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Original Article

Marine conservation of multispecies and multi-use areas with various conservation objectives and targets

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Marine spatial management is an important step in regulating the sustainable use of marine resources and preserving habitats and species. The systematic conservation planning software "Marxan" was used to analyse the effect of different conservation objectives and targets on the design of a network of marine protected areas around two islands of the Azores archipelago, Northeast Atlantic. The analyses integrated spatial patterns of the abundance and reproductive potential of multispecies, the vulnerability of fish to fishing, habitat type, algae biotopes, and socio-economic costs and benefits (including fishing effort and recreational activities). Three scenarios focused on fisheries-related objectives ("fisheries scenarios", FSs) and three on multiple-use and biodiversity conservation objectives ("biodiversity scenarios", BSs), respectively. Three different protection targets were compared for each set, the existing, minimum, and maximum levels of protection, whereas conservation features were weighted according to their biologically/ecologically functioning. Results provided contrasting solutions for site selection and identified potential gaps in the existing design. The influence of the conservation objective on site selection was most evident when minimum target levels were applied. Otherwise, solutions for FSs and BSs were very similar and mostly shaped by the protection level. More important, BSs that considered opportunity cost and benefits achieved conservation targets more cost-efficiently. The presented systematic approach ensures that targets for habitats with high fish abundance, fecundity, and vulnerability are achieved efficiently. It should be of high applicability for adaptive management processes to improve the effectiveness of existing spatial management practices, in particular when fishing and leisure activities coexist, and suggest that decision-makers should account for multiple users' costs and benefits when designing and implementing marine reserve networks.

Keywords: biodiversity surrogates, fisheries management, marine spatial management, Marxan, socio-economic data, systematic conservation planning.

Introduction

There is growing awareness that the excessive human use of marine resources has contributed to significant changes in the marine ecosystem. Decreasing biodiversity, declining fish stocks, and habitat loss are among the most visible consequences (Pauly *et al.*, 2005; Sala and Knowlton, 2006; Coll *et al.*, 2010). Marine protected areas (MPAs) can be implemented to compensate for these negative impacts and as a tool to maintain biodiversity, recover fish stocks, and improve habitat quality (Russ, 2002; Gladstone, 2007; Palumbi *et al.*, 2009). Their classification varies from the regulation of particular human activities or the protection of single species to the prohibition of any human activity (IUCN, 1994). Marine reserves (MRs), the strictest approach, are considered as most effective for the maintenance of biodiversity and as fisheries management tool (Russ, 2002; Vandeperre *et al.*, 2011; Sala *et al.*, 2013). Protected fish populations are believed to recover from overexploitation and, subsequently, support adjacent fisheries via larval, juvenile, and adult spillover (Goñi *et al.*, 2008; Stobart *et al.*, 2009; Harrison *et al.*, 2012). Studies have shown that a network of several protected zones is the most effective (Gaines *et al.*, 2010), if the individual sites are viable/adequate, representative, replicated, and connected with each other (OSPAR, 2007; Ardron, 2008).

Despite an increasing number of studies on reserve size, spacing, configuration (Kaplan and Botsford, 2005; Little *et al.*, 2005;

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Vandeperre *et al.*, 2011), ontogenetic fish migrations (Afonso *et al.*, 2008a; Compton *et al.*, 2012), and larval dispersal (Fontes *et al.*, 2009; Harrison *et al.*, 2012), the optimal reserve design remains challenging (Gaines *et al.*, 2010; Agardy *et al.*, 2011). A multitude of environmental, biological, ecological, and management aspects has to be considered. Design tasks are substantially supported by the increasing availability and quality of information of species distribution and habitat maps, including predictive distribution models in data-limited situations (e.g. Guisan and Zimmermann, 2000; Schmiing *et al.*, 2013).

Systematic conservation planning is a structured approach to locate, design, and manage MRs (or other conservation areas) that represent, protect, and promote the persistence of biodiversity (Margules and Pressey, 2000). It is an efficient, flexible, defendable, and transparent process that uses the principle of complementary (i.e. priority areas achieve conservation objectives collectively; Margules and Pressey, 2000; Ardron 2008; Watson *et al.*, 2011). Systematic conservation planning involves the identification of clear objectives, the compilation of suitable data, review of existing and selection of additional conservation areas, as well as their implementation and monitoring (Margules and Pressey, 2000; Ardron 2008). Site selection is supported by decision support software that achieves conservation objectives in a cost-efficient manner; biological/ecological, habitat, and socio-economic data are integrated to produce near-optimal solutions for reserve networks (Ardron 2008; Game and Grantham, 2008).

In the present study, systematic conservation planning is used to assess an existing, partly opportunistically designed MPA network, and to generate solutions for alternative networks in two neighbouring islands of the Azores archipelago, Northeast Atlantic. This case study addresses the challenges of marine spatial management in spatially limited and isolated coastal habitats (Santos *et al.*, 1995) that are extensively used by local fisheries and the leisure industry (Diogo and Pereira, 2013, 2014; Pham *et al.*, 2013; Ressurreição and Giacomello, 2013). A set of solutions for reserve networks with either fisheries management or multiple-use (including non-extractive, recreational activities) and biodiversity conservation objectives is produced systematically by integrating (i) spatial patterns of abundances, fecundity, and spawning biomass of multi-species, (ii) the vulnerability of fish to fishing, (iii) different habitats and algae biotopes, (iv) the distribution of socio-economic values, and (v) well-defined conservation objectives and targets. The presented approach supports the planning of conservation networks in any marine ecosystem, particularly when leisure and fishing activities coexist.

Material and methods

Study area

The study area comprises coastal habitats down to the 40 m isobath around Faial and Pico Islands, archipelago of the Azores, Northeast Atlantic (Figure 1). It includes three small offshore reefs that lay inside the shallow channel between the two islands, the "Faial–Pico channel". Island Natural Parks (INPs) of both islands were declared in 2008, including marine and terrestrial sites that follow the classification of the International Union for Conservation of Nature (IUCN). These areas integrate previously designated regional, national, NATURA2000, and OSPAR MPAs. The single "no-take" zone (IUCN category I) in the study area encompasses two adjacent small sunken calderas (8 ha), the Caldeirinhas MR in Faial Island, where only previously approved scientific activities

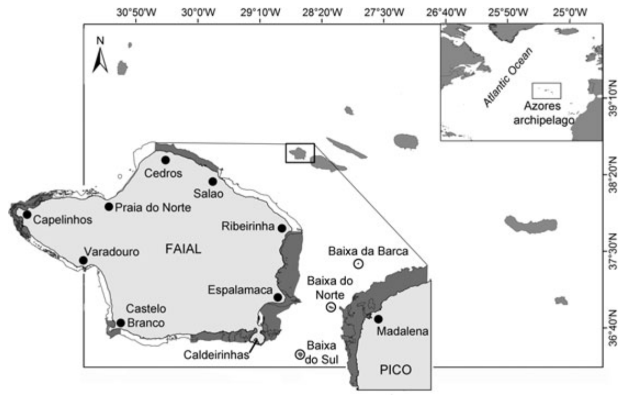

Figure 1. Map of the study area in the Azores archipelago (NE Atlantic), including three offshore reefs located in the channel between the islands of Faial and Pico. Dark grey shaded areas show the existing MPA network and black lines outline rocky habitats down to 40 m depth.

are permitted. Five other marine zones correspond to IUCN category VI but have no uniform legislation for human, extractive activities. For example, mineral exploration and extraction is conditional to authorization in just one of these zones, spearfishing is prohibited in three zones, and limpet collection in all five IUCN VI zones. Management plans are still missing for both INPs. Marine leisure activities occur in the entire study area (e.g. scuba diving, glass-bottom boat observation, whale watching, etc.).

Conservation features

Conservation features are species, habitats, or other ecological features of interest for conservation, which are given a conservation target (i.e. the amount to be included within a reserve network, Game and Grantham, 2008; Ball *et al.*, 2009). The spatial distribution of single species, the entire fish assemblage, substrate type, and biotopes were used as conservation features (Table 1). The distribution of "species features" was based on predictive species distribution maps of reef fish, derived from statistic modelling (generalized additive models, GAMs; Schmiing *et al.*, 2013, Schmiing, unpublished data). The models were based on data from underwater visual fish counts (UVCs) that identified and counted fish and grouped all individuals into four species-specific size classes (small, medium, large, very large, see Schmiing *et al.*, 2013).

Seven maps considered the relative abundance (i.e. the percentage of the maximum predicted abundance) of commercial (*Diplodus sargus*, *Labrus bergylta*, *Serranus atricauda*, *Sparisoma cretense*) and non-commercial fish species (*Abudefduf luridus*, *Coris julis*, *Sarpa salpa*), including all size classes (Schmiing *et al.*, 2013, unpublished data). Two additional maps described the probability of the presence of the non-commercial species *Symphodus caeruleus* and the commercial *Trachurus picturatus* (on a scale from 0 to 1). No sound statistic model could be produced for the abundance of these two species, thus count data were converted to presence–absence and binomial models were applied (Schmiing *et al.*, 2013).

In addition, the relative spawning biomass of mature individuals of each of the four commercial species was modelled and predicted (Schmiing, unpublished data). For this purpose, it was defined that the three largest size classes encompass mature individuals. Size classes were converted to the mean sizes in centimetres and, subsequently, the biomass was calculated based on species-specific

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Table 1. Overview of the spatial conservation features and their use in conservation scenarios.

Category	Conservation feature	Value	Scenario	Target: existing MPA (%)	Target: minimum (%)	Target: maximum (%)
Habitat	Rock	Presence	FS, BS	62 (21 km^2)	10	30
	Sediment	Presence	BS	55 (14 km^2)	10	30
Biotope	Articulated corallinaceae	Presence	BS	66 (19 km^2)	10	30
	Codium eslisabethae	Presence	BS	99 (0.03 km^2)	10	30
	Dictyota spp.	Presence	BS	63 (16 km^2)	10	30
	Halopteris filicina	Presence	BS	70 (12 km^2)	10	30
	Padina pavonica	Presence	BS	71 (8 km^2)	10	30
	Zonaria tournefortii	Presence	BS	70 (12 km^2)	10	30
Assemblage	IV, low	35–43	FS	41 (6 km^2)	5	15
	IV, medium	>43–47	FS	72 (16 km^2)	10	30
	IV, high	>47–51	FS	65 (12 km^2)	15	45
Species: abundance (all individuals)	*Abudefduf luridus*^{NC}, low	<10%	BS	60 (27 km^2)	0	0
	A. luridus^{NC}, middle	10–25%	BS	68 (6 km^2)	5	15
	A. luridus^{NC}, high	>25–50%	BS	84 (1 km^2)	10	30
	A. luridus^{NC}, very high	>50%	BS	97 (0.1 km^2)	15	45
	Coris julis^{NC}, low	<10%	BS	58 (17 km^2)	0	0
	C. julis^{NC}, middle	10–25%	BS	64 (6 km^2)	5	15
	C. julis^{NC}, high	>25–50%	BS	59 (6 km^2)	10	30
	C. julis^{NC}, very high	>50%	BS	82 (5 km^2)	15	45
	Diplodus sargus^{C}, low	<10%	FS, BS	58 (25 km^2)	0	0
	D. sargus^{C}, middle	10–25%	FS, BS	75 (7 km^2)	5	15
	D. sargus^{C}, high	>25–50%	FS, BS	73 (2 km^2)	10	30
	D. sargus^{C}, very high	>50%	FS, BS	98 (0.3 km^2)	15	45
	Labrus bergylta^{C}, low	<10%	FS, BS	63 (31 km^2)	0	0
	L. bergylta^{C}, middle	10–25%	FS, BS	52 (3 km^2)	5	15
	L. bergylta^{C}, high	>25–50%	FS, BS	49 (0.4 km^2)	10	30
	L. bergylta^{C}, very high	>50%	FS, BS	37 (0.1 km^2)	15	45
	Sarpa salpa^{NC}, low	<10%	BS	61 (25 km^2)	0	0
	S. salpa^{NC}, middle	10–25%	BS	70 (4 km^2)	5	15
	S. salpa^{NC}, high	>25–50%	BS	73 (3 km^2)	10	30
	S. salpa^{NC}, very high	>50%	BS	51 (3 km^2)	15	45
	Serranus atricauda^{C}, low	<10%	FS, BS	60 (16 km^2)	0	0
	S. atricauda^{C}, middle	10–25%	FS, BS	62 (8 km^2)	5	15
	S. atricauda^{C}, high	>25–50%	FS, BS	65 (7 km^2)	10	30
	S. atricauda^{C}, very high	>50%	FS, BS	70 (3 km^2)	15	45
	Sparisoma cretense^{C}, low	<10%	FS, BS	61 (31 km^2)	0	0
	S. cretense^{C}, middle	10–25%	FS, BS	69 (2 km^2)	5	15
	S. cretense^{C}, high	>25–50%	FS, BS	73 (1 km^2)	10	30
	S. cretense^{C}, very high	>50%	FS, BS	94 (0.1 km^2)	15	45
Species: probability of presence (all individuals)	*Symphodus caeruleus*^{NC}, low	<10%	BS	66 (19 km^2)	0	0
	S. caeruleus^{NC}, middle	10–25%	BS	54 (4 km^2)	5	15
	S. caeruleus^{NC}, high	>25–50%	BS	53 (3 km^2)	10	30
	S. caeruleus^{NC}, very high	>50%	BS	62 (8 km^2)	15	45
	Trachurus picturatus^{C}, low	<10%	FS, BS	57 (14 km^2)	0	0
	T. picturatus^{C}, middle	10–25%	FS, BS	59 (9 km^2)	5	15
	T. picturatus^{C}, high	>25–50%	FS, BS	72 (9 km^2)	10	30
	T. picturatus^{C}, very high	>50%	FS, BS	81 (3 km^2)	15	45
Species: spawning biomass (mature individuals)	*D. sargus*^{C}, low	<10%	FS	56 (19 km^2)	0	0
	D. sargus^{C}, middle	10–25%	FS	71 (9 km^2)	5	15
	D. sargus^{C}, high	>25–50%,	FS	74 (4 km^2)	10	30
	D. sargus^{C}, very high	>50%	FS	85 (2 km^2)	15	45
	S. atricauda^{C}, low	<10%	FS	62 (15 km^2)	0	0
	S. atricauda^{C}, middle	10–25%	FS	56 (10 km^2)	5	15
	S. atricauda^{C}, high	>25–50%	FS	66 (6 km^2)	10	30
	S. atricauda^{C}, very high	>50%	FS	79 (4 km^2)	15	45
Species: probability of presence of mature individuals	*L. bergylta*^{C}, low	<10%	FS	61 (15 km^2)	0	0
	L. bergylta^{C}, middle	10–25%	FS	69 (5 km^2)	5	15
	L. bergylta^{C}, high	>25–50%	FS	62 (7 km^2)	10	30
	L. bergylta^{C}, very high	>50%	FS	61 (8 km^2)	15	45
	S. cretense^{C}, low	<10%	FS	61 (14 km^2)	0	0
	S. cretense^{C}, middle	10–25%	FS	58 (2 km^2)	5	15

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Table 1. *Continued*

Category	Conservation feature	Value	Scenario	Target: existing MPA (%)	Target: minimum (%)	Target: maximum (%)
	S. cretense[C], high	>25–50%	FS	58 (4 km^2)	10	30
	S. cretense[C], very high	>50%	FS	66 (13 km^2)	15	45
Species: potential fecundity (mature females)	*D. sargus*[C], low	<10%	FS	56 (18 km^2)	0	0
	D. sargus[C], middle	10–25%	FS	62 (8 km^2)	5	15
	D. sargus[C], high	>25–50%	FS	77 (5 km^2)	10	30
	D. sargus[C], very high	>50%	FS	89 (3 km^2)	15	45
	L. bergylta[C], low	<10%	FS	61 (16 km^2)	0	0
	L. bergylta[C], middle	10–25%	FS	70 (4 km^2)	5	15
	L. bergylta[C], high	>25–50%	FS	63 (7 km^2)	10	30
	L. bergylta[C], very high	>50%	FS	59 (8 km^2)	15	45
	S. atricauda[C], low	<10%	FS	60 (20 km^2)	0	0
	S. atricauda[C], middle	10–25%	FS	58 (7 km^2)	5	15
	S. atricauda[C], high	>25–50%	FS	71 (4 km^2)	10	30
	S. atricauda[C], very high	>50%	FS	80 (3 km^2)	15	45
	S. cretense[C], low	<10%	FS	62 (31 km^2)	0	0
	S. cretense[C], middle	10–25%	FS	70 (3 km^2)	5	15
	S. cretense[C], high	>25–50%	FS	58 (1 km^2)	10	30
	S. cretense[C], very high	>50%	FS	79 (0.1 km^2)	15	45

Two objectives were analysed, FSs and BSs, each considering three different protection target levels. The area (km^2) of each conservation feature in the existing MPAs is shown in parentheses. Values of all species conservation features are expressed as percentage of the maximum predicted value of the species' abundance/probability of presence/spawning biomass/potential fecundity. IV, index of the intrinsic vulnerability of (commercial) fish to fishing.
C, commercial, NC, non-commercial.

length–weight relationships (Morato *et al.*, 2001). Model predictions were converted to a relative scale (i.e. the percentage of the maximum predicted biomass). For two species (*L. bergylta*, *S. cretense*), it was only possible to model and predict the probability of the presence of mature individuals on a scale from 0 to 1.

Additional species features considered the potential fecundity of mature females of each of the four commercial species (Schmiing, unpublished data). The number of females in UVCs was estimated via sex ratios from local populations, whenever possible per size class (Morato *et al.*, 2003; unpublished data). GAMs were applied to produce predictive distribution maps of the potential fecundity on a relative scale (i.e. the percentage of the maximum predicted fecundity).

Predictive maps of all species features were on a scale from 0 to 100% of the predicted maximum abundance or from 0 to 1 for maps resulting from presence–absence models, respectively. Each species feature was then categorized in four different classes (low: <10%, middle: >10–25%, high: >25–50%, very high: >50% for relative abundances or low: <0.1, middle: 0.1–0.25, high: >0.25–0.50, very high: >0.50 for the probability of presence, respectively; Table 1, adapted from Schmiing *et al.*, 2013).

A predictive distribution map of the intrinsic vulnerability of fish to fishing was used as "assemblage feature". This map resulted from a predictive model of the intrinsic vulnerability index (Schmiing *et al.*, 2014), considering 30 commercial species sampled in UVCs (Supplementary Figure S1). The index is based on life history and ecological characteristics of marine fish, namely maximum body length, age at first maturity, von Bertalanffy growth parameters, natural mortality rate, maximum age, geographic range, annual fecundity, and aggregation behaviour (Cheung *et al.*, 2005, 2007). It ranges from 1 to 100, with 100 being the most vulnerable (Cheung *et al.*, 2005). Long-lived, large-bodied, and slow growing species with late maturity are considered most vulnerable to fishing and have a slow recovery from exploitation (Reynolds *et al.*, 2005; Cheung *et al.*, 2007). The predicted range of the vulnerability index in the study area was on a scale from 35 to 51 and classified into three categories (low: 35–43, middle: >43–47, high: >47–51, Table 1).

"Biotope features" included spatial predictions for the presence–absence of six dominant macroalgae (articulated corallinaceae, *Codium eslisabethae*, *Dictyota* spp., *Halopteris filicina*, *Padina pavonica*, *Zonaria tournefortii*, Table 1, Supplementary Figure S1, from Tempera, 2008; Tempera *et al.*, 2012).

Habitat maps of the two main habitat types, rock and sediment, derived from multibeam surveys, were integrated as "habitat features" in the analysis (Supplementary Figure S1, Tempera, 2008; Tempera *et al.*, 2012). In total, 79 conservation features were available (Table 1, Supplementary Figure S1). The amount of each feature in the existing MPAs was estimated (Table 1).

Socio-economic data: costs and benefits

The fishing effort was used as a surrogate for the spatial distribution of the non-monetary opportunity cost for different fisheries (i.e. the negative impact an MR might have for fishers; Klein *et al.*, 2008b). Data from standardized spatial annual fishing effort (2 × 2 km grid), estimated via roving creel surveys in 2004/2005 (Diogo, 2007; Diogo and Pereira, 2013), were used for this purpose. During the surveys, a shore-based observer counted the number of spearfishers, shore anglers, recreational, commercial, and unidentified fishing vessels. Additionally, interviews were conducted with the fishers to characterize the catch (Diogo, 2007; Diogo and Pereira, 2013, 2014). The resulting fishing effort grids were refined according to expert knowledge (P. Afonso, H. Diogo, local fishers): (i) spearfishing was restricted to rocky bottom down to the 20 m isobath and the first 40 m of neighbouring sediment, (ii) shore angling was restricted to a 50 m buffer around the coast, and (iii) recreational and commercial boat fishing were restricted to occur outside this 50 m buffer. The four different fishery types were then combined to a single fishing effort layer (Figure 2) using a weighted sum based on the relative importance of each

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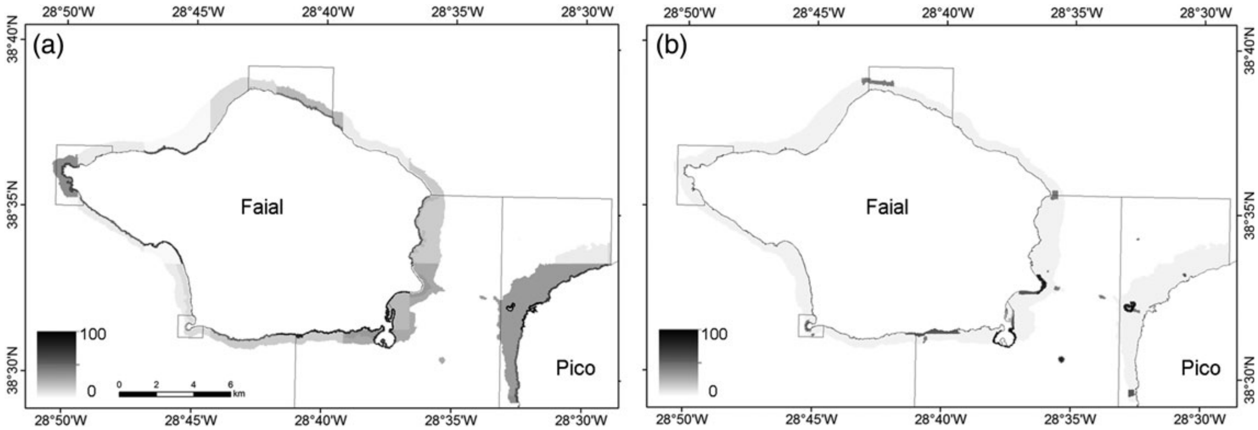

Figure 2. Spatial distribution of multiple users' costs and benefits. A relative, standardized scale from 0 to 100 was used to measure the (a) fishing effort, *F*, and (b) benefits for recreational, non-extractive activities, *R*. See text for further details. The boundaries of the existing MPAs are shown (black rectangles).

activity as a function of the total catch per species in tonnes for the years 2004/2005 (Diogo, 2007; Diogo and Pereira, 2013, 2014). Species that were absent in the UVCs, which were used to produce predictive models and maps, were excluded from these statistics to correct the catch importance. The resulting total fishing effort *F*, which ranged from 0 to 72 (standardized number of fishing operations per grid cell), was converted to a standardized, relative scale from 0 to 100.

In addition, a second cost surrogate layer that also considered the potential benefits of MRs for tourism was developed. For this purpose, the importance of recreational, non-extractive human uses, namely scuba diving and glass-bottom boat activity, were estimated via face-to-face interviews with managers of four scuba diving centres and one glass-bottom boat operator in Faial and Pico (see also Ressurreição and Giacomello, 2013). All parts of the study area were grouped into four categories according to the visiting frequency. Classes were then converted to values between 0 and 100 (not visited, 0; least visited, 33.3; regularly visited, 66.6; most frequently visited, 100). These two activities were summed up to a single non-extractive layer, assigning the same importance to both (i.e. to maintain the importance of areas that are only visited by the glass-bottom boat, Figure 2). Non-extractive, recreational activities were considered to benefit from marine conservation. This reflects the current MPA legislation that allows these activities in most of the protected zones, except inside the Caldeirinhas MR.

The non-extractive layer was then combined with the fishing effort layer *F* to calculate the total cost *C* for each planning unit (PU) of the study area:

$$C = (0.47 \times F - 0.53 \times R) + \propto,$$

where *R* is the benefit for recreational, non-extractive activities in PU *i*, and α a constant for scaling *C* to values equal to or higher than zero (i.e. to avoid negative values if the benefit outweighs the fishing effort cost surrogate). Weighting factors were based on the monetary value of landings of coastal fishing activities in 2011 for Faial and Pico Islands (excluding species that were absent in the UVCs, Diogo and Pereira, 2014) and on the estimated monetary value of diving and glass-bottom boat activities in 2011. The revenue of the glass-bottom boat was provided by the operator, and diving revenues were extrapolated using the best data available, i.e. statistics published by the Regional Statistics Service (SREA, http://estatistica.azores.gov.pt/). For this purpose, the expected number of divers was estimated and multiplied by the average price of a coastal dive package. The resulting weighting factors were almost equal, supporting the general notion of the high importance of both (artisanal) fishing (Carvalho *et al.*, 2011a; Pham *et al.*, 2013) and recreational activities in the Azores (Diogo, 2007; Diogo and Pereira, 2013, 2014; Ressurreição and Giacomello, 2013).

Reserve selection scenarios

The study area was divided into 23 710 hexagonal PUs, each with an area of 2500 m^2. The software Marxan (version 2.43, Ball *et al.*, 2009) was used to find solutions for a reserve system that achieves *a priori* defined conservation targets for the least costs, the so-called "minimum-set problem" (Possingham *et al.*, 2000; Ball *et al.*, 2009). Marxan uses simulated annealing to mathematically select a set of PUs that achieve the conservation objectives most efficiently. The score of a solution given by Marxan is the sum of the cost of the reserve system, plus its boundary length and a penalty if a conservation feature is not adequately represented in a solution. The boundary length modifier (BLM) modifies the amount of spatial clustering of PUs, and the species penalty factor (SPF) weights the relative importance of each feature (Game and Grantham, 2008).

A total of six reserve selection scenarios were run: three "fisheries scenarios" (FSs) and three "biodiversity scenarios" (BSs; Table 1). FSs analysed potential solutions for reserve networks with fisheries management purposes. They considered only distribution patterns of commercial species, the vulnerability of the commercial fish assemblage to fishing, and reef habitats. In each FS, a total of 56 conservation features were used. The fishing effort layer (*F*) was used as cost surrogate. BSs, on the other hand, analysed solutions for the protection of a wider set of conservation features, considering commercial and non-commercial species, reef and soft-bottom habitats, and algae biotopes as biodiversity surrogates. In each BS, a total of 44 conservation features were used. The total cost *C* (combination of the fishing effort *F* with the benefits of non-extractive, recreational activities) was used as cost layer.

Three different conservation targets were compared for both scenario types: the (i) "existing protection"—conservation targets were equal to the present protection of all features in the existing MPAs (FS/BS 1); (ii) "minimum protection"—targets increased

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with increasing level of each species feature (0, 5, 10, and 15% for low, middle, high, and very high levels of abundance/probability of presence/spawning biomass/potential fecundity) and assemblage feature (5, 10, and 15% for low, middle, high levels of fishing vulnerability) and were 10% for selected habitat or biotope features (FS/BS 2); (iii) "maximum protection"—targets were three times higher than in the minimum protection scenarios (FS/BS 3; Table 1). Targets of FS/BS 2 and FS/BS 3 were adapted from international guidelines recommending the protection of at least 10–30% of each marine biome or habitat (World Parks Congress, 2003; Convention on Biological Diversity, 2010). Minimum protection targets were defined to be at the lower end of this recommended range (10%) and maximum protection targets at the upper extreme (30%). The PUs inside the Caldeirinhas MR were *a priori* set to be included in the reserve selection of all scenarios. The size of this area was defined to be the minimum size of a priority area.

Each Marxan analysis was run with 100 repetitions and a billion iterations using the "Zonae Cogito" interface (version 1.74, Segan *et al*., 2011). The SPF was set to 1 for all features and the BLM was calibrated for each scenario individually, according to Stewart and Possingham (2005). For this purpose, a range of BLM values were tested and the BLM that resulted in minimization of the boundary length for a small increase in cost was chosen (Table 2). The BLM was further increased to enhance clumping and selection of larger priority areas if solutions included priority areas smaller than the Caldeirinhas reserve.

The selection frequency, or summed solution, describes the number of times each PU was selected over the 100 runs of each scenario. It was analysed as an estimate of the PUs' irreplaceability (Carwardine *et al*., 2007), i.e. the priority of protection, and classified into five categories (never selected: 0; no priority: 1–10 selections; low priority: 11–50; medium priority: 51–90; high priority: >90). Best solutions for each scenario were defined according to the lowest objective function score over all 100 runs. Network statistics of all solutions were calculated for: (i) the total extent of the protected area, (ii) the average size of each site, (iii) the number of selected sites, (iv) the mean edge-to-area ratio, (v) the percentage of the coastline integrated in network solutions, and (vi) the shortest linear distance between centres of individual sites. A Kruskal–Wallis test was used to test for differences between the parameters of the six scenarios and the statistics of the existing MPAs. Zones of the present network go beyond the 40 m isobath and thus were adapted to the depth limit of this study. As result, the three offshore reefs in the channel were considered as separate zones, and the Caldeirinhas reserve was joined to the neighbouring site of the Faial–Pico channel because they share boundaries. The costs of all solutions were compared with each other. For this purpose, the constant α, multiplied by the number of PUs of a given solution for BS 1–3, was subtracted. In addition, the cost of the existing network was calculated, using both cost surrogate layers (F, C). Site selection was compared with the existing MPAs by calculating the percentage of PUs that were (i) identified as priority areas for conservation but not included in the existing MPAs, (ii) identified as priority areas and included in the existing MPAs, (iii) not identified as priority areas but included in the existing MPAs, and (iv) not identified as priority areas and not included in the existing MPAs (adapted from Giakoumi *et al*., 2013).

Results

Six reserve selection scenarios were produced for the coastal areas of Faial and West-Pico Islands. Solutions for reserve networks are shown in Figure 3 and the selection frequency of the PUs is shown in Supplementary Figure S2. At least one priority area for conservation was selected at the northwest coast of Pico and the northwest coast of Faial, irrespective of the conservation objective and target level (Figure 3). Four of the eight sites of the existing MPAs (Cedros at the north coast of Faial and all three offshore reefs) were never selected in the Marxan solutions.

Network statistics of all scenarios vs. the existing network are summarized in Table 2. The average distance between individual sites was <12 km in all scenarios, irrespective of the conservation objective or target level. The mean edge-to-area ratio of the present MPAs was reduced to 5.01 ($\pm$2.40) if the three offshore reefs were excluded from the analysis. The Kruskal–Wallis test did not reveal significant differences. Site selection of individual reserves in FSs was visually similar to most of the solutions for BSs with the same conservation target level. Solutions for FS 1 and FS 2 did not consider the "best" solution with the lowest score, but the next best solution that produced a desirable level of compactness (i.e. minimum defined patch size). This was not achieved with a further increase in the BLM. Reserve costs, in general, increased with increasing protection target levels and costs of solutions for FSs were typically higher than of BSs (Table 2). The existing MPA network had the highest overall costs for conservation (Table 2).

Table 2. Network statistics, costs, and BLM value of solutions for three FS and BS reserve design scenarios, respectively, derived from Marxan analyses.

Scenario	Total area (km^2)	% of study area	% of coastline	Number of sites	Mean ind. size (km^2) ($\pm$SD)	Mean edge-to-area-ratio ($\pm$SD)	Mean distance between sites (km) ($\pm$SD)	Cost	BLM
FS 1	36.0	61	52	9	4.0 ($\pm$4.8)	5.7 ($\pm$3.9)	7.2 ($\pm$4.4)	141 781	2
FS 2	5.9	10	11	8	0.7 ($\pm$0.7)	8.2 ($\pm$4.1)	8.6 ($\pm$7.7)	10 961	0.6
FS 3	17.6	30	24	8	2.2 ($\pm$3.6)	7.0 ($\pm$4.6)	8.0 ($\pm$2.4)	75 034	3
BS 1	35.0	59	58	8	4.4 ($\pm$5.4)	6.1 ($\pm$4.2)	8.0 ($\pm$4.3)	55 415	4
BS 2	5.9	10	10	6	1.0 ($\pm$0.9)	8.7 ($\pm$6.5)	11.5 ($\pm$8.8)	8335	2
BS 3	20.3	34	30	10	2.0 ($\pm$3.2)	9.4 ($\pm$7.7)	6.4 ($\pm$3.0)	37 398	4
Present MPA	35.1	60	62	8	4.4 ($\pm$6.3)	14.1 ($\pm$14.9)	8.9 ($\pm$5.5)	209.938^F/ 74 852^C	–

Conservation targets for FS and BS were defined as (i) equal to the existing protection level, (ii) minimum, and (iii) maximum (see text for detailed description). FSs used the fishing effort (F) and BSs a combination of F and the benefits for recreational, non-extractive activities (C) as cost surrogate layer (standardized scales, 0–100). Average values of individual sites are reported, including standard deviations (SD). The comparison to the existing MPA network considering the 40-m-depth limit used in the study is given at the end of the table.

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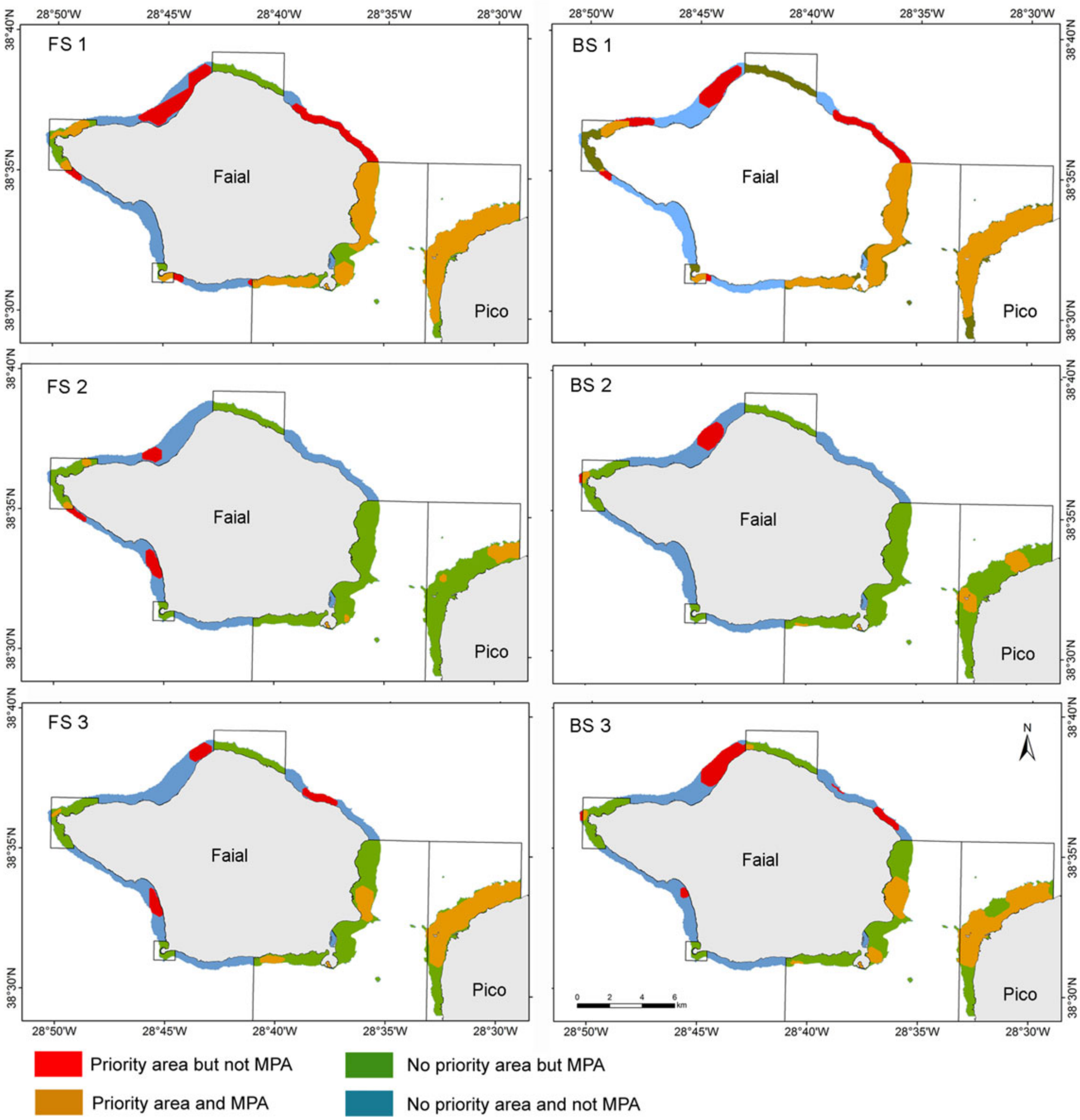

Figure 3. Priority areas for conservation in Faial and West-Pico Islands, Azores archipelago, compared with the existing MPA network. Scenarios considered either fisheries management objectives (FSs, left column) or the protection of biodiversity (BS, right column) and had (i) the present, (ii) a minimum, or (iii) maximum protection level as conservation target. Colours indicate the percentage of PUs that were (a) identified as priority areas for conservation but were not included in the existing MPAs (red); (b) identified as priority areas and were included in the existing MPAs (orange); (c) not identified as priority areas but included in the existing MPAs (green); and (d) not identified as priority areas and not included in the existing MPAs (blue; adapted from Giakoumi *et al.*, 2013). The boundaries of the existing MPAs are shown (black rectangles).

Fisheries scenarios

The solution for the FS with the existing level of protection as conservation target (FS 1) included the largest total area and average size of individual reserves of all FSs (Table 2). It was comparable to the total and average size of individual sites of the existing MPAs. The spatial overlap between priority sites and the existing MPAs was high (45% of the study area, Figure 3). This solution had the lowest edge-to-area ratio of all scenarios and the highest cost (Table 2). The solution for minimum conservation targets (FS 2) had the smallest average reserve size and the greatest distances between them. No priority areas were selected at the north (-east) coast of Faial. The spatial overlap between priority sites and the existing MPAs was small (Figure 3). Maximum conservation targets (FS 3) resulted in a solution for a reserve network with the same number of sites, but half the size of the existing MPA network. Reserves were equally distributed around Faial Island. The total cost of FS 3 was estimated at half the cost of FS 1.

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Biodiversity scenarios

The best solution for the BS with the existing level of protection as conservation target (BS 1) resulted in a zoning comparable to the solution for the FS with the same target (FS 1) and to the present network (Figure 3, Table 2). Of all six scenarios, BS 1 included the largest amount of the coastline. Site selection for the conservation of biodiversity under minimum target levels (BS 2) differed compared with FS 2 (Figure 3). The solution included a small total and average reserve size, comprised the lowest percentage of the coastline, had the lowest cost of all scenarios (Table 2), and had a minimal spatial overlap with the existing MPAs (Figure 3). Such smaller solutions exhibited the highest edge-to-area ratios (Table 2). The BS with maximum conservation targets (BS 3) had the lowest edge-to-area ratio of all scenarios and resulted in total and average reserve sizes comparable to the best solution for fisheries objectives under the same protection target level (FS 3, Table 2). Compared with FS 3, BS 3 was less costly by 50% (Table 2).

Discussion

The effective management and sustainable use of marine resources is essential to minimize conflicts between fisheries, recreation, and conservation. Arguably, reserves chosen opportunistically may be better than no reserves at all (Roberts and Hawkins, 2000), but they may be inefficient and even compromise effective conservation (Stewart *et al*., 2003).

Six reserve selection scenarios were produced with the systematic conservation planning software Marxan to analyse differences and similarities between solutions for conservation networks with different conservation objectives and protection target levels. Additionally, site selection of an existing network that was designed based on best available (scientific) knowledge, but also opportunistic approaches were assessed and compared with results of systematic conservation planning. Marxan includes or excludes PUs in a reserve scenario, assuming that all conservation features inside the reserve are fully protected (Ball *et al*., 2009). The existing MPAs, however, in the study area include mainly sites under partial protection and only one very small "no-take" MR. Therefore, conservation features were assumed as fully protected in the current MPA network to compare its design with solutions provided by Marxan.

The importance of the conservation objective was most evident for minimum conservation target levels (FS/BS 2). Although network statistics were comparable, site selection was rather different, particularly around Faial Island. The integration of different conservation features in both scenarios is probably the driving factor behind this pattern and highlights the fact that spatial management needs to be adjusted to the objectives instead of using a "one-size-fits-all" approach. Consequently, a clear formulation of objectives is significant for a successful implementation of reserves (Fernandes *et al*., 2005; Stelzenmüller *et al*., 2013). Nevertheless, spatial prioritization in FS 1 and BS 1 (or FS 3 and BS 3), with different conservation objectives but the same target, were quite similar, supporting results of other studies that demonstrate how combined solutions for fisheries management and the protection of biodiversity are feasible (Klein *et al*., 2008b, 2009; Palumbi *et al*., 2009; Gaines *et al*., 2010).

Reserve selection is also largely driven by the target protection level which may even be prevailing. For example, FSs (or biodiversity, respectively) with different protection target level (FS 1–3 or BS 1–3) varied considerably. As anticipated, reserve size increased with increasing conservation target level, whereas the edge-to-area ratio decreased. The fragmentation or compactness is essential to the functioning of a conservation network (Possingham *et al*., 2000). MRs, especially small ones, may not equally benefit all species because of their different spatial habitat uses (Kramer and Chapman, 1999; White *et al*., 2010). Some of the species whose distribution maps were integrated in this study have rather small home ranges, such as *L. bergylta* (Villegas-Ríos *et al*., 2013), *S. salpa* (Jadot *et al*., 2006), and *S. cretense* (Afonso *et al*., 2008b). Minimum conservation targets and small reserves as provided by FS 2 and BS 2 may effectively protect them and other highly sedentary species (Afonso *et al*., 2011). The size (and shape) of a reserve also influences possible edge effects. The common phenomenon of "fishing the line" (i.e. fishing activity concentrated at the reserve boundary; Kellner *et al*., 2007; Goñi *et al*., 2008) will increase with increasing boundary length. This can have considerable negative impacts on fish populations inside and close to reserve boundaries (Kellner *et al*., 2007). Similarly, with decreasing size of the reserve's core area, edge-sensitive species can be negatively influenced (Possingham *et al*., 2000). Remarkably, the mean edge-to-area ratio was up to three times higher in the existing network, although no significant differences were found between solutions. This value, however, is driven by the three small offshore reefs. In general, reserves with smaller edge-to-area ratio are easier to manage and less costly, due to having less boundaries and neighbouring areas (Possingham *et al*., 2000; Ball *et al*., 2009). Protection of larger individual sites with small edge-to-area ratio might generate biological or ecological advantages, for example, for more mobile species, such as *Pseudocaranx dentex* (Afonso *et al*., 2009) and *D. sargus* (Abecasis *et al*., 2013), but it might also increase conflicts. The number of competing user groups that are affected by the implementation of an MR, most likely increases with increasing reserve size, particularly in coastal areas. Thus, larger reserves might be easier to accept (although harder to enforce) in offshore areas, whereas smaller, more numerous reserves are preferable in coastal regions (Roberts *et al*., 2005).

Individual reserves also have to be close enough to ensure connectivity, a key aspect of MPA sustainability (Ardron, 2008). Distances between individual sites were probably adequate to promote network connectivity, considering the rather short larval dispersal distances of many reef fish (Jones *et al*., 2009) and maximum recommended distances between individual reserves of 30 km (McCook *et al*., 2009).

Marine conservation planning often uses the distribution of species (abundances or occurrences) or habitats to identify priority areas for conservation (e.g. Lourival *et al*., 2011; Giakoumi *et al*., 2013). Fish, however, often have varying ontogenetic habitat preferences and may use distinct habitats for their reproduction (e.g. spawning sites, Sala *et al*., 2003). The integration of spatial patterns of the spawning biomass and the potential fecundity of fish is proposed as surrogate to target these essential habitats in spatial prioritization. Spawning biomass also proved to be a good surrogate for reef fish connectivity (Bode *et al*., 2012). In addition, the vulnerability of fish to fishing was included as conservation feature to improve protection of most vulnerable assemblages and support their recovery. To the best knowledge, this is a novel approach.

Network statistics of solutions with the existing protection as target level (FS/BS 1) were comparable to the present MPAs. The spatial overlap between priority areas and existing MPAs was high. Thus, the current zoning scheme seems quantitatively adequate. FS 1 and BS 1, however, were less "costly" than the existing MPA. The same conservation objectives could be achieved while reducing

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possible negative effects on fisheries. The existing MPAs have high targets for all categories of the conservation features (FS/BS 1), including areas with low fish abundance or reproductive potential (spawning biomass, potential fecundity; i.e. targets >50%). Contrary to FS/BS 1, FS/BS 2 and FS/BS 3 only set highest targets for areas with maximum fish abundance or reproductive potential, because these areas are considered as preferential habitats or potential essential fish habitats (Schmiing *et al.*, 2013). Targets for potentially "low-quality" habitats are minimal to reduce the total reserve size and cost (i.e. the cost was reduced to half). This approach explicitly considers species-specific habitat preferences and integrates them in the reserve selection scenarios by giving more weight (i.e. higher targets) to ecologically or biologically most important areas. Their protection is critical to enhance the reproductive output of a population and the potential subsidy to adjacent habitats and fisheries (Sala *et al.*, 2003; Pelc *et al.*, 2010; Bode *et al.*, 2012).

Importantly, Marxan solutions do offer zoning alternatives that leave out currently protected sites and instead include new sites, particularly in the (north-) west of Faial. Up to 15% of the study area were identified as priority for conservation, but are not included in the current zoning scheme. Site selection of such priority areas was similar in at least half of the scenarios, irrespective of the conservation objective or target (i.e. close to Praia do Norte at the north- and Varadouro at the southwest coast of Faial). This demonstrates the efficiency of systematic conservation planning, i.e. the integration of clear conservation objectives and targets, and spatial patterns of biological, ecological, and socio-economic values to detect potential important gaps in protection. Such areas should receive special attention in adaptive management processes as their protection might improve the ecological functioning of a reserve network.

Marine ecosystems are an important component of the human life, providing food and other resources, but also opportunities for leisure activities. Fishing can be of considerable socio-economic local importance, particularly in small regions (Carvalho *et al.*, 2011b; Pham *et al.*, 2013). In the Azores, as much as 90% of the fishing fleet and half of the catch are attributed to small-scale, artisanal fisheries (Carvalho *et al.*, 2011a). Restricting fishing grounds might negatively impact local communities, thus a balance between the protection of fish assemblages and their exploitation is required. Consequently, the integration of socio-economic values is central for conservation planning to promote the ecological and socio-economic success of reserves, to minimize conflicts with stakeholders, and to consider "real-world" scenarios (Lundquist and Granek, 2005; Klein *et al.*, 2008a, b, 2009; Ressurreição *et al.*, 2012a, b, c). Scenarios presented in this study incorporate not only ecological, but also socio-economic information and certainly will facilitate and promote communication in stakeholder meetings. The approach presents a straightforward method to estimate cost surrogates when (spatial) monetary values are missing. Fishing effort was used as cost surrogate layer, considering commercial and recreational fisheries (Diogo, 2007; Diogo and Pereira, 2013, 2014; Pham *et al.*, 2013). However, it was calculated on a rather large spatial scale compared with the other biological and environmental data. Its refinement, particularly in coastal areas, is a potential future task that probably will improve cost layers for reserve selection scenarios.

Recreational, non-extractive activities (i.e. dive spots and areas for boat-based observations of marine life) were added as benefit to the cost surrogate layer as next step. The combination of both recreational, non-extractive activities with each other was not based on their economic importance (i.e. the glass-bottom boat was established recently and is still in the initial growth stage), instead equal weights were assigned to maintain the importance of areas that are only visited by the glass bottom boat. The overlap of both areas is very small (<1% of the study area). In the future, it will be necessary to improve the knowledge of the total annual effort and economic value of both activities, which currently have a growing tendency in the region. The protection of important recreational sites was considered as beneficial because it may enhance non-extractive, recreational activities (e.g. Angulo-Valdés and Hatcher, 2010; Ressurreição and Giacomello, 2013). This decision is in accordance with the regional legislation. Non-extractive, recreational activities are currently not forbidden in the existing MPAs, except inside the Caldeirinhas MR. Furthermore, "dive reserves" were implemented in the Azores, and elsewhere, to ban fishing and to promote scuba diving. Such activities depend on the presence and health of natural marine resources and on habitats with a high biodiversity (Rees *et al.*, 2010). Promoting sites with high recreational value also implies possible economic advantages because people pay for recreational activities (Ressurreição and Giacomello, 2013). These incomes can add up to the same economic value as fisheries (Ruiz-Frau *et al.*, 2013). People are also willing to pay for the conservation of marine biodiversity (Ressurreição *et al.*, 2011, 2012b), whereas the valuation of certain marine taxa may reflect cultural differences (Ressurreição *et al.*, 2012c).

To date, there are few examples of MR network design that explicitly integrate opportunity cost for fishing activities with benefits from tourism and related recreational activities (e.g. Watts *et al.*, 2009; Giakoumi *et al.*, 2011, 2012). The present study contributes substantially to this growing literature. BS achieved conservation targets more cost-efficiently than FSs, indicating that more biodiversity features could be conserved when taking into account multiple ocean users (e.g. fishers and scuba divers) who, in turn, could benefit from the implementation of these scenarios. Results suggest that decision-makers should explicitly account for multiple users' opportunity cost and benefits when designing and implementing MR networks. It is the first time that a systematic conservation planning software is applied in the framework of marine spatial planning in the Azores archipelago and to the best knowledge, only one other study is published for an European island in the Atlantic (García *et al.*, 2010). As such, the presented approach is believed to support the wider context of coastal spatial management, particularly in islands with limited shelf areas, and to deliver strategies for an adaptive management of MPAs.

Future studies may benefit from systematic conservation planning software that does not only differentiate between "protected" and "non-protected" zones, but also between different levels of protection, such as Marxan with Zones (Klein *et al.*, 2009; Watts *et al.*, 2009). This is particularly true for areas that experience multiple uses and restrict extractive uses but not recreational, non-extractive activities, such as in this study. An alternative approach might be to combine the solutions of different scenarios. Sites that were identified with minimum conservation targets were generally small and could be defined as "no-take" areas, whereas sites selected with maximum conservation targets could be designated as partially protected or "buffer" zones.

In conclusion, the presented approach (i) uses systematic conservation planning to assess an existing network of MPAs, (ii) integrates the abundance, reproductive potential and vulnerability to fishing of multispecies, in addition to other environmental and socio- economic criteria, (iii) gives more weight to ecologically significant habitats, (iv) suggests that multiple users' cost and benefits

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should be considered in conservation planning, and (v) identifies possible conservation gaps and produces alternative flexible solutions that aim to facilitate the dialogue between stakeholders and can be used in adaptive management to improve existing conservation networks. It is feasible in any marine region, whereas costs, benefits, conservation features, targets, and objectives can be adapted to meet the particular needs of each planning task.

Supplementary data

Supplementary material is available at the *ICESJMS* online version of the manuscript.

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