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## The effects of marine protected areas on ecosystem recovery and fisheries using a comparative modelling approach

Daniel Vilas, Marta Coll, Xavier Corrales, Jeroen Steenbeek, Chiara Piroddi, Antonio Calò, Antonio Di Franco, Toni Font, Paolo Guidetti, Alessandro Ligas, et al.

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1 **Potential journals: Biological Conservation**

2

3 **Contributions of coastal marine protected areas to fisheries sustainability**  
4 **and ecosystem recovery**

5

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## 27 **Abstract**

28 The overexploitation of many marine stocks calls for the development and  
29 assessment of tools to support recovery, rebuilding and conservation of marine  
30 resources. Here, we assessed the potential contributions to fisheries sustainability  
31 and ecosystem recovery at the local level of three multiple-use Marine Protected  
32 Areas (MPAs) in the Northwestern Mediterranean Sea ecosystem . For each MPA,  
33 we built a food-web model for each management (MU): the Fully Protected Area  
34 (FPA), the Partially Protected Area (PPA) and the Unprotected Area (UPA)  
35 surrounding the MPA. Using the nine food-web models we characterized and  
36 compared the structure and functioning of each MU, we assessed differences and  
37 similarities within and among the three MPAs, and evaluated if the ecosystem  
38 response to full protection led to specific ecosystem functional traits that are  
39 shared among the three MPAs. We showed differences among MUs in terms of  
40 ecosystem structure and functioning. Overall, FPAs presented most positive effect  
41 of protection in terms of ecosystem structure and functioning, and were followed by  
42 PPAs. The effects of protection on neighboring non-protected areas were hardly  
43 noticeable. Similarities between Cerbère-Banyuls and Medes Islands MPAs were  
44 observed, while Cap de Creus MPA showed the least benefits from protection. This  
45 is likely due to similarities in the configuration of the protection areas, levels of  
46 enforcement and establishment, and the impact of recreational and artisanal  
47 fisheries. Our study illustrates that well-enforced Mediterranean MPAs can yield  
48 local positive impacts on the structure and functioning of marine ecosystems that  
49 can contribute to fisheries sustainability.

50 **Keywords:** Management units, fully protected areas, partially protected areas,  
51 Ecopath with Ecosim, NW Mediterranean Sea

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## 53 **1. Introduction**

54 Marine ecosystems have been degraded at high rates under the cumulative impact  
55 of multiple anthropogenic activities (Costello et al., 2010; Halpern et al., 2015). In 2010, the  
56 United Nations' Convention on Biological Diversity (CBD) established a target of 10% of  
57 the ocean to be protected by 2020 ("Aichi Target 11") (CBD, 2010). MPAs are an essential

58 tool for reversing the global degradation of ocean life (Claudet et al. 2008; Babcock et al;  
59 2010). Several studies have shown that protection from fishing leads to rapid increases in  
60 abundance, size and biomass of exploited species and, sometimes, to an increase in  
61 species diversity (e.g. Claudet et al., 2010; Di Franco et al., 2018; Lester et al., 2009).  
62 However, only 3.7% of the world's ocean is protected with implemented marine protected  
63 areas (MPAs) (Sala et al. 2018).

64 MPAs can also provide socioeconomic benefits. Economic benefits may stem from  
65 the creation of employment opportunities through the establishment of non-consumptive  
66 activities such as tourism and recreation (Roncin et al. 2008), or from securing future jobs  
67 by increasing the chances of managing stocks sustainably (Sumaila et al., 2000). Fisheries  
68 benefits arise from ecological effects within protected areas spillover the boundaries of the  
69 MPA (Di Lorenzo et al. 2016). Empirical studies comprising artisanal (Stelzenmüller et al.,  
70 2008), recreational (Font et al., 2012a) and industrial bottom-trawling fishing effort  
71 (Murawski et al., 2005) showed a concentration of fishing activities in the close vicinity of  
72 the MPA boundaries. This concentration of fishing effort, also known as “fishing the line”,  
73 can reduce spillover of fish to the surrounding area and can have major implications for the  
74 effectiveness of MPAs in achieving ecological and socio-economic goals (Wilcox and  
75 Pomeroy, 2003), particularly those concerning wider stock replenishment and fisheries  
76 sustainability at a regional level.

77 Most MPAs are multiple-use MPAs (Claudet 2018). They combine different levels of  
78 protection within a spatially zoned management scheme that can be fully protected areas  
79 (FPAs), where all extractive activities are prohibited, or a type of partially protected areas  
80 (PPAs), where some extractive activities are allowed but with varying restrictions (Horta e  
81 Costa et al., 2016; Lubchenco and Grorud-Colvert, 2015) (Giakoumi et al., 2017). How  
82 multiple use MPAs are management can have strong implications in terms of the  
83 ecosystem and fisheries benefits they can provide at a more regional levels. Recent  
84 studies showed that ecological benefits can be observed in fully and highly protected  
85 areas, while lower levels of protection do not provide benefits or only when surrounding a  
86 fully protected area (Zupan et al. 2018a). In addition, when allowed in a given zone of an  
87 MPA, extractive use concentrate and can become a threat for the overall MPA (Zupan et  
88 al. 2018b)

89 While MPAs are an ecosystem-based management tool, it is still unclear how the  
90 functioning of ecosystems responds to protection. In particular, do the different levels of

91 protection in multiple-use MPAs translate into different reorganization of ecosystems? How  
92 the ecosystem response of different levels of protection transfer into fisheries benefits?  
93 Can the local ecosystem and fisheries response of multiple use-MPAs scale-up at a  
94 regional level? Here, using food-web modeling techniques, we present the first attempt to  
95 quantitatively model and compare ecosystem structural and functional trait responses to  
96 different levels of protection in multiple-use MPA. We used three Mediterranean MPAs as  
97 a case study and developed ecosystem models for each management unit in each MPA.

## 98 **2. Material & Methods**

### 99 **Study area**

100 We examined different management units (MUs) of three different MPAs in the  
101 northwestern Mediterranean Sea: the Cerbère-Banyuls MPA in France, and Cap de Creus  
102 and Medes Islands MPAs in Spain (Figure 1). Cerbère-Banyuls MPA is located in the  
103 southeastern part of France, and it covers a coastal area of 6.50 km<sup>2</sup>. This marine area  
104 spans between 0 and 60 meters and it contains three main Mediterranean habitats:  
105 calcareous seaweed bottom (*Lithophyllum*), Mediterranean seagrass meadows (*Posidonia*  
106 *oceanica*) and coralligenous bottom (*Pseudolithophyllum expansum*). There are also some  
107 protected and emblematic species that inhabit in this MPA, such as the grouper  
108 (*Epinephelus marginatus*) and the red coral (*Corallium rubrum*). Cap de Creus MPA is  
109 located in the northeastern part of Spain, it covers a coastal area of 8.19 km<sup>2</sup> and spans  
110 between 0 and 60 meters. Shallow marine bottoms surrounding the peninsula of Cap  
111 Creus are mostly rocky whereas most deep areas are muddy (Sardá et al., 2012). The  
112 coastal rocky bottoms of the Cap de Creus continental shelf are dominated by the  
113 community of coralligenous, which is mainly characterized by coralline algae and  
114 suspension feeders species (gorgonians, sponges, and bryozoans) (Iacono et al., 2012).  
115 Finally, the Medes Islands MPA is located in the northeastern part of Spain, and it covers a  
116 coastal area of 4.64 km<sup>2</sup>. Medes Islands MPA has a depth range between 0 and 60  
117 meters, and it contains a great diversity, constituting a complex ecosystem due to the  
118 existence of many different types of habitats such as *Posidonia oceanica* meadows,  
119 gorgonian communities (*Eunicella singularis* and *Paramuricea clavata*) and coralligenous  
120 bottom (Gili, 1981; Ros and Gili, 2015). It also shelters other important marine species, like  
121 the grouper and red coral (Hereu et al., 2012).

122 The main reason to select these three MPAs was their similar bathymetry range  
123 and habitat composition, and the proximity among them. The three MPAs combine

124 different levels of protection within a spatially zoned management scheme (Table 1 and  
125 Figure 1). Fully protected areas (FPAs) are areas where extractive uses are not allowed.  
126 Partially protected areas (PPAs) allow restricted uses such as traditional/artisanal fisheries  
127 or scuba-diving (Zupan et al. 2018b). Finally, we selected the unprotected areas (UPAs)  
128 surrounding the MPAs, in order to model a similar ecosystem without protection but with  
129 immediate influence from the protected region. The boundaries of the unprotected areas  
130 were set taking into account that they should have similar characteristics to the MPA, and  
131 they should be adjacent to the MPA (Figure 1).

## 132 **Modelling approach**

133 MU models were developed using Ecopath with Ecosim approach (EwE 6.6  
134 version) (Christensen et al., 2008; Christensen and Walters, 2004) and followed the best  
135 practices rules (Heymans et al., 2016). Ecopath is a mass-balanced model based on two  
136 equations. The first master equation describes the energy balance for each group, so that:

$$137 \text{ Consumption} = \text{production} + \text{respiration} + \text{unassimilated food}$$

138 The second Ecopath equation is based on the assumption that the production of  
139 one functional group is equal to the sum of all predation, non-predatory losses, exports,  
140 biomass accumulations, and catches, as expressed by the following equation:

$$141 \quad P/B_i \cdot B_i = P/B_i \cdot B_i \cdot (1 - EE_i) + \sum_j (Q/B)_{ji} \cdot B_i \cdot DC_{ji} + Y_i + NM_i + BA_i \quad \text{Eq. 1}$$

142 where  $B_i$  is the biomass,  $(P/B)_i$  is the production rate,  $(Q/B)_i$  is the consumption rate,  $DC_{ji}$   
143 is the fraction of prey  $i$  included in the diet of predator  $j$ ,  $NM_i$  is the net migration of prey  $i$ ,  $BA_i$   
144 is the biomass accumulation of prey  $i$ ,  $Y_i$  is the catch of prey  $i$ , and  $EE_i$  is the ecotrophic  
145 efficiency of prey  $i$ , that is, the proportion of production used in the system or exported.

146

## 147 **Parametrization**

148 All the models were built using the best available information and represent periods  
149 from 2000s to 2010s, mostly limited by the available biomass data on the underwater  
150 visual census (UVC). Specifically, the models we built of the Cerbère-Banyuls MPA  
151 included most of the information from 2013, while the Cap de Creus MPA and the Medes  
152 Islands MPA included most of their information from the period (2005-2008) and (2000-  
153 2004), respectively.

154 Information about species presence and their biomasses were aggregated in  
155 functional groups (FGs) of species or groups of species clustered according to key  
156 information about their trophic ecology, commercial value, and abundance in the  
157 ecosystem. We used the meta-web structure defined for the Western Mediterranean Sea  
158 model (Coll et al., 2019a) developed under the SafeNet Project<sup>1</sup> context, which contains  
159 ninety functional groups. We adapted this meta-web structure to local conditions and we  
160 removed those FGs which did not occur in the study areas. The final food-web structure of  
161 Cerbère-Banyuls MPA contains 64 functional groups (2 marine mammals, 3 seabirds, 1  
162 sea turtle, 8 pelagic fish, 25 demersal fish, 3 cephalopods, 14 invertebrates, 2 primary  
163 producers, 2 zooplankton, 2 phytoplankton and 2 detritus), and 67 functional groups for  
164 Cap de Creus MPA and Medes Islands MPA (2 marine mammals, 3 seabirds, 1 sea turtle,  
165 9 pelagic fish, 25 demersal fish, 3 cephalopods, 14 invertebrates, 4 primary producers, 2  
166 zooplankton, 2 phytoplankton and 2 detritus) (Table 2). Except in the case of FPAs, which  
167 do not have discards because all fishing extractions are forbidden, food-web structures of  
168 each MU in the same MPA were identical.

169 FGs' biomasses were obtained from different sources from the study area or  
170 surrounding areas (see supplementary material Table S1.1. for details on the  
171 parameterization of each functional group). The nine MU models share biomass data on  
172 some FGs (marine mammals, seabirds, sea turtles, pelagic fish, some invertebrates'  
173 groups, primary producers, zooplankton and phytoplankton) due to the lack of local data  
174 and/or the closeness among these MPAs. The rest of the FGs were parameterized with  
175 local information available from field studies (supplementary material Table S1.1.).

176 Biomass estimates for invertebrates' species were not frequently available and  
177 therefore we used realistic EE values to estimate the biomass of 4 FG in Cerbère-Banyuls  
178 models and 6 in Cap de Creus and Medes Islands models (supplementary material Table  
179 S1.1.). Scaling factors were used on biomass data of pelagic fish groups available from the  
180 MEDITS oceanographic survey (Bertrand et al., 1997), which does not fully cover coastal  
181 areas. Thus, this scaling enabled to adapt the estimates and get more accurate biomass  
182 estimations for these coastal areas. The scaling conversions were based on species depth  
183 distribution (extracted from *Aquamaps* – [www.aquamaps.org](http://www.aquamaps.org)) (Kaschner et al., 2013) and  
184 took into account their contribution in each FG. MEDITS trawling survey data were  
185 extracted with a new released software inside SafeNet Project, which is called MEDITS

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<sup>1</sup> <http://www.criobe.pf/recherche/safenet/>

186 data explorer v. 1.5.1 (Steenbeek, 2018a). This tool allowed us to weight the biomass by  
187 bathymetric range. We extracted MEDITS trawling survey data from the closest point to  
188 each MPA, in order to have biomass data for those species that were not present during  
189 UVC.

190 Production ( $P/B$ ,  $\text{year}^{-1}$ ) and consumption ( $Q/B$ ,  $\text{year}^{-1}$ ) rates were either estimated  
191 using empirical equations (Heymans et al., 2016), taken from literature or from other model  
192 developed in the Mediterranean Sea (Coll et al., 2019b) (supplementary material Table  
193 S1.1.). Additionally, local body lengths of reef-associated species obtained from UVC (Di  
194 Franco, 2017) were used to estimate those rates using empirical equations and local data  
195 (Pauly, 1980). In some cases, these produced different rates between MUs due to  
196 changes in the length frequency of species in protected versus not protected areas.

197 The diet information was compiled using published studies on stomach content  
198 analyses, giving preference to local or surrounding areas (supplementary material Table  
199 S1.1.). To calculate the Diet Matrix (DC) we used the Diet matrix calculator (Steenbeek,  
200 2018b). This tool automatizes the process of selecting and scaling diet data, and it  
201 generates the diet composition matrix and a pedigree index associated with each predator  
202 FG (supplementary material appendix 2, Table S2.1., S2.2., S2.3.). Due to the small sizes  
203 of the MUs investigated and the capacity to some species to move between MUs (Gell and  
204 Roberts, 2003a; Grüss et al., 2011), we set a fraction of the diet composition as import for  
205 all MUs based on the time that these species feed outside the areas and their ecological  
206 traits. This import was based on the size, behavior and ecology of species of each  
207 functional group (Froese and Pauly, 2019).

208 Fisheries data were obtained from different sources (database, literature and  
209 unpublished data) (supplementary material Table S1.1.) and were split into two fishing  
210 fleets – recreational and artisanal (except for FPA models where fishing activities are not  
211 allowed). Regarding artisanal catches, Cerbère-Banyuls catches were obtained from a  
212 local study (Prats, 2016), while for Cap de Creus and Medes Islands the data came from  
213 an official dataset of the regional government of Catalonia managed by the Institute of  
214 Marine Sciences (ICM-CSIC) (Tudó, 2017). For Cap de Creus, these landings were from  
215 Llançà, Port de la Selva, Cadaqués and Roses harbours, where the main fleet operating is  
216 coming from, and were scaled by the months fishing inside the MPA (Gómez et al., 2006).  
217 For the recreational fisheries, we extracted catches from Ivanhoff *et al.* (2010) in Cerbère-  
218 Banyuls. Cap de Creus catches came from Font and Lloret (2011a, 2011b) and Lloret *et*



219 *al.* (2008a, 2008b). Whereas, for Medes recreational catches were collected from Sacanell  
220 (2012).

## 221 **Ensuring mass-balance and assessing the quality of the model**

222 In an Ecopath model, the energy input and output of all functional groups must be  
223 balanced under some ecological and thermodynamic rules: (1)  $EE < 1.0$ ; (2)  $P/Q$   
224 [production/consumption rate or gross efficiency (GE)] ranges from 0.1 to 0.3 with the  
225 exception of fast growing groups such as bacteria; (3)  $R/A$  (respiration/food assimilation)  $<$   
226 1; (4)  $R/B$  (respiration/biomass) ranges from 1 to 10 for fish and higher values for small  
227 organisms; (5)  $NE$  (net efficiency of food conversion)  $>$   $GE$  and (6)  $P/R$   
228 (production/respiration)  $<$  1 (Christensen et al., 2008; Heymans et al., 2016).

229 Initial values of the MU models showed that the  $EE > 1$  for some functional groups:  
230 mainly small pelagic fish groups and invertebrates' groups. This could be due to unfeasible  
231 low initial biomass estimates, and two procedures were applied to solve  $EE$  higher than 1.  
232 For pelagic fish groups (mackerels, horse mackerels, European sardine, European  
233 anchovy and other small pelagic fish), we added immigration biomass because studied  
234 areas (MUs) are relatively small compared to the dispersal rate of those species (Di  
235 Franco et al., 2012). For invertebrate' groups (non-commercial decapods, sea cucumbers  
236 and other macrobenthos),  $EE$  values were fixed to 0.8 in order to calculate their biomass.

237 To balance the models, we applied a manual mass-balanced procedure following a  
238 top-down approach modifying input parameters starting from the functional groups with  
239 higher TL and considering the best practice guidelines (Heymans et al., 2016). We used  
240 the PREBAL diagnostics (Link, 2010) in order to ensure that the model followed general  
241 ecological and biological principles and to guide the balancing procedure. Several  
242 functional groups displayed too low  $P/Q$  values ( $<0.2$  or even  $<0.1$  in some cases), and in  
243 these cases reasonable  $P/Q$  values were assigned based on a previous EwE model  
244 developed in the same study area (Corrales et al., 2015).

245 The quality of the model was evaluated using the pedigree routine, which allows  
246 assigning a pedigree value for each input parameter (B,  $P/B$ ,  $Q/B$ , diet and catches)  
247 (Christensen et al., 2008; Christensen and Walters, 2004). All pedigree values were  
248 established manually except for diet pedigree values, which were obtained from the Diet  
249 Calculator software (Steenbeek, 2018b). This software computes a total pedigree value for  
250 each diet record, which is a weighted average of four field scores from four diet features

251 (region, year, type of data and method). Pedigree values were first used to determine  
252 which parameters were of lower quality and thus could be modified during the balancing  
253 procedure. Afterwards, they were used to calculate the pedigree index of each model,  
254 which vary between 0 (lowest quality) and 1 (highest quality) (Christensen and Walters,  
255 2004).

## 256 **Model analyses and ecological indicators**

### 257 ***Flow diagram***

258 The food-web structure of each MU in the three MPAs was visualized using a flow  
259 diagram. Flow diagrams were obtained using the *ggplot2* package (Wickham, 2010)  
260 implemented in R software (R Core Team, 2017) and were built from the biomass and  
261 trophic levels (TL, as outputs) of each FG, and the direct trophic interactions among them.  
262 The TL identifies the position of organisms within food webs by tracking the source of  
263 energy for each organism, and it is calculated by assigning primary producers and detritus  
264 a TL of 1 (e.g. phytoplankton), and consumers to a TL of 1, plus the average TL of their  
265 prey weighted by their proportion in weight in the predator's diet (Christensen, 1996).

### 266 ***Ecological indicators***

267 Several ecological indicators were computed to describe the state and functioning  
268 of the ecosystems. These indicators were divided into five main groups following (Coll and  
269 Steenbeek, 2017):

270 **Biomass-based.** These indicators are calculated from the biomass of components.  
271 Species biomass data are considered basic information to evaluate effectiveness in marine  
272 protected areas (Micheli et al., 2004). We included five biomass-based indicators: total  
273 biomass (TB,  $t \cdot km^{-2} \cdot year^{-1}$ ), biomass of commercial species (CB,  $t \cdot km^{-2} \cdot year^{-1}$ ), biomass of  
274 fish species (FB,  $t \cdot km^{-2} \cdot year^{-1}$ ), and Kempton Q diversity index (KI).

275 **Trophic-based.** These indicators reflect the TLs for different groups of the food web. The  
276 trophic level may indicate ecosystem “health” because it reflects fishing pressure due to  
277 the removal of predators (Christensen and Walters, 2004). We selected four trophic-based  
278 indicators: TL of the community (TLc), TL of the community including organisms with TL  $\geq$   
279 2 (TL2), TL of the community including organisms with TL  $\geq$  3.25 (TL3.25) and TL of the  
280 community including organisms with TL  $\geq$  4 (TL4).

281 **Species and size-based.** These indicators are based on species traits and conservation  
282 status. Increase in species traits such as mean length could be a direct effect of marine  
283 protected areas (Claudet et al., 2006). We selected three species-based indicators:  
284 biomass of IUCN-endangered species in the community (ES, t·km<sup>-2</sup>·year<sup>-1</sup>), mean length of  
285 fish in the community (ML, cm) and mean life span of fish in the community (MLS, year).

286 **Flow-based.** We used the Total System Throughput (TST, t·km<sup>-2</sup>·year<sup>-1</sup>), the sum of all  
287 flows in the model (consumption, export, respiration, and flow to detritus) and considered  
288 an overall measure of the “ecological size” of the system (Finn, 1976). Finn’s Cycling  
289 Index (FCI, %), the fraction of the ecosystem’s throughput that is recycled (Finn, 1976).  
290 Average Path Length (APL) defined as the average number of groups that flows passes  
291 through and is an indicator of stress (Christensen, 1995). Several additional indicators  
292 were selected because of their robustness in front of models’ comparison (Heymans et al.,  
293 2014): the ratios of consumption (Q), export (Ex) and production (P).

294 **Catch-based.** These indicators are based on catch and discard species data in the food  
295 web. They can give an idea on the potential effect on adjacent fisheries through spillover  
296 of exploited fishes from FPAs (McClanahan and Mangi, 2000). We included six indicators:  
297 total catch (TC t·km<sup>-2</sup>·year<sup>-1</sup>), total discarded catch (TD, t·km<sup>-2</sup>·year<sup>-1</sup>), trophic level of the  
298 catch (TLC), intrinsic vulnerability index of catch (VI), mean length of fish in the catch  
299 (MLC, cm) and mean life span of fish in the catch (MLC, year).

300 Additionally, the mixed trophic impact (MTI) analysis was performed to quantify  
301 direct and indirect trophic interactions among functional groups (Ulanowicz and Puccia,  
302 1990). This analysis quantifies the direct and indirect impacts that a hypothetical increase  
303 in the biomass of one functional group would have on the biomasses of all the other  
304 functional groups, including the fishing fleets. The MTI for living groups is calculated by  
305 constructing an n × n matrix, and quantifying each interaction between the impacting group  
306 (j) and the impacted group (i) is:

$$307 \quad MTI_{ji} = DC_{ji} - FC_{ij}, \quad \text{Eq. 2}$$

308 where  $DC_{ji}$  is the diet composition term expressing how much  $i$  contributes to the diet of  $j$ ,  
309 and  $FC_{ij}$  is a host composition term giving the proportion of the predation on  $j$  that is due to  
310  $i$  as a predator.

311 To identify the keystone species within the ecosystem, we estimated the  
312 keystone index (KS) of the most important reef FG (Common pandora: *Pagellus*

313 *erythrinus*; Sparidae; White seabream: *Diplodus sargus*; Common two-banded seabream:  
 314 *Diplodus vulgaris*; Common dentex: *Dentex dentex*; Red scorpionfish: *Scorpaena scrofa*;  
 315 Groupers: *Epinephelus sp*; Brown meagre: *Sciaena umbra*; Labridae and Serranidae;  
 316 Other commercial medium demersal fish; Salema: *Salpa salpa*; Mugilidae; Red mullet:  
 317 *Mullus barbatus*; and Surmulet: *Mullus surmuletus*) using 3 different methods. A keystone  
 318 species is a species that shows relatively low biomass but has a relatively important role in  
 319 the ecosystem (Power et al., 1996). We used 3 indices: (1) Power keystone indicator  
 320 (1996) ( $KS_P$ ) and (2) Libralato's keystone indicator (2006) ( $KS_L$ ), which are based on a  
 321 measure of trophic impact derived from the MTI analysis, and a quantitative measure of  
 322 biomass; and (3) Valls (2015) ( $KS_V$ ) keystone index, in which the biomass component  
 323 is based on a descending ranking. These indices are calculated as:

$$324 \quad KS_{Pi} = \varepsilon_i \frac{1}{p_i} \quad \text{Eq. 3}$$

$$325 \quad KS_{Li} = \log[\varepsilon_i(1 - p_i)] \quad \text{Eq. 4}$$

$$326 \quad KS_{Vi} = \log[IC_i \cdot BC_i] \quad \text{Eq. 5}$$

327 where  $\varepsilon_i$  represents the RTI;  $p_i$  is the contribution of the group  $i$  to the total biomass in the  
 328 food-web;  $IC_i$  is a component estimating the trophic impact of the group  $i$ ;  $BC_i$  is a  
 329 component estimating the biomass of the group  $i$ .

### 330 ***Evaluating MPAs and the role of MUs***

331 In order to determine the role of MUs in the functioning of the MPAs, results from  
 332 ecological indicators (except catch-based indicators) and keystone species were  
 333 compared among three MUs within an MPA. This comparison among MUs ecological  
 334 indicators served to capture shifts on ecosystem structure and functioning due to the level  
 335 of protection (Fortin and Dale, 2005).

336 To evaluate the differences among the three studied MPAs, results from ecological  
 337 indicators (except catch-based indicators) and keystone species were compared with  
 338 the same MUs of different MPAs. For instance, considering the FPA of three MPAs  
 339 allowed us to capture how different are these MPAs since they officially offer the same  
 340 level of protection. Despite that each MPA differs on restriction in their PPA, this multiple  
 341 comparison procedure was developed for the three MUs because the ecological theory  
 342 establishes that reserve effects should extend from FPA to beyond its boundaries as a  
 343 result of spillover (Gell and Roberts, 2003b).

### 344 ***Impact of small-scale fisheries***

345 To evaluate the impact of small-scale fisheries on the MPAs, catch-based  
346 indicators were examined between PPAs and UPAs of the three MPAs. The mixed trophic  
347 results of recreational and artisanal fishing fleets were examined to quantify the direct and  
348 indirect impact of each fleet on the functional groups for PPAs and UPAs of studied MPAs,  
349 and their potential competition and trade-offs between them.

### 350 **3. Results**

351 The pedigree index values of the MU models showed similar values among them  
352 (Figure 2), ranging from 0.41 to 0.52. The highest pedigree values were obtained for  
353 Cerbère-Banyuls, and regarding MUs, FPAs obtained slightly lower pedigree index than  
354 the other MUs.

355 The flow diagrams displayed high levels of biomass even for some high trophic level  
356 groups, especially in FPAs (Fig. 3). Also, results emphasized the complexity of these food  
357 webs regarding the number of trophic links among functional groups with important fluxes  
358 of energy from phytoplankton (FG 61 and 62) and detritus (FG 63) up to the food web.

### 359 **Ecosystem structure and functional traits**

360 Overall, biomass-based indicators displayed the same pattern between MUs  
361 (Figure 4), with the highest biomass values found in Cerbère-Banyuls, then Medes Islands  
362 and finally Cap de Creus. Conversely, the Kempton's Biodiversity Indicator (KI) decreased  
363 from Cap de Creus to Medes Islands and then Cerbère-Banyuls, which had the lowest KI  
364 value. Within MUs, the FPAs showed the highest values in terms of total and fish biomass,  
365 and they were followed by PPA in all MPAs. In Cap de Creus we observed similar values  
366 among MUs for both indicators.

367 Trophic-based indicators revealed that in general the TL of the community and TL2  
368 was higher for Cerbère-Banyuls, followed by Medes Islands, while Cap de Creus showed  
369 the lowest levels (Figure 5). Cerbère-Banyuls and Medes Islands displayed similar values  
370 for TL325 and TL4, while Cap de Creus obtained higher variance among these indicators.  
371 Specifically, Cap de Creus showed the highest TL325 values and the lowest TL4 value.  
372 Within MUs, most trophic-based indicators showed the highest values for FPAs followed  
373 by PPAs and UPAs. However, FPA in Cap de Creus displayed the lowest value of TL4  
374 (Fig. 5).

375 Flows-based indicators showed differences between MPA. Cerbère-Banyuls  
376 showed the highest values for Q/TST, TST, FCI and APL, while the lowest values for  
377 Ex/TST and P/TST (Figure 6). Inversely Cap de Creus showed the lowest values for  
378 Q/TST, whilst the highest values for Ex/TST and P/TST. Medes Islands values were  
379 located between these two MPAs. Considering MU values, flow-based results revealed  
380 higher values for FPAs followed by PPAs in most of the flows-based indicators. As an  
381 exception, Ex/TST and P/TST were higher for UPAs and lower for FPAs.

382 Results from species and size-based indicators (Figure 7) pointed out that Cerbère-  
383 Banyuls got the highest values for IUCN species B, followed by Medes Islands. Regarding  
384 ML and MLS, the highest values were also obtained for Cerbère-Banyuls, followed by Cap  
385 de Creus and then Medes Islands. Species and size-based indicators were higher for  
386 FPAs, except in Cap de Creus, where PPA had higher values of ML and MLS than FPA.

387 All keystone indices (Figure 8; Supplementary material appendix 3) for the  
388 nine models identified the same groups as keystone: Groupers, Other commercial medium  
389 demersal fish, Common dentex and Labridae & Serranidae. Particularly, Groupers  
390 obtained the highest relative total impact. These results confirmed that these functional  
391 groups play an important ecological role in these Mediterranean coastal ecosystems.  
392 Keystone indices showed different patterns among MUs and MPAs. Medes Islands  
393 models obtained the highest number of keystone species followed by Cerbère-Banyuls  
394 (Figure 8).

### 395 **Role and impact of small-scale fisheries**

396 Total catch and discards showed similar values between Cap de Creus and Medes  
397 Islands, while Cerbère-Banyuls presented the lowest values (Figure 9). IV values were  
398 similar for Cap de Creus and Cerbère-Banyuls, and the lowest IV values were found for  
399 Medes Islands. TLc was higher for Cerbère-Banyuls and Cap de Creus and lower for  
400 Medes Islands (Figure 9), and these results are in line with MLSc results (Figure 7). Also,  
401 Cerbère-Banyuls obtained the highest value for MLc while Cap de Creus got the lowest  
402 values. TC and discards exhibited higher values for PPAs than UPAs. IV indexes were  
403 higher for PPAs than UPAs except for Cerbère-Banyuls. TLc values were similar between  
404 PPA and UPA in Cap de Creus and Medes Islands, while it was higher for PPA in Cerbère-  
405 Banyuls. ML and MLS results evidenced similar values between MUs for Medes Islands,  
406 while PPA obtained higher values in Cap de Creus and lower in Cerbère-Banyuls.

407           The MTI analysis based on the recreational fisheries (Figure 10) revealed that the  
408 impact of recreational fisheries to the functional groups in the ecosystem were clearly  
409 higher than the impacts of these groups to recreational fisheries (between -0.86 and 0.84  
410 and -0.15 and 0.26, respectively). The highest impacting and impacted values were found  
411 for PPA in Cap de Creus, where impacted values were higher for PPA in Cerbère-Banyuls  
412 (positively on Sparidae and negatively on Brown meagre, Common dentex and Groupers)  
413 and Medes Islands (positively on Common two-banded seabream and Red scorpionfish).  
414 Impacts on recreational fisheries were low (close to zero) for most groups except for  
415 Groupers and Sparidae in Cerbère-Banyuls due to their higher catches and thus their  
416 importance to recreational fisheries.

417           In line with impacts of recreational fisheries, impacts of artisanal fisheries (Figure  
418 11) were higher and more fluctuating than impacts of functional groups to artisanal  
419 fisheries (ranging from -0.80 to 0.77 and -0.32 to 0.12, respectively). Generally, artisanal  
420 impact results did not evidence great differences between MUs. The highest artisanal  
421 fisheries impacting values were obtained in Cerbère-Banyuls, with some exceptions for  
422 negative impacting values of some groups (Common pandora and Red mullet) in Cap de  
423 Creus. Artisanal fisheries impacted results displayed low values (close to zero), except for  
424 Groupers group which highly negatively impacted on the artisanal fishery of PPA in Cap de  
425 Creus.

426           Overall, the MTI analysis based on fisheries did not show any pattern among MUs  
427 and MPAs, and fisheries impacting values were clearly higher and more fluctuating than  
428 impacted values (ranging from -0.86 to 0.84 and -0.32 to 0.26, respectively). Mostly,  
429 impacts of recreational fisheries were higher for the PPA of Cap de Creus, while impacts of  
430 artisanal fleet were higher for Cerbère-Banyuls. Specifically, MTI analysis revealed that the  
431 impacts of recreational fisheries were greater (positively and negatively) for Brown meagre  
432 and Groupers. On the other hand, the recreational fishery was highly (positively) impacted  
433 by Other commercial medium demersal fish and Sparidae (Figure 10). Regarding artisanal  
434 fishery, Brown meagre, Other commercial medium demersal fish and Red mullet were  
435 positively impacted by recreational fisheries, while Common dentex and Groupers were  
436 negative impacted (Figure 11). Groupers was the most impacting (negatively) group on the  
437 artisanal fishery. Recreational and artisanal fishery obtained low impacted and impacting  
438 values between them. Among them, the highest impact was for the recreational impact on  
439 artisanal in the PPA of Cap de Creus (0.28).

#### 440 **4. Discussion**

441 We built nine quantitative models to investigate the differences among MUs of  
442 three MPAs in the NW Mediterranean Sea. To our knowledge, this is the first attempt to  
443 develop a food-web model for each MUs within MPA to assess the impact of protection at  
444 local scales at the ecosystem level.

445 The pedigree index values revealed that the input data were qualitatively acceptable  
446 compared to the distribution of pedigree values in other existing models (Lassalle et al.,  
447 2014; Morissette, 2007). Also, the pedigree values of the models was comparable to that  
448 from the available MPA model in the Western Mediterranean Sea (e.g., 0.49 in (Prato et  
449 al., 2016b). However, FPAs showed the lowest pedigree values, although they are aimed  
450 to be protected and well monitored. Overall, these results highlight the need to further  
451 study and monitor MPAs within the Mediterranean Sea.

452 The flow diagram showed differences among MUs. Although the TL were similar  
453 among three MUs in each MPA, some commercial functional groups (e.g. FG 26 –  
454 Groupers in Cerbère-Banyuls) showed higher values for FPAs. This pattern could be due  
455 to the effect of protection in these areas which may be connected with the complexity of  
456 the food web and the maturity of the ecosystem (Odum and Barrett, 1971).

457 Ecological indicators showed differences among MUs and pointed out de strong  
458 benefits of FPAs (Sala et al., 2017). FPAs (also known as no-take areas) are widely  
459 recognized as a powerful tool for an ecosystem approach to fisheries management and for  
460 biodiversity conservation (Claudet et al., 2008) and several studies described their positive  
461 effects on biomass (Guidetti et al., 2014), trophic levels (Guénette et al., 2014), mean  
462 length (Claudet et al., 2006), mean life span (Guénette and Pitcher, 1999) and biomass of  
463 IUCN species such as groupers (Claudet et al., 2008). In our study, consumption rate over  
464 total system flows was higher in FPAs than PPA and UPA, because of the higher the  
465 biomass in the ecosystem the higher the consumption in the ecosystem. Libralato *et al.*  
466 (2010) found similar results for another Mediterranean MPA due to the effect of protection.  
467 The same pattern was found for TST, APL and FCI, in which the value of the indicator  
468 increases with the level of protection. These indicators suggested lower stress, more  
469 maturity, more ecological size and higher resilience (Christensen, 1995) for FPAs  
470 ecosystems. In line with these results, Sala *et al.* (Sala et al., 2017) highlighted the  
471 potential benefits of FPAs and pointed out that these areas are more resilient than UPAs.  
472 In addition, most biomass-based, trophic-based, species-based and flows-based results



473 obtained from PPAs demonstrated their role as buffer zones (Giakoumi et al., 2017), so  
474 they may confer biomass enhancement compared to UPAs although FPAs produce  
475 greater benefits (Lester and Halpern, 2008; Sciberras et al., 2015).

476 In contrast, the biodiversity index (KI) did not show the expected pattern as  
477 biodiversity is expected to increase with protection (Costello and Ballantine, 2015). This  
478 controversial result could be due to the available data, which came from different studies  
479 for each MPA and year. Therefore as more exhaustive species biomass data are available,  
480 the biodiversity index becomes more reliable (Claudet, 2013; Hereu Fina et al., 2017). In  
481 addition, export ratio and production ratio results showed higher values for UPAs because  
482 the biomass of several FGs (such as Sparidae or Groupers) were quite high to support  
483 their feeding rates. These FGs would migrate beyond the boundaries of the modelled  
484 ecosystem (MU), which are relatively small compared to the dispersal rate of those  
485 species (Di Franco et al., 2012).

486 The indicators displayed differences among MUs within each MPA, especially in  
487 the case of Cerbère-Banyuls and Medes Islands MPAs, but not for Cap de Creus.  
488 Probably, these pattern could be explained by their enforcement, reported to be a key  
489 factor to promote direct and indirect reserve effects (Guidetti et al., 2008). The lack of  
490 enforcement is one of the most relevant issues concerning MPAs in the Mediterranean  
491 context (Fenberg et al., 2012). Claudet and Guidetti (2010) recognized that an MPA  
492 without enforcement and compliance is just a paper park and no reserve effects can be  
493 expected. This could be the case of Cap de Creus, in which our results did not show the  
494 same pattern found for the other two MPAs. In fact, Lloret *et al.* (2008b, 2008a) reported a  
495 lack of enforcement and a low level of compliance in Cap de Creus, particularly on  
496 minimum landing sizes of certain species or lacking fishing license. This is in contrast with  
497 Cerbère-Banyuls and Medes Islands MPAs, in which ensuring compliance and  
498 enforcement promote a high level of ecological effectiveness (Di Franco et al., 2014). Our  
499 results are also consistent with previous studies performed in Cerbère-Banyuls (Harmelin-  
500 Vivien et al., 2008) and Medes Islands (Harmelin-Vivien et al., 2008; Sala et al., 2012),  
501 which demonstrated reserve effects for those MPAs and reported higher biomass in FPAs  
502 with a rapid declining from FPAs outward. Giakoumi *et al.* (2017) revealed significant  
503 stronger biomass effect for FPAs than PPAs and higher fish density in older, better  
504 enforced, and smaller MPA. Considering that Cap de Creus has the smallest FPA and is  
505 the newest (Table 1) and least enforced MPA in the study, our results suggest that the

506 year of establishment and level of enforcement have a strong effect on MPA effectiveness.  
507 These pattern among MPAs were also found by E Costa *et al.* (2016) who presented a  
508 novel classification system for MPAs which ranges from 1 (fully protected areas) to 8  
509 (unprotected areas). In this scale, Cerbère-Banyuls obtained a rate of 4.7, being a highly  
510 protected area, and Medes 6.4, being less well protected (e Costa et al., 2016). Cap de  
511 Creus was not included in this study, but we could assume higher MPA index because of  
512 its small FPA and its lack of enforcement. Overall, our results call for enhancement on the  
513 regulations, increasing the surface of FPAs and the enforcement of management rules in  
514 Medes and Cap de Creus MPA.

515 All keystone indices for the nine models pointed out the same functional  
516 groups as keystone. Among them, Grouper and Common dentex were highlighted as  
517 keystone groups in previous western Mediterranean MPAs models (Prato et al., 2016a;  
518 Valls et al., 2012).  $KS_L$  (Libralato et al., 2006) clearly showed that the keystone role of  
519 species increases with the level of protection, so the highest  $KS_L$  values were found in  
520 FPAs. It demonstrates its possible application to capture biological and ecological effects  
521 of protection because it tends to underestimate the role of groups with low biomass (as  
522 Libralato et al., 2010).  $KS_P$  (Power et al., 1996) showed a different pattern because it  
523 assigns high keystone to FGs with high abundance allowing confusion between  
524 dominant and structuring species and overrepresenting rare species (Coll et al., 2013).  
525 The  $KS_V$  (Valls et al., 2015) showed higher values for species such as Groupers and  
526 Common dentex in non-fully protected areas (PPAs and UPAs) because it avoids  
527 underrepresenting rare species and aims at better balancing the two components of the  
528 keystone definition, impact and abundance.

529 MPAs are considered an important tool to manage coastal fisheries (Claudet et al.,  
530 2006), and enforcement is a key aspect to achieve these goals. Our results show that well-  
531 enforced MPAs can enhance small-scale fishery (artisanal and recreational) by spillover  
532 effect and promote the sustainability of local fisheries (Forcada et al., 2009; Goñi et al.,  
533 2011; Sala et al., 2013). According to the spillover effect, total catch and discards were  
534 higher in PPAs than in UPAs for all MPAs. Exceptionally, results of MUs in Cap de Creus  
535 differed substantially from Cerbère-Banyuls and Medes Islands. This could be related to  
536 the position of its FPAs, which is not located in the core of the MPA and surrounded by the  
537 PPA, which is the common MPA design in the Mediterranean Sea (Gabrié et al., 2012) and  
538 could reduce PPAs effectiveness. High values of total catch and discards in Cap de Creus

539 could be related to the non-compliance, as well as to the lack of geolocated catch data,  
540 which could have biased our results. Non-compliance could result in some fishing inside  
541 FPAs and PPAs, which may reduce the effectiveness of their potential biological,  
542 ecological and fisheries benefits (Roberts, 2000), in accordance with above conclusions  
543 obtained from other ecological indicators (e.g. Biomass-based indicators). In addition,  
544 estimates of IV and ML from Cap de Creus could be explained as a failure in the  
545 enforcement. On the other hand, we hypothesized that TLc and MLS values were obtained  
546 as a result of the targeted species by the recreational fishery in each MPA, and the  
547 different level of protection in PPAs (Ivanoff et al., 2010; Sacanell, 2012). Considering the  
548 concept of “fishing the line”, which can jeopardize the success of an MPA (Kellner et al.,  
549 2007), understanding spatial-temporal patterns of fishing effort around a MPA is a key  
550 aspect to manage and assess these areas. In this context, Stelzenmüller *et al.*  
551 (Stelzenmüller et al., 2008) found a local concentration of fishing effort around the borders  
552 of Cerbère-Banyuls and Medes Islands MPAs, in accordance with our results.

553         The mixed trophic impact analysis results highlighted the impact of recreational  
554 fisheries in Cap de Creus among the rest of studied MPAs, in agreement with the failure  
555 on the sustainable management and its contribution to adjacent fisheries because of non-  
556 enforcement (Lloret et al., 2008b, 2008a). PPAs showed higher impacted values than  
557 UPAs, which suggest a higher contribution of PPAs to the recreational fishery. Although  
558 PPAs may differ on their level of protection (Lester and Halpern, 2008), for example  
559 spearfishing is not allowed inside the PPA of Cerbère-Banyuls (Font et al., 2012b), these  
560 results confirmed their ecological effects and benefits for adjacent fisheries. For instance,  
561 recreational fishery impacted negatively on Groupers in all MPAs, which reflected that this  
562 is a target species. The recreational fishery is impacted positively and negatively by  
563 Groupers which could be explained by feeding behavior and fleet behavior such as  
564 competition between fishery and FG. Groupers were negatively impacted by artisanal  
565 fishery because of their catch, especially in Cerbère-Banyuls (Prats, 2016). Regarding the  
566 artisanal fishery, the highest negative impacting values were represented by targeted  
567 species for this fleet, while positive values may represent FGs which are prey of those  
568 targeted species. Cerbère-Banyuls obtained higher impacting values than other MPAs.  
569 Since it is a well enforced MPA, it could represent higher benefits or catches than other  
570 MPAs. In addition, most highlighted FGs by their high impacting values in artisanal fishery  
571 matched with the keystone species, which support that non-enforced marine protected  
572 areas may compromise positive effects of these ecosystems (Claudet and Guidetti, 2010).

573 Our results encourage fishermen compliance, which is key for fisheries' success such as in  
574 Torre Guaceto MPA, where co-management involving fishers, scientists and managers  
575 increased fisheries catches (Guidetti and Claudet, 2010).

576 Even though our work showed that quantitative food-web modelling techniques can  
577 be useful to assess MPAs effects on the structural and functional traits of marine  
578 ecosystems and their adjacent fisheries, some limitations were found during this study.  
579 One of the main hurdles was the lack of local data for some FGs identified in previous  
580 MPAs modelling studies (Prato et al., 2016b; Valls et al., 2012). For example, literature  
581 including benthonic biomass estimations inside MPAs is mainly focused on sea urchin,  
582 gorgonian and *Posidonia* (e.g.(Hereu et al., 2012; Schvartz and Labbe, 2012). However,  
583 studies about other important benthic groups such as sponges or crustaceans are scarce.  
584 Additionally, spatial-temporal series of catches and fleet distribution would improve the  
585 analysis on the effect of MPA potential benefits on the recreational and artisanal fishery.  
586 We did not find studies on artisanal and recreations fishery including estimates of temporal  
587 catches in Cap de Creus and Medes Islands MPAs. Collecting time series of fishing  
588 activities surrounding MPAs is a monitoring priority, as previously highlighted (Prato et al.,  
589 2016b; Valls et al., 2012). A recent published study (Lloret et al., 2019), which focused in  
590 the northwestern Mediterranean Sea, emphasized the importance of differentiating  
591 between fishing methods or gears when studying the impacts on vulnerable species in  
592 MPAs which could be accomplished if data are available. Moreover, our results could be  
593 biased by the oceanographic conditions and the structure of the modelled MPAs  
594 (Heymans et al., 2016). In addition, and despite that these MPAs are closely located, the  
595 biomass estimations to develop the models came from different reference years which  
596 could limit the comparison among MPAs as a result of different environmental conditions  
597 (Sala et al., 2012). The structure, habitat and design of the MPAs also could represent  
598 another limitation (Charton et al., 2000) because we are modelling different environments:  
599 archipelago, coastal with covered FPA and coastal exposed FPA.

600 Despite these limitations, this work represents to our knowledge the first attempt to model  
601 management units within a protected area and it provides the basis to assess the role of  
602 these areas within a network of MPAs using a food-web modelling approach. These results  
603 highlight the capability of the EwE modeling approach to capture protection effects in such  
604 small areas despite data limitations. Our results suggest that enforcement can have an  
605 impact regarding the potential benefits of MPAs at the local scale, and a lack of

606 enforcement is noticeable in surrounding areas. Protected areas can increase their  
607 maturity and resilience and show potential benefits for small-scale fisheries that act in their  
608 surroundings when these areas are well-enforced. Future assessments on the role of  
609 these MUs should assess their role within a network to quantify their impacts at a sub-  
610 regional (e.g. Northwestern Mediterranean) and regional (e.g. Western Mediterranean)  
611 geographic scales.

## 612 **Acknowledgments**

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## 617 **Supplementary data**

618 Additional Supplementary material may be found in the online version of this article:

619 **Appendix 1.** Supplementary tables: Cerbère-Banyuls , Cap de Creus and Medes MPAs  
620 functional groups species composition and methods and references used to estimate the  
621 basic input parameters of the nine Ecopath model (Table S1.1.); Input parameters and  
622 outputs estimate for Cerbère-Banyuls (Table S1.2.), Cap de Creus (Table S1.3.) and  
623 Medes Islands (Table S1.4.) MU models.

624 **Appendix 2.** Supplementary tables: Diet composition matrix for the MU models of  
625 Cerbère-Banyuls (Table S2.1.), Cap de Creus (Table S2.2.) and Medes Islands (Table  
626 S2.3.) MPA.

627 **Appendix 3.** Supplementary tables: Keystone indexes and Relative Total Impact values  
628 for the functional groups included in Cerbère-Banyuls (Table S3.1), Cap de Creus (Table  
629 S3.2) and Medes Islands (Table S3.3) MU models. FPA: Fully Protected Area; PPA:  
630 Partially Protected Area; UPA: Unprotected Area.

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632 **References**

633 Automatic citation updates are disabled. To see the bibliography, click Refresh in the Zotero tab.

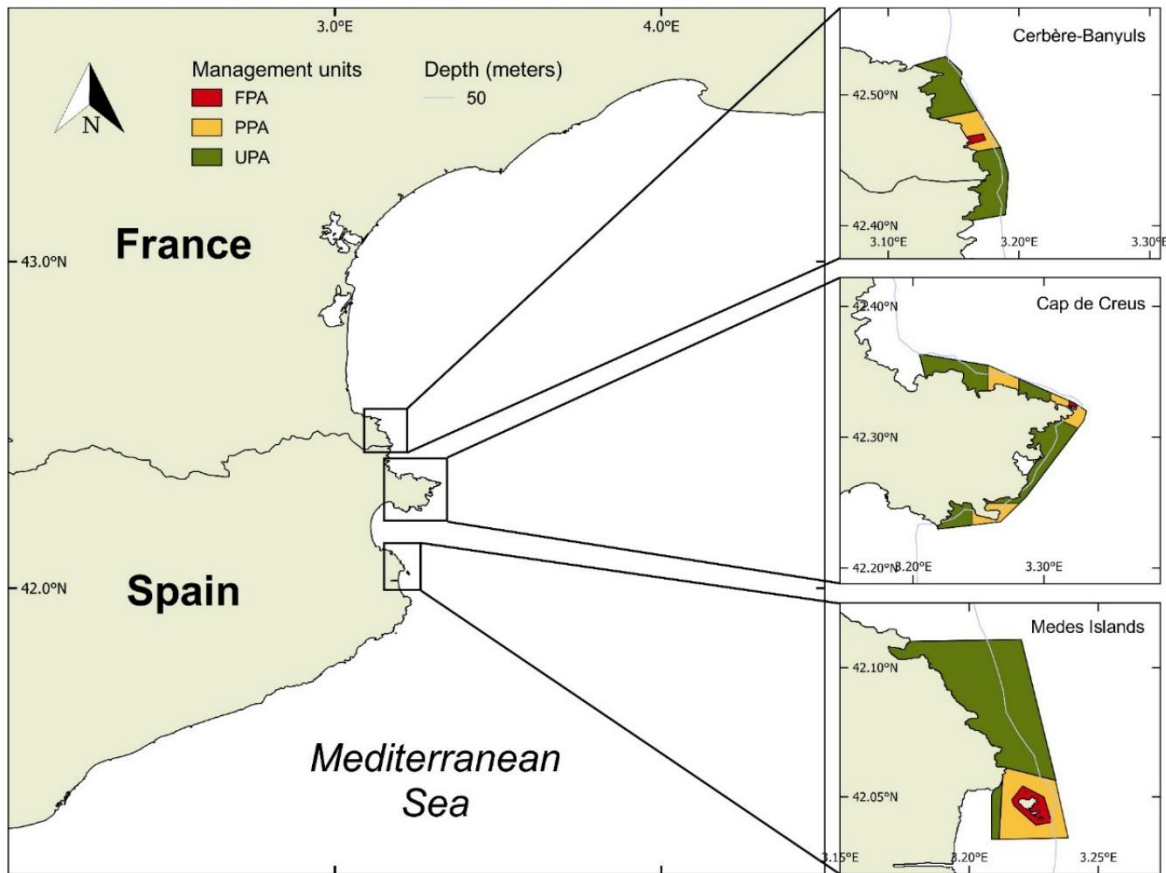
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636 **Figures and Tables**

637

638 **Figure captions**



639

640 **Figure 1.** Location of three modelled MPAs and their MUs (FPA: fully protected area; PPA:  
641 partially protected area; UPA: unprotected area) in Cerbère-Banyuls, Cap de Creus and  
642 Medes Islands.

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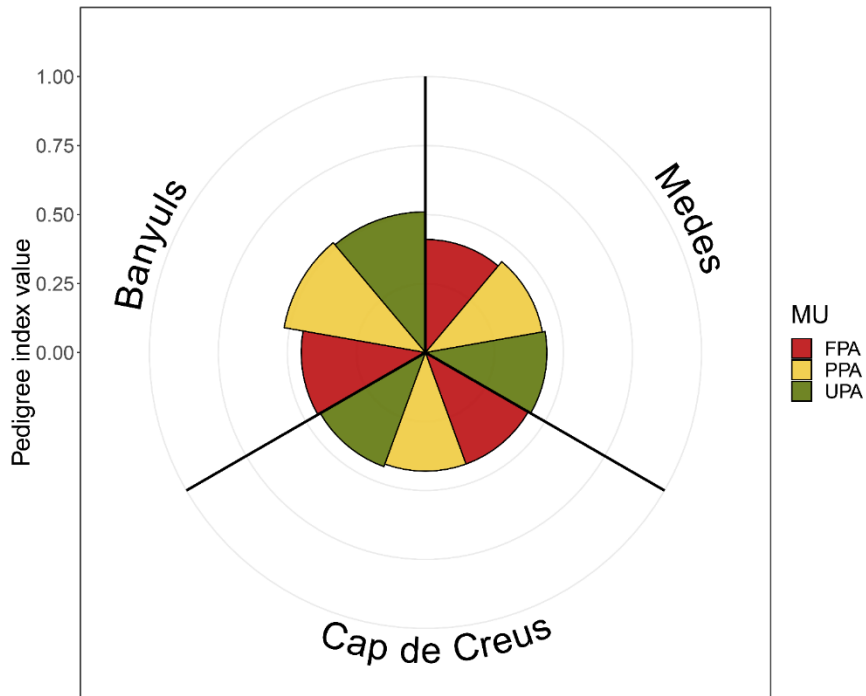
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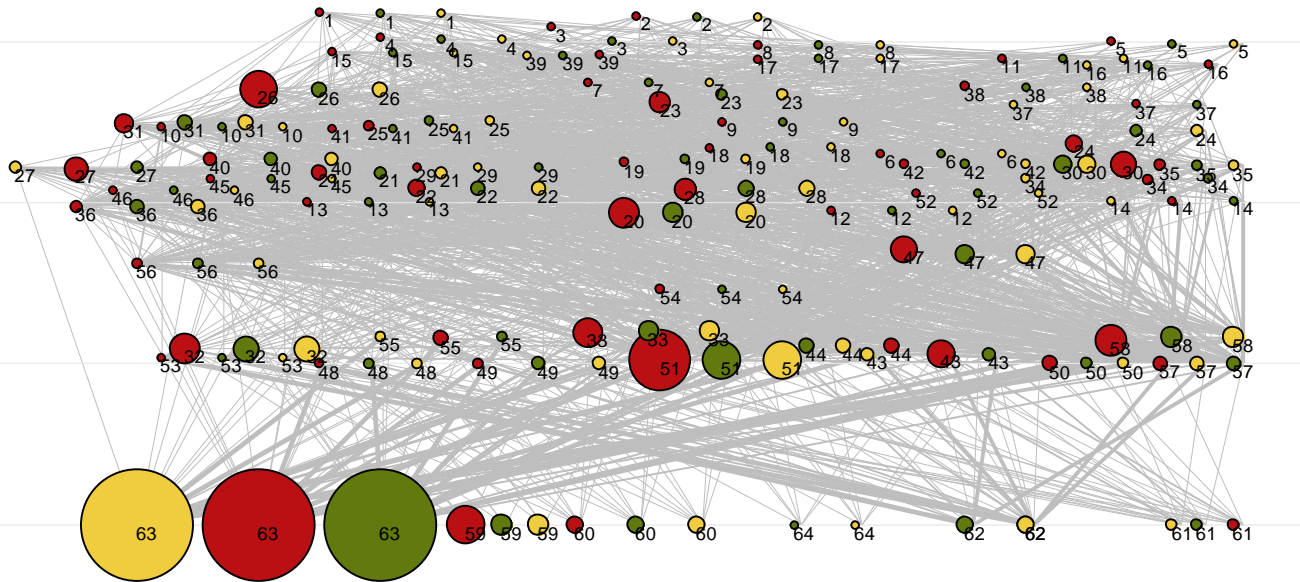
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650 **Figure 2.** Pedigree index values of three MUs models (FPA: fully protected area; PPA:  
 651 partially protected area; UPA: unprotected area) for each MPA (Cerbère-Banyuls, Cap de  
 652 Creus and Medes Islands).



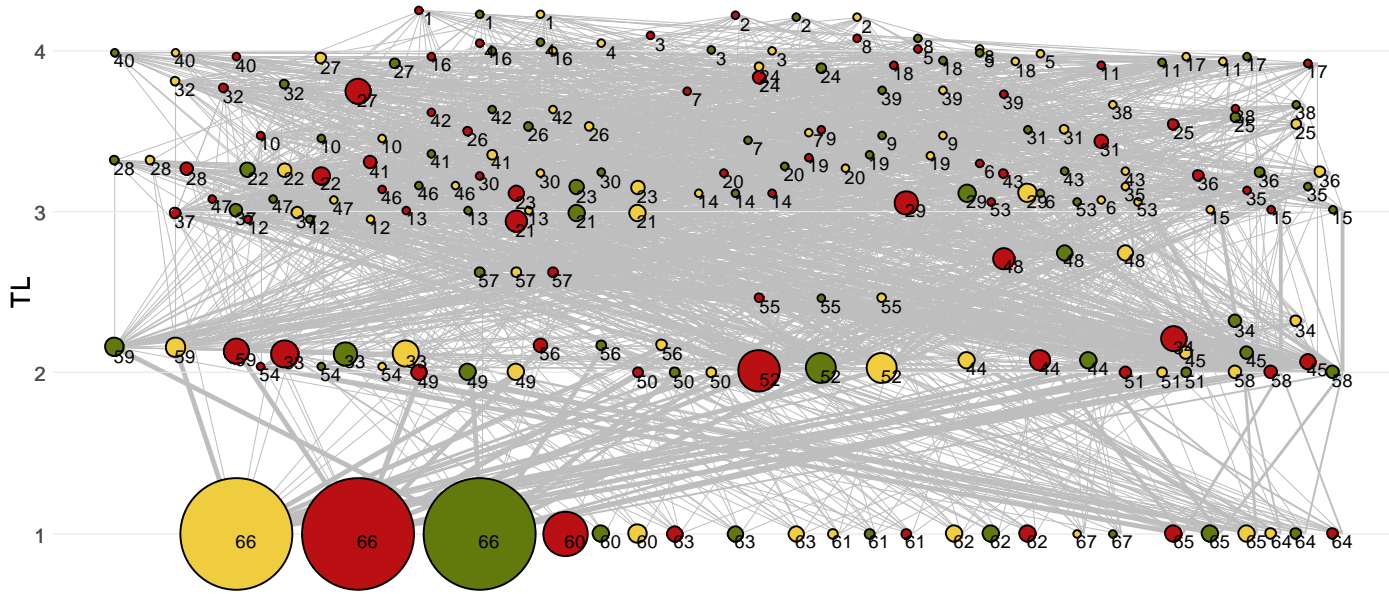
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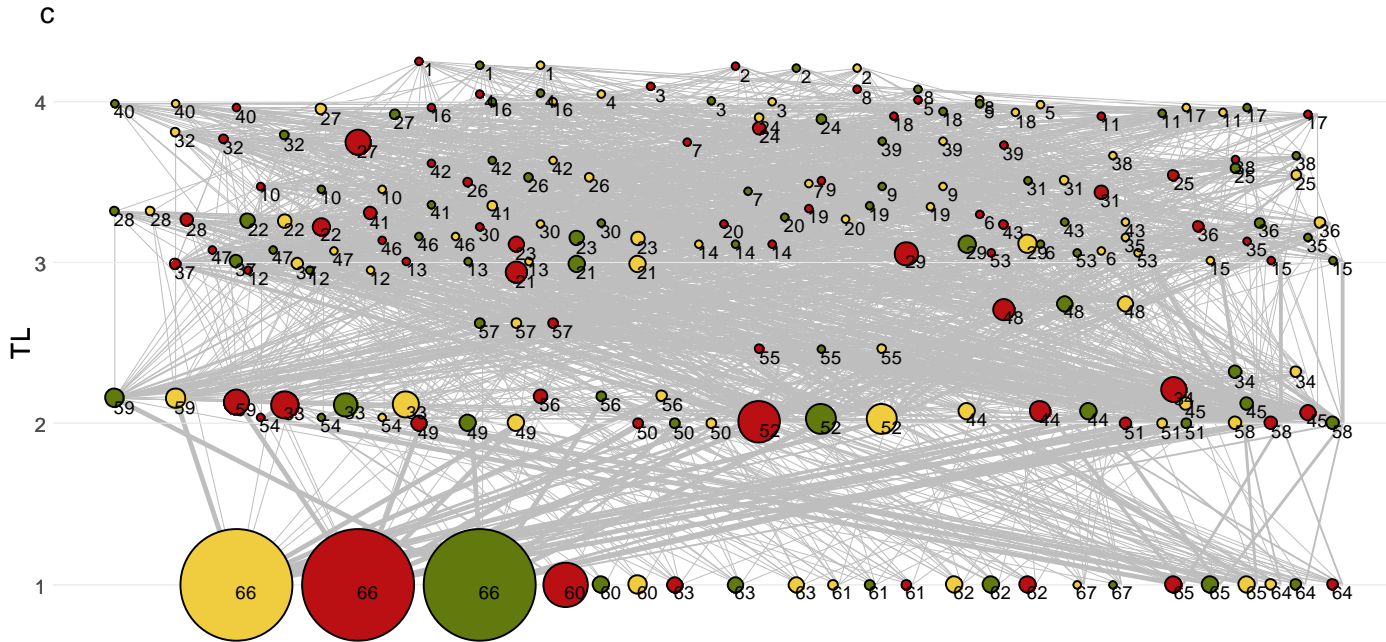
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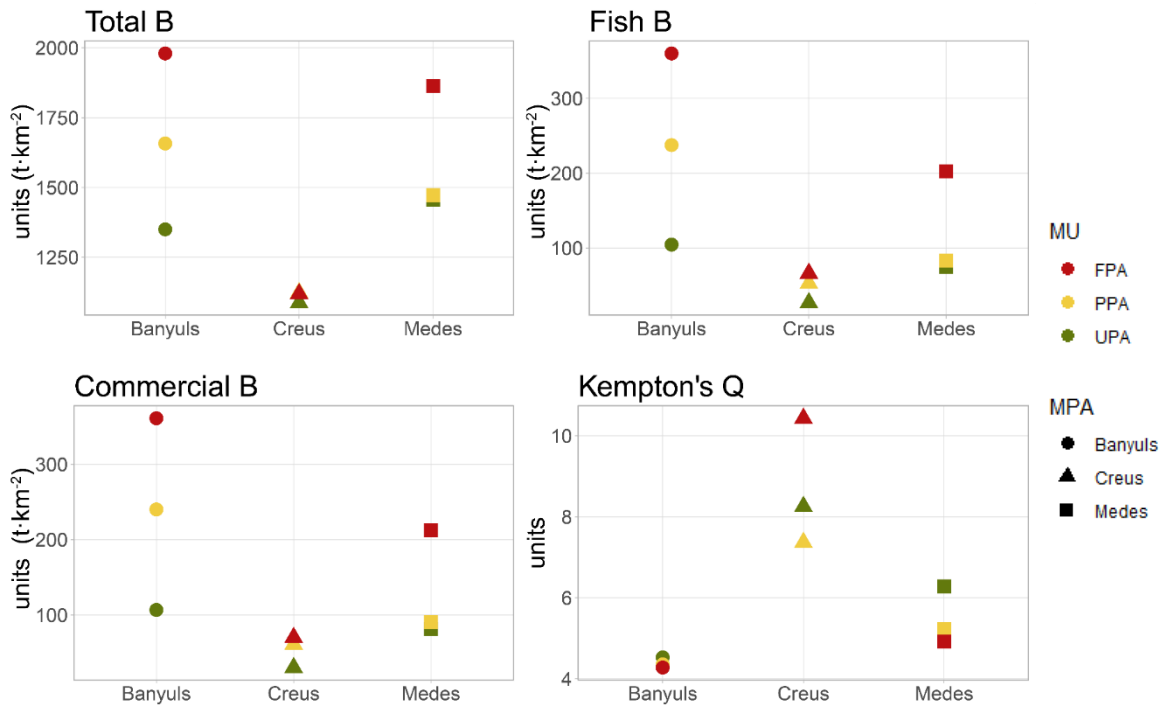
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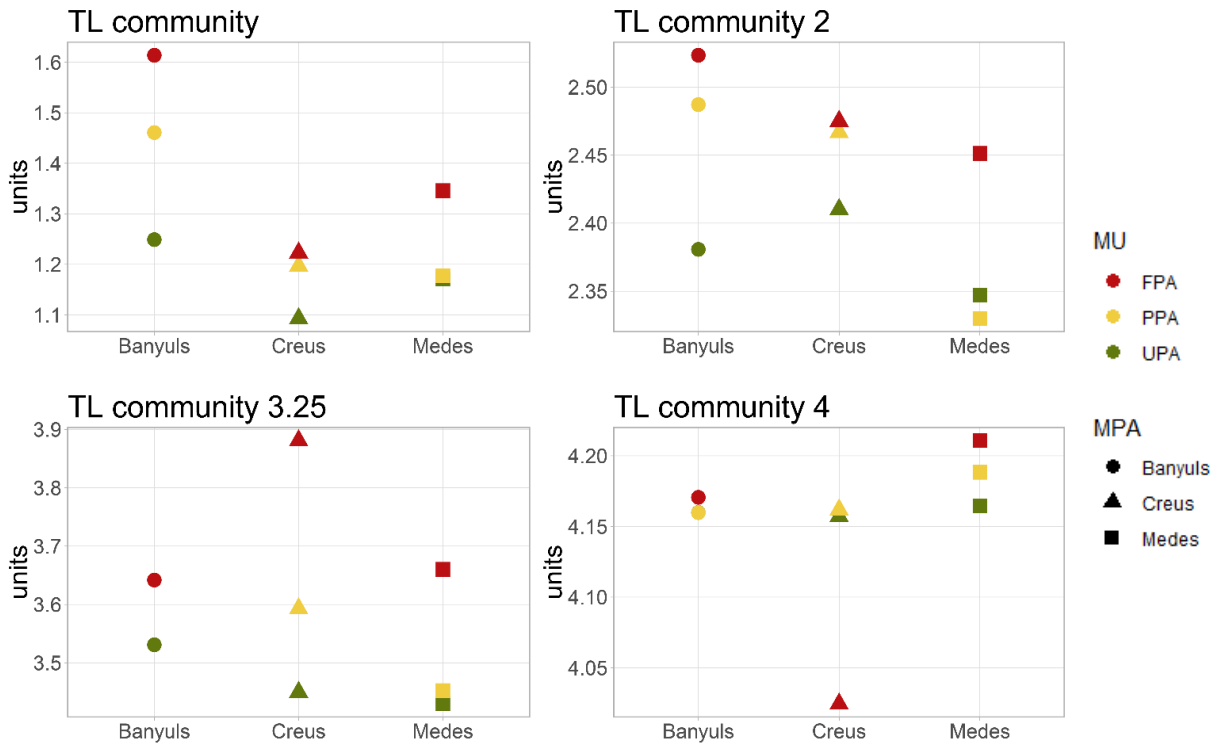
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**Figure 3.** Flow diagram of fully protected areas (FPAs) of (a) Cerbère-Banyuls MPA, (b) Cap de Creus and (c) Medes Islands MPA models organized by Trophic Levels (y-axis). The size of each circle is proportional to the biomass of the functional group. The wideness of the connecting lines is proportional to the magnitude of their flows.



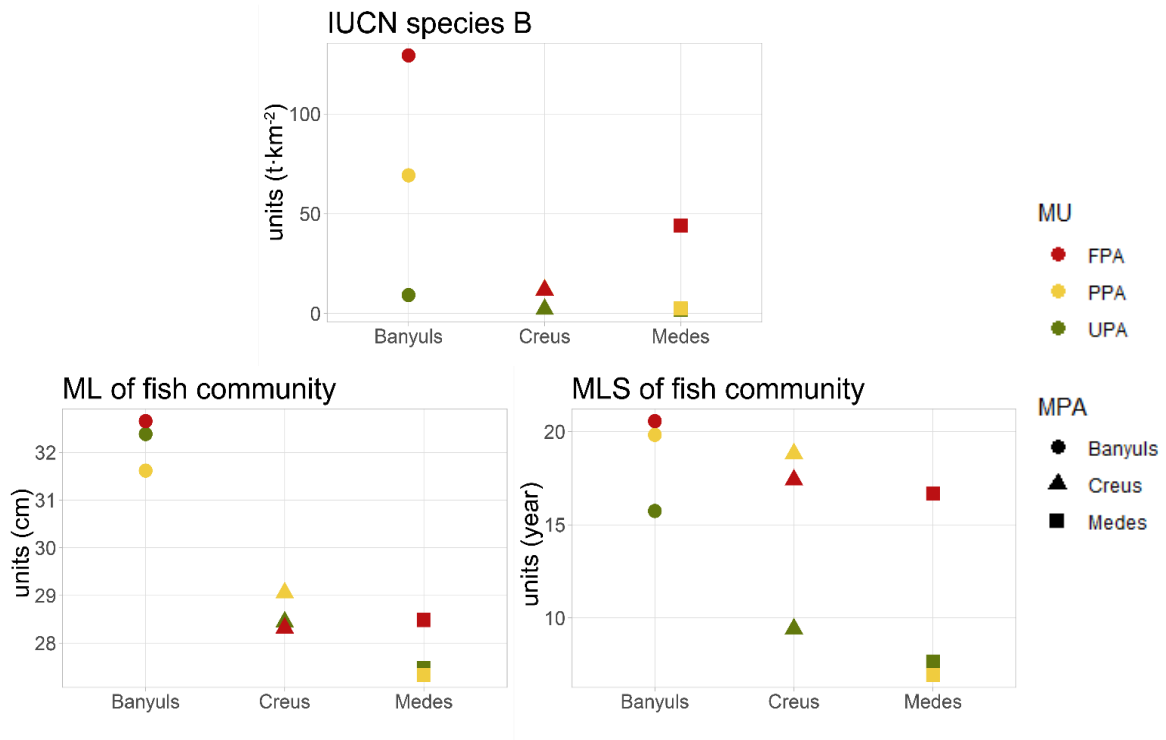
**Figure 4.** Biomass-based indicators of three MUs (FPA: fully protected area; PPA: partially protected area; UPA: unprotected area) models for each MPA (Cerbère-Banyuls, Cap de Creus and Medes Islands).



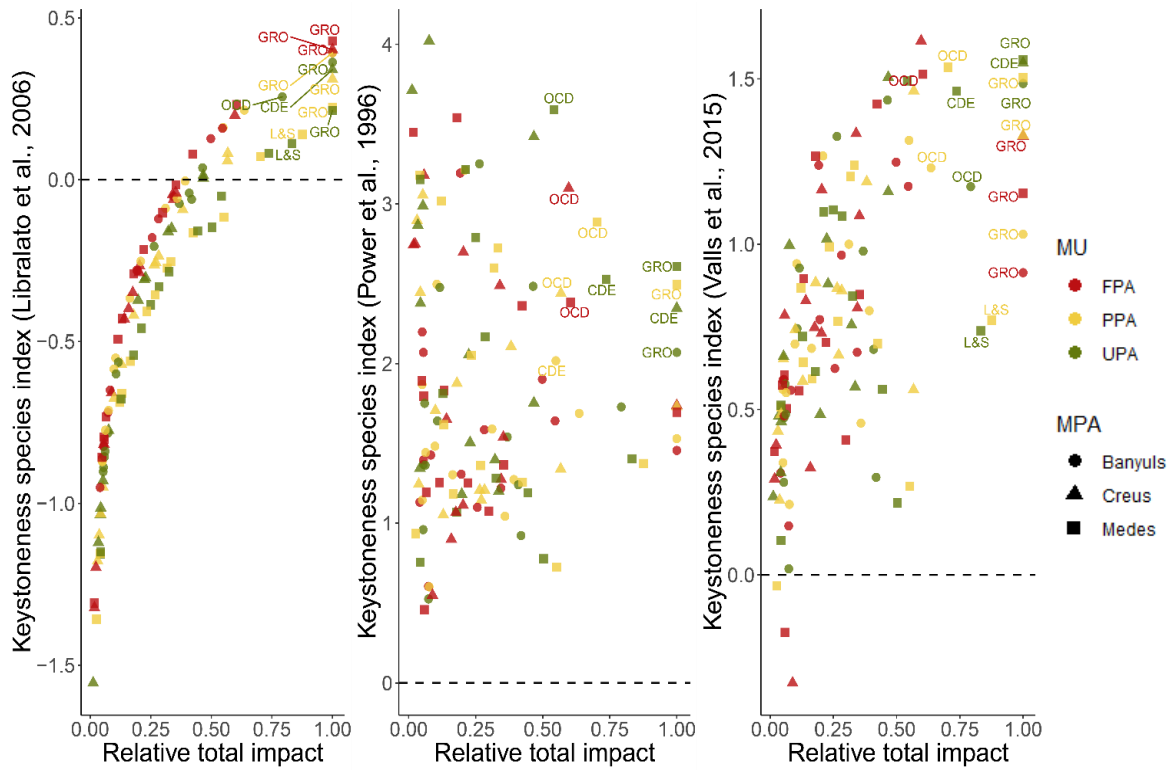
**Figure 5.** Trophic-based indicators of three MUs (FPA: fully protected area; PPA: partially protected area; UPA: unprotected area) models for each MPA (Cerbère-Banyuls, Cap de Creus and Medes Islands).



**Figure 6.** Flow-based indicators of three MUs (FPA: fully protected area; PPA: partially protected area; UPA: unprotected area) models for each MPA (Cerbère-Banyuls, Cap de Creus and Medes Islands).

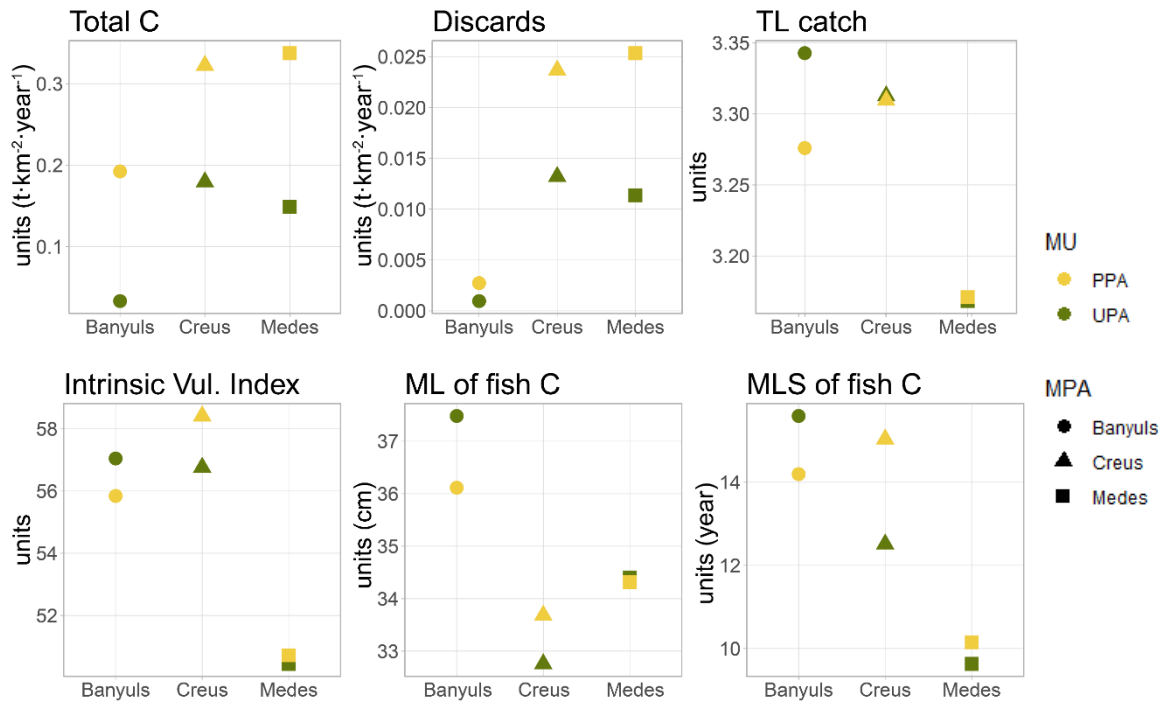


**Figure 7.** Species-based indicators of three MU models (FPA: fully protected area; PPA: partially protected area; UPA: unprotected area) for each MPA (Cerbère-Banyuls, Cap de Creus and Medes Islands).

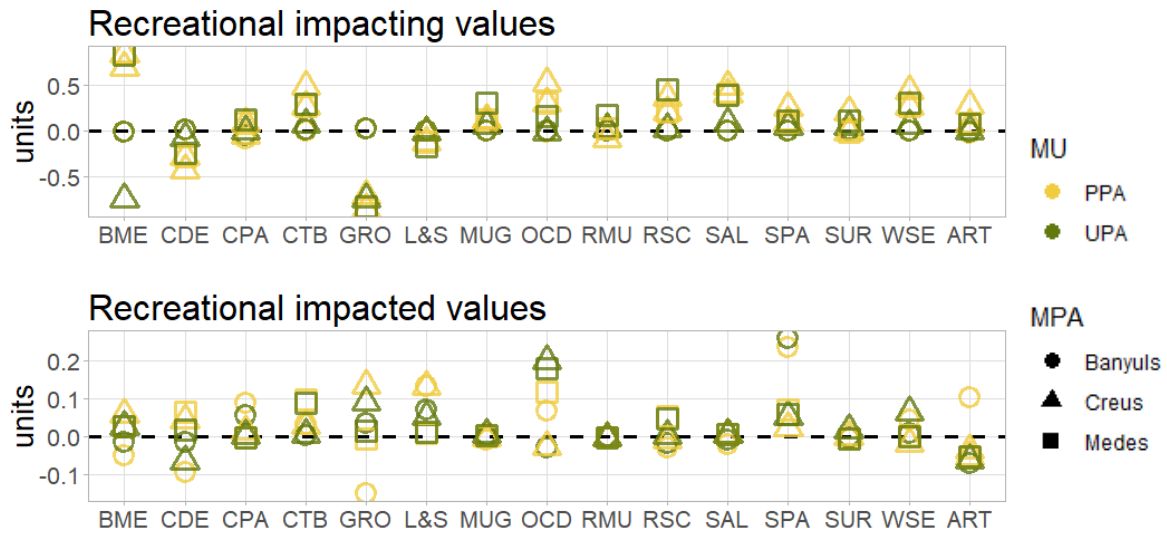


**Figure 8.** Keystone Index analysis of three MU (FPA: fully protected area; PPA: partially protected area; UPA: unprotected area) models for each MPA (Cerbère-Banyuls, Cap de Creus and Medes Islands). The acronyms identify the functional group with highest keystone index and relative total impact. (GRO – Groupers; CDE – Common dentex; OCD – Other commercial medium demersal fishes; L&S – Labridae and Serranidae).

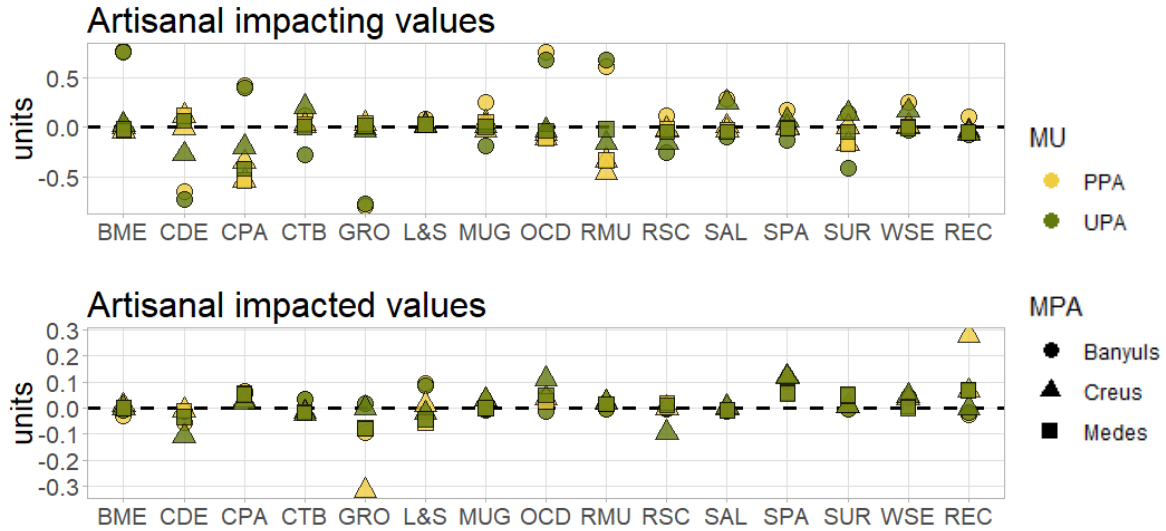




**Figure 9.** Catch-based indicators of three MUs (PPA: partially protected area; UPA: unprotected area) models for each MPA (Cerbère-Banyuls, Cap de Creus and Medes Islands).



**Figure 10.** Recreational impacting and impacted values of three MUs (PPA: partially protected area; UPA: unprotected area) models for each MPA (Cerbère-Banyuls, Cap de Creus and Medes Islands). (BME – Brown meagre; CDE – Common dentex; CPA – Common pandora; CTB – Common two-banded seabream; GRO – Groupers; L&S – Labridae and Serranidae; MUG – Mugilidae; OCD – Other commercial medium demersal fishes; RMU – Red mullet; RSC – Red scorpionfish; SAL – Salema; SPA – Sparidae; SUR – Surmulet; WSE – White seabream; ART – Artisanal fishery).



**Figure 11.** Artisanal impacting and impacted values of three MUs (PPA: partially protected area; UPA: unprotected area) models for each MPA (Cerbère-Banyuls, Cap de Creus and Medes Islands). (BME – Brown meagre; CDE – Common dentex; CPA – Common pandora; CTB – Common two-banded seabream; GRO – Groupers; L&S – Labridae and Serranidae; MUG – Mugilidae; OCD – Other commercial medium demersal fishes; RMU – Red mullet; RSC – Red scorpionfish; SAL – Salema; SPA – Sparidae; SUR – Surmulet; WSE – White seabream; REC – Recreational fishery).

## Tables

Table 1. Surface area (km<sup>2</sup>) covered by management unit (MU) and year of creation of each marine protected area (MPA) and their year (Cerbère-Banyuls, Cap de Creus and Medes Islands).

<b>MPA</b>	<b>Year of creation</b>	<b>MU</b>		
		<b>FPA</b>	<b>PPA</b>	<b>UPA</b>
Cerbère-Banyuls	1974	0.65	5.85	35.00
Cap de Creus	1998	0.21	7.98	22.37
Medes Islands	1983	0.39	4.24	14.85

Table 2. Functional groups included for the MU models showing those present (P) or absent (A) in each MPA. FG = Functional Group. Discards only are included in PPA and UPA models.

FG number	FG name	Cerbère-Banyuls	Cap de Creus	Medes Islands
1	Bottlenose dolphins	P	P	P
2	Striped dolphins	P	P	P
3	Endangered and pelagic seabirds	P	P	P
4	Gulls and cormorants	P	P	P
5	Terns	P	P	P
6	Loggerhead turtles	P	P	P
7	Non-commercial large pelagic fishes	P	P	P
8	Other large pelagic fishes	P	P	P
9	Mackerels	P	P	P
10	Horse mackerels	P	P	P
11	Other medium pelagic fishes	P	P	P
12	European sardine	P	P	P
13	European anchovy	P	P	P
14	Round sardinella	A	P	P
15	Other small pelagic fish	P	P	P
16	Anglerfish	P	P	P
17	European conger	P	P	P
18	European hake	P	P	P
19	Poor cod	P	P	P
20	Common pandora	P	P	P
21	Sparidae	P	P	P
22	White seabream	P	P	P
23	Common two-banded seabream	P	P	P
24	Common dentex	P	P	P
25	Red scorpionfish	P	P	P
26	Scorpaenidae	P	P	P
27	Groupers	P	P	P
28	Brown meagre	P	P	P
29	Labridae and Serranidae	P	P	P
30	Flatfishes	P	P	P
31	Other commercial medium demersal fish	P	P	P
32	No commercial medium demersal fish	P	P	P
33	Salema	P	P	P
34	Mugilidae	P	P	P
35	Red mullet	P	P	P
36	Surmulet	P	P	P
37	No commercial small demersal fish	P	P	P
38	Small-spotted catshark	P	P	P
39	Rays and skates	P	P	P
40	Torpedos	P	P	P
41	Coastal benthic cephalopods	P	P	P
42	Benthopelagic cephalopods	P	P	P
43	Other benthic cephalopods	P	P	P
44	Bivalves	P	P	P
45	Gastropods	P	P	P
46	European lobster	P	P	P
47	Other commercial decapods	P	P	P
48	Non-commercial decapods	P	P	P
49	Purple sea urchin	P	P	P
50	Other sea urchin	P	P	P
51	Sea cucumbers	P	P	P
52	Other macro-benthos	P	P	P
53	Jellyfish	P	P	P
54	Salps and other gelatinous zooplankton	P	P	P
55	Red coral	P	P	P

56	Other corals and gorgonians	P	P	P
57	Macrozooplankton	P	P	P
58	Meso and microzooplankton	P	P	P
59	Suprabenthos	P	P	P
60	Mediterranean seagrass	P	P	P
61	Other seagrasses	A	P	P
62	Erected algae	A	P	P
63	Seaweeds	P	P	P
64	Small phytoplankton	P	P	P
65	Large phytoplankton	P	P	P
66	Detritus	P	P	P
67	Discards	P	P	P