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Priorities for fisheries in marine protected area design and management: Implications for artisanal-type fisheries as found in southern Europe

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Summary

Much has been written in recent years regarding the advantages of marine protected areas (MPAs) as conservation tools. The benefits to fisheries have commonly been cited as primary motives in favour of the establishment of MPAs. To date, a good deal has been theorised with regard to the benefit of MPAs to fisheries in their adjacent areas, but there has been little empirical evidence to support or refute hypothetical claims. Considerations for fisheries' benefits are different to those of ecological benefits in several respects. Economically, fishers' livelihoods often depend on the marine reserve being successful. It is not enough to establish that populations of fish are growing due to protection; stocks, as well as individual fish have to be sufficiently large to be catchable by the industry. Furthermore, restrictions in fishable area ought to be compensated for by increases in catches over time. In terms of the biology of the fish themselves, evidence has shown that heavily exploited commercial fish stocks can take much longer to recover from over-exploitation than previously expected. Although there have been several studies that consider the effects of export and spill-over, there have been few that focus on the patterns that these phenomena might have on the surrounding fisheries; many assume that ecological patterns will manifest in the fishery with time. Recently, assessment methods and predictive models have been suggested for fisheries (e.g. Rapfish, Ecopath/Ecosim), some of which have been adapted specifically for MPAs. In this paper we review recent progress in the field of MPA research with particular focus on fisheries assessment. We also identify priorities, and knowledge gaps, for determining and accurately predicting the benefits of MPAs to fishers.

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Introduction

Marine protected areas (MPAs) have received a lot of attention in recent times with regard to their benefits in ecological conservation. Although fisheries are regularly declared to benefit from MPAs (e.g. Lauck et al. 1998; Pomeroy et al. 2005; Russ et al. 2004) there has been a paucity of evidence to this effect (Russ, 2002). More attention, however, has been given in recent years to the status of fisheries resources in terms of empirical investigations (e.g. Claudet et al., 2008; Côté et al. 2001; Sale et al., 2005) and the benefits that MPAs might provide to overexploited fish stocks.

MPA theory is based on some very basic ecological principles, for example, that areas of the natural environment will return to a richer, more natural state if human interference is removed or limited. Success of an MPA is measured by comparing the protected area with either data from the same area before protection was established (e.g. De Loma et al., 2008; Dufour et al. 1995; Pastoors et al. 2000), or by making comparisons with geographically separate but similar environments that are under pressure of human interference (e.g. Kamukuru et al., 2004; Maliao et al. 2004; Pérez-Ruzafa et al. 2006). Of course, the success of a protected region depends very much on the state of the area before protection was put in place as well as the underlying dynamics of the system in question. Under disturbance theory, it is too simplistic to assume that an area highly impacted by disturbance will "recover" to its prior state, particularly where a variety of possible stable states exists (Connell, 1978). Consequently, it is not uncommon for MPAs to evolve in an unexpected manner in terms of the number of species and densities of individuals within its boundaries once protection has been established (e.g. Dufour et al., 1995). The time involved in recovery is also difficult to predict and evidence has shown that in extreme cases ecosystems and species populations, in particular fish stocks, can take much longer than expected to recover from over-exploitation (Hutchings, 2000; Micheli et al. 2004). Jouvenel et al. (2004) for example, showed that recovery of fish populations in a Mediterranean marine park was still underway after five years of protection but full recovery, to the extent where fisheries begin to benefit, is likely to take a much greater amount of time.

A large proportion of succession research has focused on the terrestrial realm (Young et al. 2001) but whether terrestrial or marine, there are factors that need to be taken into consideration even after the source of disturbance has been removed or mitigated. In either the marine or terrestrial realm, recovery of an ecosystem can continue to be vulnerable to succession-limiting factors such as recruitment limitation (where the quantity of available propagules for restoration is restricted, which may happen if fish populations are driven below their maximum sustainable yield (MSY)) and dispersal limitation (the inability of potentially dominant species to arrive at the protected site thus altering the ability of a site to recover) (see Young et al., 2001 and references therein). Further complications arise where the interspecific relationships within the ecosystem change (Gascuel et al. 2005).

Marine reserve theory differs from terrestrial reserve theory, in terms of modelling at least, in that the former is predisposed to focus on preserving species richness or particular habitat while the latter gives greater attention to the protection of specific species (Gerber et al., 2003). In the marine realm, stakeholders have tended to take a short-term perspective, hoping to see benefits from marine protection within the first five years (Airamé et al., 2003). Unrealistic expectations are liable to present themselves especially when fisheries management or enhancement is stated as an objective of MPAs; an aim that was noted as occurring in 53% of the 30 case studies reviewed by Boersma and Parrish (1999). With regard to the aims of southern European MPAs, although fisheries enhancement has been commonly cited as a protection priority (Table 1, also see Francour et al. 2001), much of the supporting research has focused on assessing conservation status (e.g. Claudet et al., 2006; [17]Claudet et al., 2008; Dufour et al., 1995; Jouvenel et al., 2004). In contrast, northern European MPAs have taken the form of fish boxes or fisheries closures (see Hoffman & Vestergaard, 2006 for a review), and the associated research has been more specifically fisheries oriented (Crean & Wisher, 2000; Pastoors et al., 2000; Rijnsdorp, 1998; Rijnsdorp et al. 2001).

How MPAs might contribute to fisheries management

Advantages of marine protection to fisheries have focused on two key concepts – "spill-over" and "export" (Gell & Roberts, 2003; Russ, 2002). The former suggests that as fish populations within the core marine reserve (where fishing is assumed to be prohibited for the sake of theory) increase, adult fish will naturally migrate or spill-over into surrounding areas. The second assumes that when

Table 1. Characteristics of some southern European MPAs selected for consideration in the EMPAFISH project

Sites/MPA	Country	Coastal/Island	Year of creation (regulations modified)	Total size MPA (integral size) (ha)	Reserve size (ha)	Protection objectives
Cabo de Palos	Spain	Mixed inshore	1995	1898	270	Fisheries enhancement
Cerbére/Banyuls	France	Coastal	1974	617.4	65	Conservation
Côte Bleue (Carry-le-Rouet & Cap Couronne)	France	Coastal inshore and offshore	1983, 1996	85/210	85/210	Protection and restoration of natural habitats, fisheries enhancement
Medes Islands	Spain	Inshore island	1983	511	93	Conservation
Columbretes Islands	Spain	Mid-continental Shelf	1990	4400	1862.6	Fisheries enhancement
Sinis Mal di Ventre	Italy	Mixed	1997	25,673	529	Environmental protection
Bouches de Bonifacio	France	Mixed	1998	80,000	1200	Environmental protection
San Antonio	Spain	Coastal	1993	250	250	Fisheries enhancement
Serra Gelada–Benidorm Islets	Spain	Coastal including islets	2005	4920	–	Protection of biodiversity
Tabarca	Spain	Island	1986	1400	120	Fisheries enhancement
Anti-trawling zones in SE Spain[a]	Spain	Coastal	1989, 1996	20–800	–	Fisheries enhancement, protection of sea grasses
La Graciosa	Spain	Islands inshore and offshore	1995	70,700	1225	Fisheries enhancement, conservation
La Restinga	Spain	Islands inshore and offshore	1996	750	180	Fisheries enhancement, conservation
Monte da Guia	Portugal	Island inshore and offshore	1980 (2000)	443	10	Conservation, scientific research, tourism
Formigas Islets/Dollabarat Bank	Portugal	Offshore islands	1988 (2003)	52,527	52,527	Conservation, scientific research, tourism
Gulf of Castellammare	Italy	Mixed inshore/offshore	1990	20,000	–	Fisheries enhancement
Malta fisheries management zone	Malta	Island inshore and offshore	1971 (2004)	6735		
Tuscany Archipelago	Italy	Island	1996	56,766	6147.4	Conservation, tourism
Ustica	Italy	Island	1986	15,951	60	Environmental protection

Adapted from Vandeperre et al. (2006).

[a]The Spanish anti-trawling zone consists of several reefs that were established at different times and span a variety of habitats and depths.

protected individuals reach maturity and spawn, their eggs and larvae will be carried in the water column to unprotected regions, supporting and enhancing populations outside the marine reserve boundary that may not have the same density of spawning adults. Export is notoriously difficult to estimate since dispersal characteristics and the scale at which dispersal occurs is still largely unknown for many species (Carr & Reed, 1992; Gell & Roberts, 2003; Paddock & Estes, 2000). Even spill-over, which has been seen to occur, can be extremely individual, species and/or habitat specific (Codling, 2008; Murawski et al. 2005). Further, spill-over effects are often limited in instances where site fidelity is demonstrated by adult fish, a phenomenon that occurs particularly in coastal zones (Topping et al. 2005); MPAs are frequently located in these areas. In the case of both spill-over and export it is assumed that the marine reserve will act as a reservoir for the surrounding, unprotected, or less-protected waters (Gell & Roberts, 2003; Roberts, 1998).

Although there have been several studies that consider the effects of export and spill-over, there have been few (e.g. Codling, 2008; Harmelin-Vivien et al., 2008; Murawski et al., 2005; Sanchirico & Wilen, 2000; Wilcox & Pomeroy, 2003) that focus on the patterns that phenomena such as spill-over and export might have on the surrounding fisheries (Roberts, 1997). Many studies assume that ecological assessments will manifest in the fishery with time. Others, although they attempt to understand the benefits MPAs offer fisheries, often illustrate the benefits of protection to populations rather than to the fishery itself (Claudet et al., 2006; Harmelin et al. 1995; Rius, 1997). For example, some studies demonstrate the difference between protected fish populations and those of unprotected areas by experimentally fishing inside the no-take zone, albeit with a research permit, but such studies have little relevance to the fishery itself as they simply demonstrate the effects that might be seen in areas that are completely closed, areas which, ironically, will probably never be available to fishermen as fishing grounds. Where spill-over has been evidenced, effects appear to be very local in nature, i.e. within 5 km of the reserve boundary (Goñi et al. 2004; Harmelin-Vivien et al., 2008; Murawski et al., 2005).

In studies focusing on the abundance of species of commercial interest, results have been promising in some southern European reserves. At the French reserve of Cote Bleue, for example, a reserve composed of two sites, Carry-le-Rouet and Cape Couronne, replenishment of fish stocks has been observed to be positive, with increased abundances of the most valuable target species coupled with greater sized individuals (Jouvenel et al., 2004).

Where MPAs have been employed in the form of permanent or temporary fisheries closures, although benefits of closed areas are not a guaranteed success (Pastoors et al., 2000), such closures can improve spawning stock biomass for some species (e.g. Cadrin et al. 1995; Goñi et al. 2001) particularly where closures occur at nursery grounds (Horwood et al. 1998). Closures, even where temporary, can result in increased abundance, size and improved sex-ratios (Goñi et al., 2001). Alternating opening and closing of fishing grounds, however, may prove counter-productive since re-openings have been associated with pulses of increased fishing pressure (Murawski et al., 2005).

Although some proponents of the benefits of MPAs argue that they are indispensable in fisheries management (Gell & Roberts, 2003) the complexities involved in managing fish resources involve an intricate mix of biological, economical, social and cultural issues and MPA approaches are most likely best considered alongside other management measures (Degnbol et al., 2006).

What models tell us

MPA assessment has been addressed in the literature in one of two ways (Petellier et al., 2008): (i) using empirical data and statistical analysis; and, (ii) using dynamic predictive models. Fisheries-specific predictive models have been devised (e.g. Rapfish, Ecopath/Ecosim) (Christensen & Walters, 2004) and even adapted to include the effects of MPAs (Alder et al. 2002; Doyen & Bene, 2003; Mahévas & Pelletier, 2004; Pitcher et al. 2002) but applications of such models to realistically assess the effectiveness of MPAs is lacking. In addition, recent evidence has been conflicting regarding the predicted effects of marine reserves on adjacent fisheries (Sanchirico et al. 2006; White et al. 2008).

Both strategic and tactical models have been developed for MPAs (Gerber et al., 2003). Strategic models explore general design features of MPAs such as size, connectivity and location, while tactical models look at more specific localised considerations for particular regions and/or species. Models have addressed a wide variety of fisheries-related MPA parameters such as: ideal no-take area size for maximising yields (Crowder et al. 2000); population dynamics (Beattie et al. 2002);

connectivity (Hastings & Botsford, 2003); determination of reserve location (source–sink models) (Crowder et al., 2000); fishing effort allocation; spill-over/export (Guénette, Pitcher, & Walters, 2000; Pérez-Ruzafa et al., in press); and, the movements of individuals between protected and unprotected waters (Codling, 2008). For a concise review of fisheries models see Pelletier and Mahévas (2005). Despite this, models suffer in some instances since they aim to be general, grossly overestimating the amount of area needed for protection, for example (Dahlgren & Sobel, 2000; Pezzey et al. 2000). Where specific situations are modelled (e.g. Pelletier & Mahévas, 2005) favourable outcomes are achieved more efficiently. Petellier et al. (2008) highlight the scale dependency of most MPA evaluation approaches. Small-scale in-depth studies are bound by observation limitation but allow for a complete understanding of snapshots from an entire community, e.g. through experimental fishing or underwater visual census (UVC). Larger-scale studies are usually better evaluators of ecosystem health and sustainability of practices and resources but are often restricted to a single species, community of fishery (Doyen & Bene, 2003).

Tactically, one of the main effects of establishing a marine reserve or MPA is that access is immediately restricted to an area that was previously open to fishing. In this respect fishing effort is either reduced or dislocated to other areas (Stelzenmüller et al., 2008). Modelling has indicated that the benefits to fish stocks by effort restriction through MPAs are similar to those that can arise due to traditional fisheries management (Hastings & Botsford, 1999). In the majority of models reviewed by Gerber et al. (2003) marine reserves were predicted to benefit fisheries in cases where fishing effort was high enough before protection to significantly reduce recruitment levels, although relocation of effort was not considered in most models. In some cases, where fishing effort is low, models have indicated that fishing productivity will be reduced when marine reserves are large (Holland & Brazee, 1996). Regardless, the regulation of effort and the level to which restrictions are imposed are crucial to the commercial productivity of MPAs (Kritzer, 2004).

Strategic models have gone some way to informing us about the factors that might best favour one management approach over another. Guénette and Pitcher (1999), for example, have illustrated the effects of stock enhancement both within and without MPAs where fishing intensity approaches MSY, reserve benefits being linked somewhat to the size of the area they encompass and the size of the spawning population they protect. Similarly, Doyen and Bene (2003) demonstrated that MPAs can promote sustainability of fisheries resources where 25% of the stock area is protected, highlighting the importance of accounting for uncertainty in MPA allocation.

Methods of evaluating the performance of MPAs have also been elucidated with the aid of computer models (e.g. Alder et al., 2002). Petellier et al. (2008) compiled a series of indicators, from empirical and model-based literature, that can be used to test the effectiveness of MPAs according to different time-scales. Many models designed specifically for marine reserves are relatively recent (Gerber et al., 2003). The range of existing models is explored in some detail in Gerber et al. (2003).

When evaluating reserves in terms of the returns in the form of profits to the fishing industry, reserves designed to spatially encompass the larval dispersal distance of key species were found to be most profitable to the industry (White et al., 2008). Interestingly, profits were consistently higher where marine reserves were managed optimally compared with optimally managed quota-system management (White et al., 2008). It is very interesting that even sub-optimal reserve management, allowing for escapement and including effort regulation, provided equivalent or improved profits to commercial fisheries.

In any event, the advantages of MPAs, like any other management tool, are dependent on the level of compliance and implementation (Kritzer, 2004). It remains doubtful, however, whether MPAs offer real benefits to fisheries in cases where fisheries can be effectively managed by more traditional means (Holland & Brazee, 1996; Pelletier et al., 2005; Russ, 2002; Willis et al. 2003). Micheli et al. (2004), for example, showed that a significant proportion of species, up to a third in some cases, showed negative impacts of protection. Further, for Mediterranean fishes, Guidetti and Sala (2007) showed that trophic cascades can be strengthened inside marine reserves, particularly where species feeding on urchins are favoured.

Where models have been applied to actual fisheries (e.g. Drouineau et al. 2006) methods of increasing yields were determined. Although the methods for increasing catches are species specific, in the case of *Nephrops norvegicus*, Drouineau et al. (2006) identified management strategies, including MPAs that could potentially increase yields within a 10-year period. Some studies have shown that reserves are most likely to improve harvest biomass only in regions that have been previously heavily over-exploited (Holland &

Brazee, 1996; Smith & Wilen, 2003). In fact, models that do not aim to impose marine reserve criteria but rather seeking ways to maximise fisheries harvests have found that at least the inclusion of some no-take area gives the optimum harvest results (Neubert, 2003).

Evidence from empirical research: the case for southern Europe

The aim of the European Marine Protected Areas as tools for Fisheries Management and Conservation (EMPAFISH) project was to focus on fisheries in southern Europe (Stelzenmüller et al., 2008; Vandeperre et al., 2006). MPAs in southern Europe, particularly in the Mediterranean region, share some aspects in common, a significant number of which are sited in shallow rocky coastal areas (Harmelin, 2000) and or in insular regions (Francour et al., 2001) (also see Table 1). Many of the MPAs located in this area are older (Badalamenti et al. 2000 and references therein), smaller reserves that have similar conservation objectives (Vandeperre et al., 2006), and MPAs are a popular management tool across the entire Mediterranean region (Badalamenti et al., 2000; Harmelin, 2000, Rius, 1997). In several cases, the fisheries associated with these MPAs share some common factors too, often being small-scale, low-tech fisheries with similar target species (Goñi et al., 2004; Harmelin, 2000; Vandeperre et al., 2006). Given that the Mediterranean area is composed of a variety of sovereign states, there are a number of management and enforcement strategies in place in the region (Badalamenti et al., 2000; Francour et al., 2001), as such, generalisations across not just the Mediterranean, but southern Europe as a whole, should be treated with caution.

In a large-scale study of MPAs in southern Europe, Claudet et al (2008) demonstrated the importance of reserve size and the amount of time elapsed since the implementation of protection in an ecological study of the effects of MPAs on commercial species. This research showed that the density of fish within the marine reserve was proportional to the size of the no-take reserve but was inversely proportional to the size of the restricted-take area. This essentially corroborates that "edge effects" occur, and occur more strongly when the reserve area is small or productivity is low relative to the size of the boundary (Bartholomew et al., 2008; Kellner et al. 2007; Pérez-Ruzafa et al., in press), a result that was verified by research performed under the EMPAFISH project (Claudet et al., 2008).

Empirical research tends to focus on the benefits of MPAs to the ecosystem or to populations of particular species. For example, Halpern (2003) demonstrated that biomass inside a reserve is higher than in surrounding areas. Similar findings have emerged through other research (Garcia-Charton et al., 2004, Goñi et al. 2006; Harmelin-Vivien et al., 2008; Pelletier et al., 2005; Russ et al., 2004). In terms of southern Europe, in particular, Harmelin et al. (1995) showed that the abundance of species commonly targeted by local fisheries in the region of Carry-le-Rouet reserve in France were significantly higher. Dufour et al. (1995) however found this to be less obvious at the Cerbére-Banyuls reserve. In a study of several Mediterranean MPAs, Harmelin-Vivien et al. (2008) detected a gradient of fish biomass that decreased from more-protected to less-protected areas, supporting the contribution of spill-over to fisheries resources.

Contrary to previous studies, Claudet et al. (2008) found that larger no-take areas result in greater abundances of commercial species. Not surprisingly, many studies have found that when fishing pressure is removed, greater abundances are found inside marine reserves compared to adjacent fished areas (Claudet et al., 2008; Ojeda-Martinez et al. 2007; Paddock & Estes, 2000; Rius, 1997). Mean size of individuals was also greater inside protected areas in some cases (Garcia-Rubies & Zabala, 1990). Such results are useful but do not estimate the contribution these improved resources make to adjacent fisheries. There are few studies to date, of which we are aware, that attempt to evaluate the benefits seen in the local fisheries (Badalamenti et al., 2000; Vandeperre et al., 2006; Vandeperre et al., unpublished data).

Interestingly, Claudet et al. (2008) also noted that there was no significant effect of protection on smaller individuals (juveniles) of commercially caught species within reserves as compared with outside. With regard to larval biomass, Mediterranean studies have shown levels to be high in the area of Medes Islands MPA in both protected and unprotected areas of the sea (Sabates et al. 2003). Adult biomass, on the other hand, was sometimes found to be lower for larger individuals in unprotected areas, varying according to habitat characteristics (Macpherson et al. 2002).

Mediterranean examples have illustrated phenomena that are the converse of what might be expected from a protected population. Planes et al. (2000) for example, indicated that recruitment success within MPAs can be equal or even lower than without, an occurrence which is more likely due to increased abundance and size of predators (Macpherson et al., 2002). Macpherson et al. (1997) showed that recruitment rates from a variety of Mediterranean MPAs spanning France, Spain and Italy demonstrated no significant difference in recruitment within protected zones compared with unprotected zones, mortality rates being similar in both protected and reference sites, refuting to some degree the theory of MPAs acting as ecological sinks or receiving areas for recruits (Roberts, 1998).

These examples illustrate the complexity and heterogeneity of the marine environment in the Mediterranean and its confounding influence on measuring the effects of protection. Protection effects have been far from clear-cut, with other local factors being seen to contribution to variations in biomass and fish densities (Macpherson et al., 2002; Sabates et al., 2003).

Evidence of benefits of MPAs to fisheries

One of the current interests of marine protection in southern Europe is that of reducing and controlling trawling activity. In several regions of south eastern Spain (Ramos-Esplá et al. 2000) and Italy (Relini et al. 2007), for example, artificial reefs have been deployed with the twin objective of preventing trawling and increasing habitat surface area for the accommodation of vulnerable fish species (Caddy, 2000; Jensen, 2002). At least for some trawl ban areas, these increases were also not found to be accompanied by altered trophic structures between the reserve and unprotected areas (Badalamenti et al., 2000, 2008).

Although there is evidence that reserves promote improved fisheries resources, i.e. increased abundances of commercial species (Claudet et al., 2008; Halpern, 2003; Mosqueira et al. 2000), there is little evidence of the contributions these make to the surrounding fisheries (for exceptions see: Goñi et al., 2004; McClanahan & Kaunda-Arara, 1996; McClanahan & Mangi, 2000). Declines in several Mediterranean species have been noted in recent decades (Briand, 2000; Stergiou et al. 1997), manifesting both in recreational fisheries as well as in professional exploits (Coll et al., 2004).

Work by Micheli et al. (2004) indicates that the benefits of MPAs to target species of commercial fish is greater than for non-target, showing increased abundances and higher trophic levels inside protected areas. Similar work by Mosqueira et al. (2000) corroborated evidence of increases in abundances of target species. In a large-scale study of southern European MPAs, Claudet et al. (2008) also identified increased abundances of typically targeted individuals, both in terms of species and size range within reserves, a pattern that was not seen for non-target commercial and non-commercial individuals.

However, there remains a need for further small-scale investigations of effort intensity and (re) distribution (e.g. Murawski et al., 2005; Ragnarsson & Steingrimsson, 2003, Stelzenmüller et al., 2008; Wilcox & Pomeroy, 2003). Concentration of effort around the reserve boundary has been a common phenomenon in MPA research (Goñi et al., 2004; Murawski et al., 2005; Wilcox & Pomeroy, 2003). Stelzenmüller et al. (2008) detected similar patterns across selected EMPAFISH protected areas. Although one study demonstrated that the revenue per unit effort was found to be higher in the areas immediately adjacent to reserve boundaries, catches varied more greatly than in surrounding areas (Murawski et al., 2005).

Considerations for future management of low-tech fisheries

One problem with MPAs is that "success" or "effectiveness" criteria and methods of evaluation are not often set (Alder et al., 2002). Evaluating the effectiveness of an MPA in terms of whether resources have increased, decreased or stabilised is useful, but it is not a measure of sustainability of the relevant populations, or the ecosystem as a whole (Boesch, 1999). High-quality evaluation of fisheries effects of MPAs needs to take a two-pronged approach; monitoring of the fishery in addition to monitoring of the resource. In practice, the information necessary to monitor the fishery has proved limiting to the management of fish stocks for decades since obtaining accurate catch information from the fishing vessels involved is tricky (Francour et al., 2001). Particularly, with regard to Mediterranean artisanal fishers, Himes (2003) has shown how fishermen feel distanced from the management process. It is crucial, therefore, to involve fishers in conservation strategies.

In order to determine the status of the industry as a response to marine protection, information required corresponds to that needed for ecological studies, i.e. detailed records of catches from before and after implementation of the MPA. Unfortunately, in many cases, fisheries data for small-scale, low-tech fisheries lack spatial references. Knowledge of spatial aspects of a fleet's behaviour not only allows for greater insight into effort concentration (Murawski et al., 2005; Stelzenmüller et al., 2007; Stelzenmüller et al., 2008; Wilcox & Pomeroy, 2003) but also allows for confounding factors such as natural habitat variation, or the "habitat effect" (Bayle-Sempere & Ramos-Esplá, 2003; Garcia-Charton & Pérez-Ruzafa, 1999; Macpherson et al., 2002) to be taken into account. Habitat complexity and its influence on ecosystem processes has only recently begun to be understood (Levin, 1992). Garcia-Charton and Pérez-Ruzafa (1999) emphasise the importance of considering the underlying habitat complexity when designing such evaluative studies.

In terms of fisheries priorities, it is important to monitor the health of fish stocks in order to manage fisheries resources for the future. In an uncertain environment where species have proved more vulnerable than previously imagined (e.g. Hutchings, 2000), such monitoring and research is crucial. With respect to MPAs, where stocks are often over-exploited from the outset or close to depletion, such research is pressing. It is common, however, and also unhelpful to confuse stock assessment studies with evaluations of the effect of MPAs on fisheries. Of course the health of the fishery resource is crucial to the health of the fishery but a healthy resource is not necessarily exhibited as increased catches and landings to the fisherman, particularly where the resource is present in a no-take zone (e.g. Goñi et al., 2004; Horwood et al., 1998; Micheli et al., 2004). It is prudent, therefore, to understand the trends in catches and landings that manifest in actual fisheries around reserves before leaning too heavily on claims that they will benefit from closed regions. In terms of fisheries, the sacrifices made in the name of marine protection are higher than those in other sectors. When a portion of a traditional fishing ground is closed or restricted fishermen suffer a reduction in spatial resources and, usually, forced relocation (Murawski et al., 2005). In order for fisheries to benefit from marine protection, therefore, augmentation of stocks in surrounding areas due to spill-over and export must match or exceed the volume of the resource lost through the reduction in fishing area, which unfortunately is not always the case (McClanahan & Mangi, 2000).

Research on the success of marine protection must go hand in hand with more fisheries-specific investigations in order, for example to determine which fishing strategies will offer some level of sustainability to recovering stocks (e.g. Goñi et al., 2003). Although there is a paucity of research on MPAs in northern Europe in comparison to southern Europe, approaches to MPA evaluation in the latter could benefit from some of the approaches taken in the north. In southern Europe, evaluation of MPAs is often carried out using ecological sampling methods such as UVC (e.g. Claudet et al., 2008; Garcia-Charton et al., 2004; Garcia-Rubies & Zabala, 1990). Using fisheries logbook data (Pastoors et al., 2000) or onboard sampling of catches, commercially caught or via experimental fishing (Badalamenti et al., 2008; Goñi et al., 2003; Goñi, Quetglas, & Renones, 2004), would not only focus the evaluation of MPAs on the benefits of fisheries specifically but would also bring research strategies across Europe into alignment.

Finally, management and surveillance remains a troublesome issue (Francour et al., 2001) and this needs to be dealt with in order to ensure that there is a true protected area in place before attempting to evaluate such sites.

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