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North East Atlantic vs. Mediterranean Marine Protected Areas as Fisheries Management Tool

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The effectiveness of management initiatives implemented in the context of the European Common Fisheries Policy has been questioned, especially with regard to the Mediterranean. Some of the analyses made to compare the fishing activity and management measures carried out in the North East Atlantic and in the Mediterranean do not take into account some of the differentiating peculiarities of each of these regions. At the same time, they resort to traditional fisheries management measures and do not discuss the role of marine protected areas as a complementary management tool. In this respect, the apparent failure of marine protected areas in the North-East Atlantic compared with the same in the Mediterranean is challenging European fishery scientists. Application of the classical holistic view of ecological succession to the functioning of fishery closures and no-use areas highlights the importance of combining both management regimes to fully satisfy both fishery- and biodiversity-oriented goals. We advocate that an optimal management strategy for designing an MPA to protect biodiversity and sustain fishing yields consists of combining a network of no-use areas (close to their mature state) with fish boxes (buffer zones maintained by fishing disturbance in a relatively early successional stage, where productivity is higher), under a multi-zoning scheme. In this framework, the importance of no-use areas for fisheries is based on several observations: (1) They preserve biological diversity at regional scale, at all levels—specific, habitat/seascape, and also genetic diversity and the structure of populations, allowing natural selection to operate. (2) They permit the natural variability of the system to be differentiated from the effects of regulation and to be integrated in appropriate sampling schemes as controls. (3) They maintain the natural size and age structure of the populations, hence maximizing potential fecundity, allowing biomass export to occur from core to regulated areas, dampening the fluctuations derived from deviations from the theoretical optimal effort in the fishing zone.

Keywords: North East Atlantic, Mediterranean, Marine Protected Areas, fisheries, management

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INTRODUCTION

The effectiveness of management initiatives implemented in the context of the European Common Fisheries Policy (CFP), operative since 1984, has been put into doubt and the fisheries management system in Europe has been strongly criticized (Froese, 2011a,b). Immediately, providing data on the evolution of some traditional fisheries, Cardinale (2011) countered those claims. Since then, attention has focused particularly on the situation in the Mediterranean Sea, where the alarming decline of its fish stocks is a matter of increasing concern (Vasilakopoulos et al., 2014; Cardinale and Scarcella, 2017). This is especially evident when comparing the effectiveness of fisheries management in the North East Atlantic and the Mediterranean. According to these reviews, NE Atlantic fish stocks have been gradually recovering as a result of the decrease in fishing pressure following implementation of the EU's CFP during the past decade (Cardinale, 2011), while European Mediterranean fish stocks seem to be out of control, and regulations are often poorly enforced (Vasilakopoulos et al., 2014; Cardinale and Scarcella, 2017).

Most of the above mentioned analyses focus on stocks caught by trawlers and purse seines, both characteristic of open waters and relatively deep bottoms, and propose management actions based on traditional fisheries management tools consisting of limiting juvenile exploitation, harvesting species a few years after maturation, and changes in selectivity and exploitation rates (Selig et al., 2017). It is expected that these measures should maximize long-term yields and halt stock depletion (Hilborn and Ovando, 2014), and are in accordance with the EU's CFP, which requires that fish stocks should be exploited at a level that generates the maximum sustainable yield (MSY; European Commission, 2006; Vasilakopoulos et al., 2014).

However, Mediterranean and North East Atlantic regions differ in oceanographic and climatic processes, human impacts, cultural heritage, spatial scales and heterogeneity and size of habitats and populations, that condition the nature of the fisheries (Smith and Garcia, 2014). Therefore, any proposal to improve management strategies should consider other complementary actions, especially taking into account that local, small-scale, coastal, artisanal fisheries constitute an important component of the idiosyncrasy of Mediterranean fisheries. Neither should it be forgotten that an important contribution of this fishing activity is developed in coastal lagoons where stock dynamics and the characteristics of the fishing gears used are difficult to incorporate in traditional approaches to fishing management (Pérez-Ruzafa and Marcos, 2012).

One aspect not explicitly considered in the above mentioned reviews (Vasilakopoulos et al., 2014; Cardinale and Scarcella, 2017) and that should be taken into account is that, after the failure of traditional fisheries management measures (Waters, 1991), marine reserves have been strongly advocated as an ideal tool for the management of coastal fisheries (Plan Development Team, 1990; Roberts and Polunin, 1991; Dugan and Davies, 1993; Agardy, 1994; Gerber et al., 2002). Indeed, analysis of global trends in world fisheries points to the urgency of implementing non-conventional approaches, including the establishment of marine reserves, which enabled the apparent sustainability of pre-industrial fisheries (Pauly et al., 2005). As a result, a large number of marine protected areas (MPAs) have been established in recent decades throughout the world, including the EU (Jones et al., 1993; Lubchenco et al., 2003; Fenberg et al., 2012; Devillers et al., 2015; Batista and Cabral, 2016). Beyond this, "the establishment of MPAs is an important contribution to the achievement of a good marine environmental status" under the European Marine Strategy Framework Directive (Directive 2008/56/EC) and are considered an affordable way to mitigate and promote adaptation to climate change (Roberts et al., 2017).

Formally, the terms marine protected area (MPA) and marine reserve are not exactly synonymous (**Table 1**) and, in fact, there are a large number of conservation entities, with different levels of protection, permitted uses, and management measures.

As a fisheries management tool, a marine reserve is a no-take zone where it is forbidden to extract organisms in any way, except, in some cases, when required for scientific monitoring (Roberts and Hawkins, 2000; Halpern and Warner, 2002).

For its part, the term MPA is a more general concept applied to defined geographical areas, which are recognized, and managed, by law or other effective regulations, to preserve marine ecosystems and their associated ecosystem services and attributes, including biodiversity, species populations, cultural values, or economic resources such as fisheries production (Dudley, 2008; Thomas et al., 2014). Such areas can take a high variety of forms and designs (Planes et al., 2006), denominations (http://oceanservice.noaa.gov/facts/mpa.html), accepted uses (Mazor et al., 2014b) and management objectives at all levels of government and spatial scales (Portman et al., 2012; Giakoumi et al., 2013). In the UK, for example, MPAs include Special Areas of Conservation according to the EU Birds and Habitats Directives, Marine Nature Reserves' and Sites of Special Scientific Interest, the main aim of most of them being conservation of biodiversity, while very few are designed for managing fisheries (Gubbay, 2006). Indeed, throughout the world, the vast majority of MPAs allow fishing and extractive activities, as well as other commercial or recreational practices such as boating or scuba-diving (Thomas et al., 2014).

Despite this multiciplity of objectives, MPAs are viewed in Europe and worldwide as the best way to protect fishing resources and conserve marine biodiversity (Gaines et al., 2010a; Costello and Ballantine, 2015; Lubchenco and Grorud-Colvert, 2015). Many European MPAs have a common objective to keep harvested populations below the overfishing threshold, while maximizing sustainable yields (Fenberg et al., 2012; European Environment Agency, 2015), a second objective being to prevent the loss of biodiversity due to human erosion (Pauly et al., 2002, 2005).

The expected benefits of MPAs include preserving the spawners and the natural size and age structure of populations, maintaining assemblage structure and ecosystem equilibrium, maintaining genetic diversity and facilitating the recovery of stocks in over-exploited areas through the exportation of eggs and larvae to neighboring areas. At the same time they allow the development of research in non-impacted ecosystems that can be used as control areas in experimental sampling designs and as reference conditions for environmental impact and ecological status assessments (García-Charton et al., 2008; Wood et al., 2008; Lester et al., 2009; Fenberg et al., 2012).

Here (1) we provide a comprehensive review of the effects of MPAs as a fisheries management tool, (2) we highlight the differences in their use and expected benefits between Northeastern and Southern-Mediterranean- Europe, and (3) we apply the classical holistic view of ecological succession to the

TABLE 1 | Definitions of the most common existing figures and terminology for the protection of marine areas.

Key term	Definitions
Marine Protected Area	The IUCN, after the more specific initial definition (IUCN, 1988, 1994), actually aligns the meaning of MPA with the definition of a "protected area" as "a clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (Dudley, 2008). More concretely, Marine Protected Areas (MPAs) are "any marine area set aside under legislation to protect marine values". Such values include conservation, commercial, species enhancement, scientific importance, historic, recreational, scenery or aesthetics, cultural, etc. (Day and Roff, 2000). "They come in a variety of forms and denominations, as marine sanctuaries, marine reserves, fully protected marine areas, no take zones, estuarine research reserves, ocean or marine parks, or marine wildlife refuges, which may have different implications according to the country in which they are established" (FAO, 2011; http://oceanservice.noaa.gov/facts/mpa.html).
Multiple-use MPA	"MPAs typically comprise fluid and dynamic marine ecosystems, have a high diversity of habitats and species within an area and contain highly migratory marine species. This complexity often dictates the need for multiple objectives and complex management schemes" (Dudley, 2008). In the marine environment, this is particularly important and zoning is recommended in the IUCN best practice guidelines on MPAs as the best way of ensuring protection and managing multiple-use protected areas (Kelleher, 1999).
Marine Reserve	"Marine reserves are a specific type of MPA that achieves the preservation of the biodiversity and other values in a strictly protected area, where activities that remove animals and plants or alter habitats are prohibited, except as needed for scientific monitoring. They have become an important tool for both marine biodiversity protection and fisheries management" (Roberts and Hawkins, 2000; Dudley, 2008; http://www.protectplanetocean.org/introduction). A marine reserve or "no take" MPA is a highly protected type of MPA where removing or destroying natural or cultural resources is prohibited. (NOAA access 06/06/2017 http://marineprotectedareas.noaa.gov/aboutmpas/). "They usually allow human access and even some uses, but prohibit the extraction or significant destruction of natural and cultural resources. Also coincide with the no-take level of protection established in some whole or multiple-use MPAs, and so they are often called a "no-take" MPA" (http://marineprotectedareas.noaa.gov/pdf/helpful-resources/factsheets/mpa_classification_may2011.pdf; http://www.protectplanetocean.org/collections/introduction/introbox/reserves/introduction-item.html).
No-Take zone	"This protection category is not compatible with any removal of marine species or modification, extraction or collection of marine resources, with scarce exceptions such as scientific research. Human visitation is limited, to ensure preservation of the conservation values. Setting aside strictly protected areas in the marine environment is of fundamental importance, particularly to protect fish breeding and spawning areas and to provide scientific baseline areas that are as undisturbed as possible. They may comprise a whole MPA or frequently be a separate zone within a multiple-use MPA, seen as "cores" surrounded by other suitably protected areas" (Dudley, 2008).
Buffer zone	"Areas around a core protected zone that are managed to help maintain protected area values" (Dudley, 2008). They preserve the entire protected zone from potentially damaging external influences and are essentially transitional areas where appropriate economic activities are permitted and where sustainable resource management practices can be developed (Bennett and Mulongoy, 2006; Dudley, 2008).
Fish Box	"Some sites, such as fish spawning aggregation areas or pelagic migratory routes, are critically important and the species concerned are extremely vulnerable at specific and predictable times of the year, while for the rest of the year they do not need any greater management than surrounding areas. The EU has encouraged the establishment of such conservation "boxes" or "fishery closure areas" within which seasonal, fulltime, temporary or permanent controls are placed on fishing methods and/or access" (Dudley, 2008; http://www.protectplanetocean.org/introduction/introbox/glossary/glossary/introduction-item.html#marres).
Network of MPAs	"Set of discrete MPAs within a region or ecosystem that are connected through complementary purposes and synergistic protections, at various spatial scales, and with a range of protection levels designed to meet objectives that a single reserve cannot achieve. A network of MPAs could focus on ecosystem processes, certain individual marine species, or cultural resources. For example, an ecological network of MPAs could be connected through dispersal of reproductive stages or movement of juveniles and adults" (IUCN-WCPA, 2008; http://www.protectplanetocean.org/introduction/introbox/glossary/glossary/introduction-item.html#marres). Connectivity and its scales play here a relevant role.

functioning of fisheries closures and no-use areas, in order to put light into this debate.

A POWERFUL MANAGEMENT TOOL FOR BIODIVERSITY CONSERVATION

The effects of protection on fish structure inside reserves have been demonstrated in numerous studies (e.g., McClanahan and Mangi, 2000; Claudet et al., 2008, 2010; García-Charton et al., 2008; Lester et al., 2009; Guidetti et al., 2014; Sciberras et al., 2015), although there are still several aspects that need confirmation and further efforts must be made to understand the complex mechanisms and processes that clearly produce positive effects in some cases and negative or neutral effects in others (Gaines et al., 2010a; D'agata et al., 2016; Gill et al., 2017). Basically, MPAs are expected to protect critical spawning stock biomass of species from fishery-related depletion (Bell, 1983; García-Charton et al., 2004; Claudet et al., 2006), so that they recover and maintain a natural size and age structure in the populations, hence maximizing potential fecundity, allowing biomass exportation from core to regulated areas (Reñones et al., 1999; Goñi et al., 2003; Brito et al., 2006; Harmelin-Vivien et al., 2007; Dimech et al., 2008; Hackradt et al., 2014; Di Lorenzo et al., 2016). They also reestablish ecological interactions (Guidetti, 2006a,b) and preserve biological diversity at a regional scale (but see Klein et al., 2015) at all levels—specific, habitat/seascape, and also genetic diversity and populations structure (Pérez-Ruzafa et al., 2006)—, allowing the force of natural selection to operate.

These effects can take place in a relatively short time. In a review of 80 reserves, Halpern and Warner (2002) found that most assemblage descriptors, such as diversity, density, average organism size, and biomass inside the reserves, reach levels comparable to control areas within the first 1–3 years and are maintained in reserves for up to 40 years. Other empirical studies show that the time taken to detect the first direct effects on target species is around 5 years, while the detection of indirect effects on other taxa takes 10–15 years (Babcock et al., 2010) or more (Claudet et al., 2008), a significantly longer time but still short enough to be perceived at the scale of human perception of changes, and within the 5–20 years time-horizons used in cost-benefit analysis or community planning (Tonn et al., 2006; Hunt and Taylor, 2009; Pahl et al., 2014).

MPAS AS FISHERIES MANAGEMENT TOOL

The interest of MPAs as fishery management tool lies mostly on their potential to improve artisanal fisheries of high-value species in surrounding fishing grounds (Kerwath et al., 2013; Di Franco et al., 2016; Lloret et al., 2017). At the same time, they intend to favor the economic development of the area through the establishment of services related to tourism and diving activities providing an alternative source of inputs also for fishermen (Angulo-Valdés and Hatcher, 2010; Rees et al., 2015; Pascual et al., 2016). The key question for fisheries management is whether MPAs are effective only inside the protected area or whether they are also useful for maintaining productivity in the surrounding exploited grounds.

MECHANISMS OF EXPORTATION

Marine fish dispersal that would benefit fisheries in the vicinity of an MPA may occur via several mechanisms: egg and larval dispersal (Cowen et al., 2000; Abesamis et al., 2016, 2017), the trophic or reproductive migrations of adults (Green et al., 2015), nomadic or ontogenetic movements (Grüss et al., 2011), and, density-independent, home-range movements by individuals across reserve boundaries and home-range relocation because of density-dependent factors (Rakitin and Kramer, 1996; Russ and Alcala, 1996; Kramer and Chapman, 1999; Pérez-Ruzafa et al., 2008). Although, all these mechanisms potentially increase the catch in surrounding areas, and hence likely benefit fisheries by helping to recover and maintain target populations, full benefits of MPAs would require protecting also other mechanisms determining reproductive success, such as those necessitating essential habitats (Elliott et al., 2016)—e.g., spawning grounds (Erisman et al., 2015; Sadovy de Mitcheson, 2016) and recruitment areas (Cheminée et al., 2017). On the other hand, some authors differentiate "ecological spillover" (i.e., the net export of juvenile, subadult and adult biomass from MPAs outwardly driven by density-dependent processes) from "fishery spillover" (i.e., the proportion of this biomass that can be fished, taking into account regulations and accessibility; Di Lorenzo et al., 2016).

Although, some of these processes are difficult to demonstrate in the field (Harmelin-Vivien et al., 2008; Lester et al., 2009; Di Lorenzo et al., 2016), the ability of MPAs to improve adjacent fisheries has also been related to indirect evidence, such as spatial redistribution of fishing effort (Murawski et al., 2005; Goñi et al., 2008; Stelzenmüller et al., 2008; Cabral et al., 2016) and the direct recording of catches by commercial fishing (Vandeperre et al., 2011).

In the last decade, modeling approaches have also analyzed the exportation of individuals on the basis of diffusive processes dependent on density gradients (Gerber et al., 2003; Neubert, 2003; Kellner et al., 2007; Pérez-Ruzafa et al., 2008), which is consistent with observed patterns in fish abundance and fishing effort distribution. Other models incorporate movement patterns—migration, ontogenetic movements, etc. (e.g., as those reviewed by Grüss, 2014).

Protection also produces a rapid response in fishermen's behavior. From the first year, fishing effort and "catch per unit effort" (CPUE) tend to concentrate at the boundaries of the MPAs and gradually increase with time. Using a meta-analytical approach to investigate the effects of protection on adjacent fisheries based on 28 data sets from seven southern European MPAs, Vandeperre et al. (2011) found clear effects on the surrounding fisheries, both as regards the CPUE of the target species and, especially, on the CPUE of the marketable catch.

These effects mainly depend on the time of protection and on the size of the no-take area. The CPUE of both the marketable catch and target species increased gradually by 2–4% per year over a long period (at least 30 years). On average, the protected areas provided catches about 2.4 higher than those from the non-protected areas (Vandeperre et al., 2006).

Despite this quick response, the effects can be hidden by fishermen's behavior. Up to 75% of fishing gears can be deployed within 1 km of the MPA (Murawski et al., 2004, 2005) and most catches in other studies were concentrated within 10 km from the reserve boundaries (Kellner et al., 2007; Goñi et al., 2008; Stelzenmüller et al., 2008). The rapid concentration of fishing effort near the boundaries of the reserve causes a rapid fall in CPUE (Pérez-Ruzafa et al., 2008; Vandeperre et al., 2011), so that a successive increase/decrease/recover of yields ultimately leads to pre-reserve levels being exceeded (Hopf et al., 2016). However, in very small MPAs or when the protected area is located peripherally within the metapopulation, the recovery time runs from several years to decades and reserves are sometimes unable to support the increased mortality so that the metapopulation collapses (Hopf et al., 2016).

Although, some empirical studies showed that the spatial response of fishing boats to the implementation of an MPA greatly depends on the local particularities (fishing modalities, spatial distribution of habitats, prevailing winds, currents regime, etc.), precluding an oversimplified assumption about redistribution of fishing effort (Cabral et al., 2016), usually, the relocation of fishing effort leads to a "fishing the line" harvesting tactic (Kellner et al., 2007).

WHO BENEFITS MOST FROM MPAS?

MPAs are not only a good fisheries management tool. The economic revenues of MPAs have been claimed as being beneficial for all stakeholder groups, especially in southern European MPAs, where the demand for diving activities increases exponentially and fishing and scuba-diving are the main coexisting uses (Roncin et al., 2008; Fenberg et al., 2012; Sala et al., 2013). A broad socio-economic field survey covering 12 case studies in southern Europe showed a variety of situations, from MPAs where commercial fishing is the major economic stake (up to 88% of total incomes locally generated by fishing and diving), to MPAs where recreational activities have a dominant economic role (where incomes generated by commercial fishing can amount to <5% of the money generated by scuba diving; Roncin et al., 2008). In general, recreational diving is prevailing in coastal areas, while fishing is more associated with MPAs far from the coast, where diving activities are difficult (Roncin et al., 2008). On average, the amount of locally generated income by fishing and diving in the southern European MPAs studied by Roncin et al. (2008) represented at least 2.3 times the management costs.

According to Mangi and Austen (2008), there is a high level of satisfaction among users of marine reserves. However, most stakeholders consider MPAs that have been established for longer periods of time offer greater benefits for conservation than fisheries and that such conservation benefits gradually increase with the time of protected area management. Of note is the fact that the main reasons given by divers to justify the use of MPAs as diving sites are fish abundance and the presence of some spectacular or "emblematic" species (e.g., grouper in the Mediterranean). However, the evaluation of MPAs as areas to benefit fisheries decreased with the time an area is protected due to a gradual change in fishermen's perception.

The belief amongst fishermen in the potential of MPAs to deliver fisheries objectives declines with time, coinciding with the time taken by the relocation process of fishing effort to reach a "fishing the line" equilibrium. After this time, all fishermen in a gradient from the MPA boundary would attach the same CPUE, explaining why after this time the perception of fishermen could become more neutral, while the perception of benefits for divers and diving operators is still increasing.

FISH BOX VS. MPAS

Despite the benefits provided by marine reserves, after more than 25 years of research into their effectiveness, fishery managers question European scientists about the apparent failure of north European MPAs to reach the expected fisheries objectives (Pastoors et al., 2000; Daw and Gray, 2005; Beare et al., 2010). Moreover, if we compare them with Mediterranean MPAs, which are generally viewed by researchers and, more importantly, by users as being successful (Guidetti et al., 2014), the question arises as to the possible causes of such divergence.

In classical fisheries science, it is increasingly assumed that the concept of MSY, as a reference for the catch rate, should be reinterpreted as an upper limit rather than a management target, by replacing the traditional goal of maximizing the catch by that of maximizing economic profit or even minimizing the impact of fishing (Froese et al., 2011, 2016a,b), although this is not being implemented yet in national and EU fishing policies (Marchal et al., 2016). This requires the introduction of a range of management tools that permits an overall reduction in exploitation rates. Implementing the best management tools may depend on the local context, but there is an overall view that a combination of traditional fisheries management measures (catch quotas, closed seasons, community management) coupled to strategically placed fishing closures, ocean zoning, increased selectivity of fishing gear, and economic incentives, holds much promise for the restoration of marine fisheries and ecosystems (Worm et al., 2009; Froese et al., 2015). In theory, except for sedentary species, for which reserves have important advantages, the management of fisheries through setting up reserves and management through effort control may produce identical yields under a reasonable set of simplifying assumptions corresponding to a broad range of biological conditions (Hastings and Botsford, 1999).

In Europe, MPA design differs between Atlantic and Mediterranean areas (**Figure 1**). Northern MPAs (the so-called fish boxes or fisheries closures; Pastoors et al., 2000) are generally very large (hundreds of thousands of hectares), and are intended to protect one or a few target or by-catch species (e.g., plaice, sole, cod, herring, sprat, haddock). In this case, fisheries regulations typically consist of traditional fishing regulations, banning specific gears, and/or reducing the fishing effort within the whole closed area. For their part, Mediterranean MPAs (Planes et al., 2006; Fenberg et al., 2012) are usually small (hundreds of hectares or less; Gabrié et al., 2012; Portman et al., 2012), and are in general located in areas that are biologically unique, because of the occurrence of remarkably diverse and/or complex habitats; they always include a core marine reserve, a no-use or no-take area (i.e., where all human activity is banned or any type of fishing is absolutely prohibited), and they are often bounded by a buffer area, where some fishing activity is allowed and strongly regulated by traditional approaches. Although, the difference in total size arises as the immediate option to explain differences between Northern and Southern European MPAs, empirical evidence is less clear on this: while some studies argue for large MPAs to ensure their success, other recommend smaller MPAs (Claudet et al., 2008; Fletcher et al., 2015; Hughes et al., 2016).

The proponents of an ecosystem approach to fisheries management are still struggling with finding practical ways of application (Fulton et al., 2014; Jennings et al., 2014; Link and Browman, 2014; Berg et al., 2015; Coll et al., 2015; Long et al., 2015; Patrick and Link, 2015). Most initiatives so far are based on the notion that less diverse communities are less productive (Gamfeldt and Hillebrand, 2008; Tilman et al., 2014; Strong et al., 2015). But, as Margalef (1997) pointed out [embracing Odum (1969) holistic view of succession], decreasing productivity to biomass ratios is the most likely driver of ecosystem growth when moving from early to more mature stages, once an initial "squandering" phase has been surpassed (**Figure 2**). This pattern generally does not take into account the mechanisms determining the actual sequence of replacement and the rate

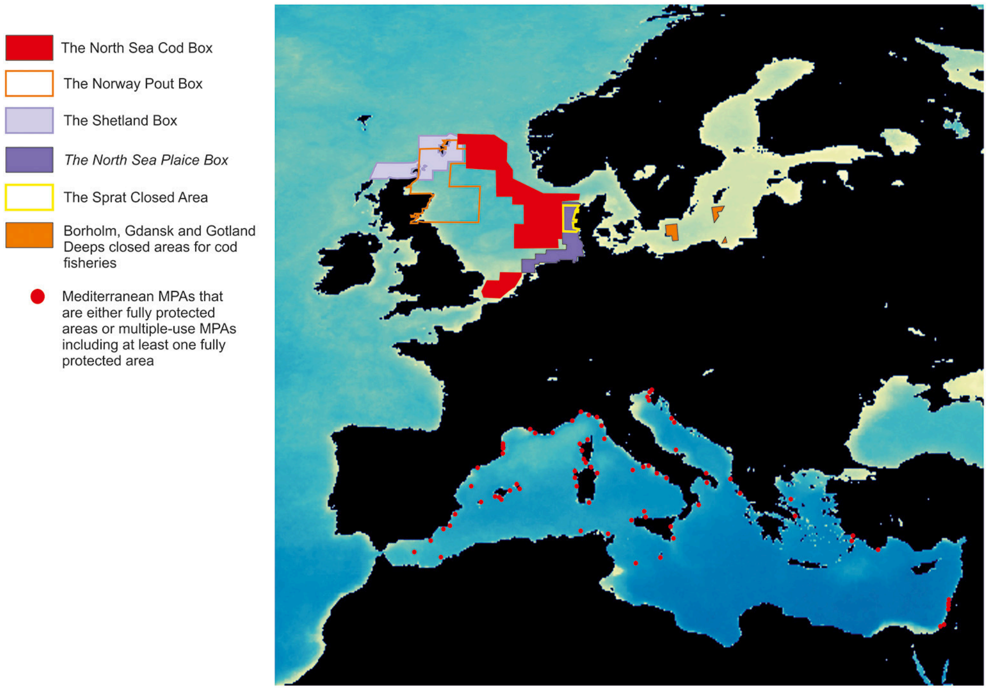

FIGURE 1 | Main fish closures in the North and Baltic Seas (Sørensen, 2006) and distribution of present Mediterranean MPAs in April 2017 (MAPAMED, 2017; http://www.medpan.org/en/mapamed). Base map corresponds to averaged maximums of Chlorophyll a for the period 2003–2010 from satellite data.

of the same (Valiela, 1995). In this context, the objectives of fishery measures of maintaining a MSY, which are those of most Northern MPAs, can only be attained in a relatively early phase of succession, when net production is maximized (**Figure 2**). The position of this stage in the succession will vary depending on the life cycle, trophic level or life-history strategy (*r* vs. *K*) (or, more generally, the average trophic level) of the species forming the catch. But this situation is achieved in a narrow successional fringe-, so that overexploitation will lead to a decrease in productivity, although the same occurs if regulatory measures move the ecosystem excessively toward more mature stages. The latter situation leads to the apparent paradox that, contrary to expectations, fishing limitations reduce fishing yields in fish boxes when more harvested (i.e., stressed by fishing disturbance, and then kept at a younger successional stage) are compared with partially protected areas, making the failure of protection measures more probable.

By contrast, Mediterranean MPAs enhance fishing yields through spillover from core no-take areas to neighboring, regulated and unprotected areas (Gell and Roberts, 2003; Tudela et al., 2005; Stelzenmüller et al., 2008) after the protected ecosystem is left to reach a state of maturity (**Figure 2**). A late successional stage is characterized by more species, longer-lived organisms, and complex food webs with a higher number of trophic levels. This mature stage is inevitable, as succession is a process of self-organization (Fath et al., 2004). Empirical studies in Mediterranean MPAs (García-Charton et al., 2004; Tudela et al., 2005; Coll et al., 2013; García-Rubies et al., 2013; Guidetti et al., 2014) showed how fish communities undergo a huge increase in target biomass—usually piscivore species within core marine reserve areas, and restoring a more "natural" community structure, provided that they are well enforced (Guidetti et al., 2008; Di Franco et al., 2016). All this leads to the general view that Mediterranean MPAs are almost always successful in achieving the planned objectives.

The above holistic ecological considerations lead us to advocate that an optimal management strategy for designing an MPA (also in the Northern Atlantic) to protect biodiversity and sustain fishing yields would be a combination of a network of no-use marine reserve areas (close to their mature state) with fish boxes (maintained by fishing disturbance in a relatively early successional stage, where productivity is higher), under a multi-zoning scheme.

OPTIMUM DESIGN: MAXIMIZING CONSERVATION, EXPORTATION AND MAINTENANCE COSTS

Zonation in an MPA

There is no one design for an MPA. Apart from the size and form, some MPAs consider only a no-take zone or marine reserve where no human activity except scientific surveys is forbidden, others also include one or several buffer zones with different relative sizes with respect to the core reserve where some gears and different degrees of regulated fishing is permitted, and other buffer zones are entirely regulated like the fish box (Planes et al., 2006; Horta e Costa et al., 2016).

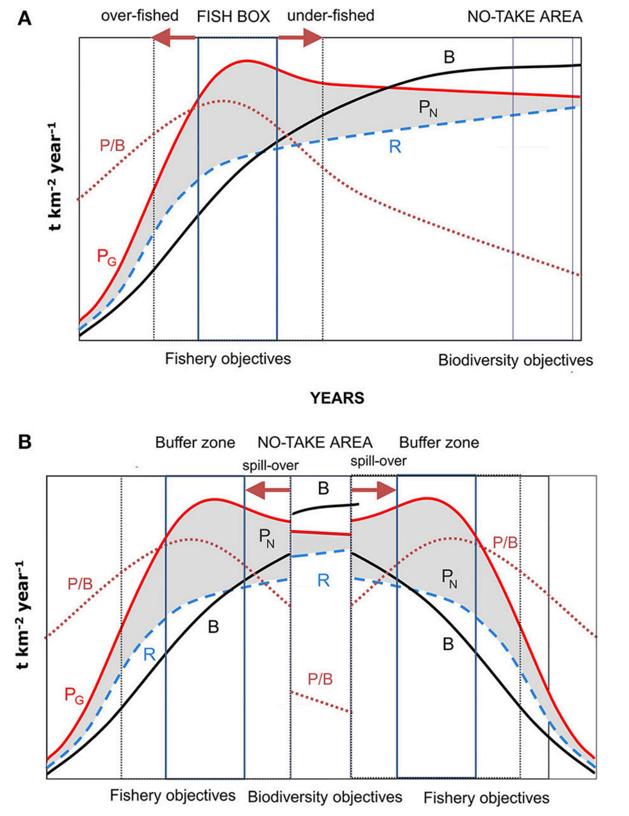

FIGURE 2 | (A) Energetic of succession in an ecosystem. P_G, gross production; P_N, net production; R, total community respiration; B, total biomass; P/B, productivity (Modified from Odum's, 1969). The fishing activity (and the objectives of fish boxes) focuses on the early stages of ecological succession, where net production is maximum. But any deviation from the optimum of fishing pressure moves the ecosystem to stages with lower net production. By contrast, conservation objectives focus on late stages, where maximum biomass is reached and the ecosystem reaches maturity and complexity, but net exploitable production is minimal. **(B)** Southern European MPAs have a central no-take area whereby ecological succession leads the ecosystem to maturity, flanked by buffer zones managed like the fish box. The export of biomass from the no take area to the buffer zone dampens the catch fluctuations derived from deviations from optimal effort in the fishing zone.

The design affects MPA effectiveness and, contrary to what might be expected, larger buffer areas seem to negatively affect both ecological aspects and fishing yields in the surrounding areas (Claudet et al., 2008; Vandeperre et al., 2011). Although, no explanation has been proposed for this, it could be related to the fact that the spill-over spatial scale is much reduced. Enlarging the buffer area implies that, although subjected to fishing regulations, the real effects of protection due to the no take zone became diluted in a longer gradient, making differences between the buffer and the free fishing areas less pronounced than with smaller buffer zones.

How Big Should an MPA Be? Size Do Matter But...

Meta-analyses performed on 58 datasets from 19 Southern European MPAs showed that the size of the no-take zone and time of protection are the main factors determining the effectiveness of an MPA for preserving the abundance and size structure of fish assemblages (Claudet et al., 2008). Similar conclusions are reached when the effects on fisheries are analyzed considering CPUE data (Vandeperre et al., 2011).

Pérez-Ruzafa et al. (2008), using a modeling approach, showed that medium size to large marine reserves (>600 ha) are to be preferred to small ones, both to maximize the protection of fish populations abundance and to improve the exportation of biomass across the marine reserve boundaries (Pérez-Ruzafa et al., 2008), thus agreeing with empirical data. Larger reserves attain between 80 and 100% of the carrying capacity in the first 5 years after protection depending on fish mobility or fishing effort in the surrounding area. However, reserves smaller than 500 m in radius have difficulties in attaining 50% of the carrying capacity, especially in the case of fishes of high mobility and subject to high fishing effort in neighboring areas (Pérez-Ruzafa et al., 2008). Furthermore, improvement in maintaining higher populations abundance or in exporting individuals increase very slowly for reserves larger than 1,500 ha (Pérez-Ruzafa et al., 2008).

The spatial scale of influence of a marine reserve is lower than might be expected. Some reviews of demersal fish and invertebrates suggest adult neighborhood sizes ranging from a few kilometres to 10 to 100 km, and for larval dispersal, of 10 to 100 km for invertebrates and 50 to 200 km for fish (Palumbi, 2004). Recent studies (e.g., D'Aloia et al., 2015; Green et al., 2015) reports even shorter average dispersal distance (<5–15 km) and highlight that self-recruitment is a common phenomenon. Field studies based on underwater visual censuses have shown that the gradient in fish abundance due to spill-over through the reserve boundary is only detectable at small spatial scales (a few 100 m; Harmelin-Vivien et al., 2008). In modeling approaches, the effect of the flux of individuals from the reserve to the fished areas is evident at <5–10 km from the boundary (Pérez-Ruzafa et al., 2008). In the case of larvae, Jessopp and McAllen (2007) found limited exchange between reserve and non-reserve areas at a relatively small spatial scale (<3 km) depending on the hydrographic and geomorphologic characteristics of the sites. In the case of models for larval exportation, changes in yield biomass during the first year are evident within 14 km of the reserve (Hopf et al., 2016).

Furthermore, although size does matter, any improvement in populations abundance or in exporting individuals decreases very quickly for reserves larger than 1,500 ha (Pérez-Ruzafa et al., 2008). Even, although strongly contested by other authors (Hughes et al., 2016), the usefulness of excessively large reserves has been questioned for improving fishery performance (Fletcher et al., 2015).

At the same time, the effectiveness of a MPA is highly dependent on users' compliance with regulations, and non-compliance is often the rule rather than the exception (Arias and Sutton, 2013; Bergseth et al., 2015, 2017; Arias et al., 2016). This makes proper surveillance and enforcement to prevent or reduce poaching to a minimum are essential in any marine reserve (Davis et al., 2004; McCauley et al., 2016). Enforcement constitutes one of the main costs in the maintenance of an MPA, something that is especially evident in MPAs far from land and in the case of recently proposed and declared mega-parks of more than 250,000 km^2 for which the development and use of relatively expensive next-generation enforcement, such as satellite and drone based patrols will be necessary (McCauley, 2014; Arias et al., 2016; McCauley et al., 2016).

Therefore, it is clear that for any marine conservation plan to be feasible and efficient it is necessary to include the costs derived both from its implementation and maintenance (Mazor et al., 2014a). The socio-economic analyses performed by Alban et al. (2008), which considered all the costs involved in MPA management for the 12 case studies considered by Roncin et al. (2008), showed that, as could be expected, total management costs increase with reserve size, but total cost per ha is minimum for integral marine reserves of between 600 and 1,500 ha (1,400–2,200 m radius).

Taking into account all these ecological, fishing and economic considerations, it may be concluded that the optimum size for marine reserves to improve conservation and fishing yields at minimum cost, would be between 600 and 1,500 ha.

Designing Networks of MPAs

It has been estimated that, to maintain biodiversity, at least a 20–30% of the ocean should be protected (Morgan, 2014; Pressey et al., 2014). At present, only about 3.6% of the ocean has some kind of protection, and just 1.6% is covered by strongly or fully protected (i.e., "no-take" reserves; Lubchenco and Grorud-Colvert, 2015); furthermore, only 2.1% of existing MPAs are actively managed (Sala et al., 2016), i.e., more than 'paper parks'. Increasing coastal area or coastal lengths under protection can be done increasing the size of marine reserves. However, as mentioned above very large marine reserves can be effective for preserving biodiversity and can have higher resilience, but are controversial as a fisheries management tool, while surveillance costs increase exponentially. A good alternative would be to establish MPA networks (Gaines et al., 2010b; Grorud-Colvert et al., 2014; Bode et al., 2016). After the results highlighted above, a network of marine reserves of around 600 ha each separated by tens of kilometres would optimize the balance between conservation efficiency and maintenance costs and have a synergistic effect on the export of biomass in fishing areas between reserves. In addition, the capacity increases and recovery rates of regional, well interconnected stocks after protection is even faster than in individual stocks, with some studies reporting that fish densities recovered in 1.5–2 years after rezoning (Russ et al., 2008).

One important aspect to consider in designing reserve networks is connectivity. Ensuring that reserve populations are connected and maintain genetic fluxes between each other and with non-reserve populations through larval or adult dispersal allows for recovery from disturbance and is a key aspect in resilience (Almany et al., 2009; Calò et al., 2013). However, it is also necessary to optimize trade-offs between connectivity and representation objectives, including species and habitat diversity, while minimizing the risk that multiple reserves will be impacted by catastrophic events (Almany et al., 2009). Spatio-temporal heterogeneity can play an important role in the emergence of homeostatic mechanisms of complex coastal ecosystems (Pérez-Ruzafa et al., 2005) and the introduction of restrictions to connectivity can develop complex structures that enhance biodiversity at genetic and taxonomic levels (Pérez-Ruzafa, 2015).

CONCLUSIONS

In just over a decade, and although many aspects remain to be investigated, much has been accomplished since the first proposals for the creation of marine reserve networks, from the uncertainties that existed (Roberts, 2000; Roberts et al., 2001; Botsford et al., 2003), to our present knowledge of how they function and the effects they have on ecological processes, fisheries and the economic activity.

The appropiate design of MPA networks allows conservation and fishery objectives to be coupled and can provide ecological and recreational services of value (Roberts et al., 2003).

Despite the fact that fisheries in both North East Atlantic and the Mediterranean Sea are governed by the European CFP, great discrepancies in performance have been observed, with recent considerable improvements in stock status in the North East Atlantic being matched by a rapidly deteriorating situation in the Mediterranean region. The control of fishing effort combined with specific technical measures, such as gear regulation, the establishment of a minimum conservation reference size and selective closure of areas and seasons, is the main management strategy adopted by Mediterranean EU countries, while Total Allowable Catches is the major regulatory mechanism in the North East Atlantic (Cardinale and Scarcella, 2017). However, these analyses focus on species fished by trawling generally at greater depths than is usual for MPAs and do not consider local shallow water fisheries, which are the most traditional in the Mediterranean.

By contrast, Mediterranean MPAs work not only by increasing fishing yields, but also by promoting the economy based on recreational diving, tourism, and meeting the objectives of biodiversity conservation, while the Fish Box of northern Europe does not reah spectations.

The holistic ecological considerations discussed above lead us to advocate that an optimal management strategy for designing an MPA (also in the Northern Atlantic) to protect biodiversity and sustain fishing yields consists of combining a network of no-use areas (close to their mature state) with fish boxes (buffer zone maintained by fishing disturbance in a relatively early successional stage, where productivity is higher), under a multi-zoning scheme. In this framework, the importance of no-use areas for fisheries is based on several observations:

- They preserve biological diversity at regional scale, at all levels—specific, habitat/seascape, and also genetic diversity and the structure of populations (Pérez-Ruzafa et al., 2006), allowing natural selection to operate.
- They permit the natural variability of the system to be differentiated from the effects of regulation and to be integrated in appropriate sampling schemes as controls.
- They maintain the natural size and age structure of the populations, hence maximizing potential fecundity, allowing

biomass export to occur from core to regulated areas, dampening the fluctuations derived from deviations from the optimal effort in the fishing zone.

Proper surveillance and enforcement, which prevents or reduces poaching to a minimum, is essential in any marine reserve to ensure its proper functioning and to introduce corrective management measures. At this point, fishermen themselves, as one of the main beneficiaries of the proper functioning of marine reserves, can play an important role in conservation and sustainable practices in a co-managed framework (Claudet and Guidetti, 2010; Hogg et al., 2013).

To evaluate the degree of success of such multi-zone MPAs, we need a suite of fishery and ecological management indicators for both fish boxes and no-use areas. Such indicators must provide a key for convergence of the recovering community in fish boxes to be examined with respect to a mature stage in a given envelope of environmental conditions, using the no-use areas as reference points. Several holistic indicators are usable (Greenstreet and Rogers, 2006; Salas et al., 2006; Shin and Shannon, 2010; Teixeira et al., 2016), including diversity measures, thermodynamically-derived functions (Jørgensen and Mejer, 1979), tropho-dynamic indexes (Tudela et al., 2005), and others more or less specific for exploited marine ecosystems (Shin et al., 2010). Research actions are urgently needed to define those suites of indicators capable of establishing the appropriate reference levels to guide management strategies to recover the increasingly collapsed fish populations in our seas (Rossberg et al., 2017).

We advocate that the optimum size of no-take zones would range between 600 and 1,500 ha, while the size of each zone within the MPA should be scaled to maximize the size of the no-take area to the detriment of buffer zones (about half the size of the no-take area). Any further improvement should come from a network of several MPAs, taking into account that the effects on fisheries improves when the distance between MPAs is no greater than a few tens of kilometres. Such a design fully meets the objective of protecting at least a 20–30% of the coastal area to maintain biodiversity. For pelagic and offshore species, spatial scales should be expanded and larger reserves may be needed, but with the same basic design model. In these cases, surveillance becomes more expensive and new methods, including technological ones, that are effective at the lowest possible cost will have to be developed, also taking into account that international, transboundary collaboration will increase conservation efficiency (Kark et al., 2009; Jay et al., 2016).

AUTHOR CONTRIBUTIONS

All authors listed have made a substantial, direct and intellectual contribution to the work, and approved it for publication.

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