UNIVERSITAT DE BARCELONA

Benthic communities' response to different trawling impact levels: generalization towards developing a Mediterranean model

Resposta de les comunitats bentòniques a diferents nivells d'impacte de pesca d'arrossegament: generalització per al desenvolupament d'un model mediterrani

Alba Muntadas Olivé

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Alba Muntadas Olivé

Tesi doctoral

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Memòria presentada per Alba Muntadas Olivé per optar al grau de doctora per la Universitat de Barcelona

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Benthic communities' response to different trawling impact levels: generalization towards developing a Mediterranean model.

I

GENERAL INTRODUCTION & OBJECTIVES

GENERAL INTRODUCTION

1. FISHERIES' MANAGEMENT FRAMEWORK: ECOSYSTEM APPROACH TO FISHERIES (EAF)

Fishing is one of the most widespread human pressures on marine ecosystems and it could be considered as one of the most harmful chronical human activities in the oceans (Jackson et al. 2001). Although fishing impact on ecosystems is not a recent concern (de Groot 1984, Osio 2012), during the last three decades it has arisen a renewed interest for a more ecological approach to this issue, focused on functional responses of the complex marine ecosystems (Thrush & Dayton 2002, Garcia & Cochrane 2005). Emerging from this interest, the term Ecosystem-based Fisheries Management (EBFM) was first described in 1998 by the US National Research council as "*an approach that takes major ecosystem components and services –both structural and functional – into account when managing fisheries*". However, the term did not find consensus in the 2001 FAO Reykjavik Conference, where participants considered that its interpretation could imply that environmental conditions prevailed over socio-economic and cultural ones (Garcia et al. 2003). Therefore, the term Ecosystem Approach to Fisheries (EAF) was adopted by the FAO Technical Consultation in 2002. Despite being essentially the same, this wording involved also socio-economic factors, linking society, economy and ecosystems, which fitted better countries' interests (Garcia et al. 2003).

The goals of EAF are ''*to balance diverse societal objectives, by taking into account the knowledge and uncertainties about biotic, abiotic, and human components of ecosystems and their interactions and applying an integrated approach to fisheries within ecologically meaningful boundaries*'' (FAO 2003). In accordance with these principles, in the 2002 revision of the Common Fisheries Policy (CFP), the European Union (EU) included the EAF in the new CFP (Lutchmann et al. 2008). Additionally, in 2008 the EU published the Marine Strategy Framework Directive (MSFD), which commit member states to integrate environmental concern in their marine policies, such as CFP, in order to achieve or maintain a Good Environmental Status (GES) by 2020 (Lutchman et al. 2008, EC2008/56). Aiming at advancing towards an EAF, the latest review of the CFP in 2013 established further measures such as maintaining populations above their Maximum Sustainable Yield (MSY), technical measures for protecting stocks and marine ecosystems, fishing capacity ceilings for member states, a discard ban, regionalization and further stakeholder involvement in the decision process (Salomon & Holm-Müller 2013). Some of these measures, as the regionalization of the CFP, had been long demanded and it is thought to be helpful for applying the EAF in different European regions by adapting management plans to regional particularities (Symes 2012). In this way, member states affected by a particular management measure can suggest changes to adapt the measure to its particular context as long as the suggested measures are at least as restrictive as the UE legislation (EC 2013/1380). The clearest example is the different management needs between Mediterranean and Atlantic fisheries. These two regions have very different fishing characteristics (see the "Mediterranean Fisheries" section for further details). Hence, a good management measure for one region might not be applicable to the other region.

Moreover, the new CFP stresses the importance of increasing the scientific knowledge on commercial stock assessment, the impact of fishing on marine ecosystems and fisheries socioeconomics aspects (EC 2013/1380), allocating specific funds for these studies (European Maritime and Fisheries Fund). This better scientific knowledge is the basis to build an EAF management plan.

EAF requires assessing ecosystems in an integrated way, taking into account environmental and socioeconomic aspects of the system to be managed. In this way, an integrated assessment will provide a framework to organize scientific knowledge in order to inform managers to design EAF measures (Levin et al. 2009, Tallis et al. 2010). Determining and defining the elements of the system of interest is the first step towards building an EAF (Fletcher 2008, Levin et al. 2009). Then, the description of these elements' role in the system, as well as their interactions, will constitute the basis to build the EAF framework. Figure I.1 introduces the fisheries system in which this thesis is based, defining its ecological and socioeconomic elements as well as its main relationships: decision makers establish management measures to regulate trawling activities (e.g. effort limitation measures) aiming to ensure the food provision for society. The impact of these trawling activities on the seabed will depend on the fishing effort level. Benthic community changes caused by trawling may affect target species and hence the food provision for society. A potential relationship is shown between “Management” and “Society” as society perception of fishing activities may influence management decisions. The studies included on this thesis assess these relationships focusing on how to better describe the fishing pressure, on understanding ecosystem functionality response under different levels of trawling effort and how to use this knowledge in management.

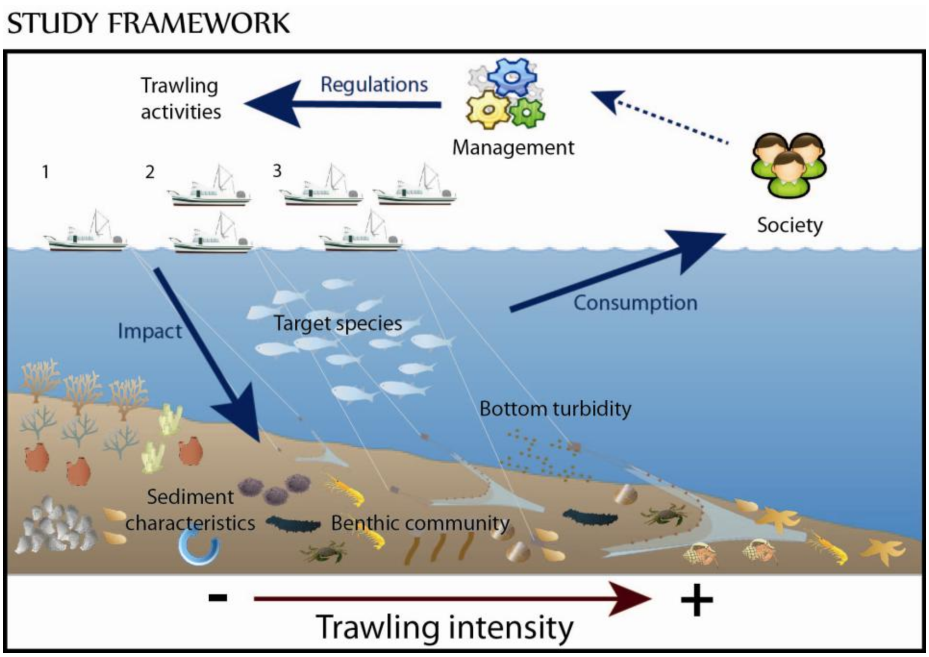


Figure I.1. Diagram representing ecological and socioeconomic elements in a fishery system (in this case trawling fisheries are represented). The numbers 1, 2 and 3 represent different qualitative levels of fishing effort, from low to high according to the number of vessels. Full arrows define actual interactions whereas the dashed arrow points out potential interactions. Black font represents the system elements and blue font the relationship between these elements. Benthic community structure under different trawling intensity is based on de Juan et al. (2009) and de Juan & Demestre (2012)

2. THE NEED TO ADOPT AN EAF: IMPACTS OF TOWED FISHING GEARS ON BENTHIC ECOSYSTEMS

Fishing technics can be grouped in two main categories: passive fishing techniques, which include the use of pots or traps, baited hooks on set lines, gill nets and drift nets and active fishing methods that usually involve towing trawls or dredges (Lleonart & Sardà 1985). They all provoke changes on marine trophic webs by targeting (and hence removing from the ecosystem) organisms belonging to the highest trophic levels (Pauly 1998). Other non-trophic relationships (e.g. facilitation or apparent competition) between species might be also affected by fishing (Dill et al. 2003). Furthermore, fishing not only affects marine organisms but also may cause a mortality increase on predator seabirds' populations (Tudela 2004, Lewison et al. 2004, Frederiksen et al. 2004). When dealing with fisheries it is also important to take into account by-catches and discards as their omission may lead to underestimations of fishing mortality (Bellido et al. 2011) . Moreover, fisheries bycatch not only affect fish or invertebrates, but also megafauna such as turtles or marine mammals (Frid et al. 2005; Lewison et al. 2004; Tudela 2004). However, the most acute and chronical impact suffered for marine ecosystems is that inflicted to seabed ecosystems, where towed gears are the ones causing the most severe damage (Dayton et al. 1995, Jennings & Kaiser 1998, Thrush et al. 1998) (Box 1).

The extend of the impact would depend on the actual level of fishing effort. Hence, in order to assess fishing impacts on benthic ecosystems, the first step is to obtain an accurate estimation of the fishing effort to

BOX 1: Overview of trawling impacts on marine ecosystems

1.Target species

References	Observations
Pauly et al. 1998 Myers & Worm 2003	There has been a transition in landings from long-lived high trophic level species to smaller size and lower trophic level species.
FAO 2014 Pontecorvo & Schrank 2014 Lleonart & Bas 2012	Catches are declining worldwide and around 90% of commercial stocks are fully exploited or overexploited

2.By-catch and discards

References	Observations
Lewinson et al. 2004 Tudela 2004	Population decreases due to bycatch mortalities of marine megafauna (seabirds, turtles, marine mammals, sharks)
Tudela 2004 Catchpole et al. 2005 Vassilopoulou et al. 2007	Otter bottom trawling discard about 45% of total catches which provoke changes on the overall trophic web structure and habitats.

3.Benthic community

References	Observations
Dayton et al. 1995 Jennings & Kaiser 1998 Thrush & Dayton 2002	Decrease of benthic fauna biomass and density. Destruction of complex habitats formed by seagrasses and sessile animals (corals, sponges, gotgonians., bryozoans...) Changes in species diversity
Bremner et al. 2003 de Juan et al. 2007 Thrush & Dayton 2010	Functional changes: Loss of large long-lived organisms, decrease of epifaunal filter feeding organisms, increase of small infauna and scavenging species Decrease of the extend of the bioturbated and bioirrigated sediment layer Productivity decrease

4. Sediment

References	Observations
Jennings & Kaiser 1998 Thrush & Dayton 2002	Sediment homogenization, loss of microstructures.
Kaiser et al. 2002 Martín et al. 2014	Sediment resuspension, contaminants remobilization Changes in sediment particle size, and organic matter content
O'Neill & Summerbell 2011 Palanques et al. 2014	Increase in near-bottom turbidity

what the community is subjected (Thrush et al. 1998, Collie et al. 2005, Atkinson et al. 2011, de Juan & Demestre 2012). There are many ways to estimate the fishing effort (e.g., swept area per time unit, catch per unit effort (CPUE) or hours at sea), but these estimations are normally obtained at too large scales to properly link effort and fishing impact on benthic communities (Piet et al. 2007, McCluskey & Lewison 2008). Effects of fishing on benthic communities are normally spatially heterogeneus, hence, an accurate small scale estimation is needed when assessing trawling effects (Piet & Quirijns 2009) (See objective 1 in the objectives section).

Trawling and dredging are normally performed on soft-bottoms and affect both, physical and biotic habitats (Fig. I.2) (Box 1). On the physical side, they are known to homogenize the seabed by eliminating microstructures that may serve as refuge for some organisms (Jennings & Kaiser 1998, Thrush & Dayton 2002). Regarding the biological community, fishing tend to remove large, sessile and long-lived organisms (Kaiser et al. 2000, Thrush & Dayton 2002, de Juan et al. 2007a). In addition, the pass of the trawl or dredging gears leave many damaged organisms that attract scavengers (Demestre et al. 2000, Rumohr & Kujawski 2000), which cause benthic communities to become dominated by small and short lived scavenging species. Moreover, bottom fishing increases near-bottom water turbidity (O'Neill & Summerbell 2011, Palanques et al. 2014), which may interfere with filtering systems of filter feeding organisms (Tjensvoll et al. 2013). Sediment characteristics are also altered by bottom trawling (e.g. change of sediment granulometry and the organic matter content on the sediments (Martín et al. 2014)), which may indirectly affect benthic community assemblages as sediment characteristics play an important role on species' distribution (Fig. I.2) (Macdonald et al. 2012).

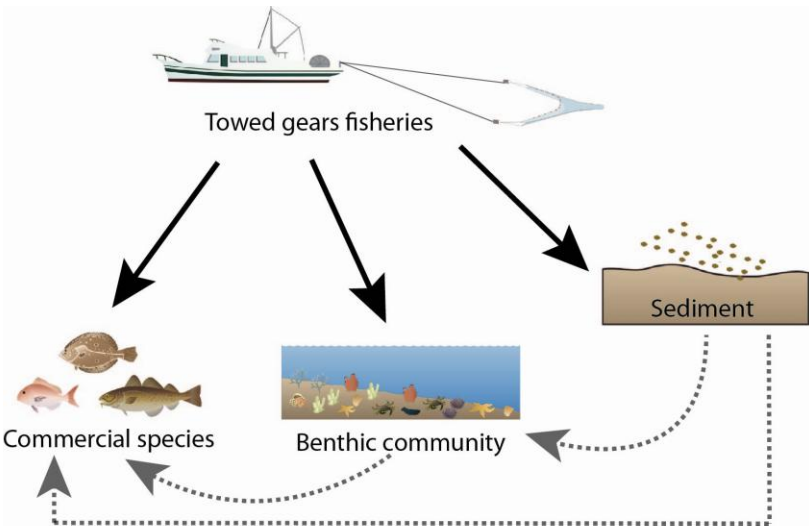


Figure I.2. Demersal towed gear's impacts on marine ecosystems. Full arrows represent direct impacts while dashed arrows represent indirect effects.

Besides, changes on benthic ecosystems can ultimately affect commercial fish populations, as fish species need particular habitat characteristics to fulfill their life cycles (Fig. I.2). This range of habitats that fish need for feeding, spawning or growth to maturity are called Essential Fish Habitats (EFH) (Auster & Langton 1998). Some benthic ecosystems affected by towed gears constitute also EFH, therefore, fishing does not only have a direct impact on commercial fish populations by direct organisms´ removal, but it also indirectly affects fish by altering their EFH (Fig. I.2). Actually, some studies have already proven that fish diet changes in heavily fished areas (Frid et al. 1999, Lloret & Sánchez-pardo 2007, Juan et al. 2007b, Smith et al. 2013). However, whether changes caused by fishing benefits or not commercial fish populations feeding on benthos is still controversial as different studies led to different conclusions (Van Denderen et al. 2013, Hiddink et al. 2011, 2008). Nevertheless, major consensus exits on the loss of organisms forming 3D habitat structures due to demersal towed fisheries, which clearly have negative effects on juveniles seeking refuge or on adults´ spawning areas (Gregory & Anderson 1997, Thrush & Dayton 2010). Hence, to maintain target species stocks under sustainable exploitation levels, ecosystem changes caused by fishing must be taken into account, which requires an EAF management plan.

2.1. Evolution in the study of trawling impacts on benthic communities: from species approach to ecosystem functioning.

Many studies have been carried out in order to understand the effects of fishing on the benthic community structure, the first approaches being based on species composition (e.g. de Groot 1984, Philippar 1998, Norse & Watling 1999, Kaiser et al. 2000). This strategy increased our knowledge on community responses, but it provided limited information on ecosystem functioning and, additionally, it was often geographically limited (Bremner 2008). To address the functionality issue, trait analysis (i.e. based on species' behavioral, morphological and life-history characteristics) has proved to be a useful approach. As different species may share the same traits, the trait approach allows to overcome geographic differences, highlighting community responses which are difficult to detect using the species approach (Bremner et al. 2006b, Tillin et al. 2006, de Juan et al. 2007a).

Moreover, ecosystem functional dynamics shows a tight link with ecosystem functional performance, which will ultimately depend on the functional roles performed by the species. Finally, the species' functional roles will be determined by the traits they hold. For example, bioturbation behavior enhances sediment-water column nutrient fluxes (Emmerson & Raffaelli 2000, Lohrer et al. 2004, Laverock et al. 2011) and polychaete tubes increase sediment stability (Snelgrove 1997, Passarelli et al. 2012). Therefore, the study of ecosystem functionality helps the understanding of ecosystem processes (e.g. Austen et al. 2002, Bolam & Eggleton 2014) and how these processes change when the ecosystem is subjected to an impact, e.g. fishing (Cooper et al. 2008, Bolam et al. 2014). The direction of this change will be determined by the traits' vulnerability/resistance to the pressure. For instance, in the trawling fisheries' case, the traits "filter feeding" and "sessile" are vulnerable traits that tend to be removed from the system and the traits "small" and "deep burrowing" are resistant traits that prevail in fished communities (Thrush & Dayton 2002, de Juan et al. 2009).

In this context, the concept of functional redundancy, i.e. functionally equivalent species, is very important when dealing with ecosystem disturbance such as fishing, as it may represent an insurance for maintaining ecosystem processes under stress (Walker 1992, Boero 1994). Although sharing the same traits, functionally equivalent species may have different responses to stress, hence the loss or decrease of a certain species may be offset by an increase of a redundant species. Therefore, redundancy is closely related to ecosystem resilience described as the capacity of a system to absorb disturbance and reorganize while undergoing change so as to retain essentially the same function, structure, identity, and feedbacks (Walker et al. 1999, Folke et al. 2004, Bremner 2008). Hence, those communities having low levels of redundancy will be more vulnerable to trawling and they will deserve special attention in management plans (Walker 1992).

However, it is difficult to know to what extend this redundancy is sufficient to allow species loss without significantly altering ecosystem processes (Snelgrove 1998) as in a pool of redundant organisms certain species might contribute more to the ecosystem process than others (Bolam et al. 2002, Solan et al. 2004). Thus, if these species disappear, ecosystem functionality may change dramatically. When this happens, the ecosystem is on the limits of its resilience and a regime shift (i.e. a broad-scale change in ecosystem species composition and function) may occur (Folke et al. 2004, Thrush et al. 2009). The regime shift leads to an ecosystem alternate state in which species interactions are reorganized and ecosystem functionality changes (Choi et al. 2004, Folke et al. 2004). Management actions should be aimed at enhancing ecosystem resilience in order to avoid a regime shift that may be very difficult to revers (Suding et al. 2004, Troell et al. 2005).

Therefore, ecosystem functionality knowledge is a key issue to design an EAF. Consequently, it has been also addressed in the studies presented in this thesis (see objectives 2 and 3 in the objectives section).

2.2. Evolution in the study of trawling impacts on benthic communities: from ecosystem functioning to ecosystem services

Lately, the functionality approach has evolved to link ecosystem functions to ecosystem services (Bremner et al. 2006a, Granek et al. 2010, Bello et al. 2010, Frid 2011). “Ecosystem services” can be defined as “the benefits people obtain from ecosystems” (e.g. climate regulation, nutrient cycling, flood prevention…) (Millennium Ecosystem Assesment 2005), hence this approach allows to include a societal point of view in ecosystem management. In this way, ecosystem functionality scales up from an ecology-focused framework to include society concerns on ecosystem studies that match with the integrated assessment needed in an EAF (Salomon et al. 2011). In this context, ecosystem services, covering ecological and socioeconomic aspects, could be used as a common language in EAF management framework in order to bring scientists’ and fisheries stakeholders’ positions closer (Granek et al. 2010, Atkins et al. 2011). Therefore, it is important to quantify the ecosystem services that benthic communities’ can deliver and to understand how they respond to fishing pressure.

Focusing on soft bottoms subjected to fishing pressure (i.e. fishing grounds) the most obvious delivered service is “food provision” through fish catches. However, fishing grounds can also deliver other ecosystem services, particularly supporting services (e.g. habitat provision or nutrient cycling) (Snelgrove 1997, Austen et al. 2002). Maintenance of these services is important to keep ecosystems in a good status, so they should be also taken into account by fisheries managers (Garcia & Cochrane 2005). Furthermore, the “food provision” service relies on other services such as secondary production or habitat provision, which are important aspects of commercial species EFH (this issue is discussed in chapter 3).

An EAF should consider globally all ecosystem services of a fishing ground, in order to get a complete picture of the system to be managed (this issue is discussed in chapter 2, section 1).

3. ECOSYSTEM APPROACH TO FISHERIES IMPLEMENTATION

Despite many countries are already leading their fisheries’ policies towards and EAF (Murawski 2007, Bianchi & Skjoldal 2008), the overall implementation of EAF in world’s fisheries is still in its infancy. Pitcher et al. (2009) evaluated the performance of EAF in 33 countries and none was rated as having a “good” overall performance while only four were rated as “adequate”. Efforts to implement EAF are being made worldwide, but the transition from single stock assessment to EAF is not easy as it entails a change of fisheries’ management paradigm (Rice 2011, Berkes 2012).

EAF management plans have been applied for Australian Fisheries (McLoghlin et al. 2008, Day 2008) and for some US fisheries (Tromble 2008), although in this last case it is not considered to be fully applied and, among others, more collaboration between scientists and policy makers is needed (McFadden & Barnes 2009, Samhouri et al. 2014).

The EAF plans in the Australian and US cases are slightly different. The basis for the Australia EAF has been the development two key tools: a comprehensive risk assessment evaluation and a harvest strategy for target stocks in every fishery (McLoghlin et al. 2008). The risk assessment evaluation can be applied to both, data-rich and data-poor fisheries and it is useful to detect knowledge gaps. The harvest strategy is periodically reviewed based on data from a permanent stock monitoring program. On the other hand, the US EAF is based on Fishery Ecosystem Plans, based on the Integrated Ecosystem Assessment (IEA) approach (Levin et al. 2009). The IEA approach is based in ecosystem modelling, ecological indicators and adaptive management simulation, hence US EAF is often founded on models outputs (Link et al. 2011). Those Fishery Ecosystem Plans have been implemented on most of the US fishing areas (Tromble 2008). Both strategies are ecologically focused, putting major efforts on understanding the ecological impacts of fishing and, while trying to balance ecosystem and society objectives, both apply the precautionary approach.

Canada has also make some steps towards an EAF, but, rather than focusing only on fisheries, they chose an integrated management approach which takes into account different ocean industries (e.g. fisheries, oil and gas development, aquaculture…) (O'Boyle & Jamieson 2006). Its main aim is the harmonization of the activities of the different industries in order to achieve common and agreed management goals (e.g. conserve ecosystem components). Stakeholders are directly involved in the planning process, which is intended to be flexible and transparent (Link et al. 2011).

In Europe, EAF has been applied in Norwegian waters (Barents Sea and Norwegian Sea), although some problems on monitoring have been reported due to lack of coordination of ministry sectors (Ottersen et al. 2011). Similarly to Canada's strategy, the Norwegian EAF plan is designed as an integrated management plan that includes other human pressures such as petroleum development and shipping as well as fisheries. An environmental assessment per sector was carried out along with an identification of vulnerable areas. Then, based on this information, an integrated management plan was implemented and a monitoring program was established (Winsnes & Skjoldal 2007, Ottersen et al. 2011).

North Atlantic fisheries are also advancing towards an EAF, implementing management tools that involve not only a species approach but also an ecosystem scope (e.g. multispecies assessment) (Frid et al. 2005, 2006). However, to achieve the integrated socio-ecological approach of an EAF plan, socio-economic indicators should be also considered to inform management. An ecosystem-based management plan was developed based on a stakeholders' consultation process followed by scientific modelling (Paramor et al. 2005). Most stakeholders considered important to reduce by-catch, especially of charismatic species, but only a few considered important to protect non-target species and habitats. They supported effort control measures, such as temporal closed areas to protect commercial species, but complete closure areas to protect benthic ecosystems received little support (Paramor et al. 2005), which resulted in a lack of progress in habitat protection measures (Frid et al. 2006). Nowhere else in Europe, including the Mediterranean region, an EAF has been implemented despite all the mentioned European laws advocating for it. This failure could be due to the scientific gaps in ecosystem dynamics knowledge as well as a lack of collaboration and communication among stakeholders involved in fisheries (i.e. politicians, scientists, fishermen, managers…) (Coll et al. 2013, Daw & Gray 2005, de Juan et al. 2012, Frid et al. 2006). Moreover, despite recognition that humans are part of marine ecosystems´ dynamics, there is still a gap between ecological sciences, social sciences and management (Samhouri et al. 2014).

Communication and understanding among all actors involved in fisheries management is a key issue if the EAF has to be successfully implemented. However, this is a difficult task often hindered by the lack of understandable information exchange approaches among stakeholders (Verweij et al. 2010, Soomai et al. 2011). For example, a scientific graph showing the evolution of spawning stock biomass may tell nothing to fishermen who are more familiar with catches or landings data. Scientists must make an effort to compile all the knowledge gathered on their studies and make this information accessible and understandable to stakeholders involved in fisheries management so that this knowledge helps in building an EAF (Coll et al. 2013, Garcia & Cochrane 2005). To this aim, one of this thesis objectives was the construction of a knowledge platform were the effects of fishing on soft-bottoms communities were presented in a comprehensive and understandable way (see objective 4 in the objective section).

3.1. EAF in the Mediterranean

Despite the need for an EAF in the Mediterranean, the current fisheries advice is still mainly based on single stock assessment (Spagnolo 2012). Sartor et al. (2014) evaluated the current information available to implement an EAF in the Mediterranean and found only an overall low-medium requirement fulfillment. This work also highlighted the imbalance in knowledge gathering for different EAF aspects: while fleet structure and species distribution is relatively well known, modelling, socio-economic and management issues are poorly assessed. Moreover, although some local initiatives to implement more ecosystem oriented management strategies exists, they have highly overlapping themes and are poorly coordinated (Coll et al. 2013).

Additionally, many problems arise when trying to implement an EAF in the Mediterranean, e.g. the lack of systematic collection of data regarding the spatial distribution of biological communities and fishing activities or the complex political and geographic situation which involves European Union member states, Middle-eastern and African countries (Coll et al. 2013, de Juan et al. 2012, Caddy 2012, Spagnolo 2012).

The Mediterranean Ecosystem Approach Strategy (ECAP) was proposed in 2005 aiming at strengthening efforts to achieve Healthy Environment Status by 2020 (similar to GES of MSFD for EU countries) (Cinnirella et al. 2014). This strategy was launched under the Mediterranean Action Plan (MAP) which is legally binding for the contracting parties (21 Mediterranean bordering countries) (http://www.unepmap.org). However, ECAP has been poorly implemented due to lack of collaboration between countries and because of limited funding and human resources of Mediterranean non EU countries (Cinnirella et al. 2014).

Recently, an interesting initiative to enhance Mediterranean countries' cooperation towards and EAF has arisen: the EMBASEAS (Eaf in the Mediterranean and BlAck SEAS) network created by CREAM (Coordinating research in support to application of Ecosystem Approach to Fisheries and management advice in the Mediterranean and Black Seas) project framework. EMBASEAS aims at promoting a scientific approach to EAF by coordinating initiatives and sharing relevant information among Mediterranean researchers as well as establishing bridges between scientists, policy makers and other sea users (EMBASEAS Network Constitutional Framework 2014).

In the context of CREAM, Coll et al. (2013) identified gaps of knowledge and lack in information on a series of Mediterranean topics needed to advance an EAF in the region. These topics included the quantification of the real impact of fishing, the description of basic ecological processes and patterns, socioeconomic subjects such as the quantification of ecosystem services and real fishing effort and fishing fleet behavior. This thesis addresses some of these topics aiming to contribute to the scientific knowledge baseline to build an EAF in the Mediterranean.

3.1.1. Marine Protected Areas (MPAs): first step towards a Mediterranean EAF

A first approach towards the establishment of an EAF management plan in the Mediterranean are the MPAs, which are seen as a good and “easy to implement” precautionary approach tool in EAF management (Stergiou 2002, Tsikliras & Stergiou 2007, García-Charton et al. 2008). Actually, it has been demonstrated that MPA establishment enhances species’ population and somatic growth as well as undisturbed nursing that leads to a successful recruitment (Tsikliras & Stergiou 2007, Harmelin-Vivien et al. 2008, Higgins et al. 2008). However, the spill-over effect in MPAs (i.e. adults migrating to adjacent waters) may be very limited and there is little evidence of its contributions to the surrounding fisheries (Harmelin-Vivien et al. 2008, Higgins et al. 2008, García-Rubies et al. 2013, but see Goñi et al. 2010). Benthic communities (infauna and epifauna) generally also experience biomass recovery within MPAs, leading to a more complex and structured communities (Babcock et al. 1999, de Juan et al. 2011). Hence, MPAs do play a role in habitat protection, but this protected habitat has to be large enough in ordrer to benefit target species (Allison et al. 1998, Browman & Stergiou 2004, de Juan et al. 2012).

In addition, very few of the Mediterranean protected surface include no-take areas as most MPAs allow recreational and artisanal fishing

(Tsikliras & Stergiou 2007, de Juan et al. 2012). Although not as harmful as trawling or dredging, artisanal and recreational fishing does have an impact on the benthic communities and resources that are meant to be protected within the MPAs (Font 2014, Purroy et al. 2014). Moreover, MPAs often attract divers and other touristic activities, which, if not controlled, may damage MPAs communities (García-Charton et al. 2008). Therefore, regulations of those activities in the MPAs have to be carefully planned, specially artisanal and recreational fisheries, whose activity is very important in the Mediterranean (Guidetti et al. 2010, Morales-Nin et al. 2010).

Furthermore, it is worth to bear in mind that the implementation of MPAs in Atlantic trawling grounds has led to an effort displacement towards areas previously holding low or no effort (Dinmore et al. 2003, Hiddink et al. 2006). This behavior is predicted to have slightly greater cumulative impacts on benthic communities (Dinmore et al. 2003), hence this effort displacement ought to be controlled. However this is not likely to occur in the Mediterranean as each port trawl fleet normally operates in its area of influence and there is no fleet movement between ports (see the next section). Nevertheless, an experience carried out in the Mediterranean (a trawl ban in the Gulf of Castellamare) lead to an effort concentration in the outer periphery of the MPA, increasing the trawling pressure in this area (Whitmarsh et al. 2002). Moreover, the concentration of the trawl fleet in the MPA borders interfered with the artisanal fishers operating in this area (Whitmarsh et al. 2002). Therefore, when implementing an MPA in trawling grounds, redistribution of the fleet has to be carefully planned.

However some authors claim that a good design of MPA networks might result in more clear benefits to fisheries (Hilborn et al. 2004, Gaines et al. 2010), but for that to be true MPAs should be designed following ecological criteria rather than sociopolitical interests as is now the case for Mediterranean MPAs (Francour et al. 2001, Coll et al. 2012, de Juan et al. 2012). For example, most of the Mediterranean MPAs belong to coastal territorial waters, failing to protect deeper water EFH (Lloret et al. 2008, Gabrié et al. 2012). Moreover, most of them are located around islands and in rocky bottoms, hence not covering potential trawling grounds (Gabrié et al. 2012).

MPAs network should be regarded as one of the EAF management tools in the wider concept of Marine Spatial Planning, i.e. managing the multiple uses (different fishing gears, recreational uses…) of marine spaces as a whole (Douvere 2008). Marine Spatial Planning organizes marine space stablishing different use zones, from zero-use areas to general access areas. In this way, fishermen reluctance towards no-take MPAs (Jones 2008) might be overcome by considering fisheries' interests as well as conservation concerns in MPAs design (Gaines et al. 2010, Salomon et al. 2011). In this context, MPAs planning goes beyond the establishment of MPAs network and will help to maintain sustainable marine ecosystems in line with the "blue economy" philosophy (i.e. an economy in which ocean ecosystems bring economic and social benefits that are efficient, equitable and sustainable (Australian Government 2012)).

4.MEDITERRANEAN FISHERIES

4.1. Fisheries' characteristics

In order to understand the difficulties facing the design and implementation of an EAF in the Mediterranean, it is worth to introduce an overview of Mediterranean fisheries' characteristics and current regulations.

As described by many authors, Mediterranean fisheries are typically multispecies and multi-gear, with different gears competing for the same resource, which hinders the application of general regulations (Farrugio et al. 1993, Lleonart et al. 1998, Papaconstantinou & Farrugio 2000, Bas 2006, Spagnolo 2012). Mediterranean fisheries are also characterised by the strong demand for fresh fish and the high production cost of exploitation, which makes market prices to be normally high. Thus, when studying Mediterranean fisheries it is important to take into account catch weight but also income (Spagnolo 2012, Caddy 2012, Maynou et al. 2013).

There are two main types of fisheries in the Mediterranean (Demestre 1986, Lleonart 1990, Papaconstantinou & Farrugio 2000, Lleonart & Maynou 2003) (Fig. I.3b):

i) Pelagic fisheries, which include seine nets that target small pelagics, such as sardine and anchovy, long line and driftnets for large pelagics, such as tuna and swordfish.

ii) Benthic-demersal fisheries, comprising trammel nets, gillnets, bottom longlines, drags, traps and, the most wide spread gear, bottom trawling characterized by multiespecific target species (e.g. hake, red mullet, angler, shrimps, lobster, sole, squid or octopus). In some Mediterranean areas semi pelagic trawlers are also allowed.

About 70-80% of the whole Mediterranean fishing fleet is composed by small-scale artisanal fisheries that use a wide range of gears (Papaconstantinou & Conides 2007, Lloret et al. 2008, Spagnolo 2012). Actually, most of the abovementioned gears are considered artisanal and only seine nets, tuna fisheries' and bottom trawling are performed with vessels larger than 12m (Lloret et al. 2008). Although these later vessels are the largest fisheries' boats in the Mediterranean, they are still small compared to the large Atlantic industrial vessels, hence, they are referred as a semi-industrial fleet rather than an industrial fleet (Farrugio et al. 1993, Lleonart et al. 1998, Papaconstantinou & Conides 2007).

Among these Mediterranean semi-industrial fleets, demersal otter-trawling is the gear having the most dramatic and chronic effects on benthic ecosystems (Tudela 2004, de Juan et al. 2007a, Sacchi 2008, Mangano et al. 2013). Otter-trawlers normally operate over extended and homogeneous soft-bottom areas on the continental shelf (i.e. fishing grounds) (Demestre 1986, Lleonart 1990). However, due to the narrowness of the Mediterranean continental shelf (Fig. I3a), the continental slope is close to the coast, which has favored deep sea fisheries (Sardà 1998a, Bas 2006, Mytilineou & Machias 2007). Therefore, trawling spreads over the upper slope and even down to submarine canyons (Demestre & Martín 1993, Sardà 2000) (Fig. I.3b).

Fishing areas are normally close to the base port, hence, fishing trips are short, usually not longer than a day, and catches are landed in a daily or bi-daily basis (Caddy 1993, Lleonart et al. 1998, Bas 2006, Spagnolo 2012).

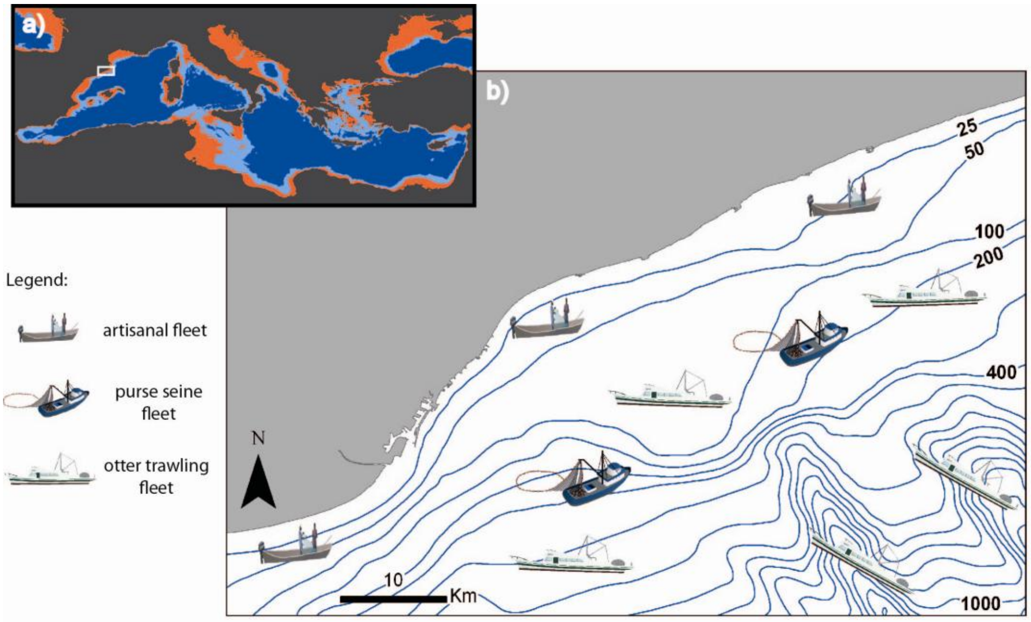


Figure I.3. a) Mediterranean Sea. The orange area represents the continental shelf (≤200 m) and the dark blue the area represents >1000 m where trawling is forbidden (EC 1967/2006). b) Detail of a coastal zone showing the distribution of the main Mediterranean fleets. (2005. Catalano-Balearic Sea - Bathymetric chart ww.icm.csic.es/geo/gma/MCB).

Seasonality of landings is another fisheries' characteristic to take into account when managing Mediterranean fisheries. Fishermen usually change fishing grounds over the year following different target species, with a spatial distribution that depends on their life cycle (Sardà & Martín 1986, Demestre et al. 1997, Martin et al. 1999, Halley & Stergiou 2005). Unfortunately, in some cases, the trawl fleet concentrates in recruitment areas (e.g. red mullet (Demestre et al.

1997, Martin et al. 1999)). This behaviour, along with the high fishing pressure exerted by trawlers all along the Mediterranean, has led to the overexploitation of many demersal stocks (Colloca et al. 2013, Lleonart 2008, Lleonart & Bas 2012).

In addition, it is important to pay attention to the special characteristics of the Mediterranean trawl fleet discards. Trawl gears have low selectivity and produce a high number of non-commercial discards (around 45% of the total annual catches (Vassilopoulou et al. 2007, Sacchi 2008, Tudela 2004)). The number and composition of discards shows high spatio-temporal variability and discarded species or individuals are normally smaller, and belong to a significant lower trophic level, than the marketable ones (Martín 2001, Sánchez et al. 2004, 2007, Vassilopoulou et al. 2007). Among these discarded species there are also many invertebrates that play important roles in the ecosystem function performance (e.g. irregular urchins such as *Schizaster canaliferus* which are important bioturbators (Lohrer et al. 2004)). Moreover, these discards provide food subsides for birds and other marine species (Vassilopoulou et al. 2007, Bellido et al. 2011, Tudela 2004), also potentially changing marine ecosystem structure and functionality. Therefore, discards are another source of concern in trawling fisheries. In this context, it is also worth to pay attention to the discard ban measure included in the new CFP, which, according to the European Commission, will encourage fishermen to make their gears more selective (Condie et al. 2013). However, the obligation of bringing to port commercial unwanted catches by 2019 (EC 2013/1380) may enhance in the future the emergence of different sources of markets for these incidental catches which might discourage fishermen to increase the gears' selectiveness (Bellido et al. 2011).

4.2. Fishing regulations in the Mediterranean

Although CFP apply to all fishing activities undertaken within the EU fishing zone, fisheries policy was originally thought for northern European and Atlantic fisheries, which make it difficult to implement in the complex cultural, geographic, biological, economic and legal-political scenario in the Mediterranean (Casado 2008). For example, whereas northern European and Atlantic fisheries are being managed mainly through the Total Allowable Catch (TAC) quotas system, this system is not viable in the Mediterranean due to the multispecies nature of its fisheries. Therefore, this regulation system is only applied to tuna and tuna-like species, as this fisheries is mainly monospecific (Cacaud 2005, Caddy 2012).

Two main European Council regulations have been written up especially for the Mediterranean fisheries (EC 1626/94 and EC 1967/2006). The EC 1626/94 set up minimum mesh sizes and minimum landing size and promotes member States to establish protected areas where fishing is restricted. The EC 1967/2006 revised the previous regulation and, being more ecosystem-oriented, describes restrictions for fishing gears, protected species, habitats and fishing protected areas and establishes control measures. However, neither has been fully implemented (Casado 2008). Moreover, the Mediterranean Sea is surrounded by 21 coastal states and only eight of them belong to the EU, hence, fisheries management in the

Mediterranean has been conducted by General Fisheries Council for the Mediterranean (GFCM) and their resolutions are adopted by the Mediterranean EU policies (Papaconstantinou & Conides 2007, Casado 2008, Caddy 2012).

Mediterranean fisheries' management stablished in the above mentioned regulations is mainly based in effort control and minimum sizes measures.

a) Effort control regulations:

Trawl gears are prohibited within 3 nautical miles of the coast or below 50 m deep in order to protect vulnerable coastal habitats such as *Posidonia oceanica* meadows (EC 1626/94). More recently, a ban for trawl gears deeper than 1000 m was established aiming to protect deep-sea slow-growing, and hence vulnerable, organisms (EC 1967/2006, Fig.I3a). Moreover, the GFCM has long recommended a reduction of the overall fishing effort in the Mediterranean (Caddy 1993, Papaconstantinou & Conides 2007). To this purpose, many countries have opted to limit the number of fishing licenses and the EU has set up measures to monitor, control and even reduce the fishing capacity, e.g. vessel's withdrawal subsidies (Cacaud 2005, Spagnolo 2012). However, technological improvement increases the fishing capacity of the remaining fleet and the overall fleet effort remains practically unchanged or has even increased (Lleonart & Maynou 2003, Spagnolo 2012, Osio 2012). Other effort limitation measures applied in the Mediterranean comprise the limitation of the fishing time (days in a week or hours in a day) and the limitation of the total individual power (Papaconstantinou & Farrugio 2000, Cacaud, 2005). Closed seasons are also widely applied in order to protect some commercial species breeding season and recruitment periods, their length varying from 1 month in the Catalan coast to 4 months in Greece (Kapantagakis 2007, Demestre et al. 2008). However, in some cases, it is difficult to protect juveniles (the main idea of the closed season) because when the fishing activity recommences the trawling fleet concentrates on fishing young recruits (Demestre et al. 2008, 1997, Lleonart & Maynou 2003, Spagnolo 2012). Therefore, fleet activity after the closed season should be regulated in order to achieve a gradual reincorporation of trawling vessels in the fishing ground, and also the spatial distribution of fleet must be controlled to minimize the catches of recruits.

b) Minimum landing size

Another widespread regulation is the minimum landings´ sizes (Demestre 1986, Lleonart et al. 1998). Theoretically, this measure is aimed at preventing immature organisms' to be caught, but some of the established minimum landings sizes are below the first maturity size. For example, for hake (*Merluccius merluccius*; one of the most important trawl target species in the Mediterranean) the minimum landing size is 20 cm, whereas females reach their maturity at 39 cm and males at 32 cm (Lleonart et al. 1998; Recasens et al. 1998, Assessment 2011).

Furthermore, the small mesh sizes traditionally used in the Mediterranean (Bas 2006, Papaconstantinou & Conides 2007) led to catches below minimum landing sizes, hence, it was necessary to improve mesh selectivity. The European Regulation 1967/2006 established a minimum cod-end mesh size of 40 mm which should be squared shaped as it is more selective than the traditionally used diamond shape mesh (Bahamon et al. 2006, Ordines et al. 2006, Sacchi 2008). However, experimental studies have shown that, even using this more selective mesh size, catches still include individuals below the minimum landing size (Sala et al. 2008). Contrary to what fisheries' sector expected, the implementation of the 40 mm squared mesh does not seem to have reduced catches biomass or income (Samy-Kamal et al. 2014), but its performance on non-commercial species discards has not been assessed. Nevertheless, the improvement on the gear selectivity alone will not be enough to protect overexploited species and ecosystems and these measure should be accompanied by the effort control measures mentioned in the previous section (e.g. an overall effort reduction or the implementation of closed areas) (Demestre et al. 1997, Colloca et al. 2013, Sacchi 2008).

5. SOFT-BOTTOM COMMUNITIES IN THE MEDITERRANEAN: STUDY AREAS

Contrary to what is generally thought, soft-bottom communities conform structured communities that perform important ecological processes and deliver many ecosystem services (Snelgrove 1997, Thrush & Dayton 2002). Many commercial species live in these soft-bottoms (Demestre et al. 2000, Kallianiotis et al. 2000, Colloca et al.

2003), hence, soft-bottom communities constitute important fishing grounds that are exposed to severe impacts.

In order to deepen our knowledge on the response of soft-bottom communities subjected to otter trawling pressure and to increase the scientific knowledge needed to design an appropriate management plan in accordance with EAF, this thesis has studied benthic communities from seven different areas in the Mediterranean (Fig. I.4). Three of the studied areas were located in the Catalan coast (in the north-western Mediterranean): Cap de Creus-CC, Medes-M and Ebre delta-D; another one in the Murcia coast (in the south-western Mediterranean): Cabo de Palos CP; two in the Italian coast: Ligurian

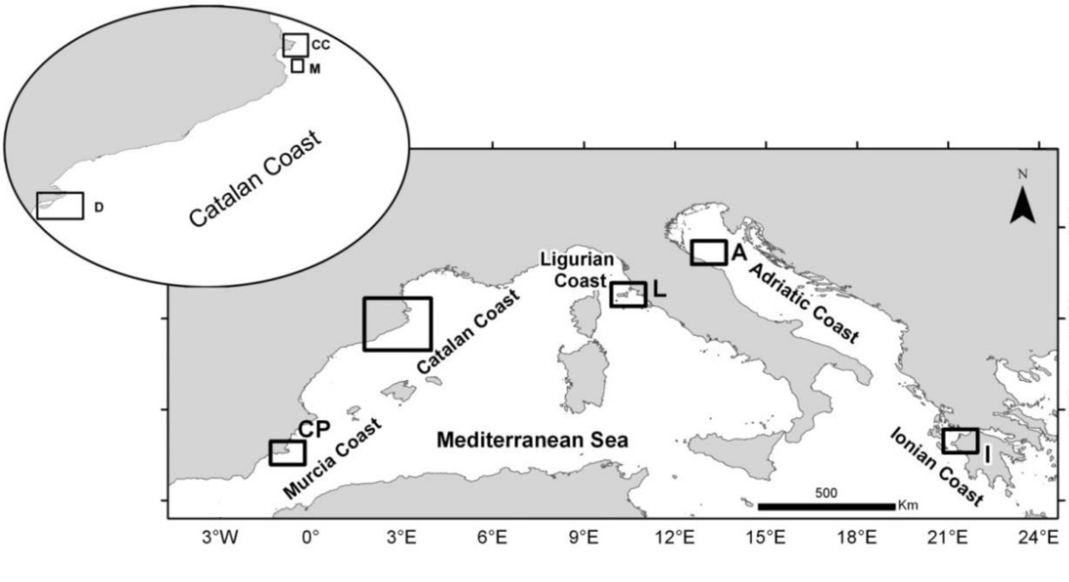


Figure I.4. Map of the areas studied in the thesis. CP: Cabo Palos, D: Delta, CC: Cap Creus, M: Medes L: Ligurian coast, A: Adriatic coast, I: Ionian coast.

coast-L and Adriatic coast-A and finally one in Greece: Ionian Coast-I.

The study sites comprise several habitat types having different sediment characteristics. M area was characterized by having coastal detritic mud. The L, I and D areas had muddy sediments, while CC and A areas had sandy-mud sediments. The CP area was characterized by maërl protruding within sandy-mud bottoms.

All these sites were located on soft bottoms in the continental shelves between 40-80 m, where commercial trawling is performed, except for M area, which was an MPA and the CC area included a portion of a MPA where trawling is banned.

Benthic community includes the epifauna (organisms that live on the sediment, although sometimes they can be buried in it, e.g., sea stars, bivalves) and the infauna (smaller organisms that live in the sediment, e.g., polychaetes, small crustaceans and bivalves). In order to correctly characterize the benthic community, both compartments must be taken into account (Jørgensen et al. 2011). Hence epifauna was studied in all areas but infauna data was only available for D, CC,L and I areas.

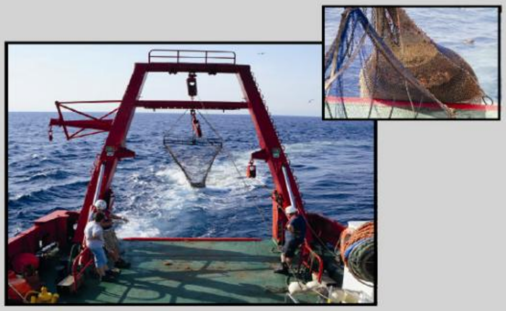
Epifaunal community was sampled with an epibenthic dredge with a 2m iron-frame aperture and a 1 cm of mesh size at the cod-end. Six samples were randomly collected in each site, the dredge (Box 2) towed for 15 minutes with a constant speed of 2.3 knots and a scanmar device attached to the dredge to ensure continuous contact with the seabed. Infauna was sampled with a 0.1 m^2 Van Veen grab (Box 2). Five replicates were randomly collected in D and 3 replicates were collected in all the other sites. To obtain the minimum sample size 5 grabs were collected per replica and pooled (de Juan et al. 2007a). All samples were collected between spring and early summer: June-July 2003 for D and A, May 2007 for M and June 2009 for the other areas.

Within this thesis the available data is used differently in the following chapters according to the aim of each chapter.

BOX 2 : Epifauna and infauna sampling gears and samples

a) Epifauna sampling

Epibenthic dredge



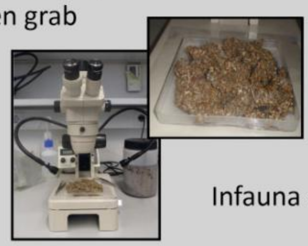
Dredge manoeuvre

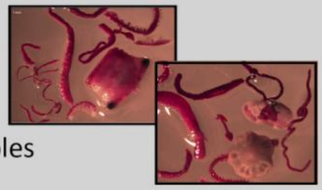
Epifauna samples

b) Infauna sampling

Van Veen grab

Grab manoeuvre





Infauna samples

THESIS OBJECTIVES

This thesis was conceived under the EAF framework, hence the objectives were set up aiming to increase the knowledge needed to design an EAF management plan in the Mediterranean.

The main aim of this thesis is to adopt an integrated approach to the benthic community (infauna and epifauna) response to commercial trawling in the Mediterranean continental shelf and assess how ecosystem service delivery, such as commercial species catches, could be compromised by this response.

This main objective was split in five specific objectives addressed in the different chapters. Each of the specific objectives represents one step towards the understanding of benthic ecosystems dynamics under trawling disturbance; the first step being a precise description of the stress (objective 1). Then, the description of the stress' effects on benthic functionality based on BTA (objective 2) and how this stress may indirectly affect commercial species (objective 3). Finally, the integration of all the acquired knowledge in a simulation model is presented (objective 4). Objective 5 represents the application of the acquired knowledge in the precedent objectives towards a possible case example of Mediterranean EAF implementation.

Specific objectives:

1. **To explore a methodology to monitor the fleet dynamics and to estimate trawling fishing effort in a benthic scale.** To achieve this objective in section 1 of **chapter 1,** we explored the potential of Vessel

Monitoring System (VMS) records as a source of vessel position combined with data of landings income and biomass from auction to monitor the spatial distribution and temporal dynamics of a North Catalonian trawling fleet. In section 2 of chapter 1 we explored the ability of three different ways of estimating trawl fishing effort (fishermen interviews, VMS and side scan sonar (SSS) images) to link effort estimation with fishing driven changes on benthic communities.

2. **To estimate benthic ecosystem response under different disturbance regimes at functional and ecosystem services levels. Chapter 2,** section 1 aims to explore how different benthic ecosystem functions behave under trawling disturbance and assess how potential changes might compromise the delivery of ecosystem services. Section 2 explores benthic community redundancy as a measure to assess ecosystem resilience to trawling.

3. **To link changes in seafloor communities due to trawling with potential effects on target species.** In **chapter 3** we compared benthic community functionality from a trawling fishing ground and a control site undisturbed for 20 years. Based on this information, an important Mediterranean target species such as red mullet (*Mullus barbatus*) was chosen to discuss the possible effects of chronic trawling on a commercial species population.

4. **To develop a conceptual and simulation model to be used as a deliberation support tool by fisheries' actors.** To address this issue, a user-friendly platform is introduced in **chapter 4,** where the benthic community is represented through biological traits known to be

vulnerable/resistant to trawling. This platform enables the user to visualize the benthic community structure of several soft-bottom areas subjected to different levels of fishing effort and provides the possibility to test how a change of this fishing effort might affect benthic community structure.

5. **To suggest management measures in the EAF framework in order to mitigate demersal trawling effects on benthic communities.** This issue is addressed in **chapter 5**, compiling the results from previous chapters as well as scientific literature information. This chapter recommends a series of measures to implement an EAF for the Catalan trawl fleet which might be extended to other Mediterranean fleets.

REPORT OF THE SUPERVISORS

Report of the advisors on the authorship and Impact Factor of the research papers included in the Ph.D.

Dr. Montserrat Demestre Alted and Dr. Silvia de Juan Mohan, supervisors of the Ph.D presented by Alba Muntadas Olivé, certify that the seven studies included in this work have been submitted to international journals that are subjected to peer review. Three of these works have been published, three are currently under revision, and one was sent to the journal on July 2015. The details for each publication and their Impact Factors are included bellow:

CHAPTER 1:

Section 1: Performance of a NW Mediterranean trawl fleet: integration of landings and VMS data to promote the implementation of Ecosystem- Based Fisheries Management.

Martín, P., **Muntadas, A**., de Juan, S., Sánchez, P., Demestre, M. (2014). Performance of a NW Mediterranean trawl fleet: integration of landings and VMS data to promote the implementation of Ecosystem- Based Fisheries Management. *Marine Policy 43*, 112–121. DOI: 10.1016/j.marpol.2013.05.009. **Q1; H index: 45; Impact Factor: 2.621**

Section 2: The need for fine-scale distribution of fishing effort to inform on an ecosystem-based management approaches: exploring three data sources in Mediterranean trawling grounds.

Demestre, M., **Muntadas, A**., de Juan, S., Mitilineou, C., Sartor, P., Mas, J., Kavadas, S., Martín, J. The need for fine-scale assessment of trawl fishing effort to inform ecosystem approach to fisheries: exploring three data sources in Mediterranean trawling grounds. Submitted March 2015 to *Marine Policy*; under revision. **Q1; H index: 45; Impact Factor: 2.621**

CHAPTER 2:

Section 1: Integrating the provision of ecosystem services and trawl fisheries for the management of the marine environment.

Muntadas, A., de Juan, S., Demestre, M. (2015). Integrating the provision of ecosystem services and trawl fisheries for the management of the marine environment. *Science of the total Environment* 506-507: 594–603. DOI: 101016/j.scitotenv.2014.11.042. **Q1; H index: 133; Impact Factor: 3.163**

Section 2: Assessing functional redundancy in benthic community response to trawling.

Muntadas, A., de Juan, S., Demestre, M. Assessing functional redundancy in chronically trawled benthic communities. Submitted

June 2015 to *Ecological Indicators*; under revision. **Q1; H index: 54; Impact Factor: 3.23**

CHAPTER 3:

Trawling disturbance on benthic ecosystems and consequences on commercial species: a northwestern Mediterranean case study.

Muntadas, A., Demestre, M., de Juan, S., Frid, C.L.J. (2014). Trawling disturbance on benthic ecosystems and consequences on comercial species: a northwestern Mediterranean case study. *Scientia Marina*, 78(S1), 53-65. DOI: 10.3989/scimar.04024.19. **Q2; H index: 45; Impact Factor: 1.247**

CHAPTER 4:

A knowledge platform to assess the effects of trawling on benthic communities.

Muntadas, A., Lample, M., Demestre, M., Ballé-Béganton, J., de Juan, S. and Bailly, D. A knowledge platform to assess the effects of trawling on benthic communities. Submitted in March 2015 to *Estuarine, Coastal and Shelf Science*; under revision. **Q1; H index: 79; Impact Factor: 2.253**

CHAPTER 5:

Integrated thinking: a multidisciplinary approach for an adaptive management of trawling fisheries in the Mediterranean.

Muntadas, A., Demestre, M., de Juan. Integrated thinking: a multidisciplinary approach for an adaptive environmental management of trawling fisheries. Submitted in September to *Environmental Science and Policy*. **Q1; H index: 60; Impact Factor: 3.018**

The Ph.D. candidate Alba Muntadas Olive has actively participated in every work included in this project, and none of these research papers will be presented in other Ph.D. projects. She has leaded five of the studies under the supervision of the two advisors. Therefore, Alba Muntadas has demonstrated full capacity to develop independent and high quality research.

Alba Muntadas involvement in each work is detailed bellow:

- Chapter 1, section 1: she contributed to data analysis, leading VMS study, and writing of the manuscript.
- Chapter 1, section 2: she contributed to data analysis, , and writing of the manuscript.
- Chapter 2, section 1: she participated in the laboratory analysis (sorting and identification of samples), conceived and designed the study, analysed the data, and leaded the writing of the paper.
- Chapter 2, section 2: she performed laboratory analysis (sorting and identification of samples), conceived and designed the study, analysed the data, and leaded the writing of the paper.

- Chapter 3: she conceived and designed the study, analysed the data, and leaded the writing of the paper.
- Chapter 4: she conceived and actively participated in the model design, in the data analysis, and leaded the writing of the paper.
- Chapter 5: she conceived and designed the study and leaded the writing of the paper.

Barcelona, 27 of July, 2015

Montserrat Demestre Alted
Marine Science Institute
CSIC

Silvia de Juan Mohan
Center for Marine Conservation
Pontificia Universidad Católica de Chile

II

CHAPTER 1:

Defining trawl fishing effort: study of fleet dynamics and trawling effort estimation at a benthic scale.

The first step to understand the effects of trawling on benthic ecosystems is to gather a good knowledge of fleet behavior as well as to obtain a precise effort estimation at the adequate scale. The former will provide information on the pressure ‘s spatial distribution and extend and the second will produce a representative value on the pressure’s intensity. This section explores different methods to meet these objectives and discuss its applicability in the EAF context.

Section 1:
Performance of a NW Mediterranean trawl fleet: integration of landings and VMS data to promote the implementation of Ecosystem- Based Fisheries Management.

Published in *Marine Policy* 43, 112–121
DOI:10.1016/j.marpol.2013.05.009
See the original publication in Appendices

Section 2:
The need for fine-scale distribution of fishing effort to inform on an ecosystem- based management approaches: exploring three data sources in Mediterranean trawling grounds.

Under revision in *Marine Policy*.

Comportament d'una flota d'arrossegament del Mediterrani nord-occidental: com la integració de dades de captures i de VMS pot contribuir a la implementació d'una gestió basada en l'ecosistema

RESUM:

La directiva marc europea sobre l'estratègia marina empeny els països membres a gestionar tots els impactes de les activitats humanes en el medi marí des d'una perspectiva ecosistèmica per tal d'aconseguir un bon estat dels ecosistemes l'any 2020. La pesca d'arrossegament és l'activitat humana que presenta un impacte més gran sobre els fons marins, però la seva gestió al Mediterrani encara està lluny de la gestió basada en l'ecosistema.

En el context d'una gestió de la pesca basada en l'ecosistema, aquest estudi explora el potencial de la relació entre el rendiment diari per vaixell d'arrossegament (o bou) estimat en captures i ingressos diaris per espècie, i la posició del mateix vaixell, coneguda a través del sistema de localització de vaixells per satèl·lit (VMS en anglès) com a eina per a la gestió de la flota. Aquesta relació permetrà saber on i quan es concentra la flota per a pescar una determinada espècie, la qual cosa pot ajudar a determinar la idoneïtat d'implantació de vedes temporals o de la limitació d'esforç pesquer en una zona determinada. Aquesta aproximació és possible a l'àrea d'estudi (la costa nord de Catalunya) perquè els vaixells retornen cada dia a port de manera que les seves captures i ingressos són registrats diàriament en la subhasta

de la confraria del port de venda. A més a més, des de 2012 tots els vaixells de més de 12m d'eslora han de portar obligatòriament un dispositiu de localització per satèl·lit.

En aquest estudi, a partir de les dades mencionades més amunt (VMS i captures i ingressos diàris), a la flota de Roses es van poder observar quatre grups de vaixells amb diferents estratègies de pesca:

1.- Bous costaners: vaixells associats a la plataforma continental que pesquen entre 50 i 100m de profunditat i que tenen com a objectiu principals el lluç (*Merluccius merluccius*), els raps (*Lophius spp.*), el moll de fang (*Mullus barbatus*), el pop blanc (*Eledone cirrosa*) i el calamar (*Loligo vulgaris*).

2.- Bous amb l'activitat centrada en els canyons submarins: aquests vaixells pesquen en les aigües profundes dels canyons (500-900m) i tenen com a principal objectiu la gamba vermella (*Aristeus antennatus*).

3.- Bous amb activitat al voltant dels 400m de profunditat: l'objectiu principal d'aquests vaixells és l'escamarlà (*Nephrops norvegicus*).

4.- Bous generalistes: aquests vaixells no presenten un calador preferencial i varien la seva estratègia al llarg de l'any segons les espècies objectiu.

Aquest estudi demostra clarament que el patró d'activitat dels bous segueix la distribució espacial de les espècies objectiu. La distribució d'aquestes espècies, que pot variar al llarg del seu cicle vital, també està determinada pel tipus de substrat i/o per la presència d'un hàbitat determinat (entre d'altres característiques ambientals). A més a més, els pics de captures o ingressos coincideixen en el temps i en l'espai amb l'època i àrea de reclutament de l'espècie objectiu (per exemple el moll de fang). Per tant, el coneixement precís de la dinàmica de la flota, condicionada pel cicle vital de les espècies objectiu, juntament amb el coneixement de les estratègies de pesca de cada vaixell, permet saber el calador i per tan l'hàbitat a on s'exerceix la pressió pesquera. Aquest coneixement pot ser utilitzat per a la gestió d'aquestes pesqueries en el context de l'aproximació ecosistèmica, per exemple limitant l'esforç en les zones que presentin hàbitats més vulnerables i que a l'hora siguin més freqüentats.

Performance of a NW Mediterranean trawl fleet: integration of landings and VMS data to promote the implementation of Ecosystem-Based Fisheries Management

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ABSTRACT

The European Union has established a framework to achieve or maintain good environmental status in the marine environment by 2020. The Marine Strategy Framework Directive requires the application of the ecosystem approach to the management of human activities, covering all sectors having an impact on the marine environment. However, fisheries in the Mediterranean are far from a systematic implementation of an ecosystem-based fisheries management (EBFM). Aiming to address this issue, this study explores the potential of the relationship between daily yield by vessel (landings and income by species) and vessel position (known via vessel monitoring system) as a tool for fleet management. This approach is possible due to the current dynamics of Mediterranean fleets, with vessels returning daily to the harbor where landings are registered as weight and income by vessel. Moreover, vessels of >15m total length have been compulsory monitored by VMS since 2005. A bottom trawl fleet that operates in the northwestern Mediterranean was chosen to

develop this approach. Different groups of trawlers were identified, which could be linked to the strategies displayed by the fishermen that were mainly driven by the target species dynamics. Accurate knowledge of the fishing targets driving the fleet dynamics and of the fishing strategies at the vessel level (i.e. fishing ground- habitat where the fishing pressure is exerted and corresponding landings) are shown to be a feasible tool for fleet management.

Keywords: Fishing fleet management, Mediterranean fisheries, Ecosystem approach, Fishing strategies, Target species dynamics, Bottom trawl.

1. Introduction

The Reykjavık Declaration, adopted at the 2001 Reykjavık Conference on Responsible Fisheries in the Marine Ecosystems, included the commitment "to advance the scientific basis for developing and implementing management strategies that incorporate ecosystem considerations and which will ensure sustainable yields while conserving stocks and maintaining the integrity of ecosystems and habitats on which they depend". One of its main tasks is to translate the generic and conceptual Ecosystem-Based Fisheries Management (EBFM) framework (i.e. extension of the conventional principles for sustainable fisheries development to cover the ecosystem as a whole), into an operational framework at regional, national, or local scales (e.g. ecosystem or fishery levels (FAO 2003, FAO 2002)). To date, practical application of this approach by competent regional organizations for fisheries and for the marine and coastal environment is still in early development stages. It is worth mentioning the pioneering initiatives undertaken in Australia (e.g. (Day 2008, McCook et al. 2010)).

In consonance with the EBFM conceptual framework, the European Union (EU) has established a framework to achieve or maintain good environmental status (GES) in the marine environment by 2020. To this aim, criteria and methodological standards to identify or achieve a GES of marine waters have been defined. These include, among others, descriptors at the species, habitat and ecosystem levels, as well as descriptors related to the fishing activity (EC Directive 2008/56, Marine Strategy Framework Directive (MSFD, EC 2010/477). The activity of the EU fishing fleets is governed by numerous regulations based on the Common Fisheries Policy, which at present is under revision. In any case, environmental concerns must be integrated in the fisheries management, in line with the MSFD. Regarding the implementation of EBFM in the Mediterranean, the General Fisheries Commission for the Mediterranean (GFCM), composed of EU and non-EU signatory countries, launched in 2012 a revision process aimed at modernizing its legal and institutional framework. This revision includes among its objectives the promotion of the ecosystem-based approach for the conservation of the marine environment and the sustainable use of marine living resources (GFCM 2012/8).

Bottom trawling is the fishing activity with the highest impact on marine ecosystems (Thrush & Dayton 2002), hence, in order to achieve a GES, it is important to properly regulate this activity. The management of bottom trawling in the EU Mediterranean waters is carried out by fishing effort control, through i) control of the fleet activity and technical measures; and ii) closures, which refer to permanent spatial closures (fishing forbidden at < 50 m and > 1000 m depth) and temporal closures. The control of the activity includes a daily limit of hours at sea. Technical measures include minimum landing size for the main target species, square- meshed net of 40 mm at the cod- end, and control of the fishing capacity (Council Regulation (EC) No 1967/2006, Recommendation GFCM/29/2005/1). Temporal closures are not homogenously implemented and, when applied, its duration differ among areas, from 1-2 months in some Spanish Mediterranean areas, 1.5 months in Italian waters, to 4 months in Greece (Demestre et al. 2008). It is worth mentioning that Total Allowable Catch (TAC) only applies to bluefin tuna. In areas other than the Mediterranean where EU fleets operate, bottom trawling is based on the exploitation of a single main fishing resource, and the landed catch is the result of fishing during one trip, which usually lasts several days or even weeks. Otherwise, despite bottom trawling in the Mediterranean can also be based on the exploitation of a single resource, it typically targets several species, their relative importance in the landings changing over the year. In general, trawling is carried out near the port base, five days a week, and the catch is freshly commercialised, daily, upon the arrival of trawlers to the port base.

MSFD proposes qualitative descriptors (e.g. biological diversity, sea floor integrity or exploitation of populations within safe biological limits) of the environmental status of marine ecosystems. Some of these descriptors have been monitored for a long time and are currently impacted by fishing (Rice et al. 2012). However, fisheries in the Mediterranean are far from a systematic implementation of an EBFM and many sensitive ecosystems (including essential fish habitats) are threatened by fishing activities (de Juan et al. 2010). The complex geopolitical situation of the Mediterranean basin, i.e. many costal states and a large proportion of international waters, implies coordinated management actions that are necessary to ensure the protection of ecosystems, but these are difficult to implement (de Juan et al. 2011). Nevertheless, despite these difficulties, the characteristics of the fishing fleet control and distribution in the Mediterranean (i.e. clear delimitations for operation of different port-based fleets) is a potential bridge towards the implementation of EBFM.

It is generally accepted that EBFM must incorporate a spatial dimension to ensure the sustainability of resources and ecosystem integrity (Thrush et al. 2010, Stelzenmüller et al. 2013). In the Mediterranean this could be partly achieved by establishing maximum effort levels on a spatial extent, including temporal or permanent closures to protect Sensitive Habitats. Moreover, technical measures could be adopted to minimise impact on ecosystems (Jennings et al. 2012). But in order to achieve this, we first need to know the spatial and temporal dynamics of the fishing fleet that will help identifying the priority areas/seasons for management. In this regard, the Vessel Monitoring System (VMS) is a promising tool for the fisheries management. Since 2005 fishing vessels exceeding 15 m of overall length must be equipped with VMS (EC 2003/2244).The potential of

VMS for fleet management has been explored in North Atlantic fisheries, by combining logbook data with VMS signals to investigate the spatial distribution of catches and effort. However, the fishing fleet dynamics in this area imply landings corresponding to several fishing days and therefore the relationship between catches and vessel position is not straightforward (e.g. Jennings & Lee 2011, Gerritsen & Lordan 2010, Witt & Godley 2007, ICES 2011). In the Mediterranean, the daily return of trawlers to port facilitates a direct link between vessel position and catches, however, VMS data have been barely analysed aiming to evaluate its potential as a tool for the management of the fishing activity (e.g. Russo et al. 2011).

Aiming to advance in the implementation of EBFM in the Mediterranean and integrate current fishing practices with management strategies, the relationship between daily yield by vessel (landings and income) and vessel position has been explored to assess its potential as a tool for fleet management. A bottom trawl fleet that operates in the northwestern Mediterranean was chosen to develop this approach.

2. Material and Methods

2.1. Study area

The northwestern Mediterranean study area, off Cape of Creus and Bay of Roses, corresponds to the fishing grounds where the trawl fleet based on the fishing port of Roses operates (around 3500 km^2; Fig. 1). The Cape of Creus continental shelf is characterised by an abrupt morphology and alternates smooth areas, where sandy and muddy sediments prevail, with rough rocky areas, mainly along the coast and in the outer shelf for depths between 95 and 130 m. Moreover, the Cape of Creus canyon breaches the shelf at a depth of 110 m (Lo Iacono et al. 2012). The shelf- slope limit, in front of the Bay of Roses, is located at 170 m depth (Ercilla et al. 1995). The fishing grounds encompass the rather narrow continental shelf, the upper slope and the submarine canyon. This heterogeneous area was chosen with a view to unveil potential different fishing strategies. The study area includes the coastal protected areas of Natural Park of Cape of Creus and Medes Islands Marine Reserve (Fig. 1).

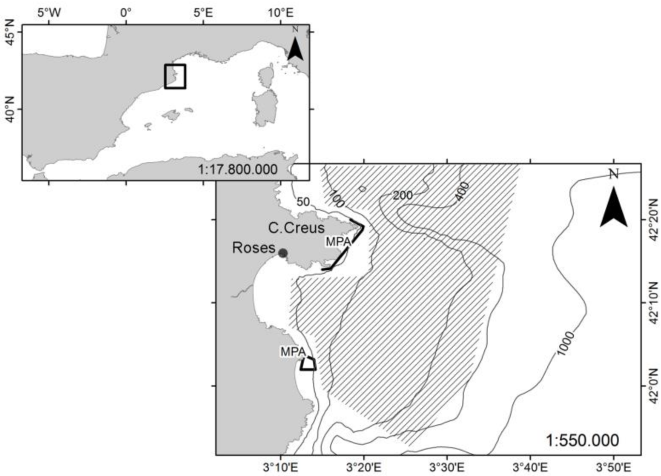

Figure 1. Study area in the northwestern Mediterranean. The striped area encompasses the fishing grounds where the bottom trawl fleet from Roses operates. The limits of the MPAs Cape of Creus (in the north) and Medes Islands (in the south) and the 50, 100, 200, 400 and 1000 m isobaths are also shown.

2.2. Data on landings, income and fishing fleet

The trawl fleet from Roses consisted of 22 vessels, most of them of an overall length > 20 m, when the study was carried out in 2009 (Table 1). Trawlers return daily from Monday to Friday to Roses harbour, where the catch is sold in the local auction. A maximum of 12 hours per day at sea is permitted. Bottom trawling license is to be used all year round, that is, fishermen cannot practice any other type of fishing.

Table 1. Gross tonnage (GT) and overall length (LOA in m) of the bottom trawlers based in Roses in 2009.

vessel	GT	LOA
1	105.6	28.5
2	24.2	14.3
3	54.4	22.4
4	34.6	17.5
5	60.5	22.9
6	21.5	15.5
7	65.2	22.9
8	93.7	25.1
9	56.7	21.6
10	61.1	23.3
11	58.9	20.4
12	65.2	23.1
13	87.4	24.9
14	61.6	24.9
15	178.5	26.5
16	74.4	24.5
17	100.1	25.0
18	77.1	22.5
19	96.7	25.4
20	100.0	27.4
21	97.1	26.3
22	96.5	27.1

Data on landings, income and activity of the trawl fleet were obtained from the daily sales slips obtained from the sale at the auction that takes place upon the arrival of the vessels at port (data source: fishing statistics elaborated by the Fisheries Department of the Generalitat de Catalunya). Data were available on daily landings by species, in weight and in income, for each fishing vessel during 2009. No contemporary data on discards were available. Data on the trawl fleet (gross tonnage (GT) and overall length in m (LOA)) were obtained from the fishing vessels census of the Spanish Ministry of Agriculture, Food and Environment.

2.3. Fleet structure based on landings and income

Cluster analysis and factor analysis approaches were used to obtain homogeneous groups of vessels at the annual scale based on the 2009 daily landings by vessel (catch and income by species). No prior assumptions of the fishery were made, avoiding the inherent subjectivity of qualitative analysis. Firstly, the specific composition of the catch (weight and income) was transformed into a landings profile (relative species composition) and data were log-transformed. Secondly, cluster analysis was applied to the log-transformed landings profile matrices using Ward's minimum variance clustering algorithm and the Euclidean distance. Species with annual landings <100 kg and annual income from the sales at the auction <600 euros were excluded because of their low relevance in the overall landings and income. The number of species retained for the analyses was 73. Thirdly, factor analysis was applied to further explore the structure of the relationship among vessels. Communalities were estimated as multiple R-square and only factors with eigenvalues > 1 were retained. The varimax rotation was applied to the factor solution. This study selected factor loadings over 0.7 to evaluate the relationship between the selected factors and the vessels. Analyses were done with the statistical package Statistica6.

2.4. Spatial distribution of fishing effort

Finally, daily position records of the vessels obtained from the VMS (Vessel Monitoring System) were available (data source: General Secretariat for Fisheries of the Spanish Ministry of Agriculture, Food and Environment) which allowed linking fishing catches with the actual corresponding fishing position. Unprocessed VMS data does not indicate whether a vessel is fishing, so a speed filter was applied to raw data. Only records from vessels with fishing activity (i.e. speed up to 4 knots (Lleonart 1990)) were taken into account and these positions were plotted using ArcGis 10.0 package to represent the spatial distribution of the fishing activity. Each plot represents the fishing positions of one vessel in 2009.

2.5 Identification of outliers

Data on daily landings by species per vessel were explored aiming to detect outlying observations, that is, values that deviate markedly from others. Outliers can indicate either an error or a very high daily landing or income for a given species in a certain time of the year. These high daily landings or income were our target. An example is presented to illustrate how daily landings, in combination with vessel position, are a potential tool for the monitoring of the fleet. To this aim monthly box-plots for the daily landings and income of one trawler (vessel no.7 in Table 1) were done and the identified outliers were also shown.

3. Results

3.1. Annual landings and income

In 2009, the trawl fleet from Roses landed around 1560 t of fish in a total of 4945 fishing days, which generated around 8.2 million euros from the sale at the auction. The activity of the fleet was highest in summer (July and August), and in January and March, with around 450 fishing days per month, and lowest in February (345 fishing days).

A total of 105 species were commercialised. Nevertheless, 94% of the landings in weight and in income was obtained with 22 and 21 species respectively (weight and income of each of these species > 0.5% of the

Table 2. Fishing port of Roses 2009: bottom trawl annual landings, expressed in tones (a) and in thousands of euros (b). Species with landings and income 40.5% of the total are detailed.

(a) landings (tones)		%	(b) income (thousands euros)		%
Merluccius merluccius	324.5	20.8	*Aristeus antennatus*	2326.2	28.3
Trachurus spp	240.9	15.5	*Merluccius merluccius*	1477.8	18.0
Micromesistius poutassou	177.5	11.4	*Nephrops norvegicus*	868.0	10.6
Eledone cirrhosa	170.4	10.9	*Lophius spp.*	631.1	7.7
Aristeus antennatus	71.5	4.6	*Eledone cirrhosa*	400.8	4.9
Lophius spp.	66.2	4.2	*Loligo vulgaris*	335.2	4.1
Nephrops norvegicus	54.3	3.5	*Micromesistius poutassou*	321.3	3.9
Mullus barbatus	43.5	2.8	*Mullus barbatus*	264.0	3.2
Illex coindetii	42.8	2.7	*Lepidorhombus boscii*	193.5	2.4
Phycis blennoides	40.3	2.6	*Trachurus spp*	181.3	2.2
Trisopterus minutus	35.0	2.2	*Phycis blennoides*	111.0	1.4
Lepidorhombus boscii	30.0	1.9	*Zeus faber*	90.2	1.1
Pagellus acarne	27.9	1.8	*Illex coindetii*	83.3	1.0
Sardina pilchardus	23.3	1.5	*Trisopterus minutus*	67.7	0.8
Loligo vulgaris	21.7	1.4	*Scomber scombrus*	59.2	0.7
Octopus vulgaris	16.7	1.1	*Sepiidae. Sepiolidae*	55.8	0.7
Triglidae	16.5	1.1	*Octopus vulgaris*	54.9	0.7
Scomber scombrus	14.6	0.9	*Pagellus erythrinus*	51.9	0.6
Engraulis encrasicolus	13.7	0.9	*Stichopus regalis*	51.2	0.6
Trigla lyra	13.6	0.9	*Trigla lucerna*	46.9	0.6
Pagellus erythrinus	12.8	0.8	*Pagellus acarne*	46.2	0.6
Conger conger	9.1	0.6	other	495.2	6.0
other	92.2	5.9			
Total	1559.0	100	Total	8212.5	100

annual total). The specific composition of the landings expressed by weight or by income provides different patterns of the fleet yield

(Table 2). European hake (*Merluccius merluccius*) was the main species in terms of landings, 324.5 t, followed by horse mackerel (*Trachurus* spp.), blue whiting (*Micromesistius poutassou)* and curled-horned octopus (*Eledone cirrhosa)*. Otherwise, the landings ranked by income clearly identified the main target species. Blue and red shrimp (*Aristeus antennatus*) was by far the species that generated the highest income (2.3 million euros). The other species that also generated high income were hake, Norway lobster (*Nephrops norvegicus*), monkfish (this category includes *Lophius budegassa* and *Lophius piscatorius*), curled-horned octopus, squid (*Loligo vulgaris*) and red mullet (*Mullus barbatus)*. Horse mackerel and blue whiting did not generate high income and were clearly behind other species highly appreciated, that despite their lower catches were ahead in the ranking based on income. Hence, in this case these two species should be considered as by- catch species. It is worth mentioning the landings (0.7t) of the royal cucumber (*Stichopus regalis*), which corresponds to its five longitudinal muscular bands that are sold as a culinary delicacy, and that expressed in total fresh weight of the organism would amount to around 7t (Ramón et al. 2010). This is the species with the highest price at the auction (> 70 euros per kg), more than twice the mean price for blue and red shrimp.

3.2. Fleet structure based on landings and income

Cluster analysis from both the landings and income profiles, by species per vessel, identified three homogeneous groups within the fleet (Fig. 2). The groups from the two ordination approaches were quite similar, although the ordination within group C was slightly different, when expressed in terms of landings or income. The first dichotomy separated the same group of seven vessels (Fig.2; vessels no. 2,3,5,7, 8, 12 and 19; codes as in Table 1; group A). Within the second dichotomy, a small group of four vessels (no. 10, 16, 20, and 21; group B) was differentiated from the remaining vessels (vessels 1, 4, 6, 9, 11, 13, 14, 15, 17, 18 and 22; group C).

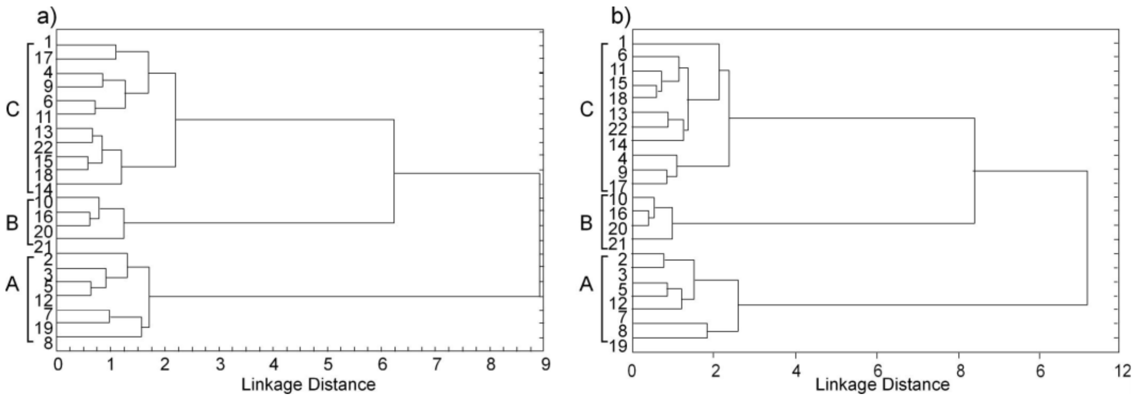

Figure. 2. Cluster analysis identified three groups of vessels (A, B and C) from the bottom trawl fleet from Roses in 2009. Analyses were applied to the log-transformed landings profiles matrices using Ward's minimum variance clustering algorithm and the Euclidean distance; clustering based on landings profiles in weight (a), and income (b). Vessel number as in Table 1.

The landings profiles by vessel, arranged according to the cluster analysis, ease the identification of the main species characterizing each vessel group. When expressed as income, landings profiles evidenced the main target species, those driving the fishing activity within each group of vessels (Table 3). Hake and horse mackerel, the species with highest landings (Table 2), were well represented in the landings by weight of the whole fleet (Table 3). The importance of

Table 3. Bottom trawl fleet from the fishing port of Roses: landings profiles (relative species composition) by vessel in 2009, expressed by weight (a) and by income from the sales at the auction (b). Profiles arranged according to the cluster analysis results shown in Fig. 2; vessel number as Table 1.

4	9	6	11	13	22	15	18	14	10	16	20	21	
17.4	14.1	13.3	17.5	25.1	29.3	22.4	30.2	20.9	21.7	18.5	16.1	15.8	1
3.4	6.1	10.9	11.0	13.7	14.9	11.5	8.8	21.2	8.8	12.2	13.9	2.3	1
13.3	21.3	9.1	20.8	19.1	11.7	16.1	13.6	13.4	1.1	2.5	4.1	1.4	
10.2	5.9	10.9	7.7	10.1	10.6	9.8	11.5	6.6	12.2	10.1	11.3	5.6	3
1.8	2.6	4.4	3.4	3.1	6.2	2.3	1.7	0.8	29.9	33.3	24.2	56.7	
6.7	4.8	9.1	6.1	3.2	3.4	5.0	4.7	3.8	2.5	2.9	4.5	3.3	
14.5	15.1	5.0	5.5	2.1	2.4	6.2	4.7	2.4	0.0	0.2	0.2	0.1	
1.9	3.2	1.9	1.5	1.1	0.7	2.1	1.2	1.3	4.3	2.4	2.1	0.6	
2.2	3.2	3.7	2.9	2.6	3.0	3.8	3.7	3.6	2.2	1.9	1.5	0.8	
7.1	7.1	7.1	6.4	2.4	2.9	4.1	3.5	2.4	3.6	3.1	2.9	3.0	
1.6	0.6	2.3	2.4	2.4	2.6	2.7	2.8	2.8	0.9	2.0	2.6	1.1	
3.9	2.7	3.0	2.4	2.0	1.5	2.8	2.1	2.2	0.3	0.8	0.9	0.3	
0.4	1.1	1.7	0.8	1.4	0.1	0.3	0.1	2.2	0.8	0.3	1.2	0.1	
0.8	0.3	0.1	0.6	0.7	0.8	0.8	2.0	2.3	0.8	0.8	0.6	0.4	
2.9	0.9	0.8	0.5	0.8	0.5	0.4	0.8	0.6	1.5	0.5	0.7	0.2	
2.2	0.8	1.4	0.5	0.8	0.4	0.3	0.5	0.8	0.7	0.4	1.0	0.3	
0.4	0.7	1.1	0.7	1.1	0.5	1.7	0.6	1.5	0.5	0.7	1.0	0.1	
0.2	0.2	0.5	0.5	1.1	2.0	0.9	1.0	1.0	1.2	1.3	0.8	0.7	
0.0	0.1	0.0	0.3	1.3	0.5	0.3	0.8	0.6	0.5	0.2	0.4	0.5	
1.6	1.2	4.4	2.7	1.1	0.7	0.8	0.8	0.9	0.1	0.5	0.6	0.0	
0.5	1.1	0.2	0.2	0.3	0.1	0.2	0.3	0.2	0.7	0.3	0.4	0.0	
0.5	0.3	1.4	1.0	0.4	0.9	0.7	0.1	1.4	0.5	0.5	0.8	0.7	
6.3	6.7	7.8	4.6	4.2	4.5	4.7	4.4	6.9	5.2	4.8	8.5	5.8	

Table
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(b) profiles by income

vessel number	1	6	11	15	18	13	22	14	4	9	17	10	16	20	21	
Aristeus antennatus	0.8	25.8	22.3	14.7	12.9	22.5	41.6	5.5	9.8	12.0	13.5	79.6	82.1	75.3	92.3	0
Merluccius merluccius	15.0	10.0	15.6	21.8	29.8	25.2	23.5	27.8	13.0	10.4	11.7	7.1	6.1	7.3	3.4	19
Nephrops norvegicus	21.8	15.2	16.8	19.0	15.8	6.2	5.4	11.7	35.7	42.1	40.3	0.0	0.2	0.2	0.0	0
Lophius spp.	15.7	13.3	10.0	9.8	9.0	7.5	5.1	10.4	8.2	6.5	6.9	1.9	2.5	3.5	1.3	13
Eledone cirrhosa	2.2	2.9	2.8	3.6	4.3	4.7	3.3	3.5	3.4	1.9	0.9	2.9	1.4	2.5	0.3	30
Loligo vulgaris	1.0	1.9	1.4	1.4	2.3	2.5	1.2	2.9	7.5	1.9	0.7	1.8	0.6	0.9	0.2	8
Micromesistius poutassou	14.5	2.6	8.5	5.2	5.0	8.4	3.4	7.3	4.1	6.3	10.4	0.2	0.4	0.8	0.1	0
Mullus barbatus	2.8	2.7	2.0	2.4	1.6	1.7	0.6	2.0	2.0	2.4	0.8	1.7	0.9	1.3	0.1	6
Lepidorhombus boscii	4.0	3.5	3.2	4.5	2.9	2.9	1.5	3.9	5.0	3.2	3.6	0.1	0.4	0.5	0.1	3
Trachurus spp	2.3	1.4	1.5	1.8	1.4	2.6	2.4	4.9	0.4	0.7	2.5	0.6	0.8	1.0	0.1	2
Phycis blennoides	1.4	3.0	3.3	2.5	2.1	1.6	1.5	2.3	3.3	3.0	2.1	0.7	0.7	0.8	0.4	0
Zeus faber	4.8	1.6	1.9	1.7	0.8	1.9	0.7	0.7	0.1	0.1	0.7	0.1	0.2	0.6	0.1	0
Illex coindetii	1.5	1.1	1.1	1.6	1.4	1.1	1.2	1.9	0.8	1.1	0.9	0.3	0.3	0.3	0.1	1
Trisopterus minutus	1.5	0.7	1.0	1.0	1.2	1.2	0.8	1.5	0.5	0.2	0.2	0.1	0.3	0.5	0.1	1
Octopus vulgaris	0.3	0.9	0.4	0.3	0.4	0.6	0.3	0.6	1.0	0.3	0.1	0.2	0.1	0.2	0.1	1
Sepiidae, Sepiolidae	0.7	1.1	0.7	0.4	0.5	0.7	0.4	0.6	0.0	0.0	0.2	0.3	0.4	0.4	0.0	1
Scomber scombrus	0.4	0.3	0.4	1.0	0.9	1.0	1.2	1.2	0.1	0.2	0.1	0.4	0.5	0.3	0.1	0
Pagellus erythrinus	0.1	0.1	0.2	0.4	0.7	0.4	0.2	0.4	0.2	0.4	0.0	0.1	0.2	0.1	0.0	0
Stichopus regalis	4.1	3.5	1.2	0.3	1.2	0.5	0.0	0.1	0.0	0.0	0.0	0.0	0.2	0.2	0.0	0
Pagellus acarne	0.0	0.9	0.6	0.2	0.1	1.1	0.0	0.5	0.1	0.2	0.1	0.1	0.1	0.3	0.0	0
other	5.0	7.3	5.2	6.3	5.7	5.8	5.3	10.2	4.5	6.8	4.2	1.9	1.6	3.1	1.1	9

hake differed among groups representing high income. T landings was high in group C, nil in group A, and did not r Curled-horned octopus repre vessels from groups A, B ar income only for some trawlers of group A. Within this group, other species that provided high income were monkfish, squid and red mullet. Blue and red shrimp was the most important species by weight in group B, its importance small in group C and nil in group A. By far, the highest income corresponded to blue and red shrimp and group B vessels. Within group C, a small group of three vessels (no. 4, 9, 17) had Norway lobster as the main species in terms of income (see also

Fig. 2b, these vessels appear as a sub-group of group C); another sub-group of five vessels (no. 13, 14, 15, 18, 22; Table 3) had hake as the main target species in terms of landings and income. Relatively low blue and red shrimp landings represented also high income for some trawlers of group C.

Factor analysis results based on the landings profiles by weight and by income were similar, as the same main groups were retained. The factors loadings for each vessel, which indicate the vessels with the highest correlation with each of the three retained factors, are given in Table 4. The first three factors explained > 90% of the total variance. With the application of the varimax rotation to the factor analysis solutions, variance was redistributed among the three factors (40%, 32% and 21% for Factors 1, 2, and 3 for the analysis based on the landings; and 39%, 29% and 25% in the case of income). Based on the factor loadings > 0.7, the vessels linked to Factors 1, 2 and 3 were identified. For both landings profiles by weight and income, factors loadings > 0.7 were for Group of vessels C and Factor 1; for Group A and Factor 2; and for Group B and Factor 3, with the exception of vessel no. 8, for which factor loadings were < 0.7 for the three factors, with landings profiles as income. For the interpretation of factors, the two-factors rotated solutions are also shown for the landings by

Table 4. Factor analysis results: factor loadings for the landings (a) and income (b); vessel number as in Table 1.

	(a) landings				(b) income		
vessel	Factor1	Factor2	Factor3	vessel	Factor1	Factor2	Factor3
1	0.9	0.4	0.2	1	0.9	0.3	0.0
2	0.3	0.8	0.2	2	0.2	0.9	0.1
3	0.3	0.9	0.3	3	0.2	0.9	0.1
4	0.8	0.3	0.3	4	0.9	0.3	0.3
5	0.4	0.9	0.3	5	0.3	0.9	0.1
6	0.8	0.3	0.4	6	0.8	0.3	0.4
7	0.2	0.9	0.2	7	0.2	0.9	0.2
8	0.4	0.7	0.4	8	0.4	0.6	0.6
9	0.9	0.3	0.3	9	0.9	0.2	0.3
10	0.3	0.4	0.8	10	0.3	0.3	0.9
11	0.9	0.3	0.4	11	0.9	0.2	0.4
12	0.4	0.9	0.2	12	0.2	0.9	0.1
13	0.7	0.5	0.4	13	0.7	0.4	0.5
14	0.7	0.6	0.3	14	0.8	0.5	0.3
15	0.8	0.4	0.4	15	0.8	0.3	0.4
16	0.4	0.4	0.8	16	0.4	0.2	0.9
17	0.9	0.2	0.3	17	0.9	0.1	0.3
18	0.8	0.4	0.4	18	0.8	0.4	0.4
19	0.4	0.8	0.2	19	0.5	0.7	0.2
20	0.5	0.4	0.8	20	0.4	0.2	0.9
21	0.3	0.1	0.9	21	0.3	0.0	1.0

weight (Fig. 3).

3.3 Spatial distribution of fishing effort

VMS data showed the overall dynamics of the fleet in 2009. One vessel within each one of the three identified vessels' groups (A, B and C) was selected to show their preferential fishing grounds (Fig. 4). The group of vessels linked to Factor 2 (Group A) corresponded to coastal trawlers that operate very close to the coast, at depths between 50-100 m; their activity at >100 m depth was limited (Fig.4a). The group of vessels linked to Factor 3 (Group B) corresponded to the trawlers targeting blue and red shrimp. As depicted by the fishing positions, the fishing grounds are the submarine canyons. Trawlers targeting

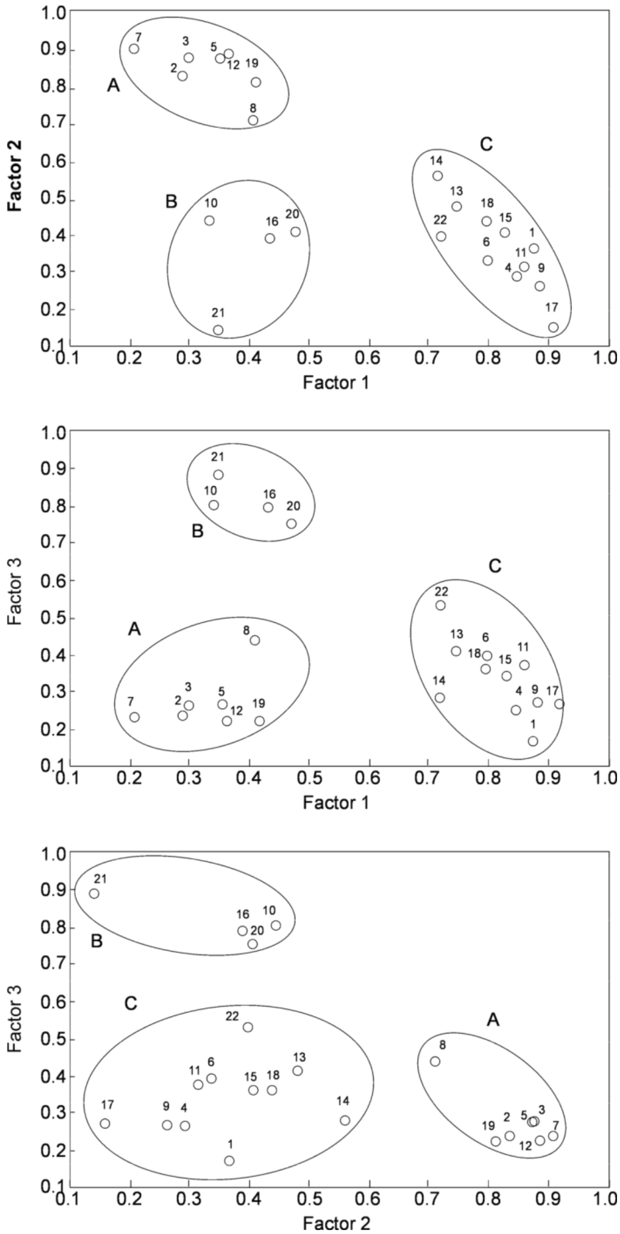

Figure 3. Factor analysis of the landings profiles in weight from the bottom trawl fleet from Roses in 2009. The plots show the position of the vessels based on the factor loadings of the first three factors. The trawlers belonging to the same vessels' group, as identified in the cluster analysis, are shown encircled; vessel number as in Table 1.

Aristeus antennatus generally perform the final haul of the day in their way back to the port, and this would correspond to those hauls between 50 and 100 m depth (Fig.4b). Finally, 11 vessels out of the 22 that made up the fleet appeared more closely linked to Factor 1 (Group C). This is a heterogeneous group (see the landings profiles, Table 3) that included trawlers with Norway lobster or hake as main target, and trawlers that additionally targeted other species as blue and red shrimp or monkfish. The preferential fishing grounds for trawlers targeting Norway lobster are located in the slope, around the 400 m isobath, between 350-450 m (Fig. 4c). Some of the vessels from

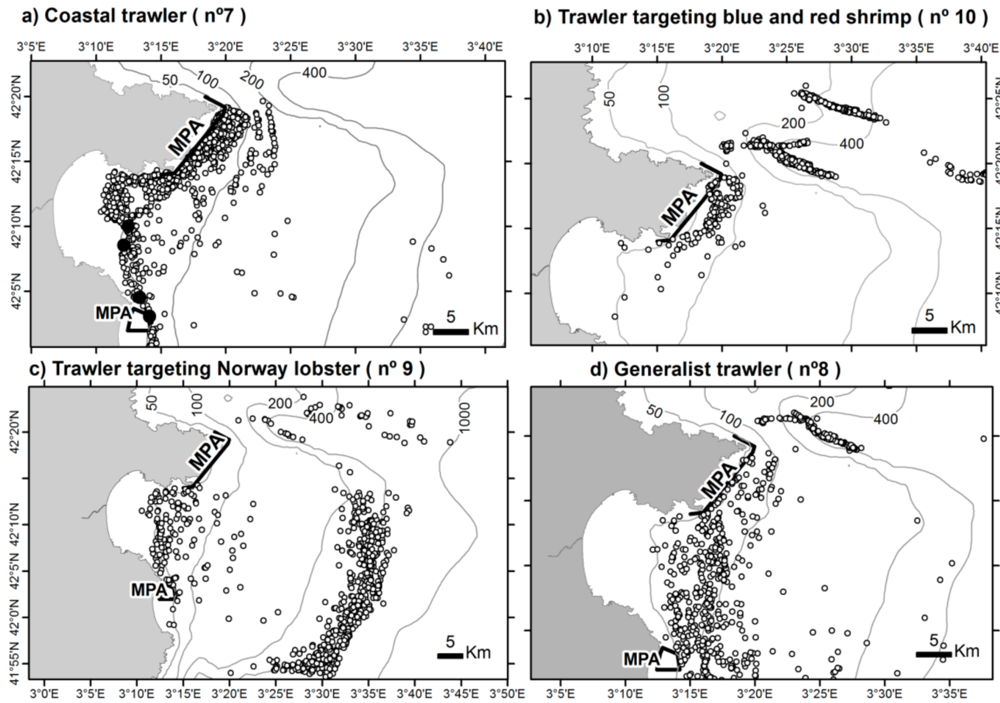

Figure 4. Fishing strategies identified in the bottom trawl fleet from Roses in 2009 through clustering and factor analysis (vessel number as in Table 1). Examples are shown of trawlers operating in the coastal zone (a; in bold, positions corresponding 17 July, explanation in the text on the identification of landings and income outliers); targeting blue and red shrimp Aristeus antennatus (b); targeting Norway lobster Nephrops norvegicus (c); and of generalist behaviour (d)

this group C, whose activity was driven by several target species, displayed a "generalist" fishing strategy by shifting between species during the year. Generalists trawlers fished in coastal grounds, over the shelf and, also, on the submarine canyons (Fig. 4d). The generalist strategy of some vessels, i.e. those that do not display a clear affinity to any of the three factors, is more evident in the factor loadings by income (Table 4b). A typical "generalist vessel" would be no. 8 (Fig. 4d), but vessels no. 13, 19 and 22 could also be considered as generalists.

3.4 Identification of outliers

As an example of the potential for the daily landings to be used as a tool for fleet management, the landings from a vessel from group A, coastal trawlers, were presented during 2009 (vessel no.7; Fig. 5). Outliers are shown in relation to the corresponding monthly average of the daily catch and income. Thus, for instance, in three days over the year (17 July, 3 August and 7 September) the daily landings of this vessel were much higher than the monthly average (> 1400 kg/day). On the 17th of July this trawler obtained its maximum daily income of the year (> 5000 euros). Since data were structured by vessel and by species, the species that generated this outstanding income could be identified and in this particular case turned out to be red mullet. Given that the position of the trawler can be identified from the VMS data, the detected outliers on the 17th of July (Fig. 5) could be linked with the position of the vessel (Fig. 4a). On the other two outlying dates for landings, the high catch was generated by *Pagellus acarne* (1400 and 1200 kg respectively on 3 August and 7 September), which did not correspond with the income peaks in August and September.

Evaluating the income outliers, the income from the 11th of August (4000 euros) and 22nd of September (3500 euros) outstand as they were not detected in the catches. These outliers were explained, in addition to red mullet, by the income of a number of species, including *Pagellus acarne, Loligo vulgaris* and *Pagellus erythrinus.*

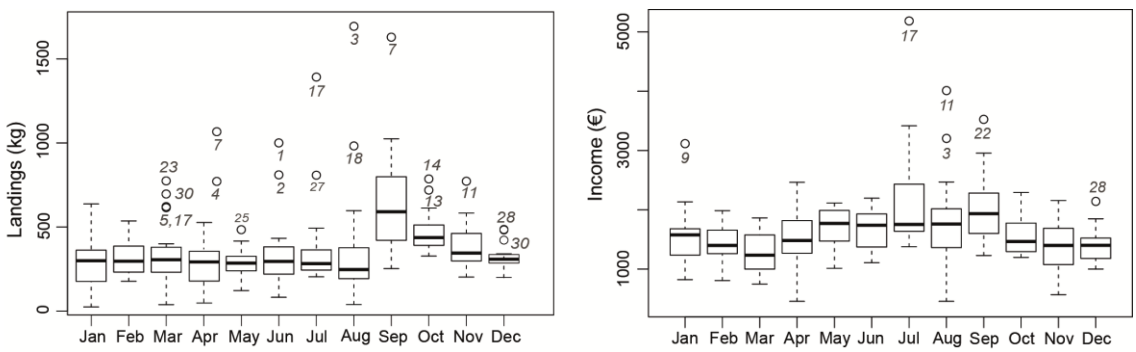

Figure 5. Example of the identification of outliers from the daily landings data by vessel: landings data by weight (kg) and income (euros) of trawler no. 7 during 2009. The boxplots include median, 25 and 75 percentiles and maximum and minimum values each month over 2009. The numbers correspond to the date of the outlier.

4. Discussion

This study identified different groups of trawlers within a bottom trawl fleet that operates in the northwestern Mediterranean, which can be linked to the strategies displayed by the fishermen. The analyses were based on the daily landings profile for each vessel, based on both catch weight and income; however, the species with the highest contribution to the annual income are assumed to drive the fishermen's decision on the choice for the fishing ground.

Results identified three groups of vessels, which can be interpreted as coastal trawlers (A), trawlers operating in the submarine canyons (group B), and trawlers fishing in the shelf break and slope (group C). The fleet structure resulting from the factor analysis and the corresponding factor loadings (Table 4 and Figs. 3 and 4) suggested that Factor1 could be linked to the fishing grounds over the deepest shelf and on the shelf break. The trawlers with highest correlation with Factor1 (group C) included those with Norway lobster as main target, that is, those that operate in the fishing grounds at further distance from the coast. The morphology of the study area determines that at short distances from the coast a wide depth range is encountered. Factor3, for which only the trawlers targeting blue and red shrimp (group B) had factor loadings > 0.7, could be interpreted as a factor depending on depth, since submarine canyons are the deepest fishing ground. Coastal trawlers (group A) showed the strongest link to Factor2, which could be associated to the shelf that in the study area is rather narrow (Fig. 1).

This interpretation is in accordance to the type of substrate in the study area, which in turn is decisive for the spatial distribution of the main target species. The submarine canyons in the Mediterranean are the habitat for blue and red shrimp (Demestre & Martín 1993, Sarda et al 1997), a highly valued commercial species. Along its life cycle, hake inhabits the shelf and the shelf break zones throughout the western Mediterranean (Massutí & Oliver 1995). Norway lobster is found in the muddy bottoms over the upper and middle continental slope (Abelló et al. 1988, Maynou et al. 1998). And coastal areas in the region harbour heterogeneous substrata (Lo Iaconio et al. 2012) that are habitat to a variety of demersal fish, represented in the catch composition. The tight relationship between the fishing strategy and the selection of fishing grounds by the concentration of target species denotes the importance of identifying Essential Fish Habitats that might be vulnerable to fishing. Indeed, Bergmann et al. (2004) suggested that the fishermen knowledge on the location and distribution of EFH is highly valuable information for management.

The selection of these habitats by the fishermen can be partly explained by the presence of three species whose income represented >10% of the annual total, and therefore are to be considered as main drivers of the fishing activity. These are blue and red shrimp, European hake and Norway lobster (Table 3). The income of the species that followed in the ranking (i.e. curled-horned octopus, squid, red mullet) may seem low at the annual scale, but this is misleading because their landings displayed a marked seasonality and most of the annual income was concentrated in a short period. The monitoring of these economically important species is essential in the context of EBFM, as policy makers need to quantify the goods and services provided by an ecosystem in order to evaluate the impact of management decisions (Beaumont et al. 2007). Importantly, the main life history traits to be taken into account in fisheries management, such as spawning areas and season, recruitment area or preferential feeding habitat, are well known for these species in the study area (e.g. Maynou et al. 1998, Recasens et al. 1998, Macpherson et al. 1981, Demestre & Fortuño 1992, Sanchez & Martín 1993, Demestre et al. 1997, Martín et al. 1999, Recasens et al 2008, Sabatés et al. 2007, Bautista-Vega 2008, Sánchez et al. 2008, Sánchez & Demestre 2010).

Fishermen develop dynamic fishing tactics and strategies as an adaptive response to changes in resource abundance, environmental conditions and market or regulatory constraints. From the way of coping with these factors, the behaviour of fishermen can be defined as specialist or generalist. The knowledge of these dynamics is essential for effective management (Smith & McKelvey 1986, Salas & Gaertner 2004). VMS data (precise trawler position) combined with data on daily landings allowed identifying the fishing strategies of each of the identified trawlers' group and, also, the fishing strategy of the individual vessel. Specialists concentrated in one area or one target species. In the study area, this typically corresponds to the fishermen targeting blue and red shrimp. But the concept of specialists would also include coastal trawlers and shelf-break-slope trawlers, with Norway lobster as main target. This is because the fishing grounds where most trawlers operate remain fairly constant over the year. Generalists would be those trawlers deploying their activity in non-clearly defined fishing grounds, which can be located from the coast to the shelf-break, and even the submarine canyons (Fig. 4d). The knowledge of the fishermen strategies as for the preferred fishing grounds and targets in a certain area or time is basic for the definition of spatial or temporal closures, aimed at the protection of a given fishing resource or habitat.

European and national policy commitments require further integration of fisheries and environmental management (Jennings et al. 2012). Therefore, data on the fleet dynamics needs to be complemented with indices of the ecosystem status (Ellis et al. 2008). Knowledge on the distribution of fishing activities should be followed by the characterization of the fished habitats (Demestre et al. 2000) and the evaluation of these habitats under the concept of GES (Hinz et al. 2013). An example of this process was presented in de Juan & Demestre (2012) that tested a Trawl Disturbance Indicator in several habitats linked to trawling grounds, including the coastal fishing grounds near the Cape of Creus that were assigned a moderate environmental status, and soft-bottoms encompassed in Medes Islands MPA, that had high status. This ecological indicator should be complemented with other indicators that overall evaluate the ecological integrity of the area, which might include, among others, target species distribution, population size (population abundance and/or biomass), level of pressure of the fishing activity, proportion of selected species at the top of the food webs (large fish, by weight). Nevertheless, these assessments should consider different scales for fishery dynamics and ecosystem conservation, as often the scales of fishing activities estimations are too large to link with ecosystem impact at the habitat scale (Thrush & Dayton 2002). In this context, the smaller-scale distribution of Mediterranean fleets is useful to identify links between fishing activity and environmental disturbance.

Fleet-based assessment is the pathway for implementation of efficient EBFM in European Seas (Gascuel et al. 2012). Fishing intensities calculated from VMS data have proven to be reliable indicators of fishing impacts for several habitats (Hinz et al. 2013, de Juan & Demestre 2012, Gascuel et al. 2012, Foley et al. 2012, Fofarty & Muraw 1998, Babcock et al. 2005). Results on the preferential fishing grounds by vessel or group of vessels have been presented here at the annual scale. Nevertheless, the areas where fishing effort and target species concentrate could also be identified at a seasonal basis, from daily VMS and landings data. Fleet dynamics and landings trends along the year, provide precise information for the spatial distribution of fishing effort and distribution and abundance of the target species, at the local scale and by vessel, thereby contributing to the implementation of the MSFD. As mentioned before, this information can be used as input for indicators of GES.

Daily landings outliers in combination with vessel position can become a useful tool for the control of the fleet activity. The assumption is that, if a daily landing or income is well-above the monthly average, the species explaining this outlier is different from the average catch, i.e. the vessel fished in grounds different from those where it deployed most of its monthly activity. This does not necessarily mean that the trawler has entered into non permitted grounds, but this outlier suggests checking which species is explaining the high landing/income and where it has been fished. The biology of the target species allows the interpretation of outliers. Thus, for instance, in the outlier example shown in Figs. 4a and 5, the species responsible for the sharp increase of daily landing was identified as red mullet while the coastal vessel position was known from VMS data. These results point to the start of the red mullet recruitment to the shallow coastal waters fishing grounds.

We propose that this combined information of vessel positions and landings, at the daily scale, has the potential to be used as a tool for fleet management. For instance, this information could provide options for the best choice regarding where and when spatial and temporal closures could be implemented, for the limitation of fishing hours per day, and even in the control of marine protected areas limits (for instance, by implementing a buffer area in case of high concentration of trawlers very close to the limits of the reserve).

In the study area different activities compete with bottom trawling for the use of the space (existence of two coastal marine reserves, small-scale and recreational fishing, and leisure- related activities). Temporal and spatial components need to be incorporated in EBFM in order to support current and future uses of marine ecosystems and maintain the delivery of valuable ecosystem goods and services for future generations Foley et al. 2010). The spatial zoning of the marine environment needs to be systematically implemented in the Mediterranean to protect sensitive areas (de Juan et al. 2012, Fogarty et al. 1998, Babcock et al. 2005). To achieve this, detailed analysis of geo-referenced data from the fleet dynamics is essential in order to consider management scenarios Stelzenmüller et al. 2013, Babcock et al. 2005).

5. Conclusions

Accurate knowledge of fishing targets driving the fleet dynamics along with knowledge on fishing strategies at the vessel level (i.e. fishing ground-habitat where the fishing pressure is exerted and corresponding landings) are a potential tool for fleet management. VMS has been shown to be a feasible tool for Mediterranean fleets management, as it helps linking landings data to their corresponding fishing ground. However, despite the use of VMS data for the management of the fleet and implementation of the EBFM is straightforward due to the existence of data on daily landings, this is an issue that needs further development. Overall, the complementation of these tools may contribute to achieve both fishing and environmental targets and, hence, to the EBFM implementation in the Mediterranean.

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La necessitat d'avaluar l'esforç pesquer de les flotes d'arrossegament a petita escala per a una bona gestió basada en l'ecosistema: exploració de tres fonts de dades diferents en caladors mediterranis.

RESUM:

Un dels paràmetres més utilitzats en el Mediterrani a l'hora de gestionar la pesca d'arrossegament és l'esforç pesquer. Per altra banda, una estimació precisa de l'esforç pesquer és la base per a un bon coneixement de l'impacte de la pesca d'arrossegament en els ecosistemes bentònics. Així doncs, un coneixement acurat de l'esforç permetrà conèixer l'impacte del bou en els fons marins i podrà ser utilitzat en la gestió d'aquestes pesqueries. La mesura d'aquest esforç es pot fer en relació a les captures, p.ex. Captures Per Unitat d'Esforç (CPUE). Però, en el marc d'una gestió més global en què també es tinguin en compte els efectes de la pesca sobre els ecosistemes marins, cal una unitat d'esforç que relacioni el bou amb els seus impactes sobre les comunitats bentòniques. Això permetrà avaluar l'impacte dels bous en els ecosistemes bentònics i poder assegurar tant un bon estat dels ecosistemes com un nivell de pesca sostenible.

En aquest treball es comparen tres mètodes diferents per a l'estimació de l'esforç pesquer: dades de pesca (entrevistes amb els pescadors i informacions obtingudes en les vendes diàries a les confraries), dades del sistema de localització de vaixells per satèl·lit (Vessel Monitoring System-VMS) i imatges del sonar de rastreig lateral (Side Scan Sonar-

SSS). L'objectiu del treball és discutir els seus punts forts i febles per a estimar l'esforç pesquer a petita escala.

Es van comparar caladors de pesca situats en 4 àrees del Mediterrani (Cap de Creus, Cabo de Palos, Mar de Ligúria i Mar Jònic), cadascuna de les quals presentava 3 zones amb un esforç de pesca diferent: alt, mitjà i baix o sense esforç. Aquestes zones es van establir a partir de la informació procedent de les dades de pesca. Les dades de VMS només estaven disponibles per a Cap de Creus i Cabo de Palos, on es va fer l'estimació pels 3 mètodes. A les altres dues zones només es van poder comprar els altres dos mètodes d'estimació.

Comparant les estimacions obtingudes pels tres mètodes podem establir que:

-La informació obtinguda de les dades de pesca és un bon punt de partida per a localitzar els caladors i les espècies objectiu de cada flota i definir aproximadament les àrees d'influència de cada port, però és massa poc precisa per a valorar l'impacte dels bous sobre l'ecosistema bentònic.

-Les dades provinents del VMS proporcionen informació independent sobre la dinàmica temporal i espacial de la flota, però la resolució espacial de les dades que actualment proporcionen els vaixells (un senyal per vaixell cada dues hores) és d'una freqüència massa baixa per a realitzar una bona estimació a nivell de l'ecosistema bentònic.

-El SSS proporciona informació sobre l'estat real del fons marí ja que en les imatges s'hi poden observar clarament les marques deixades per les portes de l'art de pesca sobre el sediment. Així doncs aquestes dades tenen una bona resolució per a estimar l'esforç pesquer a nivell bentònic. Cal, però, tenir present que el temps que perduren les marques sobre el fons depèn de la seva naturalesa (p.ex. en fons de fang perduren més que en fons de sorra). Per tant caldrà tenir en compte aquest aspecte en el processament de dades d'aquest mètode.

La informació obtinguda pels tres mètodes és necessària per a un bon coneixement de l'esforç pesquer i per a poder estimar la resposta dels ecosistemes bentònics sotmesos a diferents nivells d'esforç. Els tres mètodes mostraren un patró similar, però les variacions en els resultats podrien tenir importants conseqüències en l'avaluació dels efectes del bou. Per tant, cal tenir present l'escala a la qual es vol avaluar l'impacte per tal d'obtenir estimacions precises al nivell de les comunitats bentòniques i els hàbitats.

The need for fine-scale assessment of trawl fishing effort to inform on an ecosystem approach to fisheries: exploring three data sources in Mediterranean trawling grounds

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HIGHLIGHTS

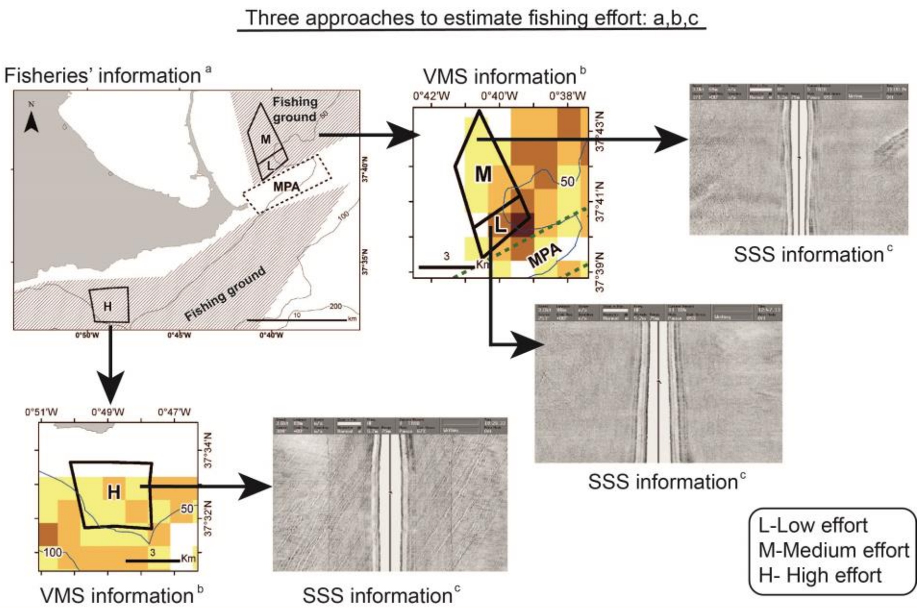
● Accurate effort estimation to evaluate trawling disturbance on benthic ecosystems.

● Feasibility of fishery data, SSS and VMS to assess trawling effort at benthic level.

● The 3 approaches were tested in 12 sites located in Mediterranean trawling grounds.

● The strengths and weakness of the 3 methods were assessed.
● Importance of fine-scale effort estimation to ensure good environmental status.

GRAPHICAL ABSTRACT



ABSTRACT

Aiming to estimate fishing effort at a small-scale that can be linked to benthic ecosystem disturbance (including target and non-target communities and habitats), three approaches were compared in 4 Mediterranean areas subjected to different trawl effort: i) information from the fishery (fishermen interviews and daily sales' records), ii) Vessel Monitoring System (VMS) data and iii) Side-Scan Sonar (SSS) records. Fishery data proved a useful baseline to characterise effort in the fishing grounds, VMS described in detail the dynamics of the fleet and SSS provided evidence of small-scale trawl activity. This paper examines the strengths and weaknesses of these 3 methods regarding their feasibility to assess the effort distribution at the smallest possible scale, but with a broad application. The findings highlight the need for accurate fishing effort estimations to evaluate trawling disturbance on ecosystems to ensure both healthy benthic ecosystems and sustainable exploitation of target species populations.

Keywords: small-scale effort estimation, trawling impact, sustainable resources, good environmental status, Mediterranean

1. Introduction

Trawling must be adequately managed to simultaneously minimise the overexploitation of resources and benthic ecosystem degradation (Thrush & Dayton 2010). Many studies have demonstrated that towed gears chronically alter benthic ecosystems, including target and non-target species and habitats (e.g. Watling & Norse 1998, Eleftheriou 2000, Kaiser & de Groot 2000, de Juan et al. 2009). Moreover, many target species are overexploited, indicating the inefficiency of past management strategies (Pauly et al. 1998, Berkes 2012, Colloca et al. 2013). Facing this failure in management, there is an urgent need to advance in the implementation of an Ecosystem Approach to Fisheries (EAF) globally through regional management plans.

The European Marine Strategy Framework Directive (MSFD), (EU 2008), requires Member States to monitor marine ecosystems subjected to disturbance and achieve a Good Environmental Status (GES) by 2020 (EU 2008). To accomplish this ambitious goal, accurate information on current threats to marine ecosystems is urgently needed, including detailed knowledge of the level of fishing effort that threatens the integrity of ecosystems (Hall 2002, Beddington et al. 2007, Svane et al. 2009). Related to this, the aim of EAF is to avoid surpassing a critical threshold for ecosystem degradation. In this context, it is of paramount importance to estimate fishing effort at a small scale linked to the variability of biological communities (de Juan & Demestre 2012, Berkes 2012). However, this is not an easy task, and few of the existing approaches for effort estimation have a fine spatial scale resolution (Rijnsdorp et al. 1998, Dinmore et al. 2003, Piet & Jennings 2005). The need for fine scale effort estimations grows in importance in the Mediterranean, where the traditional management of demersal fisheries is centred on the limitation of effort and regulation of minimum landing size (Papaconstantinou & Farrugio 2000, Lleonart & Maynou 2003).

In this study, three different approaches to estimate the spatial distribution of trawling effort were comprehensively analysed: data from the fishery, Vessel Monitoring System (VMS), and Side Scan Sonar images of the seafloor (SSS). Fishery data is a direct way to record fishing effort (e.g. days at sea and number of operational vessels per month) and this information allows to describe fleet dynamics, seasonality of landings, etc. (McCluskey & Lewison 2008, Vermard et al. 2010). Legislation to monitor European fishing vessels movement using a satellite VMS was introduced in 1998 by the European Commission (EC 2847/93) and, after a series of modifications, since 2012 it applies to all vessels over 12m (EC 1224/2009). These VMS devices installed in every vessel records its position at least every 2h, and this information is then transferred to the local fisheries authorities. VMS has become an important source of spatial and temporal effort estimations (Mills et al. 2007, Jennings & Lee 2012, Hintzen et al. 2012) and has improved our knowledge on the geographical distribution of fishing fleets (Murawski et al. 2005, Witt & Godley 2007, Stelzenmuller et al. 2008, Russo et al. 2011). SSS images record the trawl marks on the sediment caused by the "doors" of the fishing gear, showing the parallel and equidistant marks made by the two "doors" that unambiguously reveal the presence of trawlers. The SSS is also frequently used to define sediment morphology and for describing and detecting small-scale and patchy distribution of benthic structures and habitats (Brown et al. 2002, Cochrane & Lafferty 2002, Collier & Brown 2005, Lathrop et al. 2006, Lucchetti & Sala 2012, de Juan et al. 2013). Nevertheless, although this geophysical method may be considered as an efficient tool for indirectly estimating the trawling effort by a geo-acoustic mapping system, few studies exist that quantify the density of trawl tracks on the sediment to estimate the fishing effort (Vanstaen et al. 2007, Krost et al. 1990, Friedlander et al. 1999, Sartor et al. 2007, Demestre et al. 2010).

These three methods were used in 2 areas in the Mediterranean (in the coasts of Spain), while only two methods were used in the other 2 areas (in the coasts of Italy and Greece). With a few exceptions, the otter trawl fleets from the Italian and Spanish ports operate within their own fishing grounds (i.e. there is no movement of vessels between fishing grounds), bounded by the fleets from the neighbouring ports. Fishing grounds are located from 50 m to 900 m depth. Spanish trawlers generally return to port every day with 6-8 hours of effective fishing activity, and Italian trawlers follow a similar pattern with 10-12 effective fishing hours per day (Sartor et al. 2007, Demestre et al. 2010). Greek trawlers operate with different dynamics, moving across several fishing areas under Greek jurisdiction. Trawlers might return every evening to sell their landings in the nearest port or in the harbour offering the best income. Otherwise, they might stay at sea for up to two days (Kapatangis 2007).

As trawling management in the Mediterranean is mainly based in effort control, the fine-scale effort estimation could be an essential input for fisheries management aiming to achieve sustainable exploitation of resources and healthy ecosystems. Thus, the principal goal of the study is to examine the three approaches used to estimate fishing effort and to analyse their strengths and limitations. The final aim of the study is to highlight the importance of small spatial scale effort estimation to link trawling activities with their effects on benthic ecosystems.

2. Material and Methods

2.1. Case studies

The present study was carried out in four Mediterranean coastal areas: one in Italy, Liguria Coast (LC); one in Greece, Eastern Ionian coast (IC); and two in Spain, the Catalan coast (Cap de Creus - CC) and the Murcia coast (Cabo de Palos - CP) (Fig. 1). These areas are located between 40 and 90 m deep and have muddy/muddy-sand homogeneous habitats except for CP that has gravelly-sand and maërl habitats (de Juan et al. 2013). All the areas are in trawling grounds except for a section of CC area that is within the MPA of Cap de Creus (CC-MPA), where trawling activity is prohibited (DGPM 4/1998).

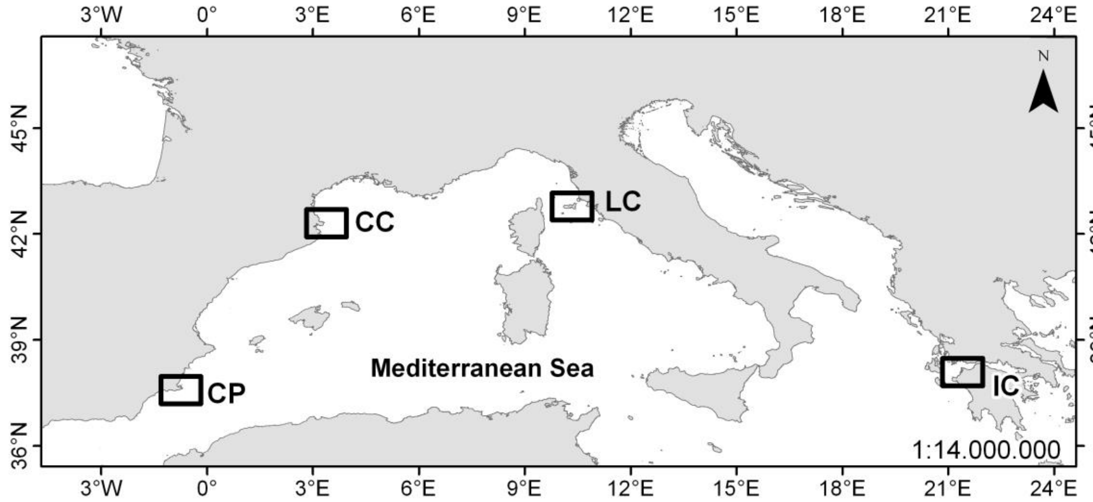


Figure 1. Location of the four study areas: two in Spain (CP and CC), one in Italy (LC) and one in Greece (IC).

2.2. Fishing effort estimations

The fishing effort data analysed varied between countries. In Italy and Greece only fishery data and SSS trawl tracks were used to estimate the effort. In the two Spanish areas, where VMS data was available, data from the three methods were analysed.

2.2.1 Fishery data

The fishing effort was estimated by combining official records including daily sales in each port with data collected from interviews with fishermen. Interviews were conducted in the most representative local ports from each area during March 2009. All fishermen working with trawling gears from each port considered for the study were asked to mark on a map the fishing grounds they mainly visited. Also, fishermen had to describe if they changed fishing grounds on a seasonal basis following the life cycle of target species. Taking into account this information, the species´ composition of landings recorded during the daily sale provided evidence on the fishing grounds visited by each vessel (Fig. 2). This approach was based on knowledge on the habitat preferences by the main target species (Martín et al. 2014). The daily sale records were also used to determine the number of days these vessels had actually worked on

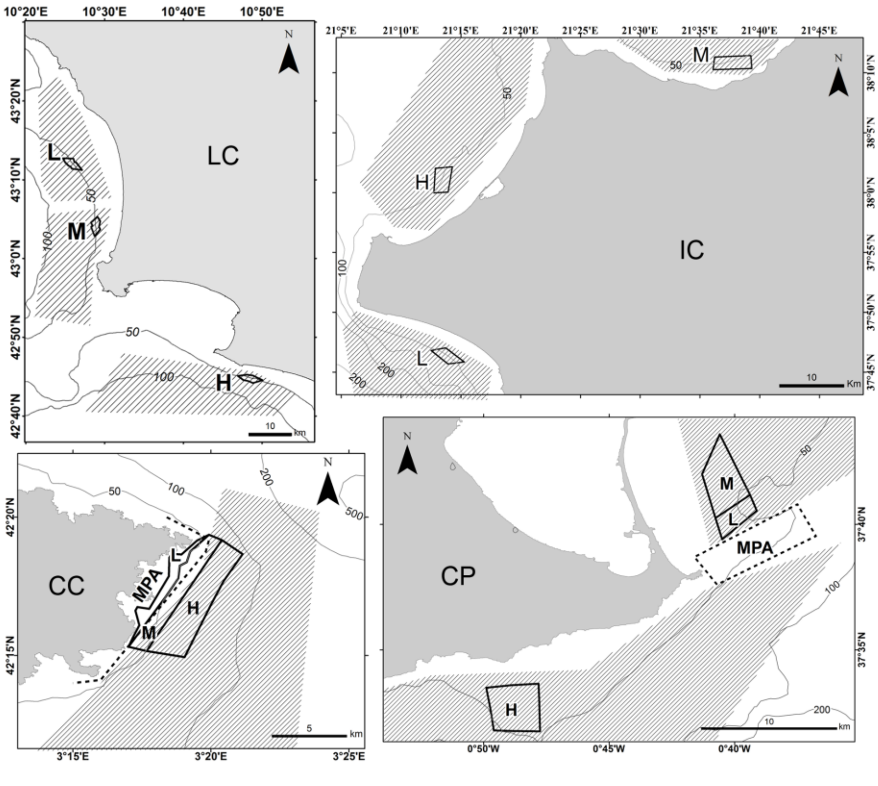


Figure 2. Location of the fishing grounds (stripped areas) in the study areas (LC, IC, CC, CP). The polygons delimit the selected sites with different levels of fishing effort (L: Low, M: Medium, H: High) within each area.

these fishing grounds.

Trawling effort was estimated as total active fishing days per month within the fishing ground. The Gross Tonnage (GT) was included in the effort estimations as different vessel characteristics have different impacts on benthic communities. The GT is considered by the EU DG MARE (EU 2009) as the most appropriate fishing capacity parameter to categorize the vessels. GT data from the trawling fleet for the period 2007-2009 were obtained from the official censuses from the Fisheries Ministry in each country.

The final fishing effort was estimated in each fishing ground as follows:

$$GT * \text{fishing days/month} \quad (1)$$

2.2.2 Satellite information: VMS data.

The VMS data from the Spanish fleets operating in CC and CP were provided by the Secretaría General de Pesca (MAGRAMA) of the Spanish government. The data analysed included daily records from 2007, 2008 and 2009. Only information from trawlers known to be fishing within the studied fishing grounds in CC and CP (from 2.2.1) was used, which resulted in nine vessels per area, 60% and 40% of each fleet respectively.

The raw VMS data does not indicate whether a vessel is fishing when it sends positional data. Vessels are fishing when the VMS signal corresponds to a speed lower than 4 knots (Lleonart & Maynou 2003, Sala et al. 2007). At greater speed the vessels are steaming rather than fishing. Hence, signals above 4 knots were removed. Signals with speeds around 0 knots were also removed from the analysis to minimize the risk of including vessels that were recovering the gear after a haul (Murawski et al. 2005, Lee et al. 2010). After filtering the data, the analysis was based on signals with a speed ranging from 0.1 to 4 knots.

The filtered VMS data set was used to represent the fishing effort distribution on a grid (Dinmore et al. 2003, Mills et al. 2007). The scale of the grid has a noteworthy consequence on the assumed impacts of fishing, as large grids tend to homogenize the fishing effort (Piet & Quirijns 2009, Hinz et al. 2013). Therefore, taking into account the objective of our study, this grid had to be designed at the lowest possible resolution because benthic communities vary at scales smaller than fishing grounds (Collie et al. 2005). In the studied fishing grounds, the benthic fauna was surveyed within 2-4 km^2 plots (de Juan et al. 2013), therefore, we used a 1 km x 1 km VMS grid.

Then, to estimate the fishing effort in each cell, the following expression was used:

$$Days\ at\ sea\ x\ GT/day = \sum_{i=n}^{i=1} \left(\frac{y}{TFVis}\right) x\ FV\_GTi \qquad (2)$$

Where *n* is the number of fishing vessels operating in the cell, *y* is the number of signals per day of a fishing vessel in that specific cell, TFV_is is the total number of signals from that fishing vessel in that day and FV_GT_i is the GT value of the vessel (Kavadas & Maina 2012).

Based on the values obtained from (2), fishing effort distribution maps were represented using ArcGis 10.0 package.

2.2.3 Side Scan Sonar (SSS) surveys

SSS surveys were performed using the C-Max CM2 model operating at 325 kHz covering a surface around 1.05 km^2 in each fishing ground. Trawl marks were estimated by Sonographs that had 75 m range with total swath coverage of 150 m. Each survey followed a design of two lines of 2-3 km long, parallel to isobaths and separated 1000 m from each other, and three lines perpendicular to the previous ones, of 1-2 km and separated also 1000 m from each other (Figs 3, 4, 5, 6). The boat was positioned by real-time differential GPS and the surveys were usually conducted during calm sea conditions.

The SSS was towed at a depth between 6-12 m above the bottom. Images obtained with the SSS were analysed by a computer-aided planimetric analysis that allows the digitalization of the trawl tracks. The digitized tracks were processed using the ArcGis software to measure the trawl length and to ultimately determine trawl marks' density. Density was estimated considering the total length of trawl marks (parallel and perpendicular) in lineal meters standardized to the total surface surveyed. From fishery data, we assumed that our studied fishing grounds showed an homogeneous effort pattern. Hence, when summing the total track length the directional effect is negligible.

The geographic latitude/longitudes were converted to UTM referred to WGS84 Datum, employed by the Global Positioning Systems.

In each fishing ground, the extent of fishing activity was estimated by relating the area with trawl marks to the total area surveyed:

Trawl Tracks Lineal Density = trawl tracks length (m) / surface (Km2) (3).

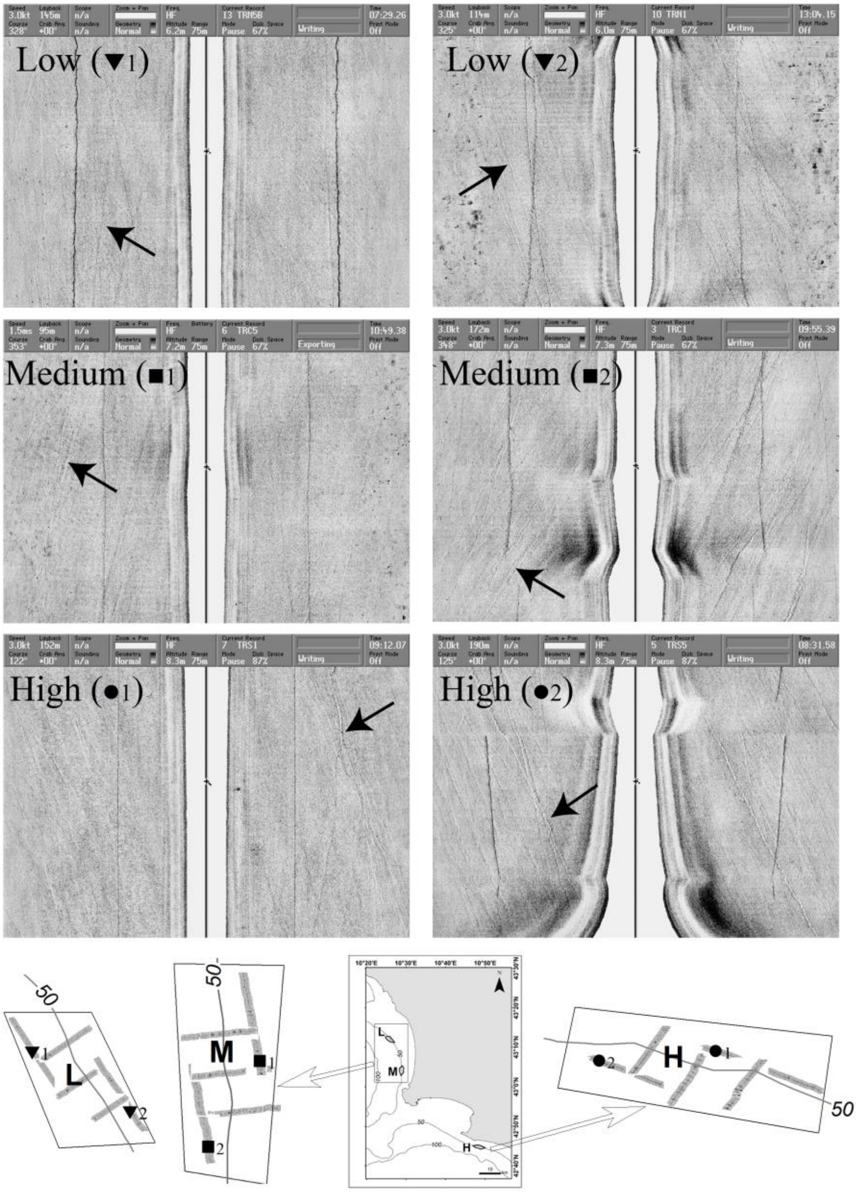


Figure 3. SSS images from each of the three LC sites. Examples of trawl tracks are marked (arrow). Image in the bottom: SSS survey's position.

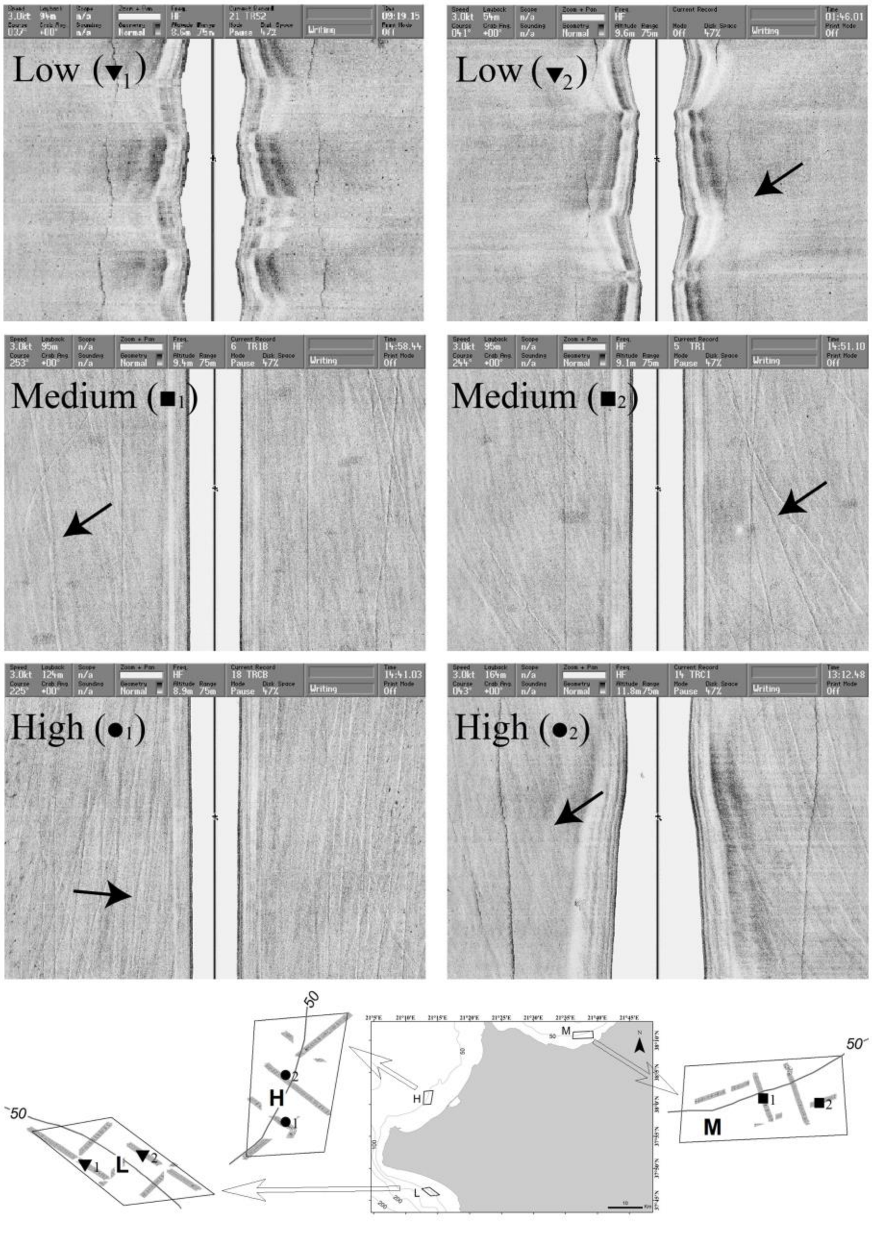


Figure 4. SSS images from each of the three IC sites. Examples of trawl tracks are marked (arrow). Image in the bottom: SSS survey's position.

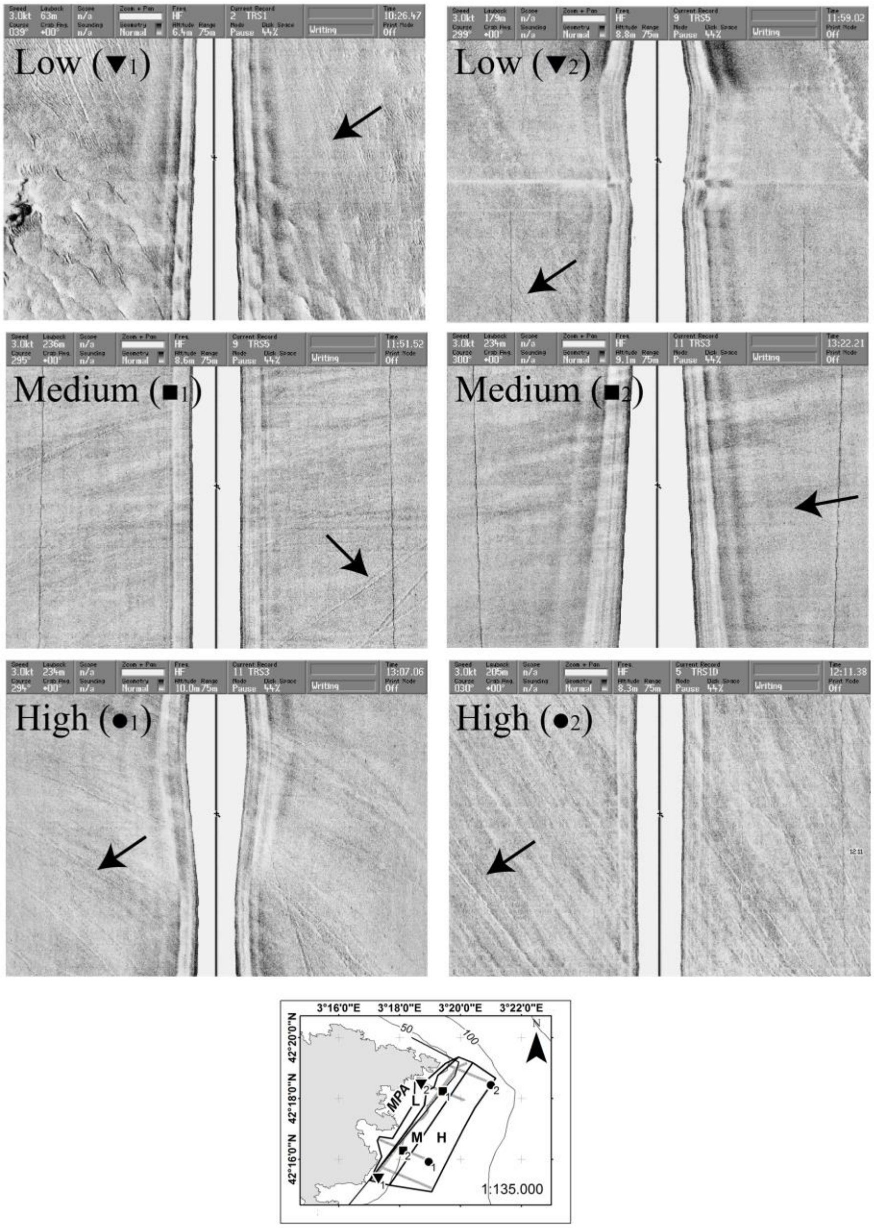


Figure 5. SSS images from each of the CC sites. Examples of trawl tracks are marked (arrow). Image in the bottom: SSS survey's position.

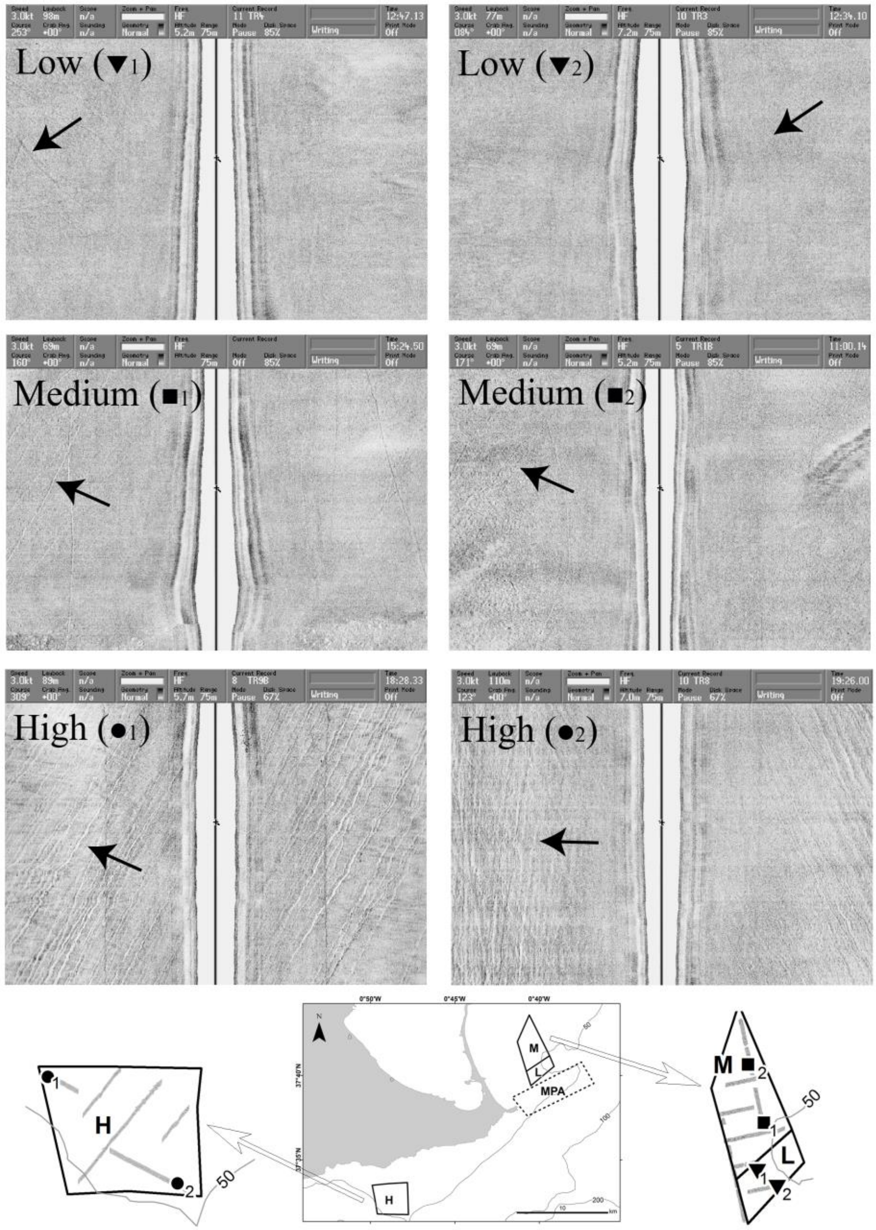


Figure 6. SSS images from each of the three CP sites. Examples of trawl tracks are marked (arrow). Image in the bottom: SSS survey’s position.

3. Results

The information obtained from the equation (1) gave consistent estimates of the fishing activity, as results showed a similar seasonal pattern over the years in each study area. Data from fishermen interviews was geographically referenced to map the dynamics of the trawl fleets over a year in the four study areas, highlighting the most visited fishing grounds. Therefore, this information is a good baseline to identify fishing grounds in each area subjected to different fishing effort levels (Fig. 2).

These results allowed the selection of 3 sites in each fishing ground subjected to different levels of fishing activity in each area: H) sites with high effort level, M) sites with medium effort level, and L) sites with low effort level (Fig. 2). In the CC area a site with no-effort, the MPA of Cap de Creus (CC-MPA), was labelled “L”. Therefore, a total of 12 sites were selected (Table 1a). It should be noted that the 3 levels of fishing effort in each area are not comparable with those from other areas (e.g. level H in a site does not necessarily compare to a level H elsewhere).

From this baseline information, SSS surveys were performed within the selected sites (Table 1b). As an example, figures 3, 4, 5 and 6 show a couple of images in each site. The SSS images in LC showed different number of trawl tracks in each site and evidenced a high fishing activity in H site, less activity in M and few in L (Fig. 3). All these images evidenced trawls tracks on the typical soft muddy-sand sediment where trawlers operate. In IC, with sandy-mud sediment, the

Table 1. Fishing effort estimations from a) fishery information and b) Side Scan Sonar images (SSS). LC: Ligurian Coast; IC: Ionian Coast; CC: Cap de Creus; CP: Cabo de Palos (see Fig.1). L: Low fishing intensity; M: Medium fishing intensity; H: High fishing intensity.

Area	a) Interview (GT*days at sea/month)	b) SSS (Trawl tracks length/km2)
LC-L	6.256	87.542
LC-M	7.930	96.954
LC-H	17.280	129.145
IC-L	5.593	86.615
IC-M	10.603	102.016
IC-H	17.255	122.131
CC-L	0	22.393
CC-M	1.322	63.171
CC-H	1.623	81.113
CP-L	1.125	48.248
CP-M	17.889	54.199
CP-H	22.881	137.243

L site was clearly different from H and M sites that evidenced higher fishing activity (Fig. 4). The analysis of CC images (Fig. 5) evidenced the presence of trawl tracks on the seabed within the MPA, revealing fishing activity within the MPA boundary. The trawl marks in this site were typical of sandy sediments. Finally the images from CP area (Fig. 6) showed an intense fishing activity in H site. The L site also had a relatively high presence of trawl marks on the seabed, just on the border of Cabo de Palos MPA (Fig. 2). In this case the sediment was muddy-sand.

VMS data from the three years produced very similar effort distribution patterns, hence, only data from 2009 is shown (Fig. 7 and

8). The VMS signals from CC and CP sites were represented in a map and overlapped with the information from 2.2.1 (Fig. 7a and 8a).

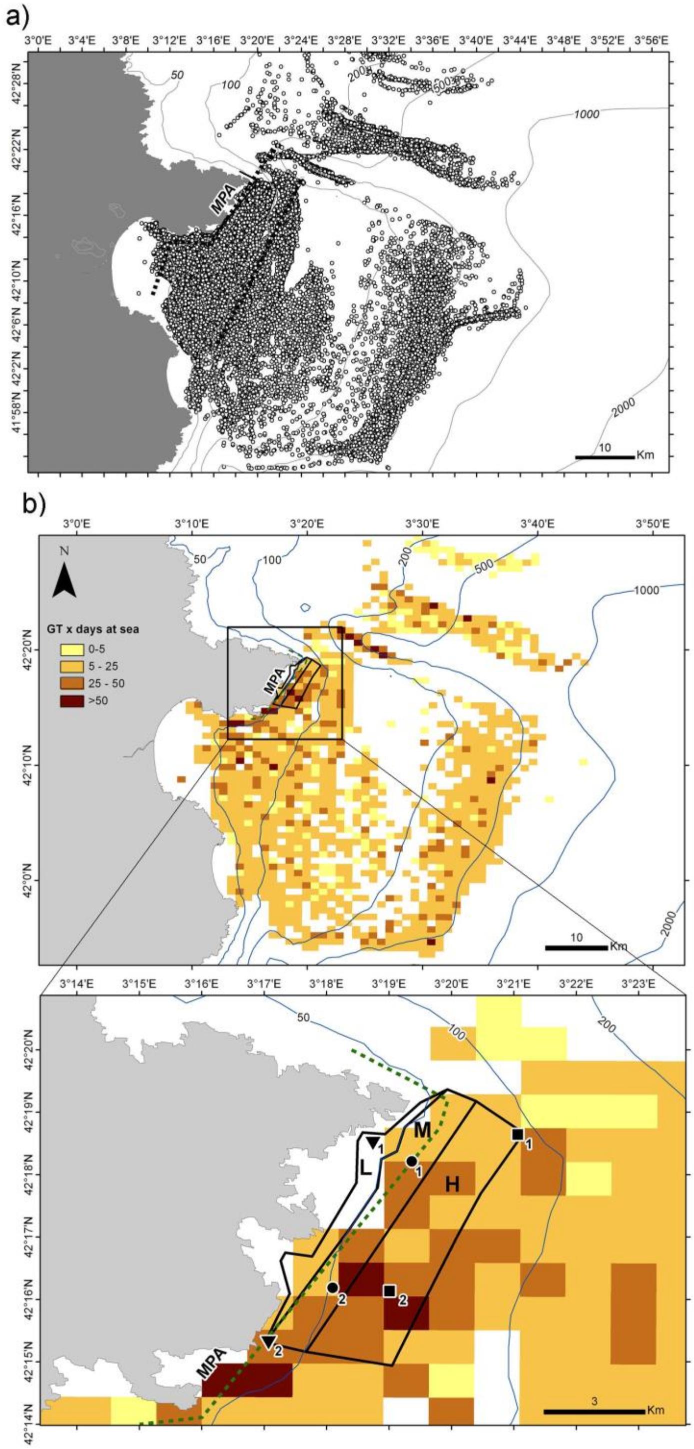


Figure 7. a) Distribution of VMS signals in CC study area (hollow circles). Discontinuous black lines represent the fishing grounds delimited from the fishery information. b) Effort calculated from 2009 VMS data on a 1x1 km grid. General view of the study area and detailed view of the study sites. Symbols in the study sites indicate SSS images' positions: triangle, CC-L; circle, CC-M; and square: CC-H (SSS images are shown in Fig. 5).

Regarding effort estimation patterns, in CC sites the nine selected trawlers generally had a coastal distribution, between 50 and 100 meters deep, in agreement with information obtained from fishermen (Fig. 7b). It is worth highlighting that fishermen information claimed null effort within site CC-L, the MPA, whereas both VMS data (Fig. 7b) and SSS images (Fig. 6) evidenced fishing activity in this site. The VMS signals were located just in the MPA border.

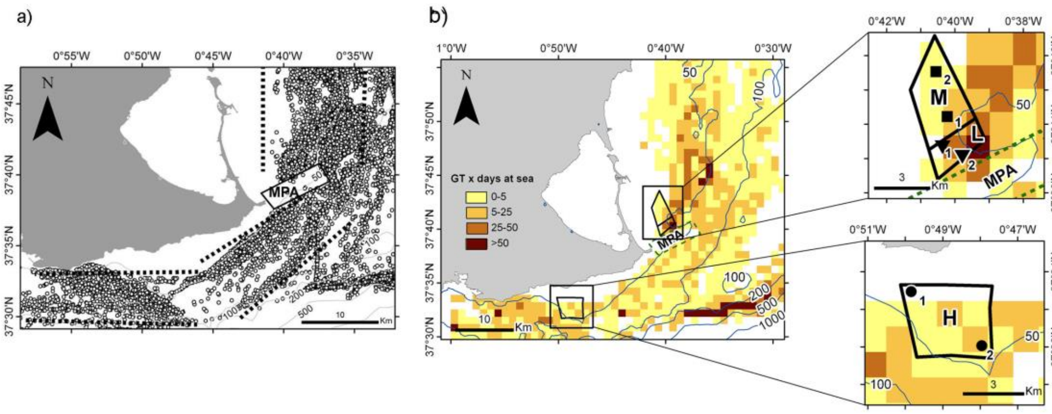


Figure 8. a) Distribution of VMS signals in CP study area (hollow circles). Discontinuous black lines represent the fishing grounds delimited from the fishery information. b) Effort calculated from 2009 VMS data on a 1x1 km grid. General view of the study area and detailed view of the study sites. Symbols in the study sites indicate SSS images' positions: triangle, CC-L; circle, CC-M; and square: CC-H (SSS images are shown in Fig. 6).

In the CP sites the trawling effort estimated from VMS (Fig. 8) also showed that the selected trawlers had a high fishing activity on shallow grounds (50-100 m). Some disagreements were detected between fishery information and VMS data. While fishery data expressed a low fishing activity in site L (Table 1a), the VMS exhibited a high effort in this site compared to M and H sites (Fig. 8b). It is also important to underline the surprisingly high effort to which, according to the VMS data, the no-take MPA was subjected to (Fig. 8b). Data from the SSS agreed with the fishery data (Table 1). Nevertheless, it should be stressed that, while data from the fishermen interviews showed a large difference in effort between L and M sites, SSS data did not (Table 1).

4. Discussion

In the present study, three approaches for trawl fishing effort estimation were examined: fishery data, VMS signals, and SSS trawl marks. The three approaches offered different levels of accuracy and each one provided results at different spatial and temporal resolutions. The three methods were complementary and their combination provided useful information to evaluate the trawl effort intensity and distribution. This information is essential to assess the effects of trawling on exploited commercial and non-commercial species at the adequate spatial scales. The adequate spatial resolution is necessary to identify ecosystem disturbances and to ultimately achieve a GES of trawling ground ecosystems (Pikitch et al. 2004).

The study pointed out that local fishery data was useful as baseline information to define the fishing activity framework in a particular fishing ground. Landings´ data were useful for providing information to obtain the daily distribution and seasonality patterns of trawl fleets, as also pointed out Bellman et al. (2005) and Alexander et al. (2009) (Table 2). In agreement with Jennings & Lee (2011), our results support VMS as a good data source to spatially define fishing grounds (Fig. 7a, 8a). The combination of these two data sources, fishery information and VMS, demonstrated the spatial complexity of fishing activities to define the spatial distribution of trawl effort in fishing grounds. Moreover, as previously observed (Witt & Godley 2007, Lee et al. 2010), VMS information provided an independent way to assess the spatial accuracy of reported fishery data (Table 2).

Table 2. Summary of the main strengths and weaknesses of the three approaches used to estimate fishing effort at a benthic scale. VMS: Vessel Monitoring System, SSS: Side Scan Sonar

	Strengths	Weaknesses
Fishermen interviews	Baseline to define fishing grounds Straightforward to obtain	Not precise enough at the benthic level Willingness of the fishermen to provide information Conditioned by fishermen subjectivity
SSS	Accurate picture of trawl tracks on the seabed Clear evidence of fishing activity Effort information at benthic community level	The persistence of the trawl tracks depends on the sediment Difficulty to implement and to process data
VMS	Independent data on the vessel position Useful to define fleet dynamics at stock level Easy to obtain information as it is compulsory for all fishing vessels above 12m	Lack of high frequency position signals Too coarse at benthic level Need several filtering techniques (e.g. filters of speed, gear) before using

A large amount of work has been done over the past years on the use of VMS to estimate trawl fishing effort. Different interpolation approaches (Hintzen et al. 2010, Russo et al. 2011) and point summation methods (Dinmore et al. 2003, Mills et al. 2007) have been discussed to interpret VMS registers and to estimate the number of times trawlers have crossed the seabed of a fishing ground. Some authors have also explored the possibility to use VMS data for benthic impact assessment (Tillin et al. 2006, Gerritsen et al. 2013), but, for this purpose, it is essential to specify that more frequent time intervals for VMS records than the current 2-hour interval between signals (EC 404/2011) are needed. Furthermore, the covered geographic scale is too large to encompass the heterogeneity of benthic communities and habitats (Witt & Godley 2007, Skaar et al. 2011, Hinz et al. 2013). In agreement with these studies, our results based on a point summation methodology for 1km x 1km cells, showed that the current 2 hours interval between signals is not enough to obtain an accurate effort estimation at benthic scale (Table 2).

On the other hand, the use of SSS records gives an unmistakable location of trawl paths. The images of the trawl tracks offered an accurate geographic positioning of the fishing activity distribution over the seabed. Therefore, this approach could provide an accurate assessment to link effort distribution with benthic communities and habitats. The SSS results provided effort estimations consistent with benthic community state in the study sites (Table 2). Trawl track density in CP-L hinted that effort in this site might be higher than what fishery information stated, and CP-M might have lower effort than what fishery information suggested. This is in accordance with de Juan et al. (2013) that suggested that the CP-M site harbours a benthic community with high density of vulnerable species that is not reflecting the high trawling effort allocated to this site. This site stood as an outlier regarding an analysis linking benthic community composition and direct effort estimations.

The SSS method allowed an estimation of effort at a spatial scale smaller than VMS estimations. For example, M sites in both CC and CP areas seemed to be barely fished when assessed by VMS data, whereas the SSS images evidenced a high density of trawl tracks at a smaller scale. On the other hand, results from VMS data indicated high effort in CP-L, whereas SSS images for this site did not evidence such a high effort. This inconsistency could be explained by a concentration of vessels in the border of the MPA without effective fishing. Other differences were in CP-H site, where the VMS data indicated lower effort than estimates by SSS images. VMS data in CC and CP sites offered pertinent spatial and temporal distribution of trawl fleets and SSS records provided a true picture of the seabed. SSS images evidenced the presence of trawling disturbance in the protected area of CC-L or at depths shallower than 50 m in CC-M and CP-H (Fig. 7b, 8b), where trawl fishing activity is banned (Demestre et al. 2010).

To accurately estimate the fishing disturbance on benthic communities through approaches useful for management decisions, there is a need of biological indicators of benthic ecosystem disturbance that must be tightly linked to the effort level as the disturbance factor. Such analysis would be useful to identify changes in benthic ecosystems through time in relation with variability in fishing activities (Greenstreet et al. 1999, Demestre et al. 2008, Frid 2012), or to monitor community recovery after cessation of fishing activities (Pipitone et al. 2000, Dinmore et al. 2003, Hiddink et al. 2006). In de Juan & Demestre (2012) trawling effort and benthic community status were simultaneously estimated in order to evaluate the performance of an indicator related to the level of benthic community disturbance. Nevertheless, in agreement with that study and as evidenced by the present study, the link between community change and effort intensity could be improved with the computation of trawl tracks from SSS. It is important to take into consideration that the detection of trawl marks and its permanence on the seabed depends of the local physical conditions (Smith et al. 2007, Palanques et al. 2014). For example, the maërl sites in CP provided large inconsistencies between the three methods, and SSS methods to estimate track density are not common in this type of habitat. Hence, additional work is required to improve the adaptation of the SSS method to a variety of sediments (Table 2). Nonetheless, tracks can persist up to a year on mud and sandy-mud (Palanques et al. 2001) that would allow assessing the cumulative impact on benthic communities and habitats that could represent higher disturbance levels than isolated effort values (Stelzenmuller et al. 2008).

According to the MSFD recommendations (EU 2008), information from SSS could be useful for a broad comparison of the state of soft-bottom communities across EU continental shelves. A generalised use of SSS could produce a trawl fishing effort database from SSS track maps of the seabed exploited by trawlers, similarly to what was conducted several years ago with the elaboration of bathymetric maps with acoustic surveys. This information, which could be standardized over different habitat types, could be used as a responsible and efficient methodology for determining the level of fishing pressure on benthic ecosystems. The final impact of the SSS database would be a direct implementation by integrating in that database data on target and non-target species and habitats to adequately manage ecosystems according to an EAF.

The main components of the EAF and the factors that can affect them, e.g. indicators, predation, productivity, sediment, growth rate, reproduction (Jennings 2005, Hollowed et al. 2011, Frid 2012) may be affected by variations caused by trawling on habitats and benthic communities. Therefore, the establishment of the maximum levels of trawling effort that can be sustained by benthic ecosystems may help to reduce the breakdown of resources as well as hampering life strategies such as spawning, recruitment, feeding or migration (Rice 2011).

This work provided strong evidences of the requirement to link the three methods introduced in order to advance towards an accurate methodology to estimate the fishing effort and progress in understanding the effects of trawling on communities and habitats within fishing grounds. The three approaches showed a similar general pattern. However, variability in results has important consequences for the evaluation of trawling disturbance that may lead to inadequate management decisions (Table 2). The importance of accurate effort estimations is highlighted in the context of the EAF, and relies on the need to study trawling activity and its effects on benthic communities and habitats at similar small spatial scales to work towards achieving a GES in trawling grounds.

Acknowledgements

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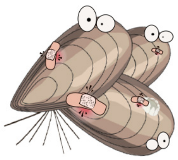
III

CHAPTER 2:

Trawling disturbance effects on benthic ecosystem functionality and ecosystem services.



Understanding ecosystem dynamics under trawling pressure is essential to correctly manage trawling fisheries. In this context, ecosystem functionality is a key process, as it is on the basis of ecosystem services delivery. This chapter addresses this issue from the point of view of trawling effects on ecosystem service delivery units (i.e. Ecosystem Serive Providers) and discussing the role of function redundancy on fishing ground's ecosystem resilience.

Section 1:
Integrating the provision of ecosystem services and trawl fisheries for the management of the marine environment.

Published in *Science of total Environment* 506-507, 594–603.
DOI: 101016/j.scitotenv.2014.11.042
See the original publication in Appendices

Section 2:
Assessing functional redundancy in chronically trawled benthic communities.

Under revision in *Ecological Indicators*.

Gestió dels sistemes marins: integració dels serveis ecosistèmics en la gestió de la pesca de d'arrossegament

RESUM:

Els serveis que proporcionen els ecosistemes bentònics són el resultat d'una sèrie de funcions que estan relacionades entre si i depenen les unes de les altres. Tot i que el servei més obvi que proporcionen les comunitats bentòniques dels caladors de pesca és la "provisió d'aliment" (captures), aquest servei depèn d'altres serveis bàsics com per exemple l'estructura d'hàbitat o la producció secundària.

El concepte de serveis ecosistèmics està sent molt utilitzat en gestió, ja que permet l'enllaç societat-ecosistema. Normalment, però, aquests serveis s'estudien des del punt de vista econòmic, deixant de banda el valor biològic o ecològic que poden tenir. Per això en aquest treball hem volgut abordar aquest aspecte biològic-ecològic dels serveis ecosistèmics basant-nos en l'estudi de la funcionalitat bentònica mitjançant el mètode dels trets biològics (Biological Traits Approach-BTA). Per a l'estudi es va analitzar l'epifauna de les comunitats bentòniques de 13 àrees situades en caladors de pesca d'arrossegament en 6 zones del Mediterrani. Cada àrea està sotmesa a diferents nivells d'esforç pesquer.

El mètode dels BTA consisteix en identificar les característiques biològiques que millor representin diferents aspectes de la morfologia, comportament, història de vida i altres caràcters de les espècies. Aleshores, per cada tret, se suma l'abundància de les espècies que el

presenten, aconseguint una matriu "d'abundància de trets". Aquesta matriu representa una aproximació indirecta a l'estudi de la funcionalitat del sistema, ja que per a estudiar pròpiament aquesta funcionalitat caldria mesurar processos duts a terme per la comunitat bentònica (p.ex. podem suposar que un organisme bioturbador contribueix al reciclatge de nutrients en el sistema, però no mesurem directament el flux de nutrients que allibera amb la seva capacitat bioturbadora).

A partir d'aquí, també es quantificaren indirectament cinc funcions ecosistèmiques considerades clau en els fons tous propis de la pesca d'arrossegament, els anomenats caladors de pesca: producció, reciclatge de nutrients, acoblament bento-pelàgic, segrest de carboni i estructura d'hàbitat. Cada funció és estimada mitjançant l'abundància i la biomassa del que anomenem proveïdors de serveis ecosistèmics (Ecosystem Service Providers- ESP) definits a partir dels trets biològics que presenten les espècies bentòniques de l'epifauna. Per exemple, els organismes grans de vida llarga que viuen adherits al substrat conformen l'ESP "Grans proveïdors d'estructura tridimensional" contribuint d'aquesta manera a la funció ecosistèmica d'"Estructura d'hàbitat", un servei ecosistèmic de suport. Algunes espècies que entrarien dins aquesta classificació són les gorgònies (p.ex. *Leptogorgia sarmentosa*) o la maneta de mort (*Alcyonium palmatum*).

Finalment, per tal d'estudiar l'efecte de l'esforç pesquer i les variables ambientals (granulometria del sediment, temperatura, contingut de matèria orgànica del sediment, terbolesa, etc..), en la funcionalitat (trets biològics) i en les funcions ecosistèmiques (ESPs), es van construir models additius generals (General Additive Models-GAM). En el cas dels ESPs es van utilitzar tan les estimacions a partir de l'abundància com de la biomassa. Els models GAM permeten saber si l'esforç pesquer i/o les variables ambientals afecten significativament els trets biològics i els ESP així com si aquest efecte és positiu, negatiu o no lineal.

Els resultats mostraren que la funcionalitat del sistema i els ESPs estan molt influenciats tant pel tipus de sediment com per l'esforç pesquer. Així doncs, es pot concloure que les característiques del sediment afavoreixen d'entrada la presència de certs ESPs i l'esforç pesquer impacta aquestes comunitats pre-establertes variant-ne l'estructura.

D'altra banda, el resultats obtinguts pels ESP són lleugerament diferents segons s'estimin amb biomassa o abundància, la qual cosa ressalta la importància de la unitat escollida a l'hora d'avaluar les funcions ecosistèmiques i la necessitat de considerar cada funció per separat.

Els resultats mostren la importància de mantenir els ecosistemes bentònics en bon estat per tal d'assegurar la provisió de serveis ecosistèmics. Per tant la gestió dels caladors de pesca ha de tenir en compte aquest aspecte assegurant d'aquesta manera una pesca sostenible. Així doncs, cal una visió global del sistema per a dur a terme una gestió integral de les pesqueries. La filosofia del DPSIR (Drivers – Pressures – State Change – Impact– Response: Activadors – Pressions - Canvi d'estat – Impacte - Resposta) sorgeix en aquest context com un mètode de representació de les relacions entre els diferents actors del sistema. El DPSIR presentat en aquest treball vol ressaltar el paper de la societat com a *Activadors*, ja que actuaria com a un grup de pressió que podria influenciar les decisions dels organismes gestors amb capacitat per a canviar la *Pressió* (pesca d'arrossegament) exercida sobre les comunitats. Aquesta *Pressió* provoca un *Canvi d'estat* en les funcions ecosistèmiques i en els estocs d'espècies comercials, la qual cosa implica un *Impacte* en la provisió de serveis. Així doncs és molt important conscienciar la societat de la necessitat de conservar els ecosistemes bentònics en bon estat per a mantenir els estocs comercials dins un nivell d'explotació sostenible. Aquesta informació és bàsica per a provocar una *Resposta* dels organismes gestors que tingui en compte tots els serveis ecosistèmics a l'hora de gestionar els caladors de pesca.

Management of the marine environment: integrating the provision of ecosystem services and trawling fisheries

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HIGHLIGHTS

• Ecosystem Service Providers (ESP) represented a link between function and services.

• ESP variability was mainly affected by sediment type and fishing effort.

• A DPSIR (Drivers – Pressures – State Change – Impact – Response) model is presented.

• The DPSIR aims to inform integrated management to ensure services provision.

• All socio-ecological system components are essential to adopt integrated management.

ABSTRACT

Disturbance of benthic communities by trawling may compromise ecosystem service delivery, including fisheries' catches. In this work, we assessed services from an ecological point of view by using the concept of Ecosystem Service Providers (ESP), i.e. ecological units that perform ecosystem functions that will ultimately deliver ecosystem services. Thirteen Mediterranean sites subjected to different levels of fishing effort were studied and, from a range of environmental variables included in the study, we found ESPs to be mainly affected by fishing effort and grain size. Our results suggest that ESPs' distribution depend on the habitat type and this natural variability influences ESP response to trawling. In order to summarize these complex relationships, we present a DPSIR (Drivers – Pressures – State Change – Impact – Response) framework adapted to trawling fisheries. This integrative tool aims to inform managers about the interactions among all actors involved in the management of ecosystem services in trawling grounds so as to help in the design of management strategies.

Key words: Benthic community, Ecosystem function, Ecosystem Service Providers (ESP), Fisheries, Integrated management, Mediterranean.

1. Introduction

Marine ecosystems provide a wide range of services that are essential for human well-being. (Beaumont et al. 2007, Townsend et al. 2011). However, human activities impacting ecosystems might compromise the service delivery, hence, these activities must be subjected to an appropriate management (Levin & Lubchenco 2008). For instance, fisheries catches are declining world-wide (Pontecorvo & Schrank 2014), which demonstrate that, management strategies have not been appropriate for maintaining healthy ecosystems that ensure the provision of the "fish for food" service. In fact, trawl fishing is broadly recognised as causing one of the most severe impacts on benthic ecosystems (Thrush & Dayton, 2002), including the modification of functional components (Bremner et al. 2003, de Juan et al. 2007).

Ecosystem services delivered by benthic communities are the product of many ecosystem functions that are interrelated (Townsend et al. 2011, Rees et al. 2012, Snelgrove et al. 2014). Moreover, benthic community structure not only regulates benthic ecosystem function, but also water column processes, e.g. nutrient fluxes which are positively related to primary production in the water column (Lohrer et al. 2004). In the fisheries context, trawling disturbance may compromise the service of food provisioning and also indirectly other services (e.g. bioturbation and nutrient cycling), which might ultimately be related with the food provisioning (e.g. productivity of fishing grounds). Therefore, all the functions related to benthic communities should be considered under an integrated management approach (Rees et al. 2012), with conservation targets that go beyond the food provisioning service.

To adopt an integrated management approach, it is necessary to understand community dynamics under stress, for which a functional approach (e.g. Biological Trait Analysis-BTA) has proved to be very useful (Bremner 2008). As ecosystem services depend on benthic functionality (e.g. deep burrowing fauna increase the oxygen flow into the sediment and extend the total denitrification zone, stimulating nutrient cycling that is involved in the ecosystem regulating service (Lohrer et al. 2004, Beaumont et al. 2007)), BTA has been proposed as a useful framework to assess the provision of ecosystem services (Bolam & Eggleton 2014, Bremner et al. 2006a, Frid 2011). However, the estimation of ecosystem services provision normally addresses economic or social values (Beaumont et al. 2007, Boyd & Banzhaf 2007) and rarely focuses on ecological measures (Kremen & Ostfeld 2005, but see Townsend et al. 2011, Snelgrove et al. 2014). In this context, the concept of Ecosystem Service Providers (ESP) aroused (Kremen & Ostfeld 2005, Cognetti & Maltagliati 2010) aiming to encompass a group of organisms sharing particular biological traits (BT) related with certain ecosystem functions and, ultimately, with the provision of services. Variation in functional components under stress is linked to changes in BT, which are in turn linked to species composition (Bremner 2008). This, altogether, will determine the ESP dynamics. The understanding of these changes will increase our knowledge on the links between ecosystem dynamics and service delivery.

On the other hand, socio-ecological links must be accurately defined if we aim to achieve an integrated management approach and this requires appropriate methodological tools (Daily et al. 2009). The DPSIR framework (Drivers–Pressures–State change–Impact–Response) has been proposed as a systems-based approach that captures key relationships between society and the environment, note the tight link with the ecosystem services concept. Moreover, the DPSIR is regarded as a philosophical context for structuring and communicating policy-relevant research about the environment to non-scientists (Mangi et al. 2007, Atkins et al. 2011).

The principal aim of this study is to understand how commercial trawling compromises benthic ecosystem function performance and ultimately ecosystem services delivery. The specific goal is to assess the variability of ESPs under different environmental conditions and subjected to variable trawling intensity. We analyzed the functional components as surrogates for ecosystem functions of benthic communities from 13 sites located in fishing grounds in the Mediterranean that were subjected to different levels of trawling effort. In these fishing grounds, variability in BT could be linked to i) trawling activity, ii) local environmental factors or iii) area location. Through this approach we discuss if these factors drive ecosystem function variability in our study area and how could this ultimately compromise the delivery of ecosystem services. Finally, management implications for the variability of ecosystem services are discussed in a socio-ecological context under the DPSIR framework, highlighting the importance of benthic ecosystem services conservation to sustain healthy fisheries.

2. Materials and methods

2.1. Study areas and site characterisation

Six areas in the Mediterranean Sea were selected to carry out this study: 3 in the coast of Spain (Cap de Creus-CC, Ebre Delta-D and Cabo de Palos-CP), two in Italy (Ligurian Sea-L and Adriatic Sea-A) and one in Greece (Ionian Sea-I) (Fig.1). Four of the study areas (CC, CP, I, L) were further divided in 2-3 sites subjected to different levels of fishing effort (Low or no effort, Medium and High) (Fig.1). The sites were located on soft bottoms in the continental shelves between 40-80 m where commercial trawling is performed, but the CC area included a no-take Marine Protected Area (CCL). The L, I and D areas had muddy bottoms (over 90% of mud); CC and A areas had sandy-mud bottoms; and the CP area was characterized by maërl protruding within sandy-mud bottoms (de Juan et al. 2013, Table 1).

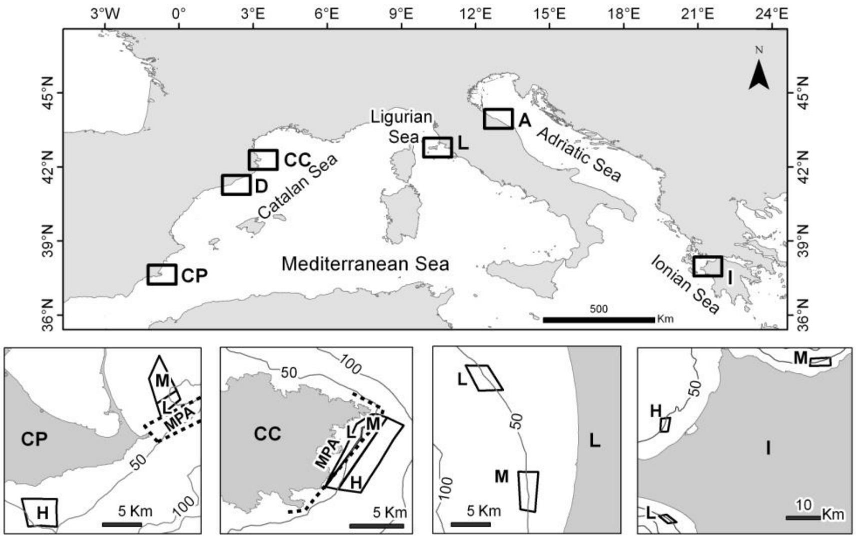

Figure 1.Map of the study areas. The detailed maps below include the sites with different fishing effort within each area. MPAs are bounded by stripped lines in CC and CP areas. CP: Cabo Palos, D: Delta, CC: Cap Creus, L: Ligurian Sea, A: Adriatic Sea, I: Ionian Sea. L: low, M: medium and H: high effort.

Fishing effort was estimated with two different approaches: fishermen interviews combined with registers from fisheries associations, and side-scan sonar surveys (SSS). The former allowed to estimate fishing effort as GT*days at sea/month and provided first-hand information of commercial trawling fleets operating in each site. The SSS approach allowed the estimation of trawl marks density on the seafloor considering their total length per surface within each site (trawl tracks density/km^2)(see de Juan et al. 2013 for further details) (Table 1).

Table 1. Summary of the variables recorded in the study sites. Fishing effort according to two different estimations: (FE1) fishermen interviews (GT*days at sea/month) and (FE2) Side Scan Sonar (trawl tracks density/km^2); environmental data: gravel, sand and mud percentages, D50 (μm), total organic carbon (TOC), total carbon (TC), bottom turbidity (mg/l), bottom temperature (Tº), bottom chlorophyll a (Chla-OBS), bottom oxygen (mg/l) and photo-synthetically active radiation (PAR).

Sites	FE1	FE2	%Gravel	%Sand	%Mud	D50	%TOC	%TC	Turbidity	Tº	Chl a	Oxygen	PAR
CC-L	0	22393	5.3	70.5	24.2	190.1	0.25	8.62	2.54	14.1	0.18	6.99	-
CC-M	1322	63171	0	14.38	85.62	14.47	0.55	3.79	2.37	13.9	0.21	7.01	-
CC-H	1623	81113	2.3	36.1	61.6	27.95	0.29	3.55	2.26	13.8	0.19	7.01	-
D	45125	116835	0	0.51	99.5	4.5	0.61	-	4	15.7	-	-	-
CP-L	900	48248	2.4	65.1	32.5	251.2	0.2	8.92	28.68	15.2	0.06	7.7	16.59
CP-M	14453	54199	2.4	65.1	32.5	251.2	0.2	8.92	28.21	15.1	0.04	7.65	12.66
CP-H	21966	137243	0	32.46	67.54	33.1	0.87	5.95	33.95	14.7	0.04	7.75	0.015
L-L	6256	87542	0	5.62	94.38	9.99	0.76	2.82	25.54	14.9	0.05	7.53	61.79
L-M	7930	96954	0	7.72	92.28	12.42	0.54	2.84	41.57	14.9	0.06	7.6	12.00
A	21823	32278	0	63.83	36.17	76.64	-	-	-	-	-	-	-
I-L	6216	91749	0	2.87	97.13	10.62	0.49	2.33	23.87	14.7	0.15	6.54	3.25
I-M	10833	102017	0	1.63	98.37	7.7	0.57	2.25	52.41	14.6	0.18	6.74	13.57
I-H	14146	122131	0	5.52	94.48	11.84	0.29	2.30	33.25	14.9	0.05	7.2	11.35

2.2. Data collection

All samples were collected in early summer (June-July) of 2003 for D and A and 2009 for the other areas, the sampling protocol being consistent across areas. Six samples of epifauna were randomly collected at each site with an epibenthic dredge (see de Juan et al. 2013 for further details). Only three replicates were collected at CCL site due to bad weather conditions. Epifaunal organisms were generally identified to species and the number of individuals and biomass for each species was recorded and standardized to $1000m^2$.

Three-five Van Veen grabs' were collected randomly in each site. The surface sediments from these grabs were retained for the analysis of grain size and carbon content. Grain size was analyzed with laser diffraction (using Horiba La-950v2 particle size analyser). Total carbon (TC) was analyzed using a TruSPec Leco analyser. After removing the inorganic carbon from the sediment sample using HCL vapors, the sample was reanalyzed to obtain the total organic carbon (TOC). These data were not available for A study area (Table 1).

Near-bottom oxygen, temperature, Chlorophyll a, Photo-synthetically Active Radiation (PAR) and turbidity values were available from CTD profiles in the sites from CC, CP, L and I areas. Turbidity was also available for D (Table 1).

2.3. Biological Trait Analysis

Fourteen biological traits covering different aspects of the organisms' biology were selected and broken down into categories (Table 2) (Bremner et al. 2003, Muntadas et al. 2014).

Following representativeness criteria, 83 out of 289 epifaunal species were selected. We selected those species recorded (a) in at least five of the six replicas or (b) in at least three replicates and with more than three individuals or (c) weighting more than 10g in at least one of the three replicas.

Each of these 83 taxa was scored for its affinity to trait categories following the "fuzzy scoring" method using a scale of 0–3 (0=no affinity to 3=high affinity)(see Bremner et al., 2003 for further details). This assignment was based on available literature, experts' knowledge and information from the BIOTIC database (http://www.marlin.ac.uk/biotic/). When no information on a particular trait was available, information for closely related taxa (genera or family) was considered (Bolam & Eggleton 2014). If no information was found, zero values were entered for each category and the taxon did not contribute to the calculation of trait weightings.

The frequency of the 14 traits' categories in the 13 sites was calculated by weighting the category scores by the abundance of each taxon exhibiting that category (Charvet et al. 1998). In the resulting sample by trait table, we detected high correlation between some of the categories as for example, habit, mobility and environmental position are generally related traits in the benthos (Norling et al. 2007, Frid

Table 2. Set of biological traits and categories used to describe epifaunal functional components. Underlined codes in brackets were used in Table 3. Traits and categories in bold were those removed in the reduced traits table (Table 2a) (see methods).

Biological trait	Categories
Environmental position	**Epibenthic** **Endobenthic**
Habit	**Attached (at)** **Bed forming** **Burrow dwelling** **Encrusted** **Erect (er)** **Free living**
Growth form	**Crustose soft** **Cushion** **Arborescent** **Vermiform** **Tubiculous** **Globose** **Turbinate (tr)** **Stellata** **Articulate** **Bivalved (bv)** **Pisciform**
Mobility	Swimmer Crawler Burrow Attachment
Bioturbation	**Not relevant** Surface Subsurface Deep to surface
Feeding mode	Deposit feeder Filter/Suspension feeder Opportunistic/Scavenger Predator Grazer
Size	Small (<5cm) Medium (5-10cm) Large (>10cm)

Table 2. Continuation

Biological trait	Categories
Fragility	**Fragile** **Intermediate** **Robust**
Regeneration potential	Yes **No**
Asexual reproduction	Yes **No**
Reproductive frequency	Continuous reproduction **1 reproductive event per year** 2 or more reproductive events per year
Type of larvae	**Brooding/Viviparous** **Lecitotrophic** **Planktotrophic**
Life span	<1yr 1-5yr >5yr
Sexual maturity	**<=1yr** >1yr

2011). Consequently, this table was explored for correlation among traits' categories and a reduced data set was designed trying to avoid high correlation ($\rho > 0.85$). To obtain this reduced traits' table (Table 3a) some of the original traits and categories were removed (in bold in Table 2), whereas other trait categories were combined to form new categories (highlighted in bold in Table 3a). The final sample by trait table used to perform the statistical analyses was calculated in the same way as for the original traits data set but using the reduced data set.

2.4. Data analysis

Similarity between each pair of samples was calculated using the Bray-Curtis index after a square root transformation of the data. Afterwards, similarity percentages within areas were obtained with the SIMPER routine using PRIMER 6 & PERMANOVA statistical package (Anderson et al. 2008).

To test for significant differences between sites and areas trait composition, we used multivariate generalized linear models (mGLM) with a negative binomial distribution ("mvabund" package in R, Wang et al., 2012). The multivariate test statistic was based on the likelihood ratio (LR) and an adjusted p-value was calculated for each trait with a step-down Monte Carlo resampling algorithm with 500 resamples. Traits with significant effects of sites, areas or their interaction term were selected for further analysis.

General Additive Models (GAM) were used to assess the effects of environmental variables on the selected traits and on all ESPs. The error structure was quasi-Poisson and the most parsimonious models were identified using the GCV scores (Wood & Augustin 2002). The environmental variables included in the analysis were: grain size (sand, mud and gravel percentages), TC and TOC content, bottom turbidity, oxygen, temperature, Chlorophyll a, PAR, and bathymetry (maximum, minimum and depth range for each sample). Fishing effort was also included in the model (data from fishermen interviews and from SSS). Prior to analysis, these variables were explored for significant inter-correlation in order to exclude from the models one of two variables with correlation higher than 0.85.

Table 3. a)Reduced set of biological traits (see methods),b)results of mGLM test on all areas except for CP and c)results of GAM best models for selected traits. % Dev, percentage of explained deviance by the selected model.*p<0.05,**p<0.01,***p<0.001 .“-“ no model was estimated;“+/-“ positive or negative effect,“s()” spline effect.

a) Reduced set of BT		b) MVabund			c) GAM		
Trait	**Categories**	**Area**	**Site**	**Area:site**	**Model**	**%Dev**	**Factors**
Mobility + bioturbation	Swimmer	ns	ns	ns	-	-	-
	Crawler	**	*	ns	***	64.6	s(mud)***, +effort *
	Burrow surface bioturbator	ns	ns	*	***	57.4	s(mud)***, -effort *
	Burrow subsurface bioturbator	ns	ns	ns	-	-	-
	Burrow deep to surface bioturbator	ns	ns	**	***	76.6	s(mud, effort)***
Feeding mode +mobility	Deposit feeder	ns	ns	ns	-	-	-
	Attached filter feeding	ns	ns	*	***	95.6	s(sss)***, s(oxigen)***, s(max depth)***
	Burrowing filter feeding	*	ns	ns	***	82.8	s(mud)***,+effort***
	Opportunistic/Scavenger	*	*	ns	***	59.8	s(mud)**, +effort*, -TOC*
	Predator	ns	ns	ns	-	-	-
	Grazer	*	ns	**	***	84.3	s(mud)***
Size	Small (<5cm)	**	*	ns	***	68.8	s(sss, mud)***
	Medium (5-10cm)	*	ns	ns	***	84.8	s(mud)***
	Large (>10cm)	ns	ns	ns	-	-	-
Regeneration	Yes	ns	ns	ns	-	-	-
Asexual reproduction	Yes	*	ns	**	**	66.7	s(mud)***, -sss***, +temp***
Reproductive frequency	Continuous reproduction	**	*	ns	***	72.4	s(mud)***
	2 or more reproductive events per year	ns	ns	*	***	61.8	s(sss, mud)***
Type of larvae	Brooding/Viviparous	**	ns	**	***	68.4	s(mud)**, s(sss)*
	Lecitotrophic	ns	ns	ns	-	-	-
	Planktotrophic	ns	ns	ns	-	-	-
Life span	=<1yr	ns	ns	**	**	86.7	s(mud)***
	>1yr	ns	ns	ns	-	-	-
Sexual maturity	>1yr	**	ns	ns	***	66.2	s(mud)***, +sss***

3. RESULTS

3.1. Functional variability across sites

According to the SIMPER analysis, samples in D site had the highest average within-area similarity regarding the trait abundance (90.3%), followed by samples in L (88.9%), in A(82.6%), in I (80%), in CC (76.8%) and finally samples in CP (60.4%) having the lowest similarity. Within-site similarity followed the same pattern.

The sample by trait table (Table 3a) for all sites was explored with mGLM analysis and no significant differences were detected for functional traits' overall abundance between areas or sites, or their interaction. The CP data (with the lowest within-area similarity) had high variability among samples and this high variability could mask differences between areas. The analysis was rerun removing all the CP samples and results showed significant differences for Area (p=0.002), Site (p=0.006) and Area:site (p=0.002). Post-hoc analysis for individual traits showed no consistent pattern (Table 3b).

GAM analyses were performed on the traits' categories that evidenced significant differences between areas and/or sites (results from the mGLM analysis, Table 3b). Mud had a significant non-linear effect on all these traits' categories, except for "attached filter feeding" (Table 3c). Fishing effort also had a significant effect, either linear or non-linear, on most traits' categories. Other environmental variables had also significant effects on some traits' categories: positive effect of temperature on "asexual reproduction", negative effect of TOC on "opportunistic/scavenger" and oxygen and maximum depth had non-linear effects on "attached filter feeding".

3.2. Link between BTs, ecosystem function and ESPs

In order to explore the performance of the ecosystem functions production, nutrient cycling, bentho-pelagic coupling, carbon sequestration and habitat structure in our study sites, 17 ESPs were defined as a combination of traits' categories (Table 2) known to be linked to these functions (Table 4).

For instance, the traits considered to be linked to the nutrient cycling function were "size", "bioturbation" and "feeding mode" and four related ESPs were designed: high, moderate and small-scale bioturbators and diffuse sediment mixers. As an example of ESP definition, small burrowing species, either surface or subsurface bioturbators, and having other feeding type than deposit feeding (i.e. *Centrocardita aculeata* or *Lesuerigobius suerii,* filter feeder and predator respectively) were classified as "diffuse sediment mixers" (Table 4).

Abundance and biomass of taxa exhibiting the traits assigned to each ESP were added obtaining an "ESP abundance/biomass per sample" table. Following this approach, we estimated both the quantity and weight of components contributing to a particular ecosystem function. The general ecosystem service category to which a specific function contributes is provided in Table 3 (categories as described in MEA 2005).

Table 4. a) Reduced set of biological traits (see methods), b) results of mGLM test on all areas except for CP and c)results of GAM best models for selected traits. % Dev, percentage of explained deviance by the selected model.*p<0.05,**p<0.01,***p<0.001 .“-“ no model was estimated;“+/-“ positive or negative effect,“s” spline effect.

ECOSYSTEM FUNCTION	RELATED BIOLOGICAL TRAITS	REFERENCES	ESP type	SPECIES EXAMPLE	ECOSYSTEM SERVICE TYPE
Production	Size Life span Reproductive frequency Sexual maturity	Robertson, 1979 Jennings et al., 2001	T1: Very fast turnover	*Chlamys opercularis* *Alpheus glaber*	**Provisioning**
			T2: Fast turnover	*Liocarcinus depurator* *Clausinella brogniartii*	
			T3: Medium turnover	*Bolinus brandaris* *Anseropoda placenta*	
			T4: Slow turnover	*Ascidia mentula* *Squilla mantis*	
			T5: Very slow turnover	*Dercitus bucklandi* *Leptogorgia sarmentosa*	
Nutrient cycling	Size Bioturbation Feeding mode	Lohrer et al., 2004 Thrush and Dayton, 2010	T6: High bioturbators	*Astropecten aranciacus* *Squilla mantis*	**Regulating**
			T7: Moderarte bioturbators	*Anseropoda placenta* *Solenocera membranmacea*	
			T8: Small-scale bioturbators	*Alpheus glaber* *Nucula sp.*	
			T9: Diffuse sediment mixers	*Centrocardita aculeata* *Lesuerigobius suerii*	
Bentho-pelagic coupling	Size Feeding type	Bremner et al., 2006a Snelgrove, 1997	T10: Large filter feeders	*Alcyonium palmatum* *Ascidia mentula*	**Regulating**
			T11: Medium filter feeders	*Ocnus planci* *Acanthocardia echinata*	
			T12: Small filter feeders	*Ascidiella scabra* *Centrocardita aculeata*	
Carbon sequestration	Size Life span Growth form (tr and bv)	Bremner et al., 2006a Frid, 2011	T13:Large or moderate carbon sequestration	*Pecten jacobeus* *Ostrea edulis*	**Regulating**
			T14: Little carbon sequestration	*Acanthocardia paucicostata* *Bolinus brandaria*	
Habitat structure	Size Life span Habit (er and at)	Thrush and Dayton, 2010 Bremner et al., 2006a	T15: Large 3D structure	*Alcyonium palmatum* *Leptogorgia sarmentosa*	**Supporting**
			T16: Moderate 3D structure	*Ascidia mentula* *Ostrea edulis*	
			T17: Simple structure	*Ascidiella scabra* *Sertella beaniana*	

3.3. Ecosystem function performance: distribution of ESPs across sites

ESPs regarding to production, bentho-pelagic coupling and habitat structure were numerically dominant in CP sites (Fig.2). Organisms performing high bioturbation (ESP type 6) dominated the nutrient cycling function in all areas except in CP and CC, where smaller-scale bioturbation prevailed. Carbon sequestration was almost absent from A area. Otherwise, organisms linked to little carbon sequestration (ESP

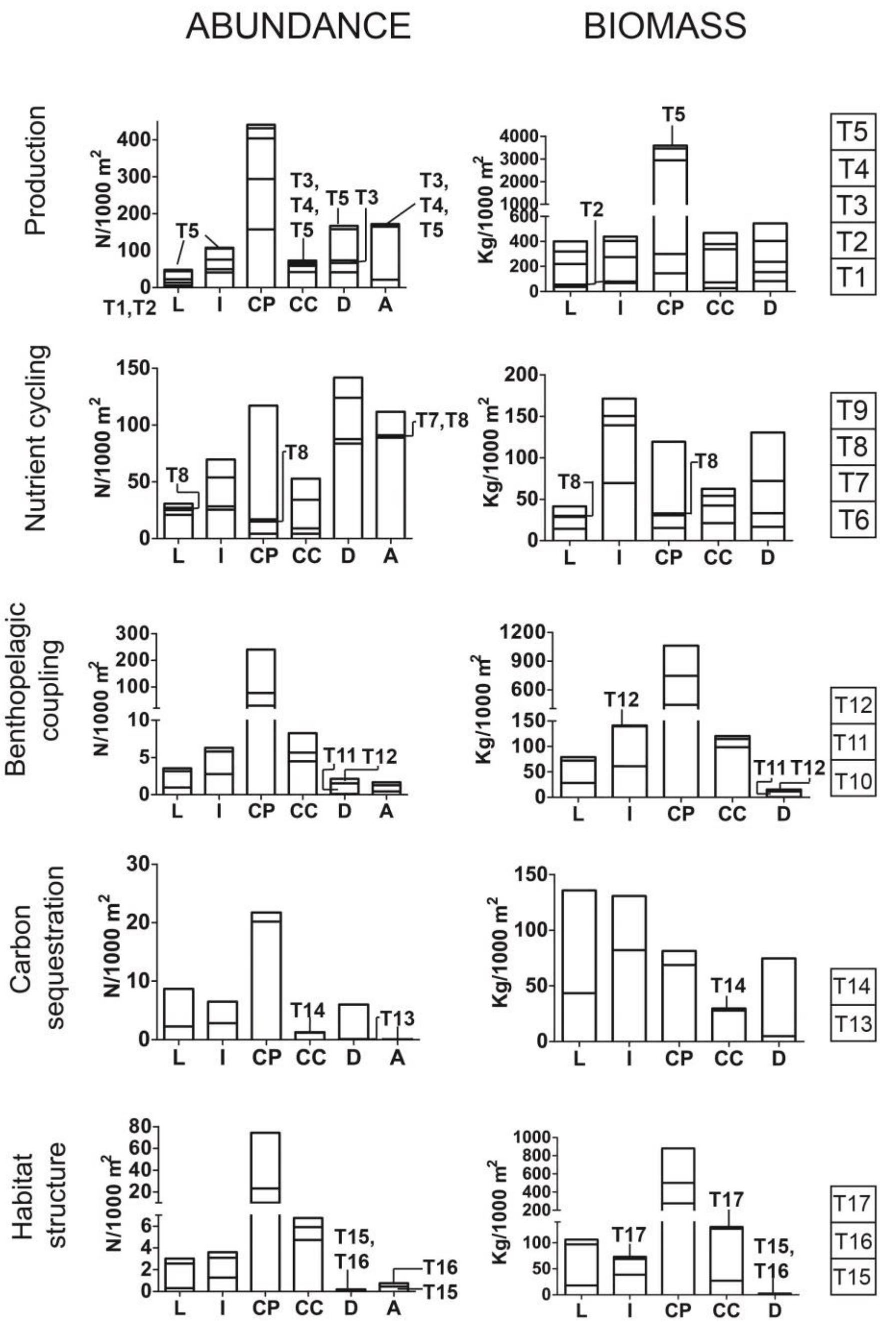

Figure 2.Accumulated abundance and biomass for epifaunal ESPs related with each ecosystem function across areas. Right bars indicate the order in which each ESP type is placed in the graphics, which is also written into the graphic when low values make it difficult to interpret. ESP codes follow those in Table 4.

type 14) dominated in L, I and especially in D area. Organisms linked to large/moderate carbon sequestration (ESP type 13) were numerically more abundant in CP (Fig.2). Biomass estimations followed similar patterns, although generally large ESPs gained importance (e.g. ESP type 5 or ESP type 10). However, biomass patterns for carbon sequestration changed compared to that shown by abundance data. While CP area stands out in the abundance estimation, L and I areas surpassed CP area in weight. For this function, it is also worth to note the high importance of little sequestration organisms (ESP type 14) in L, I and D areas when considering their weight.

The abundance of the ESPs with larger contribution to the function provision was consistent across sites within the same area except for large carbon sequestration in CP, which were more abundant in CPL and CPM than in CPH (Fig.3). The relative importance of function abundance across sites was the same as in figure 2, but in figure 3 the importance of high-rate bioturbators in D and A areas was more evident. The environmental variables having the strongest effects on ESPs were sediment and fishing effort, which played a role in almost all degrees of function performance (Table 5). For the biomass estimations, only in four cases none of these variables were included in the model: very slow turnover, moderate and small-scale bioturbation and important 3D structure. Other variables such as maximum depth, temperature, TOC, and turbidity also affected the organism types' distribution (Table 5). Results were slightly different depending on whether we considered abundance or biomass; for instance, large filter feeders (ESP type 10) were affected by sediment variables but not by effort if considering abundance, while they were affected by effort but not by sediment variables if considering biomass.

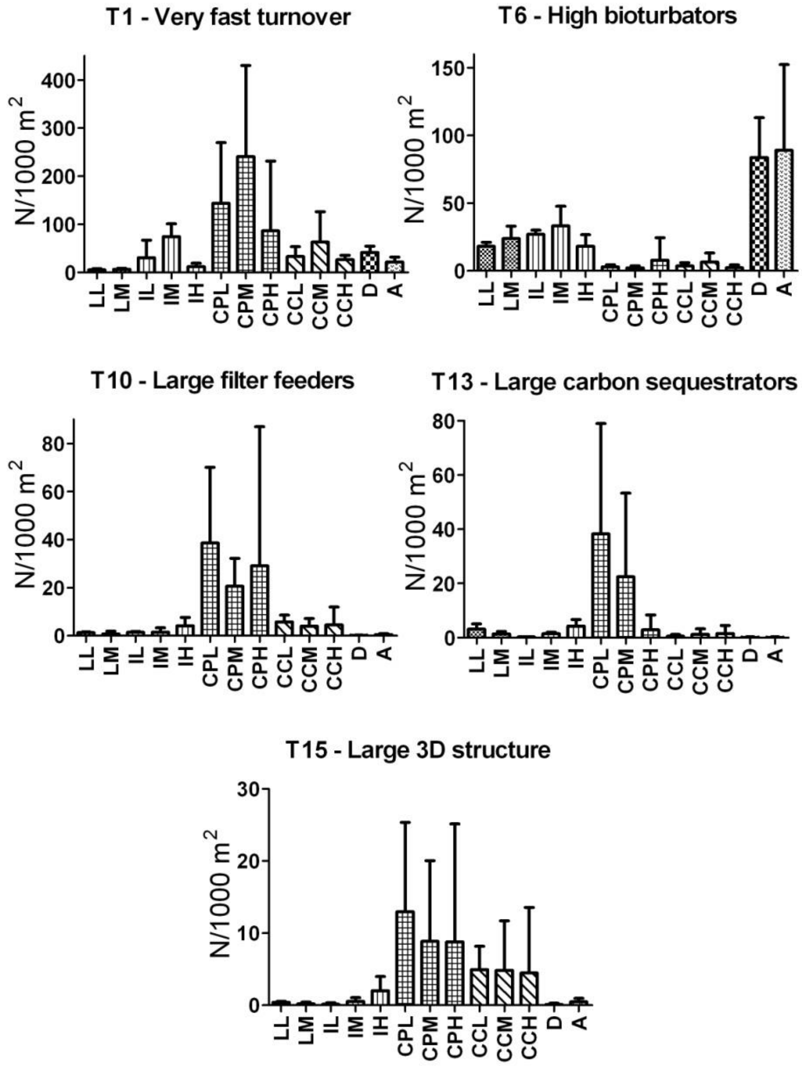

Figure 3.Abundance of the main ESPs across sites. Letters following the site names stand for the fishing effort levels: L:low; M:medium and H:high

4. DISCUSSION

4.1. From benthic functionality to ecosystem services delivery

Understanding ecosystem functionality and the external factors that modulate it is the first step towards understanding the mechanisms that govern the delivery of ecosystem services (Townsend et al. 2011). A number of studies have focused on benthic functionality in fishing grounds (e.g. Bremner et al. 2003, de Juan et al. 2007, Tillin et al. 2006) and, in accordance with them, our results evidenced that epifaunal functional composition from soft-bottom fishing grounds was highly conditioned by trawling intensity (Table 3c). Previous studies, also conducted in trawling grounds, did not find consistent relationship between sediment type and functional assemblages (Bremner et al., 2006b, Tillin et al. 2006), although Barberá et al. (2012), who studied a range of habitats including maërl biocenosis, found a significant correlation between some functional components and the mud content in sediments. Our study, which included a variety of soft sediments (from bare mud to maërl beds) showed significant effects of sediment on functional composition, evidencing the importance of multi-site studies that encompass habitat variability.

This sediment dependence was evident in our data as CP samples, with heterogeneous sediments (de Juan et al. 2013), had high within-site variability which masked statistical differences among the other more homogeneous areas. It is noteworthy to point out that sediment generally had non-linear effects on functional components that are difficult to extrapolate into a general cause-effect mechanism.

Table 5. Results of GAM for selected ESPs a) abundance and b) biomass. *p<0.05, **p<0.01,***p<0.001."+/-" positive or negative effect,"s" spline effect. ESP type codes follow those on Table 4.

ESP type	a) Abundance		
	Model	%Dev	Factors
T1	***	60.9	s(sss)***
T2	***	72.5	s(sss)***
T3	***	64.5	D50:sss***, s(max.depth)**
T4	***	56.9	s(effort)**, -max. depth***, turbidity**
T5	***	59.8	s(sss)***, s(Tº)***
T6	***	90.2	s(mud)***, turbidity**, effort***
T7	***	12.6	-mud:TOC**
T8	**	81.7	s(mud)***, –TOC***
T9	***	72	s(mud)***, -sss***
T10	**	85	s(oxygen)***, s(mud)***, -Tº*
T11	***	45.02	-mud***, Tº***
T12	**	81	s(mud)***, -effort**
T13	*	77.4	s(sss)***, s(turbidity)**
T14	***	53.7	s(max.depth)***, D50**
T15	*	46.8	s(D50, sss)***
T16	*	68.2	s(sss)***
T17	***	89.6	s(max. depth)**, sss**, -mud***

Table 5. Continuation

ESP type	b) Biomass		
	Model	%Dev	Factors
T1	**	57.2	s(D50)***, s(turbidity)**, -Tº*
T2	***	57.4	s(effort)*, s(D50)**
T3	***	47.9	s(mud)***
T4	***	48.5	s(D50)***, +sss**
T5	***	61	s(PAR)***, -turbidity**
T6	***	62.8	s(sss)**, s(turbidity)***
T7	***	44.2	s(max.depth)**, +turbidity,***, TOC**
T8	**	64.3	s(max.turbidity)**, +max.depth*
T9	**	72.1	s(effort)***,s(D50)***, -Tº*,+ turbidity**
T10	***	59.7	s(effort)***, +oxygen***, TOC**
T11	***	52.9	s(D50)***
T12	***	87.2	s(D50)**, s(turbidity)***
T13	***	42.6	s(Tº)*,-TOC*,-effort**
T14	***	72.9	s(SSS)***
T15	***	59.9	s(TOC)*, +max depth**, +Tº***
T16	***	64.7	s(TOC)***, s(effort)*
T17	***	80.5	s(D50)***

When scaling up from ecosystem functional components (BTs) to function provision (ESPs), other variables besides effort and sediment characteristics became significant. This suggests an increasing complexity of ecological interactions, which has important implications for the estimation of service provision. However, whatever metric is chosen, generally fishing effort had a significant effect on the function performance (Table 5). These are important findings as fishing intensity is the only variable susceptible to be managed in our case study, meaning that a change on fishing effort would affect function performance and hence the overall ecosystem service delivery.

ESPs abundance patterns were consistent across sites within an area, suggesting that sediment characteristics favour the settlement of particular ESP. These patterns could hinder the trawling effects assessment when dealing with a wide range of habitats. That is, trawling will certainly affect ESPs (Table 5), but effects will depend on the environmental context, e.g. the initial effects of trawling would be different in the maërl grounds in CP compared to the muddy bottoms of D (Kaiser et al. 2006).

The relative importance of each ESP was different among areas depending on the chosen parameter (abundance or biomass) (Fig.2). As expected, ESPs related to the trait “large size” generally gained relative importance when assessing function weight, but, in some cases, biomass patterns were also conditioned by some “small size” ESPs. For example, in L and D areas, biomass of the ESP “little carbon sequestration” was more prevalent than “large carbon sequestration”.

Therefore, for this particular case, the question remains, does a large quantity of small organisms and lower quantity of large organisms equally contribute to the carbon sequestration? However, to actually address this question, functions should be quantitatively measured on different scenarios (e.g. rates of nutrient cycling, carbon retention, etc. (Lohrer et al. 2004, Ståhl et al. 2004)). In this particular case, the species that contributed more to little carbon sequestration was *Bolinus brandaris*, a medium-sized and medium-length life span gastropod common in Mediterranean trawled soft-bottoms (Martín et al. 1995). In these communities, the pooled biomass of this species might partially counteract the lack of large long-lived bivalves to perform carbon sequestration. However, to measure carbon sequestration, turnover rates should be also taken into account, which is generally difficult to estimate due to the lack of life-history traits information for many species (Bolam & Eggleton 2014).

Medium-size organisms may also gain importance with regards to large organisms, as happened for habitat structure in CC area. In this case, the ESP “large 3D structure” comprised a large number of gorgonians, which were lighter than the ascidians and sponges involved in the “moderate 3D structure” ESP. However, higher biomass of medium size organisms did not mean an increase in habitat structure, as smaller organisms do not provide the same 3D structure. As Bolam & Eggleton (2014) already pointed out, these results highlight the importance of the chosen metric in assessing function performance, and the need to evaluate each function separately.

4.2. ESP as an indicator of good management practices

Consideration of ESP and how they will be affected by different activities allows a better linkage between the final services people perceive and supporting ecosystem functions that may otherwise be ignored in decision-making (Townsend et al. 2011, Rees et al. 2012). But a single ecosystem function usually links with many services (Townsend et al. 2011) and ecosystem functions and services are interrelated creating a matrix of trade-offs and synergies (Snelgrove et al. 2014). For example, in our case study, nutrient cycling and bentho-pelagic coupling, that enhance the nutrient exchange between sediment and water column making nutrients available for primary producers, might negatively interact (i.e. a trade-off): the ESP with larger contribution to the function nutrient cycling (i.e. high bioturbators) was more abundant in muddy areas subjected to high-moderate fishing pressure (L, I, D, A), whereas it was almost absent from the sandiest areas CC and CP. On the other hand, bentho-pelagic coupling prevailed in these sandy-gravel areas, CC and CP. Despite the sediment differences, the sites within CC and CP (except for CPH) held relatively lower fishing intensity compared to the other sites. Hence, fishing effort might impose a trade-off between bentho-pelagic coupling and nutrient cycling in our study areas (note that the two BT used to describe the ESP contributing more to the bentho-pelagic coupling function, “large” and “filter feeding”, are very vulnerable to trawling (Thrush & Dayton 2002)). The challenge for management is to deal with these trade-offs and synergies, considering appropriate spatial and temporal scales and tightening the beneficial loops to ensure ecosystem service provisioning (Levin & Lubchenco 2008, Daily et al. 2009).

ESP approach based on BTA provides a promising framework to assess ecosystem functions as a first step to scale up to ecosystem services. Furthermore, ecosystem services can be integrated into a DPSIR framework which might help to understand how benthic ecosystems and management decisions interact and what is the role of society in this relationship (Mangi et al. 2007, Atkins et al. 2011).

In the present work, the DPSIR framework has been adopted in order to illustrate all the actors involved in an integral ecosystem management for trawling fisheries. Using this tool, we aimed to summarize in an understandable way the complex socio-ecological interactions occurring in trawling grounds, in which the society plays a very important role towards the achievement of a Good Environmental Status (Fig.4). The different DPSIR elements were considered as follows:

Drivers: We considered society to act as a main *Driver* in the fisheries context through its food demand and/or its willingness to preserve marine ecosystems. In this context, society may act as an interest group that will influence management practices through feedback processes and it will push managers to implement those measures to better fit its interests. In this way, society has an indirect but important effect on the final ecological ecosystem status.

Pressures: Society need for food will pose a *Pressure* (trawling activities) on marine environment which can be divided in two "sub-pressures" that will affect different ecosystem components: catches and gear effects.

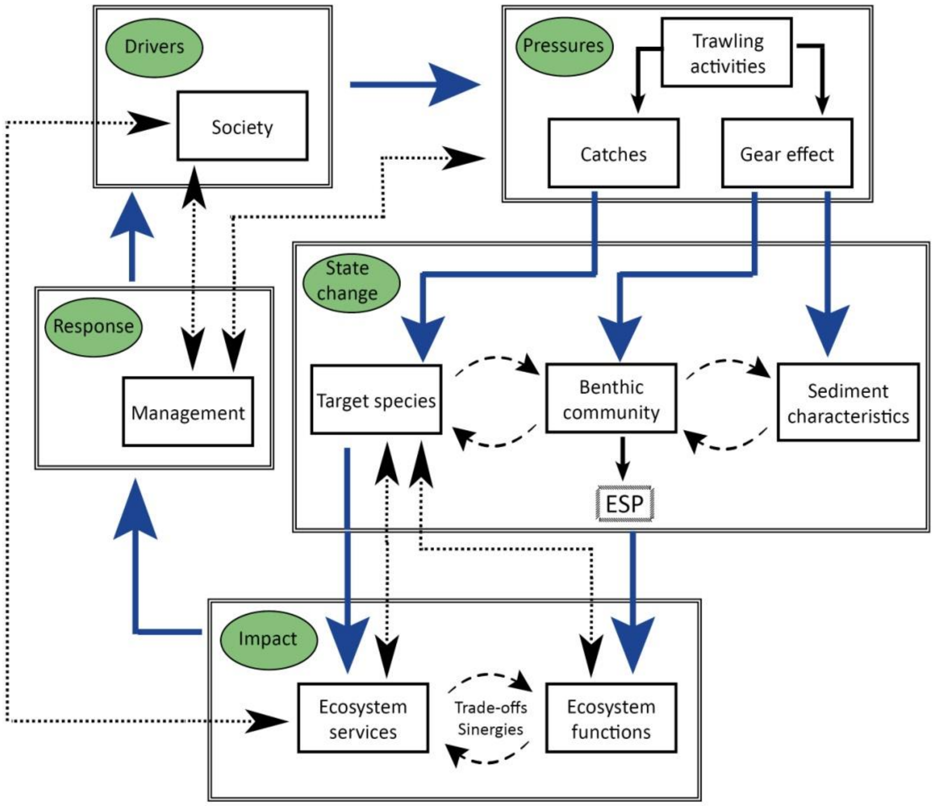

Figure 4.DPSIR framework adapted to the trawling fisheries context. Each double-lined box corresponds to one of the DPSIR elements which may include one or several ecosystem/society components (little boxes inside). Blue arrows represent DPSIR effects while black dashed arrows between double-lined boxes represent feedbacks between ecosystem/society components. Dark dashed arrows within double-lined boxes represent relationships among the elements of that particular DPSIR element. See the text for a detailed description.

State change: The sub-pressure "catches" will lead to a *State change* on target species and the sub-pressure "gear effect" will affect the seabed and non-target communities, resulting on a *State change* of benthic community structure and composition as well as environmental characteristics. It is important to bear in mind that all these ecosystem components (target species, benthic communities and sediments) are interrelated, hence, the direct changes caused by trawling may also indirectly affect the other components (Muntadas et al. 2014).

Impact: *State change* of all ecosystem components will subsequently affect ESPs, causing an *Impact* on ecosystem functions and, hence, services delivery. Moreover, *State change* on target species will directly have an *Impact* on the service “fish for food”. As mentioned above, it is also important to take into account synergies and trade-offs among ecosystem functions and services, which may modulate their performance (Snelgrove et al. 2014). Note that impacts on ecosystem functions and services may also feedback, generally in a negative way, on target species (Muntadas et al. 2014).

Response: The *Impact* on ecosystem services delivery may lead to a management *Response* to change the *Pressure,* e.g. change of fishing effort levels (Fig.4). This response may be driven by a society demand as it perceives that it is no longer benefiting from such ecosystem services. However, this management *Response* through effort level change could be (a) an effort increase aiming to maintain the catches, which would represent a decrease on the CPUE (Catch per Unit Effort) (Sardà 1998b), or (b) an effort decrease aiming to improve habitat condition and maintaining exploitation in a sustainable level. Consequently, as we assume that society pressures would be the *Driver* of this management change, it is important to arise public awareness of the key role benthic communities’ play in ensuring service delivery and the importance of the maintenance of non-economic services (e.g. habitat provision) for the delivery of economic services such as “fish for food”.

5. Conclusions

We propose an approach to estimate ecosystem functions through ESPs, focusing on the ecosystem components rather than on economic value of a trawled system. This approach is meant to understand benthic processes as a previous step to scale up to ecosystem service assessment. We focused our analysis on fishing grounds in which, as a direct benefit for society, the most evident delivered service is commercial catches. However, this is not the only service these ecosystems provide and, in fact, this service relies on other ecosystem functions whose improvement could actually benefit fish production, e.g. the habitat structure improvement directly benefits demersal fish stocks. Trawling implies a trade-off between food provision and other services, as it negatively affects key species involved in ecosystem functions such as bioturbation, habitat structure or carbon storage (e.g. lack of large carbon sequestration in trawled areas). At this point, scientists need to arise society awareness, in order to adopt good management practices that ensure fisheries production as well as other services that directly or indirectly benefit the society. Furthermore, in order to ensure the provision of benthic ecosystem services, relationships between ecosystems and society must be defined in an understandable way so as to appropriately inform managers. To this aim, we adapted a DPSIR framework to our case study summarizing the main socio-ecological relationships between ecosystem functions and society which highlights the need for an integrated management approach in order to ensure ecosystem services provision.

Acknowledgements:

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Avaluació de la redundància funcional en comunitats bentòniques sotmeses a la pesca d'arrossegament

RESUM:

La redundància funcional, és a dir, espècies diferents que realitzen la mateixa funció, és un dels aspectes a tenir en compte quan s'avalua la resiliència dels ecosistemes. La redundància pot jugar un paper important en el manteniment de l'estructura i la funcionalitat ecosistèmica quan la comunitat es veu sotmesa a una pressió (p.ex. la pesca d'arrossegament), ja que dues espècies funcionalment equivalents poden respondre de manera diferent a l'estrès i la disminució d'una es pot veure compensada pel manteniment o l'augment de l'altra. En aquest context, les espècies rares (poc abundants en l'ecosistema) poden adquirir una especial importància, ja que contribuirien a incrementar el pool de funcions augmentant d'aquesta manera la redundància funcional del sistema.

En aquest treball proposem dues maneres diferents de mesurar la redundància en les comunitats bentòniques, totes dues basades en una aproximació mitjançant el trets biològics de les espècies:

a) Trets comuns: la redundància funcional d'un tret es mesura a partir de la seva abundància, és a dir, un tret és redundant quan està present en la comunitat en "grans quantitats".

b) Trets generalitzats: la redundància funcional d'un tret es mesura a partir del número d'espècies que presenten aquell tret.

L'objectiu d'aquest treball era el d'avaluar el resultat d'aplicar aquestes dues mesures en comunitats bentòniques de fons tous sotmesos a diferent intensitat de pesca d'arrossegament i determinar si aquestes dues mesures es veuen afectades de manera diferent per aquesta activitat. Paral·lelament, també es va analitzar la influència de les espècies rares en les dues mesures de redundància.

Aquestes dues mesures es van aplicar a les comunitats d'infauna i epifauna de 8 àrees del Mediterrani sotmeses a diferent nivell d'esforç pesquer. Els trets biològics escollits per a determinar-ne la redundància (mesures indirectes de la funcionalitat de l'ecosistema) estan relacionats amb funcions clau en aquests ecosistemes com ara el reciclatge de nutrients, l'estabilitat del sediment, l'acoblament bento-pelàgic i la provisió d'hàbitat. A més a més, per tal de determinar la influència de les espècies rares en la redundància, les dues mesures es van estimar tenint en compte la llista completa de taxons determinats en cada comunitat i tenint en compte només una selecció dels taxons més abundants.

Els resultats van determinar que el nombre de trets redundants era significativament diferent entre les diferents àrees d´estudi, però la identitat dels trets redundants a les diferents àrees era bàsicament la mateixa. Per altra banda, es van detectar diferencies al comparar les dues mesures de trets: a) en general més trets eren considerats redundants al considerar la mesura dels trets generalitzats i b) alguns trets que eren considerats redundants segons la mesura de trets generalitzats no ho eren considerant la mesura dels trets comuns i vice-versa. Aquests resultats mostren que la informació que aporten les dues mesures és complementària, de manera que si només se'n té en compte una es pot arribar a conclusions errònies sobre la redundància funcional del sistema. En resum, les dues mesures s'haurien d'utilitzar conjuntament per tal d'obtenir una visó global de la redundància de la comunitat.

Per altra banda, els resultats mostraren clarament que la mesura de trets comuns estava significativament afectada per la pesca d'arrossegament, mentre que els resultats no foren tan clars per la mesura de trets generalitzats. No obstant això, cal tenir present que les comunitat estudiades representen comunitats afectades històricament per la pesca i que per tant els trets més vulnerables hi estarien presents tan sols en quantitats molt reduïdes, quedant només aquells trets més resistents (p.ex. carronyaires o espècies mòbils). Aquest context de l'estudi podria explicar perquè no es detecta un efecte significatiu de la pesca d'arrossegament en la mesura dels trets generalitzats, ja que el que estaríem observant és l'efecte de diferents nivells de pesca sobre l'abundància d'aquests organismes més resistents. Així doncs, malgrat els resultats, no podem obviar la mesura de trets generalitzats.

Finalment, es va trobar que en les comunitats estudiades les espècies rares no afecten el nombre total de trets redundants i també es va comprovar que aquestes espècies presenten fonamentalment els mateixos trets que les espècies més abundants. Això permet concloure que per tal de simplificar el màxim possible les mesures de redundància, en el nostre cas d'estudi només caldria tenir en compte les espècies més abundants.

Assessing functional redundancy in chronically trawled benthic communities

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HIGHLIGHTS

- Functional redundancy was assessed in soft-bottom communities subjected to trawling.
- Redundancy was measured as traits abundance and trait richness using a BTA.
- A combination of the two measures should be used in a redundancy assessment.
- The role of rare species in both measures was also studied.

ABSTRACT

Fishing disturbance on ecosystems leads to changes in community structure and composition, which may have drastic implications for ecosystem functional performance. Functional redundancy, defined as species sharing similar functional roles, is a community property that plays an important role in preventing functional changes in ecosystems under pressure. In this study, we suggest that functional redundancy may be achieved through trait abundance (i.e. large amounts of a trait, hereafter "common traits"), or through trait richness (i.e. large numbers of distinct taxa exhibiting the same trait, hereafter “widespread traits”). We assessed the variability of both measures obtained from epifaunal and infaunal communities in soft-bottom trawling grounds. Sampling sites were located in four Mediterranean areas that were subjected to different levels of trawling effort. Common and Widespread traits measures were based on the analysis of biological traits linked to key soft-bottoms functions such as nutrient cycling, bentho-pelagic coupling and habitat provision. The role of rare species in both measures was also assessed and we observed that, in our study sites, rare species generally exhibited the same traits as the most abundant species. Common and Widespread traits measures provided complementary information on benthic functional redundancy. Thus, we suggest that a combination of the two measures should be used to appropriately assess benthic functional redundancy in trawling grounds. Redundancy being a component of ecosystem resilience, this functional redundancy evaluation is important to assess ecosystem integrity.

KEY WORDS: BTA, trawling, functionality, resilience, soft-bottoms, Mediterranean

1. Introduction

Functional traits composition is strongly linked with ecosystem processes and services, as species’ functional traits are responsible for functional performance (Díaz & Cabido 2001, Heemsbergen et al. 2004, Kremen & Ostfeld 2005, Muntadas et al. 2015). Several studies have demonstrated that ecosystem functional components may change in response to natural and anthropogenic sources of stress, indirectly causing variability of ecosystem processes (Larsen et al. 2005, Mouillot et al. 2013, Suding et al. 2008). Moreover, previous studies proved that ecosystem functional components can be effectively assessed through a biological trait approach (BTA) (de Juan et al. 2007, Bremner et al. 2003, Bremner 2008). Consequently, information on communities' trait composition and structure may be used in bio-monitoring as a proxy to assess the ecosystem status (de Juan & Demestre 2012, de Juan et al. 2014, Menezes et al. 2010).

Disturbance on ecosystems may lead to a significant reduction in species densities and ultimately to species' extinction, which could cause a functional extinction if the species´ functional role in the ecosystem is unique (Andersen et al. 2011). Fonseca & Ganade (2001) estimated that 75% of the species in a community might be lost before the first functional group disappears, but they assumed random extinction that would rarely be the case in nature, as pressures tend to principally affect certain species or certain traits (Martins et al. 2012, McIntyre et al. 2007, Schweiger et al. 2007). For instance, fisheries principally target large predators, drastically reducing this functional group (Pauly 1998), and trawl fishing is known to remove large and sessile filter feeding species (Collie et al. 1997, de Juan et al. 2009). These responses have important consequences for ecosystem functionality, as the order in which species are lost would ultimately govern the ecosystem functional performance (McIntyre et al. 2007, Solan et al. 2004).

Redundancy, understood as different species sharing similar functional roles, would provide an insurance for maintaining functional diversity of biological communities in ecosystems subjected to stress (Lavorel & Garnier 2002, Walker 1992, Yachi & Loreau 1999). Accordingly, functional redundancy has been related to ecosystem resilience, i.e. the capacity of a system to retain essentially the same function, structure, identity, and feedbacks when facing a disturbance (Folke et al. 2004, Walker et al. 1999). But, for that to be true, redundancy should to be defined as "functionally equivalent species having different responses to stress" (Bremner 2008, Folke et al. 2004, Fonseca & Ganade 2001). If a set of species have similar functional traits but the species are equally vulnerable to disturbance, functional redundancy does not increase the resilience of that function (Hughes et al. 2005, Schweiger et al. 2007). Moreover, the magnitude of functional redundancy also depends on how functional traits interact with the environment (Murray et al. 2014). Hence, high redundancy does not always directly translate to a higher resilience of ecosystem functions.

Despite species richness estimates are essential for diversity maintenance and ecosystem stability (Díaz & Cabido 2001, Tilman & Downing 1994), species diversity per se is not an indicator for functional redundancy (Folke et al. 2004, Guillemot et al. 2011, Hughes et al. 2005). However, species composition and not only abundance might be critical, as rare species can play an important role in redundancy as they may be functionally equivalent to dominant species but may show different responses to stress (Boero 1994, Walker et al. 1999). In this context, rare species become important for the maintenance of functions in disturbed ecosystems, where the abundance of common species might have been depleted (Boero 1994, Lyons et al. 2005, Walker et al. 1999). These studies highlight the importance of testing the role of rare species in stressed systems (Ellingsen et al. 2007).

One of the most harmful pressures on marine ecosystems is fishing, especially trawling on benthic ecosystems (Thrush & Dayton 2002). Functional assemblages of benthic ecosystems are modified by trawling (Bremner et al. 2003, de Juan et al. 2007, Fleddum et al. 2013) and removal of target species leads to non-random depletion of particular functions (Martins et al. 2012). These effects, linked to low species redundancy found in some marine assemblages, may lead to functional extinction (Guillemot et al. 2011, Micheli & Halpern 2005). However, trawling is known to favour certain traits such as scavenging or high motility, while decreases vulnerable traits like sessile and filter feeding (de Juan et al. 2007, Jennings & Kaiser 1998, Thrush & Dayton 2002). Thus, in chronically trawled areas redundancy of particular resistant traits might increase because only species that share those traits would survive. Otherwise, biological traits that are vulnerable to trawling might become extinct.

In this work, we studied the functional redundancy of benthic communities (epifauna and infauna) from eight sites subjected to different intensities of commercial trawling in Mediterranean continental shelves. The functional redundancy was estimated with two different measures based on a BTA approach. We propose that functional redundancy might be achieved through trait abundance (i.e. large amounts of a trait), hereafter called “common traits”, or through high trait richness (i.e. large numbers of distinct taxa possessing the same trait), hereafter called “widespread traits”. Using these two metrics, we evaluated how trawling effort levels affected the patterns of trait redundancy in our study sites. Moreover, as the performance of these two metrics will be related with their response to trawling disturbance, we tested if common and widespread traits were differently affected by trawling effort. We also studied the influence of rare species on the two redundancy measures to evaluate whether it is necessary to include the whole community in the assessment, or a species subset would be enough in order to favour a rapid assessment. Finally, we discuss how these two measures might be used as a proxy to assess the resilience of key functions in soft-bottom habitats subjected to chronic fishing disturbance, such as nutrient cycling, sediment stability, benthopelagic coupling and habitat provision (Bremner et al. 2006, Muntadas et al. 2015, Snelgrove 1997).

2. Materials and methods

2.1. Study areas

Four study areas were selected in the Mediterranean: two in the north-eastern coast of the Iberian Peninsula (Cap de Creus-CC and Ebre delta-D), one in the north-western coast of Italy (Ligurian coast-L) and one in the south-western coast of Greece (Ionian coast-I) (Fig. 1). The four areas were located on soft-bottoms, around 40-80 m depth, where commercial trawling is performed. The L, I and D areas had muddy bottoms (99.5%, 96.6% and 93.3% of mud respectively), while CC area had a higher sand content (40.3% of sand). Fishing effort was qualitatively characterized in the study areas after conducting fishermen interviews that allowed us to further subdivide each area in several sites with different levels of fishing effort: L: low, M: medium and H: high in I area; L and M in L area; M and H in CC area; and H in D area (Fig. 1) (see Muntadas et al. 2015 for further details of the methodology adopted). In total, we studied eight sites subjected to variable trawling disturbance. Fishing effort in each site was further assessed with Side Scan Sonar (SSS), which allowed the estimation of trawl marks´ density on the seafloor by quantifying their total length per surveyed surface within each site (trawl tracks density/km^2) (see de Juan et al. 2013 for further details).

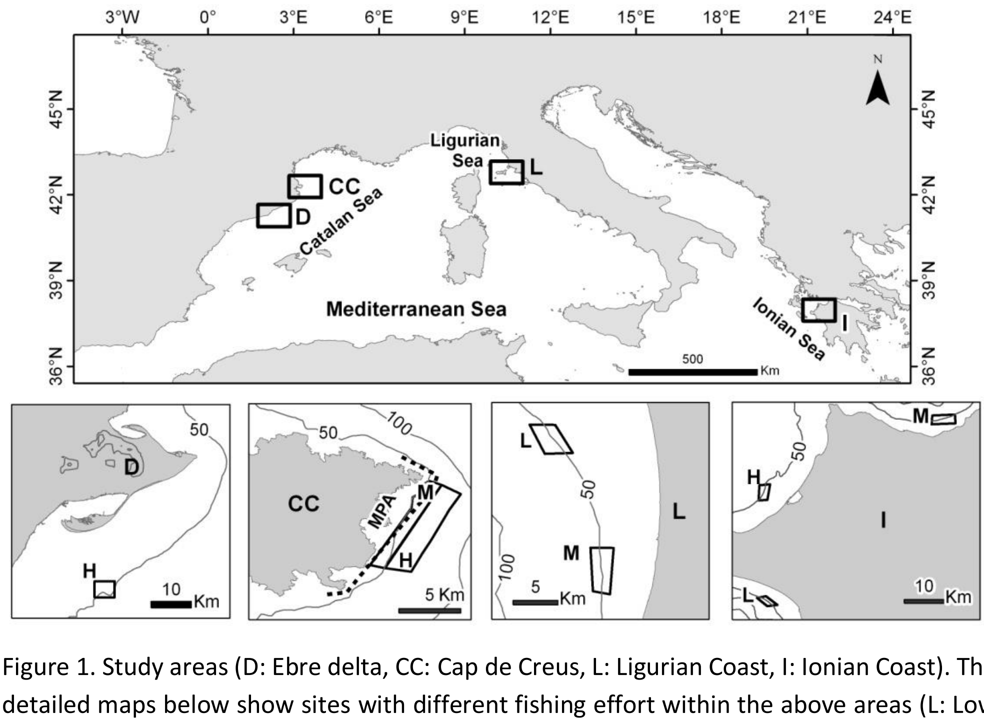

Figure 1. Study areas (D: Ebre delta, CC: Cap de Creus, L: Ligurian Coast, I: Ionian Coast). The detailed maps below show sites with different fishing effort within the above areas (L: Low effort; M: Medium effort; H: High effort). Stripped line in CC area delimitates a MPA.

2.2. Data collection

All replicates were collected in early summer (June-July) of 2003 for D area and 2009 for the other areas. Macroinfauna was sampled with a 0.1 m^2 Van Veen grab. Five replicates were randomly collected in D, whereas 3 replicates were randomly collected in the other sites. To determine the number of grabs per replica needed for a minimum sample size, a species accumulation curve was done. The asymptote was reached with the accumulation of 5 grabs (Demestre 2006), hence 5 grabs were collected and pooled per replica. All individuals retained in a 1 mm mesh were counted and identified to the lowest practical taxonomical level (generally family or genera). Abundance was standardized to 1 m^2. The surface sediments from these grabs were retained for the analysis of grain size, which was analyzed with laser diffraction (using Horiba La-950v2 particle size analyser). Data on the median grain size (D50-μm) and the percentage of mud, sand and gravel in the sediment was available for each site (see Muntadas et al. 2015 for further details).

Epifauna was sampled with an epibenthic dredge with a 2 m iron-frame aperture and a 1 cm cod-end. Six samples were randomly collected in each site, the dredge was towed for 15 minutes with a constant speed of 2.3 knots and a scanmar device attached to the dredge to ensure continuous contact with the seabed. Only 2 replicates were collected at IL site due to technical problems. Epifaunal organisms were generally identified to species and the number of individuals for each species was recorded and standardized to 1000 m^2.

2.3. Trait classification

Nineteen biological traits (BT) belonging to four different categories (environmental position, habitat-motility, size and feeding method) were selected to describe the infaunal community composition (Table 1). Those traits were linked to important functions in soft-bottoms like sediment stability (Snelgrove, 1997) and bioturbation, which enhances nutrient cycling (Frid, 2011; Thrush and Dayton, 2010). The categories habitat and motility were merged as previous data exploration showed high correlation ($\rho > 0.85$) between BTs from these two categories (see Muntadas et al. 2015 for further details). Moreover, habitat and mobility are generally related traits in the infauna and, combined, partly determine certain ecological functions (e.g. highly motile and burrowing organisms importantly contribute to nutrient cycling) (Frid 2011, Norling et al. 2007).

Twenty BTs belonging to five different categories (environmental position, habitat, motility, size and feeding method) were selected for the epifaunal community (Table 2). Those traits were related to key epifaunal functions in soft-bottoms like bioturbation (Thrush and Dayton, 2010), benthopelagic coupling and habitat structure (Bremner et al. 2006, Muntadas et al. 2015, Thrush & Dayton 2010). Moreover, the selected traits for infauna and epifauna are related to trawling vulnerability/resistance (Bremner et al. 2003, de Juan et al. 2007, 2009).

All the identified taxa (104 infaunal taxa and 182 epifaunal taxa) were scored for its affinity to each BT using a scale of 0-3 (0= no affinity to 3

Table 1. Infaunal biological traits (BT) used in the redundancy analysis.

Categories	**Biological traits**
Environmental position	Epibenthic
	Endobenthic top 2cm
	Deep buried
	Other (within another organism)
Habitat-motility	Bivalve-sedentary
	Tube-sedentary
	Tube-motile
	Burrow-limited motility
	Burrow-freely motile
	Free living-freely motile
	Other (within another organism)
Feeding method	Deposit feeding
	Filter feeding
	Opportunistic/Scavenger
	Grazer
	Commensalism
Size	Small (<1cm)
	Medium (1-3 cm)
	Large (> 3 cm)

= high affinity). The score was assigned using the 'fuzzy scoring' method that allowed the taxa to exhibit more than one BT of a given category as long as the total score per category was 3 (Bremner et al., 2003). This assignment was based on published accounts of the biology of each taxa, books and websites of various scientific institutions (e.g. the BIOTIC database maintained by the Marine Biological Association UK http://marlin.ac.uk/biotic/) (see Muntadas et al. 2015 for further details on this approach). In some cases, no information was found for a particular infaunal taxa and trait category (16 taxa regarding the category "environmental position" and 2 taxa regarding the category "feeding method"). These taxa were not taken into account for that particular trait category when assessing the redundancy metrics; therefore, our redundancy measures are conservative.

Table 2. Epifaunal biological traits (BT) used in the redundancy analysis.

Categories	**Biological traits**
Environmental position	Epibenthic Endobenthic
Habitat	Attached Bed forming Burrow dwelling Encrusted Erect Free living
Mobility	Swimmer Crawler Burrow Sessile/Sedentary
Feeding method	Deposit feeding Filter feeding Opportunistic /Scavenger Predator Grazer
Size	Small (<5cm) Medium (5-10 cm)

2.4. Functional redundancy measures

In order to investigate the role of rare taxa in community functional redundancy, two datasets for both infauna and epifauna were produced: a dataset with the whole taxa list (containing common and rare taxa - BT_{whole}) and a dataset with selected taxa (containing only common taxa - $BT_{selection}$). The whole infaunal BT list (hereafter called BT_{iw}) was reduced to a subset of 36 taxa. The selection included taxa accounting for more than 1% of the total abundance in a site and recorded in the 3 replicas (i.e. representativeness and dominance criteria, no biomass data was available for infauna); this reduced dataset was named BT_{is}. The dataset containing all the epifaunal species (hereafter BT_{ew}) was reduced to 47 species selected as the most common ones to build a reduced dataset (BT_{es}). Species on the BT_{es} dataset were selected following three representativeness and dominance criteria: species recorded (a) in at least five of the six replicas, or (b) in at least three replicates and with more than three individuals, or (c) weighting more than 10 g in at least one of the three replicas.

Once all the datasets were produced, the next step was to estimate the two redundancy measures: common traits and widespread traits.

a) Common traits: To build a common traits' matrix per site, firstly, the frequency of occurrence of BT in the eight sites was calculated by weighting the BT scores by the abundance of each taxon exhibiting that BT (Charvet et al. 1998). This procedure was applied to BT_{iw}, BT_{is}, BT_{ew} and BT_{es} data sets, resulting in four replica by trait matrices, containing the weighted abundance of BT in each replica (see Muntadas et al. 2015 for further details on the methodology applied).

Then, to quantify the number of common traits per site (i.e. the number of traits redundant regarding trait abundance), a reference value was needed to define the minimum abundance a BT had to exhibit to be considered redundant. Thus, four trait abundance accumulation curves were built from the traits' matrices (BT_{iw}, BT_{is}, BT_{ew} and BT_{es}) using the *rankabundace* function available in the BiodiversityR package (R 3.1.2 version). Specifically, the *rankabundance* function was used to accumulate the proportional abundance of BTs (specific BT abundance/total abundance of BTs) in a plot. This analysis resulted in four trait abundance accumulation curves (Fig. 2). An inflexion point was identified for each curve and the abundance of the BT in the inflexion point was used as a reference value (Fig. 2). The chosen point matched two criteria: 1) the BT accumulated to that point accounted for at least 95% of the accumulated abundance, or 2) the BT at the inflexion point contributed to at least 2% to the overall abundance. In each replica, all the BTs having an abundance equal or higher to the abundance of the BT identified as the inflexion point were counted in order to get the number of common traits per replica. This methodology was applied to the four data sets BT_{iw}, BT_{is}, BT_{ew} and BT_{es} , using the selected reference point in each case (Fig. 2).

b) Widespread traits: Prior to counting the number of taxa having any affinity for each BT (i.e. fuzzy coding > 0), the fuzzy coding score was pondered to avoid giving the same weight to a taxa having a score of <3 for a particular BT and to a taxa having a score of 3 for the same BT (i.e. a taxa with a 3 score only exhibits this trait, while a <3 score represents plasticity, or uncertainty, in the exhibition of a trait). Ponderation was done by dividing each score by 3 (i.e. the highest possible score). Then, to obtain the number of species per BT, the pondered scores for each BT for all taxa were summed in each replica. Taxa having a score of 3 counted as 1 taxa with that particular BT, while taxa holding a 0.5 score counted as “0.17 taxa” for that BT. This procedure was applied to BT_{iw}, BT_{is}, BT_{ew} and BT_{es} obtaining four

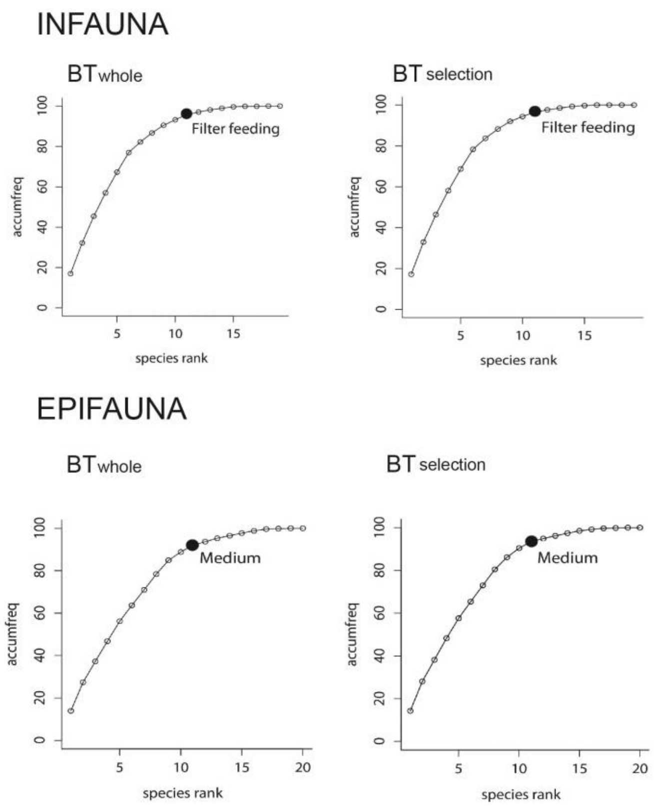

Figure 2. Infaunal and epifaunal trait abundance accumulation curves (common traits) for the whole data set (BT_{whole}) and selected taxa ($BT_{selection}$). Black points show the selected inflexion point for each redundancy measure. The trait identity in the inflexion point is also provided.

replica by trait matrices containing the number of taxa per replica having a specific BT.

The *rankabundance* function was also applied to these four widespread traits´ matrices, producing 4 trait richness accumulation curves (Fig. 3). An inflexion point was also identified for each curve following the same abovementioned criteria and the number of taxa exhibiting the BT identified in the inflexion point was used as a reference value (Fig. 3). For each replica, all the BT represented by a number of taxa equal or higher to the number identified in the inflexion point were counted in order to get the number of "widespread traits" per replica. This methodology was applied to BT_{iw}, BT_{is}, BT_{ew} and BT_{es} (Fig. 3).

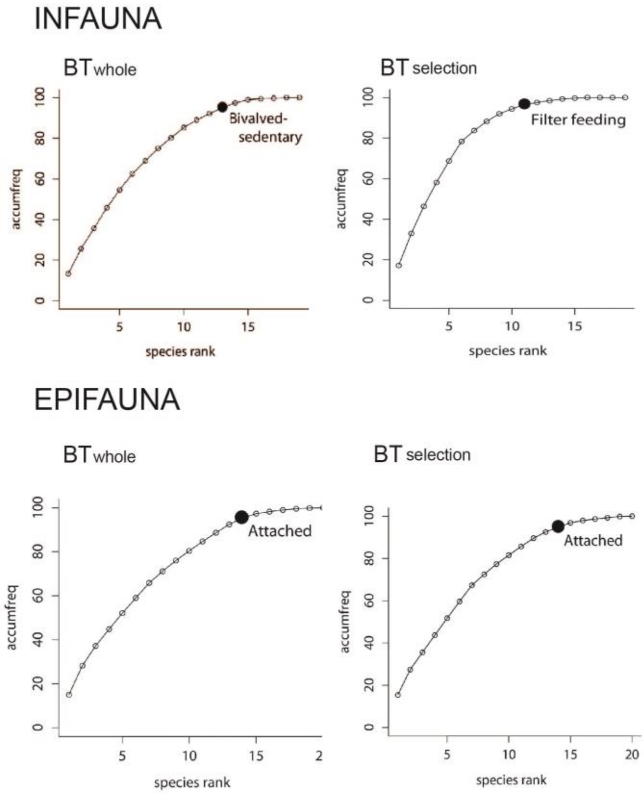

Figure 3. Infaunal and epifaunal trait richness accumulation curves (widespread traits) for whole data set (BT_{whole}) and selected taxa ($BT_{selection}$). Black points show the selected inflexion point for each redundancy measure. The trait identity in the inflexion point is also provided.

2.5. Data analysis

Mann-Whitney tests were performed to detect significant differences between redundancy estimates and to analyse the influence of the inclusion of rare species in the redundancy measures. Tests were performed with GraphPad Prism 5. Kruskal-Wallis tests were performed to analyse significant differences in redundancy measures among sites using the function *kruskal* from the package *agricolae*, which allows the performance of multiple comparison test.

The R package *classInt*, specifically the function *classIntervals*, was used to define three effort levels (Low, Moderate and High) based on the fishing effort values estimated by SSS. The first interval comprised the sites holding the lowest fishing effort values (IL, CCM and CCH), the second interval included the sites having moderate fishing effort (LL and LM) and the last interval included the sites subjected to the highest trawling effort (IM, IH and D). General Linear Models (GLM) were used to assess the effects of fishing effort (based on the previously established categories) and sediment variables on epifaunal redundancy metrics (i.e. widespread and common traits). As all sediment variables were highly correlated, only one variable (D50 or the percentage of mud) was used per model (Muntadas et al. 2015). GLM were done with the R package *mgcv*, the error structure was Poisson (tested with the goodness of fitness function *cbind*) and the most parsimonious models were identified with the lowest AIC scores (Wood & Augustin 2002). It was not possible to apply GLM models to infaunal redundancy metrics due to the low number of samples available (n=26). A Kruskal-Wallis test was used instead. All tests were performed with R v.3.1.2 (www.R-project.org).

3. Results

After applying the procedure described in section 2.4 of the methods to the infaunal and epifaunal datasets, we obtained the number of redundant traits per measure (common and widespread traits) and per replica (see section 2.2.). Figure 4 shows the average number of redundant traits per site and per redundancy measure (common traits and widespread traits for both BT_{whole} and $BT_{selection}$ datasets) for infauna and epifauna. In both cases, the number of redundant traits was higher when measured as widespread (i.e. based on trait richness) than when measured as common traits (i.e. based on trait abundance).

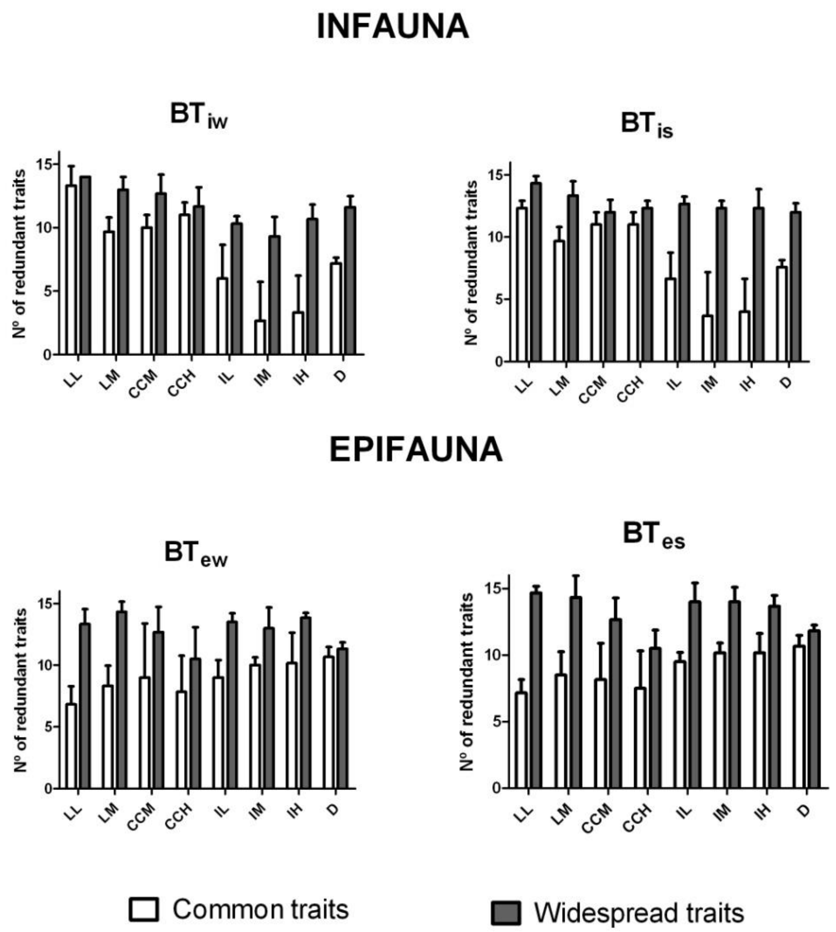

Figure 4. Average and standard deviation of the number of redundant traits for both measures: common and widespread traits. Measures represent infauna and epifauna, and the whole taxa list (BT_{iw}, BT_{ew}) and a selection of the most abundant taxa (BT_{is}, BT_{es}). X axis labels correspond to study sites: D: Ebre delta, CC:Cap de Creus, L: Ligurian Coast, I: Ionian Coast.

3.1. Did functional redundancy described as common traits or as widespread traits differ among sites?

A Kruskal-Wallis test was applied to the four functional redundancy measures to test for significant differences between sites (degrees of freedom =7). All tests detected significant differences among sites (p-value<0.05), except for the infaunal widespread traits measure based on BT_{is} (χ^2= 10.8, p-value=0.15). Widespread traits measures were not clustered according to effort levels. Otherwise, common traits measures evidenced a more clearly difference between the sites holding the highest effort and the other sites, especially for epifauna (Table 3). For epifaunal common traits, the sites subjected to the highest fishing pressure showed higher redundancy mean values, whereas the contrary occurred for infauna (Table 3).

In order to assess if differences among sites in the redundancy metrics also involved differences in the BTs composition, the identity of the redundant traits per site was registered. Tables 4 and 5 contain information on the identity of the redundant traits per site and per measure for infauna and epifauna respectively. The infaunal widespread traits´ measure based on BT_{is} was not included as differences between sites were not detected. For common traits´ measures, redundant BT based on $BT_{selection}$ were not included because the resultant trait list for $BT_{selection}$ and for BT_{whole} evidenced a matching identity of redundant BT obtained with the two approaches. espite for both common and widespread traits some unique traits appeared in specific sites (e.g. “Tube sedentary” for infaunal common traits in LL site and “Bed forming” for epifaunal common traits in IH site), the redundant traits’ composition was similar across sites. Therefore, the number and not the identity of redundant traits caused the main differences among sites for a redundancy measure (Tables 4 and 5).

More evident differences were observed between redundancy measures. The infaunal traits “Bivalve-sedentary” and “Tube motile” were considered redundant with the metric widespread traits but not

Table 3. Groups resulting from the Kruskal -Wallis (factor Site) multiple comparison test (KW groups) for all redundancy measures for infauna and epifauna and considering the whole taxa list (BT_{whole}) and a selection of the most abundant species ($BT_{selection}$). Effort category for each site is also provided (SSS). Mean = Pairwise multiple comparison mean rank sums.

	Widespread traits (BT_{whole})				Widespread traits ($BT_{selection}$)				Common traits (BT_{whole})				Common traits ($BT_{selection}$)			
Epifauna	Site	Mean	KW group	SSS	Site	Mean	KW group	SSS	Site	Mean	KW group	SSS	Site	Mean	KW group	SSS
	LCM	34.67	a	M	LCL	35.17	a	M	D	33.50	a	H	D	33.67	a	H
	ICH	29.17	ab	H	LCM	29.75	ab	M	ICM	29.08	ab	H	ICH	30.25	ab	H
	ICL	25.50	ab	L	ICL	29.25	abc	L	ICH	28.583	ab	H	ICM	30.08	ab	H
	LCL	25.42	ab	M	ICM	28.92	abc	H	ICL	22.25	ab	L	ICL	24.75	ab	L
	ICM	23.75	ab	H	ICH	25.08	abc	H	CCM	21.75	ab	L	CCM	18.67	ab	L
	CCM	21.83	ab	L	CCM	18.33	bc	L	LCM	18.58	ab	M	LCM	18.67	ab	M
	CCH	11.00	b	L	D	11.75	c	H	CCH	15.58	ab	L	CCH	15.58	ab	L
	D	10.67	b	H	CCH	6.25	c	L	LCL	10.50	b	M	LCL	9.83	b	M
Infauna	Site	Mean	KW group	SSS	Site	Mean	KW group	SSS	Site	Mean	KW group	SSS	Site	Mean	KW group	SSS
	LCL	24.00	a	M	-	-	-	-	LCL	24.83	a	M	LCL	24.33	a	M
	LCM	19.67	ab	M					CCH	21.00	a	L	CCH	20.33	a	L
	CCM	18.00	abc	L					CCM	18.33	ab	L	CCM	20.33	a	L
	CCH	13.50	abc	L					LCM	17.33	abc	M	LCM	16.67	ab	M
	D	13.10	abc	H					D	11.00	bcd	H	D	10.80	bc	H
	ICH	8.67	bc	H					ICL	8.50	cd	L	ICL	8.67	bc	L
	ICL	7.00	bc	L					ICH	4.50	d	H	ICH	4.33	c	H
	ICM	4.33	c	H					ICM	4.17	d	H	ICM	4.33	c	H

with common traits metric. Moreover, "Tube sedentary" was frequently redundant for widespread traits, while it was infrequent in the common traits measure (Table 4). The epifaunal trait "Burrow dwelling" was redundant only for the common traits measure, whereas, the traits "Attached", "Swimmer", "Sessile/sedentary" and "Filter feeding" were redundant only for the widespread traits measure (Table 5). Moreover, the trait "Deposit feeding" was redundant only for the widespread traits measure based on the BT_{es} dataset.

Table 4. Infaunal redundant traits´ distribution in all sites, 6 sites, 3-4 sites and restricted to one site for the common traits (based on BT_{iw}) and widespread traits (based on BT_{iw}) measures. Traits highlighted in bold are those traits redundant when considering widespread traits but not when considering common traits.

	Common traits	Widespread traits
All sites	Endobenthic top 2cm Deep buried Burrow freely motile Deposit feeding Medium	Endobenthic top 2 cm Deep buried Burrow freely motile Free living-freely motile Deposit feeding Filter feeding Opportunistic/Scavenger Small Medium Large
5-6 sites	Burrow limited motility Opportunistic/Scavenger Small	Epibenthic Burrow limited motility
2-4 sites	Free living-freely motile Filter feeding Large	**Bivalved sedentary** Tube sedentary
1 site	Epibenthic Tube sedentary	**Tube motile**

Table 5. Epifaunal redundant traits distribution in all sites, 6 sites, 3-4 sites and restricted to one site for the measures of common traits (based on BT_{ew}), widespread traits (based on BT_{ew} and BT_{es}). Traits highlighted in bold are those traits redundant when considering widespread traits but not when considering common traits and vice-versa. * highlights differences between BT_{ew} and BT_{es}

	Common traits	Widespread traits (BT_{ew})	Widespread traits (BT_{es})
All sites	Epibenthic Endobenthic Free living Crawling Burrow Opportunistic/Scavenger Predator	Epibenthic Endobenthic Free living Crawling Burrow Opportunistic/Scavenger Predator Small Medium Large	Epibenthic Endobenthic Free living Crawling Burrow
7 sites	Large	**Swimming** **Filter feeding**	**Swimming**
5-6 sites	Small	**Attached** **Sessile**	**Attached** **Sessile**
4 sites	**Burrow dwelling**	-	**Deposit feeding***
1 site	**Bed forming** Medium	-	-

3.2. Did fishing effort have significant effects on the two functional redundancy measures?

None of the GLM models done with the epifaunal widespread traits as response variable were significant. For common traits measures, fishing effort significantly affected both, the metric calculated using the whole dataset (BT_{whole}) and the reduced dataset ($BT_{selection}$) (Table 6). Sediment variables were also included in the model but they had no significant effects on the redundancy measures.

Table 6. GLM results for epifaunal redundancy measures considering the whole taxa list (BT_{whole}) and a selection of the most abundant species ($BT_{selection}$). SSS_cat: effort categories established using the function *classIntervals* (Low, Medium, High) (see section 2.5), DF: Degrees of freedom. D50: median grain size

GLM: Common traits (BT_{whole}) ~ SSS_cat + D50, family = poisson					
Model intercept : <0.001					
	DF	Deviance	Residual DF	Residual deviance	Pr (<Chi)
Null			43	31.51	
SSS_cat	2	6.34	41	25.17	p<0.05
D50	1	1.25	40	23.91	p>0.05

GLM: Common traits ($BT_{selection}$) ~ SSS_cat + D50, family = poisson					
Model intercept : <0.001					
	DF	Deviance	Residual DF	Residual deviance	Pr (<Chi)
Null			43	22.40	
SSS_cat	2	6.66	41	15.74	p<0.05
D50	1	1.61	40	14.12	p>0.05

Regarding the infaunal metrics, all Kruskal-Wallis tests detected significant effects of trawling on all redundancy measures (Table 7). However, the common traits´ measure evidenced stronger differences among effort classes than the widespread traits´ measure. Moreover, multiple comparison tests for the common traits´ measure showed a more consistent pattern, with the high effort group clearly different from moderate and low effort level groups.

Table 7. Kruskal-Wallis test results for infauna redundancy measures (degrees of freedom =2) considering the whole taxa list (BT_{whole}) and a selection of the most abundant species ($BT_{selection}$). *p-value < 0.05, **p-value < 0.01, ***p-value < 0.001. Groups: result of the multiple comparison test on effort categories, L:Low effort, M=Medium effort, H:High effort(see section 2.5.)

Redundancy measure	BT_{whole}	$BT_{selection}$
Common traits	χ^2= 14.04 p-value = 0.0008908*** Groups = a: M,L b: H	χ^2= 13.79 p-value = 0.001014** Groups = a: M b: H,L
Widespread traits	χ^2= 11.21 p-value = 0.003672** Groups = a: M,L b: H	χ^2= 8.86 p-value = 0.01191* Groups = a: M b: H,L

3.3. Did rare species have an influence on the functional redundancy measures?

In order to test the role of rare species in the redundancy measures, Mann-Whitney tests between the whole dataset (BT_{whole}) and the reduced dataset ($BT_{selection}$) were carried out for each redundancy metric, widespread and common traits, for the infaunal and epifaunal data sets. None of the tests detected significant differences between data sets:

a) Infauna (degrees of freedom= 50): common traits, Mann-Whitney U= 317; widespread traits, Mann-Whitney U= 267.5 (p-values >0.05).

b) Epifauna (degrees of freedom= 86): common traits, Mann-Whitney U= 956; widespread traits, Mann-Whitney U= 854 (p-values >0.05).

Therefore, rare species did not influence the number of redundant traits in neither common nor widespread traits measures for both infauna and epifauna.

4. Discussion

The assessment of ecosystem resilience is especially important in disturbed areas, as ecosystem properties that might contribute to enhance resilience, such as functional redundancy, tend to be compromised in disturbed ecosystems (Graham et al. 2013, Laliberté et al. 2010). In this context, we underline the importance of exploring ways to measure functional redundancy as one component of ecosystem resilience.

We examined the performance of two different metrics, based on BTA, to assess benthic communities' functional redundancy in soft-bottom trawling grounds. These metrics represent two functional redundancy attributes of benthic communities: common traits (i.e. traits with high frequency of occurrence, independent of the number of taxa exhibiting the trait) and widespread traits (i.e. the number of taxa exhibiting the same trait). A higher number of traits were redundant when assessed as widespread traits than when assessed as common traits. These results point out that despite many taxa may share the same trait, those taxa may not be very abundant and, hence, the common traits measure was low. Moreover, it arises the question of how much level of a trait is needed to ensure the related function in an ecosystem (Andersen et al. 2011, Sekercioğlu et al. 2004). Organisms performing a particular function may be present in a community in such low densities that their contribution to the ecosystem performance is negligible and, hence, they could be considered functionality extinct (Sekercioğlu et al. 2004). It is also worth to remind that the species number and how they interact with the environment, not only their biological traits, play a crucial role in maintaining ecosystem processes and stability (Murray et al. 2014; Tilman & Downing 1994).

The species' functional role may vary under different environmental conditions, making the redundancy context-dependent (Wellnitz & Poff 2001, Biles et al. 2003) For example, species that are capable of both, filter and deposit feeding (e.g. *Acanthocardia tuberculata* or *Magelona sp.*), might predominantly deposit feed if the turbidity caused by trawling difficult the use of their filter feeding mechanisms (Hill et al. 1999, Tjensvoll et al. 2013). The use of the fuzzy coding allowed reflecting the behaviour of changing the functionality depending on environmental conditions. However, the fuzziness assignation sometimes reflects uncertainty rather than real knowledge on species biology, as scoring of some species is based on genera or family information (Fleddum et al. 2013, Muntadas et al. 2015). This inevitably adds an error to any functional redundancy metric. Additionally, rare species' BTs are normally less known (Tyler et al. 2012) and approximations at genus or family level have to be used instead (Bolam & Eggleton 2014, Fleddum et al. 2013), which increases the uncertainty of measures. In our case, with some exceptions (Table 5), rare species did not influence any of the redundancy measures analysed. Hence, in agreement with Ellingsen et al. (2007), rare species would hold a range of traits similar to the most abundant species. Therefore, we suggest excluding rare species from the functional redundancy measures in order to keep the indicators as simple as possible and based on cost-effective monitoring (Reza & Abdulla 2011)

It should also be discussed whether both measures, widespread and common traits are needed to assess functional redundancy. Another important aspect of community functional resilience is the diversity of responses to disturbance, i.e. species sharing the same trait but responding differently to stress (Folke et al. 2004, Standish et al. 2014, Walker et al. 1999). The response diversity concept links with our proposed widespread traits measure, as the more species sharing the same trait, the higher probability that some species respond differently to stress. On the other hand, Hewitt et al. (2008) emphasized the importance of organisms' densities rather than the presence/absence of individual traits in driving ecosystem functional processes. Hence, the widespread traits´ measure alone could provide a misleading idea of the overall functional resilience. For example, for the infaunal dataset, the trait "tube sedentary" was frequently redundant based only on the widespread traits´ measure (Table 4). Therefore, although the "tube sedentary" trait was exhibited by a range of taxa in our study sites, this trait may not be abundant enough to significantly contribute to the sediment stability function (Luckenbach 1986, Snelgrove 1997). A similar issue was observed for the epifauna, where traits linked to important functions, such as habitat structure (“attached” and “sessile”) (Bremner et al. 2006a, Snelgrove 1997) and benthopelagic coupling (“filter feeding”) (Bremner et al. 2006, Thrush & Dayton 2010) (Table 7), were redundant regarding the widespread traits´ measure but not regarding the common traits measure. These traits were widely represented but in low densities, which suggests that the low density of organism exhibiting the traits would not fulfil the related functions habitat structure or bentho-pelagic coupling.

The opposite effect, a trait only redundant measured as common traits, also occurred but to a lesser extent (e.g. the epifaunal “burrow dwelling” trait) (Table 7). In this case, the trait was abundant but hold only by a few species. Hence, the functions linked to that trait would exhibit low resilience, as they would be highly vulnerable to the negative effects of anthropogenic or environmental stress on those species. For example, Bellwood et al. (2003) evidenced that the overfishing of a single parrotfish species in an Indo-Pacific coral reef (a key species for the bioerosion function in its natural habitat) lead to a change in ecosystem function from a reef calcification state to a carbonate accumulation state. However, in our specific case study, trawling disturbance was not expected to particularly affect burrow dwelling species (de Juan et al. 2009, Thrush & Dayton 2002). However, we cannot discard vulnerability of these organisms to other sources of impact, which underlines the trait has low resilience. If this situation had been observed with a trait vulnerable to trawling and linked to an important function, such as the trait “attached” for “habitat provision”, management should focus on enhancing the redundancy of these traits (i.e. specifically protecting the species exhibiting these traits, or protecting areas with high densities and representation of these traits).

For epifauna, only the common traits measure was significantly affected by trawling. Otherwise, both infaunal widespread and common traits measures were significantly affected by trawling (Tables 5, 6). However, Kruskal-Wallis test is not as powerful as GLM, and multiple comparison grouping was not clear for the widespread traits infaunal measure. Apparently, these results point out that trawling affects the overall trait abundance but not the number of species holding these traits. Hence, it might be argued that the widespread traits would not be a good measure for trawling impact assessment. Nevertheless, our study sites had been chronically disturbed by trawling and the study did not comprise a non-fished area. In these historically disturbed benthic communities, the most vulnerable traits would have been already reduced or eliminated (de Juan et al. 2009). Actually, traits resistant to trawling (e.g. scavenger, endobenthic, free living) (Jennings & Kaiser 1998, Thrush & Dayton 2002) were considered redundant in all sites, whereas less sites showed redundancy for highly vulnerable traits like attached, erect or filter feeding.

This study context might explain why we did not detect marked trawling effects on the widespread traits measures, as we might be observing the effects of different effort levels on the abundance of those organisms resistant to trawling activities. For example, infaunal overall abundance is known to decrease in fishing grounds (Hinz et al.

2009, Kaiser & Spencer 1996), which might explain the lower value of common traits measure on high effort sites, most of these common BTs being resistant to trawling (Table 4). Therefore, despite not being clearly affected by trawling in our study sites, we cannot disregard the widespread traits measure.

We conclude that the two functional redundancy measures were complementary and a combination of the two metrics is proposed as an indicator of ecosystem functional resilience in soft-bottoms. Relying on a single measure might lead to wrong conclusions about the resilience of certain functions, which might in turn lead to imprecise ecosystem health assessments. Traditionally, redundancy has been defined as the number of species performing the same function (i.e. sharing the same traits, similar to the definition of widespread traits measure) (Guillemot et al. 2011, Naeem 1998, Walker 1992). But this definition does not take into account the role of species´ abundance on functional redundancy. Bello et al. (2007) introduced the idea of species´ abundance on a redundancy measure and observed that redundancy decreased when it was assessed based on species abundance rather than on species presence/absence alone. These results agree with our findings, confirming that common traits and widespread traits are complementary and give information on different aspects of functional redundancy, which may allow a more accurate assessment of community functional resilience.

5. CONCLUSIONS

There are different approaches to estimate functional redundancy, either by considering the species abundance or the species identity. We focused on these two different aspects of functional redundancy (trait abundance and trait richness), which provided complementary information that we consider essential if these measures would contribute to the assessment of the ecosystem status in trawling grounds. Although trawling effort only significantly influenced the common traits measure, we cannot exclude the widespread traits measure from functional redundancy assessment in fishing grounds. Therefore, we suggest using the two metrics as an indicator of ecosystem functional resilience in soft-bottoms. Rare species did not influence the overall number of redundant traits, hence, a species' subset would be enough to measure functional redundancy, simplifying the suggested indicators. Functional redundancy being a component of ecosystem resilience, we underline the importance of increasing our understanding on the broad-scale effects of commercial trawling on functional redundancy if we are to manage fisheries to maintain the resilience of natural systems.

ACKNOWLEDGMENTS

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IV

CHAPTER 3:

Potential indirect effects oftrawling disturbance on commercial species.

Following the former chapter issue, benthic functionality changes due to trawling may also affect commercial species as these changes may alter the resources' Essential Fish Habitat. This chapter addresses this matter focusing on a very important Mediterranean commercial species, the red mullet *(Mullusbarbatus).*

Trawling disturbance on benthic ecosystems and consequences on commercial species: a northwestern Mediterranean case study.

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Pertorbació de la pesca d'arrossegament en els ecosistemes bentònics i conseqüències en les espècies comercials: Un cas d'estudi en el Mediterrani Nord-Occidental.

RESUM:

Els canvis provocats per la pesca d'arrossegament en els ecosistemes bentònics poden alterar els habitats essencials que els peixos utilitzen per a poder desenvolupar el seu cicle de vida, la qual cosa pot afectar negativament els estocs comercials. Per tal d'abordar aquest tema, aquest estudi analitza els efectes potencials que els canvis funcionals en les comunitats bentòniques provocats per la pesca d'arrossegament poden ocasionar en la població d'una espècie objectiu amb molta importància comercial: el moll de fang (*Mullus barbatus*).

Per a l'estudi s'escollí el calador de Sant Carles de la Ràpita al delta de l'Ebre, on aquesta espècie és un dels principals objectius de la flota d'arrossegament a més a més de ser una zona important per al seu reclutament. Aquest calador presenta un nivell d'impacte per pesca d'arrossegament variable al llarg de l'any: a) un període de veda de dos mesos a l'estiu (juliol i agost), b) un període d'esforç baix poc abans de la veda i c) un altre període d'esforç elevat immediatament després. Una altra característica important d'aquest calador és que hi podem trobar una zona no apta per a la pesca d'arrossegament on no s'hi ha pescat durant més de 30 anys. Aquesta zona es va utilitzar com a control.

Per a avaluar la resposta de la comunitat bentònica a l'impacte del bou, es van utilitzar mostres d'infauna i epifauna recollides durant els tres períodes anteriorment mencionats, tan en el calador com en la zona control. Així mateix es van analitzar les diferències entre els components funcionals de la infauna i l'epifauna entre la zona control i el calador, i entre els diferents períodes amb diferents nivells de pesca utilitzant el mètode dels trets biològics (Biological Traits Approach-BTA). Aquest mètode consisteix en escollir una sèrie de característiques biològiques (morfològiques, de comportament i patrons alimentaris entre d'altres) i assignar-ne una a cada una de les espècies estudiades. Aleshores, per cada mostra i tret se suma l'abundància de totes les espècies que el presenten. D'aquesta manera s'aconsegueix una matriu amb l'abundància del trets per mostra a partir de la qual s'estudia la funcionalitat del sistema.

Per altra banda, es van analitzar dades de captures comercials de moll de fang obtingudes a partir dels registres de la subhasta de venda diària de la confraria de Sant Carles de la Ràpita. Amb aquestes dades es va observar la peculiar dinàmica pesquera del moll del fang, la qual presenta un gran augment de les captures just després del període de veda, coincidint amb l'època de reclutament de l'espècie.

L'estudi evidencia que la variabilitat intra-anual en els nivells d'esforç té uns efectes limitats en la comunitat bentònica, mentre que les diferencies entre la zona control (no pesca) i el calador de pesca són més evidents. En aquest últim cas es van poder observar canvis principalment en l'abundància dels trets relacionats amb la maduresa sexual, l'esperança de vida i la mida corporal per a la infauna i amb la mida corporal i l'esperança de vida per a l'epifauna. En cap de les dues zones (control/pesca) es detectà la presència significativa de trets relacionats amb organismes creadors d'estructura d'hàbitat.

Els efectes observats en el calador podrien beneficiar els adults de la població de moll de fang, ja que es va observar un augment de la infauna de mida mitjana-gran en aquesta zona (relacionada amb una maduresa sexual més tardana i una major esperança de vida) que podria representar una major disponibilitat d'aliment per a aquests individus que s'alimenten d'aquest tipus d'organismes. Per contra, els reclutes del moll de fang resultarien perjudicats per la pesca, ja que el canvi funcional de les comunitats bentòniques del calador faria disminuir la seva font d'aliment (invertebrats bentònics de mida petita). Durant l'estiu però, coincidint amb l'època de reclutament a l'àrea, aquests petits reclutes de moll podrien veure's lleugerament beneficiats per un petit increment d'aliment a causa d'un possible augment de la producció secundària. A nivell d'estructures biogèniques de protecció, tant els adults com els reclutes en sortirien perjudicats, a causa de la manca d'organismes estructuradors creadors d'hàbitat (pèrdua d'hàbitat essencial).

Per tant, l'efecte global de la pesca d'arrossegament sobre la població del moll de fang, considerant els efectes negatius indirectes provocats per l'alteració de les comunitats bentòniques i l'hàbitat, juntament amb els efectes directes de l'elevada pressió pesquera exercida sobre els reclutes després de la veda, pot afectar l'estoc reproductor la qual cosa empitjoraria la situació dels reclutes.

Una gestió basada en l'ecosistema hauria de considerar aquests impactes en l'hàbitat essencial del moll i prendre mesures per a millorar-ne l'estat (p.ex. limitació de l'esforç en zones de reclutament o incorporació progressiva de la flota després de l'època de veda).

Trawling disturbance on benthic ecosystems and consequences on commercial species: a northwestern Mediterranean case study

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SUMMARY

Trawling is known to disturb benthic communities and habitats, which may in turn indirectly affect populations of commercial species that live in close association with the seabed. The degree of impact on both benthic communities and demersal species depends on the fishing effort level. This may vary over the year because of the fleet dynamics, which are in turn normally driven by the main target species' life cycle. In this study we describe changes in benthic functional components of a northwestern Mediterranean fishing ground that represents a recruitment area for an important target species (red mullet, *Mullus barbatus*). This fishing ground experiences a varying intensity of fishing effort over the year and benthic functional components under different levels of trawling were compared with an unfished, control area. Traits related to sexual maturity and life span for infauna and body size and life span for epifauna were found to vary with fishing activity. Potential effects of these changes on ecological functioning and the impact on red mullet population are discussed. The

development of fisheries management plans under an ecosystem based fisheries management (EBFM) requires the links between target species and benthic communities' disturbance due to fishing practices to be explicitly considered.

Keywords: Benthos: BTA; Functioning; Mullet fisheries; Trawling; NW Mediterranean.

Perturbación de la pesca de arrastre en ecosistemas bentónicos y sus consecuencias en las especies comerciales: un caso de estudio en el Mediterráneo Noroccidental

Resumen: Se sabe que la pesca de arrastre provoca una perturbación en los hábitats y ecosistemas bentónicos, lo cual a su vez puede afectar indirectamente a las poblaciones de especies comerciales que viven en estrecha relación con el fondo marino. El nivel de impacto en las comunidades bentónicas y en las especies comerciales depende en ambos casos del nivel de esfuerzo pesquero. Este esfuerzo puede variar a lo largo del año, ya que la dinámica de la flota está normalmente determinada por el ciclo vital de las especies objetivo. En este estudio se describen cambios en los componentes funcionales del bentos de un caladero del Mediterráneo noroccidental que constituye un área de reclutamiento para una importante especie objetivo como es el salmonete de fango (*Mullus barbatus*). Este caladero experimenta variaciones de la intensidad de esfuerzo pesquero a lo largo del año. Los componentes funcionales del bentos sometidos a estos niveles variables de esfuerzo fueron comparados con los de una zona control que no está sometida a la pesca. Los resultados muestran que características relacionadas con la madurez sexual y el periodo de vida para la infauna y con el tamaño corporal y el periodo de vida para la epifauna variaron con el esfuerzo pesquero. En el trabajo se discuten los efectos potenciales de estos cambios en la funcionalidad del ecosistema y su impacto en la población de salmonete. Para desarrollar planes de gestión pesquera en el marco de la gestión basada en el ecosistema (EBFM) se requiere que estas relaciones entre la perturbación de las comunidades bentónicas debida a la pesca y las especies objetivo sean claramente consideradas.

Palabras clave: Bentos; BTA; Funcionalidad; Pesquería del salmonete; Pesca de arrastre; Mediterráneo noroccidental.

1. Introduction

Trawling is widely held to be the human activity with the greatest impact on continental shelves all over the world (Jennings and Kaiser 1998, Thrush et al. 1998). Towed bottom fishing gears severely disturb the seabed and there are many studies highlighting their negative effects on benthic communities and habitats (e.g. Dayton et al 1995, Kaiser et al. 2000, Smith 2000). These habitats might provide critical environments for various life stages for many commercial species, i.e. spawning, recruitment and growth habitats, and are sometimes termed essential fish habitats (EFH) (Auster and Langton 1998). Alterations to the seabed may therefore indirectly affect commercial species populations, especially for those species living in close relationship with benthos and feeding on it (de Juan et al. 2007a, Fanelli et al. 2010).

The degree of seabed alteration and potential consequences on commercial species will depend on the intensity of the fishing effort. Therefore, it is important to understand the responses of benthic ecosystems to variations in trawling intensity. Communities vary over time and space in response to natural variability (de Juan & Hewitt 2014), which can be confounded with the temporal and spatial dynamics of anthropogenic disturbance factors (Koch et al. 2009). In this context, it is essential to consider the effects of the stressors over time and space, as i) the temporal frequency of the activity might condition the ability of the systems to recover between disturbance events, ii) the spatial intensity of stressors might be linked to the existence of less disturbed areas that, through connectivity mechanisms, can contribute to the ecosystem recovery (Thrush et al. 2013, Planes et al. 2006), and iii) natural variability can influence the cumulative effects as natural oscillations of communities overlap with the stressor effects.

The effects of different levels of fishing pressure on habitats and benthic communities have been explored over spatial gradients of trawling disturbance (Thrush et al. 1998, Jennings et al. 2001, Collie et al. 2005, de Juan & Demestre 2012) and temporal closures (Smith 2000, Hiddink et al. 2006a, Demestre et al. 2008). However, the effect of the temporal dynamics of trawling fleets (i.e. differences in fishing effort intensity over time in the same fishing ground) on benthic ecosystems are still unknown and cumulative impacts of trawling activities over time probably further compromise the resilience of communities (Hinz et al. 2009).

The European Marine Strategy Framework Directive (EMSFD) established by the European Commission 2008 (EC2008/56) encourages Member States to move towards an ecosystem-based fisheries management (EBFM) in order to protect ecosystems goods and services that marine ecosystems provide. Consequently, it is important to take into account the link between habitat and commercial species in management in order to move towards an EBFM if the goals of the EMSFD are to be met. There are a number of interactions through which fishing activity may influence commercial species (the principal ones are depicted in Figure 1):

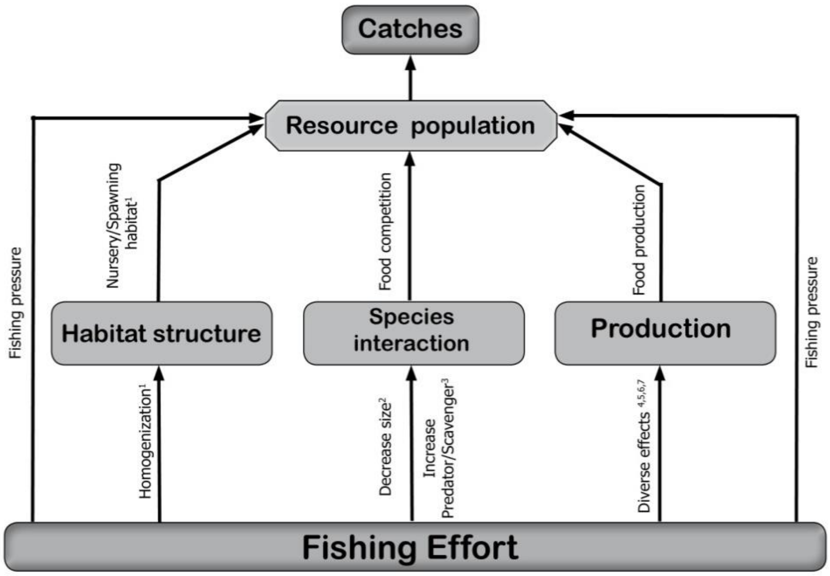

Figure 1. Conceptual model depicting the effects of fishing on commercial species. See the text for more details. References: [1] Jenning and Kaiser 1998,[2]de Juan et al. 2009, [3]Rumohr & Kujawski 2000, [4] Jennings et al. 2001, [5]Jennings et al. 2002, [6] Queirós et al. 2006, [7]Hiddink et al. 2006b

i) Production: ecosystem production is represented as a food source for demersal commercial species. Production by small infauna does not seem to be affected by high fishing intensity (Jennings et al. 2002), although it can increase at moderate levels of disturbance (Jennings et al. 2001), which may benefit species feeding on small opportunistic fauna (Rijnsdorp & Vingerhoed 2001). On the other hand, production by larger infauna and epifauna would decrease in heavily trawled areas (Jennings et al. 2001, Hiddink et al. 2006b, Queirós et al. 2006), which may result in a food impoverishment for fish. Benthic carnivorous fish consume larger prey as they grow (Lukoschek and McCormick 2001), so a decrease in larger infauna might principally affect adult populations and the most economically important components of the stock (Fanelli et al. 2010). Another important food source for several demersal species, especially during the juvenile phase, is suprabenthos, whose abundance and biomass could also be affected by trawling (de Juan et al. 2007a, Fanelli et al. 2011).

ii) Habitat structure: important negative consequences of trawl fishing activity have also been described at the benthic habitat level. Habitat structure that provides shelter and favours the establishment of spawning and nursery habitats may also be altered as trawling activity is known to homogenize habitat structure (Jennings & Kaiser 1998, Thrush et al. 2001).

iii) Species interactions: changes observed in trawled areas, such as changes in epifaunal composition, might also affect commercial species as some epifaunal species compete with commercial fish for food. The epifaunal size decrease caused by trawling (de Juan et al.

2007b) might actually benefit commercial species by releasing them from large potential competitors, but the increase in predator and scavenging species (Rumohr & Kujawski 2000) could increase the food competition.

Current management strategies of the Mediterranean trawl fisheries imply effort limitation (temporal and spatial closed areas, engine power limitation, licence controls, etc.) and technical measures (minimum landing sizes, mesh size, etc.) (Caddy 1993, Smith 2000, Lleonart and Maynou 2003). All these measures focus mainly on the commercial species and are far from an integrated ecosystem approach.

Mediterranean trawl fisheries are characterized by seasonal dynamics which are mainly driven by the life cycle of the main target species (Martín et al. 1999, Martín et al. 2014). These characteristics, added to implementation of a closed season for many fleets as a management measure, leads to a pattern of uneven fishing effort over the year.

One of the main demersal commercial species in the study area, a trawl fishing ground in the northwestern Mediterranean, is red mullet (*Mullus barbatus*), which constitutes an average of 7.2% of the total catches and 7.9% of the total income. Although these may seem low figures, it is important to take into account the multispecies nature of Mediterranean trawl fisheries, with typically fewer than five species exceeding 5% of the total catch (Sánchez et al. 2004, Martín et al. 2014). Red mullet is a species closely linked to benthic ecosystems with a well-known biology and life cycle (Demestre et al. 1997, Demestre et al. 2000). The study area is part of a nursery ground for red mullet, as muddy sediment and depth around 15-60 m constitute the typical characteristics for this species juvenile habitat (Lombarte et al. 2000, Fiorentino et al. 2004). Likewise, the deepest zones of the study area, 50-80 m, constitute part of a reproductive habitat for this species (Machias & Labropoulou, 2002, Fiorentino et al. 2004).

In the study area, fishing effects on benthic communities were assessed by characterizing it as a chronically impacted seabed (de Juan et al. 2007b). Moreover, analysis of a two-month closed period in this area revealed changes in abundance of some epifaunal mobile species (Demestre et al. 2008). In the present study, we increased our current knowledge by evaluating ecosystem responses to seasonal dynamics of trawling activities, having an unfished site as reference and the potential consequences on the exploited red mullet.

With the aim of advancing on EBFM approaches, taking into account all the effects depicted in Figure 1 and the estimations of fishing effort, the goal of the present work is to link changes in benthic functional structure with indirect effects on red mullet population under the following assumptions:

i) changes in benthic production will affect red mullet's food provision;

ii) homogenization of habitat structure affects both nursery and spawning habitat for red mullet's, and

iii) changes in epifaunal assemblage composition affect interspecific competition between epifaunal species and red mullet.

2. Materials and methods

2.1. Characteristics of the study area

The study was conducted on a muddy fishing ground located in the northwestern Mediterranean. This fishing ground spreads over 400 km^2 in a depth range between 30 and 80 m, and the study area covers a depth range of 40-60 m (Fig. 2). Within this study area, samples were taken from a fished site and from a control site that had remained undisturbed for 20 years due to the presence of the remains of an oil platform (Fig. 2) (see de Juan et al. 2007b for details). The fishing ground was operated by Sant Carles de la Ràpita trawling fleet that, with 69 vessels, is the most important trawling fleet in the Catalonia region (northeast Spain). This fleet showed variable activity throughout the year: fishing effort was high during autumn and

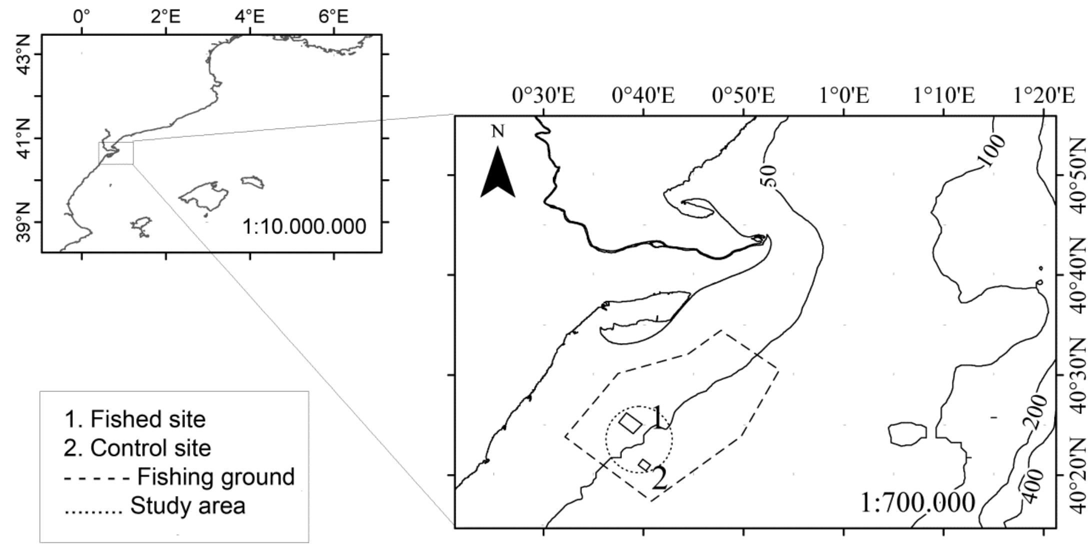

Figure 2. Location of the study area in the Catalan Sea off the Ebro Delta (northwestern Mediterranean.

winter, low in spring and at the beginning of summer and observed a closed season during July and August. Data on fishing effort was obtained from St. Carles de la Ràpita fishermen's association (see Demestre et al. 2008 for further details). Several abiotic characteristics of the area were measured during the benthic sampling, which was timed to cover seasonal pattern of fishing effort (Table 1). The mud sediment content was almost 100% over the whole study period and the temporal variability was not significant, which characterizes the area as a muddy habitat.

Table 1. Study area characterization over the sampling cruises. C, Control site; F, Fished site; %OM, organic matter percentage; %Mud, mud percentage on sediment;

Cruise	27 to 30 June 2003		14 to 17 July 2003		28 to 31 July 2003		19 to 22 August 2003	
Site	C	F	C	F	C	F	C	F
Turbidity (mg/l)	1.48	2.38	1.65	1.37	1.25	2.66	1.13	1.72
% OM	0.55	0.59	0.48	0.59	0.54	0.64	0.63	0.7
%Mud	99.5	99.59	99.48	99.46	99.23	99.52	99.29	99.47
D50	2.65	4.53	2.73	4.58	2.68	4.41	2.67	4.55
Effort level	-	Low	-	Closed	-	Closed	-	Closed

Cruise	26 to 29 September 2003		17 to 17 November 2003		18 to 21 June 2004	
Site	C	F	C	F	C	F
Turbidity (mg/l)	4.57	5.76	3.27	8.11	1.89	2.88
% OM	0.61	0.68	0.51	0.63	0.52	0.61
%Mud	99.38	99.43	99.33	99.5	99.33	99.51
D50	2.82	4.78	2.75	4.82	2.75	4.67
Effort level	-	High	-	High	-	Low

2.2. Collection and processing of samples

Samples of epifauna and infauna were collected during seven experimental cruises. Epifauna was collected with a surface dredge, similar to a 2 m beam-trawl with a 1-cm cod-end, and infauna with a 0.1-m^2 Van Veen grab. On each cruise, a total of three epifaunal and five infaunal replicates were randomly collected at both fished and control sites. To collect the minimum sample size, estimated from species accumulation curves, the surface dredge was towed for approximately 15 minutes for each replicate and five grabs were collected per replicate. Epifaunal and infaunal organisms were identified to the lowest possible taxonomical level, generally species for epifauna and genera for infauna, and counted (see de Juan et al. 2007b for details).

2.3. Data on landings and income

Data on landings and income from 2000 to 2011 were obtained from records from the local fish auction that takes place upon the arrival of vessels at port (data source: fishing statistics elaborated by the Fisheries Department of the Catalan government). Data were available on daily landings by species (weight and income) for each fishing vessel.

2.4. Trait classification to characterize benthic communities

Eleven biological traits covering aspects of the benthic organisms' morphology, feeding patterns and life histories were selected to represent benthic community. The biological trait approach (BTA) allows community structure and functionality to be better represented in order to link them to the ecosystem services that the community can provide. In our case, a benthic ecosystem from a fishing ground, one of the main ecosystem services that this community provides is food production.

These 11 traits were broken down into categories. For example, feeding type was separated into the categories deposit feeder, filter/suspension feeder, opportunist/scavenger and predator (Table 2). The trait "age at sexual maturity" was treated differently for infauna and epifauna data due to the different life histories of these two groups, with most of the infauna taxa reaching the sexual maturity before a year time and epifauna species maturing later. To reduce the task to a manageable size, the data sets were reduced. For the infaunal assemblage, 25 of 147 taxa were used, comprising the species that contributed 80% of the total abundance plus those that, though not among the most abundant species, were continuously present in three of the five replicates either at fished or control sites over the study period. The epifaunal data set was reduced to 17 of 96 species, which accounted for 95% of the total abundance and met the frequency of occurrence criteria.

Each taxon in the database was scored for its affinity to each trait category using a scale of 0–3 (0 = no affinity to 3 = high affinity). The score was given using the 'fuzzy scoring' method, which allowed the taxa to exhibit more than one category of a given trait as long as the total score per trait was 3 (Bremner et al., 2003). This assignment was based on published accounts of the biology of each species and information codified in the BIOTIC database maintained by the Marine Biological Association UK (http://www.marlin.ac.uk/biotic/). This source of information was complemented with a literature review for

Table 2. Biological traits and categories used to describe functional components of benthic communities. Categories used in (i) infaunal and (e) epifaunal analyses

Trait	Categories
Feeding behaviour	Deposit feeders
	Filter/suspension feeders
	Opportunistic/scavengers
	Predators
Food type	Invertebrates
	Carrion
	Detritus
	Plankton
	Microorganisms
	Nekton
Fragility	Fragile
	Intermediate
	Robust
Living habit	Tube dweller
	Permanent burrow dweller
	Free-living
Size	Very small <1 cm
	Small 1-2 cm
	Small-medium 3-10 cm
	Medium 11-20 cm
	Medium-large 21-50 cm
Flexibility	None <10 degrees
	Low 10-15 degrees
	High >45 degrees
Life span	< 1 y
	1-2 y
	3-5 y
	>5 y
Age at sexual maturity	< 1 y (i)/ ≤1 y (e)
	≥ 1 y (i)/> 1 y (e)
Adult movement	Sessile
	Crawl
	Swim
	Burrow
Reproduction frequency	Continuous
	1 reproductive event per year
	2 or more reproductive events per year
	Less than annual
Type of larvae	Direct development
	Short planktonic (<1 week)
	Long planktonic (>1 week)

potential regional differences, i.e. studies conducted in the Mediterranean. When information was not available at the species level we used data based on accounts of other members of the genera or, rarely, family (only in 7.6% of cases for infauna and 3.2% for epifauna). When no information on a particular trait was available for a taxon, zero values were entered for each category and the taxon did not contribute to the calculation of trait weightings.

The frequency of each trait category in the dataset was calculated by weighting the category scores by the abundance (number of individuals per m^2) of each taxon exhibiting that category (Charvet et al. 1998). This resulted in a sample by trait table that showed the abundance of biological traits at each station over the study period.

2.5. Statistical analysis

Similarity between each pair of samples was calculated using the Bray-Curtis index after a square root transformation of the data to reduce the influence of dominant traits/species. A PERMANOVA analysis was used to test for significant differences between sites (control and fished as fixed factors) and time (fishing effort periods, i.e. before, during and after closure, as fixed factor). The trait data were further analysed with the SIMPER routine to determine which traits accounted for the significant dissimilarities identified by PERMANOVA. Then, the most important traits highlighted by SIMPER (traits showing ratio of dissimilarity to standard deviation [diss/sd] >1.5 and being among the ones summing 50% of cumulative contribution to dissimilarity) were selected for univariate analyses. When traits had a normal distribution and homogeneity of variances, a two-way ANOVA was performed to test for the factors treatment and effort period. If traits were not normally distributed (even after log transformation), a Kruskall-Wallis test was performed instead. All the multivariate analyses were carried out using the PRIMER6 & PERMANOVA statistical package (Anderson et al. 2008). Univariate analyses were performed using the R program, v.2.11.0.

3. Results

3.1. Landings and fleet dynamics in the study area

Figure 3 shows the percentage of mantis shrimp (*Squilla mantis*), hake (*Merluccius merluccius*) and red mullet (*Mullus barbatus*) landings and income with respect to the total landings and income in the study area. The importance of these three species remained consistent over the years, representing around 25% of the total landings and income. The “other” percentage comprises on average 53 species, most of them accounting for less than 5% of the total catches and only sporadically (e.g. in one or two years of the data series) more than 5%.

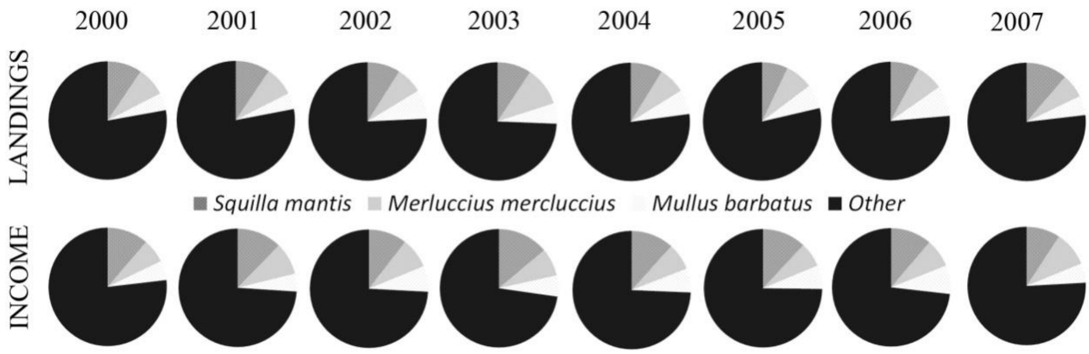

Figure 3. Percentage of landings and income for the main target species of the Sant Carles de la Ràpita trawling fleet.

Despite an overall decrease since 2007, red mullet landings consistently followed the same trend, with a peak of catches in September/October (the recruitment months for this species), which accounted for almost 20% of the total landings (Fig. 4A, B). Figure 4C shows how this peak in red mullet catches coincided with the high effort period. However, red mullet landings dropped sharply in late autumn (November) whereas the decreasing trend of fishing effort was smoother as mantis shrimp landings were maintained over the winter (Fig 4C). This figure shows how fleet dynamics followed red mullet population, as the highest landings and effort occurred just

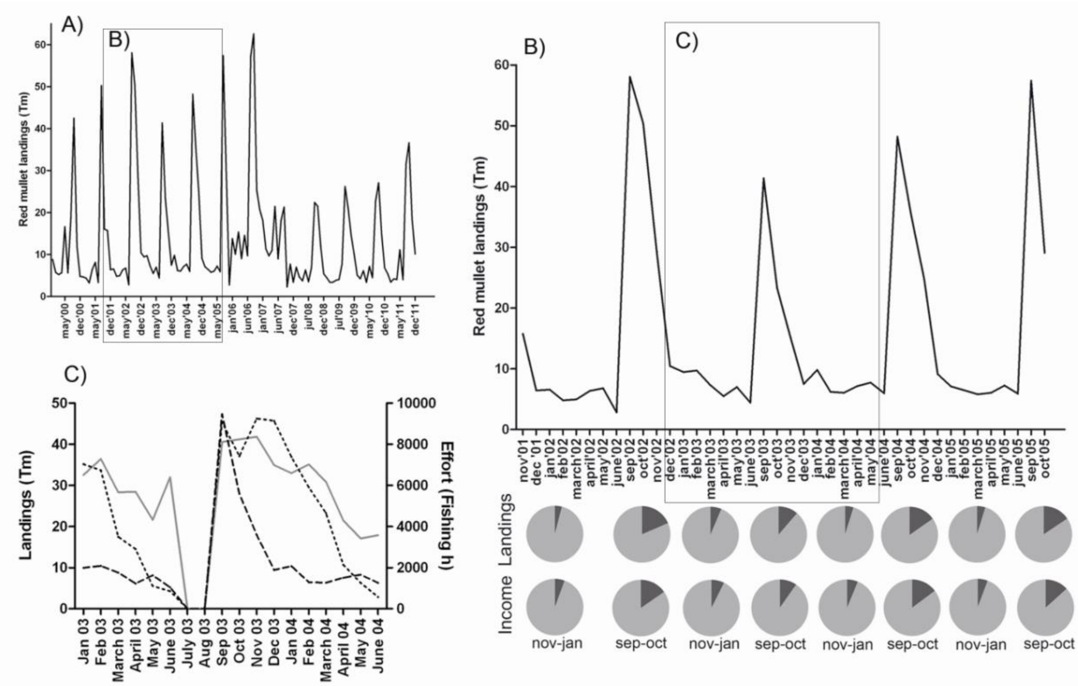

Figure 4. A) Evolution of red mullet catches over the years. B) Zoom from November 2001 to October 2005. Pie charts show the percentage of landings and income of red mullet (black area) with respect to total landings and income, respectively (grey area). C) Landings of *Mullus barbatus* (stripped line) and *Squilla mantis* (dotted line), and fishing effort (grey continuous line) over the study period (2003-2004).

after the closed season, when the trawling fleet gathered on this fishing ground to fish red mullet recruits (Demestre et al. 1997, Martín et al. 1999).

3.2. Functional changes in the benthic communities

PERMANOVA analyses highlighted significant differences for both infauna and epifauna between sites (control vs. fished) and time (different effort regimes), and site:time interaction for infauna (Table 3). Pairwise tests performed for infauna within fished sites showed significant differences between before closure and the closed season ($p<0.01$) and between before closure and after closure periods ($p<0.05$). Furthermore, differences between times for epifauna were all independent of the site, and pairwise tests performed across all samples showed significant differences between before closure and after closure periods ($p < 0.01$) and between closed season and after closure periods ($p < 0.01$). Ordination of samples in a multi-dimensional scaling plot did not reflect any clear pattern and was not included.

Table 3. Infauna and epifauna PERMANOVA results for the fixed factors time and site. Significant p-values are highlighted in bold.

INFAUNA	df	MS	Pseudo F	p-value
Time	2	158.56	3.87	**0.008**
Site	1	1199.40	29.33	**0.001**
Time:site	2	246.35	6.02	**0.003**
EPIFAUNA	df		Pseudo F	p-value
Time	2	460.56	5.70	**0.002**
Site	1	474.93	5.88	**0.006**
Time:site	2	83.05	1.03	0.354

SIMPER analysis for infaunal samples highlighted the traits “sexual maturity at less than 1 year” and “life span of less than 1 year” as the principal traits driving the differences between fished and control sites, both being more abundant in the control site. “Medium-large size” (more prevalent at the fished site) was also an important trait discriminating sites (Table 4).

SIMPER analysis performed only for infaunal fished samples revealed “high flexibility”, “intermediate fragility” and “<1 y sexual maturity” as the main traits driving the differences between before closure and closed season periods, although they showed relatively low diss/sd (Table 4). All these traits were more abundant during the closed season. “Filter feeding” and “2 or more reproductive events per year”, with relatively high diss/sd and also more abundant during the closed season, were important in discriminating between these two periods. Finally, “crawl” and “filter feeding” were the traits making the difference between the before closure and after closure period, both being more abundant in the latest (Table 4).

The SIMPER routine for epifaunal samples highlighted the traits “medium”, “>5 y life span”, “low flexibility” and “medium-large” as the most important traits driving the differences between control and fished sites, all of them being more abundant at the control site (Table 5). As the PERMANOVA test showed no significant interaction between site and time for epifaunal samples (Table 3), a SIMPER test based on the whole data set was performed to identify the traits driving the differences between effort regimes. This analysis highlighted “no flexibility” , “deposit feeder” , “permanent burrow

Table 4. Results of SIMPER analysis for infaunal trait abundance, comparing fished (Ab. Fished) and control (Ab. Control) sites. Fished vs control analysis was based on the whole data set, whereas effort regime analysis was based only on fished site samples. Cut-off to traits list applied at 50% of cumulative contribution to dissimilarity (Cum. Contrib.%); traits with a ratio of dissimilarity to standard deviation (diss/sd) >1.5 are highlighted in bold.

FISHED VS CONTROL (Average dissimilarity=12.64)				
Traits	Ab. Fished	Ab. Control	Diss/SD	Contrib %
<1 yr (sex.mat)	24.29	36.98	**1.61**	5.44
<1yr (life span)	24.05	34.9	**1.5**	4.77
High flexibility	41.92	47.32	1.38	4.31
Direct development	32.34	39.36	1.45	4.15
Medium-large	21.47	12.13	**2.8**	4.09
Tube dweller	18.15	26.1	**1.5**	3.62
Permanent burrow dweller	24.25	31.7	**1.54**	3.49
Detritus	41.44	46.8	1.38	3.51
Filter feeder	20.47	27.95	**1.51**	3.45
Small size	21.33	28.89	**1.51**	3.28
Burrow	44.44	48.03	1.33	3.25
Small-medium size	31.28	35.2	1.4	3.24
BEFORE vs CLOSED (Average dissimilarity=10.03)				
Traits	Ab. Before	Ab. Closed	Diss/SD	Contrib %
High flexibility	40,13	45,89	**1,54**	4.29
Intermediate fragility	40,88	46,93	**1,56**	4.18
<1yr (sex.mat.)	21,93	28,21	**1,57**	3.86
Burrow	42,51	46,73	1,43	3.85
1 repr. event/year	35,39	41,18	1,46	3.72
Direct development	30,35	35,67	1,42	3.69
Detritus	39,49	43,93	1,40	3.61
<1yr (life_span)	22,42	27,92	1,41	3.52
Deposit feeder	37,95	42,12	1,40	3.33
2+repr. events/year	3,37	8,94	**1,76**	3.26
Filter feeder	17,72	23,27	**1,70**	3.15
Free living	33,90	37,92	1,47	3.14
Small-medium size	30,61	33,56	1,43	3.07
Very small size	18,38	22,17	1,08	3.03
Long planktonic	16,69	20,28	1,44	2.88
Free living	33,90	37,53	1,38	2.86

Table 4. Continuation

BEFORE vs AFTER (Average dissimilarity=8.65)				
Traits	Ab. Before	Ab. After	Diss/SD	Contrib %
High flexibility	40,13	45,97	1,46	4.62
Intermediate fragility	40,88	46,56	1,43	4.12
Burrow	42,51	46,61	1,44	3.86
Direct development	30,35	35,28	1,44	3.84
Detritus	39,49	43,83	1,42	3.81
small-medium size	30,61	34,40	1,47	3.7
Crawl	9,56	15,01	**1,78**	3.66
Deposit feeder	37,95	42,40	1,40	3.61
1 repr. event/year	35,39	40,27	1,48	3.48
<1yr (sex.mat.)	21,93	25,81	1,37	3.34
Filter feeder	17,72	22,52	**1,56**	3.27
Continuous reproduction	25,00	28,18	1,39	2.97
Long planktonic	16,69	20,28	1,44	2.88
Free living	33,90	37,53	1,38	2.86

dweller", "burrow" and "detritus" as the traits discriminating between the before and after closure periods and between the closed season and after closure periods. All these traits were more abundant in the before closure period and the closed season respectively. "Fragile", more abundant during the closed season, was also found to be important in discriminating between the closed season and the after closure period (Table 5).

Though these traits were highlighted by SIMPER analysis, it should be noted that average dissimilarities among fishing effort periods were low (10.03 and 8.65 for infauna and 16.33 and 13.83 for epifauna) (Tables 4, 5). Average dissimilarities between control and fished areas were also low (12.64 for infauna and 12.98 for epifauna).

Figures 5 and 6 show the trends of two traits for infauna (Fig. 5) and epifauna (Fig. 6) over the study period. These traits were highlighted

Table 5. Results of SIMPER analysis for epifaunal traits abundance, comparing fished (Ab. Fished) and control (Ab. Control) sites. All analyses were based on the whole data set. Cut-off to traits list applied at 50% of cumulative contribution to dissimilarity (Contrib.%); traits with a ratio of dissimilarity to standard deviation (diss/sd) >1.5 are highlighted in bold.

FISHED VS CONTROL (Average dissimilarity=12.98)				
Traits	Ab. Fished	Ab. Control	Diss/SD	Contrib %
High flexibility	15,10	13,11	1,26	4.43
Medium size	1,09	5,00	**2,37**	4.42
3-5yr (life span)	15,13	13,81	1,24	4.29
>1yr(sex.mat)	15,40	15,37	1,25	4.18
Crawl	17,48	15,86	1,25	3.93
Intermediate fragility	19,10	19,49	1,36	3.81
>5yr (life span)	5,59	8,83	**1,68**	3.8
1 repr. event/year	20,05	20,37	1,35	3.73
Long planktonic	19,42	19,61	1,37	3.71
Small-medium size	19,77	18,55	1,29	3.69
Free living	18,06	18,47	1,43	3.67
Low flexibility	5,41	8,22	**1,66**	3.42
Medium-large size	3,80	6,57	**1,72**	3.23
BEFORE VS AFTER (Average dissimilarity = 16.33)				
Traits	Ab. Before	Ab. After	Diss/SD	Contrib %
No flexibility	15,46	9,15	**2,16**	6.16
3-5yr (life span)	13,01	15,00	1,36	4.71
>1yr(sex.mat)	14,10	15,53	1,37	4.68
High >45	12,91	14,61	1,40	4.45
Intermediate fragility	19,96	17,43	1,45	3.97
Deposit feeder	9,99	6,03	**2,11**	3.95
Permanent burrow dweller	11,30	7,35	**1,68**	3.94
1 repr. event/year	20,98	17,93	1,44	3.94
Long planktonic	20,27	17,40	1,43	3.92
Burrow	9,93	5,90	**2,06**	3.9
Detritus	10,30	6,36	**2,03**	3.89
Small-medium size	20,01	16,99	1,35	3.72

Table 5. Continuation

CLOSED VS AFTER (Average dissimilarity = 13.83)				
Traits	Ab. Close	Ab. After	Diss/SD	Contrib %
No flexibility	14,48	9,15	**2,10**	6
1 repr. event / year	21,21	17,93	1,44	4.4
Long planktonic	20,43	17,40	1,42	4.25
Small-medium size	20,04	16,99	1,44	4.16
Permanent burrow dweller	10,92	7,35	**1,82**	4.11
Intermediate fragility	20,10	17,43	1,40	4.08
Deposit feeder	9,54	6,03	**2,07**	4.01
Detritus	9,81	6,36	**2,08**	3.99
Burrow	9,33	5,90	**2,08**	3.86
Fragile	7,12	3,94	**1,80**	3.84
Free living	19,07	16,85	1,39	3.61
Crawl	17,27	15,84	1,42	3.29

by the SIMPER routine as being important in discriminating between fishing effort periods. Infaunal “filter feeding” and “two or more reproductive events” showed a decreasing trend over the study period at the control site but, whereas for “filter feeding” there seems to be no change at the fished site, in “two or more reproductive events” there was an increasing trend until the end of the closed season and afterwards this trait abundance decreased. Fishing effort values are overlapped in the figure, showing that epifaunal “deposit feeder” and “fragile” organisms were more abundant at fished and control sites when effort was low or null, in the autumn months. Plots for other traits are not included as they showed similar trends to those in Figures 5 and 6: “Filter feeder” trend for infauna was similar to “high flexibility" and highly similar to "<1 y of life span", and"<1 y sexual maturity". On the other hand, the "deposit feeder" trend for epifauna was highly similar to "detritus", "burrow" and "no flexibility", and the "fragile" trend was highly similar to "permanent burrow dwelling".

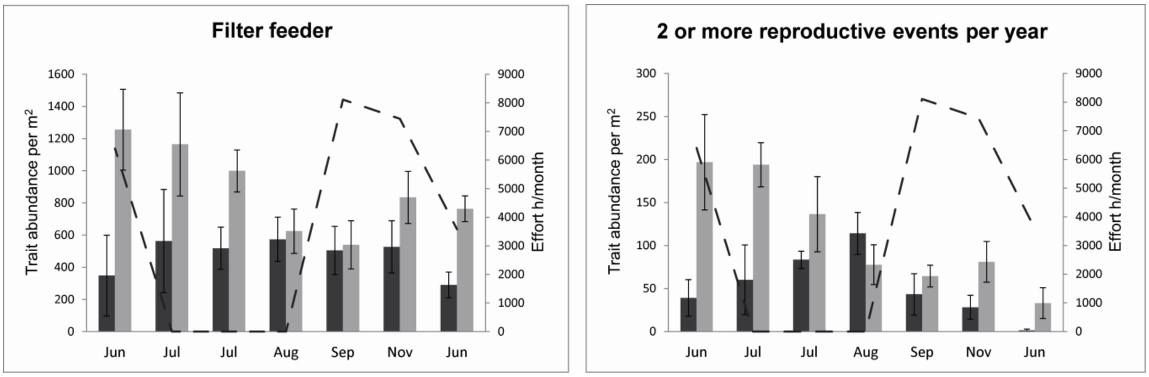

Figure 5. Mean (±se) trait abundance of infauna at fished (black bars) and control sites (grey bars). Dashed line shows fishing effort. June corresponds to before closure period, July and August to closed season and September and November to after closure period.

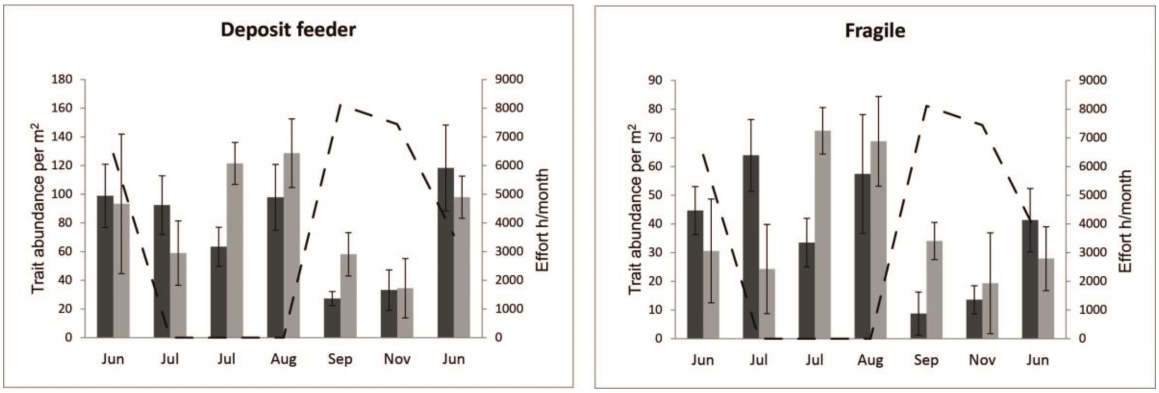

Figure 6. Mean (±se) trait abundance of epifauna at fished (black bars) and control sites (grey bars). Dashed line shows fishing effort. June corresponds to before closure period, July and August to closed season and September and November to after closure period.

Table 6 shows that differences over fishing effort periods (time) for almost all these traits were significant, confirming that they were important in discriminating between these periods. Factor time was not significant for infaunal "filter feeder" and thus all the traits show a similar trend; however, the site:time interaction was significant, suggesting an influence of fishing effort periods or seasonality on these traits' changes. Traits highlighted by SIMPER as the ones driving the differences between control and fished sites also showed significant differences for the factor site.

Table 6. Summary of ANOVA/Kruskal-Wallis and pairwise tests for traits highlighted by SIMPER routine (traits showing a ratio of dissimilarity to standard deviation [diss/sd] >1.5 and being among the 50% of total contribution). * p<0.05, ** p<0.01, *** p<0.0001, ns=non-significant, b=before closed, c=closed season, a=after closed[1]; the specific comparisons were not significant.

Infauna	**Site**	**Time**	**Site: time**
<1 y (life span)	***	ns	*
<1 y (sex.mat)	***	ns	*
Medium-large size	***	ns	ns
Tube dweller	***	ns	*
Permanent burrow dweller	***	ns	**
Intermediate fragility	ns	* (c≠a, b)	*
High flexibility	*	ns	*
Filter/suspension feeder	***	ns	*
Crawl	*	* (b≠a, c)	ns
2+ repr. events/y	***	*[1]	*
Epifauna	**Site**	**Time**	**Site: time**
Deposit feeder	ns	* (a≠b, c)	ns
Detritus	ns	* (a≠b, c)	ns
Burrow	ns	* (a≠b, c)	ns
No flexibility	ns	* (a≠b, c)	ns
Permanent burrow dweller	ns	*** (a≠b, c)	ns
Fragile	ns	***all times	ns
Medium size	***	ns	*
Medium- large size	***	ns	*
Low flexibility	***	ns	*
>5 y (life span)	**	ns	ns

4. Discussion

4.1. Potential effects of fishing on red mullet population caused by functional changes in benthic communities

Different fish species have different habitats requirements, which could be more or less resilient to trawling impacts (Kaiser et al. 1999). The Sant Carles de la Ràpita fishing ground, our case study, is part of a nursery habitat for red mullet (Lombarte et al. 2000, Fiorentino et al. 2004), which is one of the most important commercial fish in the study area (Sánchez et al. 2007). Moreover, this species was chosen as a case study because it is a species closely related to the benthic system, with a well-known biology and life cycle which leads to particular catch dynamics (Fig 4A.). The existence of a nursery habitat is supported by the high catches of new recruits observed after the closed season, in September-October, which constitutes a specific characteristic of this species fisheries (Martin et al. 1999). Recruitment is a critical step for most fish life cycles and decline in recruitment may have important consequences for adult commercial stock (Bundy & Fanning 2005, Caddy 2014). Therefore, the protection of benthic communities and habitat structure, which provide food and shelter for young fish, is essential to maintain a healthy adult stock.

As red mullet lives in a close relationship with the benthic environment for feeding, reproduction and refuge, this species might be particularly affected by the chronic alteration of benthic ecosystems (Auster & Langton 1998, Caddy 2014). In this study we aimed to assess potential negative effects of trawling on red mullet due to changes on benthic functional components in the fishing ground (Fig. 7). Regarding the food availability, small, short-lived infaunal organisms were more abundant at the control site, which might contribute to higher food production for red mullet in this site,

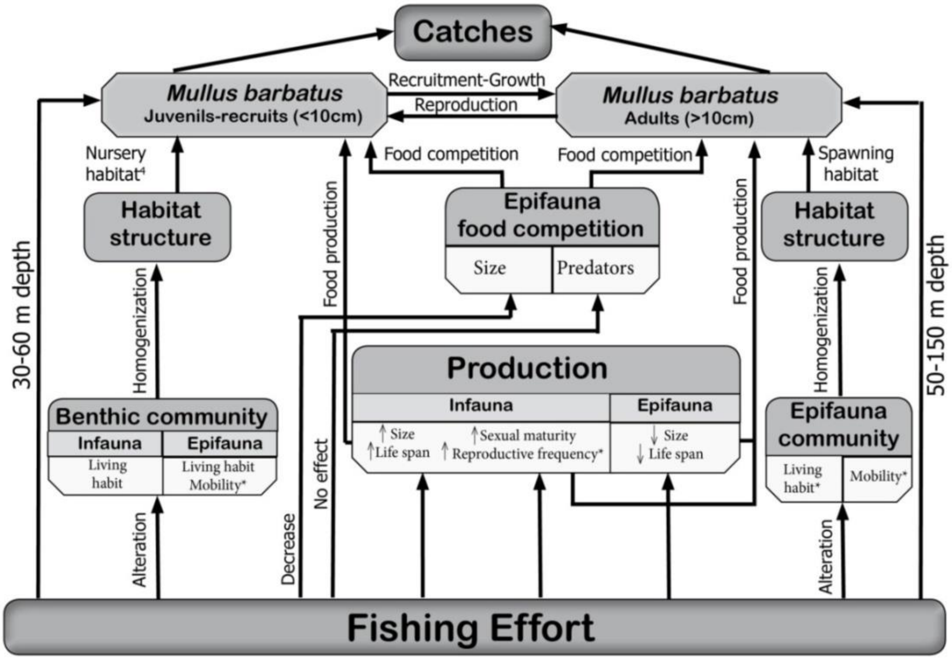

Figure 7. Complex interactions and main potential effects of changes in benthic community due to trawling on red mullet population. *Traits changing over different fishing effort regimes; the other traits showed differences between fished and control sites.

especially for young recruits that feed on smaller prey (Machias & Labropoulou 2002). Moreover, during the closed season in summer, production of small infauna at the fished site may increase due to the higher prevalence of traits related to rapid reproduction (Fig. 7). The closed season coincides with the early phase of recruitment of red mullet (Fiorentino et al. 2008), so young red mullet recruits might benefit from higher food production in summer months. On the other hand, medium-large infaunal organisms dominated the fished site (Fig. 7), which could represent higher prey abundance for adult red mullet in the fishing ground. These results seems to contradict the current understanding of the effects of fishing on marine benthos, suggesting that communities in areas with a long history of fishing would consist mainly of small organisms (Jennings et al. 2001, Queirós et al. 2006). However, all medium-large taxa in the study area were deep burrowers that might avoid trawling disturbance (Brown et al. 2005), and also generally opportunistic-predator feeders that usually benefit from fishing activity (Frid et al. 2000).

Some of the observed trends in infaunal traits over the study period matched the fishing effort pattern, but the variability in infaunal abundance would more likely be related to seasonal patterns as the closed season in the fishing ground is too short to allow ecosystem recovery (Zajac 2003, Kaiser et al. 2006). The decreasing trend for almost all infaunal traits at the control site in summer and early autumn (closed and after closed season) could indicate a decrease in the overall infaunal abundance consistent with the temporal cycles shown by macroinfauna in the Mediterranean (Sardá et al. 1999). In contrast, the reduction of this seasonal trend at the fished site could be due to chronic fishing disturbance, which could alter the natural macroinfaunal cycles (de Juan et al. 2007b). However, this lack of trend in the fished site cannot be unequivocally linked to the fishing activities as natural processes could also play a role, resulting in an additive effect of both fishing and environmental variability (Koch et al. 2009).

In the epifaunal community, as expected and in agreement with de Juan et al. (2009), traits related to long life span and large size were more abundant at the control site (Table 5). This means that epifauna productivity (production/biomass) would be lower in the control area, although biomass production would be higher in this area. These large long-living organisms were fishes and large bivalves (e.g. *Citharus linguatula* or *Acanthocardia echinata*) that are not red mullet preys and, if they are predators, might compete with red mullet for prey. For example, *Citharus linguatula*, feeds in this area on decapoda and other small crustaceans (Juan et al. 2007a), which are potential prey for red mullet (Aguirre 2000), so these two fish partially share the trophic niche. Therefore, although adult red mullet could potentially find more food at the control site, this food would be less available due to interspecific competiveness (Fig. 7).

Regarding the habitat structure, as the control site is slightly deeper than the fished site, it might be a spawning area for red mullet. However, no traits related to habitat structure (e.g. sessile emerging epifauna) were highlighted in the analysis. In general, the whole fishing ground holds a homogenized community with a reduced habitat structure due to historical trawling disturbance in the area (de Juan et al. 2007b, 2009)(Fig. 7). Nevertheless, physical habitat is also important in creating habitat structure for recruits and spawners, and ROV images showed higher abundance of sediment structures such as ripples, mounds and pits at the control site (Demestre 2006). Changes in the trait “burrow” or “permanent burrow dwelling”, which seemed to occur in similar abundance at fished and control sites over the sampled months, would not benefit red mullet spawning as these traits would destabilize the sediment with negative consequences for the creation of habitat structure (Lohrer et al. 2008). However, changes in these traits may affect bioturbation activity which may positively affect ecosystem production (Thrush and Dayton 2002, Lohrer et al. 2004) and indirectly benefit red mullet population in terms of food availability by enhancing primary production.

4.2. Management considerations

Associating demersal fish with their habitats is very critical to the definition of EFH and to correctly managing those EFH impacted by trawling activities (Kaiser et al. 1999). However, management strategies such as closed seasons are principally implemented to protect vulnerable steps of commercial species' life cycles such as spawning and recruitment, focusing only on commercial species stock and not taking into account protection of benthic communities. Short-term closed seasons, such as the one implemented in the Sant Carles de la Ràpita fishing ground, do not enable benthic community recovery between successive periods of impact, especially on stable muddy bottoms where communities might take years to recover (Kaiser et al. 2006). Furthermore, short-term closures result in a concentration of the trawl effort and landings immediately after the closed season rather than in a more equitable pattern throughout the year (Martin et al. 1999). This concentration of fishing effort is also accompanied by an increase in discards (Sánchez et al. 2007), which indicates a higher level of disturbance on benthic communities. An adequate management regulation should progressively increase the fleet capacity after the closed season to avoid resource depletion and the highest disturbance levels on ecosystems.

Red mullet are an important commercial species in the Mediterranean, being one of the main target species for trawling fleets (Caddy 1993, Tserpes et al. 2002). In this case, the red mullet life cycle drives the fleet dynamics, as the highest fishing effort coincides with the highest landings (Fig. 4C), which are composed mainly of red mullet recruits (Demestre et al. 1997, Martín et al. 1999). Though the red mullet population could apparently tolerate this level of exploitation due to its high turnover rate, the yield per recruit shows evidence of overexploitation (Demestre et al. 1997) and this dominance of young specimens in landings makes the stock highly vulnerable to recruitment changes (Tserpes et al. 2002). Therefore, it would be advisable to protect red mullet nursery and spawning areas. Actually, Fiorentino et al. (2008) reported an increase in the number of recruits and a wider recruitment period after a trawl ban in the Gulf of Castellammare, exemplifying that an implementation of a permanently closed area does benefit reproduction success of this species. Moreover, a benthic community in a permanent closed area will have the possibility to recover from trawling impact, which might benefit not only red mullet but also other commercial species (e.g. the structured soft-bottom community observed in the Medes Island MPA, de Juan et al. 2011).

This work shows that changes in the effort regime within a year only had limited consequences for benthic community structure, whereas changes between a non-fished control site and a fished site were clearly evident. The observed changes at the fished site might benefit adult red mullet, as their food provision will be higher due to an increase in medium-large infauna and to lower interspecific trophic competition. However, red mullet recruits will be negatively affected by functional changes caused by fishing as their food provision might overall decrease, although they could benefit from a short-term increase in food production during summer. Moreover, both adults and recruits will suffer from lack of protection of habitat structures. Thus, the overall effect of trawling on the red mullet stock, considering the high fishing pressure on recruits and the indirect negative effects caused by ecosystem disturbance, could be a decrease in the spawning stock that will worsen the recruit's stock situation.

This study highlights the idea that permanent closure areas, which would allow recovery of the benthic ecosystem, restructuring habitats and communities, might be more beneficial for commercial species and their habitats than temporary closures.

Acknowledgements

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V

CHAPTER 4:

Knowledge integration: building a platform to simulate trawling effects on benthic community and to raise stakeholders' awareness of trawling driven changes on benthic ecosystem.

Raising public (and particularly managers') awareness of trawling effects on benthic communities and the importance of keeping habitats in a good condition to keep sustainable fisheries is a key point towards the implementation of an EAF. Therefore this chapter presents a platform that simulates potential changes on benthic communities caused by trawling and show them in an understandable way so as anyone can get an idea of the impact magnitude.

A knowledge platform to assess the effects of trawling on benthic communities.

Under revision in *Estuarine, Coastal and ShelfScience*.

Una plataforma de coneixement per a avaluar els efectes de la pesca d'arrossegament en les comunitats bentòniques.

RESUM:

En el marc actual de l'Enfocament Ecosistèmic a la Pesca (Ecosystem Approach to Fisheries-EAF) hi ha molts més actors involucrats en les decisions de gestió que en el marc tradicional de la gestió d'estocs. Mentre que abans tan sols els polítics i els científics prenien part en les decisions, en l'EAF altres actors involucrats en el món de les pesqueries com els mateixos pescadors, les ONGs mediambientalistes o la societat en general també hi participen. Això fa que l'elaboració d'un pla de gestió sigui molt més complexa, ja que els diferents participants presenten diferents antecedents i defensen diferents interessos.

Per tal d'arribar a un acord consensuat i que el pla de gestió pugui desenvolupar-se amb èxit, calen eines d'intercanvi d'informació fàcilment comprensibles per tots els actors involucrats en el desenvolupament del pla gestor. Així doncs, amb l'objectiu de treballar en la línia de la gestió ecosistèmica i per a integrar la informació adquirida d'una manera comprensible per a tots els actors implicats, aquest treball presenta una plataforma de coneixement que mostra l'impacte de la pesca d'arrossegament en 18 comunitats bentòniques mediterrànies d'una manera atractiva i entenedora. Aquestes comunitats estan situades en caladors de fons tous de la plataforma continental. Així mateix, aquesta plataforma és interactiva

ja que també permet a l'usuari simular un canvi d'esforç en una d'aquestes comunitats i visualitzar l'impacte d'aquest canvi en l'estructura bentònica.

La plataforma està dissenyada en quatre blocs:

1.- En el primer bloc s'hi pot trobar una explicació dels objectius i la utilitat d'aquesta plataforma.

2.- El segon bloc està destinat a la descripció dels mètodes utilitzats en la construcció de la plataforma: l'estimació de l'esforç pesquer, la representació de la comunitat bentònica i l'aproximació estadística per a la construcció de l'eina de simulació. L'estructura bentònica està representada en forma de trets biològics, és a dir, de característiques biològiques enlloc d'espècies. La composició específica està molt influenciada per les condicions ambientals, la qual cosa pot emmascarar els impactes de la pesca. Així doncs, com que diferents espècies poden compartir els mateixos trets, l'aproximació mitjançant trets biològics permet "solucionar" les diferències regionals al mateix temps que permet detectar els impactes de la pesca. Els trets seleccionats estan directament relacionats amb la resistència/vulnerabilitat de les espècies a la pesca d'arrossegament, la qual cosa permet visualitzar els efectes de la pesca en l'estructura i la funcionalitat de la comunitat bentònica.

3.- En el tercer bloc s'hi pot trobar informació sobre les comunitats incloses en la plataforma: la seva ubicació, la seva composició específica i de trets biològics així com gràfics que permeten comparar aquesta composició entre les diferents comunitats.

4.- Finalment, el quart bloc està destinat a l'eina de simulació. En base a estudis previs, en la simulació es considera que el canvi depèn de la intensitat de l'esforç a què està sotmesa la comunitat així com del tipus de sediment. La granulometria del sediment es pot dividir en tres fraccions: grava, sorra i fang (mida del gra de més a menys de diàmetre). En aquest cas es va decidir incloure el percentatge de la fracció sorra en la simulació. L'usuari pot escollir la comunitat més semblant a la de la seva àrea d'interès, simular un canvi d'esforç en aquesta comunitat i veure els efectes potencials que aquest canvi provocaria. A més a més, si l'usuari disposa d'informació sobre l'esforç pesquer, la granulometria del sediment i la composició bentònica de la seva comunitat d'interès, pot entrar aquestes dades a la base de dades de la plataforma i simular una variació d'esforç en la seva comunitat. D'aquesta manera l'usuari podrà simular els canvis que pot provocar la pesca d'arrossegament en la seva comunitat d'interès.

La simulació també calcula un índex que engloba la informació de tots els trets en relació a la vulnerabilitat/resistència a l'arrossegament en un sol nombre que es pot considerar un indicador de l'estat general de l'ecosistema bentònic.

En l'article es mostren els resultats d'aplicar la simulació a quatre de les 18 comunitats incloses a la plataforma, incrementant-ne i reduint-ne l'esforç actual en un 50%. En general els resultats obtinguts són els esperats segons els estudis previs en caladors d'arrossegament: els trets vulnerables augmenten en abundància quan l'esforç disminueix i vice-versa. Així mateix, l'abundància dels trets resistents disminueix quan l'esforç disminueix i vice-versa. Tot i que per a alguns trets es van observar resultats contraris als esperats, els canvis en l'índex global van ser els esperats, la qual cosa indica la importància de considerar tots els trets conjuntament per a determinar l'estat de l'ecosistema.

La simulació no té un caràcter predictiu absolut i només pretén mostrar tendències generals de la comunitat enfront un canvi en l'esforç pesquer que serveixin com a eina d'intercanvi d'informació entre els actors involucrats en la gestió de les pesqueries. En aquest context, la plataforma presentada en aquest article vol ser una eina de conscienciació de l'impacte que provoquen les arts d'arrossegament en els ecosistemes bentònics. En un futur desenvolupament, la plataforma vol mostrar com aquests canvis que afecten a les comunitats bentòniques també afecten a les espècies comercials, per tal de tenir una eina que mostri de manera integrada els efectes de la pesca d'arrossegament.

A knowledge platform to assess the effects of trawling on benthic communities

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HIGHLIGHTS

● Understandable deliberation support tools to share fisheries' knowledge are needed.

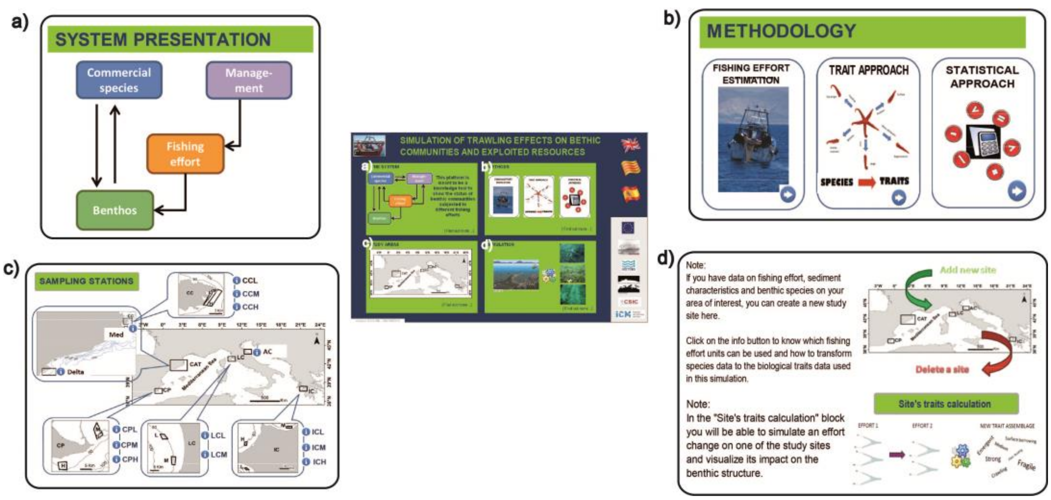
● The presented platform interface is easy to use by any fisheries' decision actor.

● The platform shows trawling impact on benthic communities in a comprehensible way.

● The platform provides a simulation tool to assess fishing effort changes' effects.

● The user may test their own community of interest's dataset.

GRAPHICAL ABSTRACT



ABSTRACT

To implement an Ecosystem Approach to Fisheries (EAF) management, it is necessary that all decision actors involved in fisheries management are aware of fishing impacts and agree with the adopted measures. In this context, there is a need of tools to share understandable knowledge on the ecosystem effects among these decision actors. When managing trawl fisheries under an EAF, one of the main concerns is the impact that trawling gear causes on the benthic communities. Therefore, using the ExtendSim® software, we developed a platform with a user-friendly interface that combines a simulation model based on existing knowledge and data collection and representation of predicted trawling impacts on the seabed. The platform aims to be a deliberation support tool for fisheries' decision actors and, simultaneously, raise public awareness of the need of good benthic community knowledge to manage fisheries correctly. The simulation procedure assumes that trawling affects benthic communities with an intensity that depends on the level of fishing effort exerted on benthic communities and on the habitat characteristics (i.e. sediment grain size). Data to build the simulation comes from epifaunal samples from 18 study sites located in Mediterranean continental shelves subjected to different levels of fishing effort. In this work, we present the simulation outputs of a 50% fishing effort increase and decrease in four of the study sites which cover different habitats and different level of fishing effort. We discuss the platform strengths and weaknesses and potential future developments.

Keywords: Benthos, trawling, EAF, simulation platform, ecosystem management, fishing effort, Mediterranean

1. Introduction

During the past decades, a society concern for the environmental consequences of trawling has arisen and fisheries management policies have moved from a target species´ stock-based management towards an Ecosystem Approach to Fisheries (EAF) management, which takes into account fisheries impacts on non-target species and habitats as well as on target species (Carter 2013, Lucthman et al. 2008, Garcia & Cochrane, 2005). This change of management paradigm not only entails a wider environmental scope but also more actors involved in the process; i.e., while only scientists and politicians took part in the decision process of the species´ stock-based framework, managers, fishermen and NGOs are also involved in the EAF process (Carter 2013). Consequently, a more complex discussion context arises when applying EAF.

The different actors playing a role in EAF management have different backgrounds and different interests, which leads to controversy in the design of management measures (Daw & Gray 2005, Garcia & Cochrane 2005, Toonen & Mol 2013, Carter 2013). For instance, while NGOs claim that no-take zones must be included in spatial management plans, fishermen are obviously reluctant to do so (Toonen & Mol 2013). Moreover, scientific advice for management normally involves large uncertainties which can lead to lack of credibility within the other stakeholders´ groups (Daw & Gray 2005).

However, in order to successfully apply a management plan, all the actors involved in the process must reach an agreement. Hence, appropriate tools to share understandable information are needed for a successful EAF and that's the purpose of the knowledge platform presented here.

In order to advise policy makers on the implementation of management plans, the scientific community has already developed several tools to test management strategies under different EAF scenarios, e.g., end-to-end models as Ecopath with Ecosim (EwE) (Christensen & Walters 2004), Object-oriented Simulator of Marine ecOSystem Exploitation (OSMOSE) (Travers et al. 2010), Atlantis (Fulton et al. 2005, Pinnegard et al this issue) or European Regional Seas Ecosystem Model (ERSEM) (Allen & Clarke 2007). However, benthic compartments that include demersal species (e.g., species targeted by trawling) and its relations with benthic communities are poorly developed in these models (Rose et al. 2010). Indeed, among all these models, only Atlantis could be seen as a tool to assess fisheries' habitat modification (Plagányi 2007).

The main environmental impacts of trawling fisheries are those caused on benthic habitats (Thrush & Dayton 2002). Fish species need particular habitat characteristics for feeding, spawning or growth to maturity, which constitute their Essential Fish Habitats (EFH) (Auster & Langton 1998). Therefore, changes on habitat structure might also affect target species populations and models focused on the benthic community are needed to manage trawling impacts on the seabed. Some examples can be found in the literature, for example Lundquist et al. (2010) introduced a model to illustrate benthic community recovery after an anthropogenic disturbance and Fujioka (2006) formulated a model for habitat reduction due to fishing but this author did not define what “habitat” meant. Dichmont et al. (2008) developed a model for prawn fisheries in Australia that estimated the biomass change of benthic species due to trawling and Duplisea et al. (2002) built a size-based model to estimate the impact of trawling in benthic communities, which differentiates between hard and soft-bodied fauna. The latter was further developed to estimate changes on benthic biomass and productivity caused by trawling in different habitats (Hiddink et al. 2006).

Despite simpler than end-to-end models, the aforementioned benthic-focused models are strongly science-orientated and still have a number of parameters and outputs that might be difficult to understand by non-scientific fisheries’ actors. Fishermen, managers, policy makers, scientists and NGOs have different backgrounds and hence, are used to different approaches for capturing and processing the information (Verweij et al. 2010). For example, different actors use different comparison modes, e.g., in target species´ stock focused management, fishermen compare current catch data with data from past catches. Otherwise, policy makers and scientists compare the most recent values of spawning stock biomass (SSB) with a predefined precautionary reference level. These two approaches may lead to different conclusions and hence controversy among decision actors (Verweij et al. 2010). Therefore, efforts should be driven towards building deliberation support tools to share understandable information among decision actors. Moreover, any stakeholder should be able to manipulate this information. A database that provides data for both scientific and management purposes about benthic species and habitat sensitivities is already available in www.marlin.ac.uk, which also reports trawling effects on benthic communities (Hiscock & Tyler-walters 2006). However, this platform does not allow users to run scenario modelling.

The knowledge platform we present here aims to compile current knowledge about changes on benthic community structure due to trawling in a dynamic and visual way. The platform offers a simulation tool that allows the user to test how the change in fishing effort on a given fishing ground would potentially affect the existing community. This way of sharing knowledge and scenario simulation among stakeholders has proved to be a good approach to deal with other complex socio-ecosystems contexts, such as freshwater management, microbiological contamination in coastal zones or the interaction of jellyfish with fisheries and tourism (Ballé-Béganton et al. 2010, Mongruel et al. 2011, 2013, Ballé-béganton et al. 2012, Tomlinson et al. this issue).

This platform also aims at raising society, and particularly decision makers', awareness about the importance of a good benthic community knowledge to improve the management of trawl fishing resources. In further developments, this platform will aim at informing stakeholders of the target species population status regarding the benthic community structure.

2. Materials and methods

The knowledge platform has been developed using the ExtendSim® software (ExtendSim 9).

2.1. Study areas

To build the platform we used epifaunal data from 7 areas in the Mediterranean Sea, three in the Catalan coast (in the north-western Mediterranean - CAT): Cap de Creus (CC), Medes (M) and Ebre Delta (D); one in Cabo de Palos (in the south-western Mediterranean CP); one in the Ligurian Sea coast (LC); one in the Adriatic Sea coast (AC); one in the Ionian Sea coast (IC) (Fig. 1). CC, CP and IC areas were further divided in 3 different sites that were subjected to different

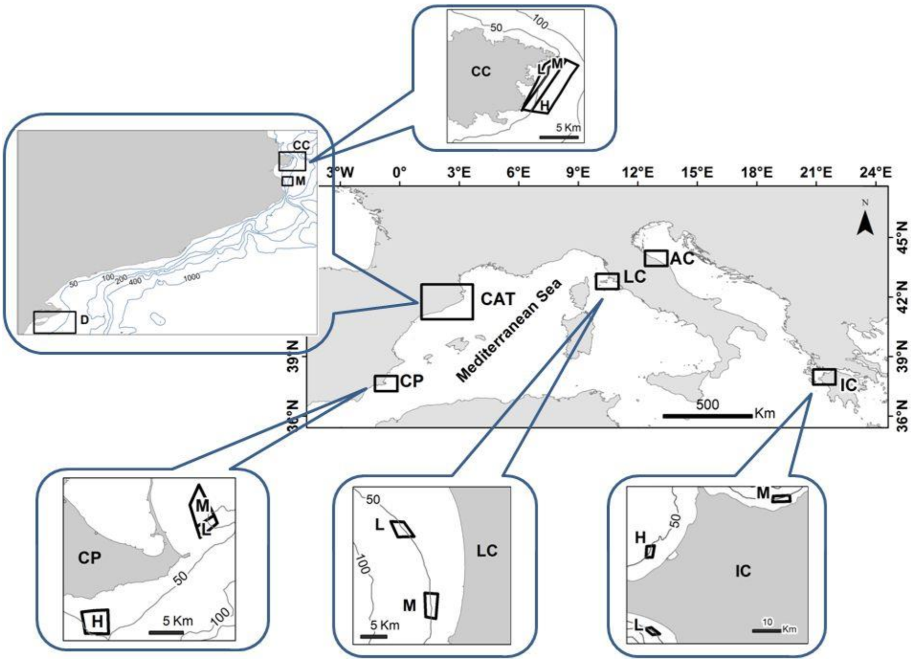


Figure 1. Mediterranean study sites where data used to build the platform were collected. CC: Cap de Creus, M: Medes; D: Delta, CP: Cabo de Palos, LC: Ligurian Coast, IC: Ionian Coast, AC: Adriatic Coast. For the zoomed sites: L: Low or no effort, M: Medium effort, H: High effort.

levels of fishing effort (Low or no effort, Medium and High, hereafter CCL, CCM and CCH for Cap de Creus, CPL, CPM and CPH for Cabo de Palos and ICL, ICM and ICH for Ionian sea coast). The LC area was divided in two sites with low and medium effort (hereafter LCL and LCM) (Fig. 1). Medes and CCL sites were within a Marine Protected Area. In total, we used data from 18 sites.

The LC, IC and D areas had muddy bottoms (99.5%, 96.6% and 93.33% of mud respectively), while CC and AC had sandy-mud bottoms (40.33% and 63.83% of sand respectively). The CP area was characterized by maërl beds protruding within sandy-mud bottoms (de Juan et al. 2013). The M area was characterised by coastal detritic mud.

2.2. Fishing effort estimation

Two different fishing effort estimations were available in our study areas: 1) fishermen interviews along with information from fisheries associations and 2) Side-scan sonar surveys (SSS) (Table 1). The former provides first-hand information about fleet movement in the fishing grounds, which allowed to estimate fishing effort as GT*days at sea/month. The SSS approach allowed the estimation of trawl marks density considering their total length per surface within each site (trawl tracks density/km^2) (see de Juan et al. 2013 for further details).

2.3. Data collection

All samples were collected between spring and early summer from different years: June-July 2003 for D and AC, May 2007 for M and June 2009 for the other areas. Epifaunal community was sampled with an epibenthic dredge with a 2 m iron-frame aperture and a 1 cm cod-end. Six samples were randomly collected at each site, the dredge towed for 15 min with a constant speed of 2.3 knots and a scanmar device attached to the dredge to ensure continuous contact with the seabed. Only 3 replicates were collected in CCL site and 2 in ICL due to bad weather conditions. Epifaunal organisms were generally identified to species and the number of individuals for each species was recorded and standardized to $1000 m^2$.

Table 1. Fishing effort estimations in the sites used to build the platform.

Site	Fishermen data (GT*days at sea/month)	SSS (Trawl tracks density/km2)
LCL	6256	87542
LCM	7930	96954
ICL	5593	86615
ICM	10603	102017
ICH	17255	122131
CCL	0	22393
CCM	1323	63171
CCH	1623	81114
CPL	1125	48248
CPM	17889	54199
CPH	22881	137243
AC	21893	60772
D	45125	116835
M	0	0

2.4. Biological Traits Approach

In order to represent the benthic community in a simple and understandable way we chose a BTA instead of a species approach, i.e., species' morphological, behavioural and feeding characteristics were used instead of the species identity. Although not being as a

straightforward approach as the species' composition, the BTA approach offers several advantages. Species composition is highly influenced by local environmental conditions, which can mask the consequences of fishing impact. For instance, Bremner et al. (2003) found a geographical gradient in species´ composition along the English channel due to tidal action, sand transport and temperature gradient. However, this gradient disappeared in the BTA analysis that highlighted other functionality patterns. Therefore, as different species may share the same traits, the trait approach allows overcoming regional differences. Additionally, this approach proved successful in identifying fishing effects on benthic communities (Tillin et al. 2006, de Juan et al. 2007, Bremner 2008) and can be used as an indicator of ecosystem functioning (Lyons et al. this issue). Moreover, this approach reduces the number of variables (i.e. the species dataset is reduced from 300 to 22 variables) and thus simplifies the analysis.
However, despite the advantages of using this functionality approach (Muntadas et al. 2015), we emphasize the importance of the species identity that helps to understand the study systems, hence scientists should check the species data set to better understand the results of the functionality approach and to take biodiversity into account.

Five traits exhibiting different categories were chosen to characterize the epifaunal benthic community (Fig. 2). These traits, position on the seabed, feeding, motility, size and other attributes (e.g. fragility, vermiform, hard shell), were selected because they are easily assigned to any species and are essential attributes for an indicator of trawling impact (de Juan et al. 2009, de Juan & Demestre 2012) (Fig. 2).

Traits were assigned to each species based on available literature, experts' knowledge and information from the BIOTIC database (http://www.marlin.ac.uk/biotic/) (see Muntadas et al. 2015 for further details). In order to obtain the total trait abundance per replica, the abundance of all species in a replica having a particular trait category were added up (see de Juan et al., 2007 for further details of the approach).

Traits used in the platform				
Position	**Feeding**	**Motility**	**Size**	**Other attributes**
Emergent	Filter feeding	Sessile (attached)	Large >10 cm	Fragile
Surface	Deposit feeding	Sedentary	Medium 5-10 cm	No protection
Surface burrowing	Predators	Motile (crawlers)	Small <5cm	Flexible
Deep burrowing	Scavengers	Highly motile (swimmers)		Strong
				Regeneration
				Vermiform
				Hard shell

Legend
- Very resistant*
- Resistant*
- Vulnerable*
- Very vulnerable*

*Vulnerability regarding to trawling

Figure 2. Biological traits and their categories used in the platform.

2.5. Statistical approach

The model is based on multivariate regressions linking trait abundance to the trawling effort and to a sediment characteristic in each site.

a) Trait estimation

Present abundance of each trait category per replica was standardized to a 0-1 scale by dividing the abundance of the trait by the sum of the abundance of all trait categories in that replica. The standardized trait data was used along with sediment sand content data and the SSS effort data to estimate regression coefficients in the following expression:

$T^{*}=A_{S}\cdot SSS + A_{Sa}\cdot Sand + B$ (1)

Where T* is the estimated trait category abundance for a given fishing effort determined as trawl tracks density/km^2 (SSS) in a soft bottom with a particular sand content (Sand). Sand content represents the sediment characteristics (i.e. grain size), an important factor when evaluating fishing effects on benthic communities as different sediment types will be differently affected by trawling (Kaiser et al. 2006, Muntadas et al. 2015). A_{S}, A_{Sa} and B are the regression coefficients estimated for each trait category in figure 2. A Student test allows us to test the hypothesis of correlation with fishing effort. For the traits “small” and “predator” we cannot accept this hypothesis (p-value are 0.72 and 0.56 respectively) and hence the expression 1 cannot be applied to these traits and they are marked as “undetermined” in the simulation outputs. For some other traits (deep burrowing, surface burrowing, deposit feeding, high motile and flexible) this hypothesis remains questionable because p-value is in [0.10, 0.30], but we considered the expression 1 still valid for these traits under the practical significance hypothesis (Gibbons & Pratt 1975). However, these results show that these traits are little impacted by fishing effort.

To build the expression 1, we chose the SSS effort estimation instead of the fishermen interviews because, as the SSS provides an actual picture of trawl tracks on the seabed (Collie et al. 1997, Friedlander et al. 1998), it better reflects the potential effects of fishing effort on epifauna. However, SSS effort estimations are not always available, and fishermen interviews are more straightforward to obtain.

Therefore, by using the available estimations in our study sites, a regression law was defined to allow the calculation of effort in GT*days at sea/month from the trawl tracks density/km^2 estimation (fishing effort values are here divided by 100000 in order to be of the same order as traits variables):

Trawl tracks density/ km^2 = 1.76* GT*days at sea/month + $0.57_{(2)}$

This expression would be used if the user wants to add their own study site data and has no available SSS data to estimate the fishing effort.

All the above regression parameters were calculated using the software SPAD 2003 version 5.6.

Finally, the simulation tool calculates the trait category abundance in a given site under a new fishing effort (T') considering the current trait category abundance (T) using the following expression:

T'= T+ (T*'-T*) $_{(3)}$

Where T* and T*' are the estimated values of a given trait category for the current and the new effort respectively, calculated by the expression (1). T and T' are respectively the current and new estimated abundance of the trait category in that site. Using T*' and T* we obtain a gradient (T*'-T*) which is assumed to be caused by effort change and which is used to modify the present trait category abundance T. T' is the value represented in the simulation output.

b) Condition to pressure index (CPI)

In order to provide an idea of the average presence of vulnerable/resistant traits on a site, we developed a “condition to pressure” index calculated by the following expression:

$$\text{Index}= (2+2^* RR+R-V-2VV)/4 \quad (4)$$

Where RR is the sum of the abundance of all the very resistant traits in a site (in green in figure 2), R is the sum of the abundance of all the resistant traits in the same site (in orange in figure 2), V the sum of the abundance of all vulnerable traits (in yellow in figure 2) and VV the sum of the abundance of all very vulnerable traits (in red in figure 2).

This formula gives a value between 0 and 1, 0 meaning that all the traits in the site are very vulnerable to trawling and 1 meaning that all the traits are very resistant to trawling. Therefore, the value of this index resides in the representation with a single number of the overall community state regarding the pressure, i.e. the trawling effort.

As for the trait abundance estimation case (expression 3), in order to estimate the CPI under a new fishing effort (CPI’) the platform uses the following expression:

$$CPI' = CPI + (CPI^{*\prime} - CPI^*) \quad (5)$$

Where CPI* and CPI*’ are the estimated CPI values for the current and the new effort respectively, calculated by the expression (4) using T* and T*’ values respectively. CPI and CPI’ are respectively the current and new estimated CPI values. Using CPI*’ and CPI* we obtain a gradient (CPI*'- CPI*) which is assumed to be caused by effort change and which is used to modify the present CPI value.

3. Results

3.1. The knowledge platform

The platform is designed in four blocks (Fig. 3). The first block is dedicated to the platform presentation, where the user can find a brief description of the objectives and contents of this platform. A second block is dedicated to the platform methodology. Within this second block, the user will find three sections analogous to the previous sections 2.2, 2.4, 2.5 (i.e. fishing effort estimation, biological traits approach and statistical approach). In the section "biological traits approach", besides the biological traits' methodology, the user will find detailed information of the data used to build the simulations (Fig. 4). The third block describes the 18 study sites located in the Mediterranean where the sampling was carried out (see section 2.2).

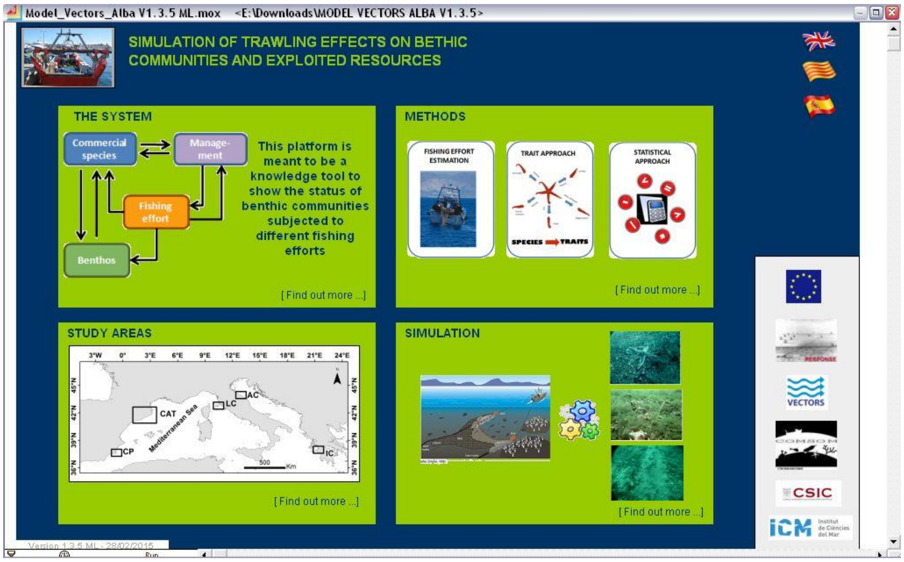


Figure 3. Image of the platform front-page that introduces the four blocks.

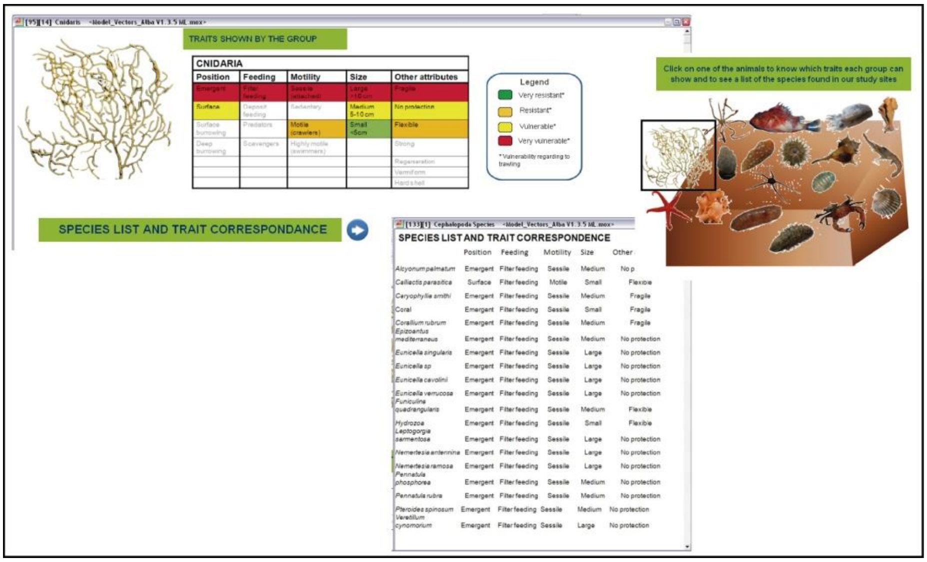


Figure 4. Information included in the "Materials and Methods" block within the section "biological traits' approach". Example providing information about Cnidarian biological traits and species recorded in the study sites.

Moreover, it provides the user with detailed information about the epifaunal community found in each site, as well as community comparison among sites (see an example in Fig. 5). Finally, the fourth block corresponds to the simulation tool, an example of which is provided in section 3.2.

These four main blocks are not totally independent from each other and the user may need information on, for example, the trait approach (block 2) when consulting community structure in a given area (block 3). In such cases the platform provides an information button to move quickly from one block to another (Fig. 6).

The platform has also the capacity to communicate with other software such as Adobe Acrobat to display the PDF files. This allows including further information about the topics displayed in each block (Fig. 6).

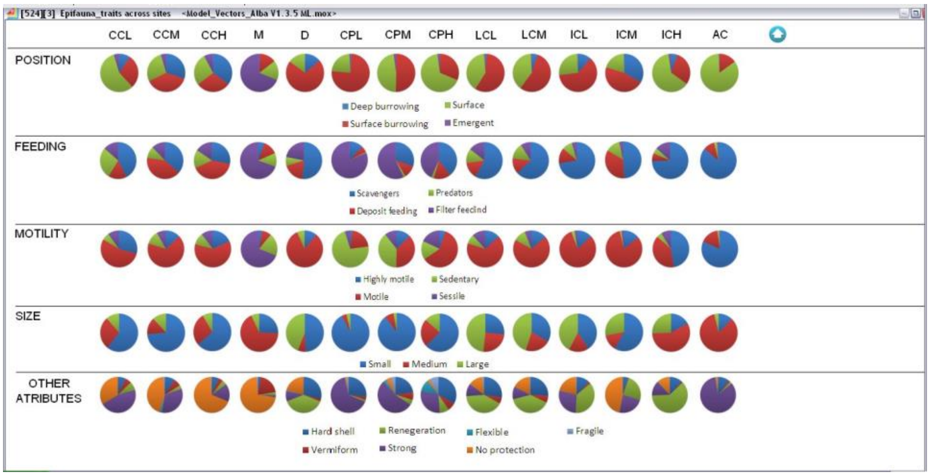


Figure 5. Example of trait composition comparison among sites included in the block "Study areas".

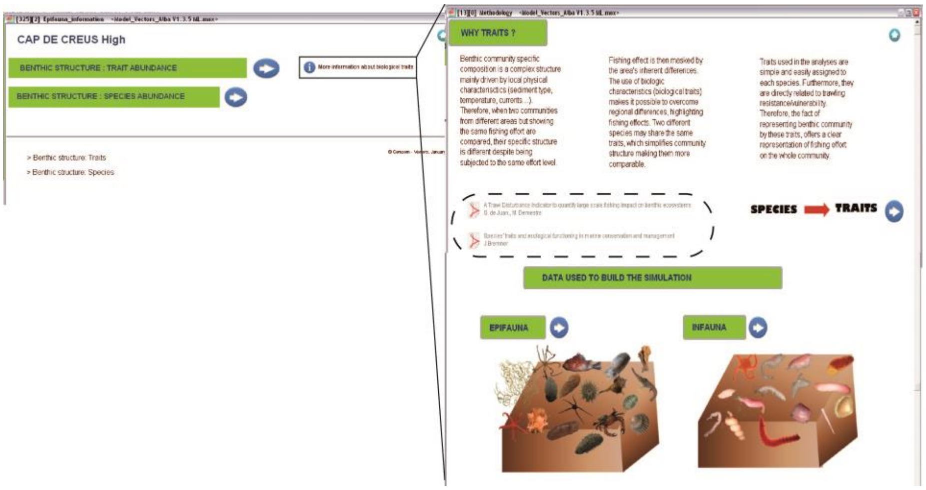


Figure 6. Example of the connection between blocks through the information button. The image shows an example of the "Material and Methods" block that includes a link to files that contain more information (highlighted by the stripped line).

3.2. Simulation block outputs

Figures 7 and 8 show the simulation results for and increase and decrease of 50% in fishing effort in the sites D, ICL, CCH and CPM. The site D was chosen because it is the muddiest site holding the highest effort (99.5% mud; 116835 trawl tracks density/km^2). The sites ICL, CCH and CPM are subjected to moderate trawling effort (86615, 81114 and 54199 trawl tracks density/Km^2 respectively) having variable sediment content (96.3%, 61.6% and 32.5% of mud respectively). Therefore, the simulation outputs of these four sites will be useful to compare 1) the response of benthic communities inhabiting similar habitats under different effort regimes (D and ICL) and 2) the response of benthic communities from different habitats under similar effort regimes (ICL, CCH and CPM).

Generally, as expected, the very vulnerable traits decreased when increasing the effort and vice-versa. Regarding the very resistant traits, the opposite happened; these traits predominantly increased when increasing the effort and vice-versa. However, some unexpected outcomes aroused. Surprisingly, the trait “large” size trait increased when augmenting the fishing effort and decreased when reducing effort in all cases. Also, “fragile”, a very vulnerable trait, increased when the effort was raised and decreased when the effort was diminished at all sites with the exception of D site. Another inconsistency was detected in D site as when reducing the effort almost all very vulnerable traits (except for “filter feeding”) decreased; however, these very vulnerable traits also decreased when the effort increased. On the other hand, the very resistant trait “vermiform” decreased in all sites when augmenting the effort. The trait “deep burrowing” decreased in both cases, when augmenting and when reducing the fishing effort in CPM site (Fig. 7 and 8).

CPI values increased in all sites when augmenting the effort and decreased when reducing the effort, but the magnitude of increase was higher in the less muddied sites, while the magnitude of decrease was higher in the muddier sites (Table 2).

Table 2. Condition to Pressure Index values of D, ICL, CCH and CPM sites under the current effort pressure and for 50% increase and decrease of the current effort.

	50% effort increase	Current effort	50% effort decrease
D	0.686	0.643	0.504
ICL	0.654	0.630	0.531
CCH	0.652	0.552	0.533
CPM	0.664	0.541	0.523

4. Discussion

The knowledge-based platform was built combining the capacity to represent information in an understandable way with simulation capacities (Ballé-Beganton et al. 2010). The presented platform fulfills these requirements as we have developed a simulation procedure from raw benthic species´ abundance data that is compiled in a biological traits table (Fig. 7 and 8). As both capabilities, providing information and simulation, are difficult to get with a single modeling tool, the platform has the possibility of interfacing with other software (e.g., PDF reader, GIS and other executables). Another important property that the platform should exhibit for a fruitful dissemination towards managers and decision-makers is the portability. The portability is attained with the availability of a free demo version of ExtendSim® which is able to load and run a full model. The portability to an Apple Mac version of ExtendSim® has not been attempted but this might be done with minor adjustments. We have to mention that a difficulty of the approach resides in the design of software that

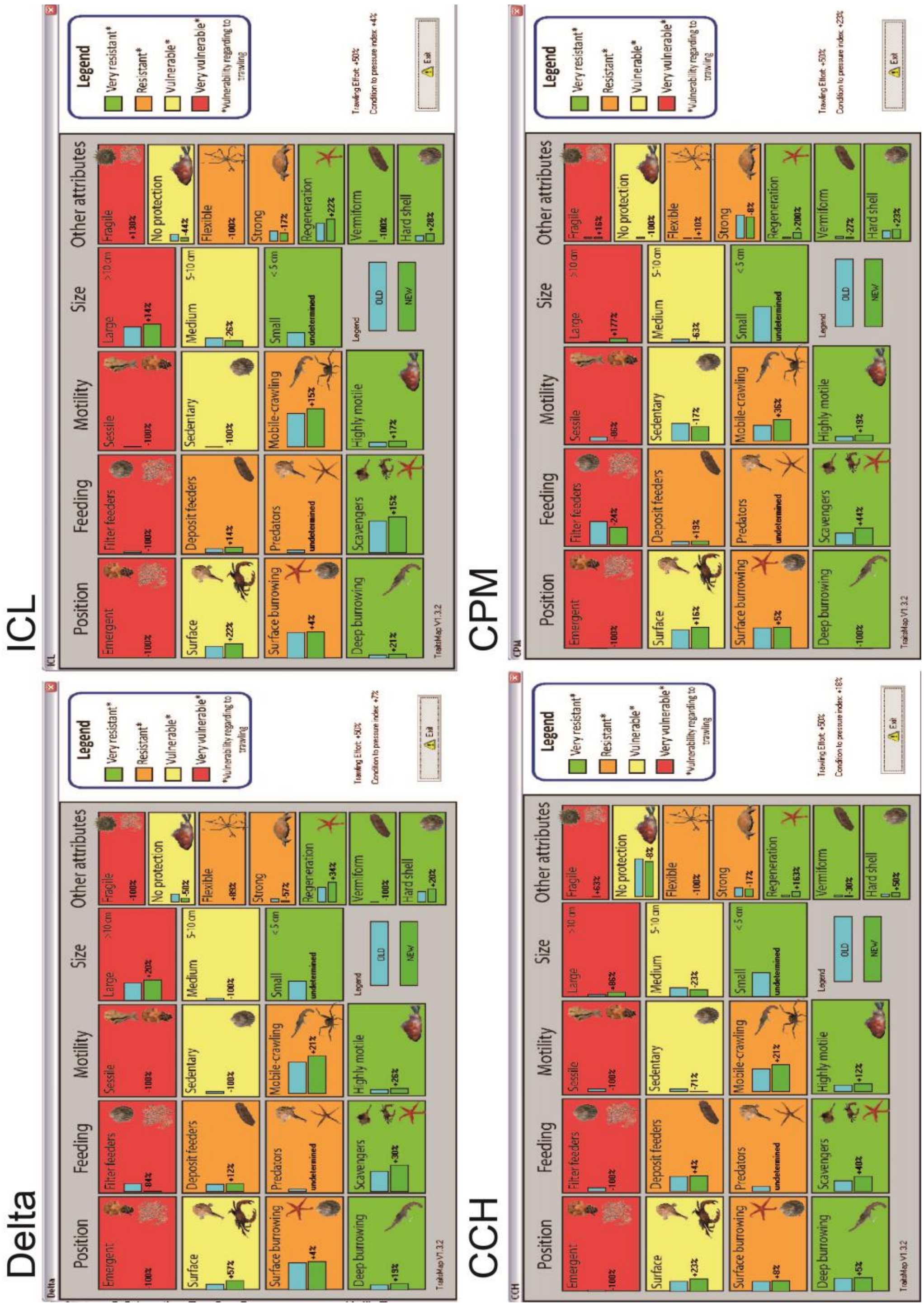


Figure 7. Simulation for a 50% increase in fishing effort in four different sites: Delta, ICL, CCH and CPM. Blue bars (“old”) represent trait abundance under the current fishing effort, while the green bars (“new”) represent trait abundance under the simulated effort (i.e., 50% increase).

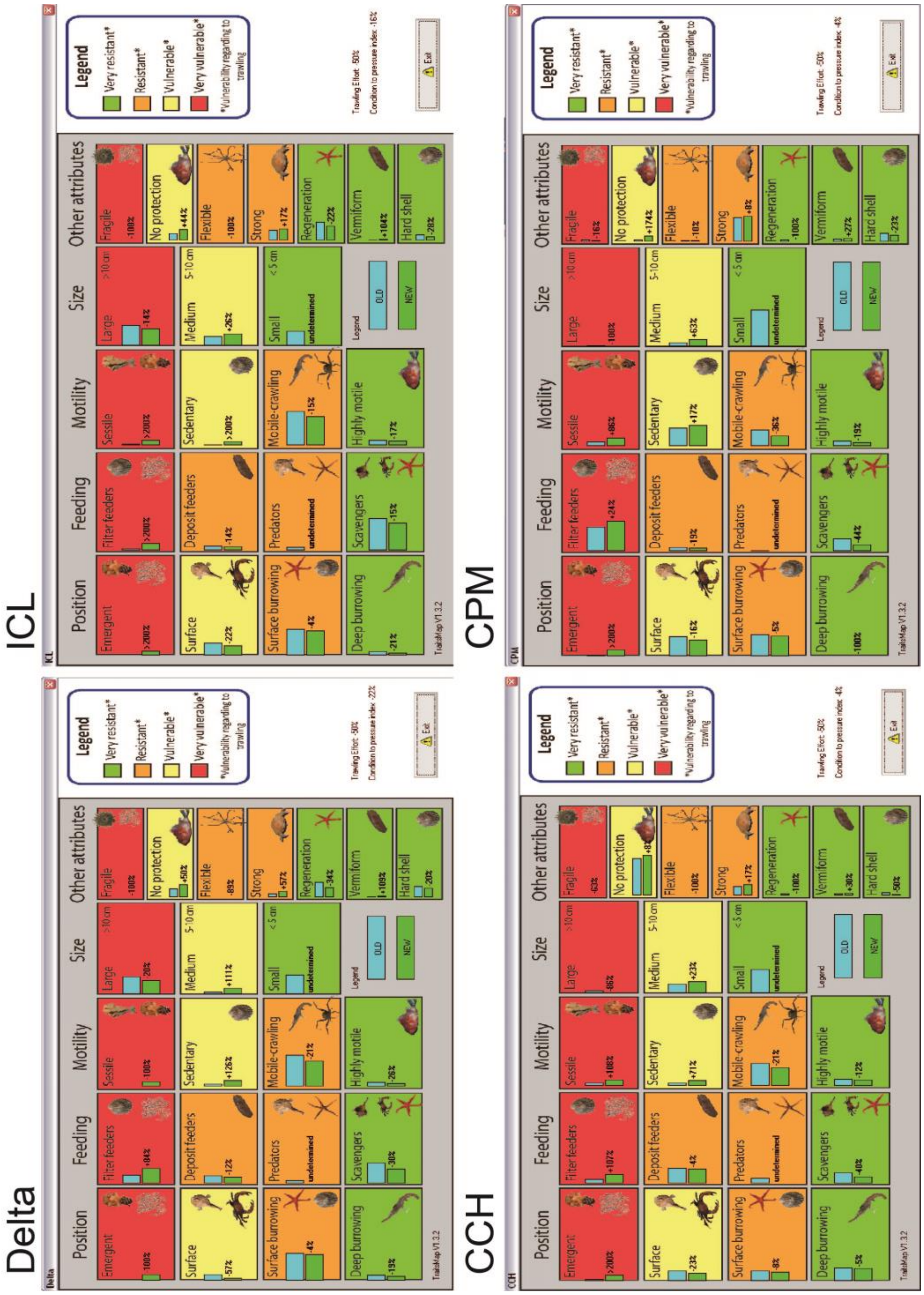


Figure 8. Simulation for a 50% decrease in fishing effort in four different sites: Delta, ICL, CCH and CPM. Blue bars ("old") represent trait abundance under the current fishing effort, while the green bars ("new") represent trait abundance under the simulated effort (i.e., 50% decrease).

respects its decision-support-tool finality. It remains an open problem to design graphical representations of scientific results that are accessible to stakeholders and are not obliterated by science standards and conventions.

Despite a large amount of data was available, a model forecasting the impact of trawling on one trait cannot be yet envisaged. The primary reason lies in the lack of temporal data for measuring the impact of an increasing or decreasing trawling on the different study sites. But we can argue that all traits were found in all sites, the abundance of the categories within them varying depending on the effort level and their vulnerability to trawling. Moreover, the described trait categories corresponded to a common ecosystem (soft-bottom on continental shelves in the Mediterranean Sea) and the trawling practices were similar in all sites. In consequence, there are elements that argue in favor of a kind of ergodicity that should overcome the lack of temporal data. Nevertheless, our model needs to be seen in the light of this hypothesis and cannot be considered as a model for prediction and forecasting of changes in benthic communities, but as a tool to show trends of potential changes on benthic communities. Consequently, our model should be framed in the proper perspective of a pragmatic communication and information exchange tool among fisheries' actors.

Simulation outputs generally showed a vulnerable traits´ decrease and a resistant traits´ increase when fishing effort was raised. However, contrary to what we expected, fragile organisms, which are known to be vulnerable to fishing (de Juan et al. 2009), increased when increasing effort and decreased when diminishing effort. Nevertheless, the category fragile organisms include some bivalves and sea urchins

that may also exhibit resistant traits such as small sizes or burrowing behaviour. Moreover, the trait “fragile”, despite being “very vulnerable” to trawl fishing, was not the main vulnerable trait driving differences between fished and unfished areas (generally, “filter feeding” and “sessile” were the very vulnerable determining differences) (Strain et al. 2012, Muntadas et al. 2014). Besides, the correlation of the trait “fragile” with fishing effort was not very strong (see section 2.5). On the other hand, the increase of the trait “large” size when decreasing effort might be due to the increase of large mobile/highly motile scavengers and predators such as sea stars and fish. As mentioned in section 2.4., results like this highlight the importance of going back to check species identity in order to better understand ecosystem functionality as biodiversity plays an important role in functioning (Palumbi et al. 2009, Snelgrove et al. 2014)
Moreover, it is important to stress that it is the overall trait abundance composition and not each trait category abundance on its own which allows to assess the general community status (de Juan & Demestre 2012). In this context, the simulation tool in the platform calculates the Condition to Pressure Index (CPI) to provide this overall community status overview.

According to the CPI, a fishing effort increase would mainly affect the sandy-mud CCH site and the maërl site CPM. Actually, fishing is known to greatly affect biogenic habitats as the maërl beds from CP area (Kaiser et al. 2006, Collie et al. 2009). The little increase of CPI in muddy bottom communities may either reflect a higher trawling resistance, or that these intensively trawled communities have already attained an alternative steady state dominated by organisms low vulnerable to trawling (de Juan et al. 2009, Thrush & Dayton 2010). Otherwise, when decreasing effort, these muddy bottom communities were the ones that, overall, recovered the most vulnerable traits, which suggests a higher recovery potential. These results are consistent with Strain et al. (2012) who found that some sessile filter feeding species associated to muddy bottoms, such as *Nemertesia antenina* or *Suberites carnosus,* grew rapidly when fishing effort decreased. However, it is worth to bear in mind that most of our study sites were muddy, thus predicted effects for these communities will be more consistent than for gravel/sand communities. This might be improved with the addition of new sites covering a wide range of fishing effort and grain size characteristics in the platform database. These new sites would be used to recalculate the regression parameters increasing their robustness.

Our simulation does not take into account recovery by migration of mobile adult organisms from neighboring sites or by larval dispersal at larger scales (de Juan et al. 2014). This omission could be one reason explaining why almost all very vulnerable traits in D site decreased when reducing the effort. The original abundance of the traits “emergent”, “sessile” and “fragile” in the D site (i.e. trait abundance at current/old effort) was 0, hence, no recovery could take place without migration. This is a very important aspect to take into account for further development of the simulation, as migration from nearby less damaged patches plays a very important role in the community recovery (Lundquist et al. 2010, Lambert et al. 2014). But the existence of these less damaged patches must be recorded and, to our knowledge, in D area the benthic communities from small to medium scales are heavily trawled (de Juan et al. 2009). This might also explain why deep-burrowing organisms did not increase in CPM when increasing the effort, as the original abundance of this trait was also 0. Moreover, maërl sediments such as those on CPM does not favor the settlement of this type of organisms (Bremner et al. 2006, Barberá et al. 2012). These results evidence an habitat-dependent response to trawling. But as mentioned earlier, in order to define consistent patterns for each habitat type, in the future more sites should be incorporate in the model. Therefore, this platform's results are rather conservative because, as we did not take migration into account, the recovery potential of our studied communities might be higher. Including recovery by migration, although increasing the accuracy of outputs, would also make the parametrization more complex as it would be necessary to consider the species' dispersal traits (i.e. type of larvae or reproduction frequency, which are unknown for many species (Tyler et al. 2012)) and the community patches' distribution at medium scales (Lundquist et al. 2010, de Juan et al. 2014). Our model, despite providing coarser estimations, is potentially applicable to any soft-bottom trawled area, as no previous knowledge of community distribution is required.

Currently, if the user is interested in testing the potential response of the benthic community from their site of interest, the platform allows for the addition of a new site where epifauna data, sediment sand content and effort estimation have been recorded. The user has to provide the standardised abundance of each trait category following the methodology of section 2.3 (an example with a small species set is provided in the platform's Materials and Methods block), the percentage of sediment sand content and an estimation of the trawling effort in their area of interest (calculated in trawl tracks density/km^2 or GT*days at sea). The effort of this new site could be changed in order to get a representation of the benthic community under this new effort, but the rules to estimate this new community traits' composition are those drawn from the study sites presented in this work. However, a link to connect this platform with R (www.R-project.org) is under development. This connection would allow including in the estimation the data from the new site to increase the robustness of the simulation.

Besides epifauna, another important component of benthic ecosystems is infauna, i.e. organisms living within the sediment (e.g. polychates, amphipoda, bivalves). Many target species of the trawl fleet feed on infauna, particularly during juvenile stages (Labropoulou et al. 1997, Hiddink et al. 2008). Thus, including an infauna compartment in the simulation will provide important information in order to build the link benthos-commercial species (many commercial species feed on infauna) as well as helping to have a more complete picture of the benthic community subjected to trawling disturbance.

Diet changes on target species due to fishing are well documented (Frid et al. 1999, Badalamenti et al. 2008), however, the overall effect of these changes are controversial. For instance, while Hiddink et al. (2008) found that fishing may benefit plaice (*Pleuronectes platessa*) in the North Sea, Hiddink et al. (2011) found the contrary to be true in the Irish Sea and other species in the same study seemed to be unaffected by trawling. Nevertheless, wider consensus exists in the fact that fishing homogenizes the sediment micro-structure and composition and simplifies 3D benthic structure, which might affect fish ability to find refuge against predators (Gregory & Anderson 1997, Thrush & Dayton 2010). Therefore, benthic community status plays a very important role in maintaining target species stocks and should not be ignored when building fisheries' management plans (Archambault et al, this issue). As shown by Muntadas et al. (2014), the functional trait composition of benthic communities is tightly lined with demersal fish species; the inclusion of this link would be an asset for the simulation platform and a further step to represent an integrated framework of trawled ecosystems.

Acknowledgements

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VI

CHAPTER 5:

Towards a Mediterranean EAF

Despite the implementation of EAF plans in some fisheries, seabed ecosystems and habitats are still largely ignored in management plans. Based on the knowledge gathered in this thesis as well as literature information, this chapter suggests an integral EAF management plan for the Catalan trawling fleet highlighting the importance of adopting benthic habitats' focused management measures.

Integrated thinking: a multidisciplinary approach for an adaptive enviro mental management of trawling fisheries.

Submitted to Environmental Science and Policy.

Pensament integrat: un enfocament multidisciplinari per a una gestió adaptativa de la pesca d'arrossegament al Mediterrani.

RESUM:

La filosofia de l'aproximació ecosistèmica a la gestió de les pesqueries (Ecosystem Approach to Fisheries – EAF), implica tenir en compte tots els elements del sistema pesquer (ecosistèmics, socials i econòmics) a l'hora de dissenyar un pla de gestió, tant en la seva implementació com en l'avaluació d'aquesta implementació. A més a més, en tot aquest procés és molt important que hi participin tots els actors involucrats en les pesqueries per tal que les regulacions establertes tinguin un alt grau de consens, la qual cosa comporta una major voluntat de complir-les.

Un dels elements ecosistèmics més importants en les pesqueries d'arrossegament són les comunitats bentòniques que estan sotmeses a l'impacte dels arts de ròssec. Malauradament, però, aquest és un element molt ignorat en els plans de gestió d'aquestes pesqueries.

Aquest treball recomana una sèrie de mesures i eines de gestió per a la implementació d'un pla de gestió integrat per a la flota d'arrossegament catalana que tingui en compte els ecosistemes bentònics a més a més de les espècies objectiu i els elements socio-econòmics. S'ha escollit la flota pesquera d'aquesta zona perquè ha estat àmpliament estudiada considerant diversos objectius (flota, esforç, captures, especies, etc.) i aquesta informació representa una

bona base per a definir possibles mesures de gestió. Aquest pla de gestió és extensible a qualsevol flota d'arrossegament mediterrània, ja que en general les característiques d'aquest tipus de pesca són semblants a tot el Mediterrani.

Per tal de minimitzar l'impacte sobre els ecosistemes bentònics, primerament s'haurien de prendre mesures per evitar l'augment de l'esforç pesquer (p.ex. aturar l'expedició de noves llicències), la qual cosa evitaria un augment de la pressió sobre especies i comunitats bentòniques que veuen afectades les seves funcions a alts nivells de pesca. Però mantenir el nivell d'esforç pot no ser suficient per a conservar la integritat dels fons marins. Per tant, també s'haurien d'incentivar millores tecnològiques en l'art que en minimitzessin l'impacte sobre el fons marí. Així mateix, establir vedes que coincideixin en el temps i en l'espai amb els períodes de fresa i reclutament de les espècies objectiu, alhora que redueix l'esforç sobre els estocs explotats, ajuda a disminuir l'impacte a què estan sotmeses les comunitats i els hàbitats bentònics i que ocasiona una desestructuració i pèrdua de diversitat important. Tot i així, les vedes temporals no permeten la completa recuperació de les comunitats bentòniques, i per tant també s'hauria de considerar la implementació progressiva d'algunes àrees permanentment tancades a la pesca que asseguressin la provisió dels serveis ecosistèmics d'aquestes comunitats.

Per altra banda, aquestes mesures adreçades a la protecció dels ecosistemes bentònics s'han de complementar amb altres mesures dirigides específicament a mantenir els estocs pesquers dins uns límits d'explotació sostenible: talles mínimes de captura coincidents amb la talla de primera maduresa, adoptar tècniques per a detectar quan s'estan pescant massa individus juvenils i establir vedes a temps real (és a dir, tancar temporalment una àrea quan s'hi detecti una elevada concentració de juvenils), o establir quotes de captura per a les espècies objectiu principals en les èpoques de fresa i reclutament.

Una eina útil en el marc de la gestió segons l'EAF pel que fa a la implicació dels diferents actors involucrats en les pesqueries, és la plataforma de coneixement presentada en el capítol anterior. Aquesta plataforma permet avaluar els possibles efectes d'una mesura que impliqui un canvi d'esforç pesquer en la comunitat bentònica i per a conscienciar els actors de la importància del paper d'aquestes comunitats en el sistema pesquer.

L'avaluació del pla de gestió, efectuada per un comitè de cogestió amb representació de tots els actors implicats, comporta dues fases. La primera fase consistiria en un seguiment científic exhaustiu per a ratificar el bon funcionament de les mesures establertes. La segona fase seria un seguiment a llarg termini de l'estat dels elements del sistema. Per a això calen indicadors adequats per a tots els elements del sistema: pesquers, ecològics, socio-econòmics i biològics. Si en alguna fase del seguiment es detectés que amb l'aplicació de les mesures establertes no s'aconsegueix el resultat esperat, s'haurien de revisar aquestes mesures (gestió adaptativa).

Integrated thinking: a multidisciplinary approach for an adaptive management of trawl fisheries in the Mediterranean.

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ABSTRACT

The implementation of the Ecosystem Approach to Fisheries (EAF) in trawl fisheries demands a paradigm change from traditional management in both, the management design and the monitoring of management effectiveness. Stakeholders should be involved from the beginning of the process and benthic ecosystem health status has to be taken into account when designing the EAF plan. In the Mediterranean fisheries, some advances have been made regarding stakeholder's participation in management plans, but benthic ecosystems and habitats are still largely ignored. In this work, we focus on a well-known North-western Mediterranean trawling fleet, the Catalan fleet, and recommend a series of measures, tools and indicators to successfully implement the EAF framework. Moreover, we suggest adopting a two phase monitoring program: phase one consisting on a thorough scientific evaluation of the EAF regulation measures outcomes and phase two consisting on a long-term routine monitoring. Finally, we stress the importance of following a multiactor

participatory approach, which may enhance appropriate regulations' enforcement.

Keywords: Benthic communities, EAF, monitoring, stakeholders, trawling, Mediterranean

1. The Ecosystem Approach to Fisheries paradigm

The emergence of the Ecosystem Approach to Fisheries concept in 2002 consolidated the increasing concern on the environmental effects of fishing (FAO 2003). The overall objective of the EAF is to promote sustainable fisheries while maintaining healthy marine ecosystems. This objective should be achieved by protecting the ecosystem structures and processes that are necessary for the maintenance of fisheries and, in general, of the goods and services societies demand (Link 2002, Thrush & Dayton 2010). Hence, an EAF management plan requires an integrated approach that takes into account ecological (biotic and abiotic components) as well as socioeconomic elements (Levin et al. 2009). EAF management plans have been implemented for a number of fisheries around de world (e.g. Australia (McLoughlin et al. 2008), US (Link et al. 2011) or Norway (Ottersen et al. 2011)), but all of them present execution problems and are not fully implemented.

Fisheries' bottom towed gears (i.e. trawls and dredges) cause the most severe damage on seabed ecosystems, as they chronically modify benthic communities and habitats' structure (Dayton et al. 1995, Jennings & Kaiser 1998), impairing many ecosystem processes (Thrush & Dayton 2010). Soft-bottom communities from trawling grounds contribute to relevant ecosystem processes (e.g. bioturbation, habitat structure, or carbon sequestration), which in turn support many ecosystem services (e.g. nutrient cycling, habitat provision, or climate regulation) (Snelgrove, 1997; Snelgrove et al., 2014). Moreover, the most important direct service from fishing grounds, target species, depends on some of these ecosystems processes as habitat structure (Muntadas et al. 2014). Trawling causes changes in the ecosystem functional components that may compromise these processes and related services (Muntadas et al. 2015). The Marine Strategy Framework Directive (MSFD) (EC2008/56) compels the European Union member states to achieve a Good Environmental Status (GES) of marine ecosystems by 2020. MSFD defines eleven descriptors to achieve a GES, the most representative in the fisheries framework being D1 (biological diversity), D3 (fish stocks), D4 (marine food webs), D6 (Sea floor integrity) and D9 (contaminants in fish and seafood). Unlike the other descriptors, D3 and D9 have already been considered in the traditional Target Resource Oriented Management (TROM) paradigm. Trawling mainly affects D6 which is related to D1 (one of Sea Floor Integrity attributes is species composition (Rice et al. 2010)) and in demersal fisheries it can also be related to D4, as demersal fish prey on benthic organisms (Denderen et al. 2013). Therefore, Sea Floor Integrity, as well as ensuring the provision of ecosystem services for future generations, would ultimately benefit target species, contributing to the advance towards sustainable fisheries.

Scientific knowledge on benthic ecosystem structure and functional role, their response to trawling disturbance are essential to adopt an

EAF. However, information on fisheries' socioeconomic elements is also necessary (Hughes et al. 2005) and, for a successful EAF implementation, a key element is stakeholders' involvement in the process (Reed 2008, Curtin & Prellezo 2010). Stakeholders' participation in the management plan enhances transparency and promotes that fisheries' actors feel co-responsible of the final agreement which increases their willingness to enforce management measures (Reed 2008). Fisheries' stakeholders (e.g. policy makers, scientists, fishermen, fishery industry, consumers and NGOs) must agree on the measures to be implemented and must understand the importance of adopting these measures. Hence, it is essential to promote a participatory framework and to develop tools aiming to make scientific information available and understandable for non-scientific stakeholders (Leslie & Mcleod 2007, Soomai et al. 2011). Therefore, the advance towards a general implementation of EAF demands a major change in the fisheries' management philosophy, as it implies dealing with multiple disciplines and multiple objectives, and expanding the management scope (i.e., cooperative and multilevel co-management) (Berkes 2012).

Another important step for the adoption of an EAF is the implementation of an adaptive management plan (Levin et al. 2009) and to evaluate its effectiveness through a monitoring program (Leslie & Mcleod 2007, Levin et al. 2009). In this way, management measures may be changed if the management objectives are not attained. Nowadays, monitoring programs are already implemented for a number of fisheries (e.g. Link et al. 2011; Lleonart et al. 2014; Ottersen et al. 2011), but metrics are still mainly focused on target and bycatch species. Therefore, monitoring programs also demand a philosophical change, as, in accordance with EAF objectives, the ecosystem integrity has to be taken into account when assessing the outcomes of a management plan.

The EAF represents an overall paradigm change from the Target Resource Oriented Management (TROM), which focuses on fleet dynamics and target species. The EAF goes beyond the TROM and, besides fleet dynamics and target species, involves an integral vision of the fisheries system that takes into account all the components of this system, including ecosystems and societies.

2. Present situation in the Mediterranean Sea

Fisheries in the Mediterranean have a long tradition and constitute an important socio-economic sector (Lleonart & Maynou 2003, Franquesa 2004). On the other hand, the Mediterranean Sea harbors high biodiversity and many sensitive habitats, such as deep water corals or submarine canyons (Coll et al. 2010, de Juan & Lleonart 2010), which should be preserved from anthropogenic disturbance. Another distinctive feature of the Mediterranean is its complex cultural, geographic, biological, economic and legal-political scenario, which difficult the adoption of common fisheries policies (Casado 2008; Caddy 2012). The Mediterranean is surrounded by 21 states, only 8 belonging to the EU, and some North African and Levantine countries have no funding and/or no interest on ecosystem conservation and restriction of fishing activities (Casado 2008; Cinnirella et al. 2014).

Despite a range of regulations advocating for an EAF in all the bordering countries (Mediterranean Ecosystem Approach Strategy - ECAP (Cinnirella et al. 2014)) and in the EU countries (Marine Strategy Framework Directive- MSFD (EC 2008/56), or the latest review of the Common Fisheries Policy - CFP (EU 1380/2013)), fisheries are still far from an integral adoption of an EAF (however, note that these regulations are only binding for EU countries). There are existing regulations aimed at preserving sensitive habitats, such as the spatial ban for trawl gears within 3 nautical miles off the coast or below 50 m deep (to protect *Posidonia oceanica* medows (EC 1626/94)), and the ban for trawling below 1000 m deep (to protect deep-sea slow-growing, and hence vulnerable, organisms (EC 1967/2006)). On the other hand, in order to increase fishing gear selectivity, and contribute to a reduction of the ecosystem impact of trawlers, the European Regulation 1967/2006 established a change in the mesh shape from 40 mm diamond to 40 mm squared or 50 mm diamond. This regulation implied also the ban for some artisanal Mediterranean boat seine fisheries, which were considered as towed gears, that operate in shallow waters (7-30 m) using small mesh sizes (Lleonart et al. 2014). In some cases, this situation drove to the emergence of participatory management approaches to ask for a repeal of the law for this this small-scale fishery as the targeted species cannot be fished with another gear type. Local co-management plans were applied for the sand-eel fishery (*Gymnammodytes cicerelus and Gymnammodytes semisquamatus*) (Lleonart et al. 2014), for the picarel (*Spicara smaris*) (Bacci 2009) and for transparent gobi (*Aphia minuta*) (o.p. ittici sud adriatico 2014; BORM 2013; Lleonart et al. 2014). Although these plans imply an advance towards an EAF regarding stakeholders' involvement, none of them consider the effects on the ecosystem components beyond the by-catch species and sensitive habitats such as *P. oceanica* meadows. Other management plans based on the participatory approach for semi-industrial Mediterranean fisheries have also emerged in recent years, e.g. the shrimp trawl management plan for Palamós fleet (North Catalonia, Fig. 1) (Gorelli et al. 2014) and the plan for North Adriatic small pelagic fisheries (Recommendation GFCM/37/2013/1). Again, these plans represent an advance regarding stakeholders' involvement, but the effects on the ecosystem components are not considered.

2.1. Characteristics of trawl fisheries in Catalonia

The present document focuses on an extensively studied area in the Mediterranean Sea, the Catalan coast (north-east Spain). Based on the acquired broad knowledge on the trawl fisheries system in the area, we recommend an EAF management plan for the trawl fleet in the Catalan autonomous region, which might be extrapolated to other Mediterranean trawl fisheries where such an extensive system knowledge is also available. Trawl fisheries along the coast in Catalonia have been analysed from different perspectives: fisheries (Demestre & Martín 1993, Lleonart & Maynou 2003, Sánchez et al. 2007, Martín, Muntadas, et al. 2014), target species (Demestre et al. 1997, Recasens et al. 1998, Martin et al. 1999), bycatch species (Carbonell et al. 1998, Martín 2001, Sánchez et al., 2007), discards effects on seabirds (Oro 1996, Navarro et al. 2009) and on benthos (Bozzano & Sardà 2002) and impacts on the seabed biological communities (Sanchez et al.

2000, de Juan et al. 2007, 2009, Demestre et al. 2008) and physical habitats (Palanques et al. 2001, Martín et al. 2014). The management plan recommended in this work takes into account all the existing knowledge, and specific examples are based on red mullet (*Mullus barbatus*). This species was chosen because it represents a major target species for the Mediterranean trawl fisheries and a key species in the study area (Demestre et al. 1997, Martin et al. 1999). Furthermore, red mullet has a well-known biology, distribution and related fisheries dynamics (Suau & Vives 1957; Demestre et al. 1997, Lombarte et al. 2000; Aguirre 2000).

The Catalan trawling fleet has decreased markedly during the last decade, from 356 trawlers registered in 2003 to 256 in 2014 (http://agricultura.gencat.cat). This fleet is distributed amongst 14 harbours (Fig. 1). A two-month seasonal closure is implemented in the ports from the southern region (hereafter Tarragona's ports, which constitute 49% of the total Catalan trawl fleet): in May-June in the northern ports, and in July-August in the southern ports (Fig. 1a). It is worth to mention that the next fishing port south of Catalonia, but located in a different administrative autonomous region (Vinarós, Fig. 1a), also implements a two-month closure in July-August. The other harbours along the Catalan coast may choose to implement a one-month closure whenever they deem it necessary, but there is no regular closed season implemented. In the northern Tarragona, the May-June closed season is claimed to target red mullet spawning, as it overlaps the species' spawning season (Suau & Vives 1957). On the other hand, the July-August closed season in southern Tarragona ports

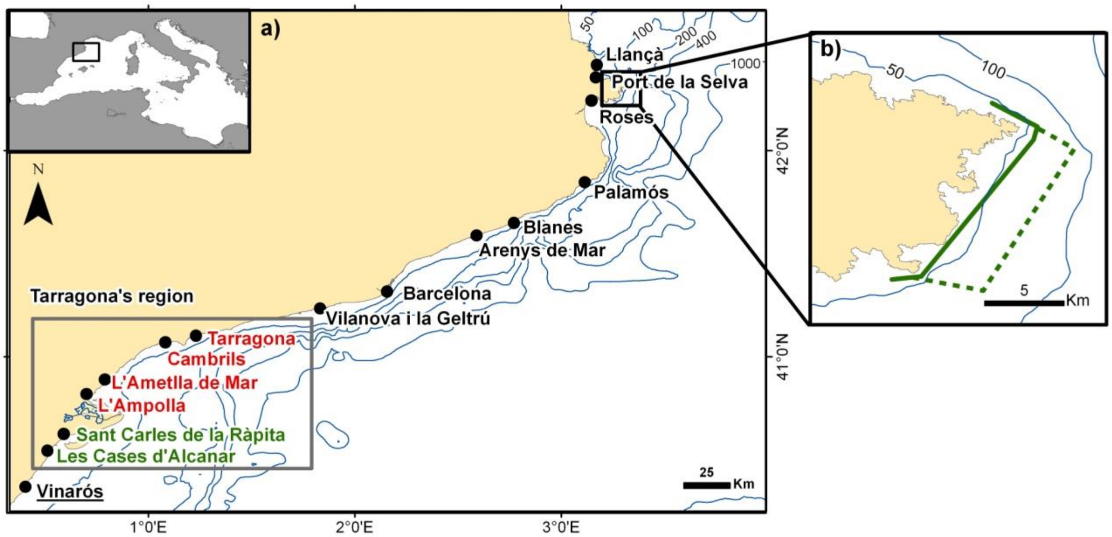

Figure 1. a) Situation of the 17 harbors in Catalonia autonomous region. The grey rectangle includes Tarragona's region ports, the northern ones (typed in red) having a closed season in May-June and the southern ones (typed in green) having a closed season in July-August. Underlined port name corresponds to other administrative autonomous region, but it also has a closed season in May-June. b) Zoom of the Cap de Creus area showing the limits of the present MPA (solid line) and the suggested extension (stripped line).

matches the start of red mullet's recruitment period, when larvae settle on benthic habitats (Suau & Vives 1957, Demestre et al. 1997). This closed season is aimed at protecting red mullet early recruits. However, the market value for red mullet's recruits is higher than for adults. Therefore, higher market demand drives fisheries dynamics, as fishermen tend to target specifically young recruits after the closed season, which leads to a peak in September/October catches (Demestre et al. 1997, Muntadas et al. 2014). Nevertheless, the percentage of catch below the legal minimum landing size (11 cm; EC 1967/2006), has decreased in the last decade, probably due to the implementation of the more selective 40 mm squared cod-end (STECF

2014). However, despite these measures, red mullet stocks are still overexploited in the area (STECF 2014). Besides the problem of overexploited stocks, the management measures implemented in Catalonia region are target species' focused measures ("TROM" paradigm) that do not take into account an EAF integral framework.

To evaluate the exploitation status of a stock, it is essential to correctly classify the landed species. Moreover, the combination of landings' data with fleet dynamics and target species' biology allows to determine when and where the fleet concentrates to catch a specific species, which may hint when and where a closed season should be implemented to protect periods of the life-history of a species (Martín, Muntadas, et al. 2014). In Catalonia, landings data are available on a daily basis from auction reports, as vessels return every day to their home-port to sell their caches. These auction records should properly reflect landings composition. However, this is not always the case. For example, two different mullet species coexist in the Mediterranean: *Mullus barbatus* and *Mullus surmuletus* and the existing auction reports do not tell apart the two species, refering indifferently to *M. barbatus* or *M. surmuletus,* or reporting the catches under the generic *Mullus spp.* Nevertheless, trawl catches are mainly composed by *M.barbatus*.

3. An integral and adaptive EAF management plan: a case study in the Catalan trawl fisheries framework.

Figure 2 synthetises the necessary elements for an EAF management plan for the Catalan trawl fleet. It highlights the importance of stakeholder involvement in all EAF steps (EAF plan design and EAF plan enforcement and monitoring) as well as the importance of the adaptive nature of the plan.

Some of the suggested tools, indicators and measures are already used in the traditional TROM approach, but here are considered in the broader and integral EAF.

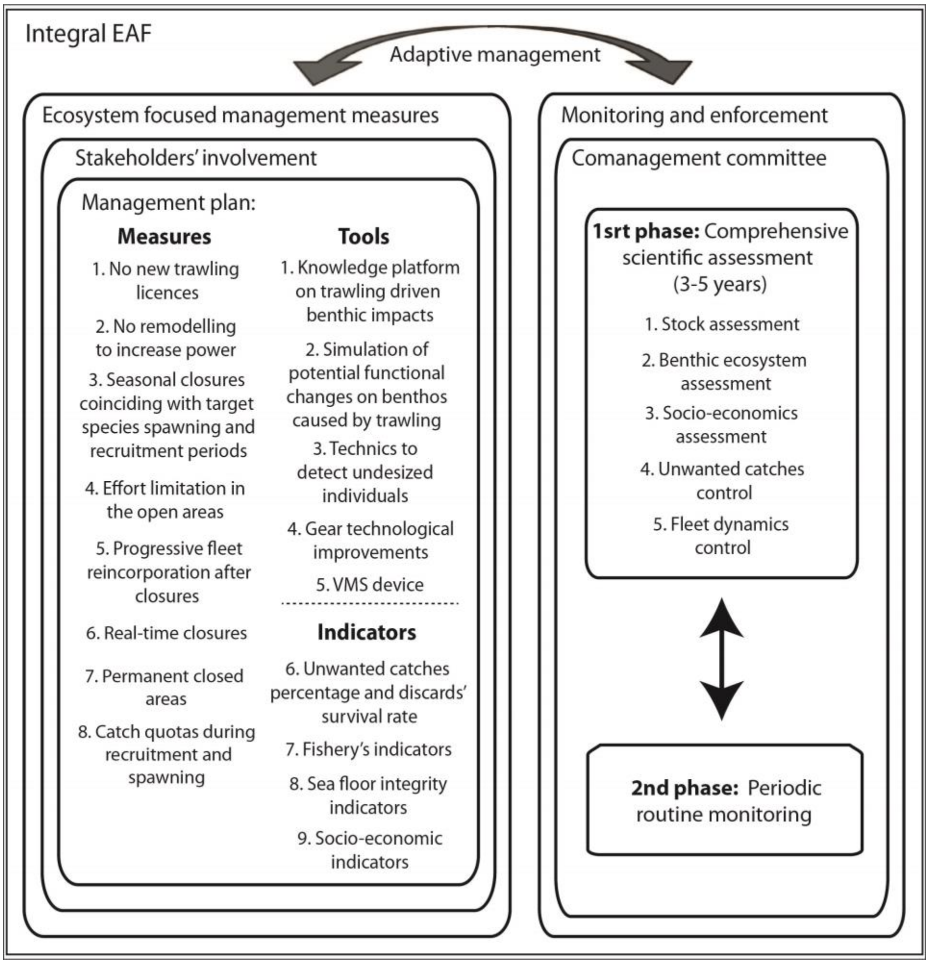

Figure 2. Integral EAF management plan suggested for the Catalan trawl fleet. See the text for further details.

3.1. Paradigm change: ecosystem-focused management measures.

The following subsections suggest how an integral EAF management plan, taking into account the "Sea Floor Integrity" along with target species stocks and socioeconomic interests, might be attained for the Catalan trawling fisheries system. The recommended management measures consider red mullet fishery as a case study, but they may be extended to other target species (e.g. hake, *Merluccius merluccius* or angle fish, *Lophius spp.*) and to all trawl activities in the Mediterranean. Paragraphs in italics highlight the recommended tools and regulation measures for the management plan (Fig. 2), while paragraphs in bold represent an example of applying these measures in well-known specific cases/areas of the Catalan coast: the red mullet fisheries' and the Cap de Creus area (Fig. 1b).

3.1.1 Ecosystem focused measures: protecting seabed communities and target species

As previously mentioned, benthic ecosystems from soft-bottom fishing grounds perform many relevant ecosystem functions (Snelgrove 1997; Snelgrove et al. 2014). One of these functions is secondary production, which is linked with the food source for target species (Muntadas et al. 2014). In general, scientific studies observed that the production of muddy-bottom communities is negatively affected by high levels of trawling (Queirós et al. 2006, Bolam et al. 2014). Most of the trawling grounds along the Catalan coast are located on muddy bottoms (Lleonart 1990), hence, a fishing effort increase might compromise secondary production. Moreover, benthic ecosystems subjected to high levels of trawling pressure harbour less structured benthic communities than those subjected to lower fishing pressure (de Juan et al. 2009, de Juan et al. 2011). This highlights that the service "provision of habitats" for demersal species might be compromised in heavily trawled areas. Furthermore, other important functions such as carbon sequestration and bentho-pelagic coupling are also compromised at high levels of trawling (Muntadas et al. 2015).

> *In order to avoid an increase of the present fishing effort, which would negatively affect seabed communities, no new trawling licences should be given, the total number of fishing hours should not be increased and the existing trawl vessels should not undergo remodelling that increase its power (Fig. 2).*

A regulation that ensures the *status quo* in the fishing effort might not be enough to ensure seafloor integrity, as the abovementioned ecosystem services and processes could be already compromised (e.g., in the most heavily fished areas the habitat provision service is almost null) (de Juan et al. 2011, de Juan & Demestre 2012, Muntadas et al. 2015). Moreover, important target species, such as red mullet or hake, are overexploited at the current effort levels (STECF 2014). Closed seasons are a measure that contributes to fishing effort reduction, releasing benthic communities from trawling pressure for a specific period. Closed seasons are already in force in some areas of the Catalan coast, but they are not homogeneously implemented. If these closed seasons coincide spatio-temporally with especially vulnerable life cycle stages of target species, such as recruitment and spawning, they would contribute to prevent overfishing and they would also protect benthic communities constituting their Essential Fish Habitats.

In order to reduce trawling disturbance on benthic communities constituting EFH, as well as preventing target species overfishing, temporal closures coinciding spatiotemporally with spawning and/or recruitment might be implemented (Fig. 2).

Focusing on the red mullet's example, a closure around 150 m during spring and early summer (May-June) (the area and period of red mullet's spawning (Suau & Vives 1957)**) and a closure around 50 m in summer (July-August) (red mullet early recruits migrate from 20-40 m to deeper waters on summer** (Suau & Vives, 1957)**) might be implemented in the Catalan coast. In this way, there would always be an open area for trawlers, which would allow fishing activity all year round and, hence, ensure fresh fish supply for markets.**

However, an effort limitation to protect benthic communities (e.g., limiting the number of vessels per day) in the areas where fishing is allowed in both closed seasons (during spawning and during recruitment) should be also implemented, as the closure of one area might lead to an effort concentration in the area where fishing is allowed (Fig. 2).

Then, special attention should be paid to the fleet dynamics during the recruitment period occurring after the closed season. When small individuals of the target species have a higher value than the larger ones, the fleet concentrates after the closed season to target young recruits, as it happens with red mullet in *Sant Carles de la Ràpita* fishing grounds (Fig. 1) (Demestre et al. 2008, Muntadas et al. 2014).

A regulation aimed at implementing a progressive reincorporation of the fleet after the closed season that overlaps with recruitment periods of the target species should be also implemented. For example, during the month following the closed season each individual trawler might operate only in alternate days or during fewer hours per day (Fig. 2).

Nevertheless, it must be kept in mind that seasonal closures are too short to allow benthic community recovery (Kaiser et al. 2006, Demestre et al. 2008) and the effort reduction measures suggested might be insufficient to minimise negative impacts on benthic ecosystems and ensure an appropriate provision of certain ecosystem services (e.g. habitat provision) (Muntadas et al. 2015). For example, the non-recovery of north Atlantic cod stock, despite the fishing moratoria, might be related to the loss of appropriate nursery habitats for young recruits (Bundy & Fanning 2005, Gårdmark et al. 2011). Sensitive habitats, including those harbouring high biodiversity or presence of vulnerable organisms, and Essential Fish Habitats, that might be composed by complex benthic structures, should be protected from trawling in order to prevent further degradation (de Juan & Lleonart 2010, de Juan et al. 2012).

Permanent closed areas should be progressively implemented in particularly vulnerable areas to ensure seafloor integrity and hence the provision of benthic communities' ecosystem processes and services at a long term (Fig. 6).

For example, to optimize already existing measures, the current MPA in Cap de Creus (North Catalonia, Fig. 1b) could be extended offshore, as it harbours complex structured communities (Lo Iacono et al. 2012, de Juan et al. 2013, Muntadas et al. 2015)**. This example is in line with the Convention of Biological Diversity (CBD) targets, as it aims at protecting representative habitats to reach the objective of protecting 10% of marine coastal areas by 2020 (https://www.cbd.int/sp/targets/). Moreover, the fishing ground that would be closed is located very near to the coast and is frequently inaccessible due to bad weather conditions. Hence, we presume that the extension on this MPA would have little consequences for the local fleet and would be acceptable for all fisheries stakeholders.**

Permanent closures are difficult to implement due to fishermen reluctance, but designing MPA size and location within a regional MPA network may enhance both, fisheries and conservation benefits (Salomon et al. 2011). For instance, Fiorentino *et al.* (2008) observed that a trawl closed area benefited red mullet population, and Morris and Green (2014) discussed how a MPA combined with other effort control measures around the permanent closed area may enhance cod fishery. Moreover, as suggested in the above example, the extension of existing MPAs offshore, when done through a participatory an transparent process (note the importance of stakeholders participation throughout the process), has proved to result in an effective enforcement of a MPA network (Gleason et al. 2010).

Additional measures to protect target species populations might also be adopted. Firstly, catching immature individuals should be avoided.

Minimum landing sizes should be in accordance with length at first maturity (Fig. 2)

For example, red mullet reach sexual maturity in its first year, males at 12.5 cm and females at 13 cm (Aguirre 2000)**, while minimum landing size is 11 cm. Hence, minimum landing size for this species should be at least of 13 cm.**

On the other hand, the use of echo sounder and technics of underwater vision techniques (Rosen et al. 2013) that enable the visualization of the potential catches, allow fishermen to decide whether the haul should be conducted depending on the proportion of undersized individuals detected by these techniques. Moreover, these techniques might lead to the implementation of the real time closures, which consists on temporally closing an area if a critical mass of undersized individuals is detected. This area would be considered as a potential nursery area (Kempf 2010, Needle & Catarino 2011).

Advanced techniques to detect and avoid undersized individuals should be adopted. If the catch in a specific area is mostly composed by undersized individuals, vessels should move to alternative fishing grounds and that area might be temporally closed (real time closure) (Needle & Catarino 2011) *(Fig. 2).*

Catch quotas in the Mediterranean are only implemented for tuna and tuna-like species, as the multi-specific catches of trawl vessels difficult the implementation of quotas (Cacaud 2005; Caddy 2012). However, to protect spawners and young recruits, catch quotas on these especially vulnerable live stages could be established.

Catch quotas for the main target species during recruitment and spawning periods might be implemented (Fig. 2).

As previously mentioned, red mullet recruits are targeted by fishermen after the closed season in summer, as recruits are highly appreciated by consumers. Therefore, a catch quota for these young recruits could be implemented.

All the recommended restrictive effort and catches measures are not normally welcomed by fishermen, as reducing fishing time/areas and establishing quotas means reducing catches. However, as there would be less fish available in the market, prizes are expected to rise and fishermen working fewer hours would represent a social improvement, as it happened in the sand eel fishery experience from the study area (personal communication). Moreover, target species focused regulations are specifically aimed at increasing resources stock, which might lead to a mid/long-term increase in catches.

Further measures for habitat protection are gear technological modifications to reduce physical disturbance of benthic ecosystems. These measures may be aimed at reducing bycatch by modifying the net with different devices (Brewer et al. 1998), or at mitigating seabed impact by modifying trawl doors, weights, sweeps and bridles (Valdemarsen et al. 2007). These technical modifications should not significantly affect fishing effectiveness in order not to provoke an increase of the overall fishing effort (Valdemarsen & Suuronen 2003).

Gear technological modifications to reduce physical disturbance on benthic communities and habitats should be tested and adopted (Fig. 2)

3.1.3 Management tools and indicators

A key element for a successful EAF management is stakeholders' engagement and comprehension of the problem (Reed 2008, Curtin & Prellezo 2010). Therefore, stakeholders should understand the importance of maintaining seafloor integrity and how trawling compromises this integrity.

For this purpose a knowledge platform that gathers information of trawling impacts on benthic communities and introduces it in an easily and understandable way has been built (manuscript under review) (Fig. 2).

The platform represents benthic communities based on the Biological Traits Analysis (BTA) rather than on species composition as BTA has been proved to be a good methodology to assess trawling impacts in commercially exploited fishing grounds (de Juan et al. 2007, Bremner 2008). Moreover, biological traits are linked to community functionally, which will ultimately determine ecosystem services and processes (Bolam et al. 2014, Muntadas et al. 2015). Target species depend on these ecosystem services and processes, hence this might be a good approach to raise stakeholder's awareness about the importance of maintaining seafloor integrity.

In order to define acceptable trawling effort levels, it is necessary to determine how an effort change would potentially alter benthic communities. A tool that enables the comparison between current benthic community status and the potential status under different effort regimes would be useful to weigh the pros and cons of the suggested effort measures.

> *The knowledge platform contains a simulation tool that estimates potential functional changes on benthic communities due to effort modifications and facilitates its visualization in a user-friendly interface (Fig. 2).*

On the other hand, the assessment of the effect of all the suggested measures on benthic ecosystems, target species stock and the fisheries socio-economic context, will require the development of appropriate indicators to be used during the monitoring process (Leslie & Mcleod 2007, Samhouri et al. 2014). Hence, besides the traditional stock assessment reference points (e.g. Fmsy, Bmsy (Santoro et al. 2013)) and socio-economic indicators (e.g. productivity of labour, average wage (Bonzon 2000)), indicators to determine benthic ecosystem status are also needed. For example, Condition to Pressure Index provided by the simulation tool in the knowledge platform, or the Trawling Disturbance Index (TDI) proposed by de Juan & Demestre (2012), might be used to assess benthic ecosystem status.

Indicators covering all fisheries elements (target species, benthic ecosystems and stakeholders) should be used and their results integrated in order to assess the overall management plan successfulness.

Unwanted catches should also be controlled and reduced, as they are an additional source of ecosystem impact (Vassilopoulou et al. 2007). The first indicator on trawling unwanted catches should be the percentage of unwanted catches, which should be as low as possible. The recently implemented 40 mm squared cod-end (EC 1967/2006, but fully implemented on 2010) has proved to be more selective than the traditional diamond mesh, reducing the catch of immature individuals of target species (Bahamon et al. 2006, Sala et al. 2008). However, its effectiveness to reduce unwanted catches (mainly benthic invertebrates) has not been demonstrated. Part of unwanted catches is also commercialized, and the remaining unwanted catches fraction is returned to the sea, the discards, which should also be quantified. However, not all species included in the discarded fraction are equally resistant, and survival rates can greatly differ. Some experiments have shown low mortality of crustaceans and varying fish survival rates depending on the species (Bergmann & Moore 2001; Figuerola et al. 2001), but more data is needed to appropriately quantify the impact of discards on benthic communities.

The total percentage of unwanted catches, the discarded fraction and survival rate of this discarded fraction should be assessed, as it will also determine trawling impact on benthic ecosystems.

Moreover, they might be also used as an ecosystem impact indicator (Fig. 2).

3.2. Monitoring and enforcement

As mentioned in section 1, monitoring is a very important step to advance towards an EAF. This step is crucial to assess if the proposed regulatory measures have been capable to reach the objective to attain a GES of marine ecosystems, and of overall sustainability of target stocks. Firstly, in accordance with a participatory approach, a management committee composed by all fisheries' stakeholders should be constituted (Lleonart et al. 2014). This committee will discuss monitoring outcomes based on the previously mentioned target species' stocks, benthic ecosystem, economic and societal indicators, in order to reach the pre-established objectives.

The monitoring process should be divided in two phases. The first phase consisting on a thorough scientific follow up on the system performance under the established management measures, and a second phase of routine monitoring. The first phase is aimed at testing the management plan successfulness, and the second phase consists on a long-term monitoring of the system status (Fig. 2).

Phase one should last around 3-5 years (depending on the life span of the most important target species) and should consist on a comprehensive scientific assessment of the target stocks, benthic ecosystems and fisheries' socio-economic indicators evaluated in a integrative way. Catches and benthic ecosystems should be evaluated on a monthly or bimonthly basis using the indices mentioned in the previous section. Fishermen should cooperate with scientists by providing daily data on the fishing situation, and the catches and unwanted catches composition. The results of these surveys should be assessed in combination with the socio-economic indicators which should also be evaluated in a monthly or bimonthly basis.

Phase two should start once the results from phase one evidence that the suggested measures are appropriate to attain a GES. Already established surveys such as MEDITS project (Bertrand et al. 2002), onboard or port observers (BOE 3/2001), or the annual report of the European Scientific, Technical and Economic Committee for Fishing (STECF) may be useful at this monitoring stage. Landing records in auction reports would be also useful but, as mentioned at the end of section 2.1., landed species should be correctly classified. Furthermore, as well as these control measures for target species, a benthic ecosystem survey protocol should be established. This protocol might be based on the previously mentioned indicators and might be undertaken on an annual basis.

In both phases, monitoring outcomes should be evaluated by the management committee and, according to the adaptive management principles, management measures might be modified if management objectives are not attained (Westgate et al. 2013). Moreover, if during the second phase of monitoring a great change in the system is detected, a comprehensive scientific study (i.e. phase one of the monitoring program) might be carried out again.

Concurrently to these monitoring measures, enforcement should be also carried out. This is a very important component of a fisheries management plan, as it is necessary to ensure that all trawlers enforce the regulations to appropriately assess the effectiveness of the implemented measures (Lostado et al. 1999). If all measures are being strictly followed but management is not giving the expected outcomes, then a revision of the management measures should be done. However, if a non-compliance of the regulations is detected, they should be appropriately enforced before management outcome's evaluation.

The Vessel Monitoring System (VMS) was introduced in 1998 by the European Commission (EC 686/97) with the aim to control fishing activities and also for security purposes. VMS has proved useful as a control tool to follow fleet movements (Holmes et al. 2011, Needle & Catarino 2011). Alternatively, records from trawl fishing vessels of the Automatic Identification System (AIS) might also be used (Mazzarella et al. 2014).

The previously mentioned onboard or port observers (BOE 3/2001), may be also useful to check the enforcement of target species based measures, such as minimum landing sizes or catches quotas.

> *The management plan should also incorporate a system to check management measures compliance (e.g. VMS device, SSS, onboard or port observers) (Fig. 2).*

These enforcement measures are already implemented and, therefore, are also appropriated for the EAF plan recommended in this document.

However, other measures could be also implemented. For example, for the enforcement of gear technological improvements, economical incentive might be useful to encourage fishermen to modify fishing gears (Valdemarsen & Suuronen 2003). Moreover, although few, there are still some trawlers below 12 m length. Despite not being compulsory according to the European regulations, they should be also asked to incorporate a VMS device. Another possible measure to effectively control the trawling activity could be to carry out Side Scan Sonar (SSS) surveys to detect whether trawlers actually fish in permanent closed areas.

4. Final thoughts

This document suggests a possible implementation of an integral EAF on the trawling fleet from the Catalonia region. This protocol fits within the wider framework of the European Union regulations such as MSFD and CFP. This case study was chosen due to the available knowledge on fisheries and benthic ecosystems in the area. However, the suggested measures might also be applied to other Mediterranean trawl fisheries, as the trawl fleet dynamics are similar across the Mediterranean region (Lleonart et al. 1998). In this case, the General Fisheries Commission for the Mediterranean (GFCM) or the Mediterranean Advisory Council (MEDAC) should take the lead to coordinate the management committee and a network of fisheries research Mediterranean institutions might undertake the scientific monitoring. Moreover the management plan should also include the creation of an external committee to review the EAF products and enhance the process' transparency (Samhouri et al. 2014) . Moreover, a knowledge exchange network should be created in order to promote experience sharing to enhance the management plan successfulness. This knowledge exchange network should rely on the easy availability of monitoring survey and enforcement results, which, nowadays, is not always the case (Hinz et al. 2013). The creation of an open data internet site might facilitate data exchange. This network of information exchange should arise from the commitment of all Mediterranean countries sharing similar fisheries and should be based on public funding.

ACKNOWLEDGEMENTS

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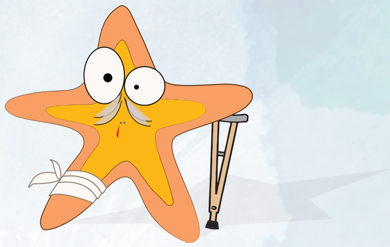
VII



Global discussion & Conclusions

GENERAL DISCUSSION

Marine ecosystems are complex adaptive systems with non-linear responses to stress that are not yet fully understood (Folke et al. 2004, Levin & Lubchenco 2008, Koch et al. 2009). This makes the Ecosystem Approach to Fisheries (EAF) challenging as scientists and managers have to deal with large uncertainties. Marine ecosystems, and particularly benthic ecosystems, perform many ecosystem functions and processes whose performance may change due to trawling activities (Frid 2011, Bolam et al. 2014). In order to understand this potential change, this thesis assessed benthic ecosystem response to trawling from a benthic functionality point of view, aiming to improve the current knowledge on benthic ecosystem dynamics under fishing disturbance. Moreover, in order to adequately implement an EAF, ecological and fisheries' knowledge needs to be integrated with societal needs, which is often difficult due to trade-offs between these needs and ecosystem conservation (Salomon et al. 2011). For this purpose, the concept "ecosystem services" has arisen to bring societal interests and ecological dynamics together (Boyd & Banzhaf 2007, Díaz et al. 2015). Ecosystem services are based on the interrelation of many ecosystem functions and processes and on how their combination results in a service for society. The studies presented in the thesis discuss functions of benthic ecosystems within the ecosystem services framework with the aim to scale up from the functionality knowledge to ecosystem services (Townsend et al. 2014, Snelgrove et al. 2014), and thus make an effort of linking ecosystems with societies.

The assessment of benthic response to trawling is undertaken within the broader context of an EAF. Using an integrated approach, the studies deal with all the elements of the fishing system, from fleet dynamics and fishing effort (Chapter 1) to the effects of fishing disturbance on benthic community functionality and related ecosystem services (Chapter 2), and the potential effects these changes on functionality might have on target species (Chapter 3) (Fig. D.1). The thesis also stresses the importance of stakeholders' involvement in fisheries' management by providing tools to enhance communication and information exchange among scientists and other stakeholders (Chapter 4). Finally, it suggests an EAF management plan for a case study, the Catalan trawling fisheries, based on the existing knowledge on the fisheries system of this case study (Chapter 5) (Fig. D.1).

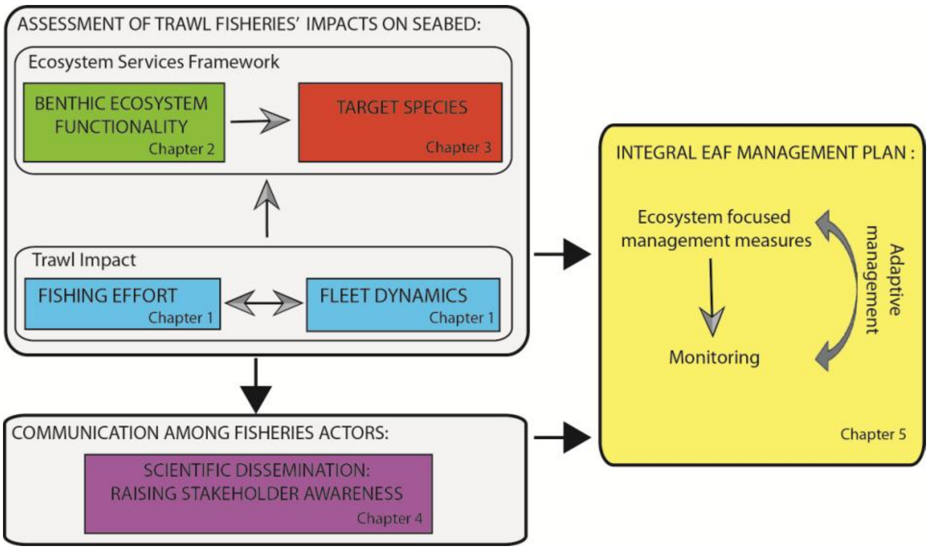


Figure D.1. Thesis contents and relationships among them.

1. ASSESSING TRAWLING IMPACTS ON THE SEABED WITHIN AN ECOSYSTEM SERVICES FRAMEWORK: FROM BENTHIC ECOSYSTEMS' FUNCTIONALITY TO EFFECTS ON TARGET SPECIES

1.1. Trawl impact

In trawling grounds, benthic ecosystem dynamics depend mainly on the level of fishing effort to what these communities are subjected (Thrush et al. 1998, Collie et al. 2005, Atkinson et al. 2011, de Juan & Demestre 2012). This disturbance causes benthic communities' functionality change which might provoke further changes in ecosystem functions performance and may also affect target species population (de Juan et al. 2007, Bolam et al. 2014, Johnson et al. 2015). The EAF paradigm aims at minimizing fisheries ecosystems impacts, hence knowledge on trawling effects on benthos is necessary to reduce to its impact.

Fishing effort disturbance overlaps with benthic communities' variability, which operate at scales smaller than fishing grounds. Therefore, in order to correctly assess fishing impacts on seabed ecosystems, fishing effort should be quantified at similar small scales (Collie et al. 2005, de Juan et al. 2013). Many studies have evaluated the effects of fishing on benthic communities along gradients of fishing intensity (Thrush et al. 1998, Craeymeersh et al. 2000, Blyth et al. 2004, Collie et al. 2005, Tillin et al. 2006), but only few (e.g. Craeymeersh et al. 2000, Blyth et al. 2004) have estimated the fishing effort at small scales (including variability within fishing grounds, e.g. 1x 1 miles grid).

This thesis tested the performance of three different methods, fisheries knowledge, Vessel Monitoring system (VMS) data and Side Scan Sonar (SSS) images, to estimate fishing to assess related ecosystem disturbance. Based on this thesis results, Box 3 summarizes the performance of each approach to assess several elements of the trawl fisheries' system (see figure I.1). Knowledge on these elements (i.e. fishing dynamics, benthic ecosystems and direct and indirect effects of trawling on target species) is essential to implement an EAF, as management measures should be designed based on current scientific knowledge of the system. Box 3 also highlights the importance to combine the three effort estimation methodologies in order to assess ecosystem effects and ultimately achieve a good EAF plan. The following paragraphs provide a summary of the strengths and weakness of the three abovementioned effort estimation approaches.

BOX 3: The performance of different effort estimation approaches to assess fisheries ecosystem disturbance

Effort data source	Fishing dynamics	Benthic ecosystems and habitat	Target species*	Target species**	EAF plan design
Fisheries	☆☆	☆☆	☆☆☆	☆	
VMS	☆☆☆	☆	☆☆	☆	☆☆☆
SSS	–	☆☆☆	–	☆☆☆	

Legend:
– no use
☆ limited use
☆☆ good performance
☆☆☆ very good performance

* direct impacts - catches
** indirect impacts - habitat changes

Fisheries knowledge is useful to approximately establish fishing ground boundaries, but it is too coarse to be directly related with benthic communities' heterogeneity (Box 3). VMS signals have also been used for benthic impact assessment (Piet & Hintzen 2012, Gerritsen et al. 2013) using high resolution grids, which should be able to detect within-fishing ground effort heterogeneity. However, these studies used VMS signals with 2 h interval, which might lead to inaccurate effort estimations (normally effort underestimation at epifauna level) that would ultimately affect the link between fishing effort and benthic ecosystem status (Deng et al. 2005, Skaar et al. 2011, Lambert et al. 2012) (Box 3). In accordance with these studies, our work demonstrates that, if VMS are to be used for benthic ecosystem assessment, more frequent signals (~ 30 min) are needed. However, even using high frequency signals, VMS data is not a precise approach to estimate fishing effort at benthic ecosystem spatial scale. Due to the original purpose of the VMS devices, to control fleet movements, VMS data is oriented at monitoring larger scale processes, (e.g. fleet dynamics) and the objective is not to record data at small-scale. Among the three approaches, SSS is the one that gives more detailed information at a small scale (Box 3). As SSS methodology provides detailed sea bed images, it allows observing trawl tracks on the seabed, the actual trawl impact on marine bottoms.

Effort estimations from fisheries data and SSS were used on the assessment of ecosystem responses to trawling based on changes in benthic functional components. Both estimations were well correlated with benthic communities' functions (production, nutrient cycling, benthopelagic coupling, carbon sequestration and habitat structure), but SSS, as expected, showed slightly better results (12 out of 20 GAM models significantly influenced by trawling using SSS vs. 8 out of 20 using fishery data). Thus, whenever possible, SSS effort estimations should be used in benthic impact assessment, as they provide a more accurate link between trawling activities and benthic communities' responses. Nevertheless, when using the SSS approach it should be bored in mind that trawl tracks' permanence on the seabed depends on the sediment type (Smith et al. 2007, Palanques et al. 2014). Therefore, SSS might not be an appropriate effort estimation methodology in high dynamic sediments such as sand.

On the other hand, it is not always possible to get the data from all three sources (fisheries data, VMS and SSS) and, normally, the source of information most straightforward to obtain is fisheries data (McCluskey & Lewison 2008). This limitation has been taken into account when building the simulation of the knowledge platform in Chapter 4, as the model accepts different sources of data (SSS and fisheries data effort estimations).

Although SSS estimations provide an accurate picture of trawl activity at the benthic community scale, they do not provide information about spatio-temporal fleet dynamics (Box 3), which is also an important aspect to take into account when managing fisheries (Stelzenmüller et al. 2008). Fleet dynamics offer information on when and where fishing activity concentrates, which contributes in the design of spatio-temporal closures and/or effort reduction strategies to reduce fishing effort in the most vulnerable areas (Branch et al. 2006, Jennings & Lee 2012). VMS signals, being an independent data source, have proved to be very useful for this purpose (Box 3) (Bertrand et al. 2008, Jennings & Lee 2012). As an example, figure D.2

shows the movements of the coastal trawler from Chapter 1-Section 1- Fig. 4a over the year. In this chapter, the general fishing strategy of the vessel was described, but its seasonal activity was not assessed. Fishermen change fishing grounds along the year following target species life cycle (Colloca et al. 2009) and understanding these dynamics is important for fisheries' management. Thus, to detect this seasonal activity, yearly data has to be break down to monthly data. The representation of VMS data in Figure D.2 shows how this vessel concentrates its activity around the -50 depth line from July to October. The two species increasing its landings especially from June to October are *Mullus barbatus* (red mullet) and *Pagellus acarne* (axillary seabream), but, according to catches´ income, fishermen must be targeting *M. barbatus*, as, of the two species, this is the one with highest income (see graphs embedded on Fig. D.2). This assumption is confirmed by the fact that summer-early autumn (July to September-October) coincides with the recruitment period of *M. barbatus*, and this species is known to recruit around -50 m (Suau & Vives 1957). *P.acarne* would constitute a by-catch due to the fact that this species share habitat with *M. barbatus* (Demestre et al. 2000). Therefore, coastal trawlers change fishing grounds during this period of the year to target *M. barbatus* young recruits, a typical characteristics of this species' fisheries (Demestre et al. 1997, Martin et al. 1999) also observed in Chapter 3.

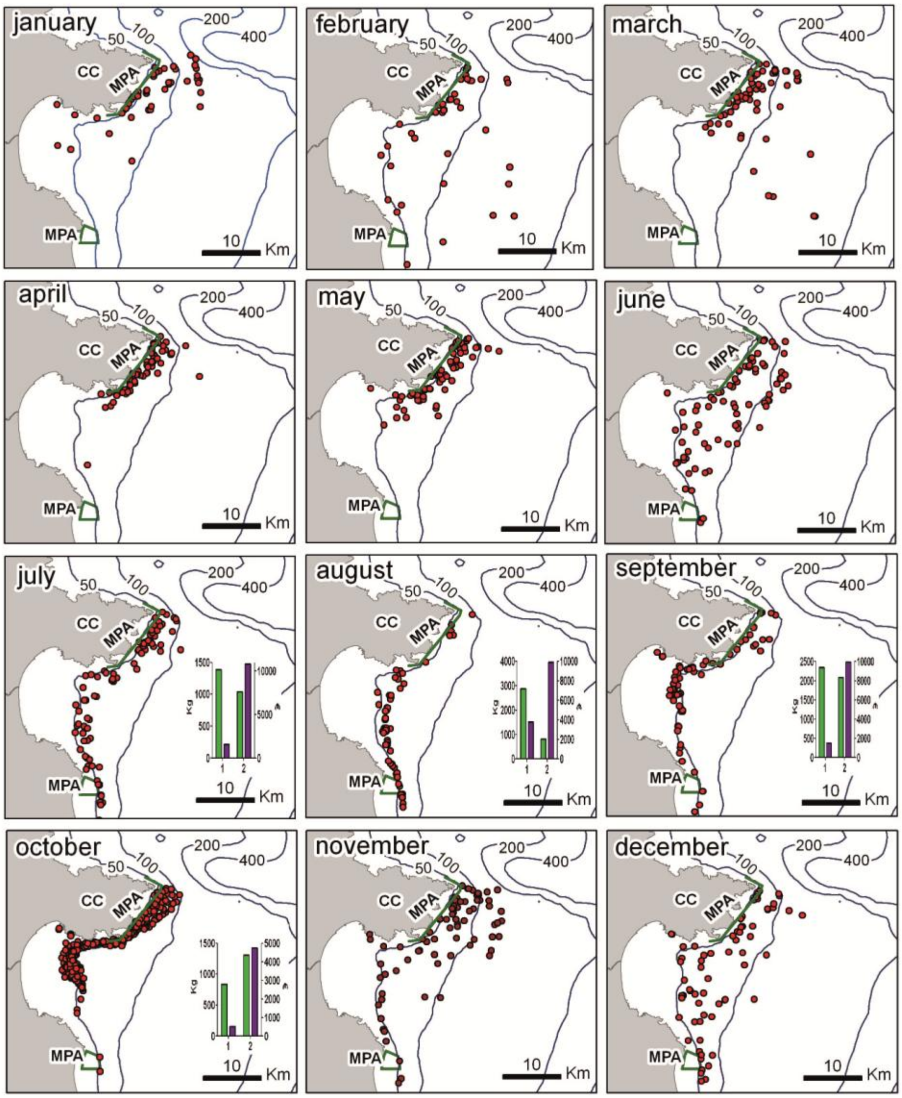


Figure D.2. VMS signals showing vessel dynamics over the year 2009 of the costal trawler from the Chapter 1-Section 1 Fig.4a. CC: Cap de Creus. Graphs: 1- *Pagellus acarne,* 2- *Mullus barbatus*. Green bars correspond to landings (Kg) and purple bars to income (€)

This example highlights the importance of the area around 50 m depth as an Essential Fish Habitat (EFH) for red mullet and other species such as *P. acarne*. VMS data has been used to estimate the distribution of target species (Campanis et al. 2008, Bertrand et al. 2008) and, combined with landings data, to explore catches' spatial distribution and target species' CPUEs (Fonseca et al. 2008, Gerritsen & Lordan 2010). The presented example shows that it is possible to go a step further and combine landings and VMS data with existing knowledge on target species life-cycle, which allows detecting EFH that should be considered for protection. Moreover, SSS data was also available for the area in front of Cap de Creus. This SSS data showed that the fishing grounds in front of Cap de Creus were subjected to relatively low effort compared to other studied fishing grounds in this thesis. Those areas subjected to less fishing activity hold more structured benthic communities, hence, they should be protected from fishing disturbance. Furthermore, as those areas are less visited by trawlers, fishermen might be less reluctant to potential effort reduction measures. In this context, SSS data allows detecting differently impacted areas within an EFH, which might help in establishing protected areas limits. Therefore, all levels of information (fishery data, SSS, VMS and species life cycles) are needed to gather information for an EAF management plan design (Box 3).

2.2. Ecosystem Services framework

In the EAF context, ecosystem dynamics knowledge has to be integrated on the society dimension, hence this thesis assessed benthic ecosystem response to trawling within the ecosystem services framework. "Ecosystem services" is a complex concept that integrates different levels of system understanding: species' biological/functional traits and the consequent function performance, the service provision related to each function, and finally the "ecosystem use", i.e. benefits humans obtain from ecosystems (MEA 2005). The assessment of ecosystem functioning by Biological Traits Analysis (BTA), has already been addressed by several authors (Tillin et al. 2006, de Juan et al. 2007, Bremner 2008). Considering this background knowledge, this thesis explored the potential of BTA to go beyond ecosystem functionality (i.e. ecosystem traits distribution) and used it as a proxy to assess benthic ecosystem function performance, and hence related ecosystem services, in trawled grounds. Although this methodology does not allow the estimation of function rates, it is useful to evaluate ecosystem function response to trawling. The ecosystem function and services assessment in this thesis is done from an ecological perspective, based on the ecosystem components and highlighting its importance to properly understand the final "ecosystem use". Chapter 2 focused mainly on key supporting ecosystem services delivered by soft-bottom communities (e.g. nutrient cycling, sediment stability or habitat provision) and chapter 3 discussed how benthic functionality change due to trawling may affect target species populations and hence trawling catches, i.e. the main direct ecosystem service humans perceive as a benefit from fishing grounds (Fig. D.3).

The conservation of benthic habitats is very important to maintain viable target species' stocks (Caddy 2013). Target species' populations, besides direct fishing impact, may be indirectly affected by fishing activities due to trawling driven changes on seabed ecosystems. Demersal target species live in close relationship with benthic ecosystems and depend on them to fulfil their live cycle (i.e. feeding, spawning, seeking refuge, etc.) (Auster & Langton 1998, Kaiser et al. 1999). Several studie s have focused on how trawling may change

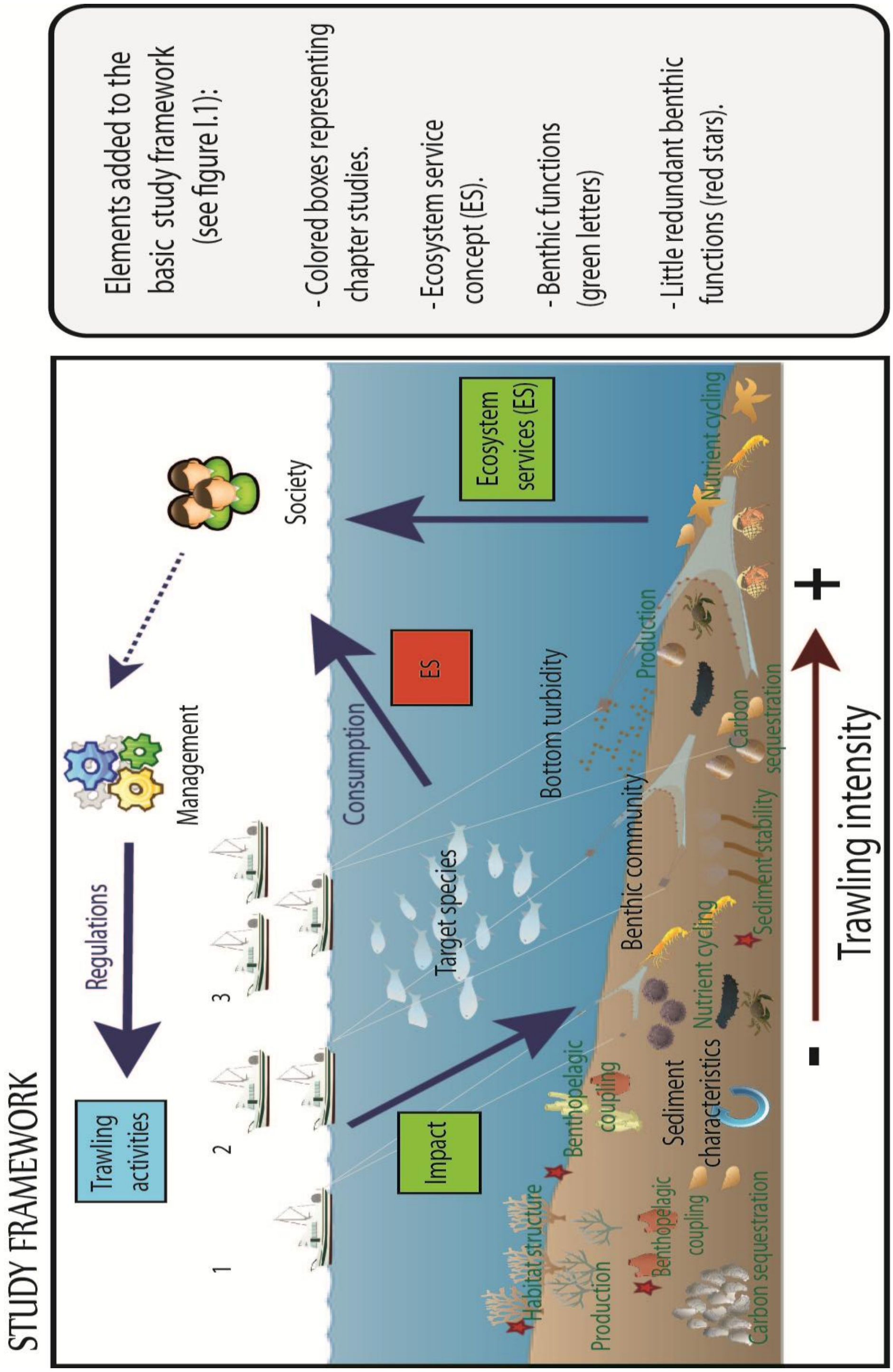

STUDY FRAMEWORK
Trawling activities
Regulations
Management
Society
1
2
3
Impact
Consumption
ES
Ecosystem services (ES)
Target species
Bottom turbidity
Benthic community
Habitat structure
Production
Benthopelagic coupling
Carbon sequestration
Sediment characteristics
Nutrient cycling
Sediment stability
Carbon sequestration
Production
Nutrient cycling
-
+
Trawling intensity
Elements added to the basic study framework (see figure I.1):
- Colored boxes representing chapter studies.
- Ecosystem service concept (ES).
- Benthic functions (green letters)
- Little redundant benthic functions (red stars).

Figure D.3. Diagram representing ecological and socioeconomic elements in a fishery system (in this case trawling fisheries are represented). The numbers 1, 2 and 3 represent different qualitative levels of fishing effort, from low to high according to the number of vessels. Full arrows define actual interactions whereas the dashed arrow points out potential interactions. Black font represents the system elements and blue font the relationship between these elements. Colored rectangles represent the elements studied in this thesis: blue-first chapter; green-second chapter; red-third chapter; purple-fourth chapter; yellow-fifth chapter. Benthic community structure under different trawling intensity is based on de Juan et al. (2009) and de Juan & Demestre (2012). The main ecosystem functions provided by benthic community are shown in green. Red stars mark functions showing low redundancy (according to Chapter 2-Section 2 results).

commercial species feeding habits (Badalamenti et al. 2008, Hiddink et al. 2011, Smith et al. 2013, Johnson et al. 2015), however, to our knowledge, no studies exist that consider the overall effect of trawling on a target species' life cycle. Chapter 3 in this thesis represents a first insight on this topic, highlighting an overall negative indirect impact of trawling on red mullet's population. However, chapter's 3 study only discussed the potential indirect effects of trawling on red mullet population in a heavily trawled area. To define more accurately these indirect effects, a comparative study of at least two different target species stocks of the same species from areas having markedly different trawling intensity, and hence presumably holding differently structured benthic communities, should be done.

The conservation of benthic habitats for target species means also maintaining ecosystem functionality. In order to assess ecosystem function performance in trawling grounds, this thesis suggests the use of the "Ecosystem Service Provider" (ESP) concept, i.e. an ecological unit performing certain functional process (e.g. a set of benthic species that are large filter feeders contributing to bentho-pelagic coupling). ESPs were found to respond to trawling, proving their usefulness in the assessment of trawling impact on benthic ecosystems. However, ESPs were also highly influenced by habitat characteristics, which hindered the assessment of trawling impacts on the related functions and services (see the main soft-bottoms ecosystem functions in Fig. D.3). Not all the habitats included in the study were equally represented: homogenous muddy sediments were well represented, whereas only few replicas for sandy, gravelly and maërl habitats were included. Hence, an analysis including additional samples of these habitats that tend to be more heterogeneous than muddy habitats might help in telling apart variability conditioned by the habitat type from responses to trawling impacts. Furthermore, although the study included a range of sites with variable effort levels, only one site was a no-take MPA and, hence, only one habitat having null (or almost null, as SSS images detected some trawl tracks within the MPA) effort was represented. As the threshold of change on trawled communities might be low (Bremner et al. 2003, de Juan et al. 2009), future studies should include more sites holding low or null effort than sites holding high fishing effort, therefore, including in the study sites over a fine gradient of low effort intensity.

To overcome the influence of habitat type on benthic responses to trawling, some studies chose to include only sites with similar sediment characteristics (Jennings et al. 2001, Collie et al. 2005, Tillin et al. 2006, Mangano et al. 2014) or to remove the effect of sediment characteristics using partial canonical correspondence analysis and partial redundancy analysis (Craeymeersh et al. 2000). However, in agreement with our findings, many studies have shown that different habitats respond differently to trawling (Kaiser & Spencer 1996, Collie et al. 2000, Kaiser et al. 2006, Queirós et al. 2006). Hence, it is important to include a range of habitats in trawling impact studies to appropiately assess variability of responses. Trawling grounds are subjected to a natural habitat variability that should be appropiately adressed in the study of benthic ecosystem response to trawling. This understanding might be achieved by performing a metanalysis of studies carried out in the Mediterranean on fishing grounds having different sediment characteristics, paying special attetion to the underestudied more heterogeneous habitats (e.g. studies in maërl habitats such as Barberá et al. 2012).

Despite the confounding effects of sediment characteristics in the assessment of benthic responses to trawling, this thesis studies found the functions carbon sequestration and benthopelagic coupling to be vulnerable to trawling in our study sites (see all the considered functions in Fig. D.3). Moreover, combining the results of the two studies in chapter 2, a clear pattern emerged: the functions performed by ESPs exhibiting vulnerable traits to trawling (i.e. functions: sediment stability, habitat structure and benthopelagic coupling; traits: tube dwelling, attached/sessile and filter feeding) had lower redundancy at higher trawling intensity levels (Fig. D.3). For example, the trait “filter feeding”, which is linked to vulnerable ESPs performing the benthopelagic coupling function, showed low redundancy in the study sites with high effort levels. The lower redundancy shown by these traits implies that trawling might compromise them, altering benthic system functionality. For example, the potential reduction of sediment stability and benthopelagic coupling may unbalance the storing and cycling of nutrient service, releasing too many nutrients and/or pollutants (Snelgrove 1997, Chauvaud et al. 2000, Snelgrove et al. 2014). This phenomenon may be also enhanced by the sediment disruption caused by trawling gears (i.e. cumulative impacts) (Palanques et al. 2001).

In this context, redundancy of functional traits is an ecosystem property that may prevent the loss of an ecosystem function. A function having high redundancy may be more resilient (i.e. capable of withstand disturbance maintaining ecosystem functionality), as species performing that function may have different responses to trawling and the loss of one species might be offset by an increase of another redundant species (Folke et al. 2004, Levin & Lubchenco 2008). Therefore, EAF strategies aimed at maintaining ecosystem resilience (Levin & Möllmann 2015) should take functional redundancy into account. In this context, special attention must be paid to areas holding low redundant ecosystem functions, and human activity within these vulnerable areas should be carefully managed (Douvere 2008, Ottersen et al. 2011). In conclusion, a vulnerable area would exhibit high abundance of vulnerable ESPs when the traits held by these ESPs are vulnerable to trawling and/or are low redundant traits. For example, the ESP “Large filter feeder” is considered a vulnerable ESP as it is composed by the traits “large” and “filter feeding” that are vulnerable to trawling; moreover “filter feeding” was a low redundant trait, increasing the overall vulnerability of this ESP. Preservation of the areas with high densities of vulnerable ESPs would allow conserving indirect services (Rees et al. 2012), which might ultimately benefit target species and hence support direct ecosystem services as fishing catches.

2. ADVANCING TOWARDS AN EAF MANAGEMENT PLAN: THE NEED OF COMMUNICATION AMONG FISHERIES' ACTORS

Valuing ecosystem functions and their related ecosystem services helps in understanding ecosystem function response to trawling, but, in order to incorporate this knowledge on ecosystem management, the ecosystem services approach must be integrated in management decisions (Daily et al. 2009). This step needs stakeholder's understanding of the importance of ecosystem functions and services that the system to be managed provides. Therefore, scientific knowledge must be communicated in an understandable way to fisheries' actors. Stakeholders must participate in fisheries management decisions (i.e. participatory approach) and thus appropriate deliberation support tools have to be used (Reed 2008).

In this context, this thesis introduced two tools to promote a participatory approach in fisheries management: a DPSIR (Drivers – Pressures – State Change – Impact – Response) conceptual framework, to inform of all the factors involved in trawl fisheries and the relationships among them, and a knowledge platform specifically informing on the most ignored relationship: the state change produced by trawl gears on benthic communities.

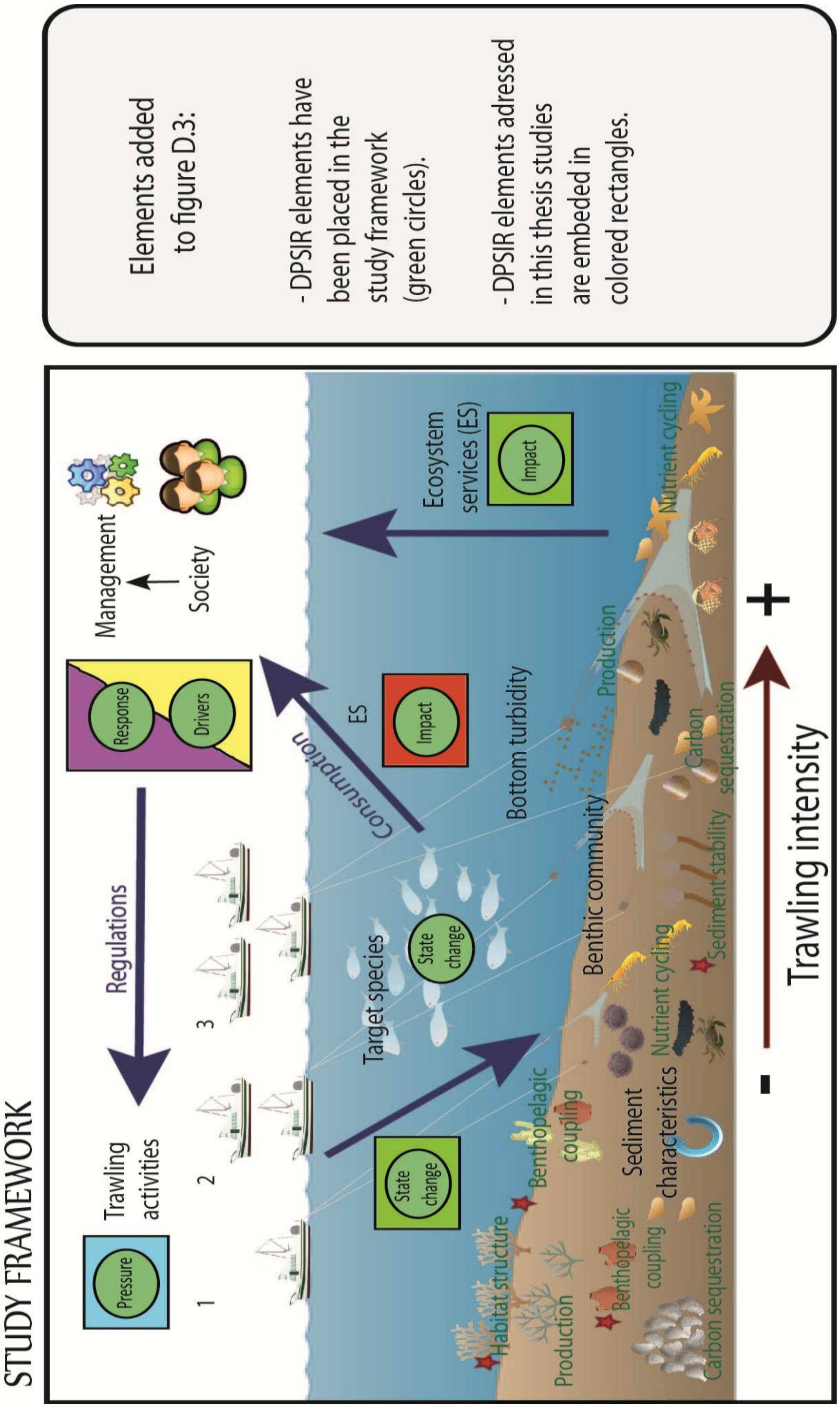
The DPSIR framework allowed representing the trawl fisheries' system in a simplified version, which is thought to make communication between policy makers and scientists easier (Mangi et al. 2007, Atkins et al. 2011, Ojeda-Martínez et al. 2012). Figure D.4 shows where the DPSIR elements are positioned in the trawl fisheries' system, denoting which of the studies presented in this thesis addresses each element.

Drivers:

> Society was considered an important *Driver* that demands actions that could motivate the EAF implementation (Fig. D.4). Society behaviour towards fisheries' management, e.g. requesting information about how a fish has been harvested in order to decide whether they buy it or not (i.e., responsible consumption), may imply management changes towards an EAF (see for example https://www.msc.org/track-a-fishery). Moreover, society votes politics that will implement the management plans, hence, the important role of society as drivers of change.

Pressure:

> Describing this component in detail constitute the basis or an appropriate assessment of the benthic communities' *State Change*. Chapter 1 addressed this issue exploring different methods to estimate fishing effort. Moreover, it provided useful information on fleet dynamics (e.g. where and when the fleet concentrates), which also constitutes a *Pressure* component.



STUDY FRAMEWORK
Pressure
Trawling activities
1
2
3
Regulations
Response
Drivers
Management
Society
State change
Target species
State change
Consumption
ES
Impact
Ecosystem services (ES)
Impact
Bottom turbidity
Benthic community
Habitat structure
Production
Benthopelagic coupling
Benthopelagic coupling
Sediment characteristics
Carbon sequestration
Nutrient cycling
Sediment stability
Carbon sequestration
Production
Nutrient cycling
Trawling intensity
-
+
Elements added to figure D.3:
- DPSIR elements have been placed in the study framework (green circles).
- DPSIR elements adressed in this thesis studies are embeded in colored rectangles.

Figure D.4. Diagram representing the DPSIR elements (green circles) in this thesis study framework. DPSIR elements embedded in colored rectangles represent the elements studied in this thesis: blue-first chapter; green-second chapter and red-third chapter. The numbers 1, 2 and 3 represent different qualitative levels of fishing effort, from low to high according to the number of vessels. Black font represents the system elements and blue font the relationship between these elements. Benthic community structure under different trawling intensity is based on de Juan et al. (2009) and de Juan & Demestre (2012). The main ecosystem functions provided by benthic community are shown in green. Red stars mark functions showing low redundancy (according to Chapter 2-Section 2 results).

State change on benthic communities and Impact on ecosystem services:

The *State Change* of benthic communities due to trawling disturbance may lead to possible *Impacts* on society through the impairment of ecosystem services. The study of benthic community responses to trawling allows describing how benthic functionality changes when facing fishing pressure (Bremner et al. 2003, de Juan et al. 2007) and the effect these changes might have on ecosystem functions provision, determining its resilience to trawling disturbance (Frid & Caswell 2014, Lambert et al. 2014). This knowledge may allow defining thresholds of change beyond which the benthic community may not be able to deliver certain ecosystem services (Folke et al. 2004, Thrush et al. 2009, Levin & Möllmann 2015). But, in order to maintain the integrity of ecosystems, management has to act before this threshold is surpassed. Hence, reference levels or benthic status indices have to be designed to determine acceptable levels of pressure (de Juan & Demestre 2012). In this context, and as previously mentioned, chapter 2 highlighted the vulnerability of some key benthic ecosystem functions in fishing grounds: habitat provision, benthopelagic coupling and sediment stability. Moreover, it suggested two measures to assess benthic community redundancy, which, after further testing, might be potentially used as indicators to establish acceptable levels of fishing pressure. Indicators must meet a series of characteristics: respond to stress in a predictable manner, be sensitive to small variations in stress, be simple and cost-effective, predict changes that can be reverted with management actions, be adjustable to different scenarios, be applicable to extensive geographical areas and must also be easily understood by stakeholders (Salas et al. 2006, Reza & Abdullah 2011). Redundancy measures presented in chapter 2 have proved to respond to stress (i.e. trawling disturbance) in a wide geographical range and, as rare species were found not to significantly influence the measures, they could be calculated using only the most representative species, keeping the index cost-effective. However, the other characteristics have not been tested and they should be addressed before proposing these redundancy measures as practical management indicators. Moreover, as the two redundancy measures are complementary, we need to consider how information from both measures should be combined in a global redundancy indicator.

State change on benthic communities and Impact on the ecosystem service "food provision":

State Change of benthic communities may indirectly affect target species populations due to EFH alteration, causing an *Impact* on the ecosystem service “food provision”. Chapter 3 stressed the relevance of soft-bottom ecosystems as EFH for red mullet and hinted potential indirect impacts of trawling on this target species population due trawling disturbance of benthic ecosystems. Therefore, Chapters 2 and 3 highlighted the relevant role of benthic ecosystems for the trawl fisheries system and, hence, the importance of considering benthic ecosystem status in EAF management plans.

Response:

In order to raise managers’ awareness of the importance of considering benthic communities in EAF management plans, the role of these communities in the fisheries system has to be clearly communicated. To this aim, Chapter 4 compiled the information of trawl driven changes on benthic communities from this thesis study areas on a knowledge platform. The platform introduces the scientific knowledge of trawling effects on benthic ecosystem functionality in an attractive visual way, so as to share this knowledge with all the actors involved in the fisheries’ system and enhance communication and coordination among them. Moreover, the platform includes a simulation block, which enables the user to test the effects of an effort level change on benthic community structure. The importance of this management *Response* is also considered in Chapter 5, where an EAF plan focused on maintaining the seafloor integrity, while supporting sustainable fisheries, is proposed. Management measures suggested in chapter 5 were designed bearing in mind the integral trawl fisheries' system depicted in figure D.5., by discussing potential trade-offs between maximizing catches and maintaining benthic ecosystems in a good status.

The knowledge platform in chapter 4 is suggested to be used in the chapter's 5 EAF management plan implementation. However, this knowledge platform is still in its first development steps. In the future the platform should communicate with R open-access software in order to automatically recalculate the simulation regression laws when a new study site was added. The recalculation would be done using the additional data included with this new site, together with the existing data, which would increase the simulation robustness. Moreover, it should also include data of trawling effects on infauna in order to have a more complete picture of the benthic ecosystem (Jørgensen et al. 2011) and to include the benthic component that corresponds to the prey of target species (Smith et al. 2013, Johnson et al. 2015). Finally, considering Chapter 3 results as a basis, the simulation should scale up to include trawling effects on target species in order to provide stakeholders with a more complete evaluation tool for fisheries management.

Despite the importance of benthic communities and habitat structure for the life-cycle of target species and for the provision of the ecosystem services societies need, ecosystem protection measures are still the main weakness of EAF management plans (Gårdmark et al.

2011, Caddy 2013, Snelgrove et al. 2014). Hence, tools like the knowledge platform, and concretely the simulation bloc within the platform, are needed in order that stakeholders' understand the role of seabed ecosystems in fisheries performance. This understanding would encourage the inclusion of benthic impact assessment in trawl management plans. Simulations to test different management scenarios and to assess how management actions are likely to affect the chosen indicators, are very useful when designing an EAF management plan (Tallis et al. 2010). As management measures in the Mediterranean are mainly based on effort regulations, the simulation in the knowledge platform might be used for this purpose.

Actually, as previously mentioned, Chapter 5 includes the knowledge platform in the set of tools recommended to implement a Mediterranean EAF plan. This plan stresses the important role of benthic ecosystems to sustain trawling fisheries and suggests how benthic ecosystem status might be taken into account in trawl fisheries management. As most of the suggested measures imply reduction of fishing effort and catches, one of the main difficulties to implement this plan would be to make fishermen aware that these measures are necessary to maintain long-term viable target species stocks. Moreover, long-term monitoring plans are costly and the inclusion of the benthic assessment would increase this cost even more. Hence, political will to invest in such a plan would be another key point for its successfulness. The suggested management plan might be implemented in different phases along the Catalan coast (e.g. starting in the southern ports and adding sequentially the northern ports) and, if it results in a successful implementation, it might be extended to the neighboring trawl fleets which present similar dynamics.

3. SCIENTIFIC DISSEMINATION

a) Fisheries' actors focused dissemination

This thesis has been mainly focused on the ecological elements of the fisheries' system, and it has not introduced the suggested communication tools (the DPSIR conceptual framework and the knowledge platform) to the non-scientific community. Hence, the following step should consist on seeking fisheries actors' appraisal of these tools. This might be done through stakeholders workshops, where they would be introduced to the tools and would be asked to complete an evaluation questionnaire (Janssen et al. 2014). For instance, stakeholders might be asked to weight the importance of each DPSIR element in the whole system according to their opinion, which would provide information of their perception of the fisheries' system. In such workshops, it would be also possible to use the knowledge platform as a deliberation support tool in the context of a Multi-Criteria Decision Analysis (MCDA) applied to trawling fisheries (Kiker et al. 2005). MCDA is used for evaluating different decision alternatives for systems in which multiple stakeholders with different points of view and interests are involved. They are very flexible in handling complex information and allow to assess the systems' trade-offs to draw feasible management strategies (Brown et al. 2001, Kiker et al. 2005, Janssen et al. 2014). Therefore, it would be interesting to apply a MCDA methodology to the trawling system analyses throughout this thesis. For example, stakeholders might be introduced to different potential management scenarios including some or all the management measures suggested in Chapter 5. Each scenario should also take into account potential socio-economic consequences. The simulation in the knowledge platform might be used to show benthic ecosystem status under each scenario. Then, stakeholders might be asked to rank these management scenarios and to suggest another possible combination of measures if they think a better choice exists. Finally, they might be asked to form groups of people with different interests and to rank again the scenarios reaching a group agreement. The final aim would be to reach a consensus on the set of management measures that should be implemented optimizing ecological and economic interests.

b) General public focused dissemination

It would be very useful to carry out a questionnaire addressed to the general public, in order to know their perception about ecosystem services delivered by soft-bottom benthic communities (Himes 2007, Hutchison et al. 2013). Contrary to the above-mentioned MCDA approach, this questionnaire should address a broader topic than the fisheries' management-oriented proposed before. Questions should address benthic ecosystem ecology values, asking people about their perceptions (i.e. cultural values, benthic ecosystem role in fisheries and wider marine system) (de Juan et al. 2015). This questionnaire would help to evaluate the link ecosystem-society, which would ultimately determine society role as Drivers. Ultimately, this approach would point out if scientists are efficiently communicating the importance of benthic communities in marine ecosystems, or more effort should be put on disseminating activities.

Scientific dissemination is essential to raise public awareness of the importance to keep ecosystems in a healthy status and to enhance the role of society as drivers (Sheavly & Register 2007, Demestre & Masó 2012). The increase of knowledge resulting from dissemination activities helps in building a society ownership feeling towards the environment and a willingness to preserve it (Fanini et al. 2007). Therefore, dissemination activities with the aim to introduce benthic ecosystem ecological values and potential threats to the general public (beyond fisheries stakeholders) should be also carried out. For example, a workshop in which the public is introduced differently structured benthic communities under different conservation status might be designed. Concurrently, the public might be also introduced the same number of trawl fleets corresponding to different effort levels as well as different benthic ecosystem properties (e.g. the ecosystem capacity of recycling nutrients or providing shelter for organisms). Then, the public might be asked to link each benthic community status with its related fishing effort level and ecosystem properties. This activity, explaining the consequences of the different ecosystem conservation status in real world scenarios, might help to improve public knowledge of benthic ecosystem properties and the importance to maintain ecosystems in good conservation status.

The concern for ecosystem impacts of fisheries is already included in the new European Common Fishery Policy (EU 1380/2013), meaning that there is a political awareness of the importance of the ecosystem status. The society is also pushing for the implementation of these policies, asking for information about how seafood has been fished and choosing to buy sustainable fished seafood (https://www.msc.org/). However, discussions around how to implement fisheries policies are still mainly focused on target species (e.g. http://www.ecologistasenaccion.es/article30480.html) as they are directly linked to economics. Despite the relationship between target species and their ecological environment, normally economic interests prevail and it is difficult to overcome the existing trade-off between socio-economic and conservation interests (Kempf 2010, Salomon et al. 2011). Moreover, scientific knowledge is sometimes limited, which also hinders the EAF implementation (Coll et al. 2013, Sartor et al. 2014). Aiming at filling this knowledge gap, this thesis provides a methodology to be used in ecosystem function assessment (ESPs and redundancy measures) and a simulation tool to show potential changes in benthic communities' structure due to changes on fishing effort. Moreover it suggests how an EAF management plan might be implemented in a well-known fisheries context.

GENERAL CONCLUSIONS

● Information from fishermen interviews and auction reports allows the identification of fishing grounds and helps determining the spatial-temporal distribution of the resources. It is also useful to detect areas holding different fishing effort levels.

● Vessel Monitoring System is a good independent data source to describe fleet dynamics at a broad scale. Although it is too coarse to define fishing effort at a benthic ecosystem level, it allows delimiting fishing grounds in time and space.

● Side Scan Sonar surveys provide an actual picture of trawl impact on the seabed. Therefore, Side Scan Sonar information gives very accurate effort estimation at the benthic level.

● Based on the previous conclusions, the use of a combination of the three approaches is suggested to inform on an Ecosystem Approach to Fisheries as they provide a characterization of the vessel's fishing strategies, spatio-temporal distribution of targeted species and the best small scale effort estimation.

● Ecosystem Service Providers, defined as groups of epifaunal organisms based on shared biological traits and their link to ecosystem functions, are a good approach to assess the response of benthic community functionality to trawling.

● Trawling significantly affects Ecosystem Service Providers, although the response is difficult to define due to the influence of habitat variability. Sediment type conditions the presence of certain Ecosystem Service Providers and trawling will act upon this natural variability. Therefore, the relative abundance of each Ecosystem Service Providers varies depending on trawling effort and sediment characteristics.

● The study of a range of sites subjected to different effort levels allows the detection of general patterns in benthic community response to trawling. However, in order to disentangle fishing and sediment effects, more samples from heterogeneous sediments than from homogeneous sediments should be included in the study.

● The metric chosen to estimate Ecosystem Service Providers, epifauna biomass or abundance, influences the estimation of the Ecosistem Service Providers contribution to the ecosystem function, hence, it is important to assess each function separately and determinate which is the best approach in each case.

● Infaunal and epifaunal benthic functional redundancy should be assessed by focusing on both trait abundance (Common traits) and trait richness (Widespread traits), as the two measures give complementary information. The use of only one of the measures might lead to a misleading idea of the overall functional redundancy and, ultimately, benthic community resilience.

● Infaunal and epifaunal rare species in chronically trawled fishing grounds do not significantly increase the benthic ecosystem trait richness. Hence, redundancy measures may be estimated focusing only on the most abundant species, resulting on a cost-effective indicator.

● Ecosystem Service Providers exhibiting vulnerable traits to trawling and performing functions such as sediment stability, habitat structure and benthopelagic coupling, presented also traits having low redundancy at high fishing intensity. Therefore, these functions might be compromised at high levels of fishing intensity in our study area.

● Changes in the functional components of benthic communities that might have negative effects on Essential Fish Habitat combined with the exploitation of red mullet young recruits might result in an overall negative effect on the red mullets' stock.

● We consider that society may potentially play an important role in demanding management Responses and advance in the implementation of an EAF, hence society was placed as a Driver in the trawl fisheries DPSIR conceptual framework.

● The knowledge platform is aimed at sharing knowledge on trawling impacts on benthic communities in a participatory approach context. This platform shows in an understandable and attractive visual way the trawling impact on the structure of benthic communities from our study sites.

● The simulation bloc included in the knowledge platform shows potential benthic functionality responses to an effort change. This simulation may be used as a support deliberation tool to evaluate the consequences on benthic ecosystems of potential management scenarios.

● Based on the observed fisheries dynamics and effects of trawling on benthic communities and targed species, an integrated EAF management plan in a case study area is presented aiming at maintaining sea floor integrity while supporting sustainable fisheries.

● The suggested plan pays special attention on recommending benthic ecosystem focused measures as it is normally the most largely ignored element on fisheries management. Therefore, benthic status indicators should be used in monitoring as well as the traditional stock assessment reference points.

VIII

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IX

Resposta de les comunitats bentòniques a diferents nivells d'impacte de pesca d'arrossegament; generalització per al desenvolupament d'un model mediterrani.

RESUM

INTRODUCCIÓ GENERAL

1. EL MARC DE GESTIÓ DE LES PESQUERIES: ENFOC ECOSISTÈMIC DE LES PESQUERIES

La pesca és una de les pressions humanes més esteses en el medi marí, i pot ser considerada com una de les activitats més perjudicials per als ecosistemes oceànics (Jackson et al. 2001) Tot i que la preocupació per l'impacte de la pesca en els ecosistemes marins ve de lluny (de Groot 1984), en les últimes dècades s'ha donat un creixement renovat de l'interès per abordar aquest problema des d'un punt de vista ecològic (Garcia & Cochrane 2005). D'aquesta manera el 1998 va aparèixer el terme "Gestió de les pesqueries basada en els ecosistemes (Ecosystem-based Fisheries Management-EBFM) descrit pel Consell d'Investigació Nacional dels Estat Units, amb la idea de tenir en compte els components ecosistèmics en la gestió de les pesqueries. Però aquest terme no va trobar consens en la Conferència de la FAO a Reykjavik el 2001, ja que es va considerar que posava l'estat ecosistèmic per damunt dels interessos socio-econòmics. Finalment, el terme Enfoc Ecosistèmic de les Pesqueries ("Ecosystem Approach to Fisheries- EAF") va ser adoptat per la FAO el 2002. Tot i ser bàsicament el mateix, el nou terme inclou també factors socio-econòmics a més dels ecosistèmics, de manera que satisfeia els interessos dels estats participants (Garcia et al. 2003).

Els objectius principals de l'EAF són els de dissenyar plans de gestió pesquers tenint en compte el coneixement i les incerteses sobre els elements biòtics, abiòtics i també l'humà que formen part del sistema a gestionar, aplicant d'aquesta manera una aproximació integrada a la gestió de les pesqueries. Seguint aquests principis, el concepte d'EAF es va incloure ja en la revisió de la Política Pesquera Comú (PPC) de la Unió Europea (UE) el 2002 (Lutchmann et al. 2008). Així mateix, el 2008, la UE va aprovar la Directiva Marc sobre l'Estratègia Marina (Marine Strategy Framework Directive - MSFD), en què els països membres es comprometen a portar a terme polítiques respectuoses amb el medi marí amb l'objectiu d'aconseguir un Bon Estat Mediambiental (Good Environmental Status -GES) el 2020 (Lutchman et al 2008, EC2008/56). A més a més, l'última revisió de la PPC del 2013 inclou noves mesures amb l'objectiu de continuar avançant cap a la implementació d'un EAF (p.ex. mesures tècniques per a protegir els estocs comercials i els ecosistemes marins, obligació de desembarcar totes les captures incidentals, la regionalització de les polítiques pesqueres, més implicació dels gestors i pescadors en el disseny de les polítiques pesqueres, etc.) (Salomon & Holm-Müller 2013).

El disseny d'un pla de gestió segons la filosofia de l'EAF requereix una visó integrada del sistema a gestionar. Així doncs, el primer pas bàsic per a establir un EAF és determinar els elements d'aquest sistema, ja siguin elements naturals o socio-econòmics, i definir el seu paper en el sistema així com les relacions entre ells (Fletcher 2008, Levin et al. 2009). En el marc d'aquesta tesi s'estudiaren de manera integrada els elements involucrats en la pesca d'arrossegament: flota pesquera, espècies objectiu, ecosistema bentònic, elements de gestió de la pesqueria i la societat que se'n beneficia. Els treballs aquí presentats avaluen les relacions entre ells centrant-se en analitzar la millor manera de descriure la pressió pesquera i estudiant els processos ecosistèmics bentònics sotmesos a diferents nivells d'esforç pesquer amb l'objectiu d'utilitzar els coneixements obtinguts en la gestió pesquera.

2. LA NECESSITAT D'ADOPTAR UN EAF: ELS IMPACTES DE LA PESCA D'ARROSSEGAMENT EN ELS ECOSISTEMES

Les tècniques de pesca es poden agrupar en dues categories principals: tècniques passives i tècniques actives. Les primeres inclouen nanses, palangres, tresmalls, soltes, etc., i les segones inclouen els típics bous que arrosseguen xarxes pel fons marí, així com els rastells i gàbies (Lleonart & Sardà 1986). Totes aquestes tècniques provoquen múltiples canvis en els ecosistemes, com ara alteracions en les relacions tròfiques i no tròfiques entre organismes marins (Pauly 1998, Dill et al. 2003) i també afecten les poblacions d'ocells marins predadors (Frederiksen et al. 2004, Lewison et al. 2004, Tudela 2004). A més a més, per no subestimar la mortalitat provocada per la pesca, també cal tenir en compte les captures accidentals i el rebuig (Bellido et al. 2011) que no només afecta els peixos o els invertebrats, sinó també la megafauna com tortugues o mamífers marins (Frid et al. 2005, Lewison et al. 2004, Tudela 2004). Tot i així, l'impacte més gran de les pesqueries és el provocat pels arts d'arrossegament en els fons marins (Dayton et al. 1995, Jennings & Kaiser 1998, Thrush et al. 1998).

El nivell d'impacte dependrà en última instancia del nivell d'esforç pesquer, així doncs, primer de tot cal una estimació precisa d'aquests nivells d'esforç pesquer. Existeixen varies maneres d'estimar l'esforç pesquer (p.ex. àrea escombrada per unitat de temps, captures per unitat d'esforç (CPUE), hores al mar, etc.), però en general estan calculades a una escala massa gran que no permet una bona relació entre l'esforç i l'estat del bentos (Piet et al. 2007, McCluskey & Lewison 2008). A més a més, els efectes de la pesca en les comunitats bentòniques sovint són irregulars, per tant per a avaluar-los cal una estimació precisa a petita escala (Piet & Quirijns 2009) (vegeu l'objectiu 1 en la secció d'objectius).

L'arrossegament afecta tant l'hàbitat biòtic, com l'abiòtic. Pel que fa a l'hàbitat abiòtic, el pas de l'art homogeneïtza el fons, altera les característiques del sediment i incrementa la terbolesa del fons (O'Neill & Summerbell 2011, Palanques et al. 2014). Per altra banda, el bou tendeix a eliminar els organismes grans i sèssils (més vulnerables), deixant també organismes danyats al seu pas, la qual cosa provoca un canvi cap a comunitats dominades per organismes petits, carronyers i de vida curta (Kaiser et al. 2000, Thrush & Dayton 2002, de Juan et al. 2007). Aquests canvis en les comunitats bentòniques també poden afectar indirectament les espècies comercials, ja que aquestes necessiten unes característiques d'hàbitat determinades per a completar els seus cicles biològics, el que s'anomena Habitat Essencial (Essential Fish Habitat EFH) (Auster & Langton 1998). Per tant, per a explotar de manera sostenible les espècies comercials, s'han de tenir en compte els efectes de la pesca d'arrossegament en l'ecosistema, la qual cosa suposa un canvi de paradigma en la gestió i l'establiment d'un EAF.

2.1. Evolució de l'estudi dels impactes de la pesca d'arrossegament en les comunitats bentòniques: des de l'aproximació específica a la funcionalitat de l'ecosistema.

Són molts els estudis realitzats per tal d'entendre els efectes de la pesca en les comunitats bentòniques. Els primers estudis, basats en la composició específica de l'ecosistema (e.g. de Groot 1984, Philippart 1998, Norse & Watling 1999, Kaiser et al. 2000) van permetre augmentar el coneixement pel que fa a les respostes de la comunitat, però no aportaren gaire informació sobre funcionalitat ecosistèmica i, per altra banda, la informació que aportaven era geogràficament restringida (Bremner 2008). Per a abordar la qüestió de la funcionalitat, han proliferat els treballs basats en l'anàlisi de trets biològics (i.e. basats en característiques del comportament, de la morfologia i del cicle de vida de les espècies) (Bremner et al. 2006b, Tillin et al. 2006, de Juan et al. 2007). Aquests treballs ressalten respostes difícilment observables amb l'aproximació específica i a més a més, com que els trets biològics estan estretament relacionats amb els processos ecosistèmics, també permeten analitzar l'ecosistema des del punt de vista de les funcions ecosistèmiques (p.ex. el comportament bioturbador d'algunes espècies potencia els fluxos de nutrients entre el sediment i la columna d'aigua (Emmerson & Raffaelli 2000, Lohrer et al. 2004, Laverock et al. 2011)). Així mateix, l'estudi de la funcionalitat ecosistèmica permet entendre millor com els processos ecosistèmics canvien quan l'ecosistema es veu sotmès a algun impacte.

Per altra banda, en el context de la funcionalitat ecosistèmica, també és important el concepte de redundància, és a dir espècies funcionalment equivalents que mostren respostes diferents enfront a l'estrès, ja que aquest concepte està lligat al de la resiliència dels ecosistemes (Walker et al. 1999, Folke et al. 2004, Bremner 2008). Si s'entén la resiliència com la capacitat del sistema per a absorbir la pertorbació i reorganitzar-se per a mantenir essencialment la mateixa funció, estructura, identitat i circuits de retroalimentació (Folke et al. 2004). La redundància funcional representa una assegurança per a mantenir l'estat de l'ecosistema sotmès a un estrès. Tot i així, és difícil determinar com interaccionen la pèrdua d'espècies i la redundància per a mantenir la funció, ja que, dins un conjunt d'espècies redundants, algunes poden tenir una contribució més gran a un procés ecosistèmic que les altres (Bolam et al. 2002, Solan et al. 2004). La desaparició d'aquestes espècies podria comportar canvis dràstics en la funcionalitat de l'ecosistema provocant un canvi a gran escala en la composició i funció de les espècies de l'ecosistema en qüestió. Aquest canvi, que pot ser molt difícil de revertir, dóna lloc a la reorganització de les relacions ecosistèmiques i s'estableix un ecosistema alternatiu amb una funcionalitat diferent (Folke et al. 2004, Thrush et al. 2009).

Així doncs, el coneixement de la funcionalitat d'un ecosistema és un element clau en l'aproximació ecosistèmica a la gestió de les pesqueries i per tant és una de les qüestions tractades en els estudis presentats en aquesta tesi (vegeu els objectius 2 i 3 en la secció d'objectius).

2.2. Evolució de l'estudi dels impactes de la pesca d'arrossegament en les comunitats bentòniques: de la funcionalitat ecosistèmica als serveis ecosistèmics.

Últimament, l'aproximació funcional a l'estudi dels ecosistemes ha evolucionat cap al concepte de "serveis ecosistèmics", que permet abordar la qüestió incloent-hi una visió social (Bremner et al. 2006a, Bello et al. 2010, Granek et al. 2010, Frid 2011). Els serveis ecosistèmics es poden definir com "els beneficis que la gent obté dels ecosistemes" (MEA 2005), la qual cosa permet incloure la societat en la gestió dels ecosistemes. Així doncs, el concepte de funcionalitat ecosistèmica evoluciona per passar d'un marc exclusivament ecològic a incloure aspectes socials en l'estudi dels ecosistemes, seguint així la filosofia de la gestió integrada de l'EAF (Salomon et al. 2011). A més a més, com que el concepte de "serveis ecosistèmics" inclou elements socioeconòmics a més d'ecològics, permet acostar posicions entre científics i gestors (Granek et al. 2010, Atkins et al. 2011).

Centrant-nos en els fons marins tous sotmesos a la pressió pesquera, els anomenats caladors, el servei més obvi que se n'obté és la "provisió d'aliment" mitjançant les captures. Nogensmenys, els caladors proveeixen altres serveis ecosistèmics, sobretot serveis de suport com ara la provisió d'hàbitat o el reciclatge de nutrients, els quals també s'han de tenir en consideració a l'hora de dissenyar un pla de gestió (Garcia & Cochrane 2005). A més a més, cal recordar que el servei de provisió d'aliment se sustenta en alguns d'aquests altres serveis (e.g. producció secundària o provisió d'hàbitat) (vegeu el capítol 3). Per tant, L'EAF ha de considerar globalment tots els serveis ecosistèmics que proveeix el calador (aquesta qüestió es discuteix en el capítol 2, secció 2).

3. IMPLEMENTACIÓ DE L'ENFOC ECOSISTÈMIC DE LES PESQUERIES

Tot i que les polítiques pesqueres de molts països arreu del món estan evolucionant cap a l'aplicació de l'EAF, aquesta transició no és fàcil, ja que significa un canvi en el paradigma de la gestió de les pesqueries (Murawski 2007, Rice 2011, Berkes 2012). Tot i així existeixen algunes experiències reeixides.

L'EAF s'ha aplicat exitosament en les pesqueries australianes (McLoghlin et al. 2008, Day 2008) i en algunes pesqueries nord-americanes (Tromble 2008), tot i que amb aproximacions lleugerament diferents. En el cas australià, els plans de gestió s'han basat principalment en el desenvolupament de dues eines de gestió: els plans d'avaluació de risc que tenen en compte els canvis en l'ecosistema i les estratègies de captura dels estocs. Aquestes dues tècniques s'han aplicat a totes les pesqueries australianes i estan en constant revisió, de manera que les estratègies de captura poden canviar segons la necessitat (McLoghlin et al. 2008). Per altra banda, en els EUA, el sistema està basat en l'Avaluació Integrada dels Ecosistemes (Integrated Ecosystem Assessment- IEA) (Levin et al. 2009). En aquest sistema s'utilitzen eines com els indicadors ecològics i els models ecosistèmics, de manera que la gestió està basada principalment en els resultats d'aquests models (Link et al. 2011). Aquests dos exemples, tot i aplicar mesures diferents, se centren només en gestionar l'activitat pesquera.

A Canadà podem trobar un altre exemple d'EAF, englobat dins la visió més general de la gestió integrada dels usos marins, en què també es tenen en compte altres activitats com la indústria petrolífera o l'aqüicultura (O'Boyle & Jamieson 2006). En aquest cas es tracta d'arribar a un acord entre tots els usuaris d'una zona per tal d'establir uns objectius de gestió concrets, per exemple la conservació de determinats elements ecosistèmics.

A Europa l'EAF s'ha aplicat en aigües noruegues (Mar de Barents i Mar de Noruega), encara que s'han detectat alguns problemes en la supervisió del pla a causa de la manca de coordinació entre ministeris (Ottersen et al. 2011). De manera semblant a l'experiència canadenca, a Noruega també s'ha aplicat un pla de gestió integrada que engloba altres pressions antropogèniques a part de la pesca (Winsnes & Skjoldal 2007, Ottersen et al. 2011). Per a cada sector implicat (transport marítim, industria petrolífera, etc.) es va dur a terme una avaluació ambiental, parant especial atenció a la identificació d'àrees vulnerables, i a partir d'aquest estudi s'ha dissenyat un pla de gestió i de seguiment.

Les pesqueries del Mar del Nord també estan fent passos cap a la implementació de l'EAF, instaurant mesures de gestió que tenen en compte no només una aproximació específica sinó també una visió ecosistèmica (e.g. avaluació multiespecífica) (Frid et al. 2005, Frid et al.

2006). També es va desenvolupar un pla de gestió basat en un procés consultiu amb tots els actors implicats en les pesqueries, però només es va arribar a un acord de mínims (Paramor et al. 2005).

Malgrat tota la legislació vigent, en cap altre lloc d'Europa, tampoc en el Mediterrani, s'ha implantat una pla de gestió de les pesqueries basat en l'EAF. Aquest fracàs podria ser degut a la manca de suficient coneixement sobre la dinàmica ecosistèmica així com a la falta de col·laboració i comunicació entre els actors involucrats en les pesqueries (polítics, gestors, ONGs, científics i pescadors) (Coll et al. 2013, Daw & Gray 2005, de Juan et al. 2012, Frid et al. 2006).

La comunicació entre tots els actors involucrats en la gestió de les pesqueries és un factor clau per a la implementació de l'EAF. No obstant això, aquesta comunicació no sempre és fàcil ja que els diferents actors normalment utilitzen diverses aproximacions a l'hora d'abordar una mateixa informació (Verweij et al. 2010, Soomai et al. 2011). Per exemple, un gràfic científic que mostri l'evolució de la biomassa reproductora de l'estoc difícilment serà comprès per un pescador que no està familiaritzat amb el concepte i que es guia per dades de captures. Per tant, els científics han de fer un esforç per a presentar les conclusions de la seva recerca de manera que sigui fàcilment comprensible per tothom per tal que aquest coneixement sigui útil a l'hora d'elaborar un EAF. (Coll et al. 2013, Garcia & Cochrane 2005). En aquest context, un dels objectius d'aquesta tesi és la construcció d'una plataforma de coneixement on es presentin els efectes de la pesca en les comunitats bentòniques d'una manera fàcilment comprensible per tothom (vegeu l'objectiu 4 en la secció d'objectius).

3.1. L'EAF en el Mediterrani

Malgrat la necessitat d'adoptar un EAF, la gestió actual de les pesqueries mediterrànies encara esta basada en l'avaluació d'estocs individuals (Spagnolo 2012). Tot i que existeixen algunes iniciatives locals per a implementar una gestió més enfocada a l'ecosistema, aquestes no acaben de reeixir a causa de la seva pobra coordinació (Coll et al. 2013).

A l'hora d'implementar un EAF en el Mediterrani sorgeixen força problemes: la manca d'una recol·lecció sistemàtica de dades sobre les comunitats biològiques i les activitats pesqueres o el complex escenari geogràfic i socio-polític que inclou països membres de la UE, països de l'Orient Mitjà i països nord-africans (Coll et al. 2013, de Juan et al. 2012, Caddy 2012, Spagnolo 2012). En un treball recent, Sartor et al. (2014) van determinar que actualment només es disposa d'una petita part de la informació necessària per a implementar un EAF en el Mediterrani.

El 2005 es promogué l'Estratègia per a un Enfocament Ecosistèmic Mediterrani (The Mediterranean Ecosystem Approach Strategy - ECAP) amb l'objectiu de sumar esforços per a aconseguir un medi ambient saludable (Healthy Environment Statuts, similar a la idea del Bon estat Mediambiental de la Directiva Marc sobre l'Estratègia Marina). (Cinnirella et al. 2014). Aquesta estratègia es va emprendre sota el paraigües del Pla d'Acció del Mediterrani (Mediterranean Action Plan-MAP), de manera que fos vinculant per a tots els seus membres (21 països de la riba del Mediterrani), però no s'ha implementat adequadament per manca de recursos humans i econòmics (Cinnirella et al. 2014).

Recentment, ha sorgit una iniciativa interessant per tal d'enfortir la cooperació entre els països Mediterranis en l'objectiu d'establir-hi un EAF: el projecte EMBASEAS (Eaf in the Mediterranean and BlAck SEAS). Aquest projecte promou una aproximació científica per a l'EAF coordinant iniciatives i compartint informació entre investigadors mediterranis a la vegada que pretén establir ponts de comunicació entre científics, gestors i altres usuaris del mar (EMBASEAS Network Constitutional Framework).

El projecte EMBASEAS ha estat promogut dins en projecte CREAM (Coordinating research in support to application of Ecosystem Approach to Fisheries and management advice in the Mediterranean and Black Seas), en el qual també s'ha identificat la manca de coneixement en alguns aspectes necessaris per a la implementació d'un EAF, com per exemple la quantificació de l'esforç pesquer real, la descripció de procesos ecosistèmics bàsics, la quantificació de serveis ecosistèmics, etc. (Coll et al. 2013). Aquesta tesi aprofundeix en alguns d'aquests temes amb l'objectiu d'incrementar la base de coneixement necessària per al diseny d'un EAF mediterrani.

3.1.1. Àrees marines protegides (AMPs): un primer pas cap a l'EAF en el Mediterrani.

Les AMPs són considerades com una mesura directa d'implementar i que pot representar un primer pas cap a l'establiment d'un EAF mediterrani (Stergiou 2002, Tsikliras & Stergiou 2007, García-Charton et al. 2008). De fet, s'ha demostrat que les AMPs afavoreixen el creixement de les poblacions i dels individus així com la seva reproducció i reclutament (Tsikliras & Stergiou 2007, Harmelin-Vivien et al. 2008, Higgins et al. 2008). No obstant això, existeixen poques evidències quantitatives de la contribució d'aquest creixement poblacional a les pesqueries circumdants (Harmelin-Vivien et al. 2008, Higgins et al. 2008, García-Rubies et al. 2013). Per altra banda, les comuntitats bentòniques de les AMPs en general experimenten una recuperació i un augment de la seva complexitat (Babcock et al. 1999, de Juan et al. 2011). No obstant això, les AMPs han de ser suficientment grans perquè aquesta recuperació de l'ecosistema (hàbitats i comunitats) tingui efectes sgnificativament beneficiosos en les espècies objectiu (Allison et al. 1998, Browman & Stergiou 2004, de Juan et al. 2012).

A més a més, en la majoria de la superficie protegida es permeten activitats com la pesca artesanal i el submarinisme que si no són degudament controlades poden perjudicar les comunitats que aquestes AMP pretenen protegir (García-Charton et al. 2008, Purroy et al. 2014, Font 2014). També és important recordar que l'establiment d'una AMP en caladors d'arrossegament pot provocar una redistribució de l'esforç que pot comportar l'augment de la pressió pesquera en zones on abans no era tan forta (Whitmarsh et al. 2002, Dinmore et al. 2003, Hiddink et al. 2006). Així doncs, cal parar esment a aquest fenomen i controlar aquest desplaçament de l'esforç pesquer quan s'estableix una AMP en caladors de pesca d'arrossegament.

Nogensmenys, una xarxa d'AMPs dissenyada seguint criteris ecològics podria donar lloc a uns beneficis més clars per a les pesqueries (Hilborn et al. 2004, Gaines et al. 2010). Malauradament, la majoria de les AMP mediterrànies han estat definides seguint principalment criteris socio-polítics (Francour et al. 2001, Coll et al. 2012, de Juan et al. 2012). Això ha resultat en una sèrie de petites AMPs costaneres localitzades al voltant d'illes i en fons rocosos, on no hi són representats ni els hàbitats profunds ni els fons tous dels caladors (Lloret et al. 2008, Gabrié et al. 2012).

Les AMPs han de ser considerades com una eina de gestió dins del plantejament més ampli de l'Ordenació de l'Espai Marí (Marine Spatial Planning-MSP) que té com a objectiu gestionar conjuntament els múltiples usos del mar (Douvere 2008). El MSP permet organitzar l'espai determinant àrees amb diferents usos, des de zones on l'activitat humana hi és del tot prohibida fins a zones d'accés totalment obert. D'aquesta manera es tenen en compte al mateix temps els interessos pesquers i els conservacionistes i es contribueix a mantenir una explotació sostenible dels ecosistemes marins (Gaines et al. 2010, Salomon et al. 2011).

4. LES PESQUERIES MEDITERRÀNIES

4.1. Característiques de les pesqueries

Per tal de comprendre les dificultats que s'han d'enfrontar a l'hora de dissenyar i implementar un EAF en el Mediterrani val la pena fer un petit repàs de les característiques de les pesqueries mediterrànies i les seves regulacions actuals.

Les pesqueries mediterrànies són típicament multiespecífiques amb diferents arts competint pel mateix recurs (Farrugio et al. 1993, Lleonart et al. 1998, Spagnolo 2012). També estan caracteritzades per l'alta demanda de peix fresc i l'alt cost de l'explotació del recurs, la qual cosa fa que els preus de mercat siguin normalment alts. Així doncs, a l'hora d'estudiar les pesqueries mediterrànies cal tenir en compte els ingressos a més a més del pes de les captures (Spagnolo 2012, Caddy 2012).

Al Mediterrani s'hi poden trobar dos grans tipus de pesqueries (Demestre 1986, Lleonart 1990, Papaconstantinou & Farrugio 2000, Lleonart & Maynou 2003):

i) Pesqueries pelàgiques. Caracteritzades pels arts d'encerclament (teranyines) utilitzats en la pesca de petits pelàgics (sardina i anxova), els palangres de superfície i xarxes a la deriva per als grans pelàgics (emperador i tonyina).

ii) Pesqueries bento-demersals. Aquest sector comprèn varis arts: tresmall, soltes, palangres de fons, rastells, nanses i l'art més àmpliament utilitzat, el bou, caracteritzat per un ampli nombre d'espècies objectiu com ara el lluç, el moll de fang, el rap, la gamba, la llagosta, el llenguado, el calamar, el pop, etc. També en alguns llocs de la mediterrània hi són permesos els bous semipelàgics.

El gruix de la flota mediterrània (entre un 70-80% del total) està compost per petits vaixells de pesca artesanal que utilitzen una àmplia gamma d'arts de pesca (Papaconstantinou & Conides 2007, Spagnolo 2012). De fet, la majoria dels arts mencionats més amunt corresponen a aquest tipus de pesqueries i tan sols l'encerclament, la pesca de la tonyina i el bou es realitzen amb vaixells de més de 12 m d'eslora (Lloret et al. 2008).

Entre totes les arts mencionades més amunt, el bou és l'art que provoca un efecte més perjudicial i crònic en els ecosistemes bentònics (Tudela 2004, de Juan et al. 2007, Sacchi 2008, Mangano et al. 2013). Els bous operen principalment en caladors situats a la plataforma continental. Tanmateix, a causa de l'estretesa de la plataforma continental mediterrània, el talús continental es troba situat molt a prop de la costa, la qual cosa ha afavorit l'expansió de la pesca cap a zones més profundes (talús superior i canyons submarins) (Demestre & Martín 1993, Sardà 2000, Bas 2006). Els caladors explotats es troben normalment prop de la costa, de manera que els vaixells tornen normalment cada dia al port base per a vendre-hi les captures (Caddy 1993, Lleonart et al. 1998, Bas 2006, Spagnolo 2012).

Una altra característica de els pesqueries mediterrànies és la seva estacionalitat, és a dir, el canvi de caladors que realitzen els pescadors al llarg de l'any seguint el cicle biològic de les espècies objectiu (Sardà

& Martín 1986, Demestre et al. 1997, Martin et al. 1999, Halley & Stergiou 2005). Malauradament, en alguns casos, la flota d'arrossegament es concentra en zones de reclutament (p.ex. moll de fang (Demestre et al. 1997, Martin et al. 1999)). Aquest comportament, sumat a l'elevat esforç pesquer a què està sotmès bona part del Mediterrani, ha portat a la sobreexplotació de molts estocs comercials (Colloca et al. 2013, Lleonart 2008, Lleonart & Bas 2012).

A més a més, per a tenir una visió global dels efectes de la pesca d'arrossegament en el Mediterrani, també cal tenir en compte la part de les captures que es rebutja; aproximadament un 45% del total de les captures anuals (Vassilopoulou et al. 2007, Sacchi 2008, Tudela 2004). Els individus i les espècies rebutjades són normalment més petites i pertanyen a un nivell tròfic significativament més baix que les espècies comercials (Martín 2001, Sánchez et al. 2004, 2007, Vassilopoulou et al. 2007). Això, juntament amb el fet que aquest rebuig aporta excedents d'aliment per a ocells i espècies marines (Vassilopoulou et al. 2007, Bellido et al. 2011, Tudela 2004), provoca canvis en l'estructura funcional de l'ecosistema. En aquest context, cal tenir en compte l'obligatorietat de desembarcar totes les captures d'espècies comercials que promulga la nova PPC. Segons la Comissió Europea, la norma pretén incentivar els pescadors a fer els seus arts més selectius, però per contra podria fer emergir un mercat per a les captures incidentals que no provocaria aquest incentiu (Bellido et al. 2011).

4.2. Regulacions pesqueres en el Mediterrani

Tot i que la PPC és d'aplicació a totes les pesqueries de la UE, va ser inicialment dissenyada per a les pesqueries atlàntiques i del nord d'Europa, la qual cosa la fa de difícil implementació en el complex escenari mediterrani (Casado 2008). Per exemple, el sistema de quotes que s'utilitza per a la gestió de les pesqueries atlàntiques, no és viable en les pesqueries multiespecífiques mediterrànies i només s'aplica a la pesca de la tonyina i similars que si que és monospecífica (Cacaud 2005, Caddy 2012).

Per això, abans de l'última revisió de la PPC, es van redactar dues lleis europees especialment dirigides a les pesqueries mediterrànies (EC 1626/94 i EC 1967/2006). Tanmateix, cap de les dues ha estat completament desplegada (Casado 2008). A més a més, cal recordar que, dels 21 països que rodegen el Mediterrani, només vuit pertanyen a la UE, així doncs les normatives pesqueres en el Mediterrani estan regulades pel Consell General de Pesca del Mediterrani (CGPM, que inclou països de fora la UE) i les seves resolucions són adoptades per la normativa mediterrània de la UE (Papaconstantinou & Conides 2007, Casado 2008, Caddy 2012).

Aquestes normatives de gestió de les pesqueries al Mediterrani es basen principalment en el control de l'esforç i en l'establiment de talles de captura mínimes.

a) Normatives de control d’esforç

Els arts d’arrossegament estan prohibits a menys de 3 milles nàutiques de la costa o a menys de 50 m (EC 1626/94) i més recentment també se n’ha prohibit l’ús per sota dels 1000 m (EC 1967/2006). Per altra banda i seguint el consell de la CGPM de reduir l’esforç pesquer total, molts països han optat per limitar el nombre de llicències de pesca i la UE ha establert una sèrie de mesures per a controlar i reduir la seva capacitat pesquera (Cacaud 2005, Spagnolo 2012). No obstant això, l’avenç tecnològic incrementa la capacitat pesquera i l’esforç total pràcticament no canvia (Lleonart & Maynou 2003, Spagnolo 2012). En el Mediterrani a més a més també es limita el temps de pesca (dies a la setmana o hores al dia) i la potencia dels vaixells (Papaconstantinou & Farrugio 2000, Cacaud 2005). Una altra mesura de control d’esforç són les vedes estacionals, que en el Mediterrani poden anar d’un a quatre mesos, implantades sobretot per a protegir els períodes de reclutament d’algunes espècies comercials (Kapantagakis 2007, Demestre et al. 2008). Tanmateix, en alguns casos, la flota d’arrossegament es concentra en zones determinades just després de la veda per a pescar aquests nous reclutes, la qual cosa minimitza els possibles efectes beneficiosos d’aquesta mesura de gestió.

b) Talles mínimes de captura

Teòricament aquesta mesura pretén impedir la captura d’individus immadurs, però en molts casos la talla mínima de captura és inferior a la talla de primera maduresa. Per exemple, la talla mínima de captura del lluç és de 20 cm mentre que la talla de primera maduresa és de 39 cm per a les femelles i 32 cm pels mascles (Lleonart et al. 1998, Recasens et al. 1998, Assessment 2011).

A més a més, tradicionalment en el Mediterrani s'han utilitzat xarxes amb llums de malla molt petites, de manera que en les captures hi havia molts individus per sota la talla mínima de captura (Bas 2006, Papaconstantinou & Conides 2007). Així doncs, la regulació europea 1967/2006 va establir que la llum de malla del cóp ha de ser d'almenys 40 mm. A més a més, aquesta malla ha de ser quadrada, ja que s'ha demostrat que és més selectiva que la tradicional malla romboide (Bahamon et al. 2006, Ordines et al. 2006, Sacchi, 2008), però tot i així encara es capturen individus per sota la talla mínima de captura (Sala et al. 2008).

5. COMUNITATS DE FONS TOUS EN EL MEDITERRANI: ÀREES D'ESTUDI

Contràriament al que pot semblar, les comunitats de fons tous presenten ecosistemes estructuralment complexos que duen a terme importants processos ecològics i proveeixen molts serveis ecosistèmics (Snelgrove 1997, Thrush & Dayton 2002). Moltes espècies comercials viuen en aquests fons, per tant aquests ecosistemes constitueixen importants caladors de pesca d'arrossegament i estan exposats als seus impactes. Cal doncs conèixer com responen aquets ecosistemes a l'impacte de la pesca per tal de dissenyar plans de gestió seguint la filosofia de l'EAF.

Així doncs, per als estudis d'aquesta tesi es van escollir set àrees del Mediterrani situades en fons tous de la plataforma continental entre

40-80 m. Tres d'aquestes àrees es troben en la costa catalana (Cap de Creus-CC, Medes-M and Ebre delta-D), una altra a la costa de Murcia (Cabo de Palos-CP), dues a la costa italiana (costa de Liguri-L i costa de l'Adriàtic-A) i finalment una àrea a Grècia (costa Jònica-I). En totes aquesta àrees hi operen els bous en major o menor mesura, excepte a l'àrea M i part de l'àrea de CC que també corresponen a AMPs on l'arrossegament hi està prohibit.

Aquestes àrees comprenen diversos tipus d'hàbitats: les àrees L, I i D presenten un sediment fangós mentre que les àrees CC i A mostren un percentatge més alt de sorra en els seus sediments. L'àrea M està caracteritzada per sediments fango-detrítics costaners i l'àrea de CP presenta fons fango-sorrencs amb presència de maërl (de Juan et al. 2013).

Les comunitats bentòniques estan formades per l'epifauna (organismes que viuen sobre el sediment, tot i que a vegades també s'hi poden enterrar, p.ex. estrelles de mar, bivalves, crancs...) i per la infauna (organismes més petits que viuen dintre el sediment, p.ex. poliquets i petits crustacis). Per tal de caracteritzar correctament la comunitat bentònica cal tenir en compte els dos compartiments (Jørgensen et al. 2011), així doncs l'epifauna es va estudiar a totes les àrees mentre que la infauna només es tingué en compte a D, CC,L i I. L'epifauna es va mostrejar amb un art experimental tipus rastell que presenta un marc de ferro de 2 m d'obertura i un cóp d'1 cm de malla. La infauna es va mostrejar amb una draga Van Veen. Totes les mostres es van recol·lectar entre la primavera i principis d'estiu dels anys 2003, 2007 i 2009.

Les dades disponibles s'han utilitzat diferentment en els següents capítols segons els objectius de cadascun.

OBJECTIUS DE LA TESI

Els objectius d'aquesta tesi estan descrits en el marc de l'EAF ja que els resultats contribuiran a proveir informació necessària per a dissenyar un pla de gestió seguint aquesta filosofia.

L'objectiu principal de la tesi és estudiar de manera integrada la resposta de la comunitat bentònica (infauna i epifauna) a la pesca d'arrossegament en la plataforma continental del Mediterrani i avaluar com la provisió de serveis ecosistèmics (e.g. captures comercials) es pot veure compromesa per aquesta resposta.

Aquest objectiu principal es va dividir en cinc objectius específics que són abordats en els diferents capítols. Cada objectiu específic representa un pas cap a la comprensió de la dinàmica dels ecosistemes bentònics sotmesos a la pesca d'arrossegament; des de la descripció i estimació precisa de l'esforç de pesca (objectiu 1), fins a com la pesca pot afectar indirectament les espècies comercials (objectiu 3), passant per la descripció dels efectes de l'arrossegament en els processos funcionals (objectiu 2). Finalment, també es presenta la integració de tot aquest coneixement en un model de simulació (objectiu 4). L'objectiu 5 representa l'aplicació del coneixement adquirit en els objectius precedents en la implementació d'un EAF en el Mediterrani.

Objectius específics:

1. **Explorar una metodologia per a conèixer la dinàmica de la flota i estimar el nivell d'esforç pesquer a una escala bentònica.** Per a aconseguir aquest objectiu, en la secció 1 del **capítol 1** s'explora el potencial de les dades obtingudes a partir del sistema de control de vaixells per satèl·lit (Vessel Monitoring System - VMS) analitzades conjuntament amb dades de biomassa i ingressos de captures obtingudes en les subhastes al port per a controlar la distribució d'una flota d'arrossegament del nord de Catalunya. En la secció 2 d'aquest mateix capítol s'explora la capacitat de tres estimacions diferents de l'esforç pesquer (entrevistes amb els pescadors, VMS i imatges del sonar d'escombratge lateral (side scan sonar - SSS)) per a relacionar aquest esforç amb els canvis observats en les comunitats bentòniques.

2. **Estimar la resposta de la comunitat bentònica sotmesa a diferents nivells d'esforç pesquer des del punt de vista funcional i de serveis ecosistèmics.** La secció 1 del **capítol 2** té com a objectiu explorar com es comporten diverses funcions ecosistèmiques quan estan sotmeses a la pertorbació provocada per la pesca d'arrossegament i com els canvis potencials en aquestes funcions podrien afectar la provisió de serveis ecosistèmics. La secció 2 explora la capacitat de la redundància funcional mostrada per la comunitat bentònica per a ser utilitzada com a mesura de resiliència de l'ecosistema enfront la pesca d'arrossegament.

3. **Relacionar els canvis provocats per l'arrossegament en les comunitats dels fons marins amb els efectes potencials d'aquests canvis en les espècies comercials.** En el **capítol 3** es compara la funcionalitat dels ecosistemes bentònics d'un calador de pesca d'arrossegament i la d'una zona control que no ha estat pertorbada des de fa 20 anys. En base a aquests resultats, es discuteixen els possibles efectes del canvi en la comunitat bentònica a causa de l'arrossegament en la població d'una espècie comercial important en el Mediterrani, el moll de fang (*Mullus barbatus*).

4. **Desenvolupar un model conceptual i de simulació basat en l'EAF i que pugui ser utilitzat per qualsevol dels actor involucrats en les pesqueries.** Per tal d'abordar aquest tema, en el **capítol 4** es presenta una plataforma de coneixement fàcil d'utilitzar que representa la comunitat bentònica a través de trets biològics vulnerables/resistents a la pesca d'arrossegament. En aquesta plataforma l'usuari pot visualitzar l'estructura de la comunitat bentònica sotmesa a diferents nivells d'esforç pesquer i també ofereix la possibilitat de canviar l'esforç original per tal d'estimar els canvis potencials d'aquest canvi en la comunitat.

5. **Suggerir mesures de gestió en el context de l'EAF per tal d'atenuar els efectes de les pesqueries d'arrossegament demersals en les comunitats bentòniques.** Aquest objectiu s'aborda en el **capítol 5**, basat en els resultats dels capítols anteriors així com en informació de la literatura científica. En aquest capítol es recomanen una sèrie de mesures que es podrien adoptar per a la implementació d'un EAF en la

flota d'arrossegament catalana i que podria ser extensible a altres pesqueries d'arrossegament mediterrànies.

DISCUSSIÓ GENERAL

El disseny d'un pla de gestió segons la filosofia de l'EAF implica una aproximació integral que tingui en compte el coneixement ecològic de l'ecosistema així com els elements socio-econòmics. En aquest context ha sorgit el concepte de "serveis ecosistèmics", basat en com la interrelació de vàries funcions ecosistèmiques resulta en un servei per la societat (Boyd & Banzhaf 2007, Díaz et al. 2015). Així doncs, cal un bon coneixement de les funcions i processos ecosistèmics i de la seva resposta a la pesca d'arrossegament per tal de gestionar-los adequadament.

Per a contribuir al coneixement científic necessari per a l'aplicació d'un EAF, l'objectiu principal d'aquesta tesi és el d'avaluar la resposta de la funcionalitat de les comunitats bentòniques a la pesca d'arrossegament.

1. AVALUANT ELS IMPACTES DE LA PESCA D'ARROSSEGAMENT EN ELS FONS MARINS DINS EL MARC DELS SERVEIS ECOSISTÈMICS: DES DE LA FUNCIONALITAT DELS ECOSISTEMES BENTÒNICS FINS ALS EFECTES EN LES ESPÈCIES COMERCIALS

1.1. Impacte de l'arrossegament

La dinàmica dels ecosistemes bentònics dels caladors de pesca depèn principalment del nivell d'esforç a què estan sotmeses aquestes comunitats. (Thrush et al. 1998, Collie et al. 2005, Atkinson et al. 2011, de Juan & Demestre 2012). A més a més, la pròpia variabilitat de les comunitats bentòniques es dóna a una escala inferior a la del calador (Collie et al. 2005, de Juan et al. 2013), per tant, cal avaluar l'impacte del bou a una escala similar al bentos i aquest és el primer objectiu d'aquesta tesi. Per aquest motiu es va analitzar el resultat de tres mètodes diferents per a estimar l'esforç pesquer: entrevistes amb els pescadors i informacions obtingudes en les subhastes i les confraries, dades del sistema de localització de vaixells per satèl·lit (Vessel Monitoring System-VMS) i imatges del sonar de rastreig lateral (Side Scan Sonar- SSS).

La informació provinent de pesqueries permet establir aproximadament la situació dels caladors, però és massa poc precisa per relacionar-la directament amb l'estat de les comunitats bentòniques. Les dades provinents de VMS han estat utilitzades en alguns estudis per a avaluar l'estat de la comunitat bentònica en quadrícules d'alta resolució. (Gerritsen et al. 2013, Piet & Hintzen

2012). No obstant això, cal recordar que, tal i com també es va observar en el nostre estudi, l'actual freqüència d'enviament de senyals (un cada 2h) és massa baixa per a poder determinar l'esforç a petita escala (Deng et al. 2005, Skaar et al. 2011, Lambert et al. 2012). De totes maneres, tot i que s'augmentés la freqüència d'enviament de senyals, la pròpia naturalesa de les dades de VMS dissenyades per a controlar els moviments de la flota (un procés que es dóna a gran escala), fa que no siguin les més adequades per a avaluar processos a més petita escala com la resposta de les comunitats bentòniques a la pesca d'arrossegament. Entre els tres mètodes, el SSS és el que dóna una informació més detallada de l'impacte a nivell del bentos, ja que proporciona imatges de les marques que l'art de ròssec en el fons marí.

Les estimacions d'esforç pesquer calculades a partir de les dades provinents de pesqueries i del SSS es van utilitzar per a l'avaluació de cinc funcions ecosistèmiques produïdes per les comunitats bentòniques en diferents caladors de pesca (producció secundària, reciclatge de nutrients, acoblament bento-pelàgic, segrest de carboni i estructura d'hàbitat). Es va trobar un relació significativa de dues estimacions amb aquestes funcions, però l'esforç calculat a partir de les dades de SSS va donar millors resultats. Per tant, per a l'avaluació de l'estat de la comunitat bentònica, s'haurien d'utilitzar estimacions basades en el SSS sempre que fos possible. No obstant això cal recordar que el temps de permanència de les marques en el fons depèn del tipus de sediment i per tant aquest mètode podria no ser apropiat en sediments que presenten un alt dinamisme (p.ex. sorra) (Smith et al. 2007, Palanques et al. 2014).

Per altra banda, no sempre és possible obtenir informació de totes les fonts i normalment la que tenim més a mà és la informació de les pesqueries (McCluskey & Lewison 2008). Aquesta limitació s'ha tingut en compte a l'hora d'elaborar el bloc de simulació de la plataforma de coneixement del Capítol 4, ja que el model accepta diferents tipus de dades, estimacions d'esforç a partir de SSS i de dades de pesqueries.

A més a més, tot i que la informació obtinguda amb el SSS permet la visualització directa de les marques d'arrossegament en el sediment, no proporciona informació a nivell de dinàmica de la flota, la qual cosa també s'ha de tenir en compte a l'hora de gestionar les pesqueries (Stelzenmuller et al. 2008). En aquest aspecte, les dades del VMS constitueixen una valuosa font independent d'informació (Bertrand et al. 2008, Jennings & Lee 2012). Les dades de VMS, combinades amb dades de captures, també s'han utilitzat per a estimar la distribució de les espècies comercials (Fonseca et al. 2008, Gerritsen & Lordan 2010). Anant un pas més enllà, aquestes dades (VMS i captures) es podrien combinar també amb el coneixement del cicle biològic de les espècies objectiu, la qual cosa permetria la delimitació d'Hàbitats Essencials (EFH) per aquestes espècies susceptibles de ser protegits en plans de gestió. A més a més, amb l'obtenció de dades de SSS d'aquestes àrees, es pot detectar la variabilitat d'esforç pesquer dins de l'EFH i considerar la possibilitat de protegir aquelles zones menys impactades que presumiblement presentarien una comunitat més estructurada. Per tant, totes les fonts d'informació (pesqueries, VMS, SSS i cicle vital de les espècies) són útils per al disseny d'un pla de gestió segons l'EAF.

2.2. El marc dels serveis ecosistèmics

L'estudi de les funcions i processos bentònics es realitzà tenint en compte el marc més ampli dels serveis ecosistèmics. Aquest concepte engloba diferents nivells del sistema pesquer global, des dels trets biològics que presenten les espècies que determinen la seva funció en l'ecosistema fins a "l'ús ecosistèmic" que se'n deriva i que constitueix els benefici que n'acaba percebent la societat (MEA 2005). L'avaluació de la funcionalitat dels ecosistemes des del punt de vista dels trets biològics ja ha estat realitzada en diversos estudis (Tillin et al. 2006, de Juan et al. 2007, Bremner 2008). A partir del coneixement obtingut en aquests estudis, en aquesta tesi s'explorà la potencialitat dels trets biològics per a l'avaluació de les funcions produïdes pels ecosistemes bentònics en els caladors de pesca. Tot i que aquesta aproximació no permet el càlcul de fluxos de les funcions, es va demostrar útil per a valorar la resposta global de les funcions a la pesca d'arrossegament. Amb aquesta aproximació mitjançant trets biològics, en aquesta tesi hem volgut ressaltar la importància de la vessant ecològica del terme "serveis ecosistèmics" ja que està a la base de "l'ús ecosistèmic" final. En aquest context el capítol 2 se centrà en l'estudi de serveis ecosistèmics de suport produïts per les comunitats bentòniques (p.ex. reciclatge de nutrients o estabilitat del sediment) i, el capítol 3 analitzà com els canvis en l'ecosistema bentònic poden afectar la població d'una espècie comercial i per extensió les captures de peix, és a dir, el principal servei ecosistèmic directe que aporten els caladors de pesca.

Protegir els hàbitats bentònics és molt important per a mantenir els estocs pesquers (Caddy 2013). A més a més de l'impacte directe provocat per les captures, les espècies comercials també es poden veure indirectament afectades per la pesca a causa de l'alteració dels ecosistemes bentònics provocats per aquesta activitat. Aquest fet és especialment important per a les espècies demersals que estan estretament relacionades amb el bentos i en depenen per a completar el seu cicle biològic (Auster & Langton 1998, Kaiser et al. 1999). El capítol 3 d'aquesta tesi estudia aquest efecte indirecte de la pesca d'arrossegament sobre la població de moll de fang, evidenciant que els canvis en la funcionalitat de l'ecosistema bentònic provocats per l'art de ròssec podrien tenir un efecte global negatiu sobre la població d'aquesta espècie. Aquest estudi representa una primera aproximació a la problemàtica basat en un calador molt concret que presenta una alta intensitat de pesca. Per tal d'afinar més en l'avaluació d'aquests efectes indirectes es podria realitzar un estudi comparatiu de l'estat de dues poblacions diferents de moll de fang en dues àrees que presentin un nivell d'esforç molt diferent i que presumiblement presentarien també comunitats bentòniques d'estructura diferent.

La conservació dels hàbitats bentònics per a les espècies demersals inclou també el manteniment de les funcions ecosistèmiques produïdes pels ecosistemes bentònics. Així doncs, per tal d'avaluar la provisió d'aquestes funcions en els caladors de pesca, s'utilitzà el concepte de "Proveïdors de Serveis Ecosistèmics", és a dir, unitats ecològiques que realitzen determinats processos funcionals (Ecosystem Service Providers- ESPs). Aquests ESPs mostraren una resposta significativa a l'esforç pesquer, la qual cosa els ratifica com a bons candidats per a l'avaluació dels impactes de la pesca d'arrossegament sobre les funcions produïdes pels ecosistemes bentònics . Però també les característiques del sediment mostraren un efecte significatiu sobre els ESPs, la qual cosa dificulta l'avaluació d'aquest impacte. A més a més, no tots els hàbitats estaven representats equitativament en les mostres: hi havia més mostres dels hàbitats fangosos (més homogenis) que dels hàbitats sorrencs i de maërl. Per tant, en un futur anàlisi caldria incloure més mostres d'aquests últims. A més a més, com que el llindar de canvi d'estructura de les comunitats bentòniques és possible que estigui situat a nivells baixos de pesca (Bremner et al. 2003, de Juan et al. 2009), també caldria que l'estudi considerés més àrees que representessin un bon gradient d'esforç a nivells baixos.

Per tal d'evitar l'efecte de l'hàbitat en els estudis d'impacte pesquer alguns treballs inclouen només àrees amb el mateix tipus de sediment o recorre a mètodes estadístics per a eliminar-ne la influència (Craeymeersh et al. 2000, Jennings et al. 2001, Collie et al. 2005, Tillin et al. 2006, Mangano et al. 2014). Però és important incloure la variabilitat natural d'hàbitat en aquests estudis, ja que aquesta variabilitat es dóna dins mateix dels caladors pesquers que han de ser gestionats com un tot, no per trossos d'hàbitats homogenis. Una possible manera d'abordar aquesta qüestió podria ser a través d'un metanàlisi d'estudis d'impacte de pesca realitzats al Mediterrani, donant especial importància als habitats més heterogenis com els de maërl, però la majoria d'estudis existents d'aquest tipus estan fets en fons homogenis de sorra o fang (Smith et al. 2000, Tudela 2004, de

Juan et al. 2009, 2013, Mangano et al. 2013, però vegeu Barberá et al. 2012).

Malgrat l'efecte en certa manera distorsionant de l'hàbitat, els estudis aquí presentats mostren que les funcions de segrest de carboni i acoblament bento-pelàgic són vulnerables a la pesca d'arrossegament en les nostres àrees d'estudi. A més a més, combinant els resultats dels dos estudis del capítol 2, es va observar que les funcions produïdes per ESPs formats per trets vulnerables al bou (estabilitat del sediment, estructura d'hàbitat i acoblament bento-pelàgic), presentaren una redundància més baixa a nivells més elevats d'esforç pesquer. Per exemple, el tret "filtradors", que forma part d'ESPs vulnerables responsables de l'acoblament bento-pelàgic, presentà una baixa redundància en les àrees d'estudi amb major impacte pesquer. Aquest fet podria comprometre el proveïment d'aquestes funcions en àrees fortament impactades, alterant el funcionament dels seus ecosistemes bentònics. Per exemple, una reducció de les funcions d'estabilitat del sediment i de l'acoblament bentopelàgic podria desequilibrar el flux de nutrients, alliberant-ne massa i fins i tot podent alliberar contaminants (Snelgrove 1997, Chauvaud et al. 2000, Snelgrove et al. 2014). A més a més aquest fenomen es podria veure ampliat per la pròpia activitat disruptiva dels sediments de l'art de ròssec (Palanques et al. 2001).

Així doncs, la redundància és una propietat important dels ecosistemes que podria evitar la pèrdua d'una funció en un sistema exposat a l'impacte pesquer. En aquest context, és important identificar les àrees que presenten trets amb una baixa redundància així com ESPs vulnerables i gestionar-hi acuradament les activitats humanes que s'hi duguin a terme (Douvere 2008, Ottersen et al. 2011). La conservació d'aquestes àrees permetrà el manteniment de serveis ecosistèmics indirectes (Rees et al. 2012) que en darrer terme estan a la base dels serveis ecosistèmics directament avaluables econòmicament com les captures pesqueres.

2. DISSENYANT D'UN PLA DE GESTIÓ BASANT EN L'ECOSISTEMA: LA NECESSITAT DE COMUNICACIÓ ENTRE ELS ACTORS INVOLUCRATS EN LES PESQUERIES

Un cop avaluades les respostes de les funcions ecosistèmiques del bentos a la pesca, així com els serveis ecosistèmics relacionats, cal incorporar aquest coneixement als plans de gestió (Daily et al. 2009). Per a realitzar aquest pas, cal que tots els actors involucrats en les pesqueries entenguin la importància que tenen aquestes funcions ecosistèmiques en el global del sistema pesquer. Per tant es necessiten eines adequades de suport a la gestió per a una correcta comunicació entre actors.

En aquest context, aquesta tesi presenta dues eines per a fomentar una gestió participativa de les pesqueries: un model conceptual segons la filosofia del DPSIR (Drivers – Pressures – State Change – Impact– Response: Activadors – Pressions - Canvi d'estat – Impacte – Resposta) i una plataforma d'integració de coneixement. El DPSIR consisteix en la representació en un model conceptual dels principals elements del sistema així com les relacions entre ells, identificant-ne els Activadors, les Pressions, els Canvis d'Estat, els Impactes i les Respostes (Atkins et al. 2011, Mangi et al. 2007, Ojeda-Martínez et al. 2012). La plataforma de coneixement informa concretament sobre una d'aquestes relacions, la més ignorada en els plans de gestió: el canvi d'estat causat per l'art de ròssec en les comunitats bentòniques.

Activadors:

En aquesta tesi es va identificar la societat com un important element *Activador* del sistema en el procés per a aconseguir un EAF. En aquest context la societat actuaria com un grup de pressió, p.ex. demanant informació sobre la procedència de les captures abans de decidir si compren el producte (consum responsable) i reclamant una gestió sostenible de les pesqueries.

Pressions:

La descripció detallada dels elements de *Pressió* constitueix la base per a poder avaluar el *Canvi d'Estat* de les comunitats bentòniques a causat per aquesta *Pressió*. El capítol 1 tracta aquesta qüestió descrivint les activitats de la flota d'arrossegament, és a dir, la dinàmica i l'esforç pesquer.

Canvi d'estat de les comunitats bentòniques i Impacte sobre els serveis ecosistèmics:

El *Canvi d'Estat* de les comunitats bentòniques provocats del bou pot produir un *Impacte* sobre la societat a causa d'un desequilibri en els serveis ecosistèmics proveïts per aquestes comunitats. El coneixement de la resposta de les comunitats bentòniques a la pesca d'arrossegament pot ajudar a definir llindars de canvi més enllà dels quals la comunitat ja no és capaç de produir certs serveis (Folke et al. 2004, Thrush et al. 2009, Levin & Möllmann 2015). Els organismes de gestió han d'actuar abans que es doni aquest canvi i per tant s'han de definir nivells de referència o indicadors de l'estat de les comunitats bentòniques (de Juan & Demestre 2012). En aquest context, en el capítol 2 es destaca la vulnerabilitat d'algunes funcions clau en els fons tous dels caladors: provisió d'hàbitat, acoblament bento-pelàgic i estabilitat del sediment. A més a més s'hi suggereixen dues mesures per a avaluar la redundància funcional de la comunitat bentònica que potencialment podrien utilitzar-se com a indicadors per establir nivells d'esforç acceptables. Aquestes mesures ja han demostrat que compleixen alguns dels requisits per a ser utilitzades com a indicadors (p.ex. responen a l'estrès en àrees situades en una àmplia extensió geogràfica (Salas et al. 2006, Reza & Abdullah 2011)), però encara caldrien més estudis per a acabar de definir altres característiques. Per exemple, com es combinaria la informació complementària obtinguda amb les dues mesures en un indicador global de la redundància funcional de la comunitat.

Canvi d'estat de les comunitats bentòniques i Impacte sobre el servei de "provisió d'aliment":

El *Canvi d'Estat* de les comunitats bentòniques pot afectar indirectament les poblacions de les espècies objectiu a causa de l'alteració del seu Hàbitat Essencial, provocant un *Impacte* en el servei ecosistèmic de “provisió d’aliment”. El capítol 3 estudià aquesta qüestió ressaltant la rellevància dels ecosistemes de fons tou com a Hàbitats Essencials del moll de fang. Així doncs, els capítols 2 i 3 accentuen la importància del paper dels ecosistemes bentònics en el context de les pesqueries d’arrossegament i, per tant, la rellevància de tenir-los en compte en els plans de gestió d’aquesta activitat.

Resposta:

Per tal que els gestors prenguin consciència de la importància de considerar els ecosistemes bentònics en la gestió de les pesqueries, el paper d’aquests ecosistemes en el marc de la pesca d’arrossegament s’ha d’explicar amb claredat. Amb aquest objectiu, el capítol 4 reuneix la informació de l’estat funcional de les comunitats bentòniques d’epifauna de les àrees d’estudi en una “plataforma de coneixement”, mostrant els efectes de la pesca d’arrossegament en aquestes comunitats d’una manera visual, atractiva i entenedora. A més a més, la plataforma inclou un bloc de simulació que permet visualitzar els canvis potencials de l’estat de les comunitats bentòniques provocats per un canvi en l’esforç pesquer. La importància d’aquestes comunitats és considerada en el capítol 5, en el que es recomanen mesures per a una gestió integral de les pesqueries d’arrossegament dirigides a mantenir l’estabilitat dels sistemes bentònics de manera que sustentin una pesca sostenible. A més a més en aquest capítol es recalca que la implicació de tots els actors és clau per tal de poder impulsar una *Resposta* dels organismes de gestió encaminada a la implantació d'un EAF.

La plataforma de coneixement presentada en el capítol 4 és una de les eines recomanades en el capítol 5 per a la implementació d'un EAF a la costa catalana. No obstant això, aquesta plataforma de coneixement encara està en les seves primeres fases de desenvolupament. En futures millores, s'hauria de desenvolupar una connexió amb el programa R que recalculés automàticament les lleis de regressió de la simulació quan s'hi afegís una nova àrea d'estudi, d'aquesta manera s'incrementaria la robustesa de la simulació. A més a més també hauria d'incloure dades sobre l'impacte del bou sobre la infauna, ja que és el compartiment bentònic de què depenen moltes espècies objectiu per a alimentar-se (Smith et al. 2013, Johnson et al. 2015). Finalment, i considerant els resultats del capítol 3 com a base, la simulació també hauria d'incloure els efectes que el canvi en la comunitat bentònica podria provocar en les espècies comercials. D'aquesta manera la plataforma constituiria una eina d'avaluació més completa per a la gestió de les pesqueries.

Tot i la importància del hàbitats i les comunitats bentòniques per a les espècies objectiu i per la provisió de serveis ecosistèmics, les mesures de protecció ecosistèmica són encara el punt flac en els plans de gestió pesquers (Gårdmark et al. 2011, Caddy 2013, Snelgrove et al. 2014). Per tant, eines com la plataforma de coneixement i concretament el bloc de simulació, són necessàries per tal que tots els actors involucrats en les pesqueries entenguin el paper dels ecosistemes bentònics en les pesqueries i fomentar-ne la seva inclusió en els plans de gestió. Les eines de simulació són molt útils per a l'avaluació de diferents escenaris de gestió (Tallis et al. 2010) i, com que la gestió de les pesqueries en el Mediterrani està fonamentalment basada en la regulació de l'esforç, la plataforma es podria utilitzar per a aquest propòsit. De fet, com s'ha comentat prèviament, el capítol 5 ja inclou aquesta plataforma en les seves recomanacions per a la gestió de la flota d'arrossegament catalana. Aquest pla, basat principalment en mesures de reducció de l'esforç i de les captures per tal d'evitar la degradació dels ecosistemes bentònics, podria ser difícil d'aplicar si no s'és capaç de fer entendre als pescadors que aquestes mesures són necessàries per garantir la viabilitat dels estocs pesquers a llarg termini. A més a més, el seguiment del pla a llarg termini és costós (més si es té en compte l'avaluació dels ecosistemes bentònics), i per tant, per a la seva implementació caldria la voluntat política d'invertir-hi. Les mesures recomanades es podrien implantar per fases, començant per exemple pels ports del sud de Catalunya i, si el resultat és bo, ampliar-lo cap als ports més septentrionals.

3. DIVULGACIÓ CIENTÍFICA

a) Divulgació per als actors involucrats en les pesqueries

Aquesta tesi, però, s'ha centrat sobretot en els aspectes ecològics, i les esmentades eines de comunicació no han estat presentades a la comunitat no científica. Per tant, el següent pas seria mostrar aquestes eines a la comunitat no científica per tal que les valorin. Per a això es podrien organitzar tallers amb els actors involucrats en les pesqueries i, per exemple, proporcionar-los un qüestionari perquè quantifiquessin segons la seva opinió la importància de cada element del DPSIR en el global del sistema. També es podria utilitzar la plataforma de coneixement com una eina de suport a la deliberació en el context d'un Anàlisi de Decisió Multi-Criteri (Multi-Criteria Decision Analysis - MCDA) (Kiker et al. 2005). Aquest tipus d'anàlisis són molt flexibles i adaptables a molts tipus de situacions i s'utilitzen per avaluar diferents alternatives de gestió en sistemes en què hi ha molts actors involucrats amb diferents interessos (Brown et al. 2001, Kiker et al. 2005, Janssen et al. 2014). En el nostre cas, per exemple, es podrien avaluar diferents tipus d'escenaris de gestió basats en algunes o totes les mesures proposades en el capítol 5 i el bloc de simulació de la plataforma de coneixement es podria utilitzar per a visualitzar l'estat de la comunitat bentònica en cada escenari.

b) Divulgació per al públic en general

També seria interessant realitzar un qüestionari destinat al públic en general per tal de conèixer la seva percepció sobre els serveis ecosistèmics proveïts per les comunitats bentòniques (p.ex. valors culturals, el paper dels ecosistemes bentònics en les pesqueries i en els sistemes marins en general) (Himes 2007, Hutchison et al. 2013, de Juan et al. 2015). Aquest qüestionari serviria per a avaluar la relació ecosistema-societat, que en última instància determinaria el paper de la societat com a Activadors. Així mateix, el qüestionari també serviria per a determinar si la comunicació científica reïx en conscienciar sobre l'important paper de les comunitats bentòniques en els ecosistemes marins.

La divulgació científica és essencial per a conscienciar la població de la importància de mantenir els ecosistemes en un bon estat de conservació i per desenvolupar un sentiment de responsabilitat i predisposició a la seva conservació (Fanini et al. 2007, Sheavly & Register 2007, Demestre & Masó 2012). Per tant, cal desenvolupar activitats de divulgació com ara tallers que mostrin com canvia l'estat de les comunitats bentòniques i les seves propietats (p.ex. capacitat de provisió de funcions) quan són sotmeses a diferents esforços de pesca.

La preocupació pels impactes de la pesca sobre els ecosistemes marins està inclosa dins la nova llei europea de política pesquera (EU 1380/2013), la qual cosa implica l'existència d'una consciència política sobre la importància dels ecosistemes marins. Al mateix temps la societat també està actuant com un grup de pressió decantant-se pel consum de productes procedents de la pesca sostenible (https://www.msc.org/). Tot i així, el debat al voltant de la implementació d'aquestes polítiques pesqueres encara gira força al voltant de les espècies objectiu (p.ex. http://www.ecologistasenaccion.es/article30480.html), ja que aquest aspecte està directament lligat a assumptes econòmics i és difícil superar la disjuntiva entre interessos socio-econòmics i conservacionistes (Kempf 2010, Salomon et al. 2011). A més a més, el coneixement científic a vegades és limitant a l'hora d'aplicar una gestió segons l'EAF (Coll et al. 2013, Sartor et al. 2014). Amb l'objectiu d'ampliar aquest coneixement, aquesta tesi presenta una metodologia per a l'avaluació de les funcions ecosistèmiques en els caladors de pesca (els ESPs i les mesures de redundància) i una plataforma de simulació que mostra els canvis potencials en les comunitats bentòniques provocats per l'art de ròssec. A més a més, proposa com es podria implementar un pla de gestió en el context de l'EAF en el marc d'una pesqueria de la qual se'n té molta informació.

CONCUSIONS GENERALS

● La informació provinent d'entrevistes amb els pescadors i dels informes de les subhastes permet la identificació de la situació dels caladors de pesca i ajuda a determinar la distribució espacio-temporal dels recursos. També és útil per a detectar àrees que presentin diferent nivell d'esforç pesquer.

● Les dades del sistema de control de vaixells per satèl·lit representen una bona font independent d'informació per a descriure la dinàmica de la flota a gran escala. Tot i que és massa poc precisa per a estimar l'esforç a nivell d'ecosistema bentònic, permet delimitar la situació de caladors de pesca.

● El sonar de rastreig lateral proporciona una imatge real de l'impacte de l'art de ròssec en el fons marí. Per tant, la informació obtinguda amb el sonar de rastreig lateral proporciona una estimació d'esforç molt acurada a nivell bentònic.

● Basant-nos en les conclusions anteriors, s'aconsella la utilització conjunta de les tres aproximacions en el marc de la gestió ecosistèmica de les pesqueries, ja que proporcionen informació sobre la caracterització de les estratègies de pesca dels vaixells, la distribució espacio-temporal de les espècies objectiu i la millor estimació de l'esforç a petita escala.

● Els proveïdors de serveis ecosistèmics, grups d'organismes basats en els seus trets biològics i la seva relació amb funcions ecosistèmiques, representen una bona aproximació per a avaluar la resposta funcional de les comunitats bentòniques a la pesca d'arrossegament.

● La pesca d'arrossegament afecta significativament els proveïdors de serveis ecosistèmics, tot i que aquesta resposta és difícil de definir a causa de l'efecte de la variabilitat d'hàbitat. El tipus de sediment condiciona la presència de determinats proveïdors de serveis ecosistèmics i aleshores la pesca actua sobre aquesta variabilitat natural. Per tant, l'abundància relativa de cada proveïdor de servei ecosistèmic varia en relació a l'esforç i a les característiques del sediment.

● L'estudi d'una gamma de llocs d'estudi sotmesos a diferents nivells d'esforç permet detectar patrons generals en la resposta de les comunitats bentòniques a la pesca d'arrossegament. No obstant això, per tal de distingir els efectes de la pesca i del sediment, l'estudi hauria d'incloure més mostres de sediments heterogenis que de sediments homogenis.

● La mesura escollida per a l'estimació dels proveïdors de serveis ecosistèmics, la biomassa o l'abunància d'epifauna, influeix en l'estimació de la contribució del proveïdor de servei ecosistèmic a la funció ecosistèmica. Per tant, és important avaluar cada funció per separat i determinar quina és la millor mesura a utilitzaren cada cas.

● La redundància funcional d'infauna i epifauna s'hauria de mesurar considerant els dos aspectes, l'abundància del tret (Trets Comuns) i la riquesa del tret (Trets Generalitzats), ja que les dues mesures donen informació complementària. La utilització de només una de les mesures pot portar a una percepció enganyosa de la redundància funcional global de la comunitat bentònica i, per extensió, de la seva resiliència.

● Les espècies rares de la infauna i l'epifauna no augmenten significativament la riquesa de trets bentònics en fons crònicament impactats per la pesca. Per tant, les mesures de redundància es poden estimar utilitzant solament les espècies més abundants, d'aquesta manera se'n simplifica el càlcul.

● Els proveïdors de serveis ecosistèmics formats per trets vulnerables i que produeixen funcions com l'estabilitat del sediment, estructura d'hàbitat i acoblament bentopelàgic, mostraren una redundància més baixa a nivells més elevats d'esforç pesquer. Per tant, aquestes funcions es poden veure compromeses en zones sotmeses a elevats esforços pesquers en les nostres àrees d'estudi.

● Canvis en els components funcionals de les comunitats bentòniques que poden tenir efectes negatius en els Hàbitats Essencials del moll de fang juntament amb l'elevada pressió pesquera sobre els reclutes d'aquesta espècie, podrien tenir un efecte global negatiu en l'estoc de moll de fang.

● En aquesta tesi es considera que la societat pot jugar un paper potencialment important per a impulsar una Resposta dels organismes de gestió per avançar cap a la implementació d'una gestió basada en l'ecosistema. Per tant, en el model conceptual del DPSIR la societat és considerada com un element Activador del sistema.

● La plataforma de coneixement té com a objectiu compartir el coneixement sobre l'impacte de la pesca d'arrossegament en els ecosistemes bentònics en el context d'una gestió participativa. Aquesta plataforma mostra d'una manera entenedora, visual i atractiva l'efecte de l'art de ròssec sobre l'estructura de les comunitats bentòniques de les nostres àrees d'estudi.

● El bloc de simulació inclòs en la plataforma de coneixement mostra la resposta potencial de la funcionalitat bentònica a un canvi en la intensitat d'esforç pesquer. Aquesta simulació podria ser utilitzada com a eina de suport a la deliberació per a avaluar les possibles conseqüències en els ecosistemes bentònics de diferents escenaris de gestió.

● Basant-nos en la dinàmica observada de les pesqueries i els efectes del bou en les comunitats bentòniques i les espècies comercials, es presenta un pla de gestió integrat en el context de la gestió basada en l'ecosistema. Aquest pla basat en un cas d'estudi concret, té com a objectiu mantenir la integritat dels fons marins de manera que sustentin una pesca sostenible.

● El pla de gestió proposat para especial atenció en la recomanació de mesures enfocades a la conservació dels ecosistemes bentònics, ja que aquest és normalment l'element més ignorat en els plans de gestió pesquers. Per tant, cal que en el seguiment del pla s'utilitzin indicadors de l'estat de l'ecosistema bentònic a més a més dels tradicionals indicadors de l'estat dels estocs de les espècies objectiu.

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APPENDICES

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Performance of a northwestern Mediterranean bottom trawl fleet: How the integration of landings and VMS data can contribute to the implementation of ecosystem-based fisheries management

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ABSTRACT

The European Union has established a framework to achieve or maintain good environmental status in the marine environment by 2020. The Marine Strategy Framework Directive requires the application of the ecosystem approach to the management of human activities, covering all sectors having an impact on the marine environment. However, fisheries in the Mediterranean are far from a systematic implementation of an ecosystem-based fisheries management (EBFM). Aiming to address this issue, this study explores the potential of the relationship between daily yield by vessel (landings and income by species) and vessel position (known via vessel monitoring system) as a tool for fleet management. This approach is possible due to the current dynamics of Mediterranean fleets, with vessels returning daily to the harbour where landings are registered as weight and income by vessel. Moreover, vessels of >15 m total length have been compulsory monitored by VMS since 2005. A bottom trawl fleet that operates in the northwestern Mediterranean was chosen to develop this approach. Different groups of trawlers were identified, which could be linked to the strategies displayed by the fishermen that were mainly driven by the target species dynamics. Accurate knowledge of the fishing targets driving the fleet dynamics and of the fishing strategies at the vessel level (i.e. fishing ground habitat where the fishing pressure is exerted and corresponding landings) are shown to be a feasible tool for fleet management.

1. Introduction

The Reykjavík Declaration, adopted at the 2001 Reykjavık Conference on Responsible Fisheries in the Marine Ecosystems, included the commitment "to advance the scientific basis for developing and implementing management strategies that incorporate ecosystem considerations and which will ensure sustainable yields while conserving stocks and maintaining the integrity of ecosystems and habitats on which they depend". One of its main tasks is to translate the generic and conceptual Ecosystem-Based Fisheries Management (EBFM) framework (i.e. extension of the conventional principles for sustainable fisheries development to cover the ecosystem as a whole), into an operational framework at regional, national, or local scales (e.g. ecosystem or fishery levels [1,2]). To date, practical application of this approach by competent regional organisations for fisheries and for the marine and coastal environment is still in early development stages. It is worth mentioning the pioneering initiatives undertaken in Australia (e.g. [3,4]).

In consonance with the EBFM conceptual framework, the European Union (EU) has established a framework to achieve or maintain good environmental status (GES) in the marine environment by 2020. To this aim, criteria and methodological standards to identify or achieve a GES of marine waters have been defined. These include, among others, descriptors at the species, habitat and ecosystem levels, as well as descriptors related to the fishing activity (EC Directive 2008/56, Marine Strategy Framework Directive (MSFD; [5])). The activity of the EU fishing fleets is governed by numerous regulations based on the Common Fisheries Policy, which at present is under revision. In any case, environmental concerns must be integrated in the fisheries management, in line with the MSFD. Regarding the implementation of EBFM in the Mediterranean, the General Fisheries Commission for the Mediterranean (GFCM), composed of EU and non-EU signatory countries, launched in 2012 a revision process aimed at modernising its legal and institutional framework. This revision includes among its objectives the promotion of the ecosystem-based approach for the conservation of the marine environment and the sustainable use of marine living resources [6].

Bottom trawling is the fishing activity with the highest impact on marine ecosystems [7], hence, in order to achieve a GES, it is

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important to properly regulate this activity. The management of bottom trawling in the EU Mediterranean waters is carried out by fishing effort control, through (i) control of the fleet activity and technical measures; and (ii) closures, which refer to permanent spatial closures (fishing forbidden at < 50 m and > 1000 m depth) and temporal closures. The control of the activity includes a daily limit of hours at sea. Technical measures include minimum landing size for the main target species, square- meshed net of 40 mm at the cod- end, and control of the fishing capacity (Council Regulation (EC) no. 1967/2006; Recommendation GFCM/29/2005/1). Temporal closures are not homogenously implemented and, when applied, its duration differ among areas, from 1–2 months in some Spanish Mediterranean areas, 1.5 months in Italian waters, to 4 months in Greece [8]. It is worth mentioning that Total Allowable Catch (TAC) only applies to bluefin tuna. In areas other than the Mediterranean where EU fleets operate, bottom trawling is based on the exploitation of a single main fishing resource, and the landed catch is the result of fishing during one trip, which usually lasts several days or even weeks. Otherwise, despite bottom trawling in the Mediterranean can also be based on the exploitation of a single resource, it typically targets several species, their relative importance in the landings changing over the year. In general, trawling is carried out near the port base, five days a week, and the catch is freshly commercialised, daily, upon the arrival of trawlers to the port base.

MSFD proposes qualitative descriptors (e.g. biological diversity, sea floor integrity or exploitation of populations within safe biological limits) of the environmental status of marine ecosystems. Some of these descriptors have been monitored for a long time and are currently impacted by fishing [9]. However, fisheries in the Mediterranean are far from a systematic implementation of an EBFM and many sensitive ecosystems (including essential fish habitats) are threatened by fishing activities [10]. The complex geopolitical situation of the Mediterranean basin, i.e. many costal states and a large proportion of international waters, implies coordinated management actions that are necessary to ensure the protection of ecosystems, but these are difficult to implement [11]. Nevertheless, despite these difficulties, the characteristics of the fishing fleet control and distribution in the Mediterranean (i.e. clear delimitations for operation of different port-based fleets) is a potential bridge towards the implementation of EBFM.

It is generally accepted that EBFM must incorporate a spatial dimension to ensure the sustainability of resources and ecosystem integrity [12,13]. In the Mediterranean this could be partly achieved by establishing maximum effort levels on a spatial extent, including temporal or permanent closures to protect Sensitive Habitats. Moreover, technical measures could be adopted to minimise impact on ecosystems [14]. But in order to achieve this, we first need to know the spatial and temporal dynamics of the fishing fleet that will help identifying the priority areas/seasons for management. In this regard, the Vessel Monitoring System (VMS) is a promising tool for the fisheries management. Since 2005 fishing vessels exceeding 15 m of overall length must be equipped with VMS [15]. The potential of VMS for fleet management has been explored in North Atlantic fisheries, by combining logbook data with VMS signals to investigate the spatial distribution of catches and effort. However, the fishing fleet dynamics in this area imply landings corresponding to several fishing days and therefore the relationship between catches and vessel position is not straightforward (e.g. [16–19]). In the Mediterranean, the daily return of trawlers to port facilitates a direct link between vessel position and catches, however, VMS data have been barely analysed aiming to evaluate its potential as a tool for the management of the fishing activity (e.g. [20]).

Aiming to advance in the implementation of EBFM in the Mediterranean and integrate current fishing practices with management strategies, the relationship between daily yield by vessel (landings and income) and vessel position has been explored to assess its potential as a tool for fleet management. A bottom trawl fleet that operates in the northwestern Mediterranean was chosen to develop this approach.

2. Material and methods

2.1. Study area

The northwestern Mediterranean study area, off Cape of Creus and Bay of Roses, corresponds to the fishing grounds where the trawl fleet based on the fishing port of Roses operates (around 3500 km^2; Fig. 1). The Cape of Creus continental shelf is characterised by an abrupt morphology and alternates smooth areas, where sandy and muddy sediments prevail, with rough rocky areas, mainly along the coast and in the outer shelf for depths between 95 and 130 m. Moreover, the Cape of Creus canyon breaches the shelf at a depth of 110 m [21]. The shelf-slope limit, in front of the Bay of Roses, is located at 170 m depth [22]. The fishing grounds encompass the rather narrow continental shelf, the upper slope and the submarine canyon. This heterogeneous area was chosen with a view to unveil potential different fishing strategies. The study area includes the coastal protected areas of Natural Park of Cape of Creus and Medes Islands Marine Reserve (Fig. 1).

2.2. Data on landings, income and fishing fleet

The trawl fleet from Roses consisted of 22 vessels, most of them of an overall length > 20 m, when the study was carried out in 2009 (Table 1). Trawlers return daily from Monday to Friday to Roses harbour, where the catch is sold in the local auction. A maximum of 12 h per day at sea is permitted. Bottom trawling licence is to be used all year round, that is, fishermen cannot practice any other type of fishing.

Data on landings, income and activity of the trawl fleet were obtained from the daily sales slips obtained from the sale at the auction that takes place upon the arrival of the vessels at port (data source: fishing statistics elaborated by the Fisheries Department of the Generalitat de Catalunya). Data were available on daily landings by species, in weight and in income, for each fishing vessel during 2009. No contemporary data on discards were available. Data on the trawl fleet (gross tonnage (GT) and overall length in m (LOA)) were obtained from the fishing vessels census of the Spanish Ministry of Agriculture, Food and Environment.

2.3. Fleet structure based on landings and income

Cluster analysis and factor analysis approaches were used to obtain homogeneous groups of vessels at the annual scale based on the 2009 daily landings by vessel (catch and income by species). No prior assumptions of the fishery were made, avoiding the inherent subjectivity of qualitative analysis. Firstly, the specific composition of the catch (weight and income) was transformed into a landings profile (relative species composition) and data were log-transformed. Secondly, cluster analysis was applied to the log-transformed landings profile matrices using Ward's minimum variance clustering algorithm and the Euclidean distance. Species with annual landings < 100 kg and annual income from the sales at the auction < 600 euros were excluded because of their low relevance in the overall landings and income. The number of species retained for the analyses was 73. Thirdly, factor analysis was applied to further explore the structure of the relationship among vessels. Communalities were estimated as multiple R-square and only factors with eigenvalues > 1 were retained. The varimax rotation was applied to the factor solution.

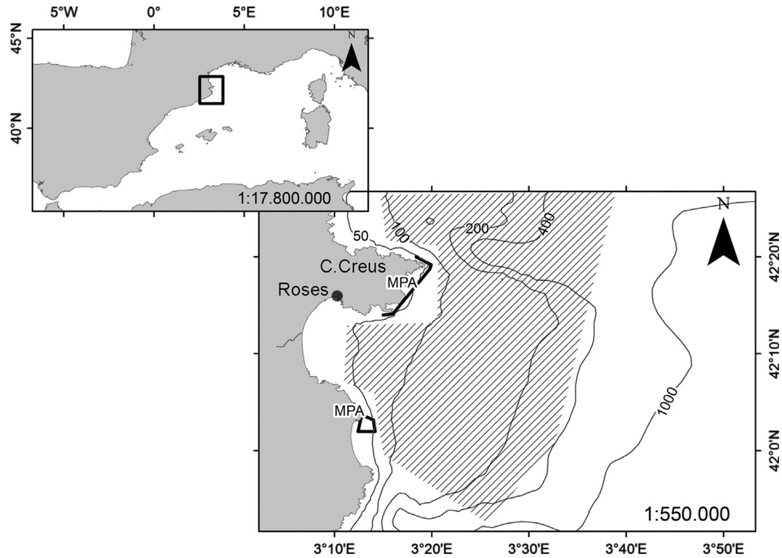

Fig. 1. Study area in the northwestern Mediterranean. The striped area encompasses the fishing grounds where the bottom trawl fleet from Roses operates. The limits of the MPAs Cape of Creus (in the north) and Medes Islands (in the south) and the 50, 100, 200, 400 and 1000 m isobaths are also shown.

Table 1
Gross tonnage (GT) and overall length (LOA in m) of the bottom trawlers based in Roses in 2009.

Vessels	GT	LOA
1	105.6	28.5
2	24.2	14.3
3	54.4	22.4
4	34.6	17.5
5	60.5	22.9
6	21.5	15.5
7	65.2	22.9
8	93.7	25.1
9	56.7	21.6
10	61.1	23.3
11	58.9	20.4
12	65.2	23.1
13	87.4	24.9
14	61.6	24.9
15	178.5	26.5
16	74.4	24.5
17	100.1	25.0
18	77.1	22.5
19	96.7	25.4
20	100.0	27.4
21	97.1	26.3
22	96.5	27.1

This study selected factor loadings over 0.7 to evaluate the relationship between the selected factors and the vessels. Analyses were done with the statistical package Statistica6.

2.4. Spatial distribution of fishing effort

Finally, daily position records of the vessels obtained from the VMS (Vessel Monitoring System) were available (data source: General Secretariat for Fisheries of the Spanish Ministry of Agriculture, Food and Environment) which allowed linking fishing catches with the actual corresponding fishing position. Unprocessed VMS data does not indicate whether a vessel is fishing, so a speed filter was applied to raw data. Only records from vessels with fishing activity (i.e. speed up to 4 knots; [23]) were taken into account and these positions were plotted using ArcGis 10.0 package to represent the spatial distribution of the fishing activity. Each plot represents the fishing positions of one vessel in 2009.

2.5. Identification of outliers

Data on daily landings by species per vessel were explored aiming to detect outlying observations, that is, values that deviate markedly from others. Outliers can indicate either an error or a very high daily landing or income for a given species in a certain time of the year. These high daily landings or income were our target. An example is presented to illustrate how daily landings, in combination with vessel position, are a potential tool for the monitoring of the fleet. To this aim monthly box-plots for the daily landings and income of one trawler (vessel no.7 in Table 1) were done and the identified outliers were also shown.

3. Results

3.1. Annual landings and income

In 2009, the trawl fleet from Roses landed around 1560 t of fish in a total of 4945 fishing days, which generated around 8.2 million euros from the sale at the auction. The activity of the fleet was highest in summer (July and August), and in January and March, with around 450 fishing days per month, and lowest in February (345 fishing days).

A total of 105 species were commercialised. Nevertheless, 94% of the landings in weight and in income was obtained with 22 and 21 species respectively (weight and income of each of these species $>0.5\%$ of the annual total). The specific composition of the landings expressed by weight or by income provides different patterns of the fleet yield (Table 2). European hake (*Merluccius merluccius*) was the main species in terms of landings, 324.5 t, followed by horse mackerel (*Trachurus* spp.), blue whiting (*Micromesistius poutassou*) and curled-horned octopus (*Eledone cirrhosa*). Otherwise, the landings ranked by income clearly identified the main target species. Blue and red shrimp (*Aristeus antennatus*) was by far the species that generated the highest income (2.3 million euros). The other species that also generated high income were

Table 2
Fishing port of Roses 2009: bottom trawl annual landings, expressed in tones (a) and in thousands of euros (b). Species with landings and income >0.5% of the total are detailed.

(a) Landings (tones)		%	(b) Income (thousands euros)		%
Merluccius merluccius	324.5	20.8	*Aristeus antennatus*	2326.2	28.3
Trachurus spp.	240.9	15.5	*Merluccius merluccius*	1477.8	18.0
Micromesistius poutassou	177.5	11.4	*Nephrops norvegicus*	868.0	10.6
Eledone cirrhosa	170.4	10.9	*Lophius spp.*	631.1	7.7
Aristeus antennatus	71.5	4.6	*Eledone cirrhosa*	400.8	4.9
Lophius spp.	66.2	4.2	*Loligo vulgaris*	335.2	4.1
Nephrops norvegicus	54.3	3.5	*Micromesistius poutassou*	321.3	3.9
Mullus barbatus	43.5	2.8	*Mullus barbatus*	264.0	3.2
Illex coindetii	42.8	2.7	*Lepidorhombus boscii*	193.5	2.4
Phycis blennoides	40.3	2.6	*Trachurus spp.*	181.3	2.2
Trisopterus minutus	35.0	2.2	*Phycis blennoides*	111.0	1.4
Lepidorhombus boscii	30.0	1.9	*Zeus faber*	90.2	1.1
Pagellus acarne	27.9	1.8	*Illex coindetii*	83.3	1.0
Sardina pilchardus	23.3	1.5	*Trisopterus minutus*	67.7	0.8
Loligo vulgaris	21.7	1.4	*Scomber scombrus*	59.2	0.7
Octopus vulgaris	16.7	1.1	*Sepiidae. Sepiolidae*	55.8	0.7
Triglidae	16.5	1.1	*Octopus vulgaris*	54.9	0.7
Scomber scombrus	14.6	0.9	*Pagellus erythrinus*	51.9	0.6
Engraulis encrasicolus	13.7	0.9	*Stichopus regalis*	51.2	0.6
Trigla lyra	13.6	0.9	*Trigla lucerna*	46.9	0.6
Pagellus erythrinus	12.8	0.8	*Pagellus acarne*	46.2	0.6
Conger conger	9.1	0.6	Other	495.2	6.0
Other	92.2	5.9			
Total	1559.0	100	Total	8212.5	100

hake, Norway lobster (*Nephrops norvegicus*), monkfish (this category includes *Lophius budegassa* and *Lophius piscatorius*), curled-horned octopus, squid (*Loligo vulgaris*) and red mullet (*Mullus barbatus*). Horse mackerel and blue whiting did not generate high income and were clearly behind other species highly appreciated, that despite their lower catches were ahead in the ranking based on income. Hence, in this case these two species should be considered as by- catch species. It is worth mentioning the landings (0.7 t) of the royal cucumber (*Stichopus regalis*), which corresponds to its five longitudinal muscular bands that are sold as a culinary delicacy, and that expressed in total fresh weight of the organism would amount to around 7 t [24]. This is the species with the highest price at the auction (>70 euros/kg), more than twice the mean price for blue and red shrimp.

3.2. Fleet structure based on landings and income

Cluster analysis from both the landings and income profiles, by species per vessel, identified three homogeneous groups within the fleet (Fig. 2). The groups from the two ordination approaches were quite similar, although the ordination within group C was slightly different, when expressed in terms of landings or income. The first dichotomy separated the same group of seven vessels (Fig. 2; vessel nos. 2,3,5,7, 8, 12 and 19; codes as in Table 1; group A). Within the second dichotomy, a small group of four vessels (no. 10, 16, 20, and 21; group B) was differentiated from the remaining vessels (vessels 1, 4, 6, 9, 11, 13, 14, 15, 17, 18 and 22; group C).

The landings profiles by vessel, arranged according to the cluster analysis, ease the identification of the main species characterising each vessel group. When expressed as income, landings profiles evidenced the main target species, those driving the fishing activity within each group of vessels (Table 3). Hake and horse mackerel, the species with highest landings (Table 2), were well represented in the landings by weight of the whole fleet (Table 3). The importance of hake differed among groups despite it remained as one of the species representing high income. The relative importance of blue whiting landings was high in group C, while it was small in group B and almost nil in group A, and did not represent high income for neither group. Curled-horned octopus represented >10% of the annual landings of vessels from groups A, B and C, but this species represented high income only for some trawlers of group A. Within this group, other species that provided high income were monkfish, squid and red mullet. Blue and red shrimp was the most important species by weight in group B, its importance small in group C and nil in group A. By far, the highest income corresponded to blue and red shrimp and group B vessels. Within group C, a small group of three vessels (no. 4, 9, 17) had Norway lobster as the main species in terms of income (see also Fig. 2b, these vessels appear as a sub-group of group C); another sub-group of five vessels (no. 13, 14, 15, 18, 22; Table 3) had hake as the main target species in terms of landings and income. Relatively low blue and red shrimp landings represented also high income for some trawlers of group C.

Factor analysis results based on the landings profiles by weight and by income were similar, as the same main groups were retained. The factors loadings for each vessel, which indicate the vessels with the highest correlation with each of the three retained factors, are given in Table 4. The first three factors explained >90% of the total variance. With the application of the varimax rotation to the factor analysis solutions, variance was redistributed among the three factors (40%, 32% and 21% for Factors 1, 2, and 3 for the analysis based on the landings; and 39%, 29% and 25% in the case of income). Based on the factor loadings >0.7, the vessels linked to Factors 1, 2 and 3 were identified. For both landings profiles by weight and income, factors loadings >0.7 were for Group of vessels C and Factor 1; for Group A and Factor 2; and for Group B and Factor 3, with the exception of vessel no. 8, for which factor loadings were <0.7 for the three factors, with landings profiles as income. For the interpretation of factors, the two-factors rotated solutions are also shown for the landings by weight (Fig. 3).

3.3. Spatial distribution of fishing effort

VMS data showed the overall dynamics of the fleet in 2009. One vessel within each one of the three identified vessels' groups (A, B and C) was selected to show their preferential fishing grounds (Fig. 4). The group of vessels linked to Factor 2 (Group

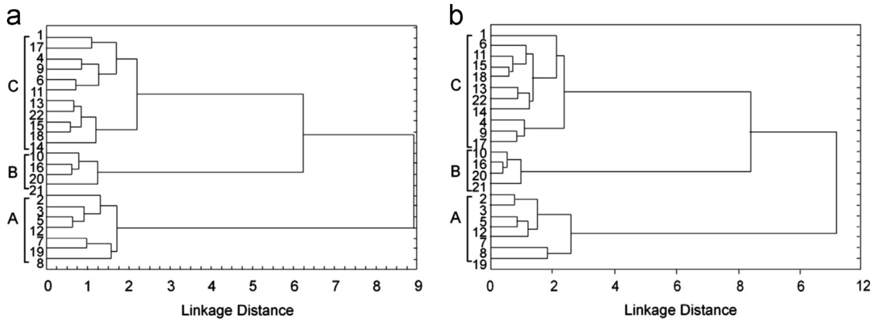

Fig. 2. Cluster analysis identified three groups of vessels (A, B and C) from the bottom trawl fleet from Roses in 2009. Analyses were applied to the log-transformed landings profiles matrices using Ward's minimum variance clustering algorithm and the Euclidean distance; clustering based on landings profiles in weight (a), and income (b). Vessel number as in Table 1.

Table 3
Bottom trawl fleet from the fishing port of Roses: landings profiles (relative species composition) by vessel in 2009, expressed by weight (a) and by income from the sales at the auction (b). Profiles arranged according to the cluster analysis results shown in Fig. 2; vessel number as Table 1.

Vessel number	(a) Profiles by weight																					
	1	17	4	9	6	11	13	22	15	18	14	10	16	20	21	2	3	5	12	7	19	8
Merluccius merluccius	14.7	12.1	17.4	14.1	13.3	17.5	25.1	29.3	22.4	30.2	20.9	21.7	18.5	16.1	15.8	15.9	23.9	26.4	17.8	15.9	19.5	33.7
Trachurus spp.	14.4	20.0	3.4	6.1	10.9	11.0	13.7	14.9	11.5	8.8	21.2	8.8	12.2	13.9	2.3	10.3	10.7	13.1	21.1	21.9	26.3	23.0
Micromesistius poutassou	31.5	30.4	13.3	21.3	9.1	20.8	19.1	11.7	16.1	13.6	13.4	1.1	2.5	4.1	1.4	0.1	0.3	1.1	2.1	0.3	2.2	1.5
Eledone cirrhosa	6.2	2.8	10.2	5.9	10.9	7.7	10.1	10.6	9.8	11.5	6.6	12.2	10.1	11.3	5.6	35.9	27.9	17.6	18.2	12.7	6.3	11.3
Aristeus antennatus	0.1	2.5	1.8	2.6	4.4	3.4	3.1	6.2	2.3	1.7	0.8	29.9	33.3	24.2	56.7	0.0	0.0	0.0	0.0	0.2	0.2	1.8
Lophius spp.	7.5	4.3	6.7	4.8	9.1	6.1	3.2	3.4	5.0	4.7	3.8	2.5	2.9	4.5	3.3	3.7	6.4	3.4	3.1	2.0	2.0	2.7
Nephrops norvegicus	5.4	12.2	14.5	15.1	5.0	5.5	2.1	2.4	6.2	4.7	2.4	0.0	0.2	0.2	0.1	0.0	0.0	0.1	0.0	0.1	1.4	0.1
Mullus barbatus	1.6	0.5	1.9	3.2	1.9	1.5	1.1	0.7	2.1	1.2	1.3	4.3	2.4	2.1	0.6	5.0	3.2	5.6	5.3	8.2	6.3	1.7
Illex coindetii	3.1	2.1	2.2	3.2	3.7	2.9	2.6	3.0	3.8	3.7	3.6	2.2	1.9	1.5	0.8	2.5	3.9	3.4	2.8	1.2	2.0	2.0
Phycis blennoides	2.3	3.6	7.1	7.1	7.1	6.4	2.4	2.9	4.1	3.5	2.4	3.6	3.1	2.9	3.0	0.0	0.1	0.6	0.1	0.2	0.8	0.6
Trisopterus minutus	2.8	0.4	1.6	0.6	2.3	2.4	2.4	2.6	2.7	2.8	2.8	0.9	2.0	2.6	1.1	2.7	2.1	3.4	2.6	1.5	2.0	2.6
Lepidorhombus boscii	2.3	2.5	3.9	2.7	3.0	2.4	2.0	1.5	2.8	2.1	2.2	0.3	0.8	0.9	0.3	3.9	2.3	1.5	1.9	0.9	1.1	1.4
Pagellus acarne	0.1	0.1	0.4	1.1	1.7	0.8	1.4	0.1	0.3	0.1	2.2	0.8	0.3	1.2	0.1	1.2	2.0	3.0	5.8	9.9	2.4	0.4
Sardina pilchardus	0.3	0.0	0.8	0.3	0.1	0.6	0.7	0.8	0.8	2.0	2.3	0.8	0.8	0.6	0.4	0.2	1.8	2.2	2.1	2.3	3.5	4.4
Loligo vulgaris	0.3	0.2	2.9	0.9	0.8	0.5	0.8	0.5	0.4	0.8	0.6	1.5	0.5	0.7	0.2	1.9	3.0	2.9	2.7	4.2	3.5	1.2
Octopus vulgaris	0.3	0.2	2.2	0.8	1.4	0.5	0.8	0.4	0.3	0.5	0.8	0.7	0.4	1.0	0.3	2.2	2.2	2.1	2.4	2.2	1.9	0.9
Triglidae	0.6	0.1	0.4	0.7	1.1	0.7	1.1	0.5	1.7	0.6	1.5	0.5	0.7	1.0	0.1	6.5	1.6	1.3	1.7	1.3	1.4	0.4
Scomber scombrus	0.5	0.1	0.2	0.2	0.5	0.5	1.1	2.0	0.9	1.0	1.0	1.2	1.3	0.8	0.7	0.4	1.1	1.5	1.0	0.8	1.5	1.2
Engraulis encrasicolus	0.1	0.0	0.0	0.1	0.0	0.3	1.3	0.5	0.3	0.8	0.6	0.5	0.2	0.4	0.5	0.1	0.8	0.9	0.7	1.3	2.1	4.5
Trigla lyra	0.8	0.4	1.6	1.2	4.4	2.7	1.1	0.7	0.8	0.8	0.9	0.1	0.5	0.6	0.0	0.1	0.4	0.8	0.6	0.4	0.4	0.3
Pagellus erythrinus	0.0	0.0	0.5	1.1	0.2	0.2	0.3	0.1	0.2	0.3	0.2	0.7	0.3	0.4	0.0	0.6	0.7	2.2	1.4	3.6	3.1	0.4
Conger conger	0.1	0.1	0.5	0.3	1.4	1.0	0.4	0.9	0.7	0.1	1.4	0.5	0.5	0.8	0.7	0.8	0.1	1.1	0.3	0.7	0.7	0.1
other	5.0	5.3	6.3	6.7	7.8	4.6	4.2	4.5	4.7	4.4	6.9	5.2	4.8	8.5	5.8	6.0	5.5	6.0	6.4	8.1	9.5	3.8

Vessel number	(b) Profiles by income																					
	1	6	11	15	18	13	22	14	4	9	17	10	16	20	21	2	3	5	12	7	8	19
Aristeus antennatus	0.8	25.8	22.3	14.7	12.9	22.5	41.6	5.5	9.8	12.0	13.5	79.6	82.1	75.3	92.3	0.0	0.0	0.0	0.0	0.9	15.2	1.6
Merluccius merluccius	15.0	10.0	15.6	21.8	29.8	25.2	23.5	27.8	13.0	10.4	11.7	7.1	6.1	7.3	3.4	19.3	26.5	29.2	21.7	19.8	38.1	24.3
Nephrops norvegicus	21.8	15.2	16.8	19.0	15.8	6.2	5.4	11.7	35.7	42.1	40.3	0.0	0.2	0.2	0.0	0.0	0.0	0.4	0.0	0.4	0.4	6.3
Lophius spp.	15.7	13.3	10.0	9.8	9.0	7.5	5.1	10.4	8.2	6.5	6.9	1.9	2.5	3.5	1.3	13.3	14.2	11.0	10.9	6.5	8.8	5.0
Eledone cirrhosa	2.2	2.9	2.8	3.6	4.3	4.7	3.3	3.5	3.4	1.9	0.9	2.9	1.4	2.5	0.3	30.1	19.3	9.4	14.7	12.6	5.5	3.3
Loligo vulgaris	1.0	1.9	1.4	1.4	2.3	2.5	1.2	2.9	7.5	1.9	0.7	1.8	0.6	0.9	0.2	8.5	11.2	10.9	10.8	16.0	4.8	14.2
Micromesistius poutassou	14.5	2.6	8.5	5.2	5.0	8.4	3.4	7.3	4.1	6.3	10.4	0.2	0.4	0.8	0.1	0.1	0.2	0.5	1.1	0.1	0.8	0.9
Mullus barbatus	2.8	2.7	2.0	2.4	1.6	1.7	0.6	2.0	2.0	2.4	0.8	1.7	0.9	1.3	0.1	6.2	5.3	8.4	9.1	13.1	2.0	9.6
Lepidorhombus boscii	4.0	3.5	3.2	4.5	2.9	2.9	1.5	3.9	5.0	3.2	3.6	0.1	0.4	0.5	0.1	3.8	3.0	2.1	3.1	1.2	2.2	1.8
Trachurus spp.	2.3	1.4	1.5	1.8	1.4	2.6	2.4	4.9	0.4	0.7	2.5	0.6	0.8	1.0	0.1	2.1	1.8	2.8	3.9	4.0	5.0	4.4
Phycis blennoides	1.4	3.0	3.3	2.5	2.1	1.6	1.5	2.3	3.3	3.0	2.1	0.7	0.7	0.8	0.4	0.0	0.1	0.4	0.1	0.1	0.5	0.5
Zeus faber	4.8	1.6	1.9	1.7	0.8	1.9	0.7	0.7	0.1	0.1	0.7	0.1	0.2	0.6	0.1	0.7	0.8	1.8	1.7	0.7	1.2	0.7
Illex coindetii	1.5	1.1	1.1	1.6	1.4	1.1	1.2	1.9	0.8	1.1	0.9	0.3	0.3	0.3	0.1	1.4	1.7	1.6	1.4	0.7	1.0	1.0
Trisopterus minutus	1.5	0.7	1.0	1.0	1.2	1.2	0.8	1.5	0.5	0.2	0.2	0.1	0.3	0.5	0.1	1.2	0.9	1.4	1.2	0.7	1.3	1.1
Octopus vulgaris	0.3	0.9	0.4	0.3	0.4	0.6	0.3	0.6	1.0	0.3	0.1	0.2	0.1	0.2	0.1	1.5	1.6	1.8	1.9	1.8	0.7	1.9
Sepiidae, Sepiolidae	0.7	1.1	0.7	0.4	0.5	0.7	0.4	0.6	0.0	0.0	0.2	0.3	0.4	0.4	0.0	1.6	2.3	1.8	2.7	1.1	0.6	0.5
Scomber scombrus	0.4	0.3	0.4	1.0	0.9	1.0	1.2	1.2	0.1	0.2	0.1	0.4	0.5	0.3	0.1	0.4	1.1	1.2	1.2	0.9	1.2	1.6
Pagellus erythrinus	0.1	0.1	0.2	0.4	0.7	0.4	0.2	0.4	0.2	0.4	0.0	0.1	0.2	0.1	0.0	0.4	0.5	2.6	0.8	2.8	0.9	3.0
Stichopus regalis	4.1	3.5	1.2	0.3	1.2	0.5	0.0	0.1	0.0	0.0	0.0	0.0	0.2	0.2	0.0	0.1	0.8	0.1	0.9	0.0	0.5	0.1
Pagellus acarne	0.0	0.9	0.6	0.2	0.1	1.1	0.0	0.5	0.1	0.2	0.1	0.1	0.1	0.3	0.0	0.3	1.0	1.3	2.7	2.8	0.1	0.8
Other	5.0	7.3	5.2	6.3	5.7	5.8	5.3	10.2	4.5	6.8	4.2	1.9	1.6	3.1	1.1	9.2	7.8	11.2	10.1	14.0	9.0	17.0

Table 4
Factor analysis results: factor loadings for the landings (a) and income (b); vessel number as in Table 1.

	(a) Landings				(b) Income		
Vessel	Factor 1	Factor 2	Factor 3	Vessel	Factor 1	Factor 2	Factor 3
1	0.9	0.4	0.2	1	0.9	0.3	0.0
2	0.3	0.8	0.2	2	0.2	0.9	0.1
3	0.3	0.9	0.3	3	0.2	0.9	0.1
4	0.8	0.3	0.3	4	0.9	0.3	0.3
5	0.4	0.9	0.3	5	0.3	0.9	0.1
6	0.8	0.3	0.4	6	0.8	0.3	0.4
7	0.2	0.9	0.2	7	0.2	0.9	0.2
8	0.4	0.7	0.4	8	0.4	0.6	0.6
9	0.9	0.3	0.3	9	0.9	0.2	0.3
10	0.3	0.4	0.8	10	0.3	0.3	0.9
11	0.9	0.3	0.4	11	0.9	0.2	0.4
12	0.4	0.9	0.2	12	0.2	0.9	0.1
13	0.7	0.5	0.4	13	0.7	0.4	0.5
14	0.7	0.6	0.3	14	0.8	0.5	0.3
15	0.8	0.4	0.4	15	0.8	0.3	0.4
16	0.4	0.4	0.8	16	0.4	0.2	0.9
17	0.9	0.2	0.3	17	0.9	0.1	0.3
18	0.8	0.4	0.4	18	0.8	0.4	0.4
19	0.4	0.8	0.2	19	0.5	0.7	0.2
20	0.5	0.4	0.8	20	0.4	0.2	0.9
21	0.3	0.1	0.9	21	0.3	0.0	1.0
22	0.7	0.4	0.5	22	0.7	0.3	0.6

A) corresponded to coastal trawlers that operate very close to the coast, at depths between 50 and 100 m; their activity at > 100 m depth was limited (Fig. 4a). The group of vessels linked to Factor3 (Group B) corresponded to the trawlers targeting blue and red shrimp. As depicted by the fishing positions, the fishing grounds are the submarine canyons. Trawlers targeting *Aristeus antennatus* generally perform the final haul of the day in their way back to the port, and this would correspond to those hauls between 50 and 100 m depth (Fig. 4b). Finally, 11 vessels out of the 22 that made up the fleet appeared more closely linked to Factor 1 (Group C). This is a heterogeneous group (see the landings profiles, Table 3) that included trawlers with Norway lobster or hake as main target, and trawlers that additionally targeted other species as blue and red shrimp or monkfish. The preferential fishing grounds for trawlers targeting Norway lobster are located in the slope, around the 400 m isobath, between 350 and 450 m (Fig. 4c). Some of the vessels from this group C, whose activity was driven by several target species, displayed a "generalist" fishing strategy by shifting between species during the year. Generalists trawlers fished in coastal grounds, over the shelf and, also, on the submarine canyons (Fig. 4d). The generalist strategy of some vessels, i.e. those that do not display a clear affinity to any of the three factors, is more evident in the factor loadings by income (Table 4b). A typical "generalist vessel" would be no. 8 (Fig. 4d), but vessels no. 13, 19 and 22 could also be considered as generalists.

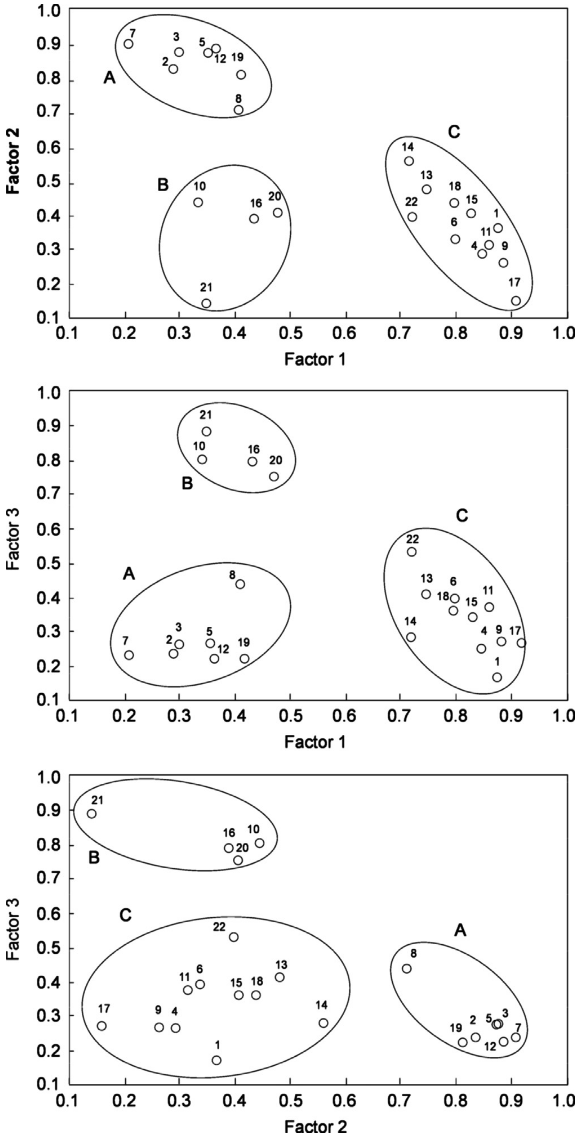

Fig. 3. Factor analysis of the landings profiles in weight from the bottom trawl fleet from Roses in 2009. The plots show the position of the vessels based on the factor loadings of the first three factors. The trawlers belonging to the same vessels' group, as identified in the cluster analysis, are shown encircled; vessel number as in Table 1.

3.4. Identification of outliers

As an example of the potential for the daily landings to be used as a tool for fleet management, the landings from a vessel from group A, coastal trawlers, were presented during 2009 (vessel no.7; Fig. 5). Outliers are shown in relation to the corresponding monthly average of the daily catch and income. Thus, for instance, in three days over the year (17 July, 3 August and 7 September) the daily landings of this vessel were much higher than the monthly average (> 1400 kg/day). On the 17th of July this trawler obtained its maximum daily income of the year (> 5000 euros). Since data were structured by vessel and by species, the species that generated this outstanding income could be identified and in this particular case turned out to be red mullet. Given that the position of the trawler can be identified from the VMS data, the detected outliers on the 17th of July (Fig. 5) could be linked with the position of the vessel (Fig. 4a). On the other two outlying dates for landings, the high catch was generated by *Pagellus acarne* (1400 and 1200 kg on 3 August and 7 September), which did not correspond with the income peaks in August and September. Evaluating the income outliers, the income from the 11th of August (4000 euros) and 22nd of September (3500 euros) outstand as they were not detected in the catches. These outliers were explained, in addition to red mullet, by the income of a number of species, including *Pagellus acarne, Loligo vulgaris* and *Pagellus erythrinus*.

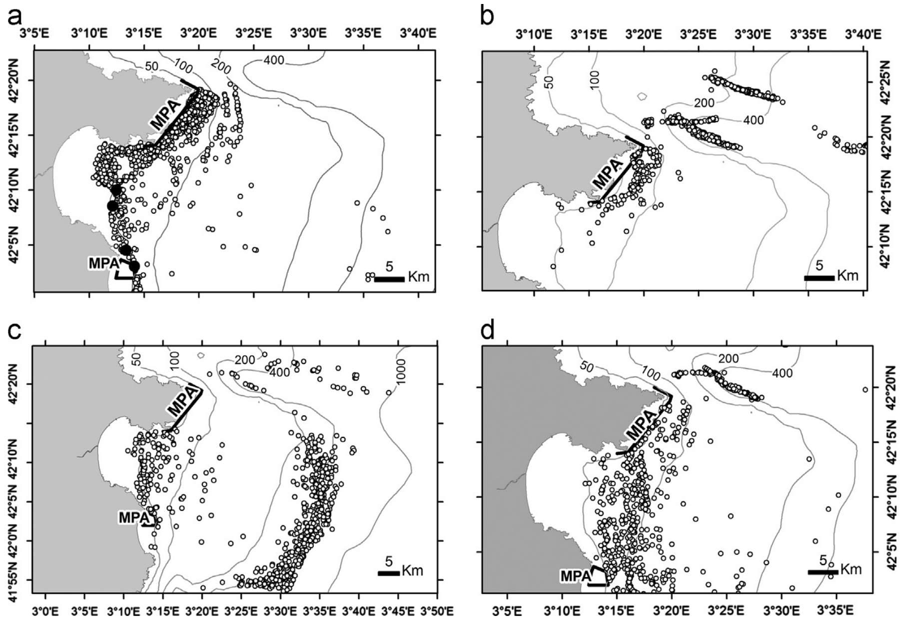

Fig. 4. Fishing strategies identified in the bottom trawl fleet from Roses in 2009 through clustering and factor analysis (vessel number as in Table 1). Examples are shown of trawlers operating in the coastal zone (a; in bold, positions corresponding 17 July, explanation in the text on the identification of landings and income outliers); targeting blue and red shrimp *Aristeus antennatus* (b); targeting Norway lobster *Nephrops norvegicus* (c); and of generalist behaviour (d).

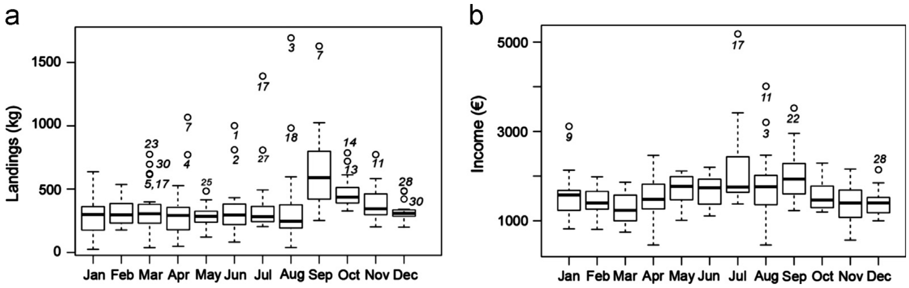

Fig. 5. Example of the identification of outliers from the daily landings data by vessel: landings data by weight (kg) and income (euros) of trawler no. 7 during 2009. The boxplots include median, 25 and 75 percentiles and maximum and minimum values each month over 2009. The numbers correspond to the date of the outliers.

4. Discussion

This study identified different groups of trawlers within a bottom trawl fleet that operates in the northwestern Mediterranean, which can be linked to the strategies displayed by the fishermen. The analyses were based on the daily landings profile for each vessel, based on both catch weight and income; however, the species with the highest contribution to the annual income are assumed to drive the fishermen's decision on the choice for the fishing ground.

Results identified three groups of vessels, which can be interpreted as coastal trawlers (A), trawlers operating in the submarine canyons (group B), and trawlers fishing in the shelf break and slope (group C). The fleet structure resulting from the factor analysis and the corresponding factor loadings (Table 4 and Figs. 3 and 4) suggested that Factor 1 could be linked to the fishing grounds over the deepest shelf and on the shelf break. The trawlers with highest correlation with Factor 1 (group C) included those with Norway lobster as main target, that is, those that operate in the fishing grounds at further distance from the coast. The morphology of the study area determines that at short distances from the coast a wide depth range is encountered. Factor 3, for which only the trawlers targeting blue and red shrimp (group B) had factor loadings >0.7, could be interpreted as a factor depending on depth, since submarine canyons are the deepest fishing ground. Coastal trawlers (group A) showed the strongest link to Factor 2, which could be associated to the shelf that in the study area is rather narrow (Fig. 1).

This interpretation is in accordance to the type of substrate in the study area, which in turn is decisive for the spatial distribution of the main target species. The submarine canyons in the Mediterranean are the habitat for blue and red shrimp [25,26], a highly valued commercial species. Along its life cycle, hake inhabits the shelf and the shelf break zones throughout the western Mediterranean [27]. Norway lobster is found in the muddy bottoms over the upper and middle continental slope [28,29]. And coastal areas in the region harbour heterogeneous substrata [21] that are habitat to a variety of demersal fish, represented in the catch composition. The tight relationship between the fishing strategy and the selection of fishing grounds by the concentration of target species denotes the importance of identifying Essential Fish Habitats that might be vulnerable to fishing. Indeed, Bergmann et al. [30] suggested that the fishermen knowledge on the location and distribution of EFH is highly valuable information for management.

The selection of these habitats by the fishermen can be partly explained by the presence of three species whose income represented >10% of the annual total, and therefore are to be considered as main drivers of the fishing activity. These are blue and red shrimp, European hake and Norway lobster (Table 3). The income of the species that followed in the ranking (i.e. curled-horned octopus, squid, red mullet) may seem low at the annual scale, but this is misleading because their landings displayed a marked seasonality and most of the annual income was concentrated in a short period. The monitoring of these economically important species is essential in the context of EBFM, as policy makers need to quantify the goods and services provided by an ecosystem in order to evaluate the impact of management decisions [31]. Importantly, the main life history traits to be taken into account in fisheries management, such as spawning areas and season, recruitment area or preferential feeding habitat, are well known for these species in the study area (e.g. [29,32–42]).

Fishermen develop dynamic fishing tactics and strategies as an adaptive response to changes in resource abundance, environmental conditions and market or regulatory constraints. From the way of coping with these factors, the behaviour of fishermen can be defined as specialist or generalist. The knowledge of these dynamics is essential for effective management [43,44]. VMS data (precise trawler position) combined with data on daily landings allowed identifying the fishing strategies of each of the identified trawlers' group and, also, the fishing strategy of the individual vessel. Specialists concentrated in one area or one target species. In the study area, this typically corresponds to the fishermen targeting blue and red shrimp. But the concept of specialists would also include coastal trawlers and shelf-break-slope trawlers, with Norway lobster as main target. This is because the fishing grounds where most trawlers operate remain fairly constant over the year. Generalists would be those trawlers deploying their activity in non-clearly defined fishing grounds, which can be located from the coast to the shelf-break, and even the submarine canyons (Fig. 4d). The knowledge of the fishermen strategies as for the preferred fishing grounds and targets in a certain area or time is basic for the definition of spatial or temporal closures, aimed at the protection of a given fishing resource or habitat.

European and national policy commitments require further integration of fisheries and environmental management [14]. Therefore, data on the fleet dynamics needs to be complemented with indices of the ecosystem status [45]. Knowledge on the distribution of fishing activities should be followed by the characterisation of the fished habitats [46] and the evaluation of these habitats under the concept of GES [47]. An example of this process was presented in de Juan and Demestre [48] that tested a Trawl Disturbance Indicator in several habitats linked to trawling grounds, including the coastal fishing grounds near the Cape of Creus that were assigned a moderate environmental status, and soft-bottoms encompassed in Medes Islands MPA, that had high status. This ecological indicator should be complemented with other indicators that overall evaluate the ecological integrity of the area, which might include, among others, target species distribution, population size (population abundance and/or biomass), level of pressure of the fishing activity, proportion of selected species at the top of the food webs (large fish, by weight). Nevertheless, these assessments should consider different scales for fishery dynamics and ecosystem conservation, as often the scales of fishing activities estimations are too large to link with ecosystem impact at the habitat scale [7]. In this context, the smaller-scale distribution of Mediterranean fleets is useful to identify links between fishing activity and environmental disturbance.

Fleet-based assessment is the pathway for implementation of efficient EBFM in European Seas [49]. Fishing intensities calculated from VMS data have proven to be reliable indicators of fishing impacts for several habitats ([47] refs. therein). Results on the preferential fishing grounds by vessel or group of vessels have been presented here at the annual scale. Nevertheless, the areas where fishing effort and target species concentrate could also be identified at a seasonal basis, from daily VMS and landings data. Fleet dynamics and landings trends along the year, provide precise information for the spatial distribution of fishing effort and distribution and abundance of the target species, at the local scale and by vessel, thereby contributing to the implementation of the MSFD. As mentioned before, this information can be used as input for indicators of GES.

Daily landings outliers in combination with vessel position can become a useful tool for the control of the fleet activity. The assumption is that, if a daily landing or income is well-above the monthly average, the species explaining this outlier is different from the average catch, i.e. the vessel fished in grounds different from those where it deployed most of its monthly activity. This does not necessarily mean that the trawler has entered into non permitted grounds, but this outlier suggests checking which species is explaining the high landing/income and where it has been fished. The biology of the target species allows the interpretation of outliers. Thus, for instance, in the outlier example shown in Figs. 4a and 5, the species responsible for the sharp increase of daily landing was identified as red mullet while the coastal vessel position was known from VMS data. These results point to the start of the red mullet recruitment to the shallow coastal waters fishing grounds.

We propose that this combined information of vessel positions and landings, at the daily scale, has the potential to be used as a tool for fleet management. For instance, this information could provide options for the best choice regarding where and when spatial and temporal closures could be implemented, for the limitation of fishing hours per day, and even in the control of marine protected areas limits (for instance, by implementing a buffer area in case of high concentration of trawlers very close to the limits of the reserve).

In the study area different activities compete with bottom trawling for the use of the space (existence of two coastal marine reserves, small-scale and recreational fishing, and leisure- related activities). Temporal and spatial components need to be incorporated in EBFM in order to support current and future uses of marine ecosystems and maintain the delivery of valuable ecosystem goods and services for future generations [50]. The spatial zoning of the marine environment needs to be systematically implemented in the Mediterranean to protect sensitive areas [11,51,52]. To achieve this, detailed analysis of geo-referenced data from the fleet dynamics is essential in order to consider management scenarios [13,52].

5. Conclusions

Accurate knowledge of fishing targets driving the fleet dynamics along with knowledge on fishing strategies at the vessel level (i.e. fishing ground-habitat where the fishing pressure is exerted and corresponding landings) are a potential tool for fleet management. VMS has been shown to be a feasible tool for Mediterranean fleets management, as it helps linking landings data to their corresponding fishing ground. However, despite the use of VMS data for the management of the fleet and implementation of the EBFM is straightforward due to the existence of data on daily landings, this is an issue that needs further development. Overall, the complementation of these tools may contribute to achieve both fishing and environmental targets and, hence, to the EBFM implementation in the Mediterranean.

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Integrating the provision of ecosystem services and trawl fisheries for the management of the marine environment

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HIGHLIGHTS

- Ecosystem Service Providers (ESP) represented a link between function and services.
- ESP variability was mainly affected by sediment type and fishing effort.
- A DPSIR (Drivers–Pressures–State Change–Impact–Response) model is presented.
- The DPSIR aims to inform integrated management to ensure service provision.
- All socio-ecological system components are essential to adopt integrated management.

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ABSTRACT

The species interaction and their biological traits (BT) determine the function of benthic communities and, hence, the delivery of ecosystem services. Therefore, disturbance of benthic communities by trawling may compromise ecosystem service delivery, including fisheries' catches. In this work, we explore 1) the impact of trawling activities on benthic functional components (after the BTA approach) and 2) how trawling impact may affect the ecosystem services delivered by benthic communities. To this aim, we assessed the provision of ecosystem services by adopting the concept of Ecosystem Service Providers (ESP), i.e. ecological units that perform ecosystem functions that will ultimately deliver ecosystem services. We studied thirteen sites subjected to different levels of fishing effort in the Mediterranean. From a range of environmental variables included in the study, we found ESPs to be mainly affected by fishing effort and grain size. Our results suggested that habitat type has significant effects on the distribution of ESPs and this natural variability influences ESP response to trawling at a specific site. In order to summarize the complex relationships between human uses, ecosystem components and the demand for ecosystem services in trawling grounds, we adapted a DPSIR (Drivers–Pressures–State Change–Impact–Response) framework to the study area, emphasizing the role of society as Drivers of change and actors demanding management Responses. This integrative framework aims to inform managers about the interactions between all the elements involved in the management of trawling grounds, highlighting the need for an integrated approach in order to ensure ecosystem service provision.

1. Introduction

Marine ecosystems provide a wide range of services that are essential for human well-being. (Beaumont et al., 2007; Townsend et al., 2011); however, the marine environment has been historically impacted by human activities (Jackson et al., 2001). Nowadays, both scientists and managers recognize that, in order to ensure the provision of ecosystem services for future generations, the broad range of human activities in our coasts and oceans must be subjected to an appropriate management (Levin and Lubchenco, 2008). Among the multiple human uses of our coasts and oceans, fisheries is acknowledged as being the principal source of disturbance that has historically and chronically altered marine ecosystems (Thrush and Dayton, 2002). Indeed, fisheries catches are declining world-wide (Pontecorvo and Schrank, 2014), implying that management strategies have failed to maintain sustainable ecosystems and to ensure the provision of the "fish for food" service.

Trawl fishing impacts cause the chronic modification of the ecosystem functioning (Bremner et al., 2003; de Juan et al., 2007). The alteration of ecosystem functioning is of high relevance for society as the services delivered by benthic ecosystems are the product of many inter-related functions (Townsend et al., 2011; Rees et al., 2012; Snelgrove

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et al., 2014). Moreover, benthic community structure does not only regulate ecosystem function of the sea floor, but also water column processes, e.g. nutrient fluxes that are positively related to primary production in the water column (Lohrer et al., 2004). Trawling disturbance may compromise the provision of food by overexploitation of target communities and degradation of benthic habitats (Jackson et al., 2001; Thrush and Dayton, 2002). Additionally, trawling activities might indirectly compromise many ecosystem functions (e.g. bioturbation and nutrient cycling) that are related with regulation and supporting services (MEA, 2005; Beaumont et al., 2007) but that might ultimately be related with the provision of food (e.g. productivity of fishing grounds). Therefore, in the fisheries context, the whole array of functional components of the seafloor should be considered under an integrated management approach (Rees et al., 2012), with conservation targets that go beyond the food provisioning service.

In order to adopt an integrated management approach, it is necessary to understand community dynamics under stress. To this aim, a functional approach based on Biological Trait Analysis (BTA) has proved useful (Bremner, 2008). Variation in functional components due to stress is linked to changes in BT, which are in turn linked to changes in the species composition (Bremner, 2008). The BTA has also been proposed as a useful framework to assess the provision of ecosystem services by biological communities (Bolam and Eggleton, 2014; Bremner et al., 2006a; Frid, 2011). Ecosystem services depend on functions performed by benthic communities (e.g. deep burrowing fauna increase the oxygen flow into the sediment, extending the total denitrification zone and stimulating nutrient cycling that is involved in the ecosystem regulating service (Lohrer et al., 2004; Beaumont et al., 2007)). But the estimation of the provision of ecosystem services normally addresses economic or social values (Beaumont et al., 2007; Boyd and Banzhaf, 2007) and rarely focuses on ecological measures (Kremen and Ostfeld, 2005, but see Townsend et al., 2011; Snelgrove et al., 2014). In this context, the concept of Ecosystem Service Providers (ESP) was proposed (Kremen and Ostfeld, 2005; Cognetti and Maltagliati, 2010) aiming to encompass a group of organisms sharing particular biological traits (BT) that were related with certain ecosystem functions and, ultimately, with the provision of services. Altogether, these links would determine the dynamics of the ESP; therefore, understanding these links would increase our knowledge on the relationships between ecosystem dynamics and service delivery.

On the other hand, socio-ecological links must be accurately defined if we aim to achieve an integrated management approach and this requires appropriate methodological tools (Daily et al., 2009). The DPSIR framework (Drivers–Pressures–State change–Impact–Response) has been proposed as a systems-based approach that captures key relationships between society and the environment, note the tight link with the ecosystem service concept. Moreover, the DPSIR is regarded as a philosophical context for structuring and communicating policy-relevant research about the environment to non-scientists (Mangi et al., 2007; Atkins et al., 2011).

The principal aim of this study is to understand how commercial trawling compromises the delivery of ecosystem services, through changes in functional components of the benthic ecosystem. The specific goal is to assess the variability of ESPs under different environmental conditions and subjected to variable trawling intensity. We analyzed the functional components of benthic communities as surrogates for ecosystem functions from 13 sites located in fishing grounds in the Mediterranean. These sites were subjected to different levels of trawling effort. Variability in BT across sites could be linked to i) trawling activity ii) local environmental factors or iii) area location. Through this approach we discuss if these factors drive ecosystem function variability in our study area and how this could ultimately compromise the delivery of ecosystem services. Finally, management implications for the variability of ecosystem services are discussed in a socio-ecological context under the DPSIR framework, highlighting the importance of benthic ecosystem service conservation to sustain healthy fisheries.

2. Materials and methods

2.1. Study areas and site characterization

Six areas were selected in the Mediterranean Sea to carry out this study: 3 in the coast of Spain (Cap de Creus-CC, Ebre Delta-D and Cabo de Palos-CP), two in Italy (Ligurian Sea-L and Adriatic Sea-A) and one in Greece (Ionian Sea-I) (Fig. 1). Fishing effort was estimated in these areas based on information gathered from fishermen interviews and from registers of landings recorded by the fisheries associations. This allowed us to estimate fishing effort as GT*days at sea/month. Four of the study areas (CC, CP, I, L) were further divided in 2–3 sites that were identified as being subjected to different levels of fishing effort (Low or no effort, Medium and High) based on the fishermen information previously obtained (Fig. 1, Table 1). Sites with “Low”, “Medium” and “High” effort were defined within each area and are not comparable among areas (e.g., site H in CC was not comparable to site H in CP; these were relative values).

All the study sites had an area of 2–4 km^2 and showed homogeneous sediments and communities (Demestre et al., 2008; de Juan et al., 2013). The sites were located on soft bottoms in the continental shelves between 40 and 80 m, where commercial trawling is performed. The CC area included a no-take Marine Protected Area (CCL). The L, I and D areas had muddy bottoms (over 90% of mud); CC and A areas had sandy-mud bottoms; and the CP area was characterized by maërl protruding within sandy-mud bottoms (de Juan et al., 2013; Table 1).

Subsequently, during the experimental cruise, a different approach for fishing effort estimation was carried out using side-scan sonar surveys (SSS). Trawl doors alter the structure of the sediments, with scars that can be up to 20–30 cm depth. These scars on the sediment show high reflectivity tracks on the SSS images (Lucchetti et al., 2012). Thus, the SSS images allow the estimation of trawl mark density on the seafloor. Density was obtained considering the total length of the trawl marks per site area (trawl tracks density/km^2) (see de Juan et al., 2013 for further details) (Table 1). This approach was consistent in all study areas; therefore, the obtained effort values could be compared across sites and areas.

2.2. Data collection

All samples were collected in early summer (June–July) of 2003 for D and A and 2009 for the other areas. The sampling protocol was consistent across areas. The potential influence of this 6-year gap between sampling periods was tested with multivariate analysis of benthic community composition (using PRIMER 6 & PERMANOVA, see Section 2.4). A MDS analysis for the whole data set was done based on Bray–Curtis similarity index, and results showed no structure in relation with the sampling year. The distribution of samples was mainly influenced by environmental variability.

The epifaunal community was sampled with an epibenthic dredge that had a 2 m iron-frame aperture and a 1 cm cod-end. Six samples were randomly collected in each site, the dredge towed for 15′ with a constant speed of 2.3 knots. A SCANMAR device was attached to the dredge to ensure continuous contact with the seabed. Due to bad weather conditions, only three replicates were collected in CCL site and two in IL site. Epifaunal organisms were generally identified to species and the number of individuals and biomass for each species was recorded and standardized to 1000 m^2.

Three to five Van Veen grabs were randomly collected in each site. The surface sediments from grab samples were retained for the analysis of grain size and carbon content. Grain size was analyzed with laser diffraction (using Horiba La-950v2 particle size analyzer). Total carbon (TC) was analyzed using a TruSPec Leco analyzer. After removing the inorganic carbon from the sediment sample using HCL vapors, the sample was reanalyzed to obtain the total organic carbon (TOC). These data were not available for A study area (Table 1).

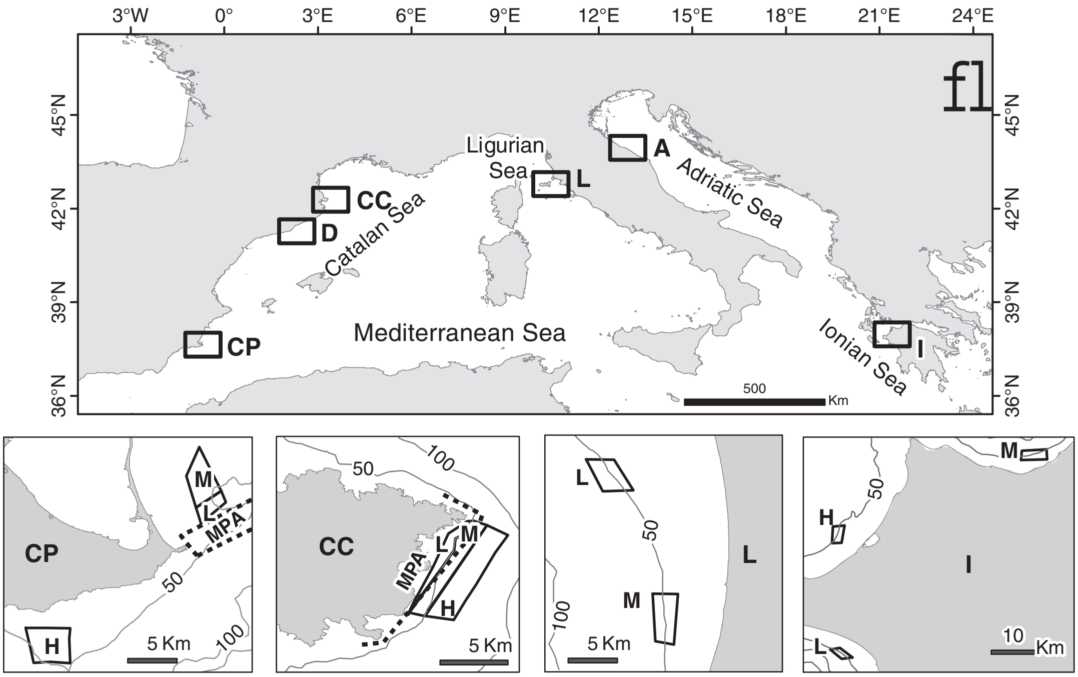

Fig. 1. Map of the study areas. The detailed maps below include the sites with different fishing effort within each area. MPAs are bounded by stripped lines in CC and CP areas. CP: Cabo Palos, D: Delta, CC: Cap Creus, L: Ligurian Sea, A: Adriatic Sea, I: Ionian Sea. L: low, M: medium and H: high effort.

The study sites were relatively small (2–4 km^2) and the sediment composition was highly homogeneous (Demestre et al., 2008; de Juan et al., 2013); also, the epibenthic dredge was towed for ca. 10, therefore, it homogenizes small-scale variability. With this sampling design, grab samples collected in the surroundings of the epibenthic dredge track were considered to be representative of the sampled habitat.

Near-bottom oxygen, temperature, chlorophyll a, photo-synthetically active radiation (PAR) and turbidity were available from CTD profiles in the study sites from CC, CP, L and I areas. Turbidity was also available for D (Table 1).

2.3. Biological trait analysis

Fourteen biological traits covering different aspects of the organisms' biology were selected and broken down into categories (Table 2) (Bremner et al., 2003; Muntadas et al., 2014).

Following representativeness criteria, 83 out of 289 epifaunal species were selected. The representativeness criteria included species recorded in at least: (a) five of the six replicas or (b) in at least three replicates and with more than three individuals in at least one of the three replicates or (c) three replicates and weighting more than 10 g in at least one of the three replicas.

Each of these 83 taxa was scored for its affinity to trait categories following the "fuzzy scoring" method using a scale of 0 to 3 (0 = no affinity to 3 = high affinity) (see Bremner et al., 2003 for further details). This assignment was based on available literature, experts' knowledge and information from the BIOTIC database (http://www.marlin.ac.uk/biotic/). When no information on a particular trait was available, information for closely related taxa (genera or family) was considered (Bolam and Eggleton, 2014). If no information was found, zero values were entered for each category and the taxon did not contribute to the calculation of trait weightings.

Table 1
Summary of the variables recorded in the study sites. Fishing effort according to two different estimations: (FE1) fishermen interviews (GT*days at sea/month) and (FE2) Side Scan Sonar (trawl tracks density/km^2); environmental data: gravel, sand and mud percentages, D50 (μm), total organic carbon (TOC), total carbon (TC), bottom turbidity (mg/l), bottom temperature (T°), bottom chlorophyll a (Chla-OBS), bottom oxygen (mg/l) and photo-synthetically active radiation (PAR).

Sites	FE1	FE2	%Gravel	%Sand	%Mud	D50	%TOC	%TC	Turbidity	T°	Chl a	Oxygen	PAR
CC-L	0	22,393	5.3	70.5	24.2	190.1	0.25	8.62	2.54	14.1	0.18	6.99	–
CC-M	1322	63,171	0	14.38	85.62	14.47	0.55	3.79	2.37	13.9	0.21	7.01	–
CC-H	1623	81,113	2.3	36.1	61.6	27.95	0.29	3.55	2.26	13.8	0.19	7.01	–
D	45,125	116,835	0	0.51	99.5	4.5	0.61	–	4	15.7	–	–	–
CP-L	900	48,248	2.4	65.1	32.5	251.2	0.2	8.92	28.68	15.2	0.06	7.7	16.59
CP-M	14,453	54,199	2.4	65.1	32.5	251.2	0.2	8.92	28.21	15.1	0.04	7.65	12.66
CP-H	21,966	137,243	0	32.46	67.54	33.1	0.87	5.95	33.95	14.7	0.04	7.75	0.015
L-L	6256	87,542	0	5.62	94.38	9.99	0.76	2.82	25.54	14.9	0.05	7.53	61.79
L-M	7930	96,954	0	7.72	92.28	12.42	0.54	2.84	41.57	14.9	0.06	7.6	12.00
A	21,823	32,278	0	63.83	36.17	76.64	–	–	–	–	–	–	–
I-L	6216	91,749	0	2.87	97.13	10.62	0.49	2.33	23.87	14.7	0.15	6.54	3.25
I-M	10,833	102,017	0	1.63	98.37	7.7	0.57	2.25	52.41	14.6	0.18	6.74	13.57
I-H	14,146	122,131	0	5.52	94.48	11.84	0.29	2.30	33.25	14.9	0.05	7.2	11.35

Table 2
Set of biological traits and categories used to describe epifaunal functional components. Underlined codes in brackets were used in Table 3. Traits and categories in bold were those removed in the reduced traits table (Table 2a) (see Materials and methods section).

Biological trait	Categories
Environmental position	**Epibenthic**
	Endobenthic
Habit	**Attached (at)**
	Bed forming
	Burrow dwelling
	Encrusted
	Erect (er)
	Free living
Growth form	Crustose soft
	Cushion
	Arborescent
	Vermiform
	Tubiculous
	Globose
	Turbinate (tr)
	Stellata
	Articulate
	Bivalved (bv)
	Pisciform
Mobility	Swimmer
	Crawler
	Burrow
	Attachment
Bioturbation	**Not relevant**
	Surface
	Subsurface
	Deep to surface
Feeding mode	Deposit feeder
	Filter/Suspension feeder
	Opportunistic/Scavenger
	Predator
	Grazer
Size	Small (<5 cm)
	Medium (5–10 cm)
	Large (>10 cm)
Fragility	**Fragile**
	Intermediate
	Robust
Regeneration potential	Yes
	No
Asexual reproduction	Yes
	No
Reproductive frequency	Continuous reproduction
	1 reproductive event per year
	2 or more reproductive events per year
Type of larvae	**Brooding/viviparous**
	Lecitotrophic
	Planktotrophic
Life span	<1 yr
	1–5 yr
	>5 yr
Sexual maturity	**<=1 yr**
	>1 yr

The abundance of the 14 traits' categories in each of the 13 sites was calculated by weighting the category scores by the abundance of each taxon exhibiting that category (Charvet et al., 1998). In the resulting sample by trait table, we detected a high correlation between some of the categories as for example, habit, mobility and environmental position are generally related traits in the benthos (Norling et al., 2007; Frid, 2011). Consequently, this table was explored for correlation among traits' categories and a reduced data set was designed trying to avoid high correlation ($\rho > 0.85$). To obtain this reduced traits' data set (Table 3a) some of the original traits and categories were removed (in bold in Table 2), whereas other trait categories were combined to form new categories (highlighted in bold in Table 3a). The final sample by trait table used to perform the statistical analyses was calculated as explained in the beginning of this paragraph.

2.4. Data analysis

The reduced trait abundance data set was squared root transformed to minimize the influence of dominant traits. Afterwards, the similarity between all samples included in the study was calculated using the Bray–Curtis index. Similarity percentages within areas and within sites were obtained with the SIMPER routine using PRIMER 6 & PERMANOVA statistical package (Anderson et al., 2008).

In order to test for significant differences in trait composition and abundance between sites and between areas, we used multivariate generalized linear models (mGLM) with a negative binomial distribution that adjusts to left-skewed data("mvabund" package in R, Wang et al., 2012). The multivariate test statistic was based on the likelihood ratio (LR) and an adjusted p-value was calculated for each trait with a step-down Monte Carlo resampling algorithm with 500 resamples. Traits with significant effects of sites, areas or their interaction term were selected for further analysis.

General Additive Models (GAM) were used to assess the effects of environmental variables on each of the selected traits and on the ESPs. The error structure was quasi-Poisson, due to over-dispersion of the data, and the most parsimonious models were identified with backward selection using the GCV scores (Wood and Augustin, 2002). The environmental variables included in the models were: grain size (sand, mud and gravel percentages), TC and TOC contents, bottom turbidity, oxygen, temperature, chlorophyll a, PAR, and bathymetry (maximum, minimum and depth range for each sample). Fishing effort was also included in the model (data from fishermen interviews and from SSS). Prior to analysis, these variables were explored for significant intercorrelation in order to exclude from the models one of two variables with a correlation higher than 0.85.

3. Results

3.1. Functional variability across sites

According to the SIMPER analysis based on the trait abundance, samples in D site had the highest average within-area similarity (90.3%), followed by samples in L (88.9%), in A (82.6%), in I (80%), in CC (76.8%) and finally samples in CP (60.4%) having the lowest similarity. Within-site similarity followed the same pattern.

The trait abundance across sites (Table 3a) was explored with multivariate GLM analysis. No significant differences were detected between areas or sites. The CP data (characterized by maërl biocenosis and with the lowest within-area similarity) had high variability among samples and this high variability could mask differences between the other areas (having over 75% of within area similarity and with homogeneous sand/mud sediments, Table 1). The analysis was rerun removing all the CP samples and results showed significant differences for Area ($p = 0.002$), Site ($p = 0.006$) and Area: site ($p = 0.002$). Post-hoc analysis for individual traits did not show a consistent pattern (Table 3b).

GAM analyses were performed for the abundance of traits' categories that evidenced significant differences between areas and/or sites (results from the mGLM analysis, Table 3b). Mud had a significant non-linear effect on all these traits' categories, except for "attached filter feeding" (Table 3c). Fishing effort also had a significant effect, either linear or non-linear, on most traits' categories. Other environmental variables had also significant effects on some traits' categories: positive effect of temperature on "asexual reproduction", negative effect of TOC on "opportunistic/scavenger" and oxygen and maximum depth had non-linear effects on "attached filter feeding".

3.2. Link between BTs, ecosystem function and ESPs

In order to explore the performance of the ecosystem functions 1) production, 2) nutrient cycling, 3) bentho-pelagic coupling, 4) carbon

Table 3
a) Reduced set of biological traits (see Materials and methods section), b) results of mGLM test on all areas except for CP and c) results of GAM best models for selected traits. % Dev, percentage of explained deviance by the selected model.*p < 0.05, **p < 0.01, ***p < 0.001."−" no model was estimated; "+/−"positive or negative effect, "s( )" spline effect.

a) Reduced set of BT		b) MVabund			c) GAM		
Trait	Categories	Area	Site	Area:site	Model	%Dev	Factors
Mobility + bioturbation	Swimmer	ns	ns	ns	–	–	–
	Crawler	**	*	ns	***	64.6	s(mud)***, + effort *
	Burrow surface bioturbator	ns	ns	*	***	57.4	s(mud)***, − effort *
	Burrow subsurface bioturbator	ns	ns	ns	–	–	–
	Burrow deep to surface bioturbator	ns	ns	**	***	76.6	s(mud, effort)***
Feeding mode + mobility	Deposit feeder	ns	ns	ns	–	–	–
	Attached filter feeding	ns	ns	*	***	95.6	s(sss)***, s(oxigen)***, s(max depth)***
	Burrowing filter feeding	*	ns	ns	***	82.8	s(mud)***,+effort***
	Opportunistic/scavenger	*	*	ns	***	59.8	s(mud)**, + effort*, − TOC*
	Predator	ns	ns	ns	–	–	–
	Grazer	*	ns	**	***	84.3	s(mud)***
Size	Small (<5 cm)	**	*	ns	***	68.8	s(sss, mud)***
	Medium (5–10 cm)	*	ns	ns	***	84.8	s(mud)***
	Large (>10 cm)	ns	ns	ns	–	–	–
Regeneration	Yes	ns	ns	ns	–	–	–
Asexual reproduction	Yes	*	ns	**	**	66.7	s(mud)***, − sss***, + temp***
Reproductive frequency	Continuous reproduction	**	*	ns	***	72.4	s(mud)***
	2 or more reproductive events per year	ns	ns	*	***	61.8	s(sss, mud)***
Type of larvae	Brooding/viviparous	**	ns	**	***	68.4	s(mud)**, s(sss)*
	Lecitotrophic	ns	ns	ns	–	–	–
	Planktotrophic	ns	ns	ns	–	–	–
Life span	= < 1 yr	ns	ns	**	**	86.7	s(mud)***
	>1 yr	ns	ns	ns	–	–	–
Sexual maturity	>1 yr	**	ns	ns	***	66.2	s(mud)***, + sss***

sequestration and 5) habitat structure in our study sites, 17 ESPs were defined as a combination of traits' categories (Table 2) known to be linked to these functions (Table 4).

For instance, the traits linked to the function nutrient cycling were "size", "bioturbation" and "feeding mode" (Thrush and Dayton, 2002; Lohrer et al., 2004). Consequently, four ESPs related with nutrient

Table 4
Main ecosystem functions provided by epifauna in our study area, their related biological traits (see reference column) and ESPs directly linked with the ecosystem services (classified after the MEA, 2005). When only some categories of a specific BT were used to design ESPs, these categories are shown in brackets (codes in Table 1). Two examples of species belonging to each ESP are also provided.

Ecosystem function	Related biological traitS	References	ESP type	Species example	Ecosystem service type
Production	Size Life span Reproductive frequency Sexual maturity	Robertson (1979) Jennings et al. (2001)	T1: Very fast turnover	*Chlamys opercularis* *Alpheus glaber*	Provisioning
			T2: Fast turnover	*Liocarcinus depurator* *Clausinella brogniartii*	
			T3: Medium turnover	*Bolinus brandaris* *Anseropoda placenta*	
			T4: Slow turnover	*Ascidia mentula* *Squilla mantis*	
			T5: Very slow turnover	*Dercitus bucklandi* *Leptogorgia sarmentosa*	
Nutrient cycling	Size Bioturbation Feeding mode	Lohrer et al. (2004) Thrush and Dayton (2010))	T6: High bioturbators	*Astropecten aranciacus* *Squilla mantis*	Regulating
			T7: Moderate bioturbators	*Anseropoda placenta* *Solenocera membranmacea*	
			T8: Small-scale bioturbators	*Alpheus glaber* *Nucula* sp.	
			T9: Diffuse sediment mixers	*Centrocardita aculeata* *Lesuerigobius suerii*	
Bentho-pelagic coupling	Size Feeding type	Bremner et al. (2006a) Snelgrove (1997)	T10: Large filter feeders	*Alcyonium palmatum* *Ascidia mentula*	Regulating
			T11: Medium filter feeders	*Ocnus planci* *Acanthocardia echinata*	
			T12: Small filter feeders	*Ascidiella scabra* *Centrocardita aculeata*	
Carbon sequestration	Size Life span Growth form (tr and bv)	Bremner et al. (2006a) Frid (2011)	T13: Large or moderate carbon sequestration	*Pecten jacobeus* *Ostrea edulis*	Regulating
			T14: Little carbon sequestration	*Acanthocardia paucicostata* *Bolinus brandaria*	
Habitat structure	Size Life span Habit (er and at)	Thrush and Dayton (2010) Bremner et al. (2006a)	T15: Large 3D structure	*Alcyonium palmatum* *Leptogorgia sarmentosa*	Supporting
			T16: Moderate 3D structure	*Ascidia mentula* *Ostrea edulis*	
			T17: Simple structure	*Ascidiella scabra* *Sertella beaniana*	

cycling were designed: high, moderate and small-scale bioturbators and diffuse sediment mixers. As an example of ESP definition, small burrowing species, either surface or subsurface bioturbators, and having other feeding type than deposit feeding (i.e. *Centrocardita aculeata* or *Lesuerigobius suerii*, filter feeder and predator respectively) were classified as "diffuse sediment mixers" (Table 4).

The abundance and biomass of the taxa exhibiting the traits assigned to each ESP were summed obtaining an "ESP abundance/biomass" data set. Following this approach, we estimated both the quantity and weight of components (ESP) contributing to a particular ecosystem function. The ecosystem functions were related to the provision of ecosystem services (Table 3), with the services defined after the Millennium Ecosystem Assessment (MEA, 2005).

3.3. Ecosystem function performance: distribution of ESPs across sites

The ESPs related with production, bentho-pelagic coupling and habitat structure were numerically dominant in CP sites (Fig. 2). Organisms performing high bioturbation (ESP type 6) dominated the nutrient cycling function in all areas except in CP and CC, where smaller-scale bioturbation prevailed. Carbon sequestration was almost absent from the A area. Otherwise, organisms linked to little carbon sequestration (ESP type 14) dominated in the D area; these organisms were also abundant in L and I areas. Organisms linked to large/moderate carbon sequestration (ESP type 13) were more abundant in CP (Fig. 2). Biomass estimations followed similar patterns, although large ESPs (e.g. ESP type 5 or ESP type 10) generally gained importance with biomass estimations. However, biomass patterns for organisms exhibiting carbon sequestration differed from patterns based on abundance estimations. While CP area stands out in the abundance estimation, L and I areas surpassed CP area in weight. For this function, it is also worth to note the high importance of organisms with little carbon sequestration (ESP type 14) in L, I and D areas when considering their weight.

The abundance of the ESPs with larger contribution to the function provision was consistent across sites within the same area except for organisms exhibiting large carbon sequestration in CP, which were

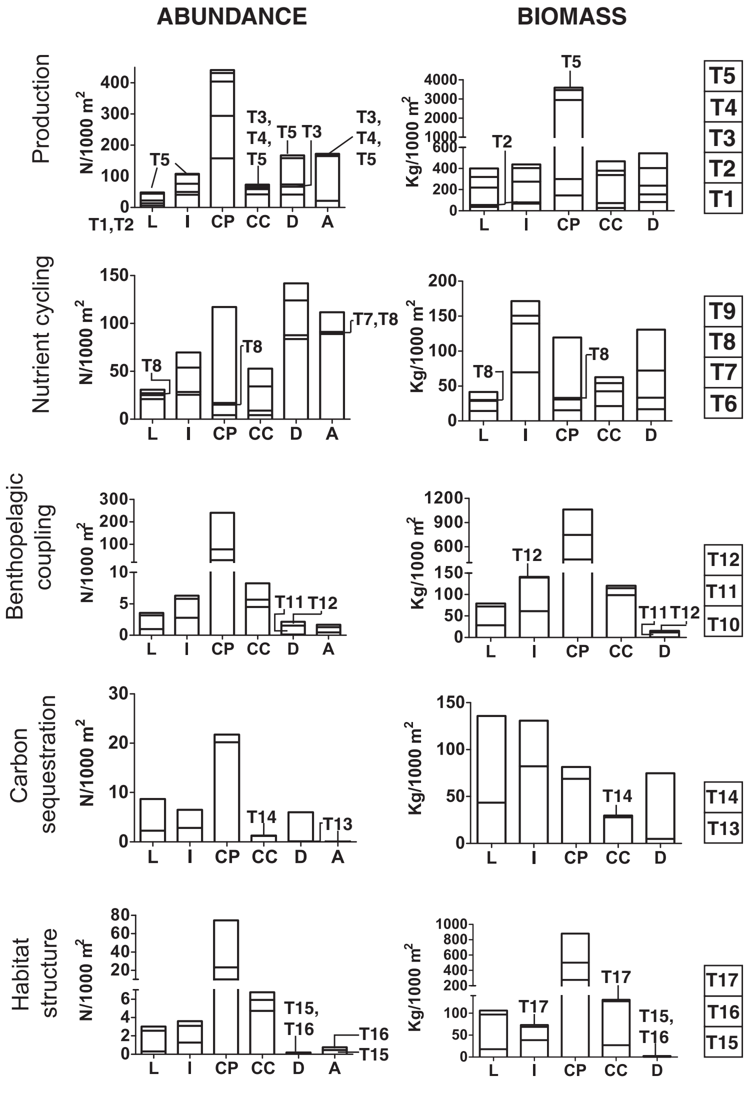

Fig. 2. Accumulated abundance and biomass for epifaunal ESPs related with each ecosystem function across areas. Right bars indicate the order in which each ESP type is placed in the graphics, which is also written into the graphic when low values make it difficult to interpret. ESP codes follow those in Table 4.

more abundant in CPL and CPM than in CPH (Fig. 3). The relative importance of the abundance of functions across sites was consistent with Fig. 2; however Fig. 3 evidences the importance of high-rate bioturbators in D and A areas.

The environmental variables having the strongest effects on ESPs were sediment type and fishing effort, which played a role in almost all degrees of function performance (Table 5). For the biomass estimations, in four cases none of these variables (i.e. sediment and fishing effort) were included in the model: very slow turnover, moderate and small-scale bioturbation and important 3D structure. Other variables such as maximum depth, temperature, TOC, and turbidity also affected the abundance of ESP (Table 5). The results were slightly different depending on whether we considered abundance or biomass; for instance, the abundance of large filter feeders (ESP type 10) was affected by sediment variables but not by effort, while their biomass was affected by effort but not by sediment variables.

4. Discussion

4.1. From benthic functional composition to ecosystem service delivery

Understanding ecosystem functioning and the environmental factors that regulate functional composition is the first step towards understanding the mechanisms that govern the delivery of ecosystem services (Townsend et al., 2011). A number of studies have focused on benthic functional composition in fishing grounds (e.g. Bremner et al., 2003; de Juan et al., 2007; Tillin et al., 2006) and, in accordance with these studies, our results evidenced that epifaunal functional composition was highly conditioned by the intensity of trawling activities in soft-bottom fishing grounds (Table 3c). Otherwise, previous studies, also conducted in trawling grounds, did not find consistent relationship between sediment type and functional assemblages (Bremner et al., 2006b; Tillin et al., 2006), although Barberá et al. (2012), who studied a range of habitats including maërl biocenosis, found a significant correlation between some functional components and the mud content in sediments. Our study, that included a variety of soft-bottoms (from bare mud to maërl beds) detected significant effects of sediment type on the functional composition. These results evidence the importance of multi-site studies that encompass habitat variability. The effects of sediment composition and heterogeneity on the structure of benthic communities were obvious in our study area as CP samples, with heterogeneous sediments (de Juan et al., 2013), had high within-site variability which masked statistical differences among the other more homogeneous areas. It is noteworthy pointing out that sediment generally had non-linear effects on functional components. This non-linearity hinders the extrapolation into a general cause–effect mechanism.

Other variables besides effort and sediment characteristics became significant when scaling up from ecosystem functional components (i.e. abundance of BTs) to functional performance and service provision (i.e. a combination of functional components that provide a specific service, ESPs). This suggests that an increasing complexity of interactions between the species and the environment regulates the provision of

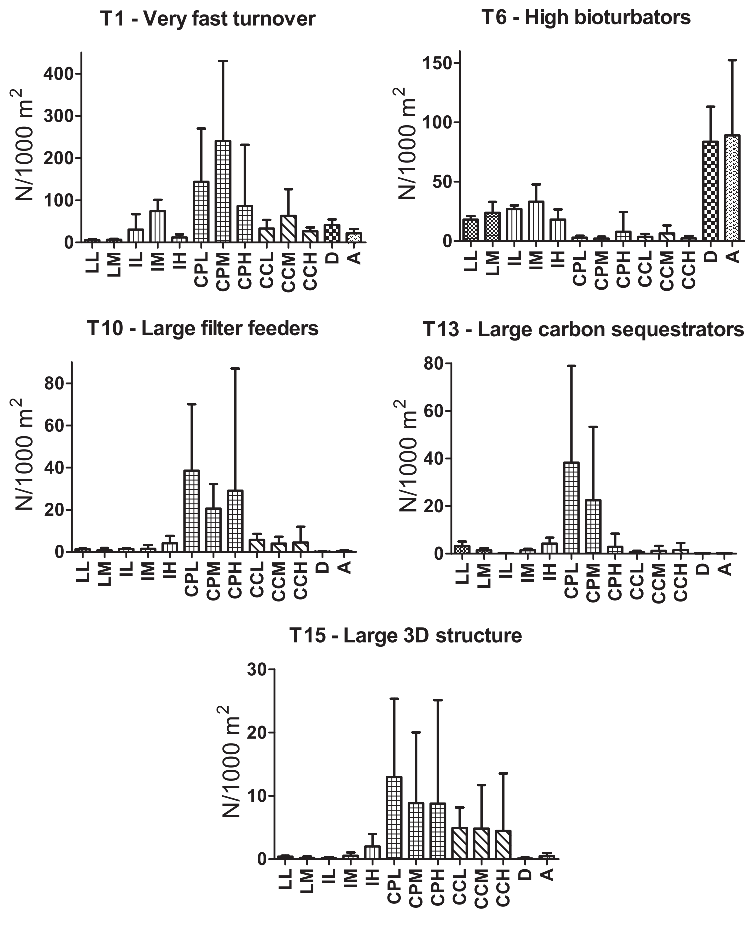

Fig. 3. Abundance of the main ESPs across sites. Letters following the site names stand for the fishing effort levels: L: low; M: medium and H: high.

Table 5
Results of GAM for selected ESPs a) abundance and b) biomass. *p < 0.05, **p < 0.01, ***p < 0.001."+/−" positive or negative effect, "s" spline effect. ESP type codes follow those in Table 4.

ESP type	a) Abundance			b) Biomass		
	Model	%Dev	Factors	Model	%Dev	Factors
T1	***	60.9	s(sss)***	**	57.2	s(D50)***, s(turbidity)**, −T°**
T2	***	72.5	s(sss)***	***	57.4	s(effort)*, s(D50)**
T3	***	64.5	D50:sss***, s(max. depth)**	***	47.9	s(mud)***
T4	***	56.9	s(effort)**, −max. depth***, turbidity**	***	48.5	s(D50)***, +sss**
T5	***	59.8	s(sss)***, s(T°)***	***	61	s(PAR)***, −turbidity**
T6	***	90.2	s(mud)***, turbidity**, effort***	***	62.8	s(sss)**, s(turbidity)***
T7	***	12.6	−mud:TOC**	***	44.2	s(max.depth)**, +turbidity,***, TOC**
T8	**	81.7	s(mud)***, −TOC***	**	64.3	s(max. turbidity)**, +max.depth*
T9	***	72	s(mud)***, −sss***	**	72.1	s(effort)***,s(D50)***, −T°*,+ turbidity**
T10	**	85	s(oxygen)***, s(mud)***, −T°*	***	59.7	s(effort)***, +oxygen***, TOC**
T11	***	45.02	−mud***, T°***	***	52.9	s(D50)***
T12	**	81	s(mud)***, −effort**	***	87.2	s(D50)**, s(turbidity)***
T13	*	77.4	s(sss)***, s(turbidity)**	***	42.6	s(T°)*, −TOC*, −effort**
T14	***	53.7	s(max.depth)***, D50**	***	72.9	s(SSS)***
T15	*	46.8	s(D50, sss)***	***	59.9	s(TOC)*, +max depth**, +T°***
T16	*	68.2	s(sss)***	***	64.7	s(TOC)***, s(effort)*
T17	***	89.6	s(max. depth)**, sss**, −mud***	***	80.5	s(D50)***

ecosystem services, which has important implications for the estimation of temporal and spatial patterns in service provision. However, generally fishing effort had a significant effect for both functional components and ESPs (Table 5). These are important findings as fishing intensity is the only variable susceptible to be managed in our case study; thus, a change of the fishing effort could affect function performance and hence the overall ecosystem service delivery.

Comparing all sampling sites across areas, we observed that the abundance patterns of ESPs were highly consistent within an area. Taking into account that sites within an area were highly homogeneous regarding the sediment composition, these results suggest that sediment type conditions the settlement of particular ESP (e.g. sessile and attached organisms prevailed in areas with coarser substrate like CP and CC). These patterns could hinder the detection of trawling effects when dealing with a wide range of habitats. Consequently, trawling will certainly affect the abundance of ESPs (Table 5), but the dimension of effects will depend on the environmental context, e.g. the initial effects of trawling would be different in the maërl grounds in CP compared to the muddy bottoms of D.

The relative importance of the ESP was different in each area depending on the chosen parameter (abundance or biomass) (Fig. 2). As expected, the ESPs that included the trait "large size" generally increased in relative importance when considering the biomass metric. However, in some cases, biomass patterns were also conditioned by some "small size" ESPs. For example, in L and D areas, biomass of the ESP "little carbon sequestration" was more prevalent than the biomass of "large carbon sequestration" ESP. Therefore, the question remains, does a large quantity of small organisms and lower quantity of large organisms equally contribute to the carbon sequestration? However, to actually address this question, functions should be quantitatively measured on different scenarios (e.g. rates of nutrient cycling, carbon retention (Lohrer et al., 2004; Ståhl et al., 2004)). In this particular case, the species that contributed more to little carbon sequestration was *Bolinus brandaris*, a medium-sized and medium-length life span gastropod common in soft-bottom trawling grounds across the Mediterranean (Martín et al., 1995). In our study area, the pooled biomass of this species might partially counteract the lack of large long-lived bivalves to perform carbon sequestration. However, to measure carbon sequestration, turnover rates should also be taken into account and this is generally difficult to estimate due to the lack of information on life-history traits for many species (Bolam and Eggleton, 2014).

Medium-size organisms may also gain importance with regard to large organisms, as happened for habitat structure in CC area. In this case, the ESP "large 3D structure" comprised a large number of gorgonians, which were lighter than the ascidians and sponges that make up the "moderate 3D structure" ESP. However, higher biomass of medium size organisms did not translate to an increase in habitat structure, as smaller organisms do not provide the same 3D structure. As Bolam and Eggleton (2014) already pointed out, these results highlight the importance of the chosen metric in assessing function performance, and the need to evaluate each function separately.

4.2. ESP as an indicator of good management practices

The evaluation of ESP abundance and biomass patterns and how these will be affected by different human activities allows a better linkage between the final services people perceive and the supporting ecosystem functions that may otherwise be ignored in decision-making (Townsend et al., 2011; Rees et al., 2012). But a single ecosystem function usually links with many services (Townsend et al., 2011) and ecosystem functions and services are interrelated creating a matrix of trade-offs and synergies (Snelgrove et al., 2014). For example, in our case study, both nutrient cycling and bentho-pelagic coupling enhance the nutrient exchange between sediments and the water column making nutrients available for primary producers; however, these two functions did not overlap in our study sites. The ESP with a larger contribution to the function nutrient cycling (i.e. high bioturbators) was more abundant in muddy areas subjected to high-moderate fishing pressure (L, I, D, A), whereas it was almost absent from the sandiest areas CC and CP. On the other hand, bentho-pelagic coupling prevailed in these gravelly-sand and mäerl areas, CC and CP. The sites within CC and CP (except for CPH) were subjected to relatively lower fishing intensity compared with the sites from the other areas. Despite sediment variability, these sites had similar abundance of the ESP bentho-pelagic coupling. Hence, the fishing effort might impose a trade-off between bentho-pelagic coupling and nutrient cycling in our study areas (note that the two BT used to describe the ESP contributing more to the bentho-pelagic coupling function, "large" and "filter feeding", are very vulnerable to trawling (Thrush and Dayton, 2002)). The challenge for management is to deal with these trade-offs and synergies at the appropriate spatial and temporal scales to encompass both ecosystem functioning and fishing activities, while identifying significant links that condition the provision of ecosystem services (Levin and Lubchenco, 2008; Daily et al., 2009).

The ESP approach based on BTA provides a promising framework to assess ecosystem functions that are ultimately related with the delivery of ecosystem services. Furthermore, ecosystem services can be integrated into a DPSIR framework which might help to understand how benthic ecosystems and management decisions interact and what is the role of society in this interaction (Mangi et al., 2007; Atkins et al., 2011).

In the present work, we adopted a DPSIR framework in order to illustrate the interaction between all the actors involved in an integral ecosystem management for trawling fisheries. Adopting this framework, we aimed to summarize in an understandable way the complex socio-ecological interactions occurring in trawling grounds, where the society demands plays a very important role in the achievement of a Good Environmental Status of trawling grounds (EC 2008/56) (Fig. 4). The different DPSIR elements were considered as follows:

Drivers: We considered society to act as a main *Driver* in the fisheries context through food demand and/or society willingness to preserve marine ecosystems. In this context, society may act as an interest group that would influence management practices through ecosystem provision–ecosystem use feedback processes by demanding managers to implement those measures that better fit society interests. Consequently, society would have an indirect but important effect on the ecosystem status.

Pressures: Society demand for sea food would impose a *Pressure* (trawling activities) on the marine environment, which can be divided into two "sub-pressures" that would affect different ecosystem components: catches and gear effects.

State change: The sub-pressure "catches" would lead to a *State change* on target species' populations (Jackson et al., 2001; Pontecorvo and Schrank, 2014) and the sub-pressure "gear effect" would affect the seabed and non-target communities (Thrush and Dayton, 2002), resulting on a *State change* of benthic communities and sediments. It is important to bear in mind that all these ecosystem components (target species, benthic communities and sediments) are interrelated, hence, the direct changes caused by trawling may also indirectly affect other ecosystem components (Muntadas et al., 2014).

Impact: The *State change* would subsequently affect ESPs, causing an *Impact* on ecosystem functions and, hence, on the delivery of services. Moreover, the *State change* on target species would have a direct *Impact* on the service "fish for food". It is also important to take into account the synergies and trade-offs among ecosystem functions and services that could modulate their performance (Snelgrove et al., 2014). Note that impacts on ecosystem functions and services may also feedback, generally in a negative way, on target species (Muntadas et al., 2014).

Response: The *Impact* on ecosystem service delivery may lead to a demand for a management *Response*, as society perceives it is no longer benefiting from the services provided by ecosystems. Therefore, society would demand for management actions, e.g. change of fishing effort levels (Fig. 4). However, a management *Response* could imply (a) an effort increase aiming to maintain fishing catches, which would represent a decrease on the CPUE (Catch per Unit Effort) (Sardà, 1998), or (b) an effort decrease aiming to ensure sustainable exploitation. As we assume that society demand would be the *Driver* of management actions, it is important to arise public awareness on the key role benthic communities' play for ensuring the delivery of both non-economic (e.g. habitat provision) and economic (e.g. fish for food) ecosystem services.

The DPSIR framework highlights that ecosystem and social components are closely related (Fig. 4); hence, the interaction between both components would determine the services provided by benthic ecosystems. Moreover, as ecosystem functions and services are interrelated, we must adopt an integrated management approach taking into

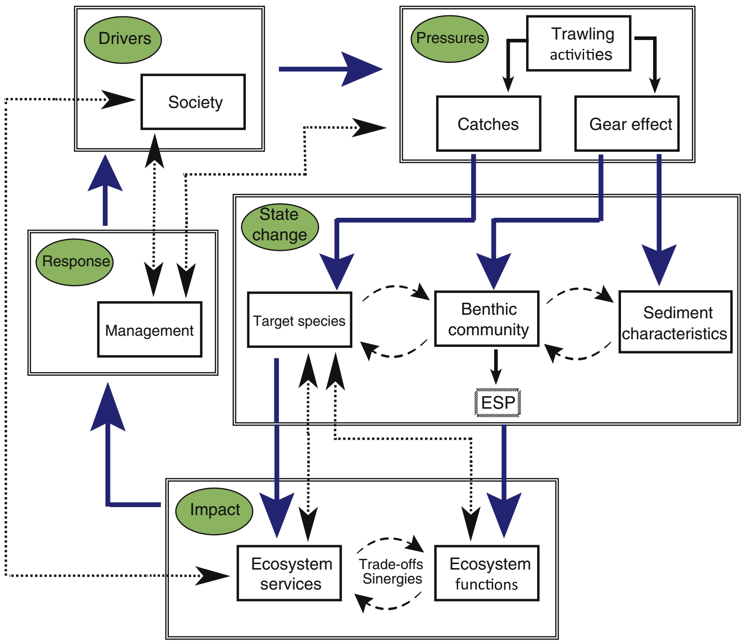

Fig. 4. DPSIR framework adapted to the trawling fisheries context. Each double-lined box corresponds to one of the DPSIR elements which may include one or several ecosystem/society components (small boxes inside). Blue arrows represent DPSIR effects while black dashed arrows between double-lined boxes represent feedbacks between ecosystem/society components. Dark dashed arrows within double-lined boxes represent relationships among the elements of that particular DPSIR element. See the text for a detailed description.

account the trade-offs and synergies among ecosystem components, as well as between ecosystems and society.

5. Conclusions

We propose an approach to estimate ecosystem functions of a trawled system through ESPs, focusing on the ecosystem components rather than on economic value. This approach aims to understand benthic functional structure as a previous step to assess services delivered by ecosystems. We focused our analysis on fishing grounds where, as a direct benefit for society, the most evident delivered service is commercial catches. However, this is not the only service these ecosystems provide and, in fact, this service relies on other ecosystem functions whose improvement could actually benefit fish production, e.g. the habitat structure improvement directly benefits demersal fish stocks. Trawling implies a trade-off between food provision and other services, as it negatively affects key species involved in ecosystem functions such as bioturbation, habitat structure or carbon storage. Nowadays, scientists need to arise society awareness on ecosystem conservation in order to stimulate good management practices that ensure fisheries catches and ecosystem health. Furthermore, in order to ensure the provision of ecosystem services, relationships between ecosystems and society must be identified in an understandable way to inform managers. To this aim, we adapted a DPSIR framework to our case study summarizing the main socio-ecological relationships in trawling grounds. Considering all the elements involved in trawl fisheries allows taking into account the complexity of natural systems. This approach that considers society and ecosystems as a whole, instead of focusing on single system elements, increases the probability to reach the desired management outcomes, i.e. the achievement of sustainable fisheries.

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The Ecosystem Approach to Fisheries in the Mediterranean and Black Seas
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Trawling disturbance on benthic ecosystems and consequences on commercial species: a northwestern Mediterranean case study

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Summary: Trawling is known to disturb benthic communities and habitats, which may in turn indirectly affect populations of commercial species that live in close association with the seabed. The degree of impact on both benthic communities and demersal species depends on the fishing effort level. This may vary over the year because of the fleet dynamics, which are in turn normally driven by the main target species' life cycle. In this study we describe changes in benthic functional components of a northwestern Mediterranean fishing ground that represents a recruitment area for an important target species (red mullet, *Mullus barbatus*). This fishing ground experiences a varying intensity of fishing effort over the year and benthic functional components under different levels of trawling were compared with an unfished, control area. Traits related to sexual maturity and life span for infauna and body size and life span for epifauna were found to vary with fishing activity. Potential effects of these changes on ecological functioning and the impact on red mullet population are discussed. The development of fisheries management plans under an ecosystem based fisheries management (EBFM) requires the links between target species and benthic communities' disturbance due to fishing practices to be explicitly considered.

Keywords: benthos; BTA; functioning; mullet fisheries; trawling; NW Mediterranean.

Perturbación de la pesca de arrastre en ecosistemas bentónicos y sus consecuencias en las especies comerciales: un caso de estudio en el Mediterráneo Noroccidental

Resumen: Se sabe que la pesca de arrastre provoca una perturbación en los hábitats y ecosistemas bentónicos, lo cual a su vez puede afectar indirectamente a las poblaciones de especies comerciales que viven en estrecha relación con el fondo marino. El nivel de impacto en las comunidades bentónicas y en las especies comerciales depende en ambos casos del nivel de esfuerzo pesquero. Este esfuerzo puede variar a lo largo del año, ya que la dinámica de la flota está normalmente determinada por el ciclo vital de las especies objetivo. En este estudio se describen cambios en los componentes funcionales del bentos de un caladero del Mediterráneo noroccidental que constituye un área de reclutamiento para una importante especie objetivo como es el salmonete de fango (*Mullus barbatus*). Este caladero experimenta variaciones de la intensidad de esfuerzo pesquero a lo largo del año. Los componentes funcionales del bentos sometidos a estos niveles variables de esfuerzo fueron comparados con los de una zona control que no está sometida a la pesca. Los resultados muestran que características relacionadas con la madurez sexual y el periodo de vida para la infauna y con el tamaño corporal y el periodo de vida para la epifauna variaron con el esfuerzo pesquero. En el trabajo se discuten los efectos potenciales de estos cambios en la funcionalidad del ecosistema y su impacto en la población de salmonete. Para desarrollar planes de gestión pesquera en el marco de la gestión basada en el ecosistema (EBFM) se requiere que estas relaciones entre la perturbación de las comunidades bentónicas debida a la pesca y las especies objetivo sean claramente consideradas.

Palabras clave: bentos; BTA; funcionalidad; pesquería del salmonete; pesca de arrastre; Mediterráneo noroccidental.

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INTRODUCTION

Trawling is widely held to be the human activity with the greatest impact on continental shelves all over the world (Jennings and Kaiser 1998, Thrush et al. 1998). Towed bottom fishing gears severely disturb the seabed and there are many studies highlighting their negative effects on benthic communities and habitats (e.g. Dayton et al 1995, Kaiser et al. 2000, Smith 2000). These habitats might provide critical environments for various life stages for many commercial species, i.e. spawning, recruitment and growth habitats, and are sometimes termed essential fish habitats (EFH) (Auster and Langton 1998). Alterations to the seabed may therefore indirectly affect commercial species populations, especially for those species living in close relationship with benthos and feeding on it (de Juan et al. 2007a, Fanelli et al. 2010).

The degree of seabed alteration and potential consequences on commercial species will depend on the intensity of the fishing effort. Therefore, it is important to understand the responses of benthic ecosystems to variations in trawling intensity. Communities vary over time and space in response to natural variability (de Juan and Hewitt 2014), which can be confounded with the temporal and spatial dynamics of anthropogenic disturbance factors (Koch et al. 2009). In this context, it is essential to consider the effects of the stressors over time and space, as i) the temporal frequency of the activity might condition the ability of the systems to recover between disturbance events, ii) the spatial intensity of stressors might be linked to the existence of less disturbed areas that, through connectivity mechanisms, can contribute to the ecosystem recovery (Thrush et al. 2013, Planes et al. 2006), and iii) natural variability can influence the cumulative effects as natural oscillations of communities overlap with the stressor effects.

The effects of different levels of fishing pressure on habitats and benthic communities have been explored over spatial gradients of trawling disturbance (Thrush et al. 1998, Jennings et al. 2001, Collie et al. 2005, de Juan and Demestre 2012) and temporal closures (Smith 2000, Hiddink et al. 2006a, Demestre et al. 2008). However, the effect of the temporal dynamics of trawling fleets (i.e. differences in fishing effort intensity over time in the same fishing ground) on benthic ecosystems are still unknown and cumulative impacts of trawling activities over time probably further compromise the resilience of communities (Hinz et al. 2009).

The European Marine Strategy Framework Directive (EMSFD) established by the European Commission 2008 (EC2008/56) encourages Member States to move towards an ecosystem-based fisheries management (EBFM) in order to protect ecosystems goods and services that marine ecosystems provide. Consequently, it is important to take into account the link between habitat and commercial species in management in order to move towards an EBFM if the goals of the EMSFD are to be met. There are a number of interactions through which fishing activity may influence commercial species (the principal ones are depicted in Fig. 1):

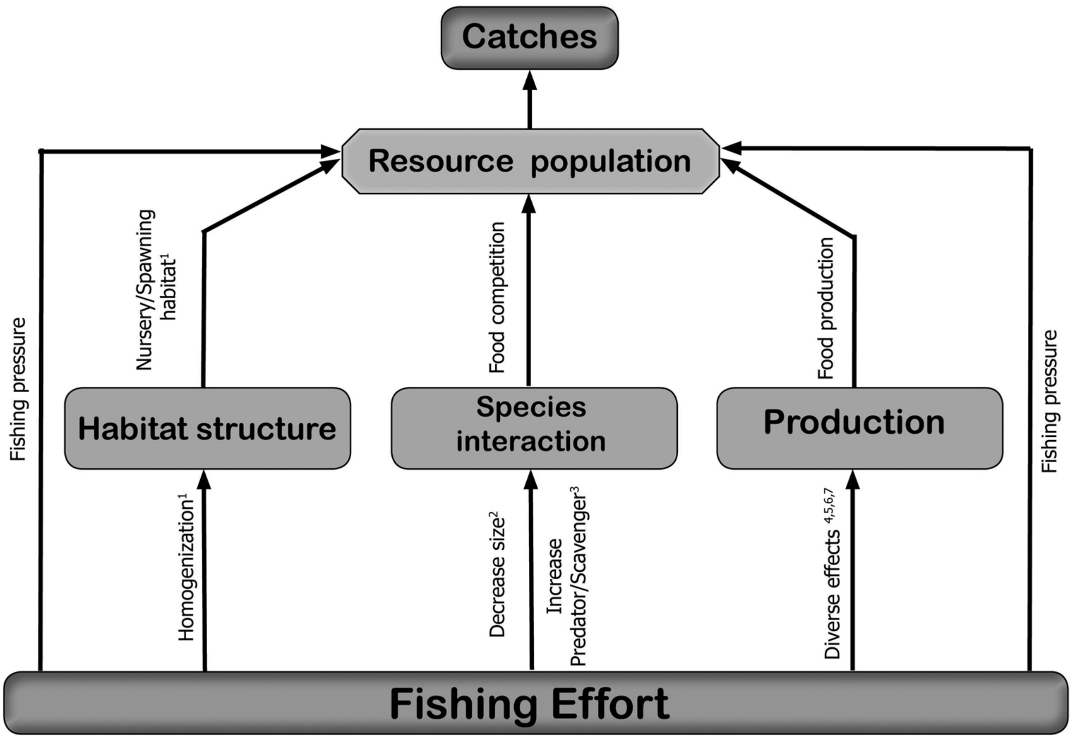

Fig. 1. – Conceptual model depicting the effects of fishing on commercial species. See the text for more details. References: [1] Jennings and Kaiser 1998,[2] de Juan et al. 2009, [3] Rumohr and Kujawski 2000, [4] Jennings et al. 2001, [5] Jennings et al. 2002, [6] Queirós et al. 2006, [7] Hiddink et al. 2006b.

i) Production: ecosystem production is represented as a food source for demersal commercial species. Production by small infauna does not seem to be affected by high fishing intensity (Jennings et al. 2002), although it can increase at moderate levels of disturbance (Jennings et al. 2001), which may benefit species feeding on small opportunistic fauna (Rijnsdorp and Vingerhoed 2001). On the other hand, production by larger infauna and epifauna would decrease in heavily trawled areas (Jennings et al. 2001, Hiddink et al. 2006b, Queirós et al. 2006), which may result in a food impoverishment for fish. Benthic carnivorous fish consume larger prey as they grow (Lukoschek and McCormick 2001), so a decrease in larger infauna might principally affect adult populations and the most economically important components of the stock (Fanelli et al. 2010). Another important food source for several demersal species, especially during the juvenile phase, is suprabenthos, whose abundance and biomass could also be affected by trawling (de Juan et al. 2007a, Fanelli et al. 2011).

ii) Habitat structure: important negative consequences of trawl fishing activity have also been described at the benthic habitat level. Habitat structure that provides shelter and favours the establishment of spawning and nursery habitats may also be altered as trawling activity is known to homogenize habitat structure (Jennings and Kaiser 1998, Thrush et al. 2001).

iii) Species interactions: changes observed in trawled areas, such as changes in epifaunal composition, might also affect commercial species as some epifaunal species compete with commercial fish for food. The epifaunal size decrease caused by trawling (de Juan et al. 2007b) might actually benefit commercial species by releasing them from large potential competitors, but the increase in predator and scavenging species (Rumohr and Kujawski 2000) could increase the food competition.

Current management strategies of the Mediterranean trawl fisheries imply effort limitation (temporal and spatial closed areas, engine power limitation, licence controls, etc.) and technical measures (minimum landing sizes, mesh size, etc.) (Caddy 1993, Smith 2000, Lleonart and Maynou 2003). All these measures focus mainly on the commercial species and are far from an integrated ecosystem approach.

Mediterranean trawl fisheries are characterized by seasonal dynamics which are mainly driven by the life cycle of the main target species (Martín et al. 1999, 2014). These characteristics, added to implementation of a closed season for many fleets as a management measure, leads to a pattern of uneven fishing effort over the year.

One of the main demersal commercial species in the study area, a trawl fishing ground in the northwestern Mediterranean, is red mullet (*Mullus barbatus*), which constitutes an average of 7.2% of the total catches and 7.9% of the total income. Although these may seem low figures, it is important to take into account the multispecies nature of Mediterranean trawl fisheries, with typically fewer than five species exceeding 5% of the total catch (Sánchez et al. 2004, Martín et al. 2014). Red mullet is a species closely linked to benthic ecosystems with a well-known biology and life cycle (Demestre et al. 1997, 2000). The study area is part of a nursery ground for red mullet, as muddy sediment and depth around 15-60 m constitute the typical characteristics for this species juvenile habitat (Lombarte et al. 2000, Fiorentino et al. 2004). Likewise, the deepest zones of the study area, 50-80 m, constitute part of a reproductive habitat for this species (Machias and Labropoulou 2002, Fiorentino et al. 2004).

In the study area, fishing effects on benthic communities were assessed by characterizing it as a chronically impacted seabed (de Juan et al. 2007b). Moreover, analysis of a two-month closed period in this area revealed changes in abundance of some epifaunal mobile species (Demestre et al. 2008). In the present study, we increased our current knowledge by evaluating ecosystem responses to seasonal dynamics of trawling activities, having an unfished site as reference and the potential consequences on the exploited red mullet.

With the aim of advancing on EBFM approaches, taking into account all the effects depicted in Figure 1 and the estimations of fishing effort, the goal of the present work is to link changes in benthic functional structure with indirect effects on red mullet population under the following assumptions:

i) changes in benthic production will affect red mullet's food provision;

ii) homogenization of habitat structure affects both nursery and spawning habitat for red mullet's, and

iii) changes in epifaunal assemblage composition affect interspecific competition between epifaunal species and red mullet.

MATERIALS AND METHODS

Characteristics of the study area

The study was conducted on a muddy fishing ground located in the northwestern Mediterranean. This fishing ground spreads over 400 km^2 in a depth range between 30 and 80 m, and the study area covers a depth range of 40-60 m (Fig. 2). Within this study area, samples were taken from a fished site and from a control site that had remained undisturbed for 20 years due to the presence of the remains of an oil platform (Fig. 2) (see de Juan et al. 2007b for details). The fishing ground was operated by Sant Carles de la Ràpita trawling fleet that, with 69 vessels, is the most important trawling fleet in the Catalonia region (northeast Spain). This fleet showed variable activity throughout the year: fishing effort was high during autumn and winter, low in spring and at the beginning of summer and observed a closed season during July and August. Data on fishing effort was obtained from St. Carles de la Ràpita fishermen's association (see Demestre et al., 2008 for further details). Several abiotic characteristics of the area were measured during the benthic sampling, which was timed to cover seasonal pattern of fishing effort (Table 1). The mud sediment content was almost 100% over the whole study period and the temporal variability was not significant, which characterizes the area as a muddy habitat.

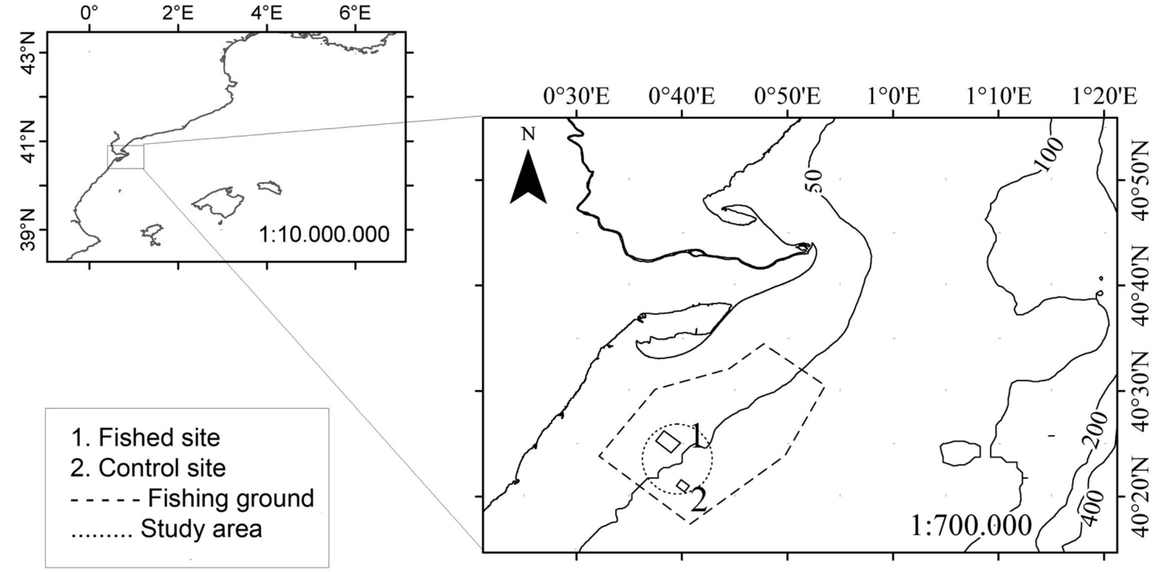

Fig. 2. – Location of the study area in the Catalan Sea off the Ebro Delta (northwestern Mediterranean).

Table 1. – Study area characterization over the sampling cruises. C, Control site; F, Fished site; %OM, organic matter percentage; %Mud, mud percentage on sediment; D50, mean grain size.

Cruise	27 to 30 June 2003		14 to 17 July 2003		28 to 31 July 2003		19 to 22 August 2003		26 to 29 September 2003		17 to 17 November 2003		18 to 21 June 2004	
Site	C	F	C	F	C	F	C	F	C	F	C	F	C	F
Turbidity (mg/l)	1.48	2.38	1.65	1.37	1.25	2.66	1.13	1.72	4.57	5.76	3.27	8.11	1.89	2.88
% OM	0.55	0.59	0.48	0.59	0.54	0.64	0.63	0.70	0.61	0.68	0.51	0.63	0.52	0.61
%Mud	99.5	99.59	99.48	99.46	99.23	99.52	99.29	99.47	99.38	99.43	99.33	99.5	99.33	99.51
D50 (μm)	2.65	4.53	2.73	4.58	2.68	4.41	2.67	4.55	2.82	4.78	2.75	4.82	2.75	4.67
Effort level	-	Low	-	Closed	-	Closed	-	Closed	-	High	-	High	-	Low

Collection and processing of samples

Samples of epifauna and infauna were collected during seven experimental cruises. Epifauna was collected with a surface dredge, similar to a 2 m beam-trawl with a 1-cm cod-end, and infauna with a 0.1-m^2 Van Veen grab. On each cruise, a total of three epifaunal and five infaunal replicates were randomly collected at both fished and control sites. To collect the minimum sample size, estimated from species accumulation curves, the surface dredge was towed for approximately 15 minutes for each replicate and five grabs were collected per replicate. Epifaunal and infaunal organisms were identified to the lowest possible taxonomical level, generally species for epifauna and genera for infauna, and counted (see de Juan et al. 2007b for details).

Data on landings and income

Data on landings and income from 2000 to 2011 were obtained from records from the local fish auction that takes place upon the arrival of vessels at port (data source: fishing statistics elaborated by the Fisheries Department of the Catalan government). Data were available on daily landings by species (weight and income) for each fishing vessel.

Trait classification to characterize benthic communities

Eleven biological traits covering aspects of the benthic organisms' morphology, feeding patterns and life histories were selected to represent benthic community. The biological trait approach (BTA) allows community structure and functionality to be better represented in order to link them to the ecosystem services that the community can provide. In our case, a benthic ecosystem from a fishing ground, one of the main ecosystem services that this community provides is food production.

These 11 traits were broken down into categories. For example, feeding type was separated into the categories deposit feeder, filter/suspension feeder, opportunist/scavenger and predator (Table 2). The trait "age at sexual maturity" was treated differently for infauna and epifauna data due to the different life histories of these two groups, with most of the infauna taxa reaching the sexual maturity before a year time and epifauna species maturing later. To reduce the task to a manageable size, the data sets were reduced. For the infaunal assemblage, 25 of 147 taxa were used, comprising the species that contributed 80% of the total abundance plus those that, though not among the most abundant

Table 2. – Biological traits and categories used to describe functional components of benthic communities. Categories used in (i) infaunal and (e) epifaunal analyses

Trait	Categories
Feeding behaviour	Deposit feeders
	Filter/suspension feeders
	Opportunistic/scavengers
	Predators
Food type	Invertebrates
	Carrion
	Detritus
	Plankton
	Microorganisms
	Nekton
Fragility	Fragile
	Intermediate
	Robust
Living habit	Tube dweller
	Permanent burrow dweller
	Free-living
Size	Very small <1 cm
	Small 1-2 cm
	Small-medium 3-10 cm
	Medium 11-20 cm
	Medium-large 21-50 cm
Flexibility	None <10 degrees
	Low 10-15 degrees
	High >45 degrees
Life span	< 1 y
	1-2 y
	3-5 y
	>5 y
Age at sexual maturity	< 1 y (i)/ ≤1 y (e)
	≥ 1 y (i)/> 1 y (e)
Adult movement	Sessile
	Crawl
	Swim
	Burrow
Reproduction frequency	Continuous
	1 reproductive event per year
	2 or more reproductive events per year
	Less than annual
Type of larvae	Direct development
	Short planktonic (<1 week)
	Long planktonic (>1 week)

species, were continuously present in three of the five replicates either at fished or control sites over the study period. The epifaunal data set was reduced to 17 of 96 species, which accounted for 95% of the total abundance and met the frequency of occurrence criteria.

Each taxon in the database was scored for its affinity to each trait category using a scale of 0–3 (0 = no affinity to 3 = high affinity). The score was given using the 'fuzzy scoring' method, which allowed the taxa to exhibit more than one category of a given trait as long as the total score per trait was 3 (Bremner et al., 2003). This assignment was based on published accounts of the biology of each species and information codified in the BIOTIC database maintained by the Marine Biological Association UK (http://www.marlin.ac.uk/biotic/). This source of information was complemented with a literature review for potential regional differences, i.e. studies conducted in the Mediterranean. When information was not available at the species level we used data based on accounts of other members of the genera or, rarely, family (only in 7.6% of cases for infauna and 3.2% for epifauna). When no information on a particular trait was available for a taxon, zero values were entered for each category and the taxon did not contribute to the calculation of trait weightings.

The frequency of each trait category in the dataset was calculated by weighting the category scores by the abundance (number of individuals per m^2) of each taxon exhibiting that category (Charvet et al. 1998). This resulted in a sample by trait table that showed the abundance of biological traits at each station over the study period.

Statistical analysis

Similarity between each pair of samples was calculated using the Bray-Curtis index after a square root transformation of the data to reduce the influence of dominant traits/species. A PERMANOVA analysis was used to test for significant differences between sites (control and fished as fixed factors) and time (fishing effort periods, i.e. before, during and after closure, as fixed factor). The trait data were further analysed with the SIMPER routine to determine which traits accounted for the significant dissimilarities identified by PERMANOVA. Then, the most important traits highlighted by SIMPER (traits showing ratio of dissimilarity to standard deviation [diss/sd] >1.5 and being among the ones summing 50% of cumulative contribution to dissimilarity) were selected for univariate analyses. When traits had a normal distribution and homogeneity of variances, a two-way ANOVA was performed to test for the factors treatment and effort period. If traits were not normally distributed (even after log transformation), a Kruskall-Wallis test was performed instead. All the multivariate analyses were carried out using the PRIMER6 & PERMANOVA statistical package (Anderson et al. 2008). Univariate analyses were performed using the R program, v.2.11.0.

RESULTS

Landings and fleet dynamics in the study area

Figure 3 shows the percentage of mantis shrimp (*Squilla mantis*), hake (*Merluccius merluccius*) and red mullet (*Mullus barbatus*) landings and income with respect to the total landings and income in the study area. The importance of these three species remained consistent over the years, representing around 25% of the total landings and income. The "other" percentage comprises on average 53 species, most of them accounting for less than 5% of the total catches and only sporadically (e.g. in one or two years of the data series) more than 5%.

Despite an overall decrease since 2007, red mullet landings consistently followed the same trend, with a peak of catches in September/October (the recruitment months for this species), which accounted for almost 20% of the total landings (Fig. 4A, B). Figure 4C shows how this peak in red mullet catches coincided with the high effort period. However, red mullet landings dropped sharply in late autumn (November) whereas the decreasing trend of fishing effort was smoother as mantis shrimp landings were maintained over the winter (Fig. 4C). This figure shows how fleet dynamics followed red mullet population, as the high-

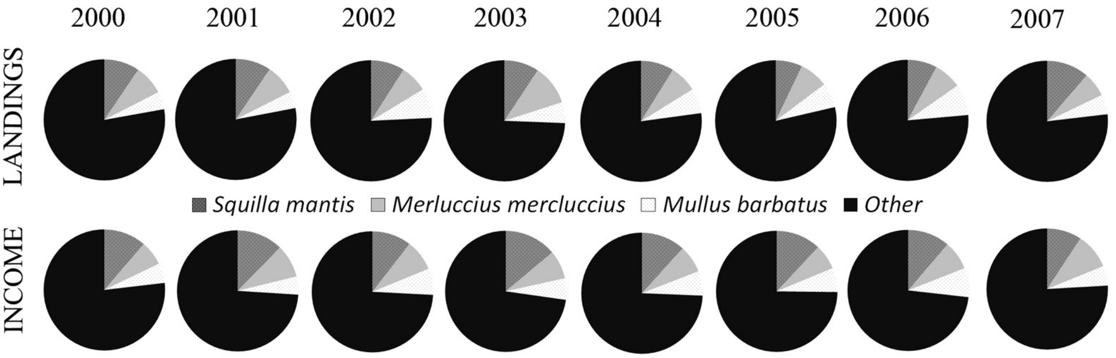

Fig. 3. – Percentage of landings and income for the main target species of the Sant Carles de la Ràpita trawling fleet.

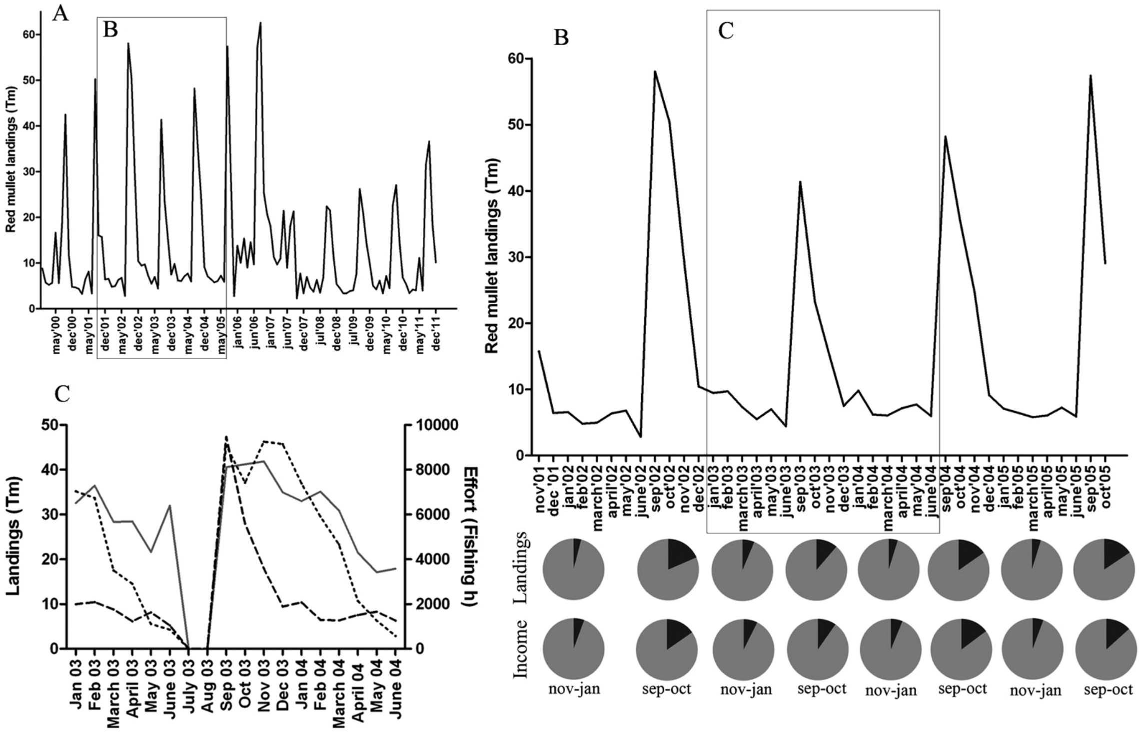

Fig. 4. – A) Evolution of red mullet catches over the years. B) Zoom from November 2001 to October 2005. Pie charts show the percentage of landings and income of red mullet (black area) with respect to total landings and income, respectively (grey area). C) Landings of *Mullus barbatus* (stripped line) and *Squilla mantis* (dotted line), and fishing effort (grey continuous line) over the study period (2003-2004).

est landings and effort occurred just after the closed season, when the trawling fleet gathered on this fishing ground to fish red mullet recruits (Demestre et al. 1997, Martín et al. 1999).

Functional changes in the benthic communities

PERMANOVA analyses highlighted significant differences for both infauna and epifauna between sites (control vs. fished) and time (different effort regimes), and site:time interaction for infauna (Table 3). Pairwise tests performed for infauna within fished sites showed significant differences between before closure and the closed season ($p<0.01$) and between before closure and after closure periods ($p<0.05$). Furthermore, differences between times for epifauna were independent of the site, and pairwise tests performed across all samples

Table 3. – Infauna and epifauna PERMANOVA results for the fixed factors time and site. Significant p-values are highlighted in bold.

INFAUNA	df	MS	Pseudo F	p-value
Time	2	158.56	3.87	**0.008**
Site	1	1199.40	29.33	**0.001**
Time:site	2	246.35	6.02	**0.003**
EPIFAUNA	df		Pseudo F	p-value
Time	2	460.56	5.70	**0.002**
Site	1	474.93	5.88	**0.006**
Time:site	2	83.05	1.03	0.354

Table 4. – Results of SIMPER analysis for infaunal trait abundance, comparing fished (Ab. Fished) and control (Ab. Control) sites. Fished vs control analysis was based on the whole data set, whereas effort regime analysis was based only on fished site samples. Cut-off to traits list applied at 50% of cumulative contribution to dissimilarity (Cum. Contrib.%); traits with a ratio of dissimilarity to standard deviation (diss/sd) >1.5 are highlighted in bold.

FISHED vs CONTROL (Average dissimilarity=12.64)					
Traits	Ab. Fished	Ab. Control	Diss/sd	Contrib %	Cum. Contrib %
<1 y (sex. mat)	24.29	36.98	**1.61**	5.44	5.44
<1 y (life span)	24.05	34.90	**1.50**	4.77	10.21
High flexibility	41.92	47.32	1.38	4.31	14.52
Direct development	32.34	39.36	1.45	4.15	18.67
Medium-large	21.47	12.13	**2.90**	4.09	22.76
Continuous reproduction	25.73	31.67	1.46	3.96	26.69
Tube dweller	18.15	26.10	**1.50**	3.62	30.31
Permanent burrow dweller	24.25	31.70	**1.54**	3.49	33.82
Detritus	41.44	46.80	1.38	3.51	37.30
Filter feeder	20.47	27.95	**1.51**	3.45	40.75
Small size	21.33	28.89	**1.51**	3.28	44.03
Burrow	44.44	48.03	1.33	3.25	47.28
Small-medium size	31.28	35.20	1.40	3.24	50.51
BEFORE vs CLOSED (Average dissimilarity=10.03)					
Traits	Ab. Before	Ab. Closed	Diss/sd	Contrib %	Cum. Contrib %
High flexibility	40.13	45.89	**1.54**	4.29	4.29
Intermediate fragility	40.88	46.93	**1.56**	4.18	8.47
<1 y (sex. mat.)	21.93	28.21	**1.57**	3.86	12.33
Burrow	42.51	46.73	1.43	3.85	16.18
1 repr. event/year	35.39	41.18	1.46	3.72	19.90
Direct development	30.35	35.67	1.42	3.69	23.59
Detritus	39.49	43.93	1.40	3.61	27.20
<1 y (life span)	22.42	27.92	1.41	3.52	30.72
Deposit feeder	37.95	42.12	1.40	3.33	34.04
2+repr. events/y	3.37	8.94	**1.76**	3.26	37.30
Filter feeder	17.72	23.27	**1.70**	3.15	40.45
Free-living	33.90	37.92	1.47	3.14	43.60
Small-medium size	30.61	33.56	1.43	3.07	46.66
very small size	18.38	22.17	1.08	3.03	49.69
BEFORE vs AFTER (Average dissimilarity=8.65)					
Traits	Ab. Before	Ab. After	Diss/sd	Contrib %	Cum. Contrib %
High flexibility	40.13	45.97	1.46	4.62	4.62
Intermediate fragility	40.88	46.56	1.43	4.12	8.73
Burrow	42.51	46.61	1.44	3.86	12.59
Direct development	30.35	35.28	1.44	3.84	16.43
Detritus	39.49	43.83	1.42	3.81	20.23
small-medium size	30.61	34.40	1.47	3.70	23.93
Crawl	9.56	15.01	**1.78**	3.66	27.59
Deposit feeder	37.95	42.40	1.40	3.61	31.20
1 repr. event/y	35.39	40.27	1.48	3.48	34.68
<1 y (sex. mat.)	21.93	25.81	1.37	3.34	38.02
Filter feeder	17.72	22.52	**1.56**	3.27	41.29
Continuous reproduction	25.00	28.18	1.39	2.97	44.26
Long planktonic	16.69	20.28	1.44	2.88	47.15
Free-living	33.90	37.53	1.38	2.86	50.01

showed significant differences between before closure and after closure periods ($p<0.01$) and between closed season and after closure periods ($p<0.01$). Ordination of samples in a multi-dimensional scaling plot did not reflect any clear pattern and was not included.

SIMPER analysis for infaunal samples highlighted the traits “sexual maturity at less than 1 year” and “life span of less than 1 year” as the principal traits driving the differences between fished and control sites, both being more abundant in the control site. “Medium-large size” (more prevalent at the fished site) was also an important trait discriminating sites (Table 4).

SIMPER analysis performed only for infaunal fished samples revealed “high flexibility”, “intermediate fragility” and “<1 y sexual maturity” as the main traits driving the differences between before closure and closed season periods, although they showed relatively low diss/sd (Table 4). All these traits were more abundant during the closed season. “Filter feeding” and “2 or more reproductive events per year”, with relatively high diss/sd and also more abundant during the closed season, were important in discriminating between these two periods. Finally, “crawl” and “filter feeding” were the traits making the difference between the before closure and after closure period, both being more abundant in the latest (Table 4).

The SIMPER routine for epifaunal samples highlighted the traits “medium”, “>5 y life span”, “low flexibility” and “medium-large” as the most important traits driving the differences between control and fished sites, all of them being more abundant at the control site (Table 5). As the PERMANOVA test showed no significant interaction between site and time for epifaunal samples (Table 3), a SIMPER test based on the whole data set was performed to identify the traits driving the differences between effort regimes. This analysis highlighted “no flexibility”, “deposit feeder”, “permanent burrow dweller”, “burrow” and “detritus”as the traits

Table 5. Results of SIMPER analysis for epifaunal traits abundance, comparing fished (Ab. Fished) and control (Ab. Control) sites. All analyses were based on the whole data set. Cut-off to traits list applied at 50% of cumulative contribution to dissimilarity (Contrib.%); traits with a ratio of dissimilarity to standard deviation (diss/sd) >1.5 are highlighted in bold.

FISHED vs CONTROL (Average dissimilarity=12.98)					
Traits	Ab. Fished	Ab. Control	Diss/sd	Contrib %	Cum. Contrib %
High flexibility	15.10	13.11	1.26	4.43	4.43
Medium size	1.09	5.00	**2.37**	4.42	8.85
3-5 y (life span)	15.13	13.81	1.24	4.29	13.14
>1 y (sex.mat)	15.40	15.37	1.25	4.18	17.32
Crawl	17.48	15.86	1.25	3.93	21.25
Intermediate fragility	19.10	19.49	1.36	3.81	25.07
>5 y (life span)	5.59	8.83	**1.68**	3.8	28.86
1 repr. event/year	20.05	20.37	1.35	3.73	32.60
Long planktonic	19.42	19.61	1.37	3.71	36.31
Small-medium size	19.77	18.55	1.29	3.69	40.00
Free-living	18.06	18.47	1.43	3.67	43.67
Low flexibility	5.41	8.22	**1.66**	3.42	47.09
Medium-large size	3.80	6.57	**1.72**	3.23	50.32
BEFORE vs AFTER (Average dissimilarity = 16.33)					
Traits	Ab. Before	Ab. After	Diss/sd	Contrib %	Cum. Contrib %
No flexibility	15.46	9.15	**2.16**	6.16	6.16
3-5 y (life span)	13.01	15.00	1.36	4.71	10.87
>1 y (sex. mat)	14.10	15.53	1.37	4.68	15.54
High >45	12.91	14.61	1.40	4.45	19.99
Intermediate fragility	19.96	17.43	1.45	3.97	23.97
Deposit feeder	9.99	6.03	**2.11**	3.95	27.92
Permanent burrow dweller	11.30	7.35	**1.68**	3.94	31.86
1 repr. event/y	20.98	17.93	1.44	3.94	35.80
Long planktonic	20.27	17.40	1.43	3.92	39.72
Burrow	9.93	5.90	**2.06**	3.9	43.62
Detritus	10.30	6.36	**2.03**	3.89	47.51
Small-medium size	20.01	16.99	1.35	3.72	51.23
CLOSED vs AFTER (Average dissimilarity = 13.83)					
Traits	Ab. Close	Ab. After	Diss/sd	Contrib %	Cum. Contrib %
No flexibility	14.48	9.15	**2.10**	6.00	6.00
1 repr. event /y	21.21	17.93	1.44	4.40	10.40
Long planktonic	20.43	17.40	1.42	4.25	14.65
Small-medium size	20.04	16.99	1.44	4.16	18.80
Permanent burrow dweller	10.92	7.35	**1.82**	4.11	22.91
Intermediate fragility	20.10	17.43	1.40	4.08	27.00
Deposit feeder	9.54	6.03	**2.07**	4.01	31.00
Detritus	9.81	6.36	**2.08**	3.99	34.99
Burrow	9.33	5.90	**2.08**	3.86	38.85
Fragile	7.12	3.94	**1.80**	3.84	42.69
Free-living	19.07	16.85	1.39	3.61	46.29
Crawl	17.27	15.84	1.42	3.29	49.58

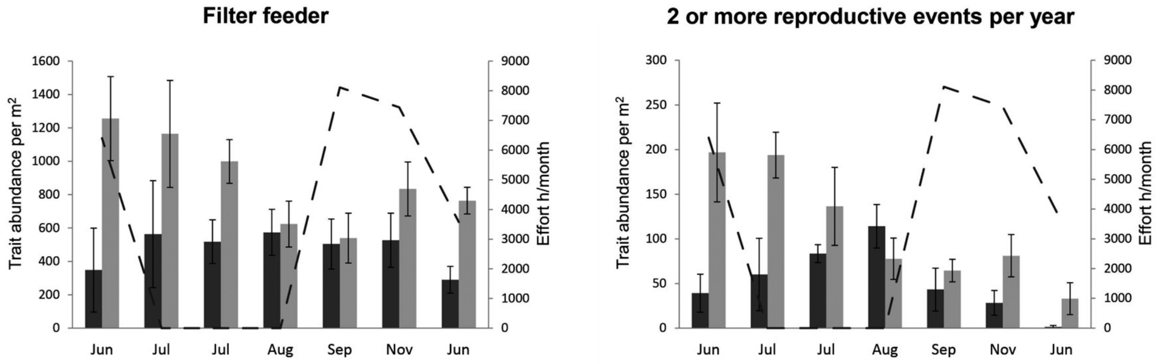

Fig. 5. – Mean (±se) trait abundance of infauna at fished (black bars) and control sites (grey bars). Dashed line shows fishing effort. June corresponds to before closure period, July and August to closed season and September and November to after closure period.

discriminating between the before and after closure periods and between the closed season and after closure periods. All these traits were more abundant in the before closure period and the closed season respectively. "Fragile", more abundant during the closed season, was also found to be important in discriminating between the closed season and the after closure period (Table 5).

Though these traits were highlighted by SIMPER analysis, it should be noted that average dissimilarities among fishing effort periods were low (10.03 and 8.65 for infauna and 16.33 and 13.83 for epifauna) (Tables

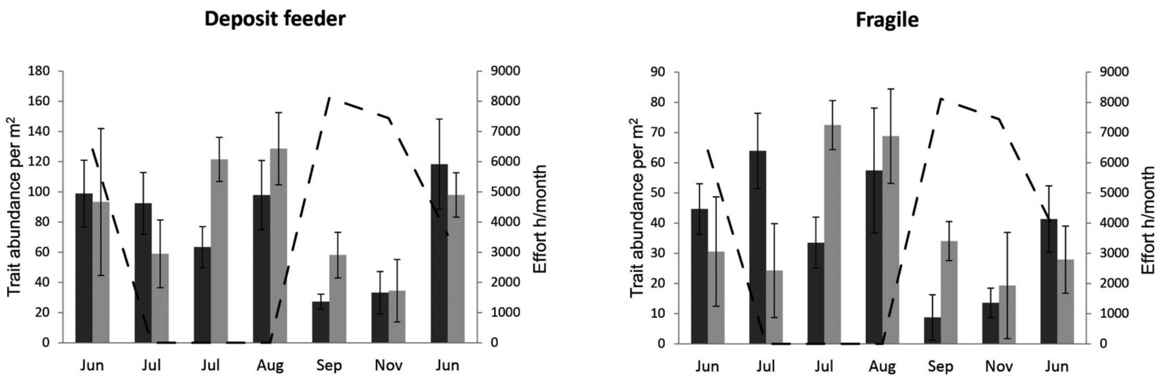

Fig. 6. Mean (±SE) trait abundance of epifauna at fished (black bars) and control sites (grey bars). Dashed line shows fishing effort. June corresponds to before closure period, July and August to closed season and September and November to after closure period.

Table 6. – Summary of ANOVA/Kruskal-Wallis and pairwise tests for traits highlighted by SIMPER routine (traits showing a ratio of dissimilarity to standard deviation [diss/sd] >1.5 and being among the 50% of total contribution). * p<0.05, ** p<0.01, *** p<0.0001, ns, non-significant; b, before closed; c, closed season; a, after closed[1]; the specific comparisons were not significant.

Infauna	Site	Time	Site: time
<1 y (life span)	***	ns	*
<1 y (sex.mat)	***	ns	*
Medium-large size	***	ns	ns
Tube dweller	***	ns	*
Permanent burrow dweller	***	ns	**
Intermediate fragility	ns	* (c≠a, b)	*
High flexibility	*	ns	*
Filter/suspension feeder	***	ns	*
Crawl	*	* (b≠a, c)	ns
2+ repr. events/y	***	*[1]	*
Epifauna	**Site**	**Time**	**Site: time**
Deposit feeder	ns	* (a≠b, c)	ns
Detritus	ns	* (a≠b, c)	ns
Burrow	ns	* (a≠b, c)	ns
No flexibility	ns	* (a≠b, c)	ns
Permanent burrow dweller	ns	*** (a≠b, c)	ns
Fragile	ns	***all times	ns
Medium size	***	ns	*
Medium- large size	***	ns	*
Low flexibility	***	ns	*
>5 y (life span)	**	ns	ns

4, 5). Average dissimilarities between control and fished areas were also low (12.64 for infauna and 12.98 for epifauna).

Figures 5 and 6 show the trends of two traits for infauna (Fig. 5) and epifauna (Fig. 6) over the study period. These traits were highlighted by the SIMPER routine as being important in discriminating between fishing effort periods. Infaunal "filter feeding" and "two or more reproductive events" showed a decreasing trend over the study period at the control site but, whereas for "filter feeding" there seems to be no change at the fished site, in "two or more reproductive events" there was an increasing trend until the end of the closed season and afterwards this trait abundance decreased. Fishing effort values are overlapped in the figure, showing that epifaunal "deposit feeder" and "fragile" organisms were more abundant at fished and control sites when effort was low or null, in the autumn months. Plots for other traits are not included as they showed similar trends to those in Figures 5 and 6: "Filter feeder" trend for infauna was similar to "high flexibility" and highly similar to "<1 y of life span", and"<1 y sexual maturity". On the other hand, the "deposit feeder" trend for epifauna was highly similar to "detritus", "burrow" and "no flexibility", and the "fragile" trend was highly similar to "permanent burrow dwelling".

Table 6 shows that differences over fishing effort periods (time) for almost all these traits were significant, confirming that they were important in discriminating between these periods. Factor time was not significant for infaunal "filter feeder" and thus all the traits show a similar trend; however, the site:time interaction was significant, suggesting an influence of fishing effort periods or seasonality on these traits' changes. Traits highlighted by SIMPER as the ones driving the differences between control and fished sites also showed significant differences for the factor site

DISCUSSION

Potential effects of fishing on red mullet population caused by functional changes in benthic communities

Different fish species have different habitats requirements, which could be more or less resilient to trawling impacts (Kaiser et al. 1999). The Sant Carles de la Ràpita fishing ground, our case study, is part of a nursery habitat for red mullet (Lombarte et al. 2000, Fiorentino et al. 2004), which is one of the most important commercial fish in the study area (Sánchez et al. 2007). Moreover, this species was chosen as a case study because it is a species closely related to the benthic system, with a well-known biology and life cycle which leads to particular catch dynamics (Fig. 4A). The existence of a nursery habitat is supported by the high catches of new recruits observed after the closed season, in September-October, which constitutes a specific characteristic of this species fisheries (Martin et al. 1999). Recruitment is a critical step for most fish life cycles and decline in recruitment may have important consequences for adult commercial stock (Bundy and Fanning 2005, Caddy in press). Therefore, the protection of benthic communities and habitat structure, which provide food and shelter for young fish, is es-

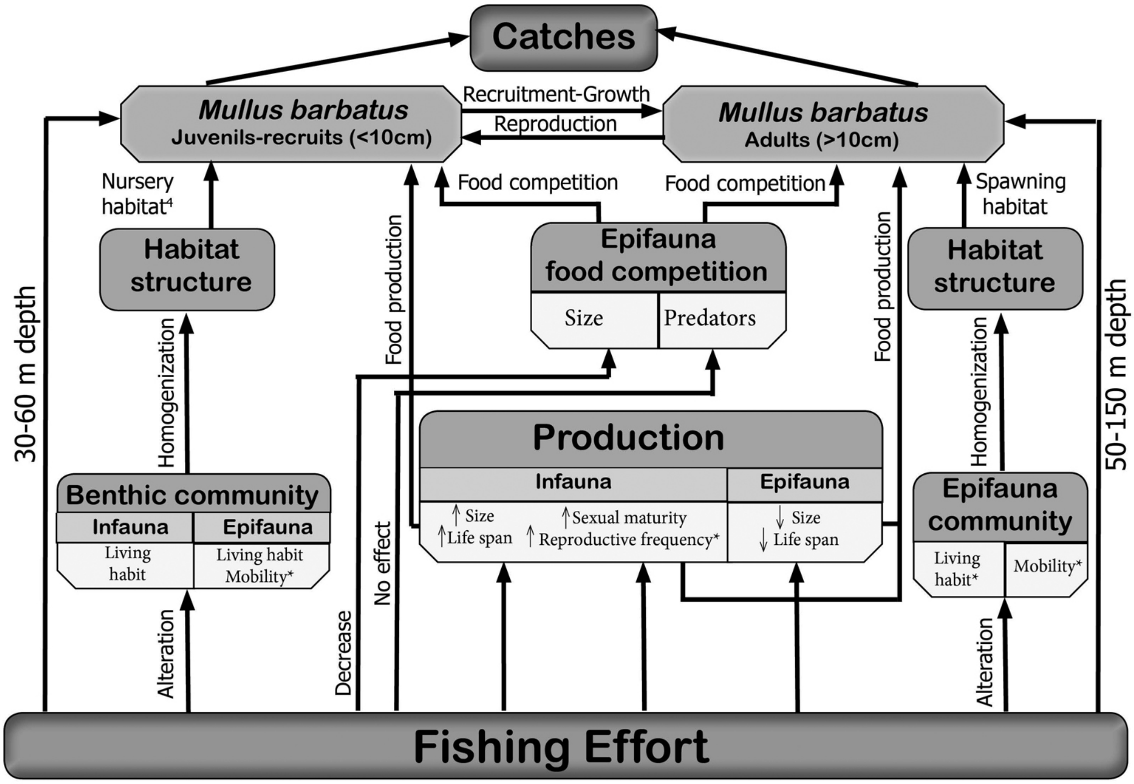

Fig. 7. Complex interactions and main potential effects of changes in benthic community due to trawling on red mullet population. *Traits changing over different fishing effort regimes; the other traits showed differences between fished and control sites

sential to maintain a healthy adult stock.

As red mullet lives in a close relationship with the benthic environment for feeding, reproduction and refuge, this species might be particularly affected by the chronic alteration of benthic ecosystems (Auster and Langton 1998, Caddy in press). In this study we aimed to assess potential negative effects of trawling on red mullet due to changes on benthic functional components in the fishing ground (Fig. 7).

Regarding the food availability, small, short-lived infaunal organisms were more abundant at the control site, which might contribute to higher food production for red mullet in this site, especially for young recruits that feed on smaller prey (Machias and Labropoulou 2002). Moreover, during the closed season in summer, production of small infauna at the fished site may increase due to the higher prevalence of traits related to rapid reproduction (Fig. 7). The closed season coincides with the early phase of recruitment of red mullet (Fiorentino et al. 2008), so young red mullet recruits might benefit from higher food production in summer months. On the other hand, medium-large infaunal organisms dominated the fished site (Fig. 7), which could represent higher prey abundance for adult red mullet in the fishing ground. These results seems to contradict the current understanding of the effects of fishing on marine benthos, suggesting that communities in areas with a long history of fishing would consist mainly of small organisms (Jennings et al. 2001, Queirós et al. 2006). However, all medium-large taxa in the study area were deep burrowers that might avoid trawling disturbance (Brown et al. 2005), and also generally opportunistic-predator feeders that usually benefit from fishing activity (Frid et al. 2000).

Some of the observed trends in infaunal traits over the study period matched the fishing effort pattern, but the variability in infaunal abundance would more likely be related to seasonal patterns as the closed season in the fishing ground is too short to allow ecosystem recovery (Zajac 2003, Kaiser et al. 2006). The decreasing trend for almost all infaunal traits at the control site in summer and early autumn (closed and after closed season) could indicate a decrease in the overall infaunal abundance consistent with the temporal cycles shown by macroinfauna in the Mediterranean (Sardá et al. 1999). In contrast, the reduction of this seasonal trend at the fished site could be due to chronic fishing disturbance, which could alter the natural macroinfaunal cycles (de Juan et al. 2007b). However, this lack of trend in the fished site cannot be unequivocally linked to the fishing activities as natural processes could also play a role, resulting in an additive effect of both fishing and environmental variability (Koch et al. 2009).

In the epifaunal community, as expected and in agreement with de Juan et al. (2009), traits related to long life span and large size were more abundant at the control site (Table 5). This means that epifauna productivity (production/biomass) would be lower in the control area, although biomass production would be higher in this area. These large long-living organisms were fishes and large bivalves (e.g. *Citharus linguatula* or *Acanthocardia echinata*) that are not red mullet preys and, if they are predators, might compete with red mullet for prey. For example, *Citharus linguatula*, feeds in this area on decapoda and other small crustaceans (Juan et al. 2007a), which are potential prey for red mullet (Aguirre 2000), so these two fish partially share the trophic niche. Therefore, although adult red mullet could potentially find more food at the control site, this food would be less available due to interspecific competiveness (Fig. 7).

Regarding the habitat structure, as the control site is slightly deeper than the fished site, it might be a spawning area for red mullet. However, no traits related to habitat structure (e.g. sessile emerging epifauna) were highlighted in the analysis. In general, the whole fishing ground holds a homogenized community with a reduced habitat structure due to historical trawling disturbance in the area (de Juan et al. 2007b, 2009) (Fig. 7). Nevertheless, physical habitat is also important in creating habitat structure for recruits and spawners, and ROV images showed higher abundance of sediment structures such as ripples, mounds and pits at the control site (Demestre 2006). Changes in the trait "burrow" or "permanent burrow dwelling", which seemed to occur in similar abundance at fished and control sites over the sampled months, would not benefit red mullet spawning as these traits would destabilize the sediment with negative consequences for the creation of habitat structure (Lohrer et al. 2008). However, changes in these traits may affect bioturbation activity which may positively affect ecosystem production (Thrush and Dayton 2002, Lohrer et al. 2004) and indirectly benefit red mullet population in terms of food availability by enhancing primary production.

Management considerations

Associating demersal fish with their habitats is very critical to the definition of EFH and to correctly managing those EFH impacted by trawling activities (Kaiser et al. 1999). However, management strategies such as closed seasons are principally implemented to protect vulnerable steps of commercial species' life cycles such as spawning and recruitment, focusing only on commercial species stock and not taking into account protection of benthic communities. Short-term closed seasons, such as the one implemented in the Sant Carles de la Ràpita fishing ground, do not enable benthic community recovery between successive periods of impact, especially on stable muddy bottoms where communities might take years to recover (Kaiser et al. 2006). Furthermore, short-term closures result in a concentration of the trawl effort and landings immediately after the closed season rather than in a more equitable pattern throughout the year (Martin et al. 1999). This concentration of fishing effort is also accompanied by an increase in discards (Sánchez et al. 2007), which indicates a higher level of disturbance on benthic communities. An adequate management regulation should progressively increase the fleet capacity after the closed season to avoid resource depletion and the highest disturbance levels on ecosystems.

Red mullet are an important commercial species in the Mediterranean, being one of the main target species for trawling fleets (Caddy 1993, Tserpes et al. 2002). In this case, the red mullet life cycle drives the fleet dynamics, as the highest fishing effort coincides with the highest landings (Fig. 4C), which are composed mainly of red mullet recruits (Demestre et al. 1997, Martín et al. 1999). Though the red mullet population could apparently tolerate this level of exploitation due to its high turnover rate, the yield per recruit shows evidence of overexploitation (Demestre et al. 1997) and this dominance of young specimens in landings makes the stock highly vulnerable to recruitment changes (Tserpes et al. 2002). Therefore, it would be advisable to protect red mullet nursery and spawning areas. Actually, Fiorentino et al. (2008) reported an increase in the number of recruits and a wider recruitment period after a trawl ban in the Gulf of Castellammare, exemplifying that an implementation of a permanently closed area does benefit reproduction success of this species. Moreover, a benthic community in a permanent closed area will have the possibility to recover from trawling impact, which might benefit not only red mullet but also other commercial species (e.g. the structured soft-bottom community observed in the Medes Island MPA, de Juan et al. 2011).

This work shows that changes in the effort regime within a year only had limited consequences for benthic community structure, whereas changes between a non-fished control site and a fished site were clearly evident. The observed changes at the fished site might benefit adult red mullet, as their food provision will be higher due to an increase in medium-large infauna and to lower interspecific trophic competition. However, red mullet recruits will be negatively affected by functional changes caused by fishing as their food provision might overall decrease, although they could benefit from a short-term increase in food production during summer. Moreover, both adults and recruits will suffer from lack of protection of habitat structures. Thus, the overall effect of trawling on the red mullet stock, considering the high fishing pressure on recruits and the indirect negative effects caused by ecosystem disturbance, could be a decrease in the spawning stock that will worsen the recruit's stock situation.

This study highlights the idea that permanent closure areas, which would allow recovery of the benthic ecosystem, restructuring habitats and communities, might be more beneficial for commercial species and their habitats than temporary closures.

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