Extending full protection inside existing multiple-use marine protected areas or reducing fishing effort outside can both deliver conservation and fisheries outcomes

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Mokrane Belharet[1], Antonio Di Franco[2,3], Antonio Calò[2,4,5], Lorenzo Mari[1], Joachim Claudet[6], Renato Casagrandi[1], Marino Gatto[1], Josep Lloret[7], Charlotte Sève[6], Paolo Guidetti[2], Paco Melià[1]

[1]Dipartimento di Elettronica, Informazione e Bioingegneria, Politecnico di Milano, Via Ponzio 34/5, 20133 Milano, Italy.

[2]Université Côte d'Azur, CNRS, UMR 7035 ECOSEAS, Parc Valrose 28, Avenue Valrose, 06108 Nice, France.

[3]Stazione Zoologica "Anton Dohrn" sede interdipartimentale di Palermo, Italy

[4]CoNISMa, Piazzale Flaminio 9, 00196 Roma, Italy.

[5]Dipartimento di Scienze della Terra e del Mare (DiSTeM), Università di Palermo, Via Archirafi 20, 90123 Palermo, Italy

[6]National Center for Scientific Research, PSL Université Paris, CRIOBE, USR3278 CNRS-EPHE-UPVD, Maison des Océans, 195 rue Saint-Jacques 75005 Paris, France.

[7]University of Girona, Faculty of Science, C/Maria Aurèlia Company 69, 17003 Girona, Catalonia, Spain.

Correspondence

Mokrane Belharet

E-mail: mokrane.belharet@gmail.com

Paco Melià

E-mail: paco.melia@polimi.it

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Abstract

1 Most fish stocks worldwide are overfished, and many fisheries management strategies have failed to achieve sustainable fishing. Identifying effective fisheries management strategies has now become urgent.

2 Here, we developed a spatially-explicit metapopulation model accounting for seascape connectivity in the Mediterranean Sea, and parameterized it for three ecologically and economically important coastal fish species: *Diplodus sargus*, *Diplodus vulgaris* and *Epinephelus marginatus*.

3 We used the model to assess how stock biomass and catches respond to changes in fishing mortality rate (F) and in the size of fully protected areas within the existing network of multiple-use marine protected areas (MPAs). For each species, we estimated maximum sustainable yield (MSY) and the corresponding values of stock biomass (B_{MSY}) and fishing mortality rate (F_{MSY}), providing crucial reference points for the assessment of fisheries management.

4 All three species are currently overexploited. Stock recovery to B_{MSY} requires a reduction of current F between 25–50% (depending on the species). This would guarantee an increase in both stock biomass (17–42%) and catch (2–13%) after a transient time of ~10–20 years. Alternatively, increasing the size of fully protected areas over fishable areas within the existing network of MPAs would lead to positive conservation effects for all three species without impairing the productivity and profitability of the fishery.

5 *Synthesis and applications.* We provide the first assessment of stock status for three coastal species in the north-western Mediterranean and evaluate the ecological and fisheries outcomes of different management strategies. Extending full protection

inside existing multiple-use marine protected areas and or reducing fishing pressure effort outside can both deliver conservation and fisheries outcomes.

Key words

Fisheries management, Metapopulation models, Coastal fish, Stock assessment, Marine conservation.

1 Introduction

Marine fisheries provide a major source of food and livelihood for hundreds of millions of people worldwide (FAO 2018). However, most of the world's fish stocks are overfished (Costello *et al.* 2016), with strong cascading impacts on both marine biodiversity (Sala *et al.* 2012; Ortuño Crespo & Dunn 2017) and societies (Golden *et al.* 2016).

Several strategies have been proposed to pursue sustainability in fisheries (Hoggarth 2006; Coll *et al.* 2013; Goetze *et al.* 2016; Carvalho *et al.* 2019). Traditional management has focused on adjusting fishing effort to levels guaranteeing maximum sustainable yield (MSY), i.e. the maximum catch that can be removed from a stock over time without depleting it. MSY and its related biological reference points, such as stock biomass (B_{MSY}) and fishing mortality rate (F_{MSY}), are benchmarks used for gauging the status of a stock or fisheries (Hilborn & Ovando 2014). Although many coastal species are key targets for small scale artisanal and recreational fisheries (Lloret *et al.* 2019), for most of them these reference points have never been assessed.

In coastal areas, multiple-use Marine Protected Areas (MPAs) can be used as a means to combine maritime spatial planning and the ecosystem approach to fisheries management (Claudet *et al.* 2006; Gaines *et al.* 2010; Melià *et al.* 2016). Their actual ecological effectiveness is affected by the presence and extent of fully protected areas within MPA borders (Zupan *et al.* 2018). Although they are often not established primarily for fisheries management (García-Charton *et al.* 2008), MPAs can provide benefits to fisheries (Russ & Alcala 2004; Di Franco *et al.* 2016) and other socioeconomic activities (Pascual *et al.* 2016). Finding a balance between biological conservation and socioeconomic viability is fundamental to ensure the consensus among stakeholders necessary for the success of MPAs (Klein *et al.* 2013; Melià 2017).

Whether benefits at the local scale (thanks to recruitment subsidy and/or spillover effects; Di Lorenzo *et al.* 2016) can scale-up and make MPAs useful tools for fisheries management also at a broader scale is still controversial (Hilborn 2015; Hughes *et al.* 2016). Quantitative tools able to describe the coupled spatiotemporal dynamics of fish and fisheries are hence crucial to assess the actual implications of proposed management measures in a realistic way (Botsford *et al.* 2009; Bastardie *et al.* 2017). Although studies linking seascape connectivity with population dynamics are scarce to date (but see, e.g., Watson *et al.* 2012; Treml *et al.* 2015), the explicit integration of these aspects into a metapopulation approach is key to understand the ecological and evolutionary dynamics of coastal marine populations, as well as to assess the long-term consequences of alternative management policies from a spatially explicit perspective (Botsford *et al.* 2009; Guizien *et al.* 2014).

Here we developed two sets of scenarios to assess the role of MPA networks as a tool to support fisheries management of three key coastal species in the north-western Mediterranean Sea. The scenarios were simulated using a biophysical metapopulation model, based on realistic patterns of connectivity estimated via Lagrangian simulations. The performances of each scenario were evaluated in terms of three indicators of conservation and socioeconomic relevance: stock biomass, fisheries catch and total value of catch. First, we tested the effects of regulating fishing mortality rates and estimated biological reference points for the three species. Second, we tested the role of the presence and size of fully protected areas in determining the bio-economic effectiveness of multiple-use MPAs. Finally, we discussed the effectiveness of the considered scenarios for achieving sustainable fisheries management objectives.

2 Methods

2.1 Case study

The study area covers the north-western Mediterranean Sea, and in particular the region located between latitudes 38.5°N– 45°N and longitudes 1°E–12°E. We focused on three fish species of high ecological and economic relevance (Guidetti *et al.* 2014) and vulnerable to small scale and recreational fishing (Lloret *et al.* 2019): the white seabream *Diplodus sargus*, the two-banded seabream *Diplodus vulgaris*, and the dusky grouper *Epinephelus marginatus*. The three species are common in the Mediterranean Sea: they thrive in littoral rocky bottoms and generally occur from a few meters down to approximately 50 m depth, although they can be found, at lower densities, even at greater depths (especially *E. marginatus*; Harmelin & Harmelin-Vivien 1999). Their bipartite life cycle is typical of the majority of coastal species, with a pelagic larval phase and a benthic adult phase (see section S1 in the Supplementary Information for further details).

2.2 Metapopulation model

We developed an age-structured, discrete-time metapopulation model, based on a biophysical model accounting for habitat suitability and oceanographic connectivity. The model describes, in a spatially explicit framework, all the key biological processes affecting the species' demographic dynamics, such as reproduction, larval dispersal, recruitment, and natural and fishing mortality. In the following sections, we concisely summarize the main features of the model; further details about its structure and formulation are given in section S2, while details on its calibration and validation are given in section S3.

2.2.1 Habitat suitability

The selected fish species have very similar habitat requirements. Therefore, we assumed the same suitable habitat (rocky and hard substrate, including coralligenous assemblages between

0–50 m depth) for all three species. Habitat was mapped using available information on bathymetry and seabed habitats from the EMODnet portal (www.emodnet.eu). Bathymetry was provided as a high-resolution raster map (1/480°; Populus et al. 2017). Seabed habitat maps were hand-corrected in QGIS software; in fact, although EMODnet maps represent the most updated georeferenced seafloor maps for the Mediterranean Sea, some areas included in our domain were associated to low confidence levels, while others completely lacked any habitat information. For these areas, we first cross-checked information on the EMODnet map with the distribution of coastline substrate types reported in Furlani *et al.* (2014), and then we analysed high-resolution satellite images from Google Earth to ascertain substrate type where the information did not match. In case of mismatch or absence of habitat information in the original map, we added a buffer of rocky substrate along the coast with its extent inversely proportional to the sea bottom slope.

2.2.2 Connectivity assessment

To evaluate seascape connectivity among local populations (i.e. among model cells), we carried out Lagrangian simulations of larval dispersal across the study area with an individual-based biophysical model. The physical component of the model was based on daily average current velocity fields made available through the Copernicus Marine Environment Monitoring Service (marine.copernicus.eu). Velocity fields, produced by the Mediterranean Sea physics reanalysis (Fratianni *et al.* 2014), had a 1/16° (~6–7 km) horizontal resolution and covered 72 unevenly spaced vertical levels. Lagrangian particles were released according to the reproductive schedule of each species and tracked for the duration of the whole larval phase. Simulations covered a 12-year-long time horizon (2004–2015). Results were aggregated across a grid with the same resolution of the ocean circulation dataset (1/16°) and used to derive a set of connectivity matrices for each species and each year. The element $c_{\{i,j,t\}} = \frac{n_{i\to j,t}}{n_{i,t}}$ of the connectivity matrix $C(t)$ is the ratio between $n_{i\to j,t}$ (i.e. the number of larvae starting from source cell *i* and successfully arriving to destination cell *j* at the end of their pelagic larval duration in year *t*) and $n_{i,t}$ (i.e. the total number of propagules released from cell *i* in year *t*). The diagonal elements of each connectivity matrix represent the retention rates of the considered cells in a specific year.

2.2.3 Protection

To describe the protection regime of each marine area, we considered three levels of protection: unprotected, partially protected and fully protected areas. Each cell within the spatial domain of the model was associated with at least one protection level. When there was more than one protection level in the same cell, we calculated the relative coverage of each protection level with respect to the total surface of the cell. To this end, we considered the 62 nationally designated Marine Protected Areas (MPAs) already established in the study area. Some are fully protected areas and some are multiple-use MPAs containing a fully protected area (Horta e Costa *et al.* 2016). Partially protected areas were identified with the portion of MPA that is not fully protected. Information on the MPAs (geographical coordinates, names, areas, establishment year, presence of fully protected areas, etc.) was derived from the MAPAMED database (medpan.org/main_activities/mapamed/). MPA perimeters were provided as georeferenced polygons, allowing us to define the geometric intersection with each cell and to calculate the corresponding surface area.

2.2.4 Population dynamics

Metapopulation dynamics were described by subdividing the stocks of the three species into subpopulations according to the same horizontal grid used for the connectivity assessment. To account for the heterogeneous distribution of suitable habitat within the study area, each cell was further subdivided into 30×30 sub-cells matching the spatial resolution of the bathymetric grid. The marine surface area A_i of each cell *i* was evaluated as the sum of the areal extent of its sub-cells with a valid bathymetric value. For each cell ***i*,** we calculated the surface area of suitable habitat A_i^{SH} as the area of the geometric intersection between the portion of cell between 0–50 m depth and the polygon of the suitable substrate. Only the cells with non-zero A_i^{SH} score (949 cells in total) were included in the metapopulation model (Fig. 1). Each sub-population was subdivided into age classes (15 for *D. sargus*, 9 for *D. vulgaris* and 20 for *E. marginatus*), whose dynamics were described by taking into account both the local demographics and the exchange of larvae under the action of the currents.

2.3 Fisheries management scenarios

Once calibrated and validated, we used the model to test different fisheries management scenarios for the three model species at the scale of the whole study area. Specifically, we investigated the response of stock biomass and catch to changes in (i) the fishing mortality rate, and (ii) the extent of fully protected areas in the current network of MPAs. In the first set of experiments, we considered a homogeneous reduction or increase of current fishing mortality rate (F_0) across the study area. In the second set of experiments, we changed the relative coverage of existing fully protected areas in the MPAs currently established in the study area, keeping the total surface area of each MPA unchanged. The area not included in the fully protected area was considered to be partially protected (i.e. with an intermediate level of fishing mortality).

For each management scenario, we performed a 50-year-long simulation with a time-averaged connectivity matrix and assuming the present distribution of the three metapopulations (as reconstructed through the calibration of the model) as the initial condition. The last ten years of each simulation were used to assess stock biomass and catch (integrated across space and averaged over time) for each species.

To evaluate the economic implications of the different scenarios tested, we estimated also the total value of catch (TVC) obtained from the fishery of the three study species. TVC was calculated as $\sum_k p_k \overline{C_k}$, where p_k is the market price of species k, and $\overline{C_k}$ is the total catch of species k averaged over the last 10 years of simulation. The relative change of TVC for each scenario was expressed as a percent change with respect to the TVC of the baseline simulation. Market prices were considered, based on an informal ex-vessel survey carried out across the study area, to be 20 EUR/kg for *D. sargus*, 18 EUR/kg for *D. vulgaris*, and 25 EUR/kg for *E. marginatus*.

3 Results

3.1 Effects of changing fishing mortality rate

The responses of stock biomass and catch of the three studied species to changes of fishing mortality rate at the scale of the whole study area are shown in Fig. 2. To make species-specific results easier to compare, we normalized biomass and catch values for each species with respect to the baseline simulation (performed under current fishing mortality, as estimated *via* model calibration). For all three species, normalized maximum sustainable yield (MSY) and the corresponding normalized stock biomass are >1, indicating that there is room for improvement over current management. Indeed, fishing mortality rates associated with MSY (F_{MSY}) are lower than current fishing mortality (F_0) for all species ($0.75F_0$ for *D. sargus*, and $0.5F_0$ for *D. vulgaris* and *E. marginatus*), suggesting that all three stocks are presently overfished.

For *D. sargus*, baseline biomass (B_0) and catch (C_0) are 83% and 98% of B_{MSY} and MSY, respectively. For the other two species, the discrepancy is even more pronounced: B_0 and C_0 for *D. vulgaris* are 64% and 90% of those associated with F_{MSY}, while for *E. marginatus* they are 58% and 87%, respectively. The relative values of B_{MSY} compared to unfished biomasses (i.e. with fishing effort set to zero across the whole study area) are 41% (for *D. sargus*), 47% (for *D. vulgaris*) and 37% (for *E. marginatus*). In parallel, the ratio of baseline biomass (B_0) on unfished biomass is 34% for *D. sargus*, 30% for *D. vulgaris*, and 22% for *E. marginatus*.

Fig. 3A shows the temporal dynamics of stock biomass over time under an MSY scenario. At the beginning of the simulations, relative biomass B/B_{MSY} is 0.85 for *D. sargus*, 0.61 for *D. vulgaris* and 0.60 for *E. marginatus*. Subsequently, relative biomass grows progressively until reaching its maximum ($B/B_{\mathrm{MSY}} = 1$). The duration of the transient required to approach B_{MSY} (i.e. for a full recovery of the stock) is ~10–20 years for the three species. Fig. 3B shows the temporal dynamics of catch (expressed, in this case, as the ratio between current catch and its present value, C/C_0) under the same scenario (MSY). Relative catches fall, during the first year of implementation of the scenario, from the present level (= 1 by definition) to approximately 0.79 for *D. sargus*, 0.55 for *D. vulgaris* and 0.54 for *E. marginatus*. Afterwards, they grow over time until reaching their maximum value (1.02 for *D. sargus*, 1.11 for *D. vulgaris* and 1.15 for *E. marginatus*). The time required to attain the present levels again ($C/C_0 = 1$) is about 16 years for *D. sargus*, 11 years for *D. vulgaris* and 10 years for *E. marginatus*.

3.2 Effects of expanding fully protected areas

Predicted responses of stock biomass and catch of the three species to changes in the relative coverage of fully protected areas (keeping fishing mortality rate at its present level F_0) are shown in Fig. 4. The effect of expanding fully protected areas on fish biomass are positive for all species and approximately proportional to the extent of full protection. When the relative coverage of full protection is set to 100% of the total protected area, the predicted increase in stock biomass relative to the baseline is 33% for *D. sargus*, 40% for *D. vulgaris*, and 61% for *E. marginatus*. On the other hand, effects on catch are species-dependent. For *D. sargus* and *E. marginatus*, catch is negatively related to the fully protected fraction (except when this ranges between its current value and 10% of the total protected area). In contrast, for *D. vulgaris* the effect of increasing the fully protected fraction is generally positive, except when the fraction is lower than the present one or >90% of the total protected area. In particular, catch of *D. vulgaris* is expected to be maximized by a fully protected area ~40% of the total protected area.

3.3 Economic consequences of the analysed scenarios

The response of total value of catch to changes in fishing mortality is shown in Fig. 5A. Under the current protection scheme, the predicted change in the total value of catch is positive for F comprised between $0.4F_0$ and F_0. The maximum value (+8%) is achieved for a fishing mortality ~60% of the present one. Beyond its maximum, total value declines progressively with increasing fishing mortalities.

The effect of changing the extent of full protection within existing MPAs on the total value of catch are shown in Fig. 5B. The maximum value (+0.5%) is achieved when the fraction of fully protected area is equal to 20%. Benefits are positive when the fraction ranges between the current value (8%) and 30%, and become negative outside this interval.

4 Discussion

We showed that the stocks of the three studied fish species (*Diplodus sargus, D. vulgaris* and *Epinephelus marginatus*) are currently overexploited in the north-western Mediterranean, and that fisheries sustainability could be reached either by reducing significantly fishing mortality in unprotected areas or by increasing the size of fully protected areas while keeping fishing constant.

Estimated current stock biomasses (B_0) are lower than B_{MSY} for the three studied species. However, the level of depletion (B_0>$0.5B_{\mathrm{MSY}}$ for all species) is such that all three species, and in particular *D. sargus*, have a good chance of recovery and avoid collapse if fishing pressure is reduced rapidly and substantially (Neubauer *et al.* 2013).

Achieving MSY requires that fishing mortality rates be significantly reduced (by one quarter for *D. sargus* and one half for *D. vulgaris* and *E. marginatus*). In practice, this could be achieved through a range of management tools including both input (e.g. gear restrictions, reduction of fishing capacity) and output controls (e.g. reduction in allowable catch). Additionally, we show that in the medium/long term (10–20 years), such a prospect of fishery recovery would simultaneously generate increases in stock biomass (17–42% depending upon species), fisheries catch (2–13%) and, consequently, revenues for the fishery sector (8% of increase in the total value of catch).

While the positive effects on stock biomass of the three studied species would be visible immediately after starting the recovery plan, our simulations suggest that the process of rebuilding catch to levels at least equal to the current ones would take more time (16 years for *D. sargus*, 11 years for *D. vulgaris* and 10 years for *E. marginatus*). During this relatively long transient period, catches may be substantially reduced, especially in the first year (about –20% for *D. sargus* and –45% for *D. vulgaris* and *E. marginatus*). To avoid excessive socioeconomic impacts (Worm *et al.* 2009) or unreported or illegal fishing (Agnew *et al.* 2009), specific measures should be targeted towards fishers during this transient period.

Enforcement of fishing effort control in unprotected areas may be difficult to put into practice, especially in the case of small scale and recreational fisheries in coastal areas. Therefore, an effective alternative strategy could be to rely on already designated MPAs and extend the coverage of full protection within the existing MPA network. Increasing the relative size of fully protected areas within multiple-use MPAs, while keeping fishing mortality rate outside MPAs at current levels, can generate positive conservation effects (increase in stock biomass) for the three coastal species. Positive effects of the size of fully protected areas on fish biomass are known (Claudet *et al.* 2008) and can be related to better inclusion of fish home ranges (Di Franco *et al.* 2018) and increase in self-recruitment through larger proportions of retained larvae (Botsford, Micheli & Hastings 2003).

Benefits on catch are species-specific and dependent on the size of the fully protected area. In our case, they are greater for the species with the longest dispersal distance (*D. vulgaris*) than for those with a narrower dispersion range (*D. sargus* and *E. marginatus*). Given that the three studied coastal species have limited adult movement (La Mesa *et al.* 2011; Di Franco *et al.* 2018), the relatively short pelagic larval phase represents the primary opportunity for dispersal and connectivity (Di Franco *et al.* 2012; Andrello *et al.* 2013; Pujolar *et al.* 2013).

Ensuring that the loss in fishing grounds is offset by gains in catch (Halpern & Warner 2003; Gaines *et al.* 2010) is key for successful fisheries management with MPAs. Here, we showed that an increase of size of fully protected areas within existing multiple-use MPAs can generate positive effects for *D. sargus* and *D. vulgaris*, both in terms of stock biomass (+2% and +15%, respectively) and catch (+0.5% and +4%, respectively), for levels of full protection between 10% and 40%, respectively, of the total protected area. In the case of *E. marginatus*, increasing the relative size of the fully protected area would not generate positive effects on catch. However, given that adult spillover was not taken into account in this study, the actual benefits on catch may be underestimated. In any case, the economic viability of the fishery (expressed in terms of total value of catch) would be preserved.

Despite the ecological and commercial interests of the studied coastal species, to our knowledge our study is the first modelling effort of its kind, fully integrating the biological and demographic characteristics of the species into a single model. We have shown that strong conservation benefits can be obtained either through non-spatial regulations, by reducing fishing effort in unprotected areas, or via area-based management strategies, by increasing the size of fully protected areas within existing MPAs (hence not increasing the size of MPAs overall). We believe this can contribute greatly to more effective management of those vulnerable species and help reconcile conservation and fisheries goals (Halpern et al. 2010).

Authors’ contributions

PM and MB conceived the ideas and designed methodology. MB developed and ran the models, with support from PM, LM, MG and RC. ADF, AC and PG contributed in acquisition and interpretation of data. MB led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

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Figure legends

Figure 1. Study area and spatial distribution of suitable habitat in each of the 949 model cells considered in this study.

Figure 2. Stock biomass and catch of the three studied species (colour coded) as functions of fishing mortality rate *F*. Biomass and catch values (averaged over the last 10 years of a 50-years simulation) are normalized with respect to baseline values for each species (obtained at current fishing mortality rate, F_0). *F* was varied by applying different multipliers to the baseline, namely: 0, 0.1, 0.2, 0.25, 0.33, 0.5, 0.625, 0.75, 1, 1.5, 2, 3, 4, and 5. Maximum Sustainable Yield for each species and the corresponding values of stock biomass (B_{MSY}) are pointed out by coloured dots near the axes, while the corresponding levels of fishing mortality (F_{MSY}) are indicated by black-bordered circles. The white, black-bordered circle identifies the baseline scenario.

Figure 3. Temporal dynamics of (A) stock biomass and (B) catch for the three studied species under a MSY management (i.e., with fishing mortality rate set to F_{MSY}). Biomasses are normalized with respect to B_{MSY}, while catches are normalized with respect to their estimated current value C_0.

Figure 4. Stock biomass and catch of the three studied species as functions of the percent coverage of fully protected areas within existing MPAs. Biomass and catch values (averaged over the last 10 years of a 50-years simulation) are normalized with respect to baseline values for each species (obtained by setting the proportion of fully protected areas over the overall size of MPAs to its present value, A_0). The white, black-bordered circle identifies the baseline scenario.

Figure 5. Percent change of the total value of catch (compared to its present value) as a function of (A) fishing mortality rate and (B) percent coverage of fully protected areas within existing MPAs.

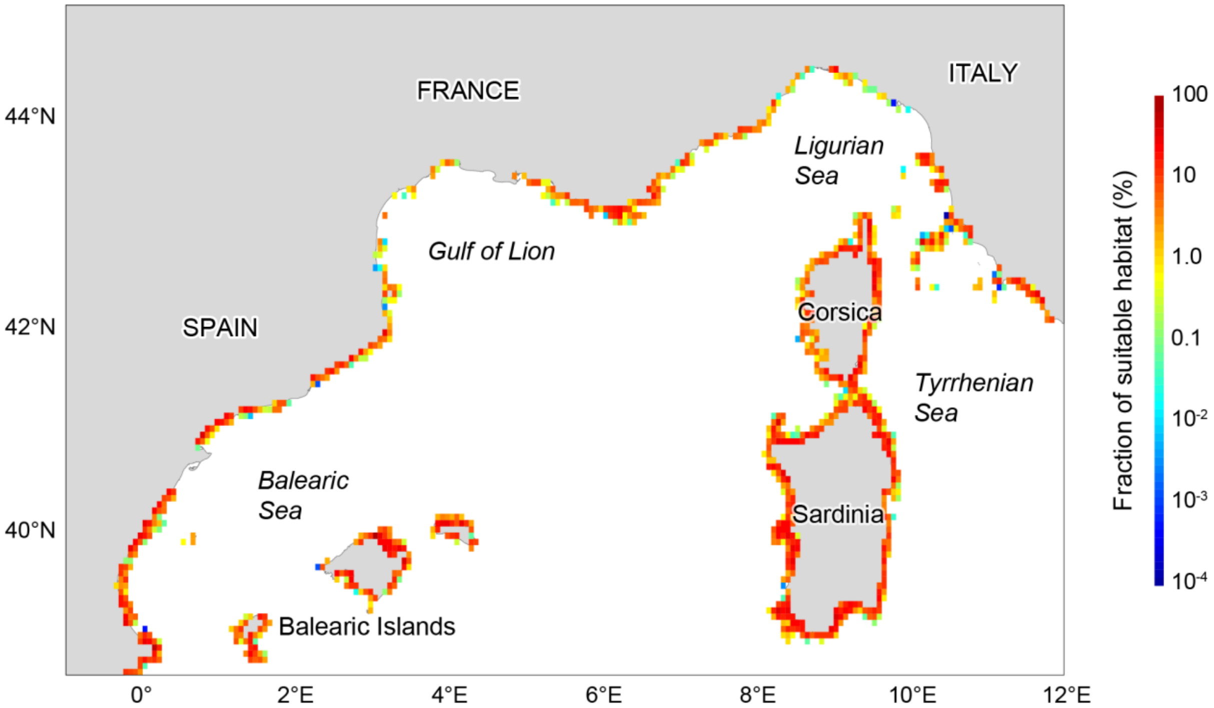

FRANCE
ITALY
SPAIN
Ligurian Sea
Gulf of Lion
Corsica
Tyrrhenian Sea
Balearic Sea
Sardinia
Balearic Islands
44°N
42°N
40°N
0°
2°E
4°E
6°E
8°E
10°E
12°E
Fraction of suitable habitat (%)
100
10
1.0
0.1
10^{-2}
10^{-3}
10^{-4}

Fig. 1

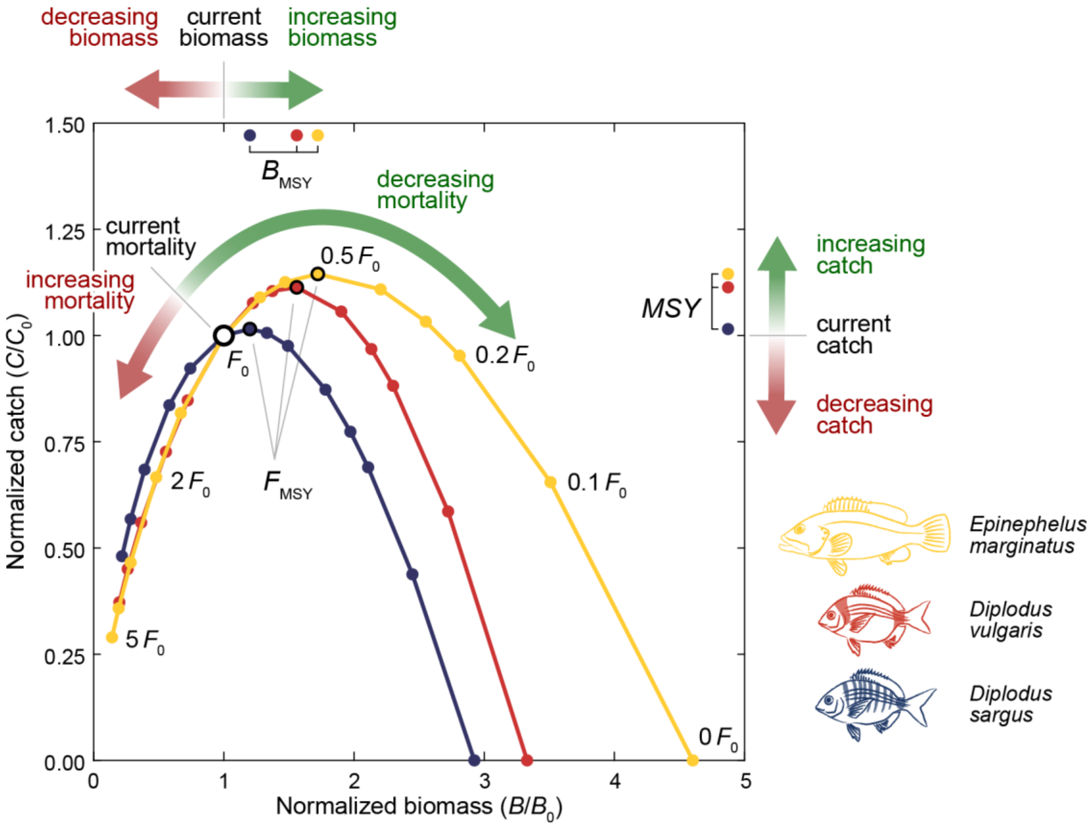

decreasing biomass
current biomass
increasing biomass
1.50
1.25
1.00
0.75
0.50
0.25
0.00
B_{MSY}
decreasing mortality
current mortality
increasing mortality
0.5 F_0
F_0
0.2 F_0
2 F_0
F_{MSY}
0.1 F_0
5 F_0
0 F_0
MSY
increasing catch
current catch
decreasing catch
Epinephelus marginatus
Diplodus vulgaris
Diplodus sargus
Normalized catch (C/C_0)
0
1
2
3
4
5
Normalized biomass (B/B_0)

Fig. 2

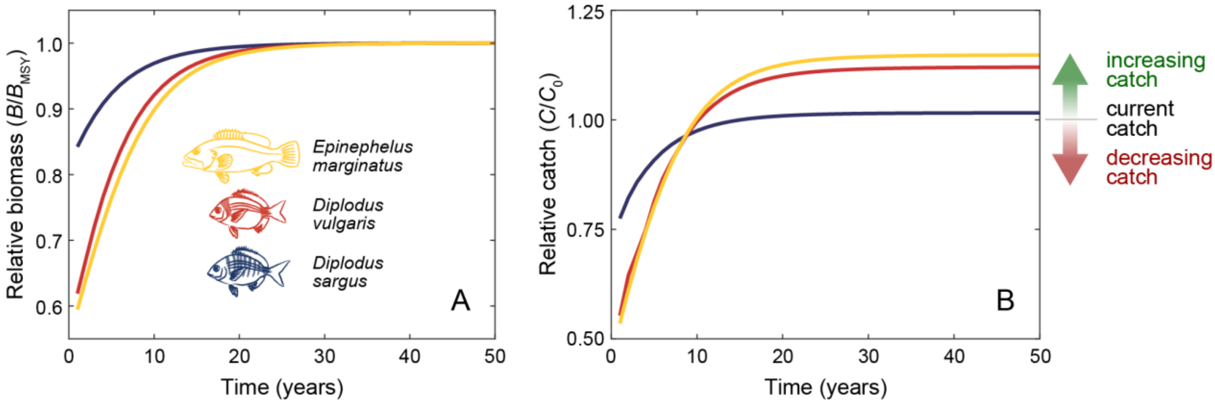

Relative biomass (B/B_{MSY})
Time (years)
Epinephelus marginatus
Diplodus vulgaris
Diplodus sargus
A
Relative catch (C/C_0)
Time (years)
B
increasing catch
current catch
decreasing catch

Fig. 3

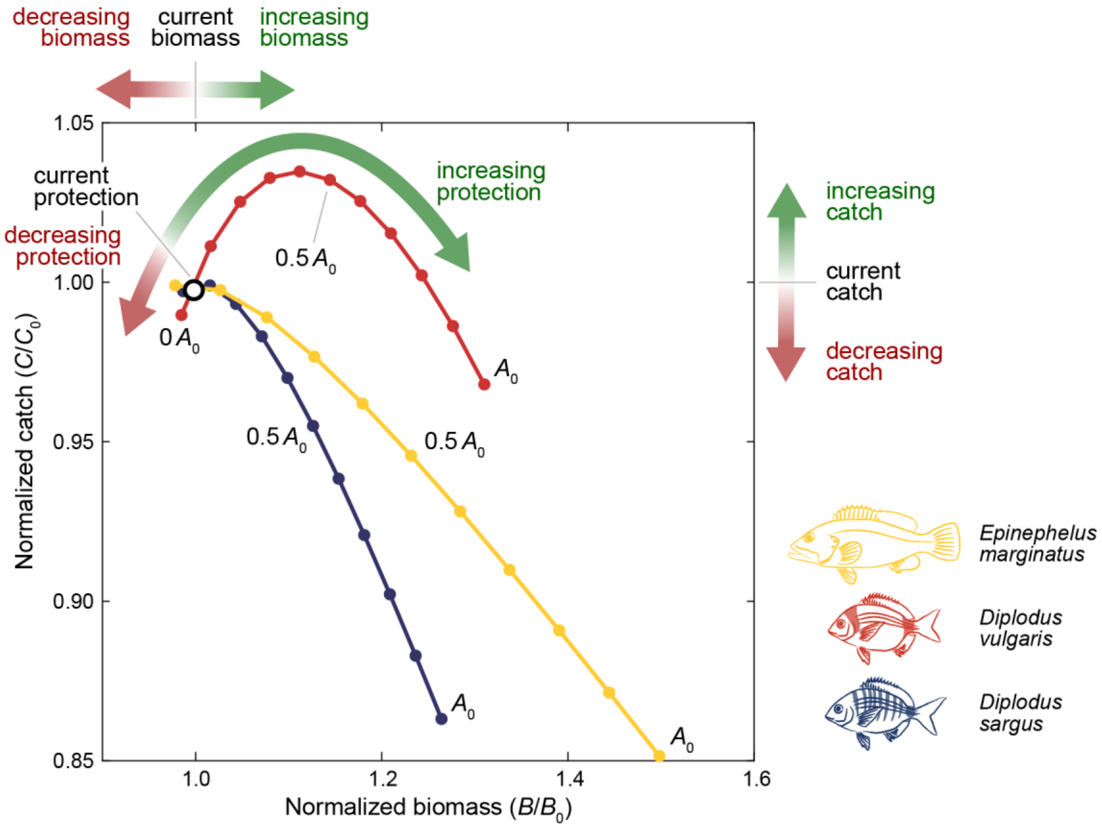
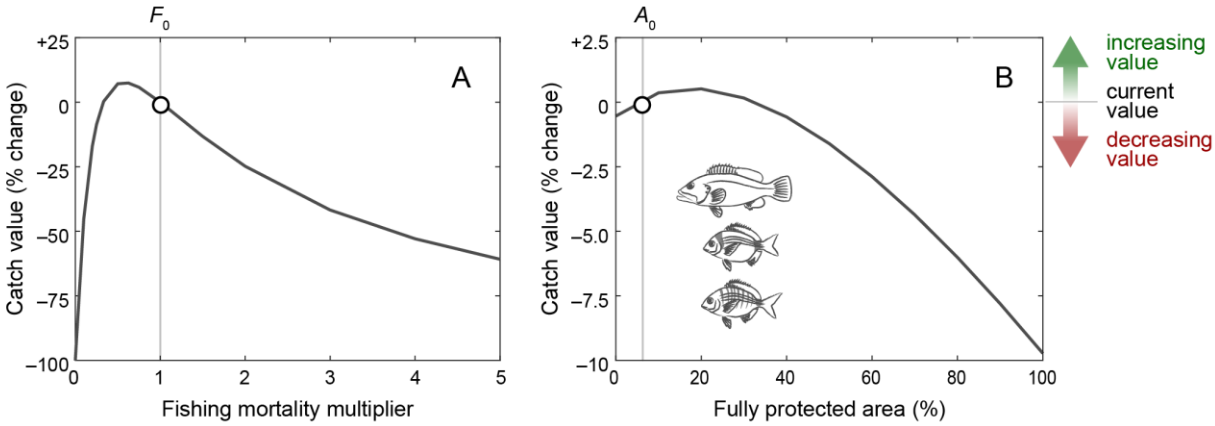

decreasing biomass
current biomass
increasing biomass
current protection
decreasing protection
increasing protection
increasing catch
current catch
decreasing catch
0.5 A_0
0 A_0
A_0
Normalized catch (C/C_0)
Normalized biomass (B/B_0)
1.05
1.00
0.95
0.90
0.85
1.0
1.2
1.4
1.6
Epinephelus marginatus
Diplodus vulgaris
Diplodus sargus

Fig. 4

536



F_0
+25
0
−25
−50
−75
−100
Catch value (% change)
0
1
2
3
4
5
Fishing mortality multiplier
A
A_0
+2.5
0
−2.5
−5.0
−7.5
−10
Catch value (% change)
0
20
40
60
80
100
Fully protected area (%)
B
increasing value
current value
decreasing value

537 **Fig. 5**