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

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1 **To be re-submitted to Ocean and Coastal Management**

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

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32

33 **Abstract**

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

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

56

## 57 **1. Introduction**

58 Coastal and marine ecosystems provide multiple ecosystem goods and services ([Costanza,](#)  
59 [d'Arge et al. 1997](#); [Martínez, Intralawan et al. 2007](#)). However, the provision of these  
60 ecosystem goods and services can be degraded as human demand for resources and the  
61 associated activities increase in number, frequency, magnitude and spatial coverage ([MEA](#)  
62 [2005](#); [Worm, Barbier et al. 2006](#); [Pauly and Zeller 2016](#)). Currently, marine ecosystems are  
63 under increasing threat from a range of stressors including overfishing, climate change,  
64 biological invasions, pollution, aquaculture and habitat degradation, directly or indirectly  
65 caused by multiple human activities ([Costello, Coll et al. 2010](#); [Halpern, Frazier et al.](#)  
66 [2015](#)). The relationships and interactive effects of multiple stressors are highly complex,  
67 and largely unknown or even highly uncertain ([Crain, Halpern et al. 2009](#); [Côté, Darling et](#)  
68 [al. 2016](#)).

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

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

94 Within the EBM, ecological modelling has been proven to be a suitable tool to merge  
95 available ecological information into a coherent form to obtain insights about how  
96 ecosystems are structured and impacted by human activities and environment, and  
97 delivering ecosystem services ([Link 2010](#); [Christensen and Maclean 2011](#)). Although  
98 quantitative models have become a key tool to assess the extent to which MPAs are able to  
99 achieve their conservation targets ([Fulton, Bax et al. 2015](#)), the understanding of MPA  
100 effects on ecological processes and ecosystem function is still scarce ([Cheng, Altieri et al.  
101 2019](#)). For instance, it is mostly unknown if protection benefits inside fully protected areas  
102 extend to partially areas and beyond, and if those benefits have the potential to cascade  
103 through the ecosystem components. There is little evidence that multi-zone MPAs (i.e.,  
104 those including both FPAs and PPAs) provide fisheries benefits inside partially protected  
105 areas and it is unknown what are the drivers of multi-zone MPAs.

106 Although considered a hotspot of biodiversity ([Coll, Piroddi et al. 2010](#)), the Mediterranean  
107 Sea has been largely altered by anthropogenic stressors for a long time ([Coll, Piroddi et al.  
108 2012](#); [Micheli, Halpern et al. 2013](#); [Halpern, Frazier et al. 2015](#)), that have determined  
109 dramatic changes on its ecosystems especially during the last 50 years ([Azzurro, Moschella  
110 et al. 2011](#); [Maynou, Sbrana et al. 2011](#); [Piroddi, Coll et al. 2017](#)). In order to address this  
111 urgent conservation challenge, a multitude of MPAs (including all forms of nationally and  
112 internationally established protected areas) have been implemented in the Mediterranean  
113 Sea in the last decades. Within this context, several studies have modelled MPAs in the  
114 Mediterranean Sea with the aim to evaluate their effects on the ecosystem and the impacts  
115 of small-scale and recreational fisheries on marine resources ([Albouy, Mouillot et al. 2010](#);  
116 [Libralato, Coll et al. 2010](#); [Valls, Gascuel et al. 2012](#); [Prato, Barrier et al. 2016](#); [Vilas, Coll](#)

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

127

## 128 2. Materials & Methods

### 129 2.1. Study area

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

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

### 147 2.2. Modelling approach – Ecopath and Ecosim

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



157 accounting for changes in predation, consumption rate, fishing and the environment  
158 ([Walters, Christensen et al. 1997](#); [Christensen and Walters 2004](#)). A short description of the  
159 modelling approach is given in the Supplementary Online Material (SOM hereafter) 1; and  
160 a detailed information of the EwE approach is documented in [Christensen and Walters](#)  
161 [\(2004\)](#) and [Heymans, Coll et al. \(2016\)](#).

## 162 **2.3. Functional groups and input data**

### 163 **2.3.1. Functional groups**

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

### 171 **2.3.2. Ecopath input data and balancing procedure**

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

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

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

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

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

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

#### 210 **2.3.4. Ecosim historical time series fitting.**

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

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

219 The models were fitted by using the Stepwise Fitting Procedure ([Scott, Serpetti et al. 2016](#)).  
220 This procedure tests alternative hypotheses related to the impact of fishing, changes in  
221 predator-prey dynamics (vulnerabilities), changes in primary production (production  
222 anomalies) or all of the above together (Table 2) ([Mackinson, Daskalov et al. 2009](#);  
223 [Heymans, Coll et al. 2016](#)). This procedure estimates different vulnerability parameters and  
224 primary productions anomalies to improve model fits by comparing model predictions to  
225 observed data using the sum of squares (SS) statistics and to find the statistically “best fit”  
226 model based on Akaike’s Information Criterion (AIC), which penalizes the  
227 overparameterization of the model ([Mackinson, Daskalov et al. 2009](#); [Heymans, Coll et al.  
228 2016](#)). In this study, the maximum number of parameters that could be estimated were 19  
229 (Cerbère-Banyuls), 51 (Cap de Creus) and 48 (Medes Islands), respectively.

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

## 233 **2.4. Model analyses and ecological indicators**

### 234 **2.4.1. Ecological indicators of initial conditions**

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

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

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

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

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

256 (4) Catch-based: based on catch, they reflect the fishing strategy of the fisheries and are  
257 used to quantify the impact of fishing ([Hilborn and Walters 1992](#); [Pauly, Christensen et al.](#)  
258 [1998](#)). We included total catch (TC  $t \cdot km^{-2} \cdot year^{-1}$ ), trophic level of the catch (TLc), intrinsic  
259 vulnerability index of catch (VI).

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

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

270 with a large and broad impact on the food web despite its low biomass ([Paine 1966](#); [Paine](#)  
271 [1969](#); [Valls, Coll et al. 2015](#)).

## 272 **2.4.2. Time-dynamic analyses**

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

283 The Monte Carlo (MC) uncertainty routine and the ECOSAMPLER plug-in were employed  
284 to assess the impact of uncertainty in Ecopath input parameters on Ecosim simulations  
285 ([Heymans, Coll et al. 2016](#); [Coll and Steenbeek 2017](#); [Steenbeek, Corrales et al. 2018](#)). We  
286 ran 500 MC simulations based on the sensibility of Ecopath input parameters obtained from  
287 the pedigree routine (SOM 3). Results from the MC simulations were used to plot the  
288 confidence intervals of the selected ecological indicators in Ecopath and to plot the 5<sup>th</sup> and  
289 95<sup>th</sup> percentile confidence intervals for the Ecosim outputs. Finally, we used the  
290 Spearman's rank correlation to evaluate the presence of significant trends (either increasing  
291 or decreasing) in model outputs (biomass, catch and ecological indicators), following  
292 previous studies ([e.g., Corrales, Coll et al. 2017](#)).

## 293 **3. Results**

### 294 **3.1. Structure and functional traits of protected areas**

295 The pedigree index values of the three MPA models showed similar values, ranging from  
296 0.41 to 0.51. The highest pedigree values were obtained for Cerbère-Banyuls (0.51),  
297 followed by Medes Islands (0.45) and Cap de Creus (0.41).

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

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

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

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

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

### 340 **3.2. Ecological impacts of protection over time**

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

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

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

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

## 374 **4. Discussion**

### 375 **4.1. Structure and functional traits of protected areas**

376 Overall, ecological indicators showed similar patterns, with the highest values in Banyuls-  
377 Cerbère MPA, followed by Medes Islands and Cap de Creus MPAs. This may be related to  
378 differences in the ecological effectiveness of the three MPAs, which is partly explained by  
379 MPA design, management and implementation features (e.g. extent of area protected, fully  
380 protected area enforcement, time since protection, MPA design) ([Claudet, Osenberg et al.  
381 2008](#); [Guidetti, Milazzo et al. 2008](#); [Edgar, Stuart-Smith et al. 2014](#); [Giakoumi, Scianna et  
382 al. 2017](#); [Di Franco, Plass-Johnson et al. 2018](#)). The lack of enforcement is one of the most  
383 relevant issues concerning MPAs in the Mediterranean context ([Fenberg, Caselle et al.  
384 2012](#)). Within this context, while Cerbère-Banyuls and Medes MPAs are considered to



385 have a high level of enforcement ([Sala, Ballesteros et al. 2012](#); [Giakoumi, Scianna et al.](#)  
386 [2017](#)), Cap de Creus has been considered a MPA only on paper due to a lack of sufficient  
387 enforcement ([Lloret, Zaragoza et al. 2008](#); [Lloret, Zaragoza et al. 2008](#)). In fact, total  
388 catches were higher in Cap the Creus than in the other two MPA; with Cerbère-Banyuls  
389 presenting the lowest total catches. Total catches are expected to be higher in well designed  
390 and enforced MPAs ([Halpern, Lester et al. 2009](#); [Di Lorenzo, Claudet et al. 2016](#)), so our  
391 results could be explained by the allowed and/or real fishing effort and the small effects of  
392 these MPAs in PPAs and adjacent areas. [Horta e Costa, Claudet et al. \(2016\)](#) presented a  
393 novel classification system for MPAs by scoring allowed uses in each management unit  
394 based on their impacts on biodiversity. In this scale, Cerbère-Banyuls obtained a rate of 4.7,  
395 being a highly protected area and Medes 6.4 being less well protected ([Horta e Costa,](#)  
396 [Claudet et al. 2016](#)). Although Cap de Creus was not included in this study, we could  
397 expect a higher MPA index because of its smaller FPA and allowed uses. These features  
398 (smallest FPA and weak enforcement) and the fact that Cap de Creus is the newest (Table  
399 1) MPA in the study, could suggest that the year of establishment, enforcement and the size  
400 of protected areas have a strong effect on MPA effects, in line with other studies ([Claudet,](#)  
401 [Osenberg et al. 2008](#); [Guidetti, Milazzo et al. 2008](#)).

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

416 small-scale and recreational fisheries on vulnerable species ([Lloret, Biton-Porsmoguer et al.](#)  
417 [2019](#)), which is an important fact to consider when establishing management plans for these  
418 species. Our results also highlighted the competition for resources between both (small-  
419 scale and recreational) fisheries. Competition between both fisheries has become an  
420 important issue in coastal areas (especially in surrounding areas of MPAs), as they target  
421 similar species and fishing grounds ([Chuenpagdee and Jentoft 2018](#); [Lloret, Cowx et al.](#)  
422 [2018](#)). In small and touristic areas such as the ones in the present study, a strong  
423 competition between small-scale and recreational fishers could be expected due to the  
424 current economic, societal and environmental challenges of the small scale fisheries and the  
425 increasing importance of recreational fisheries as a leisure activity in these areas ([Gómez](#)  
426 [and Lloret 2017](#)).

#### 427 **4.2. Ecological impacts of protection over time**

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

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

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

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

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

### 473 **4.3. Data gaps, limitations and uncertainties**

474 Overall the input data used was of acceptable quality compared to the distribution of  
475 pedigree values in other existing models ([Morissette 2007](#); [Lassalle, Bourdaud et al. 2014](#)),  
476 although they are among the lowest values in the Mediterranean Sea ([Corrales, Coll et al.](#)  
477 [2015](#)). This could be due to the challenges of modelling small and local coastal areas,  
478 where specific data is a strong requirement. In fact, limitations of available data are a  
479 common concern for most Mediterranean MPAs ([Prato, Barrier et al. 2016](#); [Vilas, Coll et](#)  
480 [al. Submitted](#)) despite there are monitoring programs inside these MPAs.

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

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

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

#### 504 **4.4. Concluding remarks**

505 Human activities have been concentrated in coastal ecosystems, resulting in a major  
506 modification of these marine areas ([Halpern, Frazier et al. 2015](#)). Since coastal areas  
507 account for a large amount of the global ecosystem services ([Costanza, d'Arge et al. 1997](#)),  
508 there is a need for the recovery of coastal marine ecosystems in order to ensure their  
509 ecological role. In addition, in recent decades, marine ecosystems have been increasingly  
510 impacted by other stressors, directly or indirectly induced by multiple anthropogenic  
511 activities ([Halpern, Frazier et al. 2015](#)). For example, sea warming and the spread of non-  
512 indigenous species represent a major challenge for conservation in Mediterranean  
513 ecosystems, especially in the Eastern Mediterranean Sea ([Lejeusne, Chevaldonné et al.](#)  
514 [2010](#); [Katsanevakis, Coll et al. 2014](#)). Organisms and ecosystems that have been already  
515 impacted by fishing are more vulnerable to these additional stressors ([Occhipinti-Ambrogi](#)  
516 [and Savini 2003](#); [Poloczanska, Burrows et al. 2016](#)). MPAs, when properly managed, have  
517 demonstrated to be an effective tool to protect target species and habitats to local stressors  
518 such as fishing ([Sala, Giakoumi et al. 2017](#)), while their role in promoting resilience to  
519 regional and global stressors such biological invasions and climate change is debated  
520 ([Giakoumi and Pey 2017](#); [Roberts, O'Leary et al. 2017](#); [Giakoumi, Pey et al. 2019](#)). In  
521 highly impacted and crowded areas like the Northwestern Mediterranean Sea, the  
522 establishment of well-designed networks of well-enforced MPA is necessary to achieve  
523 “clean, healthy and productive” oceans (Good Environmental Status) according to the  
524 Marine Strategy Framework Directive ([Fenberg, Caselle et al. 2012](#)). Therefore, there is a  
525 need to better understand how current management options can contribute to the recovery  
526 and conservation of marine ecosystems and resources.

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

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

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

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#### 552 **Author Contributions**

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

#### 558 **Tables legends**

559 **Table 1.** Surface area (km<sup>2</sup>) covered by management units (MU) and year of creation of  
560 each Marine Protected Area (MPA) of Cerbère-Banyuls, Cap de Creus and Medes Islands.

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

563 **Table 2.** Overall fitting procedure applied to the three MPA models following the  
564 methodology suggested by [Mackinson, Daskalov et al. \(2009\)](#) and [Heymans, Coll et al.](#)  
565 [\(2016\)](#).

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

### 571 **Figure legends**

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

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

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

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

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

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

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

597



598 **References**

- 599 Acosta, L. A., B. A. Wintle, et al. (2016). Using scenarios and models to inform decision making in  
600 policy design and implementation. Methodological assessment of scenarios and models of  
601 biodiversity and ecosystem services [S. Ferrier, K. N. Ninan, P. Leadley, R. Alkemade, L. A.  
602 Acosta, H.R. Akçakaya, L. Brotons, W.W.L. Cheung, V. Christensen, K. A. Harhash, J.  
603 Kabubo-Mariara, C. Lundquist, M. Obersteiner, H. Pereira, G. Peterson, R. Pichs-Madruga,  
604 N. Ravindranath, C. Rondinini and B.A. Wintle (eds.)]. IPBES. Bonn, Germany: 43-100.
- 605 Albouy, C., D. Mouillot, et al. (2010). "Simulation of the combined effects of artisanal and  
606 recreational fisheries on a Mediterranean MPA ecosystem using a trophic model." Marine  
607 Ecology Progress Series **412**: 207-221.
- 608 Azzurro, E., P. Moschella, et al. (2011). "Tracking signals of change in Mediterranean fish diversity  
609 based on local ecological knowledge." PloS one **6**(9): e24885.
- 610 Ben Lamine, E., A. Di Franco, et al. (2018). "Comparing commercial, recreational and illegal coastal  
611 fishery catches and their economic values: A survey from the southern Mediterranean  
612 Sea." Fisheries Management and Ecology **25**(6): 456-463.
- 613 Bennett, N. J. and P. Dearden (2014). "Why local people do not support conservation: Community  
614 perceptions of marine protected area livelihood impacts, governance and management in  
615 Thailand." Marine Policy **44**: 107-116.
- 616 Clarke, B., R. Thurstan, et al. (2016). "An investigation into the socio-cultural dimension of  
617 determining MPA effectiveness." Report prepared for Department of Environment, Water  
618 and Natural Resources, Adelaide.
- 619 Claudet, J. (2011). Marine protected areas: a multidisciplinary approach, Cambridge University  
620 Press.
- 621 Claudet, J., C. Osenberg, et al. (2010). "Marine reserves: fish life history and ecological traits  
622 matter." Ecological applications **20**(3): 830-839.
- 623 Claudet, J., C. W. Osenberg, et al. (2008). "Marine reserves: size and age do matter." Ecology  
624 letters **11**(5): 481-489.
- 625 Coll, M., J. Navarro, et al. (2013). "Ecological role, fishing impact, and management options for the  
626 recovery of a Mediterranean endemic skate by means of food web models." Biological  
627 Conservation **157**: 108-120.
- 628 Coll, M., I. Palomera, et al. (2006). "Trophic flows, ecosystem structure and fishing impacts in the  
629 South Catalan Sea, Northwestern Mediterranean." Journal of Marine Systems **59**(1): 63-96.
- 630 Coll, M., C. Piroddi, et al. (2012). "The Mediterranean Sea under siege: spatial overlap between  
631 marine biodiversity, cumulative threats and marine reserves." Global Ecology and  
632 Biogeography **21**(4): 465-480.
- 633 Coll, M., C. Piroddi, et al. (2010). "The biodiversity of the Mediterranean Sea: estimates, patterns,  
634 and threats." PloS one **5**(8): e11842.
- 635 Coll, M. and J. Steenbeek (2017). "Standardized ecological indicators to assess aquatic food webs:  
636 The ECOIND software plug-in for Ecopath with Ecosim models." Environmental Modelling  
637 & Software **89**: 120-130.
- 638 Coll, M., J. Steenbeek, et al. (2019). Report on the optimal configuration of the networks of  
639 management units in the study area to optimize fishing benefits and sustainability, and  
640 comparison with current networks. Sustainable Fisheries in EU Mediterranean waters  
641 through network of MPAs - Safenet MARE/2014/41 Deliverable 4.4:88.
- 642 Coll, M., J. Steenbeek, et al. (2019). Report describing the qualitative models. Sustainable Fisheries  
643 in EU Mediterranean waters through network of MPAs - Safenet MARE/2014/41  
644 Deliverable 4.1:17.

645 Coll, M., D. Vilas, et al. (2019). Report describing the quantitative models. Sustainable Fisheries in  
646 EU Mediterranean waters through network of MPAs - Safenet MARE/2014/41 Deliverable  
647 4.2:62.

648 Collie, J. S., L. W. Botsford, et al. (2014). "Ecosystem models for fisheries management: finding the  
649 sweet spot." Fish and Fisheries.

650 Corrales, X., M. Coll, et al. (2017). "Hindcasting the dynamics of an Eastern Mediterranean marine  
651 ecosystem under the impacts of multiple stressors." Marine Ecology Progress Series **580**:  
652 17-36.

653 Corrales, X., M. Coll, et al. (2015). "Ecosystem structure and fishing impacts in the northwestern  
654 Mediterranean Sea using a food web model within a comparative approach." Journal of  
655 Marine Systems **148**: 183-199.

656 Costanza, R., R. d'Arge, et al. (1997). "The value of the world's ecosystem services and natural  
657 capital." Nature **387**(6630): 253.

658 Costello, M. J., M. Coll, et al. (2010). "A census of marine biodiversity knowledge, resources, and  
659 future challenges." PloS one **5**(8): e12110.

660 Côté, I. M., E. S. Darling, et al. (2016). "Interactions among ecosystem stressors and their  
661 importance in conservation." Proceedings of the Royal Society of London B: Biological  
662 Sciences **283**(1824).

663 Crain, C. M., B. S. Halpern, et al. (2009). "Understanding and managing human threats to the  
664 coastal marine environment." Annals of the New York Academy of Sciences **1162**(1): 39-  
665 62.

666 Cheng, B. S., A. H. Altieri, et al. (2019). "Can marine reserves restore lost ecosystem functioning? A  
667 global synthesis." Ecology **100**(4): e02617.

668 Christensen, V. and J. Maclean (2011). Ecosystem approaches to fisheries: a global perspective.  
669 Cambridge, Cambridge University Press.

670 Christensen, V., C. Walters, et al. (2008). "Ecopath with Ecosim version 6 user guide." Lenfest  
671 Ocean Futures Project, University of British Columbia, Vancouver.

672 Christensen, V. and C. J. Walters (2004). "Ecopath with Ecosim: methods, capabilities and  
673 limitations." Ecological Modelling **172**(2): 109-139.

674 Chuenpagdee, R. and S. Jentoft (2018). Transdisciplinarity for small-scale fisheries governance,  
675 Springer.

676 Di Franco, A., J. G. Plass-Johnson, et al. (2018). "Linking home ranges to protected area size: The  
677 case study of the Mediterranean Sea." Biological Conservation **221**: 175-181.

678 Di Franco, A., P. Thiriet, et al. (2016). "Five key attributes can increase marine protected areas  
679 performance for small-scale fisheries management." Scientific reports **6**: 38135.

680 Di Lorenzo, M., J. Claudet, et al. (2016). "Spillover from marine protected areas to adjacent  
681 fisheries has an ecological and a fishery component." Journal for Nature Conservation **32**:  
682 62-66.

683 Edgar, G. J., R. D. Stuart-Smith, et al. (2014). "Global conservation outcomes depend on marine  
684 protected areas with five key features." Nature **506**: 216.

685 Fenberg, P. B., J. E. Caselle, et al. (2012). "The science of European marine reserves: Status,  
686 efficacy, and future needs." Marine Policy **36**(5): 1012-1021.

687 Fulton, E. A., N. J. Bax, et al. (2015). "Modelling marine protected areas: insights and hurdles."   
688 Philosophical Transactions of the Royal Society B: Biological Sciences **370**(1681):  
689 20140278.

690 Fulton, E. A., F. Boschetti, et al. (2015). "A multi-model approach to engaging stakeholder and  
691 modellers in complex environmental problems." Environmental Science & Policy **48**: 44-  
692 56.

693 Gaines, S. D., C. White, et al. (2010). "Designing marine reserve networks for both conservation  
694 and fisheries management." Proceedings of the National Academy of Sciences **107**(43):  
695 18286-18293.

696 Gell, F. R. and C. M. Roberts (2003). "Benefits beyond boundaries: the fishery effects of marine  
697 reserves." Trends in Ecology & Evolution **18**(9): 448-455.

698 Giakoumi, S., J. McGowan, et al. (2018). "Revisiting "success" and "failure" of marine protected  
699 areas: a conservation scientist perspective." Frontiers in Marine Science **5**: 223.

700 Giakoumi, S. and A. Pey (2017). "Assessing the effects of marine protected areas on biological  
701 invasions: a global review." Frontiers in Marine Science **4**: 49.

702 Giakoumi, S., A. Pey, et al. (2019). "Exploring the relationships between marine protected areas  
703 and invasive fish in the world's most invaded sea." Ecological applications **29**(1): e01809.

704 Giakoumi, S., C. Scianna, et al. (2017). "Ecological effects of full and partial protection in the  
705 crowded Mediterranean Sea: a regional meta-analysis." Scientific reports **7**(1): 8940.

706 Giron-Nava, A., C. C. James, et al. (2017). "Quantitative argument for long-term ecological  
707 monitoring." Marine Ecology Progress Series **572**: 269-274.

708 Gómez, S. and J. Lloret (2017). The Small-Scale Fisheries Guidelines as a Tool for Marine  
709 Stewardship: The Case of Cap de Creus Marine Protected Area, Spain. The Small-Scale  
710 Fisheries Guidelines, Springer: 401-420.

711 Goñi, R., S. Adlerstein, et al. (2008). "Spillover from six western Mediterranean marine protected  
712 areas: evidence from artisanal fisheries." Marine Ecology Progress Series **366**: 159-174.

713 Grorud-Colvert, K., J. Claudet, et al. (2014). "Marine protected area networks: assessing whether  
714 the whole is greater than the sum of its parts." PloS one **9**(8): e102298.

715 Grüss, A., D. M. Kaplan, et al. (2011). "Consequences of adult and juvenile movement for marine  
716 protected areas." Biological Conservation **144**(2): 692-702.

717 Guidetti, P., M. Milazzo, et al. (2008). "Italian marine reserve effectiveness: does enforcement  
718 matter?" Biological Conservation **141**(3): 699-709.

719 Halpern, B. S., M. Frazier, et al. (2015). "Spatial and temporal changes in cumulative human  
720 impacts on the world's ocean." Nature communications **6**.

721 Halpern, B. S., S. E. Lester, et al. (2009). "Spillover from marine reserves and the replenishment of  
722 fished stocks." Environmental Conservation **36**(4): 268-276.

723 Halpern, B. S., S. E. Lester, et al. (2010). "Placing marine protected areas onto the ecosystem-  
724 based management seascape." Proceedings of the National Academy of Sciences **107**(43):  
725 18312-18317.

726 Harrison, H. B., D. H. Williamson, et al. (2012). "Larval export from marine reserves and the  
727 recruitment benefit for fish and fisheries." Current Biology **22**(11): 1023-1028.

728 Hereu Fina, B., E. Aspillaga Cuevas, et al. (2017). "Seguiment del medi marí al Parc Natural del Cap  
729 de Creus i al Parc Natural del Montgrí, les Illes Medes i el Baix Ter. Memòria 2016."

730 Heymans, J. J., M. Coll, et al. (2016). "Best practice in Ecopath with Ecosim food-web models for  
731 ecosystem-based management." Ecological Modelling.

732 Hilborn, R. and C. J. Walters (1992). "Quantitative fisheries stock assessment: choice, dynamics  
733 and uncertainty." Reviews in fish biology and fisheries **2**(2): 177-178.

734 Horta e Costa, B., J. Claudet, et al. (2016). "A regulation-based classification system for Marine  
735 Protected Areas (MPAs)." Marine Policy **72**: 192-198.

736 IUCN-WCPA (2008). Establishing Marine Protected Area Networks: Making it Happen, IUCN-World  
737 Commission on Protected Areas, National Oceanic and Atmospheric Administration and  
738 The Nature Conservancy.

739 Katsanevakis, S., M. Coll, et al. (2014). "Invading the Mediterranean Sea: biodiversity patterns  
740 shaped by human activities." Marine Ecosystem Ecology **1**: 32.

741 Lassalle, G., P. Bourdaud, et al. (2014). "A toolbox to evaluate data reliability for whole-ecosystem  
742 models: Application on the Bay of Biscay continental shelf food-web model." Ecological  
743 Modelling **285**: 13-21.

744 Leenhardt, P., N. Low, et al. (2015). The role of marine protected areas in providing ecosystem  
745 services. Aquatic functional biodiversity, Elsevier: 211-239.

746 Lejeusne, C., P. Chevaldonné, et al. (2010). "Climate change effects on a miniature ocean: the  
747 highly diverse, highly impacted Mediterranean Sea." Trends in Ecology & Evolution **25**(4):  
748 250-260.

749 Leslie, H. M. and K. L. McLeod (2007). "Confronting the challenges of implementing marine  
750 ecosystem-based management." Frontiers in Ecology and the Environment **5**(10): 540-548.

751 Lester, S. E., B. S. Halpern, et al. (2009). "Biological effects within no-take marine reserves: a global  
752 synthesis." Marine Ecology Progress Series **384**: 33-46.

753 Libralato, S., M. Coll, et al. (2010). "Food-web traits of protected and exploited areas of the  
754 Adriatic Sea." Biological Conservation **143**(9): 2182-2194.

755 Lindeman, R. L. (1942). "The trophic-dynamic aspect of ecology." Ecology **23**(4): 399-417.

756 Link, J. (2010). Ecosystem-based fisheries management: confronting tradeoffs, Cambridge  
757 University Press.

758 Link, J. S. (2010). "Adding rigor to ecological network models by evaluating a set of pre-balance  
759 diagnostics: a plea for PREBAL." Ecological Modelling **221**(12): 1580-1591.

760 Link, J. S., T. Ihde, et al. (2012). "Dealing with uncertainty in ecosystem models: the paradox of use  
761 for living marine resource management." Progress in Oceanography **102**: 102-114.

762 Long, R. D., A. Charles, et al. (2015). "Key principles of marine ecosystem-based management."  
763 Marine Policy **57**: 53-60.

764 Lloret, J., S. Biton-Porsmoguer, et al. (2019). "Recreational and small-scale fisheries may pose a  
765 threat to vulnerable species in coastal and offshore waters of the western  
766 Mediterranean." ICES Journal of Marine Science.

767 Lloret, J., I. G. Cowx, et al. (2018). "Small-scale coastal fisheries in European Seas are not what they  
768 were: Ecological, social and economic changes." Marine Policy **98**: 176-186.

769 Lloret, J., N. Zaragoza, et al. (2008). "Spearfishing pressure on fish communities in rocky coastal  
770 habitats in a Mediterranean marine protected area." Fisheries Research **94**(1): 84-91.

771 Lloret, J., N. Zaragoza, et al. (2008). "Biological and socioeconomic implications of recreational  
772 boat fishing for the management of fishery resources in the marine reserve of Cap de  
773 Creus (NW Mediterranean)." Fisheries Research **91**(2-3): 252-259.

774 Mackinson, S., G. Daskalov, et al. (2009). "Which forcing factors fit? Using ecosystem models to  
775 investigate the relative influence of fishing and changes in primary productivity on the  
776 dynamics of marine ecosystems." Ecological Modelling **220**(21): 2972-2987.

777 Maris, V., P. Huneman, et al. (2017). "Prediction in ecology: promises, obstacles and clarifications."  
778 Oikos **0**: 1-12.

779 Martínez, M. L., A. Intralawan, et al. (2007). "The coasts of our world: Ecological, economic and  
780 social importance." Ecological Economics **63**(2-3): 254-272.

781 Maynou, F., M. Sbrana, et al. (2011). "Estimating trends of population decline in long-lived marine  
782 species in the Mediterranean Sea based on fishers' perceptions." PLoS one **6**(7): e21818.

783 McLeod, K. L., J. Lubchenco, et al. (2005). "Scientific consensus statement on marine ecosystem-  
784 based management." Signed by **221**: 1-21.

785 MEA (2005). (Millennium Ecosystem Assessment) Ecosystems and human well-being: biodiversity  
786 synthesis. World Resources Institute, Washington DC.

787 Micheli, F., B. S. Halpern, et al. (2004). "Trajectories and correlates of community change in  
788 no-take marine reserves." Ecological applications **14**(6): 1709-1723.

789 Micheli, F., B. S. Halpern, et al. (2013). "Cumulative Human Impacts on Mediterranean and Black  
790 Sea Marine Ecosystems: Assessing Current Pressures and Opportunities." PloS one **8**(12):  
791 e79889.

792 Morissette, L. (2007). Complexity, cost and quality of ecosystem models and their impact on  
793 resilience: a comparative analysis, with emphasis on marine mammals and the Gulf of St.  
794 Lawrence PhD thesis, University of British Columbia.

795 Occhipinti-Ambrogi, A. and D. Savini (2003). "Biological invasions as a component of global change  
796 in stressed marine ecosystems." Marine pollution bulletin **46**(5): 542-551.

797 Paine, R. T. (1966). "Food web complexity and species diversity." American Naturalist: 65-75.

798 Paine, R. T. (1969). "A note on trophic complexity and community stability." The American  
799 Naturalist **103**(929): 91-93.

800 Pauly, D., V. Christensen, et al. (1998). "Fishing down marine food webs." science **279**(5352): 860-  
801 863.

802 Pauly, D., A. Ulman, et al. (2014). "'Reported' versus 'likely' fisheries catches of four Mediterranean  
803 countries." Scientia Marina **78**(S1): 11-17.

804 Pauly, D. and D. Zeller (2016). "Catch reconstructions reveal that global marine fisheries catches  
805 are higher than reported and declining." Nature communications **7**: ncomms10244.

806 Piroddi, C., M. Coll, et al. (2017). "Historical changes of the Mediterranean Sea ecosystem:  
807 modelling the role and impact of primary productivity and fisheries changes over time."  
808 Scientific Reports.

809 Piroddi, C., M. Coll, et al. (2015). "Modelling the Mediterranean marine ecosystem as a whole:  
810 addressing the challenge of complexity." Marine Ecology Progress Series **533**: 47-65.

811 Poloczanska, E. S., M. T. Burrows, et al. (2016). "Responses of marine organisms to climate change  
812 across oceans." Frontiers in Marine Science **3**: 62.

813 Prato, G., C. Barrier, et al. (2016). "Assessing interacting impacts of artisanal and recreational  
814 fisheries in a small Marine Protected Area (Portofino, NW Mediterranean Sea)." Ecosphere  
815 **7**(12).

816 Roberts, C. M., B. C. O'Leary, et al. (2017). "Marine reserves can mitigate and promote adaptation  
817 to climate change." Proceedings of the National Academy of Sciences **114**(24): 6167-6175.

818 Rochet, M.-J. and V. M. Trenkel (2003). "Which community indicators can measure the impact of  
819 fishing? A review and proposals." Canadian Journal of Fisheries and Aquatic Sciences **60**(1):  
820 86-99.

821 Sala, E., E. Ballesteros, et al. (2012). "The structure of Mediterranean rocky reef ecosystems across  
822 environmental and human gradients, and conservation implications." PloS one **7**(2):  
823 e32742.

824 Sala, E., S. Giakoumi, et al. (2017). "No-take marine reserves are the most effective protected  
825 areas in the ocean." ICES Journal of Marine Science **75**(3): 1166-1168.

826 Scianna, C., F. Niccolini, et al. (2019). "Organization Science improves management effectiveness  
827 of Marine Protected Areas." Journal of Environmental Management **240**: 285-292.

828 Scott, E., N. Serpetti, et al. (2016). "A Stepwise Fitting Procedure for automated fitting of Ecopath  
829 with Ecosim models." SoftwareX.

830 Shin, Y.-J. and L. J. Shannon (2010). "Using indicators for evaluating, comparing, and  
831 communicating the ecological status of exploited marine ecosystems. 1. The IndiSeas  
832 project." ICES Journal of Marine Science: Journal du Conseil **67**(4): 686-691.

833 Steenbeek, J. (2018). "Diet Calculator - Quick Reference Guide. Version 0.8.5, 11 June 2018.  
834 [https://figshare.com/articles/Diet\\_calculator\\_guide\\_pdf/11369916](https://figshare.com/articles/Diet_calculator_guide_pdf/11369916). Safenet project - EU-  
835 DGMARE (MARE/2014/41). Barcelona."

836 Steenbeek, J., X. Corrales, et al. (2018). "Ecosampler: A new approach to assessing parameter  
837 uncertainty in Ecopath with Ecosim." SoftwareX **7**: 198-204.

838 Stelzenmüller, V., F. Maynou, et al. (2008). "Spatial assessment of fishing effort around European  
839 marine reserves: implications for successful fisheries management." Marine pollution  
840 bulletin **56**(12): 2018-2026.

841 Stergiou, K. I. and V. S. Karpouzi (2001). "Feeding habits and trophic levels of Mediterranean fish."  
842 Reviews in fish biology and fisheries **11**(3): 217-254.

843 Tam, J. C., J. S. Link, et al. (2017). "Towards ecosystem-based management: identifying operational  
844 food-web indicators for marine ecosystems." ICES Journal of Marine Science **74**(7): 2040-  
845 2052.

846 Ulanowicz, R. and C. Puccia (1990). "Mixed trophic impacts in ecosystems." Coenoses **5**(1): 7-16.

847 Valls, A., M. Coll, et al. (2015). "Keystone species: toward an operational concept for marine  
848 biodiversity conservation." Ecological Monographs **85**(1): 29-47.

849 Valls, A., D. Gascuel, et al. (2012). "Modeling trophic interactions to assess the effects of a marine  
850 protected area: case study in the NW Mediterranean Sea." Marine Ecology Progress Series  
851 **456**(201-214): 98.

852 Vilas, D., M. Coll, et al. (Submitted). "The effects of marine protected areas on ecosystem recovery  
853 and fisheries using a comparative modelling approach." Aquatic Conservation: Marine and  
854 Freshwater Ecosystems.

855 Walters, C., V. Christensen, et al. (1997). "Structuring dynamic models of exploited ecosystems  
856 from trophic mass-balance assessments." Reviews in fish biology and fisheries **7**(2): 139-  
857 172.

858 Worm, B., E. B. Barbier, et al. (2006). "Impacts of biodiversity loss on ocean ecosystem services."  
859 science **314**(5800): 787-790.

860

861

862