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Multi-zone marine protected areas: assessment of ecosystem and fisheries benefits using multiple ecosystem models

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Abstract

The current alarming state of many coastal ecosystems and fisheries calls for the development of tools to support recovery of exploited stocks, ensure their sustainable exploitation and protect marine ecosystems. Multi-zone Marine Protected Areas (MPAs) are often advocated to reconcile conservation and fisheries benefits. However, while there is a consensus about the ecological benefits whether such types of MPAs can really provide both benefits is still uncertain. Here, we analysed three existing Northwestern Mediterranean multi-zone MPAs (Cerbère-Banyuls, Cap de Creus and Medes Islands) using a comparative temporal ecosystem modelling approach to assess their effectiveness over time in recovering marine resources and ecosystem in the area. Our results showed differences in the ecological effectiveness of the three MPAs, potentially driven by MPA design, management and implementation features. Temporal increases of benefits were small, whenever detected, and showed slight recoveries of some target species and ecological indicators, mostly in Cerbère-Banyuls and Medes Islands MPAs. Our results confirm the benefits of protection to coastal marine resources and ecosystems when MPAs are enforced but highlight the current limitations of the three MPAs due to their small size and the significant impacts of small-scale and recreational fisheries. This study illustrates the capability to evaluate protection effects of small multi-zone MPAs with an ecosystem modelling perspective and represents the baseline to develop future scenarios of alternative management options to foster ecosystem recovery and resource rebuilding in the studied MPAs.

Keywords: Northwestern Mediterranean Sea; Ecopath model; Marine Protected Areas, Small-scale fisheries; Recreational fisheries, Ecological indicators.

1. Introduction

Coastal and marine ecosystems provide multiple ecosystem goods and services ([Costanza, d'Arge et al. 1997](#); [Martínez, Intralawan et al. 2007](#)). However, the provision of these ecosystem goods and services can be degraded as human demand for resources and the associated activities increase in number, frequency, magnitude and spatial coverage ([MEA 2005](#); [Worm, Barbier et al. 2006](#); [Pauly and Zeller 2016](#)). Currently, marine ecosystems are under increasing threat from a range of stressors including overfishing, climate change, biological invasions, pollution, aquaculture and habitat degradation, directly or indirectly caused by multiple human activities ([Costello, Coll et al. 2010](#); [Halpern, Frazier et al. 2015](#)). The relationships and interactive effects of multiple stressors are highly complex, and largely unknown or even highly uncertain ([Crain, Halpern et al. 2009](#); [Côté, Darling et al. 2016](#)).

In recognition of the complex interrelationship between marine organisms, multiple human activities, and ecosystem functioning and services, more comprehensive frameworks to manage marine ecosystems are required, such as the ecosystem-based management (EBM) ([Leslie and McLeod 2007](#); [Long, Charles et al. 2015](#)). The overall objective of this approach is to maintain ecosystems as a whole “in healthy, productive and resilient conditions allowing them to provide the needed ecosystem services to society” ([McLeod, Lubchenco et al. 2005](#)). Within this context, Marine Protected Areas (MPAs hereafter) are considered to be a key management tool for EBM as they are expected to mitigate human impacts, recover exploited resources and support their sustainable use, conserve or restore habitats and biodiversity, maintain and enhance ecosystem services and reduce conflicts between users ([Lester, Halpern et al. 2009](#); [Halpern, Lester et al. 2010](#); [Leenhardt, Low et al. 2015](#)). A MPA is a discrete area of the sea established to achieve the long-term conservation of natural resources therein ([Claudet 2011](#)). The level of protection within MPAs can vary from areas of full protection (FPAs, also called marine reserves or no-take areas), where all extractive activities (e.g., fishing) are prohibited while some non-extractive activities (e.g. diving) can be allowed; to areas of partial protection (PPAs), where some human activities (e.g. small-scale and recreational fisheries) are allowed but strictly regulated ([Horta e Costa, Claudet et al. 2016](#)). Conservation benefits of MPAs

largely vary due to their intrinsic features, including their design, the level of enforcement, their conservation goals and organization ([Guidetti, Milazzo et al. 2008](#); [Di Franco, Thiriet et al. 2016](#); [Giakoumi, McGowan et al. 2018](#); [Scianna, Niccolini et al. 2019](#)). In addition, networks of MPAs could provide greater benefits than the sum of individual MPAs benefits ([Gaines, White et al. 2010](#); [Grorud-Colvert, Claudet et al. 2014](#)). A network of MPAs is a system of individual MPAs that operates synergistically to achieve ecological objectives more effectively than individual MPAs could fulfil alone ([IUCN-WCPA 2008](#)).

Within the EBM, ecological modelling has been proven to be a suitable tool to merge available ecological information into a coherent form to obtain insights about how ecosystems are structured and impacted by human activities and environment, and delivering ecosystem services ([Link 2010](#); [Christensen and Maclean 2011](#)). Although quantitative models have become a key tool to assess the extent to which MPAs are able to achieve their conservation targets ([Fulton, Bax et al. 2015](#)), the understanding of MPA effects on ecological processes and ecosystem function is still scarce ([Cheng, Altieri et al. 2019](#)). For instance, it is mostly unknown if protection benefits inside fully protected areas extend to partially areas and beyond, and if those benefits have the potential to cascade through the ecosystem components. There is little evidence that multi-zone MPAs (i.e., those including both FPAs and PPAs) provide fisheries benefits inside partially protected areas and it is unknown what are the drivers of multi-zone MPAs.

Although considered a hotspot of biodiversity ([Coll, Piroddi et al. 2010](#)), the Mediterranean Sea has been largely altered by anthropogenic stressors for a long time ([Coll, Piroddi et al. 2012](#); [Micheli, Halpern et al. 2013](#); [Halpern, Frazier et al. 2015](#)), that have determined dramatic changes on its ecosystems especially during the last 50 years ([Azzurro, Moschella et al. 2011](#); [Maynou, Sbrana et al. 2011](#); [Piroddi, Coll et al. 2017](#)). In order to address this urgent conservation challenge, a multitude of MPAs (including all forms of nationally and internationally established protected areas) have been implemented in the Mediterranean Sea in the last decades. Within this context, several studies have modelled MPAs in the Mediterranean Sea with the aim to evaluate their effects on the ecosystem and the impacts of small-scale and recreational fisheries on marine resources ([Albouy, Mouillot et al. 2010](#); [Libralato, Coll et al. 2010](#); [Valls, Gascuel et al. 2012](#); [Prato, Barrier et al. 2016](#); [Vilas, Coll](#)

[et al. Submitted](#)). However, none of the existing studies modelled simultaneously the multi-zone nature of Mediterranean MPAs nor a series of neighbouring MPAs, potentially constituting a network of protected areas. In addition, the study of the historical effects of MPAs has not been pursued due to difficulties to gather historical data about protection effects. In this study we modelled three relatively well studied MPAs from the Northwestern Mediterranean Sea with a historical perspective: Cerbère-Banyuls, Cap de Creus and Medes Islands. Our aims were to: (1) quantify the main structural and functional traits of the three multi-zone MPAs, and (2) assess how exploitation and protection regimes affected these marine ecosystems over time, using a comparative approach to identify commonalities and differences between MPAs.

2. Materials & Methods

2.1. Study area

We studied three MPAs in the Northwestern Mediterranean Sea: Cerbère-Banyuls (France), Cap de Creus (Spain) and Medes Islands (Spain) (Fig. 1). These MPAs include one Fully Protected Area (FPA) and one (Cerbère-Banyuls and Medes Islands) or more (Cap de Creus) Partially Protected Areas (PPAs) (Fig. 1). While in the FPAs all extractive activities are forbidden, in PPAs both small-scale and recreational fisheries are allowed except in Medes Islands MPA, where recreational fisheries are prohibited.

The three MPAs comprise a similar depth range between ca. 0 and 60 meters, and several vulnerable species such as the Mediterranean seagrass (*Posidonia oceanica*), dusky grouper (*Epinephelus marginatus*) and red coral (*Corallium rubrum*), can be found in these MPAs. In addition to the protected zone scheme (FPA and PPA), we included in our study the immediate unprotected areas (UPAs) surrounding each MPA (Fig. 1 and Table 1), in order to add the adjacent non-protected waters where some species found inside the MPAs can theoretically forage and move to ([Di Lorenzo, Claudet et al. 2016](#)). The boundaries of the unprotected areas were selected taking into account that they had similar ecological and oceanographic features (i.e., same depth range and habitats) to the MPA, and they constituted part of the Parc (Cap de Creus and Medes Islands) and/or were adjacent to the MPA (Cerbère-Banyuls) (Fig. 1).

2.2. Modelling approach – Ecopath and Ecosim

The three MPA models were developed using Ecopath with Ecosim approach (EwE 6.6 version) ([Christensen and Walters 2004](#); [Christensen, Walters et al. 2008](#)). We used the static Ecopath model to provide a “snapshot” of the ecosystem in terms of flows and biomasses for a defined period of time. To develop the food-web model we used functional groups, which consist of ontogenetic fractions of a species, single species or groups of species sharing common ecological traits. The baseline models of each MPA were developed taking into account the contribution of each zone to the whole MPA based on a previous study ([Vilas, Coll et al. Submitted](#)). Afterwards, we used the time-dynamic model Ecosim, which describes the temporal dynamics of species biomass and flows over time by

accounting for changes in predation, consumption rate, fishing and the environment (Walters, Christensen et al. 1997; Christensen and Walters 2004). A short description of the modelling approach is given in the Supplementary Online Material (SOM hereafter) 1; and a detailed information of the EwE approach is documented in Christensen and Walters (2004) and Heymans, Coll et al. (2016).

2.3. Functional groups and input data

2.3.1. Functional groups

A meta-web structure defined for the Western Mediterranean Sea model (Coll, Steenbeek et al. 2019) developed under the Safenet Project (Sustainable fisheries in EU Mediterranean waters through a network of MPAs) was used and adapted to our study area. Specifically, we removed those functional groups (hereafter FGs) that did not occur in the study area (i.e., FGs that include species that inhabit deeper waters or that live offshore). The final food-web structure contained 64 FGs for Cerbère-Banyuls MPA and 67 FGs for Cap de Creus MPA and Medes Islands MPA (SOM 2).

2.3.2. Ecopath input data and balancing procedure

The Ecopath models represented a situation of the Cerbère-Banyuls MPA for 2013, while the Cap de Creus MPA and the Medes Islands MPA represented an average situation for the period 2005-2008 and 2000-2004, respectively. These periods of time to model the MPAs were selected considering the best available biomass data (SOM 2 for details on the parameterization of each functional group). Input parameters were obtained using similar procedures as those documented in Coll, Palomera et al. (2006), Corrales, Coll et al. (2015), Piroddi, Coll et al. (2015) and Vilas, Coll et al. (Submitted). Biomass and catch data were scaled using data from models of each management unit (FPA, PPA and UPA) developed previously (Coll, Vilas et al. 2019; Vilas, Coll et al. Submitted) by the percentage of the area of each management unit to the whole MPA (SOM 2 for details on parameterization of each functional group).

Production (P/B, year⁻¹) and consumption (Q/B, year⁻¹) rates were estimated through the application of empirical equations (Heymans et al., 2016), taken from literature or from

other models developed in the Northwestern Mediterranean Sea ([Coll, Palomera et al. 2006](#); [Corrales, Coll et al. 2015](#); [Coll, Vilas et al. 2019](#)).

The trophic information to populate the diet matrix was compiled using published studies on stomach content analyses, giving preference to local or surrounding areas (SOM 2). We used the Diet Matrix Calculator ([Steenbeek 2018](#)), a custom-built extraction tool that automatizes the process of selecting and scaling diet data. Drawing on a large library of published diet studies, the Diet matrix calculator tool selects the most likely suitable diet studies for a specific model area, based on a range of criteria, and generates a diet composition matrix with accompanying pedigree index for each predatory functional group. Due to the small sizes of the investigated MPA and the capacity of some species to move greatly ([Gell and Roberts 2003](#); [Grüss, Kaplan et al. 2011](#)), we set a fraction of the diet composition as import for all MPA based on the time that these species spent foraging outside the areas and biological and ecological traits such as size, behavior and ecology of species of each functional group (Table 1 in SOM 3).

To achieve mass-balance of the baseline models, we implemented a manual procedure following a top-down approach (starting from the groups with higher trophic levels) modifying appropriate input parameters (Table 2 in SOM 3) and following the best practice guidelines provided in the literature ([Heymans, Coll et al. 2016](#)). In addition, the PREBAL analysis was used to ensure that the input data complied with general ecologic principles and to guide modifications through the balancing procedure ([Link 2010](#)). Detailed explanations of these procedures are given in SOM 3 and the final diet matrixes are provided in SOM 4.

The pedigree routine ([Christensen and Walters 2004](#)) was used to evaluate the quality of the models and the uncertainty associated with the input parameters and to guide modifications in balancing the model. A detailed explanation of this routine is given in SOM 3.

2.3.4. Ecosim historical time series fitting.

The model representing Cerbère-Banyuls MPA ecosystem during the 2013–2017 period was fitted to time series of historical data, while for Cap de Creus MPA and the Medes Islands MPA, the Ecosim model was fitted to time series of 2008-2017 and 2004-2017

periods, respectively (Table 1 in SOM 5). We used historical fishing effort trends to drive the fisheries of the models, while biomass and catch time series were used to calibrate the model and compare predicted to observed results (see SOM 5 for details), respectively, following previous studies in the Mediterranean Sea and best practices of EwE ([Coll, Navarro et al. 2013](#); [Heymans, Coll et al. 2016](#); [Corrales, Coll et al. 2017](#)).

The models were fitted by using the Stepwise Fitting Procedure ([Scott, Serpetti et al. 2016](#)). This procedure tests alternative hypotheses related to the impact of fishing, changes in predator-prey dynamics (vulnerabilities), changes in primary production (production anomalies) or all of the above together (Table 2) ([Mackinson, Daskalov et al. 2009](#); [Heymans, Coll et al. 2016](#)). This procedure estimates different vulnerability parameters and primary productions anomalies to improve model fits by comparing model predictions to observed data using the sum of squares (SS) statistics and to find the statistically “best fit” model based on Akaike’s Information Criterion (AIC), which penalizes the overparameterization of the model ([Mackinson, Daskalov et al. 2009](#); [Heymans, Coll et al. 2016](#)). In this study, the maximum number of parameters that could be estimated were 19 (Cerbère-Banyuls), 51 (Cap de Creus) and 48 (Medes Islands), respectively.

Finally, we manually evaluated a range of models fits with the lowest AIC values to identify the best final model, which led to a credible behavior ([Heymans, Coll et al. 2016](#)), following previous studies ([e.g., Corrales, Coll et al. 2017](#)).

2.4. Model analyses and ecological indicators

2.4.1. Ecological indicators of initial conditions

To analyze the food-web structure of the three MPAs, we used the biomasses of selected FGs, trophic flows and trophic levels (TLs) within the flow diagram. The TL was also used to analyze the ecological position of the FG of the three MPA models ([Lindeman 1942](#); [Stergiou and Karpouzi 2001](#))

Several ecological indicators were also computed to describe the structure and functioning of the ecosystems and were divided into four groups using the ECOIND plug-in ([Coll and Steenbeek 2017](#)):

(1) Biomass-based: calculated from the biomass of components included in the food-web model, they could provide valuable information to evaluate MPAs effectiveness ([Micheli, Halpern et al. 2004](#); [Claudet, Osenberg et al. 2008](#)). We included Total Biomass (TB, $\text{t}\cdot\text{km}^{-2}\cdot\text{year}^{-1}$), Biomass of Fish species (FB, $\text{t}\cdot\text{km}^{-2}\cdot\text{year}^{-1}$), and Kempton's Q diversity index (QI).

(2) Trophic-based: reflect the TLs for different groups of the food web, provide information on the structure of the ecosystem and are used to quantify the impact of fishing ([Rochet and Trenkel 2003](#)). We selected TL of the community (TLc), TL of the community, TL of the community including organisms with $\text{TL} \geq 3.25$ (TL3.25) and TL of the community including organisms with $\text{TL} \geq 4$ (TL4).

(3) Species and size-based: based on species traits and conservation status, they could offer insights of the effects of MPAs ([Claudet, Osenberg et al. 2010](#)). We selected biomass of IUCN-endangered species biomass in the community ($\text{t}\cdot\text{km}^{-2}\cdot\text{year}^{-1}$), mean length of fish in the community (ML, cm) and mean life span of fish community (MLS, year).

(4) Catch-based: based on catch, they reflect the fishing strategy of the fisheries and are used to quantify the impact of fishing ([Hilborn and Walters 1992](#); [Pauly, Christensen et al. 1998](#)). We included total catch (TC $\text{t}\cdot\text{km}^{-2}\cdot\text{year}^{-1}$), trophic level of the catch (TLc), intrinsic vulnerability index of catch (VI).

The Mixed Trophic Impact (MTI) analysis was performed to assess the direct and indirect impact in the food web that a hypothetical increase in the biomass of one functional group would have on the biomasses of all the other functional groups in the food web, including the fishing fleets ([Ulanowicz and Puccia 1990](#); [Christensen, Walters et al. 2008](#)). To evaluate the impact of small-scale fisheries on the MPAs, the MTI was used to quantify the direct and indirect impact of each fishery on the functional groups for the studied MPAs, and their potential competition and trade-offs between them.

Finally, to identify the key species within the ecosystem (both keystone and structuring species), we calculated the keystone index developed by [Valls, Coll et al. \(2015\)](#) of the most important reef functional groups. A keystone group is defined as a predator species

with a large and broad impact on the food web despite its low biomass ([Paine 1966](#); [Paine 1969](#); [Valls, Coll et al. 2015](#)).

2.4.2. Time-dynamic analyses

Once the fitting procedure was completed, we used the best fit model to examine biomass and catch time series predicted by the model to explore the dynamics of selected functional groups. We selected four target species due to their role in terms of biomasses and commercial interest: the common two-banded seabream (*Diplodus vulgaris*), white seabream (*D. sargus sargus*), common dentex (*Dentex dentex*) and groupers (this group was mainly represented by *Epinephelus marginatus*). These species play an important role in the ecosystem as high trophic level predators or intermediate trophic species and are of great importance in small-scale and recreational fisheries. In addition, the previous selection of ecological indicators from the ECOIND plug-in was used to describe ecological changes in the ecosystem over time ([Coll and Steenbeek 2017](#)).

The Monte Carlo (MC) uncertainty routine and the ECOSAMPLER plug-in were employed to assess the impact of uncertainty in Ecopath input parameters on Ecosim simulations ([Heymans, Coll et al. 2016](#); [Coll and Steenbeek 2017](#); [Steenbeek, Corrales et al. 2018](#)). We ran 500 MC simulations based on the sensibility of Ecopath input parameters obtained from the pedigree routine (SOM 3). Results from the MC simulations were used to plot the confidence intervals of the selected ecological indicators in Ecopath and to plot the 5th and 95th percentile confidence intervals for the Ecosim outputs. Finally, we used the Spearman's rank correlation to evaluate the presence of significant trends (either increasing or decreasing) in model outputs (biomass, catch and ecological indicators), following previous studies ([e.g., Corrales, Coll et al. 2017](#)).

3. Results

3.1. Structure and functional traits of protected areas

The pedigree index values of the three MPA models showed similar values, ranging from 0.41 to 0.51. The highest pedigree values were obtained for Cerbère-Banyuls (0.51), followed by Medes Islands (0.45) and Cap de Creus (0.41).

The visualization of trophic links and flows between functional groups highlighted the complexity of these coastal ecosystems due to the large number of trophic links between functional groups and the important role of detritus (FG 65) and other macro-benthos (FG 51) in transferring energy up to the food web (Fig. 2).

The functional groups of the models ranged from trophic level (TL) of 1 for primary producers (FG. 60-67) and detritus groups (FG 66-67) to $TL = 4.2$ for both groups of dolphins (FG 1-2) (Fig. 2 and Table 3 in SOM 3). Invertebrates groups were classified with a TL between 2 and 3.5, with benthopelagic cephalopods (FG 41) showing higher TLs. Fish had TLs between 3 and 4, with the exception of salema (FG 33) and mugilidae (FG 34), which showed lower TL due to their herbivorous and detritivore behaviors. Overall, similar TLs were found for the three MPAs.

Total and fish biomass displayed similar patterns, with the highest biomass values found for Cerbère-Banyuls and Medes Islands and lower values in Cap de Creus (Fig. 3). Conversely, Kempton's Q Index presented a higher value in Cap de Creus, followed by Medes Islands and Banyuls-Cerbère (Fig. 3). With the exception of TL of the community 4, trophic based indicators also presented higher values for Cerbère-Banyuls, followed by Medes Islands and Cap de Creus (Fig. 3). Species and size-based indicators showed that Cerbère-Banyuls presented the highest values for ML of fish community and IUCN species B, followed by Cap de Creus and Medes Islands (Fig. 3), while MLS of fish community was higher in Cap de Creus, followed by Cerbère-Banyuls and Medes Islands (Fig. 3). Total catch showed similar values between Cap de Creus and Medes Islands, while Cerbère-Banyuls had the lowest value (Fig. 3). TL of the catch and the Intrinsic Vulnerability Index presented similar values in Cerbère-Banyuls and Cap de Creus, while Medes Islands had the lowest value (Fig. 3).

The keystone index identified groupers (FG 27), common dentex (FG 24), other commercial medium demersal fishes (FG 31) and non-commercial medium demersal fishes (FG 32) as keystone species in the three MPAs, followed by red scorpionfish (*Scorpaena scrofa*) (FG 25) and Scorpaenidae (FG 26) (Fig. 4).

The MTI analysis highlighted that the small-scale fisheries had the most widespread negative impact on many FG of all MPAs, especially in Cerbère-Banyuls, while the impact of recreational fisheries was more prominent in Cap de Creus and Medes Islands (Fig. 5). This analysis showed strong negative impact of fisheries on target species such as other large pelagic species (FG 8), common dentex (FG 24), and groupers (FG 27) while competitors or preys of those species may be positively impacted such as white seabream (FG 22), common two-banded seabream (FG 23) and brown meagre (*Sciaena umbra*) (FG 28) (Fig. 5). Results also highlighted that each fishery had a negative impact on itself due to self-competition for resources according to the MTI results and that competition between fisheries was complex (Fig 5). While small-scale fisheries had positive impacts on recreational fisheries in Cerbère-Banyuls MPA, recreational fisheries had slightly negative impacts on small-scale fisheries (Fig. 5a). On the contrary, small-scale fisheries had negative impacts on recreational fisheries in Cap de Creus and Medes Islands MPAs, while recreational fisheries had positive impacts on the small-scale fisheries (Fig. 5b and c).

3.2. Ecological impacts of protection over time

The best-fitted food web temporal models were obtained when trophic interactions, fishing, and primary production anomaly were included in the model configuration for all MPAs (Step 8 in Table 3). However, for all MPAs the best model was not able to reproduce the trends of white seabream, common two-banded seabream, common dentex and groupers satisfactorily, which we selected as target groups of this study. Therefore, we moved through the fitting procedure analysis to find the model that was able to reproduce the trends of most of the groups (and specifically the target groups) and was highly significant. We finally choose a model fit with 12 vulnerabilities and 3 spline points, 15 vulnerabilities and 3 spline points, and 11 vulnerabilities and 5 spline points for Cerbère-Banyuls, Cap de Creus and Medes Islands models respectively, as the best options (Step 8 in Table 3).

Observed biomass and catch time series were satisfactorily reproduced by model predictions for most of the target groups (Fig. 6 and SOM 6, 7 and 8) when using the chosen fitted model. The temporal models showed a non-significant biomass pattern of white seabream in Cerbère-Banyuls, firstly decreasing and later increasing, and in Cap de Creus, where firstly increased and later decreased, while in Medes Islands it presented a

significant decreasing biomass trend (Fig 6). The models showed similar biomass patterns for common two-banded seabream, highlighting a non-significant trend in Cerbère-Banyuls and Cap de Creus and a significant decreasing trend in Medes Islands (Fig. 6). We observed a significant decreasing biomass trend of common dentex in Cerbère-Banyuls, while in Cap de Creus and Medes this group showed non-significant biomass trends (Fig. 6). The results highlighted a non-significant biomass pattern of groupers in Cerbère-Banyuls, while they significantly increased in Cap de Creus and Medes Islands (Fig 6). However, in Medes Islands the model did not capture well the overall declining trend of observations for groupers biomass.

Regarding the temporal changes of ecological indicators, the Kempton's Index showed a non-significant trend in Cerbère-Banyuls and Cap de Creus, while in Medes Islands, the Kempton's Index significantly increased during the simulated period (Fig. 7). The TL of the community presented a non-significant pattern in Cerbère-Banyuls and Cap de Creus, while in Medes Islands significantly decreased (Fig. 7). IUCN species biomass significantly declined in Cerbère-Banyuls, while in Cap de Creus and Medes Islands presented non-significant patterns (Fig. 7). Total catches highlighted non-significant trends in Cerbère-Banyuls and Cap de Creus (in the last one firstly increasing and then decreasing), while in Medes Islands, total catches significantly decreased (Fig. 7).

4. Discussion

4.1. Structure and functional traits of protected areas

Overall, ecological indicators showed similar patterns, with the highest values in Banyuls-Cerbère MPA, followed by Medes Islands and Cap de Creus MPAs. This may be related to differences in the ecological effectiveness of the three MPAs, which is partly explained by MPA design, management and implementation features (e.g. extent of area protected, fully protected area enforcement, time since protection, MPA design) ([Claudet, Osenberg et al. 2008](#); [Guidetti, Milazzo et al. 2008](#); [Edgar, Stuart-Smith et al. 2014](#); [Giakoumi, Scianna et al. 2017](#); [Di Franco, Plass-Johnson et al. 2018](#)). The lack of enforcement is one of the most relevant issues concerning MPAs in the Mediterranean context ([Fenberg, Caselle et al. 2012](#)). Within this context, while Cerbère-Banyuls and Medes MPAs are considered to

have a high level of enforcement ([Sala, Ballesteros et al. 2012](#); [Giakoumi, Scianna et al. 2017](#)), Cap de Creus has been considered a MPA only on paper due to a lack of sufficient enforcement ([Lloret, Zaragoza et al. 2008](#); [Lloret, Zaragoza et al. 2008](#)). In fact, total catches were higher in Cap de Creus than in the other two MPAs; with Cerbère-Banyuls presenting the lowest total catches. Total catches are expected to be higher in well designed and enforced MPAs ([Halpern, Lester et al. 2009](#); [Di Lorenzo, Claudet et al. 2016](#)), so our results could be explained by the allowed and/or real fishing effort and the small effects of these MPAs in PPAs and adjacent areas. [Horta e Costa, Claudet et al. \(2016\)](#) presented a novel classification system for MPAs by scoring allowed uses in each management unit based on their impacts on biodiversity. In this scale, Cerbère-Banyuls obtained a rate of 4.7, being a highly protected area and Medes 6.4 being less well protected ([Horta e Costa, Claudet et al. 2016](#)). Although Cap de Creus was not included in this study, we could expect a higher MPA index because of its smaller FPA and allowed uses. These features (smallest FPA and weak enforcement) and the fact that Cap de Creus is the newest (Table 1) MPA in the study, could suggest that the year of establishment, enforcement and the size of protected areas have a strong effect on MPA effects, in line with other studies ([Claudet, Osenberg et al. 2008](#); [Guidetti, Milazzo et al. 2008](#)).

Our results also indicated that both the small-scale and recreational fisheries can have a notable impact affecting organisms from lower to higher trophic levels. According to our study, small-scale fisheries had the largest negative ecological impacts. In fact, this fishery tends to develop its main activity inside the MPA or in surrounding areas ([Goñi, Adlerstein et al. 2008](#); [Stelzenmüller, Maynou et al. 2008](#)). Recreational fisheries seemed to have larger impacts in the Spanish MPAs (Cap de Creus and Medes Islands MPA), while they had a negligible impact in Cerbère-Banyuls MPA, although the impact of recreational fisheries could be similar than the small-scale fisheries, as highlighted in coastal waters of Cap de Creus by [Lloret, Zaragoza et al. \(2008\)](#). This could be related to cultural reasons, differences in enforcement of each MPA and/or the reliability of catch data between study sites. In addition, our results highlighted that despite the recreational fisheries seemed to have an overall lower negative impact, it had strong negative impacts on vulnerable species such as common dentex and groupers (this group was mainly represented by *Epinephelus marginatus*) in Cap de Creus and Medes Islands MPAs, in line of the observed impact of

small-scale and recreational fisheries on vulnerable species ([Lloret, Biton-Porsmoguer et al. 2019](#)), which is an important fact to consider when establishing management plans for these species. Our results also highlighted the competition for resources between both (small-scale and recreational) fisheries. Competition between both fisheries has become an important issue in coastal areas (especially in surrounding areas of MPAs), as they target similar species and fishing grounds ([Chuenpagdee and Jentoft 2018](#); [Lloret, Cowx et al. 2018](#)). In small and touristic areas such as the ones in the present study, a strong competition between small-scale and recreational fishers could be expected due to the current economic, societal and environmental challenges of the small scale fisheries and the increasing importance of recreational fisheries as a leisure activity in these areas ([Gómez and Lloret 2017](#)).

4.2. Ecological impacts of protection over time

The temporal dynamic model of Cerbère-Banyuls predicted an increasing biomass pattern of white seabream and common two-banded seabream while common dentex and groupers largely decreased. The decrease of common dentex and groupers could be related to the impact of fishing, although fishing effort did not increase during the study period and initial fishing mortalities were not high. The increase of both seabream species could be explained by the reduction of their predators and/or by some other processes (e.g., increase in resources). In fact, common dentex and groupers are the most important predators of these species in terms of total consumption.

In contrast, the temporal dynamic model of Cap de Creus predicted a decreasing biomass pattern of white seabream, common two-banded seabream and common dentex while groupers increased. These results suggest a recovery of groupers population in Cap de Creus ([Hereu Fina, Aspillaga Cuevas et al. 2017](#)), despite the low level of enforcement, that could cause biomass reductions of their preys such as both seabream species. Similarly, the temporal dynamic model of Medes Islands highlighted decreasing trends for white seabream, common two-banded seabream and common dentex, while groupers slightly increased. These results evidence that under protection the food-web effects can play an important role and there are winners and losers as a result of the new ecological state, where results of protection can include cascading effects of predators on preys species

([Edgar, Stuart-Smith et al. 2014](#); [Cheng, Altieri et al. 2019](#)) as well as an increase in competitive interactions ([Micheli, Halpern et al. 2004](#)).

Ecological indicators play an important role within the EBM framework as they are able to describe the condition of the ecosystem, its components and its functioning, to evaluate the impact of human activities in marine ecosystems and to inform management decisions ([Shin and Shannon 2010](#); [Tam, Link et al. 2017](#)). In our study, ecological indicators showed overall contrasting results, as some indicators indicated recovery (e.g. Kempton's Q in the three MPAs) while others showed degradation patterns (e.g., TL of the community in the three MPAs). These contrasting results could respond to the limitation in historical data to represent ecosystem dynamics in the study area and could indicate limited recovery of species and ecosystems in the three MPAs.

The most notable effects of MPAs are an increase in abundance, average size and biomass inside protected areas ([Lester, Halpern et al. 2009](#); [Giakoumi, Scianna et al. 2017](#)). Due to the increased density inside the MPA, adults and juveniles may move to adjacent areas (spillover effect) and therefore, higher fisheries yields in adjacent areas are expected ([Goñi, Adlerstein et al. 2008](#); [Di Lorenzo, Claudet et al. 2016](#)). Larger abundance and sizes could entail higher reproductive potential of species, and eggs and larvae from recovered populations could then be exported to external unprotected locations, including adjacent ones ([Gell and Roberts 2003](#); [Harrison, Williamson et al. 2012](#)). In addition, MPAs could restore ecosystem functioning ([Cheng, Altieri et al. 2019](#)). Although some of our temporal results showed potential recoveries (some target species, increases of predators and declines of prey, and some ecological indicators), our study evidences that they are still far from what we would expect from the temporal protection effects of MPAs, highlighting an overall modest historical positive effect of protection on these MPAs. These results are in line with [Hereu Fina, Aspillaga Cuevas et al. \(2017\)](#), which have pointed out illegal fishing as one of the main reasons for a lack of recovery and have called for further enhance the enforcement in Mediterranean MPAs.

4.3. Data gaps, limitations and uncertainties

Overall the input data used was of acceptable quality compared to the distribution of pedigree values in other existing models ([Morissette 2007](#); [Lassalle, Bourdaud et al. 2014](#)), although they are among the lowest values in the Mediterranean Sea ([Corrales, Coll et al. 2015](#)). This could be due to the challenges of modelling small and local coastal areas, where specific data is a strong requirement. In fact, limitations of available data are a common concern for most Mediterranean MPAs ([Prato, Barrier et al. 2016](#); [Vilas, Coll et al. Submitted](#)) despite there are monitoring programs inside these MPAs.

In general, there is a lack of biomass estimates for many functional groups, especially regarding benthic invertebrates. One of the main hurdles to evaluate the impact of fishing activities is to obtain realistic estimates of total catch (official and Illegal, Unregulated and Unreported) ([Pauly, Ulman et al. 2014](#); [Pauly and Zeller 2016](#)). In our study this challenge is even higher due to the importance of recreational fishers and illegal fishing in coastal areas, specifically in MPA and surrounding areas (e.g., [Lloret, Zaragoza et al. 2008](#); [Ben Lamine, Di Franco et al. 2018](#)).

The capacity of ecosystem models to replicate trends increases with data availability and quality ([Giron-Nava, James et al. 2017](#)). Within this context, in Cerbère-Banyuls a maximum of four data points of biomass per functional group were available for the fitting procedure. The Medes Islands MPA model had the longest available time series, and thus presented a better and more informative fit, while the Cap de Creus MPA model showed an intermediate situation. This limited the capability to calibrate and validate the models and to track ecosystem dynamic status. Therefore, our results highlight the need to further monitor MPAs within the Mediterranean Sea and should inform future scientific research objectives in the area.

Despite these limitations, the models presented in this study were constructed using the best available data, including several sources of information including unpublished and *ad hoc* field data, expert knowledge and published data, and following the best practices in ecosystem modelling development ([Heymans, Coll et al. 2016](#)). The present models represent the first attempt to develop temporal MPA models that are nearly located and confirm the capability of EwE to evaluate protection effects within an ecological perspective in small MPAs even in cases of data limitations.

4.4. Concluding remarks

Human activities have been concentrated in coastal ecosystems, resulting in a major modification of these marine areas ([Halpern, Frazier et al. 2015](#)). Since coastal areas account for a large amount of the global ecosystem services ([Costanza, d'Arge et al. 1997](#)), there is a need for the recovery of coastal marine ecosystems in order to ensure their ecological role. In addition, in recent decades, marine ecosystems have been increasingly impacted by other stressors, directly or indirectly induced by multiple anthropogenic activities ([Halpern, Frazier et al. 2015](#)). For example, sea warming and the spread of non-indigenous species represent a major challenge for conservation in Mediterranean ecosystems, especially in the Eastern Mediterranean Sea ([Lejeusne, Chevaldonné et al. 2010](#); [Katsanevakis, Coll et al. 2014](#)). Organisms and ecosystems that have been already impacted by fishing are more vulnerable to these additional stressors ([Occhipinti-Ambrogi and Savini 2003](#); [Poloczanska, Burrows et al. 2016](#)). MPAs, when properly managed, have demonstrated to be an effective tool to protect target species and habitats to local stressors such as fishing ([Sala, Giakoumi et al. 2017](#)), while their role in promoting resilience to regional and global stressors such biological invasions and climate change is debated ([Giakoumi and Pey 2017](#); [Roberts, O'Leary et al. 2017](#); [Giakoumi, Pey et al. 2019](#)). In highly impacted and crowded areas like the Northwestern Mediterranean Sea, the establishment of well-designed networks of well-enforced MPA is necessary to achieve “clean, healthy and productive” oceans (Good Environmental Status) according to the Marine Strategy Framework Directive ([Fenberg, Caselle et al. 2012](#)). Therefore, there is a need to better understand how current management options can contribute to the recovery and conservation of marine ecosystems and resources.

Within the EBM, ecosystem models and ecological forecasts have become an essential analytical and decision-making tool despite their limitations due to high uncertainties and complex ecosystem characteristics ([Link, Ihde et al. 2012](#); [Collie, Botsford et al. 2014](#); [Maris, Huneman et al. 2017](#)). They have the potential to provide insights of possible future impacts on marine ecosystems and can offer guidance to decision-makers by evaluating the trade-off between different management units and identify those measures that have the potential to meet conservation objectives ([Fulton, Boschetti et al. 2015](#); [Acosta, Wintle et](#)

[al. 2016](#)). However, in order to better assess MPA effectiveness and design, there is a need to take into account economic and social attributes of the municipalities of the MPAs ([Bennett and Dearden 2014](#); [Clarke, Thurstan et al. 2016](#)). Therefore, the integration of the social and economic components into ecosystem models needs to be promoted. Also future research should try to assess how the benefits potentially delivered by MPAs in terms of species recovery and increase of fisheries catches actually affect human well-being of local communities inhabiting within and around MPAs.

This study is the baseline to develop future scenarios of alternative management options in order to maximize the impacts of MPAs to their surrounding areas and fisheries sustainability by alternative MPAs configurations. In addition, due to the vicinity of the three MPAs along a latitudinal gradient, this study is a part of a nested modelling approach with different geographic scales with the aim to assess the current effects of the actual MPA network and to perform simulations of alternative MPA network configuration ([Coll, Steenbeek et al. 2019](#)).

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Author Contributions

XC, DV, CP, JS and MC designed the study and analyses. CP, JS, JC, JL, AC, ADF, TF, AL, GP, RS, PS, PG and MC contributed with material and analysis tools. XC, DV and MC developed the models. XC, DV, CP, JS, JC, JL, AC, ADF; AL, GP, PG and MC interpreted the results. XC wrote the manuscript with assistance from DV, JC and MC and input from CP, JS, JL, AC, ADF, TF, AL, GP, RS, PS and PG.

Tables legends

Table 1. Surface area (km²) covered by management units (MU) and year of creation of each Marine Protected Area (MPA) of Cerbère-Banyuls, Cap de Creus and Medes Islands.

FPA= Fully protected Area, PPA= partially protected area, UPA= unprotected area flanking the MPA.

Table 2. Overall fitting procedure applied to the three MPA models following the methodology suggested by [Mackinson, Daskalov et al. \(2009\)](#) and [Heymans, Coll et al. \(2016\)](#).

Table 3. Results of the fitting procedure of the three MPA of Cerbère-Banyuls, Cap de Creus and Medes Islands. The table shows the statistically “best” model for each step. Vs = number of vulnerabilities estimated, PPsp = number of primary production spline points, k = number of parameters (Vs + PPsp), %IF = improved fit compared to the baseline AICc. The “best” models are highlighted in bold.

Figure legends

Figure. 1. The study area encompassing the three multi-zone MPAs in the Northwestern Mediterranean Sea with the fully protected areas (FPAs), partially protected areas (PPAs) and unprotected neighboring areas (UPAs).

Figure. 2. Flow diagram of Cerbère-Banyuls (a), Cap de Creus (b) and Medes Island (c) MPA models. The numbers identify the functional groups of the model (listed in SOM 2). The size of each circle is proportional to the biomass of the functional group. The thickness of the connecting lines is proportional to the magnitude of their trophic flows.

Figure. 3. Ecological indicators estimated for the three multi-zone MPA models of Cerbère-Banyuls, Cap de Creus and Medes Islands. Boxplot shows the distribution of values for an ecological indicator derived from the Monte Carlo routine while the dot represents the value of the indicator in the baseline Ecopath balanced model.

Figure. 4. Functional groups plotted against Keystone Index (Valls et al., 2015) and trophic level for Cerbère-Banyuls (a), Cap de Creus (b) and Medes (c) multi-zone MPA models. The numbers identify the functional groups of the model (listed in SOM 2). The size of the circles is proportional to the biomass of the functional group.

Figure. 5. Mixed Trophic Impact (MTI) analysis of the three MPA applied to the fisheries in a) Cerbère-Banyuls, b) Cap de Creus, and c) Medes Islands multi-zone MPA models.

Figure. 6. Predicted (solid lines) versus observed (dots) biomass ($t \cdot km^{-2}$) values for target species of Cerbère-Banyuls, Cap de Creus and Medes Islands multi-zone MPAs models. Blue shadows represent the 5% and 95% percentiles obtained using the Monte Carlo routine. Rho and p-values result from Spearman's rank correlation test.

Figure. 7. Temporal trends of ecological indicators of Cerbère-Banyuls, Cap de Creus and Medes Islands multi-zone MPAs models. Blue shadows represent the 5% and 95% percentiles obtained using the Monte Carlo routine. Rho and p-values result from Spearman's rank correlation test.

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