry data in conjunction with fine scale environmental data and maximum entropy modeling techniques can provide useful SDMs. The combined use of home range areas and SDMs provides a better insight into the true potential of MPAs in effectively protecting marine species, since it can reveal the size of the areas where protection is most effective and a clear, quantitative estimation of the vulnerability to fishing throughout the entire MPA. It can and should be used in identifying multispecies optimal MPA designs, whether this is done a priori or as part of an adaptive management strategy of MPAs.

In the particular case of the LSMP, the levels of protection suggest that this MPA may provide adequate protection for Senegalese sole and white seabream if compliance is adequate, but this is not the case of cuttlefish, given this species' higher levels of exposure to fisheries and very low residency.

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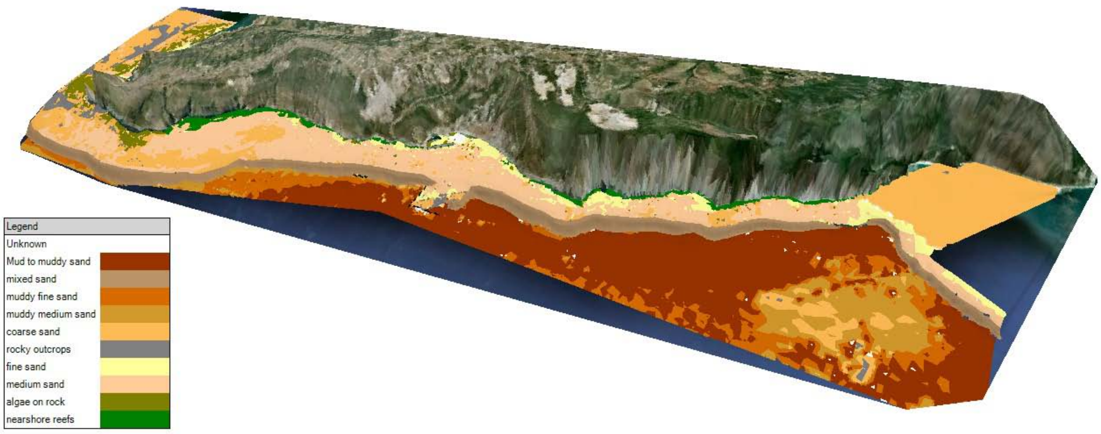
Chapter 7: Towards an adaptive management of the Luiz Saldanha Marine Park: influence of different conservation targets and costs on the design of no-take areas



This chapter is under review in the paper Abecasis D, Afonso P, Erzini K (submitted) Towards an adaptive management of the Luiz Saldanha Marine Park: influence of different conservation targets and costs on the design of no-take areas.

Chapter 7

Towards an adaptive management of the Luiz Saldanha Marine Park: influence of different conservation targets and costs on the design of no-take areas

Abstract

Currently, there is still much discussion on the most appropriate location, size and shape of marine protected areas (MPAs). These three factors were analyzed for a small coastal MPA, the Luiz Saldanha Marine Park (LSMP), for which a very limited amount of local ecological information was available at the start of its implementation in 1998. Marxan was used to provide a number of near optimal solutions considering different levels of protection for the various conservation features and different costs. These solutions were compared with the existing no-take area of the LSMP. Conservation features included eleven habitat types and distribution models for three of the most important species for the local artisanal fisheries. The human activities with highest impact in the study area (commercial and recreational fishing and scuba diving) were used as costs. The results show that the existing no-take area is actually located in the best area. However, the no-take area offers limited protection to vagile fish and covers a very small proportion of some of the available habitats. An increase in the conservation targets led to an increase in the number of no-take areas. The best solution, to target 15% of each conservation feature included three separate no-take areas totalling an area almost twice the size of the existing no-take. Yet, although providing higher level of protection to most habitats individually, each no-take area offers less protection to the fish species. An increase in 30% resulted in four no-take areas but, in this case, the protection offered to the study species was higher given their larger size.

Introduction

The design, size and location of marine protected areas (MPAs) continue to be widely debated but no consensus has yet been found (Glazer and Delgado 2006, White et al. 2009, Gaines et al. 2010). High-quality baseline information is essential not only to determine the design, size and location of MPAs but also to help during the adaptive management process, given that many times such information is not available at the time of MPA creation. Adaptive management is a formal process to improve management practices by learning from the outcomes of operational and experimental approaches (Bunnel et al. 2009). This process should be both scientific and social, requiring an open management where all stakeholders are included. The effectiveness of this approach is based on four key elements (Agardy et al. 2003, Grafton & Kompas 2005). First, it is adaptive, and intended to be self-improving. Second, it is a formal approach that combines the power of science with the practicality of management. Third, it is an on-going process for continually improving management, so the design must connect directly to the actions it is intended to improve. Fourth, although experimental approaches can be incorporated into adaptive management effectively, operational approaches and temporal and spatial scales are emphasized to allow direct connection to the efforts of managers.

Data on habitat type is considered to be the core baseline information since it can often be mapped relatively easily and because it can serve as a proxy for benthic biodiversity (Smith et al. 2006). Most authors agree that this baseline information should also include information on the movements, patterns of habitat use, home range areas and site fidelity of target species to protect (Babcock et al. 2012, Glazer and Delgado 2006). In addition, the need to include socioeconomic data is also becoming evident (Ban et al. 2009). Despite all these considerations most MPAs are designed and implemented with little baseline information and the process rarely takes into consideration all the stakeholders. This was the case of the Luiz Saldanha Marine Park (Portugal), a small coastal MPA implemented starting in 1998.

Geographic information systems (GIS) are one of the best tools to visualize the distribution of different habitats, the location of essential fish habitats and other information relevant to the design and management of MPAs. However, GIS alone does not provide stakeholders with design options, including where best to locate MPAs. The optimization of an MPA design is a problem where the goal is to find the best possible solution given a set of targets and constraints. The recent development of systematic conservation planning software has provided stakeholders with tools that present different MPA design solutions. Of the several available software options, Marxan is the most widely used given its ability to include different types of data and the easy connection with GIS (Ball et al. 2009). It has been used in studies throughout the world including the design of several MPAs such as the Channel Islands and Madagascar (Airame et al. 2003, Allnutt et al. 2012). Marxan tries to find a near optimal solution that reaches the pre-determined conservation targets while maintaining the cost of including planning units as low as possible. The costs associated with each planning unit can represent acquisition costs, management costs or opportunity costs. In marine environments opportunity costs related with commercial and/or recreational fisheries catch or effort are commonly used.

Since Marxan provides several near optimal solutions, it allows dialogue between the different stakeholder groups in order to achieve the best compromise for all user groups involved, in a transparent manner.

In this study we compared the location and design of an already implemented coastal no-take area included in the Luiz Saldanha Marine Park (Portugal) with different near optimal solutions provided by Marxan. The influence of different target levels and costs on the optimal design and location of the no-take area was tested. Besides providing a comparative framework that can be applied elsewhere this information is especially relevant to local stakeholders and managers in order to proceed with adaptive management.

Materials and Methods

This study focused on the Luiz Saldanha Marine Park (LSMP), a marine reserve located in the Setúbal Peninsula (Portugal). The LSMP was established in 1998 but management regulations were only fully implemented in 2009. Although covering an area of approximately 53 km^2, only 4.2 km^2 are included in one no-take zone where all human activities, apart from those related with scientific research and enforcement, are prohibited. The remaining area includes 21 km^2 of partially protected areas, where only octopus traps and jigs are allowed at more than 200m from the shoreline, and 28 km^2 of complementary areas where traditional fishing gears are allowed for vessels smaller than 7m and recreational angling is permitted (Figure 7.1).

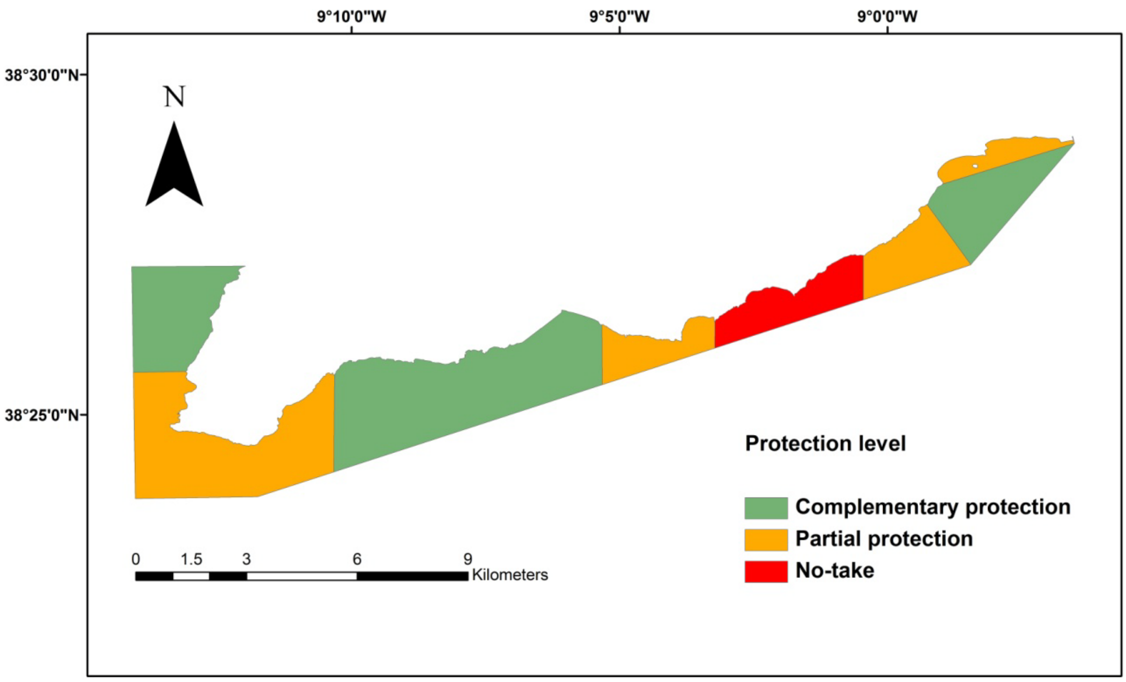


Figure 7.1. Map of the Luiz Saldanha Marine Park including the different protection level areas.

Eleven different habitat types were identified in the LSMP by the Portuguese Sea and Atmosphere Institute (IPMA) during acoustic and video surveys carried during the BIOMARES project (Cunha et al. 2011).

In addition to the information on different habitats, the habitat suitability for three key species (cuttlefish, white seabream and Senegalese sole) was also used. These species present very different life-history, ecological traits and economic values (see Chapter 6). This information comprised three different levels for each species. The first level corresponds to unsuitable areas for the species, the second to areas where the probability of presence was up to 80%, and the final level where the presence probability was over 80%.

The study area was divided in planning units consisting of regular hexagons with an area of 1600 m^2 (each side with 24.816 m). The total amount of each feature was quantified for the entire LSMP and for the no-take area only. This allowed us to know the total area of each conservation feature available in the LSMP and also the amount of protection offered by the already implemented no-take area to each of the 20 conservation features (the 11 different habitat types and the three suitability level areas for each species).

Six different scenarios with different costs and targets were run in Marxan (Table 7.1). Although there are many human activities in the region, the one with the most significant and direct impact on marine resources in the LSMP area is commercial fishing, followed by recreational fishing and scuba diving (Gonçalves, 2005). Data on fishing effort (boat based commercial and recreational fishing) was based on the results of Gonçalves (2005). The most important scuba diving locations were also taken in consideration. This data was obtained from Rodrigues (2008) and consists of a 100m buffer around the 25 most dived locations based on the number of divers in each location for four diving tour operators. Two different cost layers were considered, 1) commercial fishing only and 2) combined cost of commercial fishing, recreational fishing and scuba diving. Data on fishing effort and diving locations from a period before the implementation of the LSMP were used in order to compare with the situation when the LSMP was implemented.

Three different targets were chosen: 1) targets that were the same as the protection offered to each conservation feature by the implemented no-take area, 2) 15% of the total available area in the LSMP for each of the habitats and 5%, 15% and 30% of each species unsuitable, suitable and highly suitable areas for each species respectively and 3) 30% of the total available area in the LSMP for each of the habitats and 10%, 30% and 50% of each species unsuitable, suitable and highly suitable areas respectively (Table 7.1).

Table 7.1. Targeted protection and costs used for each scenario run in Marxan.

Scenario	Cost	Target protection
1	Commercial fisheries	Existing protection
2	Commercial and recreational fisheries plus diving	Existing protection
3	Commercial fisheries	15% habitats Spp 5%; 15%; 30%
4	Commercial and recreational fisheries plus diving	15% habitats Spp 5%; 15%; 30%
5	Commercial fisheries	30% habitats Spp 10%; 30%; 50%
6	Commercial and recreational fisheries plus diving	30% habitats Spp 10%; 30%; 50%

For each scenario, 100 Marxan models were run with $1x10^9$ iterations using the Zonae Cogito interface (Segan et al. 2011). The boundary length modifier (BLM) option was used to ensure the solutions comprised a compact set of no-take areas. Although a compact network required protecting a greater total area to meet our targets, the resulting no-take areas are more likely to be successful than a highly fragmented and dispersed network. The BLM was fine tuned following the procedures described in Game and Grantham (2008). The number of times a planning unit was selected in a run was estimated in order to identify the areas that were chosen more often, indicative of areas of higher importance.

Results

The results for scenarios 1 and 2, where the targets for all conservation features are the same as in the existing no-take area, show that the implemented no-take area is very close to the near optimal solution estimated by Marxan considering the conservation features used in this study (Figure 7.2). This was true when considering the combined costs of commercial and recreational fishing together with scuba diving but also when considering the costs of commercial fishing alone. When observing the number of times each planning unit was selected there were two highly selected areas located at opposite ends of the no-take area. Nevertheless, the most chosen planning units are inside the existing no-take area (Figure 7.3). For this reason, the amount of suitable habitats is very close to the results obtained for the existing no-take area (Table 7.2).

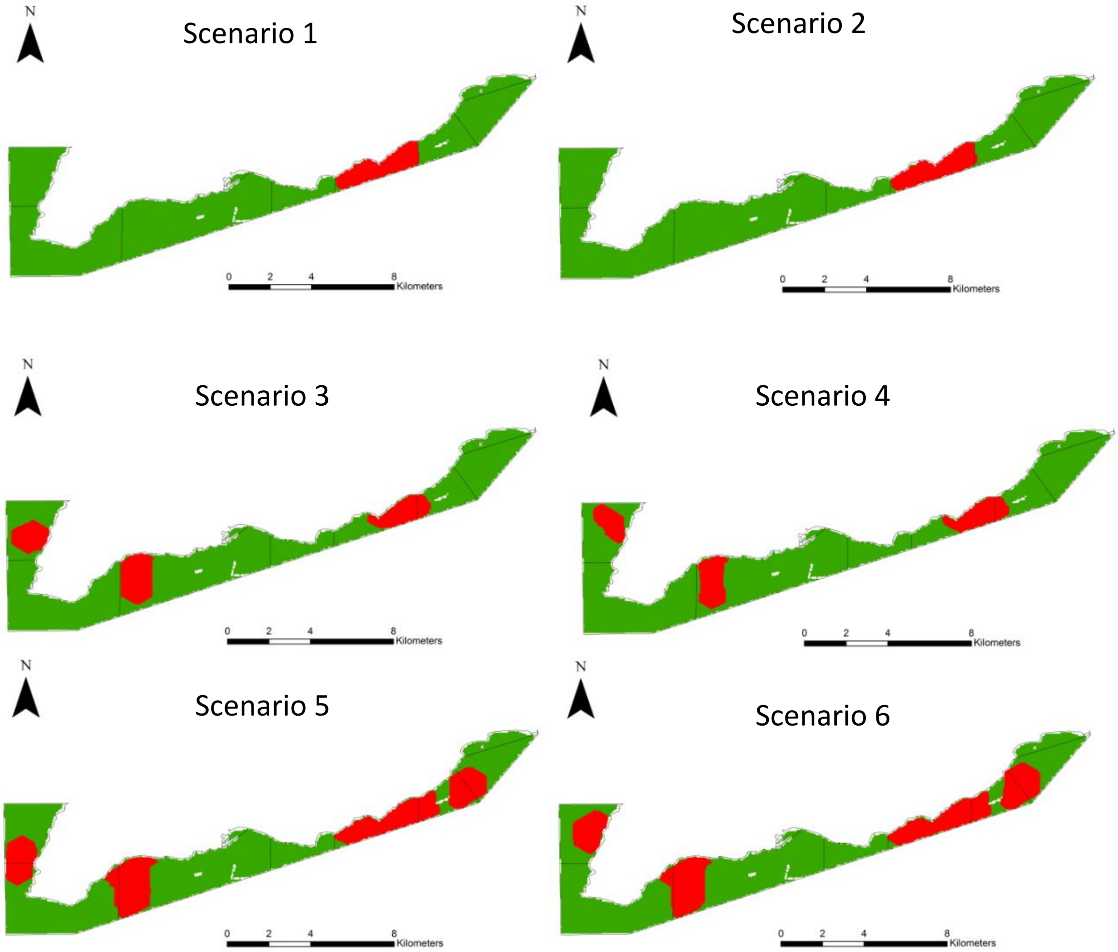


Figure 7.2. Results of the Marxan best solution for each scenario. Red areas indicate no-take areas.

With the targets set to 15% for each of the habitats and 5%, 15% and 30% of each species unsuitable, suitable and highly suitable areas respectively (scenarios 3 and 4) the solutions provided by Marxan include three distinct areas which comprise part of the existing no-take area but also areas in two complementary protection zones (Figure 7.2). Figure 7.3 shows the number of times each planning unit was selected in scenarios 3 and 4. There were no large differences in the amount of suitable habitat included in the no-take area between the existing no-take area and scenarios 3 and 4 (Table 7.2). However, in the latter, all habitats had at least 15% of their total area protected, which clearly contrasts with the protection provided by the existing no-take area where some habitats have less than 5% of the total available area protected

(Table 7.2). Scenarios 3 and 4 present three no-take areas with a total area close to twice the size of the implemented no-take area (Table 7.2).

For scenarios 5 and 6 (30% of each habitat and 10%, 30% and 50% of each species unsuitable, suitable and highly suitable) the best solutions included four no-take areas totalling an area almost 3.5 times larger than the existing no-take area (Figure 7.2). The summed solutions of the 100 Marxan runs for scenarios 5 and 6 show four large, highly selected areas including the whole already implemented no-take area (Figure 7.3).

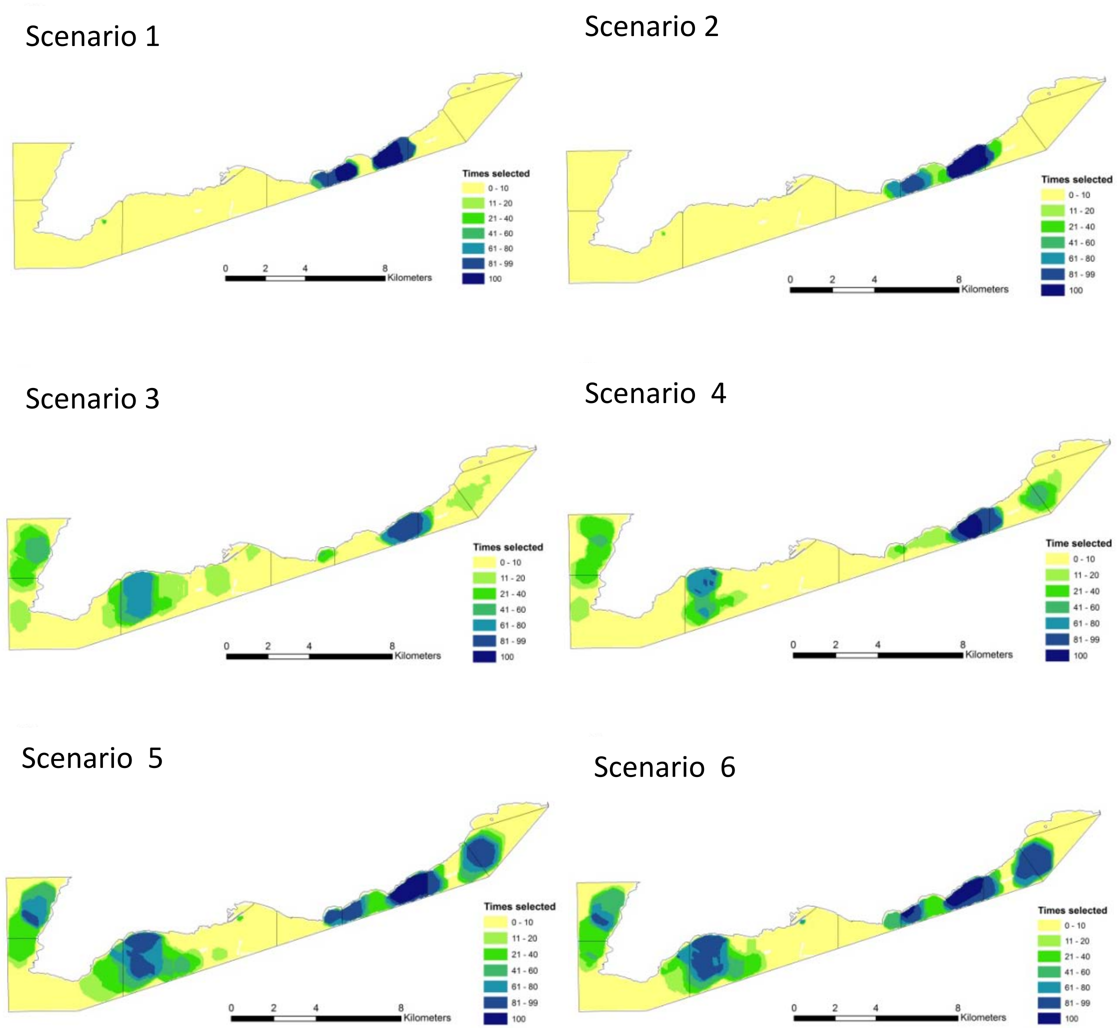


Figure 7.3. Map of the study area showing the number of times each planning unit was selected for each scenario.

Table 7.2. Area (km^2) of each conservation feature in the total MPA, existing no-take area and for each scenario. For the existing no-take area and each scenario, percentages of the total area available in the MPA are shown in parentheses.

Conservation feature	Total in MPA	Existing no-take area	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5	Scenario 6
Seabream HS	1.20	0.36 (30%)	0.35 (29%)	0.37 (31%)	0.36 (30%)	0.36 (30%)	0.6 (50%)	0.6 (50%)
Seabream S	3.50	0.66 (19%)	0.63 (18%)	0.66 (19%)	0.77 (22%)	0.67 (19%)	1.15 (33%)	1.28 (37%)
Seabream NS	44.88	2.88 (6%)	2.86 (6%)	2.83 (6%)	7.6 (17%)	6.4 (14%)	13.12 (29%)	13 (29%)
Cuttlefish HS	1.55	0.53 (34%)	0.59 (38%)	0.61 (40%)	0.46 (30%)	0.46 (30%)	0.77 (50%)	0.77 (50%)
Cuttlefish S	33.55	2.52 (8%)	2.35 (7%)	2.35 (7%)	6.36 (19%)	5.3 (16%)	10.66 (32%)	10.79 (32%)
Cuttlefish NS	14.48	0.84 (6%)	0.9 (6%)	0.9 (6%)	1.9 (13%)	1.67 (12%)	3.44 (24%)	3.31 (23%)
Sole HS	0.50	0.24 (48%)	0.31 (62%)	0.34 (68%)	0.35 (69%)	0.28 (55%)	0.33 (65%)	0.39 (78%)
Sole S	11.90	1.76 (15%)	1.67 (14%)	1.67 (14%)	3.03 (25%)	1.97 (17%)	3.98 (33%)	4.31 (36%)
Sole NS	37.18	1.89 (5%)	1.86 (5%)	1.86 (5%)	5.36 (14%)	5.19 (14%)	10.57 (28%)	10.17 (27%)
Mixed sands	4.66	0.88 (19%)	0.84 (18%)	0.84 (18%)	0.7 (15%)	0.7 (15%)	1.4 (30%)	1.4 (30%)
Nearshore reefs	0.84	0.14 (17%)	0.14 (17%)	0.14 (17%)	0.17 (20%)	0.13 (15%)	0.25 (30%)	0.25 (30%)
Algae on rock	2.89	0.04 (1%)	0.03 (1%)	0.03 (1%)	0.44 (15%)	0.43 (15%)	0.87 (30%)	0.87 (30%)
Medium sand	14.63	1.7 (12%)	1.61 (11%)	1.61 (11%)	3.35 (23%)	2.19 (15%)	4.39 (30%)	4.39 (30%)
Fine sand	2.48	0.3 (12%)	0.36 (14%)	0.36 (14%)	0.37 (15%)	0.37 (15%)	0.75 (30%)	0.75 (30%)
Rocky outcrops	1.88	0 (0%)	0 (0%)	0 (0%)	0.34 (18%)	0.28 (15%)	0.57 (30%)	0.56 (30%)

Coarse sand	14.85	0.39 (3%)	0.31 (2%)	0.34 (2%)	2.23 (15%)	2.23 (15%)	4.45 (30%)	4.46 (30%)
Muddy medium sand	0.79	0 (0%)	0 (0%)	0 (0%)	0.16 (21%)	0.12 (15%)	0.24 (30%)	0.24 (30%)
Muddy fine sand	1.93	0 (0%)	0 (0%)	0 (0%)	0.29 (15%)	0.29 (15%)	0.58 (30%)	0.59 (31%)
Mud to sandy mud	4.52	0.43 (10%)	0.53 (12%)	0.53 (12%)	0.68 (15%)	0.68 (15%)	1.36 (30%)	1.36 (30%)
Unknown	0.14	0 (1%)	0 (3%)	0 (2%)	0 (2%)	0.01 (4%)	0.03 (23%)	0.02 (12%)
No-take area	4.2	4.2 (1)	3.8 (1)	3.8 (1)	8.1 (3)	7.5 (3)	14.9 (4)	14.9 (4)

Discussion

This study focused on three of the most important fish species for the local artisanal fisheries operating in the vicinity of the LSMP. Given the limitations on data availability we used information on 11 different habitats identified in the LSMP and habitat suitability models for the three of the most important species as conservation features. Smith et al. (2009) recommend the use of habitat type as conservation features since it can act as a surrogate for benthic biodiversity. These authors also recognize that species distributions models can be used as conservation features. Regarding the cost layers, and because the true cost of all human activities is still a long way from being quantified, we opted to use the best available spatial data on the location and intensity of the major human activities taking place in the LSMP.

Our results demonstrate that, if the targets of the no-take area were to be the ones achieved by the already implemented no-take area, then its current location is the best. This was true when considering the cost of commercial fishing alone but also when taking in consideration the combined costs of commercial and recreational fishing together with scuba diving. However, the protection achieved by the implemented no-take area is not enough to effectively protect cuttlefish (Abecasis et al., 2013) and it also covers small amounts of largely available habitats such as rocky outcrops (<1%), coarse sand (<5%) or muddy fine sand (<3%). Although these habitats are not the most relevant for the three studied species, there are other commercial species for which such habitats are important (e.g. *Solea solea*, *Microchirus azevia*, *Chelidonichthys* spp.).

In contrast, the results for scenarios 3 and 4, with targets set to 15% of each habitat and 5%, 10% and 30% for species different suitability levels, show a total no-take area with almost twice the size of the existing no-take area. Nevertheless, because the best solution includes three separate no-take areas, their effectiveness might be limited since their individual area is less than 3.4 km^2 which provides little protection from fisheries (see Chapter 6). Yet, from a fisheries management perspective, these scenarios could provide a better solution since they should allow higher amounts of spillover while still offering some protection (Halpern & Warner 2003, Gaines et al. 2010, Fenberg et al. 2012). Additionally, in these solutions (scenarios 3 and 4) all identified habitats have at least 15% of the area available in the LSMP set as no-take.

The results obtained for scenario 5 and 6 (with targets set to 30% of each habitat and 5%, 10% and 30% for species different suitability levels) also contemplate several no-take areas, but because these areas are larger (two areas have more than 4.5km^2) they offer higher protection to all studied species. In addition, all habitats have at least 30% of the total available area set aside as no-take, ensuring a high level of protection. In fact, the results of the previous chapter suggest that these scenarios would provide a much lower vulnerability to fishing for all three species throughout the LSMP.

Our options for target levels were based on the literature review which suggests targets between 15% and 30% (Allnutt et al. 2012, Fraschetti et al. 2009, Green et al. 2009). However, some authors claim that, in the case of severely overfished or highly mobile species, no-take areas may not be effective as a primary management tool unless much more than 20% is closed (e.g. Murawski et al. 2000, Sale et al. 2005). Nevertheless, we suggest that target values should be reviewed for each species based on monitoring data for each particular MPA, particularly when they are the target of MPAs protection. As seen in the previous chapter, the amount of protection offered by a no-take area can vary substantially between species.

In this study we used Marxan to obtain near optimal solutions regarding the design, size and location of no-take areas for different scenarios considering different costs and conservation targets. The solutions presented in this study suggest that the performance of the LSMP could be improved with some changes in its design. These different solutions are useful for the adaptive management of the LSMP, and to the wider significance in the context of the science of MPAs. Ecosystem based conservation management, especially when considering fisheries management, must follow an adaptive management process where changes should be introduced based on the evaluation and monitoring processes of the MPA (Grafton and Kompas 2005, Hilborn et al. 2004).

The conservation planning process should be an adaptive process where the location and size of protected areas change in accordance to new constraints and availability of better data (Smith et al. 2009). Therefore, the results provided by the different scenarios analyzed, even though based on limited data, provide stakeholders with several solutions. This will allow for different approaches regarding targets of conservation features and the different costs considered. Information on other conservation features such as other species suitable/unsuitable areas and additional costs can be added to this framework in order to make the adaptive management an evolving and continuous process.

Even though the actual LSMP design only contemplates one no-take area, the enforcement is very poor and transgressions are frequent. Therefore, a solution which includes increasing the number of no-take areas would necessarily imply an increase in enforcement in order for these areas to function as true no-take areas.

One other issue would be the loss of fishing grounds for the local artisanal fisheries. Fishermen would not only loose access to a larger area, especially when comparing scenarios 5 and 6 with the actual design, but there would also be an increasing competition for the best fishing grounds. This could eventually lead to a decrease in the number of fishing boats as observed after the implementation of the LSMP (Horta e Costa et al. 2013). Yet, it is worth noting that when comparing the areas set as no-take areas in this study with the fishing hotspots found by Horta e Costa et al. (2013) there is only a small overlap with the areas presently used by fishermen. This was possible because the cost to fisheries was included in the models and therefore the most used fishing grounds would still be available in most cases.

Besides Marxan, other options can be used for systematic conservation planning, such as Marxan with zones or Zonation among others. We chose Marxan for its efficiency, flexibility and easy integration with GIS. However, it should be noted that Marxan does not tell which conservation features are important and how much area should be protected. In addition, Marxan only allows one single cost layer and its solutions do not ensure the resilience of species. The use of Marxan is not straightforward, as the process of preparing input files and learning its proper use takes time. Therefore, other approaches should be investigated and compared in the future.

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Chapter 8: General conclusions

Chapter 8

General conclusions

The main objective of this study was to evaluate the potential benefits of the Luiz Saldanha Marine Park (LSMP) for three species that are particularly important for the local small scale artisanal fisheries: the white seabream, the Senegalese sole and the cuttlefish. Given that this study started five years after the time of the MPA implementation, we also investigated whether any changes on species abundance or biomass have occurred within the limits of this the MPA.

The specific questions we tried to answer were:

> What is the size of cuttlefish, Senegalese sole and white seabream home range areas?

The home range areas (95% KUD) varied between 0.43 and 1.56 km^2 (average of 0.77 km^2) for the white seabream and between 0.54 and 2.22 km^2 (average of 1.19 km^2) for the Senegalese sole. The results obtained for cuttlefish were not coherent given the low site fidelity that this species presented.

> Do these species present site fidelity to the marine reserve?

High site fidelity was observed for the white seabream, as most individuals tagged with V9 acoustic transmitters were detected over 50% of the time (average of 70%). Some Senegalese sole also presented high site fidelity, yet the majority of the tagged individuals were more transient. The cuttlefish demonstrated no site fidelity to the study area, with only one individual spending a maximum of 39 days inside the monitored area.

> Do these species favour any particular habitats within the study area?

The white seabream demonstrated a clear preference for rocky bottoms habitats, while the Senegalese sole prefers sandy bottoms, especially medium coarse sand. A combination of short monitoring periods and large movement patterns invalidated the analysis of habitat preferences for the cuttlefish.

> Is it possible to already detect the effects of the marine reserve implementation on the abundance and/or biomass of Senegalese sole or cuttlefish?

The results of the experimental fishing trials revealed no significant differences in the abundance or biomass of Senegalese sole or cuttlefish between the periods before and after the implementation of the LSMP.

> Is the actual design of the LSMP appropriate for the protection/management of the three selected species?

Our results suggest that the existing MPA may offer appropriate protection for the Senegalese sole and the white seabream. However, it does not provide adequate protection for the cuttlefish, given this species' low site fidelity and large movements.

> Would changes in the size and location of no-take areas improve the efficiency of the LSMP?

According to the results of the different scenarios obtained with Marxan, the LSMP design could be improved in order to offer more protection to habitats that are only marginally protected with the current design and/or increase the amount of protection offered to the selected species.

Implications for the Luiz Saldanha Marine Park

The results of this study provide appropriate and useful information that can be used towards the adaptive management of coastal MPAs and the LSMP in particular. As in nearly all MPAs, the need for a multispecies based management is stressed given the multispecific nature of the artisanal fisheries operating within the LSMP and its adjacent waters. Furthermore, this is an improvement to implement the ecosystem approach to fisheries management because it adds to the effects of MPAs, which are already multispecific and species dependent (Claudet et al. 2010), an explicit multispecific consideration into its design.

While one should recognise that an MPA based on three species is unlikely to meet the requirements of all species occurring in the region, we also emphasize that the three species were selected for their contrasting characteristics (life history, economic value, etc) in order to capture as much ecological variability (hence variability effects) as possible. In addition, these species are economically important throughout the Mediterranean and ecologically similar to target species of other commercial fisheries in temperate coastal areas elsewhere. Furthermore, the results obtained in this study are representative of a wide variety of species that are important to sustain the local artisanal fisheries and fishers' livelihoods, which is one of this MPA's objectives.

For the white seabream and the Senegalese sole, the observed values of site fidelity and size of home range areas suggest that the LSMP offers an adequate level of protection to these species. In contrast, the LSMP offers little protection to the cuttlefish, given that this species shows no site fidelity and performs movements of up to more than 15 km. For migratory species such as the cuttlefish, management in the coastal zone alone has been shown to be insufficient in ensuring sustainable exploitation (Royer et al. 2006). In fact, Shipp (2003) has suggested that when species are too mobile to remain within the limits of an MPA, traditional fishery management measures such as gear restrictions, seasonal closures, or catch quotas are more effective than MPAs.

This study provided a suite of ecological information, such as habitat use, home range areas, site fidelity estimates and, especially, measures of vulnerability to fishing mortality inside the LSMP for the three contrasting species. This information is of extreme importance in understanding the implications of the LSMP for the management of these species, and may be useful for the development of an adequate MPA management plan with a design and regulations tailored to their characteristics. On the downside, this study did not provide evidence of detectable ecological effects caused by the LSMP implementation. This is most likely to be a result of the short period of time in which the LSMP's regulations have been fully implemented, given that the ecological effects of an MPA are known to be highly dependent on the age of the MPA (Claudet et al. 2008, Vandeperre et al. 2011, Leleu et al. 2012).

In the event more no-take areas are implemented, as suggested by some of the scenarios analyzed in chapter 7, the compliance could be harder to achieve given the already low enforcement observed in the only no-take area implemented. The increasing competition for fishing grounds could lead to a decrease in the number of fishing boat as observed after the implementation of the LSMP (Horta e Costa et al 2013).

General implications for MPAs

MPAs can provide both conservation and fisheries benefits, yet they should not be considered a panacea. In order to achieve sustainable fisheries and to protect biodiversity and habitats, fisheries management must include other tools such as the prohibition of damaging gears (e.g. trawl), quotas and the reduction of bycatch (Roberts et al. 2005). In fact, several authors (Shipp 2003, Roberts et al. 2005, Tetreault & Ambrose 2007) regard MPAs as a supplement and not a substitute for traditional fishery management tools. According to Roberts et al. (2005) only when combining MPAs with traditional fishery management tools will conservation and fishery goals become united.

Nevertheless, in order for MPAs to function as fishery management tools, there needs to be an examination of specific instances and specific stocks to determine potential benefits (Shipp 2003). Therefore, the objectives of MPAs should be clearly stated and the species and habitats they intend to protect should be identified early in the MPA establishment process.

Natural variability in marine ecosystems is huge and difficult to predict. This makes it even more difficult to understand the effectiveness of MPAs. This is why, to this date, there is no consensus on a framework to evaluate and manage MPAs. The framework presented in this study is widely applicable to coastal MPAs, and provides managers and stakeholders with relevant and useful information not only for the implementation phase of MPAs (establishing their size and location) but also for the adaptive management process of existing MPAs. This is a crucial aspect to improve the performance of the many coastal MPAs that represent a large proportion of the more than 5880 MPAs established throughout the world as of 2010 (Spalding et al. 2010).

Future studies and recommendations

This study focused on three of the most important species for the local small scale artisanal fisheries, yet other locally important species such as *Solea solea*, *Octopus vulgaris* and *Raja* spp. should not be overlooked, as they are also important resources for the local fisheries and information on their site fidelity and home range areas is missing. As in the present study, gathering this type of information for those species will provide extremely valuable inputs for the adaptive management of MPAs, such as distribution models and vulnerability to fishing.

One of the limitations of this study is the fact that the telemetry studies did not cover the entire range of available habitats. Not only did we have a limited number of acoustic receivers but detailed information on habitat was only available towards the end of the acoustic telemetry campaigns, preventing an optimal design of the array. The increase of the monitored area in future studies using acoustic telemetry would allow increased habitat coverage and help to achieve longer term studies on the movements and activity patterns of fishes. In the particular case of cuttlefish, given their high mobility and migrations to the nearby estuary, it would also be interesting to monitor the Sado estuary. A larger array would also benefit acoustic telemetry studies on Senegalese sole because, even though some individuals presented high site fidelity, most were undetected for large periods of time. The fact that some of them eventually returned to the study area suggests that they may not displace far. Therefore, increasing the size of the monitored area would, most probably, allow for longer periods of detection.

Monitoring MPAs should be a continuous process in order to provide managers with information on the performance of MPAs and allow for adequate adaptive management actions. Hence, we propose that experimental fishing and visual census should be carried at least twice a year in order to properly evaluate the LSMP and to provide managers with information on its benefits. In the future, studies focusing on the fisheries of the adjacent areas should also be carried out in order to clarify the role of the LSMP as a fishery management tool.

The poor enforcement observed in this particular MPA might turn the initial objectives of fisheries management and biodiversity management harder to reach. During the course of this study, infringements of the regulations were observed on several occasions. Among these were spearfishing, the use of octopus traps too close to the shore and the use of fishing nets and longlines in partial and no-take areas. Without enforcement and compliance, an MPA is just a paper park and no protection effects or benefits should be expected (Wood et al. 2008). Actual enforcement and compliance, and not the formal MPA establishment, must be considered as the true starting point of protection (Guidetti et al. 2008).

Effective MPAs should include high-quality spawning and nursery habitats (Glazer & Delgado 2006). For the species on which this study focuses such habitats are mainly found in coastal lagoons and estuaries such as the nearby Sado estuary. Therefore, efforts should be put in place to ensure the quality and protection of these essential fish habitats. Ideally, migration corridors should be included between important juvenile and adult habitats such as the LSMP. This might implicate the identification of those habitats outside the MPA. If these migration paths are not protected, individuals may be susceptible to harvest when in transit or they may be unable to migrate into critical habitats (Simpson & Mapleston 2002). Given the location of the nearby Sado estuary, an area known to act as nursery/spawning area for several species, the establishment of protected corridors between these areas and the LSMP should be investigated. In addition, studies on the movements' of earlier life stages (larvae, post-settlement) of the species would help to better understand their dispersion and the potential of the LSMP as source for seeding fished areas.

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Annexes

Range tests and detection efficiency

In order to test the diel detection patterns we set an experiment prior to the release of any tagged animals. For this, we deployed three acoustic receivers and one acoustic transmitter. This experiment lasted for 25 days during which there was a large variation in sea state, rainfall and wind speed. Daily rainfall and daily average wind speed based on forecast models were obtained from www.windguru.cz.

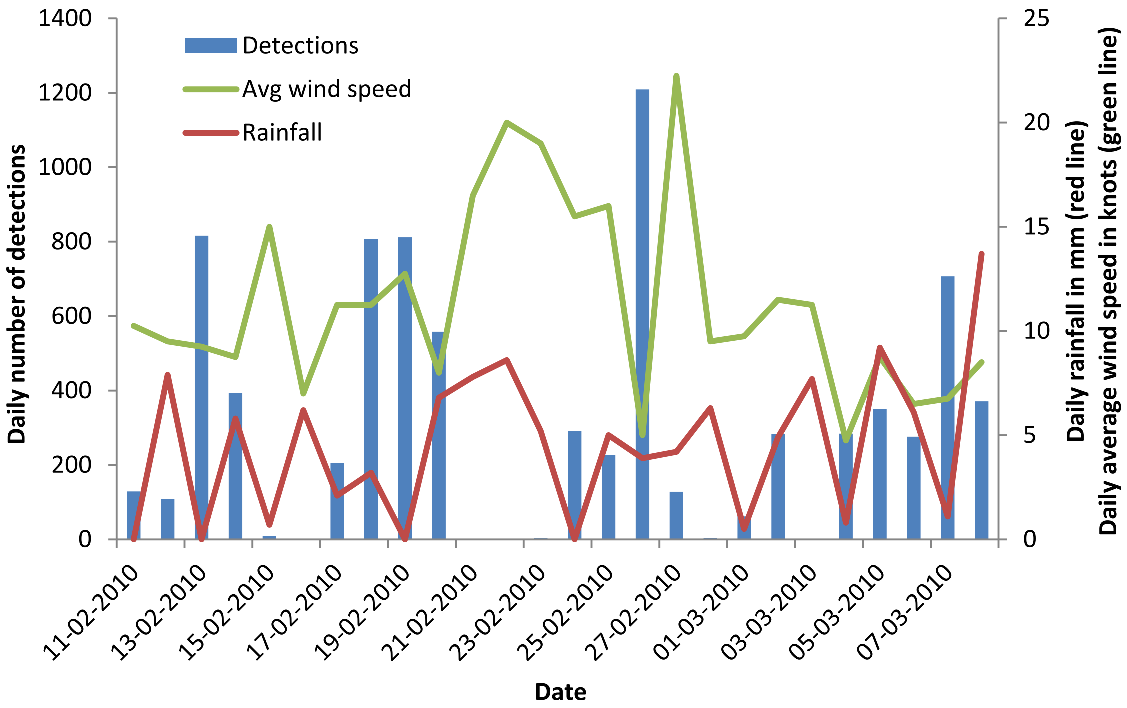

Figure A.1. Number of daily detections, daily rainfall and average daily wind speed during the tag detection experiments.

Results suggest that rainfall and especially wind speed have a great influence in the detection efficiency of the receivers. Other factors that may also influence the detection efficiency are the wave height and direction and wind direction however, these variables could not be accurately investigated.

Distance from transmitter	Code detection efficiency	Rejection coefficient
100 m	0.78	0.007
250 m	0.54	0.010
320 m	0.25	0.004

Table A.1. Code detection efficiency and rejection coefficient of acoustic receivers placed at different distances from the acoustic transmitter.

We determined two metrics of receiver performance, the code detection efficiency and the rejection coefficient. The results indicate that the detection efficiency decreases with the distance to the tag. The rejection coefficients obtained were low indicating that only a very small proportion of codes received were rejected.

To test for differences between the number of detections during the day and during the night time we used data from 24h periods with similar wind, rainfall and sea state conditions throughout. The results of the t-test reveal no significant differences between day and night time detections (t-test = 1.112, DF=22; P=0.278).

To evaluate the detection range of the acoustic tags we placed an acoustic transmitter at several known distances from the acoustic receivers for a period of one hour. We then estimated the percentage of the detections that were effectively detected by each receiver.

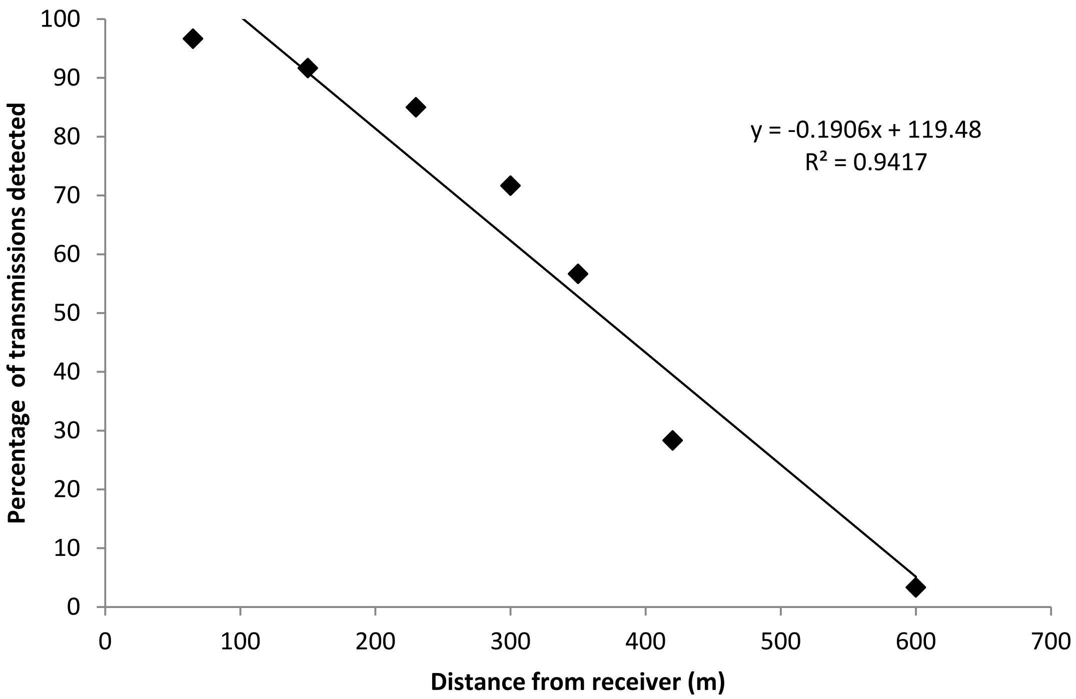

Figure A.2. Percentage of detected transmissions at different distances.

Figure A.3. Cuttlefish with an acoustic transmitter attached to the cuttlebone.

Figure A.4. Senegalese sole tagged with an acoustic transmitter.

Figure A.5. Senegalese sole tagged with an acoustic transmitter recaptured more than 300 days after release. The good condition of the fish provided evidence that the method is adequate.

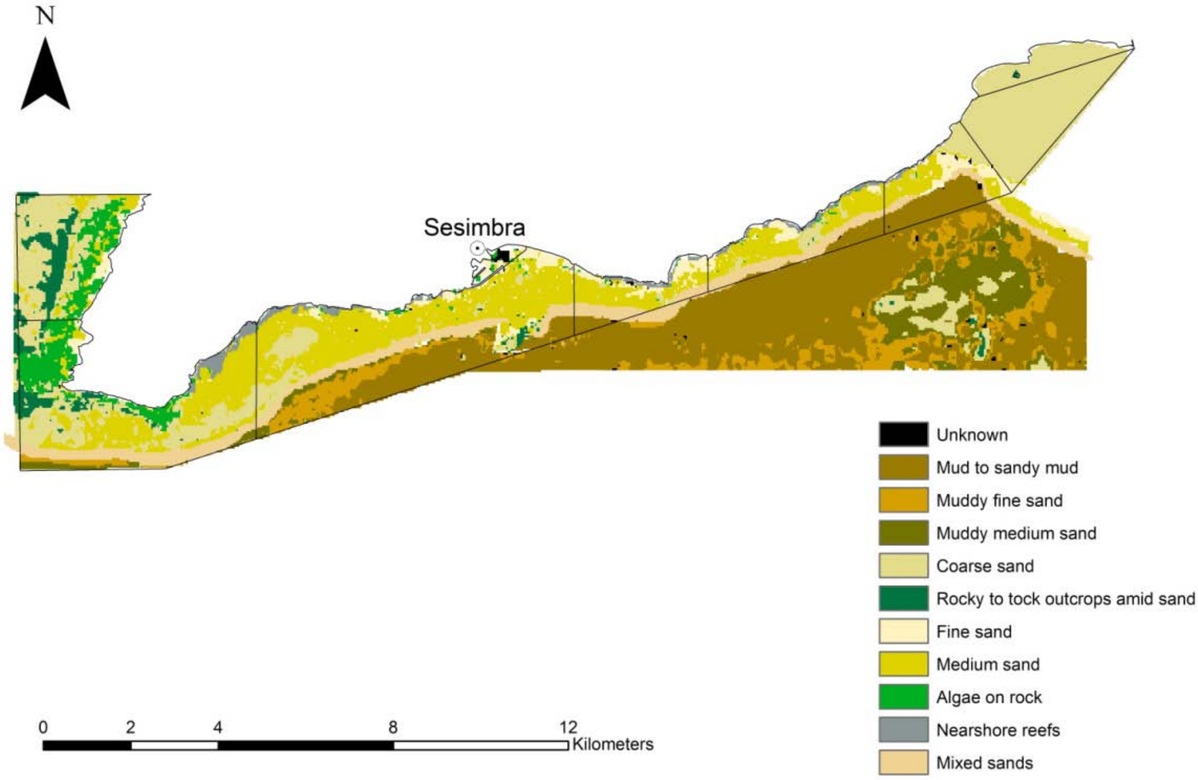

Figure A.6. Bottom substrate type based on the information provided by IPIMAR for the study area. The black line delimits the Arrábida Marine Park boundaries including the different protection areas.

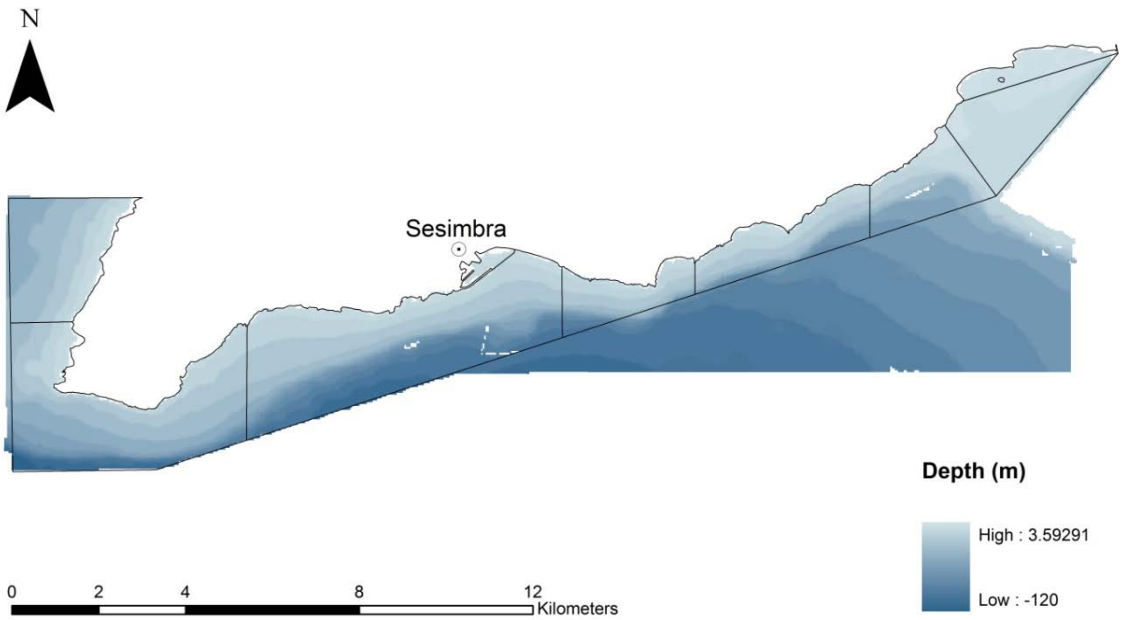

Figure A.7. Bathymetry map (m) derived from the BIOMARES project bathymetry datasets.

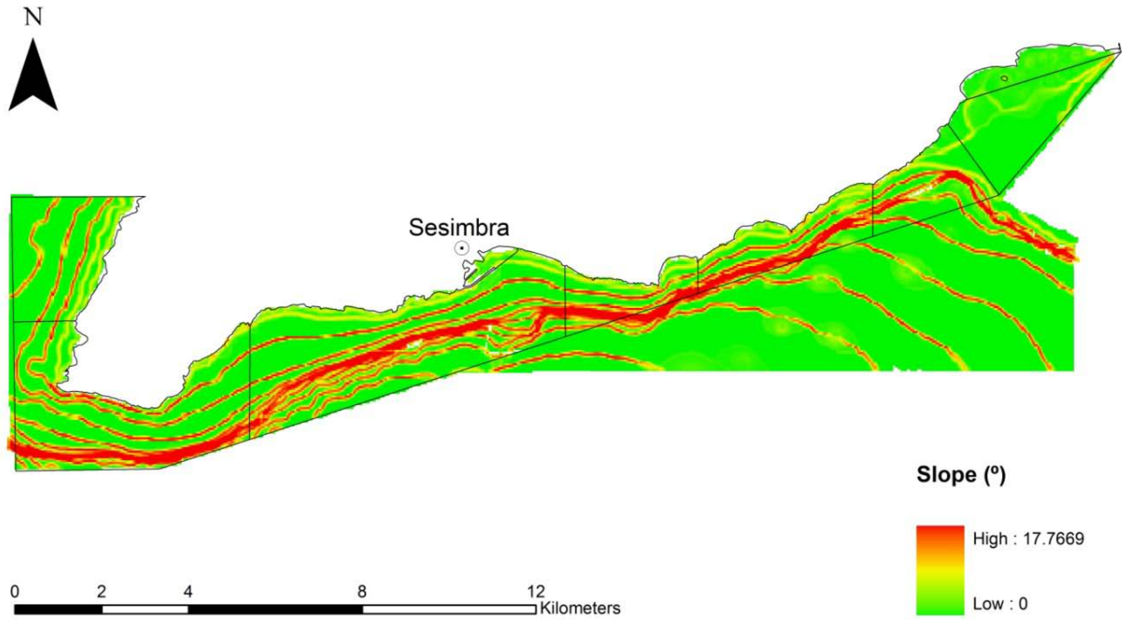

Figure A. 8. Slope map based on the bathymetry raster estimated using the BIOMARES project bathymetry datasets.

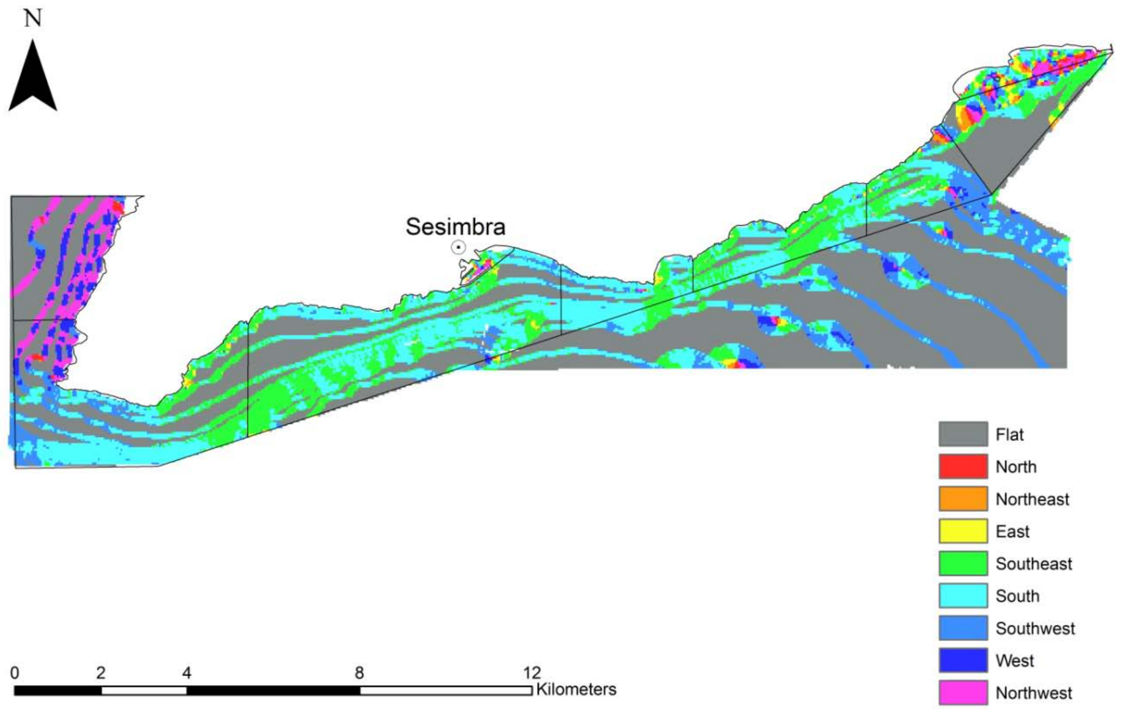

Figure A.9. Aspect (slope direction) map for the study area based on the bathymetry raster using the BIOMARES project bathymetry datasets.

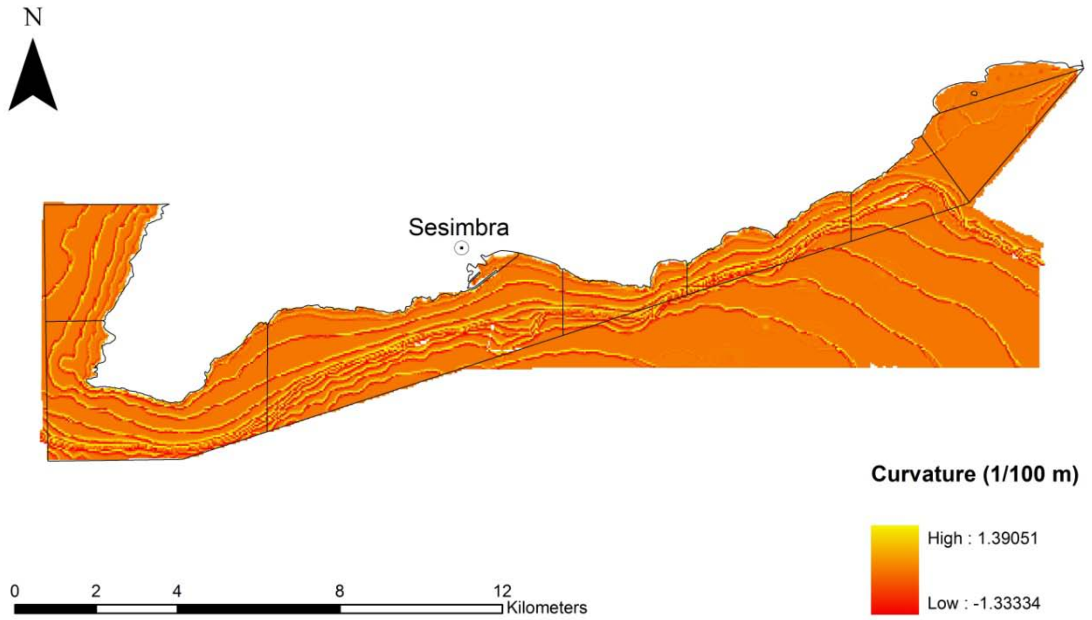

Figure A.10. Curvature map of the study area based on the bathymetry raster estimated using the BIOMARES project bathymetry datasets. Positive values indicate concave surfaces while negative values indicate convex surfaces.

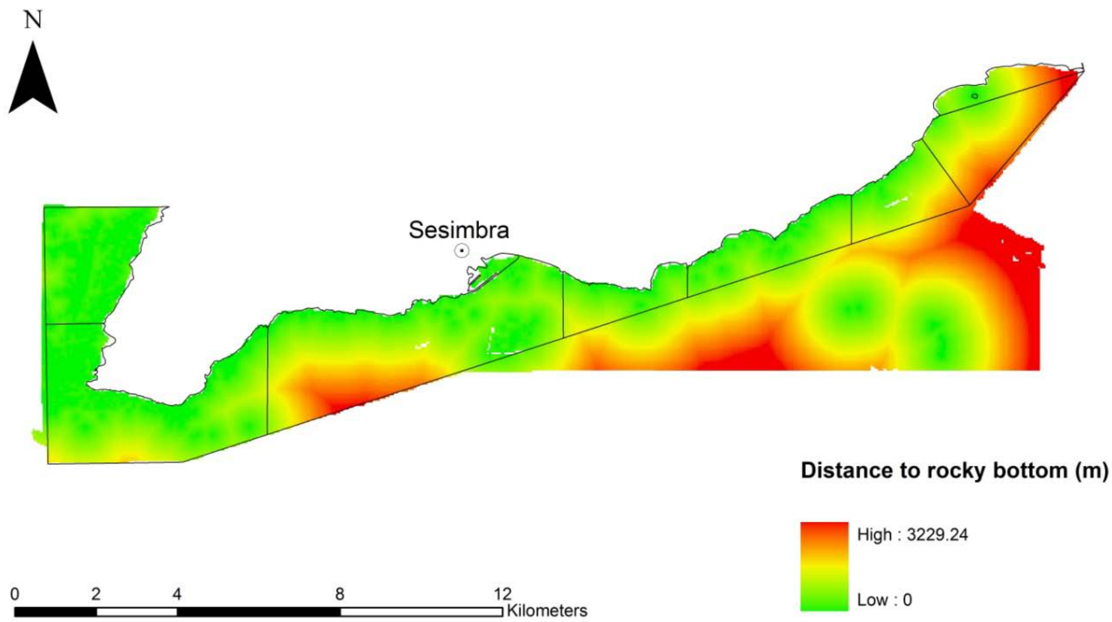

Figure A.11. Distance to nearest rock in the study area based on the information on bottom type provided by IPIMAR.

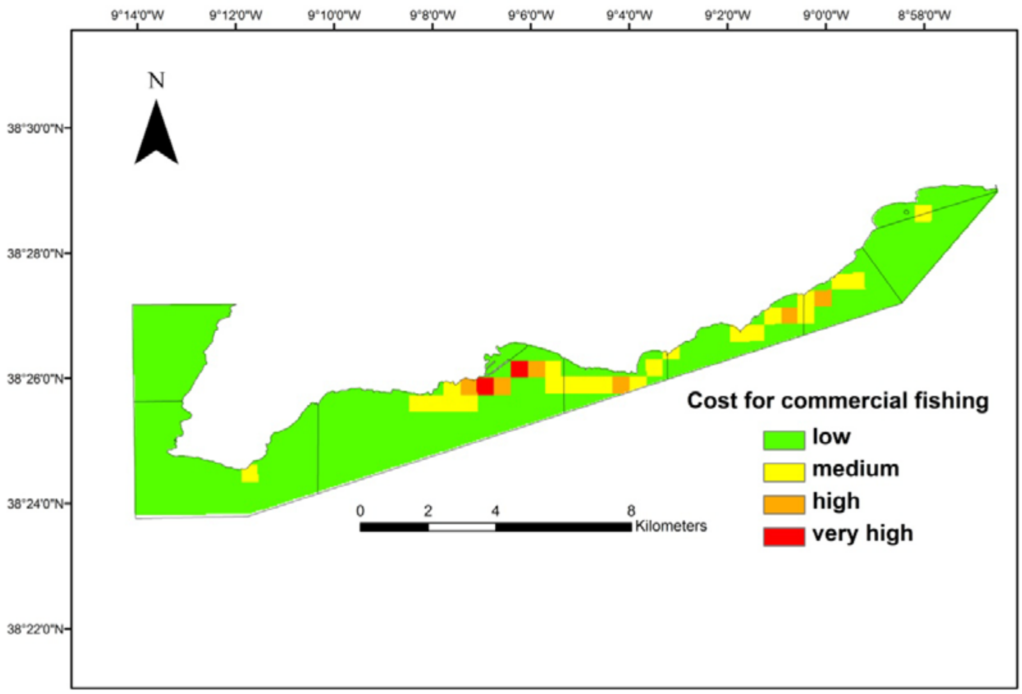

Figure A.12. Map with the cost for commercial fishing boats (previous to the implementation of the LSMP). See Gonçalves 2005 for more details.

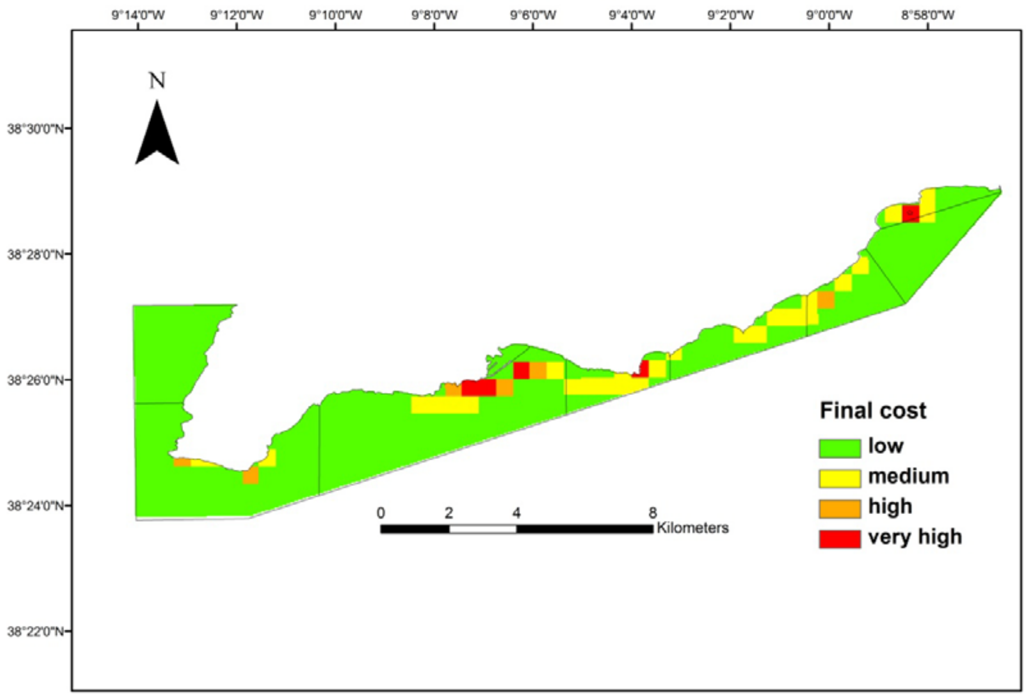

Figure A.13. Map with the combined cost of commercial fishing, recreational fishing and scuba diving (previous to the implementation of the LSMP). See Gonçalves (2005) and Rodrigues (2008) for more details.