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## The need for a protean fisheries science to address the degradation of exploited aquatic ecosystems

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27 ecosystem, include society, its economy and the services it derives from aquatic systems. Such a  
28 protean science is required to address the complexity of ecosystem functioning and of the impacts  
29 of anthropogenic pressures.

30

31 Keywords: EAFM; fisheries management; fisheries science; marine resources; models; uncertainty

32

### 33 1 Introduction

34 Fish provide 16.7 % of animal protein for the global population and more than 20 % of animal  
35 protein for 2.9 billion people (FAO 2014). The degradations of both structure and functioning of  
36 aquatic ecosystems due to overfishing, contamination, habitat fragmentation and degradation or  
37 water extraction are now widely recognised and documented (Postel and Richter 2003; Lotze et al.  
38 2006; Worm et al. 2006; Chevillot et al. 2016; in this symposium: Brind'Amour et al.; Chevillot et  
39 al.; Kaimuddin et al.; Le Luherne et al.; Olmos et al.; Vogel et al.). Among the drivers of global  
40 change, climate change is a main threat for those ecosystems affecting all biological scales (Gattuso  
41 et al. 2015): primary productivity, species distributions, and community and foodweb structure  
42 (Cheung et al. 2010). Modifications in primary production (Barange et al. 2010) and acidification  
43 due to carbon sequestration (Orr et al. 2005; Doney et al. 2009; Feely et al. 2009) potentially have  
44 drastic consequences - although these remain largely unpredictable. These impacts on aquatic  
45 ecosystems and on the services they provide require to be investigated, and possibly minimized  
46 through appropriate management (Millennium Ecosystem Assessment 2005). In this context, the  
47 12<sup>th</sup> symposium organised by the Association Française d'Halieutique (AFH, French Association for  
48 Fisheries Sciences) in 2015 illustrated that assessing the states of aquatic ecosystems and the effects  
49 of anthropogenic pressures acting at different spatial, temporal and biological scales, requires a  
50 protean science that considers simultaneously all scales, levels of complexity and dimension. This  
51 foreword does not aim to be an exhaustive analysis of upcoming challenges for fisheries sciences,  
52 but rather aims to highlight key questions that were discussed during the symposium that took place

53 in Montpellier, France (July 1-3, 2015) on “Status, functioning and shifts in marine ecosystems”.  
54 The oral presentations of this symposium are available at [http://www.association-francaise-  
halieutique.fr/conferences](http://www.association-francaise-halieu-<br/>55 halieutique.fr/conferences). Four contributions are published in the present issue.

56

57 2 A science addressing all organisational scales

58 Scales and their interactions are an important topic in ecology (Levin 1992; Chave 2013). The  
59 symposium illustrated current developments on three complementary scales.

60

61 • *On the road to the ecosystem approach for fisheries?*

62 The ecosystem approach to fisheries management (EAFM) is a paradigm born in the early 2000s  
63 following, among others, the FAO code of conduct for responsible fisheries in 1995 and the  
64 Reykjavik conference in 2002 (Garcia 2003). Implementing truly EAFM was precisely at the core  
65 of the manifesto “A future for marine fisheries in Europe” published by the AFH in 2011 (Gascuel  
66 et al. 2011). EAFM calls for a change from traditional single species fisheries management to  
67 management that considers all implications and dimensions of fishery socio-ecosystems:  
68 multispecies catches, bycatches, trophic implications, habitat degradation, socio-economic issues,  
69 governance, etc... More than ten years after its creation, EAFM is progressively being  
70 implemented. Through the modification of its working groups, the International Council for the  
71 Exploration of Sea (ICES) is for example currently trying to regionalise its advice to provide  
72 regional ecosystem advice. Recently, a working group established by the Scientific Technical and  
73 Economic Committee for Fisheries (STECF) of the European Commission to work on the  
74 development of the EAFM in European seas, proposed methods to achieve integrated fleet-based  
75 management (Gascuel et al. 2012). Ecosystem impacts are also one of the main criteria considered  
76 in fisheries’ ecolabels which have appeared since the 2000s. In this context, the discard ban  
77 implemented by the European Commission can be considered a new step within the EAFM.

78 Several tools have been developed to support EAFM, from indicators to models. The indicators  
79 developed in the context of the Water Framework Directive (Hering et al. 2010) and the upcoming  
80 indicators from the Marine Strategy Framework Directive will operationalise long-term monitoring  
81 of marine aquatic ecosystem status, which is a cornerstone of biodiversity preservation and  
82 restoration. The next challenge is to move beyond monitoring structural diversity via the application  
83 of indicators of abundance and diversity by developing and applying indicators of functional  
84 diversity. In this paper, we call this functional diversity “ecodiversity” to avoid any confusion with  
85 biodiversity and structural diversity. Such indicators need to assess the ecological functions of  
86 ecosystems to inform on the quality of their functioning. We, the fisheries scientists attending the  
87 symposium, have direct responsibilities for characterizing the ecological status of several  
88 ecosystems and some of these case studies were presented during the 2015 symposium (Lucena-  
89 Frédou et al. this symposium; Lobry et al. this symposium; Wessel et al. this symposium; Laë et al.  
90 this symposium).

91 Trophic models such as Ecopath, Ecosim (Pauly et al. 2000) and Ecotroph (Gascuel and Pauly  
92 2009) are now being applied to assess ecosystems worldwide to inform fisheries management (a list  
93 of applications can be found at EcoBase <http://sirs.agrocampus-ouest.fr/EcoBase/>). Methods have  
94 been developed to evaluate the sensitivity of model results to uncertainty in parameters (Guesnet et  
95 al. 2015; Lobry et al. this symposium) and model structure (Prato et al. 2014, this symposium). In  
96 describing trophic cascades, these models have proven useful to inform management on the direct  
97 and indirect effects of fisheries and management throughout the food web. They have raised major  
98 strategic questions concerning fisheries, such as the recent debate concerning the so called balanced  
99 harvesting approach (Garcia et al. 2015, 2012; Froese et al. in Press). During the symposium related  
100 questions were discussed in a dedicated debate on ecosystem management (Dagorn et al. this  
101 symposium). In the last decade, end-to-end models (Fulton 2010; Rose et al. 2010) such as Atlantis  
102 (Fulton et al. 2004) or Osmose (Shin and Cury 2004), have provided a more detailed description of  
103 ecosystem functioning by coupling physicochemical oceanographic descriptors with population

104 dynamics for organisms ranging from microbes to higher-trophic-levels. Such integration of all  
105 ecosystem components and their interactions offers new opportunities to explore climate change  
106 effects or those of other anthropogenic stressors on ecosystems. These models have proven useful  
107 for informing management regarding major strategic fisheries management questions (Lehuta et al.  
108 this volume), such as when to fish, where to fish or which trophic levels to harvest. During the  
109 symposium several presentations highlighted the ability of trophic models to provide diagnostics of  
110 fishing impacts at the ecosystem scale for Mauritanian waters (Meissa et al. this symposium),  
111 Tunisian waters (Abdou et al. this issue) the Bay of Biscay and Celtic Sea (Moullec et al. this  
112 symposium), the Mediterranean Sea (Halouani et al. this symposium), and at a global scale  
113 (Colleter et al. this symposium). Other integrated models like ISIS-Fish (Lehuta et al. this  
114 symposium) or Atlantis (Girardin et al. this symposium) were shown in turn to be useful for  
115 analysing the impacts of management decisions on the English Channel and the Bay of Biscay.  
116 However, most available tools focus on the ecological part of socio-ecosystems, and there is a lack  
117 of tools for addressing the socio-economic context in which fisheries operate (Gascuel et al. 2012).  
118 The ecosystem services concept formulated during the Millenium Ecosystem Assessment  
119 (Millennium Ecosystem Assessment 2005) can be a tool to fill this gap (Vanhoutte-Brunier et al.  
120 this symposium). However operational tools are still scarce and should be considered carefully  
121 specifically for dealing with moral issues regarding the value of nature (Maris 2014 this  
122 symposium). Life Cycle Assessment could also prove useful in the future to assess the sustainability  
123 of sea food production in a more integrated way (Avadí and Fréon 2013; Fréon et al. this  
124 symposium), as could viability models which, instead of predicting precise system-trajectories, only  
125 predict bounds in which the system is expected to remain sustainable (Planque et al. 2014; this  
126 symposium).

127

- 128 • *Population/species scale: still advancing*

129 The implementation of the EAFM does not mean that the traditional population scale is not

130 considered anymore, first of all because most of stocks are still assessed using mono-specific  
131 models (e.g., Folliot et al. this symposium; Marandel et al. this issue). A better understanding of  
132 species' ecology and especially of species' diets is also crucial to achieve a better understanding of  
133 ecosystem functioning. For example, Tableau et al. (this symposium) assessed the influence of the  
134 availability of trophic resources on nursery grounds on juvenile abundance of certain species. Van  
135 Beveren et al. (this symposium) focused on the control exerted by Bluefin tuna on pelagic species  
136 through predator-prey relationships. Moreover, population-based approaches are useful for  
137 understanding climate change effects on population dynamics and the adaptability of species.  
138 The coupling of population-based approaches and hydrodynamic oceanic models has proven useful  
139 for assessing the role of oceanic conditions on population dynamics and subsequently for exploring  
140 the consequences of climate change scenarios. Bertrand et al. (2011; this symposium) for example  
141 demonstrated the influence of oxygen concentrations on the distribution and abundance of various  
142 pelagic species on the Peruvian coast. At an individual level, the dynamic energy budget (DEB,  
143 Kooijman 2000) model has helped discriminate between the reproductive strategies of sardine and  
144 anchovy (Gatti et al. this symposium). Politikos et al. (2015) coupled a DEB budget model with a  
145 hydrodynamic model to infer the influence of oceanographic conditions on the reproduction of  
146 anchovy (*Engraulis encrasicolus*). Energy allocation has also been tracked through field  
147 measurements of individuals. For example, Brosset et al. (this symposium) illustrated how in situ  
148 measurements of relative condition factors of 43000 individuals can be correlated to environmental  
149 conditions to inform on the energy allocation strategy of anchovy.

150 Studying species adaptations to climate change is an active field of research, yet genetic approaches  
151 remain marginal at present in fisheries science. Levels of population differentiation, as measured by  
152 the fixation index ( $F_{st}$ ) and effective population size estimates are challenging our traditional  
153 demographic views. Bonhomme (this symposium) put forward some of these concepts and  
154 evidenced the existence of two populations of anchovy, a coastal and an offshore one, linked by  
155 gene fluxes. Demo-genetic models that combine mechanistic population dynamics and evolutionary

156 processes have been developed to explore the ability of species to adapt to climate change (e.g. Piou  
157 and Prévost 2012; Mateo et al. this symposium). Similarly, models coupling dispersion modules  
158 with climatic scenarios have been developed to predict the ability of species to adapt their  
159 distribution areas in a context of climate change (Rougier et al. 2014). Statistical approaches have  
160 also been used to explore phenological adaptations to climate change (Chevillot et al. this  
161 symposium).

162

163 • *The revolution of individual tracking methods*

164 At the lowest organisational level, individual tracking methods and the analysis of individual  
165 behaviours are making remarkable progress. The technological developments of tracking methods  
166 during the 1990s have opened new opportunities for behavioural ecology (Cagnacci et al. 2010;  
167 Jonsen et al. 2003). Meanwhile, the new paradigms in movement ecology proposed by Nathan et al.  
168 (2008) combined with the development of appropriate analytical methods such as state-space  
169 models (Jonsen et al. 2013; Joo et al. 2013) or segmentation tools (Nams 2014), have revolutionized  
170 the study of individual movements. Movement analysis can address several types of questions  
171 (Nathan et al. 2008): why organisms move, how they move, where and when they move, how the  
172 environment influences these movements, and how these components interact. Movement analyses  
173 can provide important insights into the life cycle of populations (de Pontual et al. 2015, this  
174 symposium; and during this symposium Leopold et al. for reef fish in Vanuatu or Woillez et al. for  
175 the European sea bass ), inform on key aspects of species interactions (see Bertrand et al. this  
176 symposium, on the tracking of seabirds and marine mammals, complemented with field  
177 observations in Passuni et al. this symposium) and management (Barton et al. 2015). Individual  
178 tracking methods have not only been applied to animals, but also to humans and fishing vessels.  
179 The development of Vessel Monitoring Systems has enabled fine tracking of fishing vessels  
180 resulting in a better understanding of fishers' fishing behaviour, quantification of fishing effort (Bez  
181 et al. 2011; Gloaguen et al. 2015; Vermard et al. 2010; Maufroy et al. this symposium). It can also



182 provide valuable information on the spatial distribution of target species (Walker 2010, this  
183 symposium; Gloaguen et al. this symposium). Individual vessel tracking through the analysis of  
184 onboard observations or log-books can inform on interactions between fishermen and exploited  
185 resources (Bourdaud et al. this symposium; Escalle et al. this symposium; Conte et al. this  
186 symposium; Robert et al. this symposium).

187  
188 2.2 Different levels of complexity to address multifaceted anthropogenic pressures

189 Another important ecological question debated during this symposium and related to some extent to  
190 the question of scale discussed earlier, is the issue of complexity and more specifically of model  
191 complexity. Simple models are generally considered robust and more generic. On the other hand,  
192 complex models are considered more detailed and realistic, but more sensitive and case-specific.  
193 While simple models may provide biased estimates, more complex models tend to increase  
194 estimation uncertainty (Costanza and Sklar 1985; Håkanson 1995; Burnham and Anderson 2002).  
195 The development of complex models used to be constrained by computational power. For a long  
196 time, modellers have sought the simplest model to address their question, honouring the so-called  
197 parsimony principle (Box and Jenkins 1970). Various criteria have been proposed to seek the best  
198 model, i.e. the best trade-off between detail, simplicity, data availability and ease of communication  
199 (see for example the discussion in Cotter et al. (2004) about complexity and stock assessments  
200 models and Lehuta et al, this issue).

201 The strategy of looking only for a single “best model” is now changing. In view of the uncertainty  
202 related to model selection, the development of multi-model inference methods in the early 2000s  
203 has enabled scientists to combine results from different models with different levels of complexity  
204 (Hoeting et al. 1999; Burnham and Anderson 2002). The already mentioned end-to-end models  
205 which are based on coupling several simple models describing elementary processes, or integrated  
206 models which couple different models describing various components of a system, are alternative  
207 ways to address questions. In a recent paper Evans et al. (2013) argued that simple and complex

208 models can be linked together to produce broad-scale and predictive understanding of biological  
209 systems. Combining models that have different levels of complexity appears to be an alternative  
210 strategy to benefit from the advantages of both types of models. Working with a large range of  
211 models from the simplest to the most complex is not an utopia in practice. This is because modern  
212 computational power coupled with constantly increasing data quantities enable parameterisation and  
213 exploration (Iooss this symposium) of even highly complex models. However, as debated during a  
214 dedicated session of the symposium, the next challenge is to develop frameworks to guide the  
215 combination of approaches, and how to use this combination to inform management, especially  
216 when dealing with situations in which different models provide contradictory results.

217

### 218 2.3 How should scientists address uncertainty?

219 The debate about model complexity raises the question about uncertainty which was debated during  
220 a specific session during the symposium (Bez et al. this symposium). Uncertainty arises from  
221 several sources: uncertainty in data, uncertainty in knowledge but also unknown uncertainties  
222 (Chow and Sarin 2002; O’Hagan 2004). There is a long history of dealing with these uncertainties.  
223 For example, statistical confidence intervals and sensitivity analyses (Kleijnen 1987; Faivre et al.  
224 2013) aim to quantify the impacts of uncertainties on outputs while multi-inference modelling and  
225 management strategy evaluation (Smith et al. 1999; Bunnefeld et al. 2011) aim to enhance the  
226 robustness of outputs to various sources of uncertainty. These approaches all address known sources  
227 of uncertainty but can hardly deal with “unknown unknowns”. Interestingly, rather than dealing  
228 with uncertainty, other modelling strategies have proposed focusing on the few certainties (Mullon  
229 et al. 2009; Planque et al. 2014, this symposium), eliminating impossible trajectories whilst  
230 considering all other trajectories, including “known unknowns” (sources of uncertainty we are  
231 aware of) and “unknown unknowns” (sources of uncertainty we are not even aware of).  
232 These methodological developments have greatly enhanced scientific management  
233 recommendations and form an integral part of scientific work, which must be objective and

234 transparent. Probabilistic frameworks are now widely used by scientists and translated by managers  
235 when fixing objectives. However, uncertainty cannot be restricted to a statistical or mathematical  
236 issue but raises questions about how these uncertainties should be communicated, how they are  
237 understood and how they are used by stakeholders and managers. A session was dedicated to this  
238 question during the symposium. While the “precautionary approach” states that actions should not  
239 be delayed because of uncertainties, uncertainties can be used by lobbies to cast doubt and delay  
240 actions, which is called “agnotology” (Proctor and Schiebinger 2008; Brandt 2009; Latour 2012;).  
241 In the Bluefin tuna example (Fromentin et al. 2014, this symposium), while uncertainties had been  
242 used by stakeholders to delay conservation measures in the Mediterranean Sea, they were also used  
243 by NGOs to promote those measures.

244 Moreover, scientific uncertainty is not synonymous to management uncertainty. First, management  
245 uncertainty relates to decisions not only on domain-specific probabilistic questions. Management  
246 decisions have important socio-economic consequences, sometimes much more uncertain than the  
247 ecological uncertainty (Hall 2002; Faulkner et al. 2007; Nollet and De Gelder 2000). Second, the  
248 uncertainty measures provided by scientists are often related to uncertainty around a mean state  
249 while managers are generally more interested in risks related to the occurrence of extreme events  
250 which are not always well addressed by scientists. As an example, managers can be more interested  
251 in the extinction probability of a species (binary consideration) while scientific models will often  
252 estimate the probability that the expected (i.e. mean) abundance stays at a given level and these  
253 models are not necessarily appropriate for extreme situations. This discrepancy in the type of  
254 uncertainty considered is even more dramatic given the scientific probabilistic framework is not  
255 “measurable” in the real world. In the real world, an event occurs or does not occur, and there is no  
256 way to validate a scientific statement saying that the event will occur with x % probability.

257 In this context, the question is not to ponder on whether scientists should hide uncertainties, but  
258 rather consider how they should communicate them, especially when providing advice to managers.  
259

## 260 2.4 A multidimensional science

261 The use and understanding of uncertainty by managers and stakeholders is not only an ecological  
262 issue, but also a political and sociological one. This is one example amongst others that  
263 demonstrates the potential insights from social sciences for achieving sustainable management, i.e.  
264 a management that preserve the sustainability of the resource and of all the associated goods and  
265 services. As mentioned above socio-economical aspects are still poorly considered in the  
266 implementation of the EAFM. In a recent review of the restoration of aquatic ecosystems, Wortley  
267 (2013) found that the socio-economic context had been considered in a very limited number of  
268 restoration programs. Further, mixed results of those restoration programs can often be explained by  
269 socio-political or economical barriers (Pahl-Wostl et al. 2013). On the other hand, several examples  
270 illustrate that the involvement of stakeholders can lead to unexpected effective results. The example  
271 of the transformation of the Scottish fisheries is very instructive: the unexpected mobilization of  
272 stakeholders lead to a deep transformation of the whole industry towards more sustainability (Carter  
273 2014, this symposium), an objective that the European Common Fisheries Policy had failed to  
274 achieve. During this symposium, the relevance of multidimensional monitoring was illustrated by  
275 the example of the Iroise Marine Natural Park. For this park a multidimensional and co-constructed  
276 scoreboard (Gamp et al. this symposium), an ecosystem services assessments (Vanhoutte-Brunier et  
277 al. this symposium), and consideration of anthropological aspects (Mariat-Roy et al. this  
278 symposium) were presented. Tissi re et al. (this symposium) illustrated the potential of a  
279 geopropective to co-construct a shared view of the future by several stakeholders.

280 Bio-economic models have been developed for a long time. However they are not yet used routinely  
281 to inform management. The methods proposed by Gascuel et al. (2012) can be a first step towards  
282 more integrated assessment and management of fisheries, taking into account socio-economic and  
283 ecological quantitative indicators of fisheries. However, currently no tools or frameworks exist  
284 which could incorporate qualitative anthropological, social or political criteria. Moreover, the  
285 context in which the exploitation of aquatic ecosystems takes place is not static and evolves rapidly,

286 especially with global change. This moving context is a source of barriers or opportunities that are  
287 hardly predictable, what appears impossible today may be possible tomorrow. The uncertainty  
288 around Bluefin tuna status had been a barrier to conservation measures before being an incentive. It  
289 is therefore clear that integrating constructivist approaches can help both science and management  
290 to understand the objectives and strategies of different stakeholders, to highlight barriers or  
291 opportunities and to redraw science/politics boundaries (Carter 2013).

292

### 293 3 Conclusion

294 The title of this paper calls for a protean science to address the degradation of aquatic ecosystems.  
295 By protean science we mean a multifaceted multidimensional science that works at all scales and  
296 all levels of complexity. A science that is aware of uncertainty, remains organised and integrates this  
297 complexity and which is able to adapt quickly to new challenges. Such a protean science is required  
298 to address the complexity of the functioning of socio-ecosystems which still conceals many  
299 mysteries (see for example Husson et al., this symposium) and of anthropogenic pressures that are  
300 also multiple, multidimensional and constantly changing. The upcoming challenge is most likely to  
301 build new frameworks that take the greatest benefit of this protean science to achieve effective  
302 sustainability of exploited aquatic ecosystems.

303

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312

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