

2

Ecological Effects and Benefits of Mediterranean Marine Protected Areas: Management Implications

Antoni Garcia-Rubies¹, Emma Cebrian^{1,2}, Patrick J. Schembri³, Julian Evans³ and Enrique Macpherson¹

¹Centre d'Estudis Avançats de Blanes (CEAB-CSIC), Girona, Spain

²Department of Environmental Sciences, University of Montilivi, Girona, Spain

³Department of Biology, Faculty of Science, University of Malta, Msida, Malta

Introduction

There is general consensus among scientists that marine life in the Mediterranean Sea, and in the world's oceans in general, is under considerable threat by human activities (Coll *et al.*, 2010; Micheli *et al.*, 2013). This strain on marine ecosystems worldwide has led to calls for new management approaches, especially for coastal areas (Botsford *et al.*, 1997). Such measures, for instance, include the regulation of fisheries towards more sustainable exploitation of resources and the establishment of networks of Marine Protected Areas (MPAs) (Olsen *et al.*, 2013). However, development of effective regulations for conservation must be based on sufficient knowledge and information about the protected systems. For example, it has been shown that the establishment of marine reserves that are too small or too scattered can have a reduced or nil effect on the protection of Mediterranean ecosystems (Abdulla *et al.*, 2008; Botsford *et al.*, 2009).

When appropriately designed, MPAs favour the recovery of harvested populations in the Mediterranean Sea and

elsewhere (Bell, 1983; Garcia-Rubies and Zabala, 1990; Harmelin *et al.*, 1995; Vacchi *et al.*, 1998; Claudet *et al.*, 2011; Fenberg *et al.*, 2012). The main reason for these MPA effects is the drastic reduction in overall mortality: when fishing mortality is removed or reduced, stock recovery is the most logical expected consequence (Bell, 1983). The more vulnerable to fishing a species is, the more it will respond to cessation of fishing mortality (Macpherson *et al.*, 2000). Therefore, the ecological benefits derived from these conservation units are essential for the sustainability of exploited ecosystems, and sagacious and effective management of MPAs is a key issue in an age of changing oceans and seas (Olsen *et al.*, 2013).

Ecological Benefits of MPAs

Effects on Fish Populations

Species vulnerability to fishing depends on the specific life history of the species involved (Molloy *et al.*, 2008). In general, large, long-lived, slow-growing, sedentary

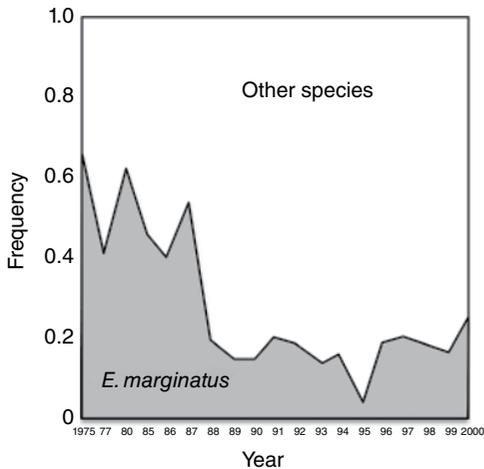


Figure 2.1 Temporal pattern of the relative frequency of species among the largest specimens captured in the regional competitions of spear-fishing in the Balearic Islands. It can be seen that the dusky grouper *Epinephelus marginatus* lost its preponderance among the biggest specimens from the end of the 1980s. Source: Redrawn from Coll *et al.* (2004).

species with low natural mortality rates are more vulnerable than small, fast-growing species with high rates of natural mortality (Cheung *et al.*, 2005). Other features, such as a late sexual maturity, a delayed sex change or a limited reproductive capability, can also exacerbate the vulnerability of a given species. Furthermore, large predatory species tend to be more targeted by fisheries than smaller species, so their populations are much more likely to be depleted in fished areas (Figure 2.1) or by special fishing methods.

The recovery process of an exploited population inside an MPA is the reverse of the process of harvesting. Initial recovery rates can be relatively fast (Garcia-Charton *et al.*, 2008; Molloy *et al.*, 2009) (Figure 2.2), but total recovery can be extremely slow for the more vulnerable species. In many cases, attaining full recovery in an MPA may even be impossible within a human lifespan (e.g. the red coral *Corallium rubrum*; Garrabou

and Harmelin, 2002). Nonetheless, species targeted by fishing generally respond to protection in a positive way compared to non-targeted species (Micheli *et al.*, 2005; Claudet *et al.*, 2006; Guidetti and Sala, 2007), leading to a net increase in biomass. However, not all targeted species respond equally to protection, with response depending on their vulnerability and also on the carrying capacity of the system.

When a population reaches the carrying capacity (K), it can be considered to have fully recovered. Although this is one of the main objectives of MPAs, it has rarely been observed in practice and there is no consensus on successful K thresholds or on the time that is necessary to achieve full recovery. There are many differing descriptions in the literature, ranging from quick recoveries in less than five years (Côté *et al.*, 2001; Halpern and Warner, 2002) to estimated recovery times of 10–40 years for apex predator species (McClanahan *et al.*, 2007) in no-take zones.

Total recovery of harvested populations in MPAs has only recently been described in the Mediterranean Sea (Coll *et al.*, 2013; Garcia-Rubies *et al.*, 2013), in spite of the large number of Mediterranean MPAs. This is probably due to the relatively young age of most of these MPAs; however, it can also be attributed to the lack of long-term studies on the changes in protected populations in most MPAs.

Total recovery can vary greatly with time since protection was implemented. For instance, Coll *et al.* (2012, 2013) showed very fast recoveries of total target fish biomass in three Balearic MPAs. Garcia-Rubies *et al.* (2013), however, observed that reaching the carrying capacity could be a long process for highly vulnerable, long-lived species such as *Dicentrarchus labrax* (20–25 years), *Diplodus cervinus* (13–16 years) and *Epinephelus marginatus* (21–24 years). Other species were still far from achieving total recovery, for example *Sciaena umbra*

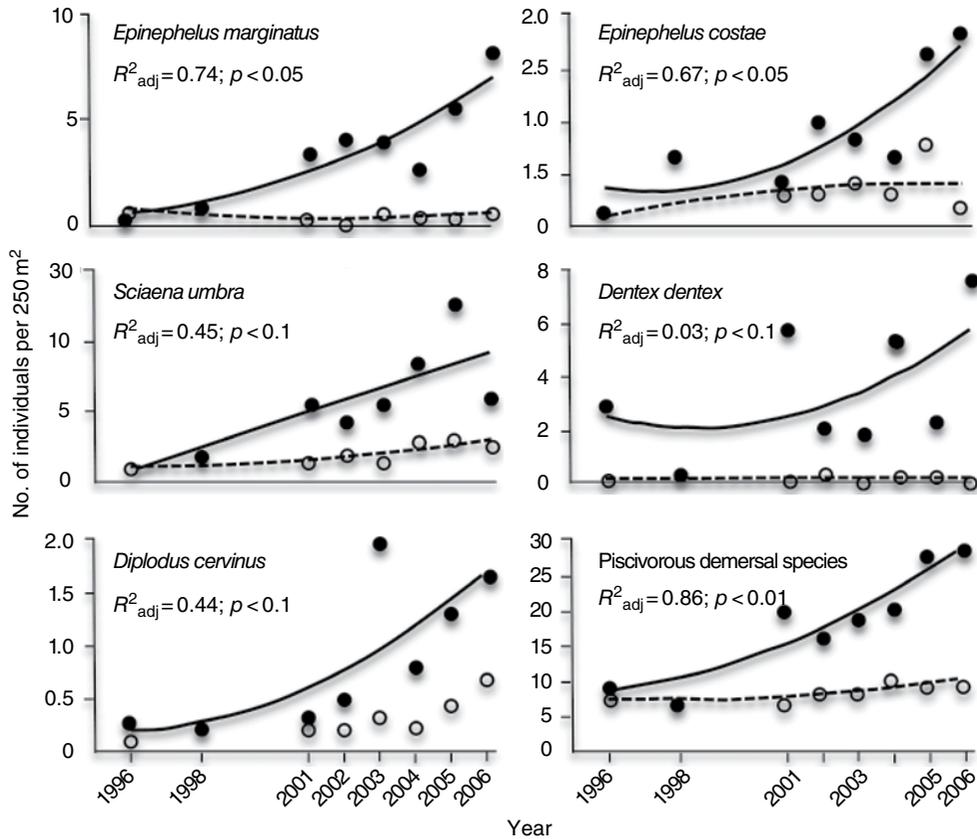


Figure 2.2 Temporal pattern of mean abundances of some large predatory species, and considering all piscivorous species together, in the Cabo de Palos – Islas Hormigas marine reserve, Cartagena, Spain (solid circles) and in an exploited control locality (open circles) after the establishment of protection measures in 1995. Although the first stages of recovery are fast in protected areas, total recovery is a long process. Source: Redrawn from Garcia-Charton *et al.* (2008).

(31–51 years), while *Dentex dentex* was still growing exponentially (see Figure 2.3).

Variations in carrying capacity values, and in the time it takes to reach them, can be explained by the effect of different environmental factors acting at small and medium scales. Achieving the maximum biomass, and the time to reach it, is a bottom-up regulated process influenced by environmental conditions (bottom features/substratum type, depth, slope and rugosity; Coll *et al.*, 2012, 2013) that favour or limit the development of the largest, long-lived species. This explains why the carrying capacity value is greater, and

the time to reach it is longer, in MPAs where environmental conditions are highly favourable (such as the Medes Islands MPA), compared to other MPAs sited in areas lacking these highly favourable conditions (Figure 2.4). Knowing the effect of these factors, one can predict how long it would take an ideal MPA to reach maximum values of K (Coll *et al.*, 2012, 2013). This ideal environment is very similar to that found in the Medes Islands marine reserve where, indeed, the value of K far exceeds that observed in the Balearic Islands MPAs, although the time required to achieve these values is much longer.

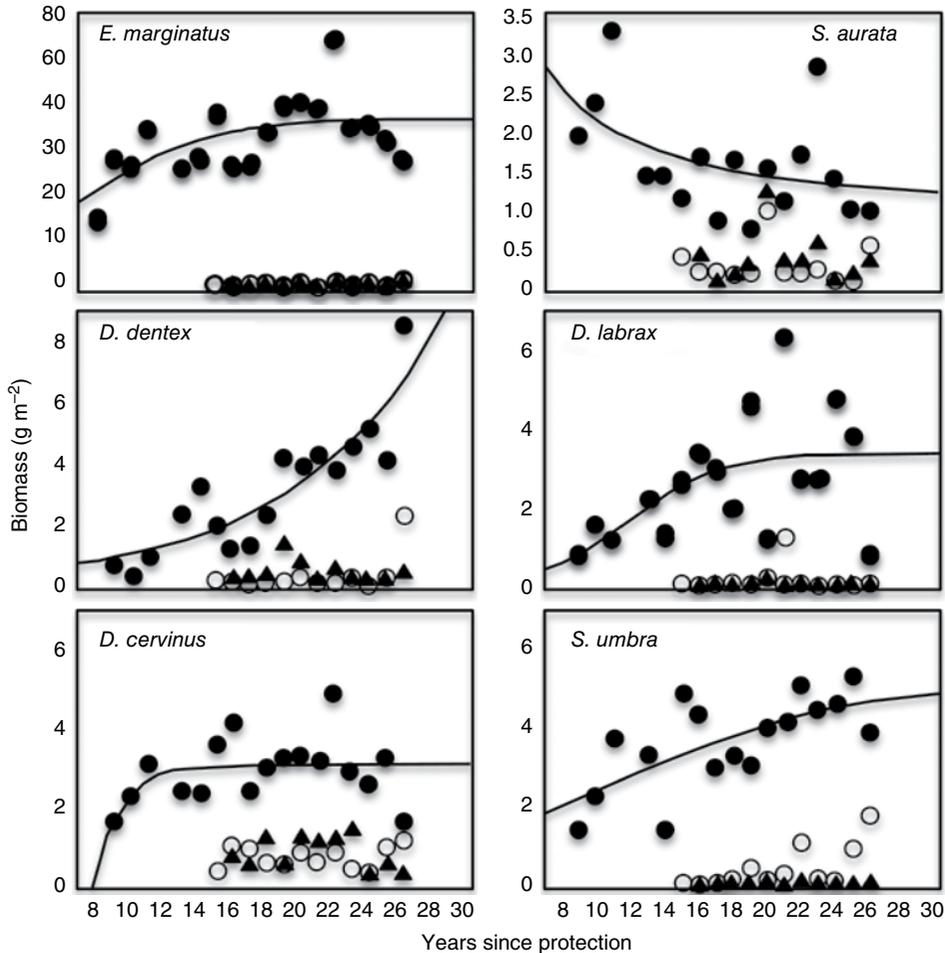


Figure 2.3 Differences in temporal patterns of biomass of six highly vulnerable fish species (*Epinephelus marginatus*, *Sparus aurata*, *Dentex dentex*, *Dicentrarchus labrax*, *Diplodus cervinus* and *Sciaena umbra*) monitored vs. time of protection for marine reserve (solid circles), partially protected reserve (open circles) and non-reserve (solid triangles), in the Medes Islands marine reserve and neighbouring coast. Source: Redrawn from Garcia-Rubies *et al.* (2013).

Ecosystem Characteristics Affecting the Benefits of MPAs

Differences in carrying capacity are just one example of the wide range of results obtained from studies of different Mediterranean MPAs; ecological effects of Mediterranean MPAs have been found to vary in both magnitude and direction (Claudet *et al.*, 2011 and references therein). Although major differences between Mediterranean MPAs could

be attributed to the level of enforcement of, and compliance with, the protection measures, which is a significant socio-cultural factor (Guidetti *et al.*, 2008; Sala *et al.*, 2012), the results can be very different even between well-protected areas. As an example, Sala *et al.* (2012) used a large range of fish biomass (from 50 to 120 g m⁻²) as the reference for a good conservation state for different well-enforced no-take areas. Such variation

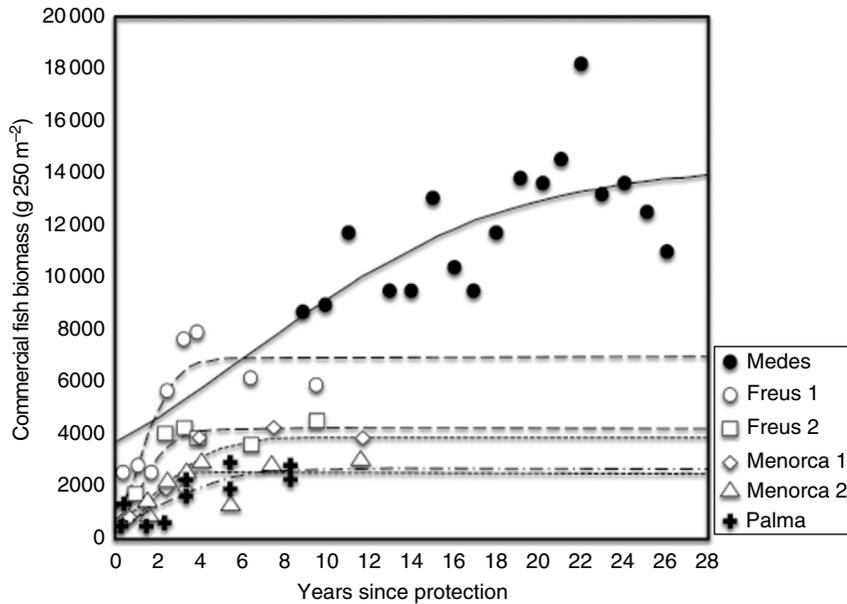


Figure 2.4 The differences in carrying capacity and recovery time between the Medes Islands MPA and three Balearic MPAs are evident despite the fact that the biomass of only six species were taken into account in the Medes Islands, whereas in the Balearic Islands MPAs total commercial fish biomass was included.

Source: Redrawn from Coll *et al.* (2012) and Garcia-Rubies *et al.* (2013).

was attributed to the idiosyncrasies of the different MPAs, that is to say: there must be factors other than good enforcement that define fish biomass in any given well-protected no-take zone (Figure 2.5).

Some obvious factors such as age and size of MPAs have not been taken into account until recently (Guidetti and Sala, 2007; Claudet *et al.*, 2008; Molloy *et al.*, 2009). The effects of these factors have been mainly assessed indirectly by comparing MPAs of different ages through meta-analysis (Guidetti and Sala, 2007; Claudet *et al.*, 2008), whereas studies comparing a temporal evolution of single MPAs, or differences between different-sized coetaneous MPAs, are practically non-existent (but see Garcia-Rubies *et al.*, 2013 and Coll *et al.*, 2012, 2013). Age and size of the MPAs are among the main factors affecting the results of protection. Guidetti and Sala (2007) found that the

response of fish assemblages to protection was significantly related to reserve age only when evaluated at functional level, whereas reserve size did not appear to influence fish assemblages in terms of either species or functional level. In contrast, Claudet *et al.* (2008) found that the age of an MPA was less important than its size, and the size of the buffer zone, in determining commercial fish density in 12 Mediterranean MPAs, although commercial fish density increased at a rate of 8.3% per year in no-take protected zones.

Other factors such as depth range have rarely been taken into account in studies comparing protected and non-protected zones. In most studies, sampling depth is typically fixed within a narrow range, and no assessment of how protection effects vary with depth is carried out.

The role of environmental factors seems to be fundamental in explaining the effects

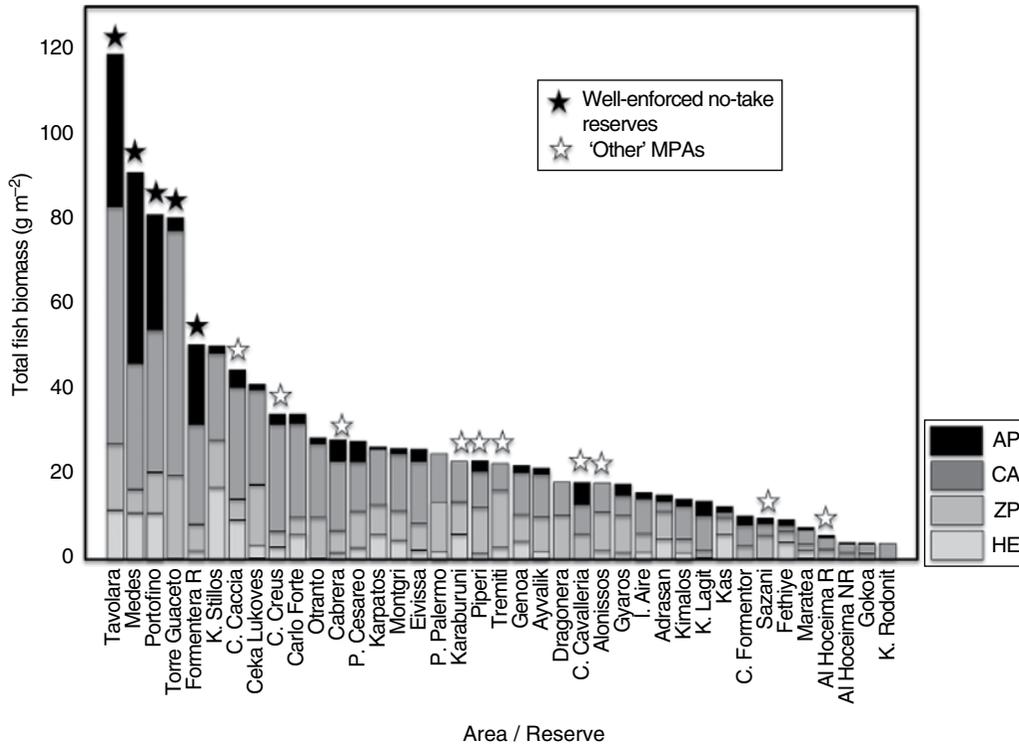


Figure 2.5 Total fish biomass in several MPAs and areas open to fishing in the Mediterranean Sea. The differences between well-enforced no-take reserves and other MPAs are obvious, but the differences between different well-enforced protected areas are also striking, indicating that factors other than effective protection have a strong influence on fish biomass. AP, apex predators; CA, carnivores; ZP, zooplanktivores; HE, herbivores and detritivores. *Source:* Redrawn from Sala *et al.* (2012), with some sites added to the original figure.

of MPAs (Garcia-Charton and Pérez-Ruzafa, 1999). The biomass of exploited populations in any no-take zone is clearly a result of a bottom-up process (Garcia-Rubies *et al.*, 2013). Knowing the key factors that regulate fish biomass is therefore of paramount importance in the design of future Mediterranean MPAs. However, one has to recognize that the combination of factors that must be present to lead to enhanced biomass at small and medium spatial scales can greatly reduce the number of potential candidate sites. In any case, rare privileged hotspots should be prioritized in any future conservation project in the Mediterranean Sea.

Partially Protected Areas

Most of the 677 Mediterranean MPAs (as included in Gabrié *et al.*, 2012) are barely protected against fishing. In fact, 507 of them are Natura 2000 areas with no specific management to avoid or limit extractive activities. Excluding the vast Pelagos Sanctuary (87 500 km²), the area covered by coastal MPAs amounts to 18 965 km² (0.4% of the total surface of the Mediterranean Sea), while only 207 km² (0.012% of the total surface of the Mediterranean) can be considered as an actual fully protected no-take area. Out of the 170 true MPAs, 80 supplied management information indicating that

only 31% (3390 km²) have some kind of special management, whether or not this includes fishing limitations. In short, only 0.14% of the Mediterranean Sea surface is known to enjoy some management so the vast majority of the Mediterranean MPAs are nothing more than partially protected areas (Gabri  *et al.*, 2012).

In spite of the fact that most MPAs are partially protected areas, studies on the benefits of partial protection are extremely scarce. Partially protected zones can have a broad range of protection regulations, going from well-protected zones, with limited fishing activities, to merely ‘paper-parks’ without any specific management or effective protection. This variability leads to a great diversity of results that, even when positive, are usually inferior to those obtained when full protection is in force. Many authors consider partially protected areas as inefficient (Denny and Babcock, 2004; Claudet *et al.*, 2008; Di Franco *et al.*, 2009; Lester *et al.*, 2009). In some cases, partially protected areas can even be counter-productive since they attract fishermen eager to fish near a no-take area, thus leading to an increase in fishing effort (Stelzenmuller *et al.*, 2007). This is possibly why large buffer zones can even have negative impacts on the overall ecological effectiveness of MPAs (Claudet *et al.*, 2006).

However, some buffer zones have shown positive trends although this mostly depends on habitat characteristics, as in the case of no-take zones, and current fisheries regulations (Coll *et al.*, 2012). Even the same regulation can lead to different results in two separate partially protected areas belonging to the same MPA. That was the case of the Freus of Eivissa and Formentera MPA, where one of the partially protected areas did not show any sign of improvement while in the other the commercial fish biomass increased by 330%. The differences were due to a combination of habitat features and fishing pressure. The seascape of the first

partially protected area had a low rugosity and few boulders, allowing fishing with trammel nets very close to the coast. In contrast, the second partially protected area had a highly complex rocky bottom with more large boulders making it very difficult to cast trammel nets there.

The buffer zone of the Medes Islands MPA showed a limited progression after 10 years of partial protection (only spear-fishing is absolutely banned there, while commercial and recreational fishing is regulated), and only three (*Epinephelus marginatus*, *Sciaena umbra* and *Dentex dentex*) out of six species studied by Garcia-Rubies *et al.* (2013) showed a positive trend in this zone. Moreover, total mean biomass of these species was 13 times lower than that observed in the no-take zone (see Figure 2.3). The modest recovery in the buffer zone could be explained by a limited spillover from the no-take zone (Garcia-Rubies *et al.*, 2013).

When protective measures are effective, and the partially protected areas are located where suitable environmental conditions prevail, the results can be quite surprising. For example, in the Nord de Menorca marine reserve one of the partially protected areas achieved a higher biomass of commercial fish than that attained in the no-take zone (Figure 2.6). Lack of spear-fishing and particularly suitable rocky bottoms in this partially protected area were the main factors allowing such a high fish biomass in spite of a moderate level of exploitation (Re ones *et al.*, 1999; Lloret *et al.*, 2008).

In general, one can conclude that the benefits of partially protected zones depend on the regulation of fishing activities, limiting the fishing effort and banning the most effective fishing methods, as well as the environmental conditions prevailing in the zone. The environmental factors that determine the success of no-take areas as fish biomass producers are exactly the same in the case of partially protected zones.

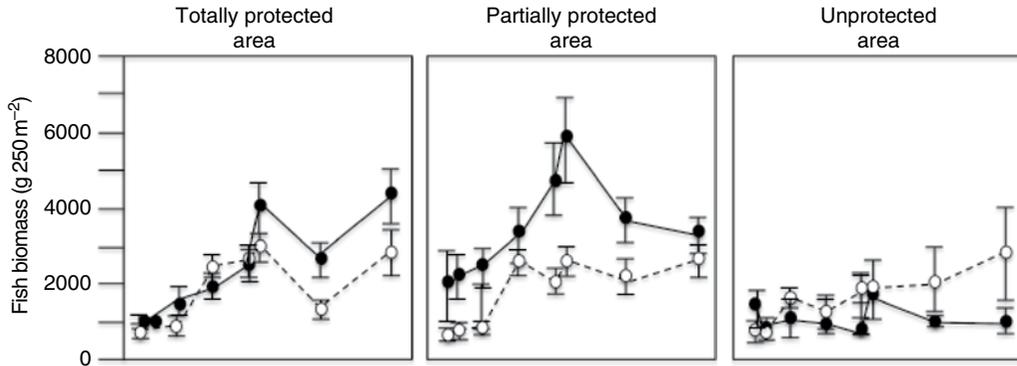


Figure 2.6 Mean commercial fish biomass at Nord de Menorca marine reserve at three protection levels; note that fish biomass at site 1 (closed circles) in the partially protected area is clearly superior to the biomass observed at site 2 (open circles) of the no-take area. *Source:* Redrawn from Coll *et al.* (2012).

Effects on Communities (Secondary Effects of Protection)

Fishing tends to remove a substantial part of exploited populations and this has been the pattern all around the Mediterranean Sea for millennia. New fishing methods such as spear-fishing have worsened the situation of large predatory fish from the second half of the 20th century onwards, especially in the Western Mediterranean (Spain, France and Italy). There are almost no historical quantitative data about the abundance of such species 70 years ago, but some books and movies from the 1940s show a very different picture from the current situation (Coll *et al.*, 2004; Guidetti and Micheli, 2011). One can conclude that these large predators have been seriously depleted from Mediterranean littoral rocky bottoms during the last 70 years or so.

The effects of such continued exploitation have led to a rarefaction of many species including many elasmobranchs (Ferretti *et al.*, 2008), monk seals *Monachus monachus* (Durant and Harwood, 1992), sea turtles, and blue fin tuna *Thunnus thynnus*. It could be concluded that there are no pristine sites left in the Mediterranean Sea (Sala *et al.*, 2012). Fishing can be considered as the main stressor of the littoral rocky reef

communities, from which large predators have nearly disappeared following the widespread process of so-called fishing down the food webs (Pauly *et al.*, 1998).

Marine Protected Areas, whether including no-take zones or well-regulated partially protected zones, have demonstrated the recovery of large predatory and carnivorous fish, which reached much higher biomass values than in open areas (Sala *et al.*, 2012; Garcia-Rubies *et al.*, 2013). In fact, top predator biomass in well-protected MPAs can account for nearly half of the total fish biomass in the most successful Mediterranean MPAs (Sala *et al.*, 2012). For instance, apex predators represent up to 49% of the total fish biomass observed at the Medes Islands MPA. Although the dusky grouper has already reached the carrying capacity there (Garcia-Rubies *et al.*, 2013), the biomass of other large predators, such as *Dentex dentex*, is still increasing and has not yet reached an asymptote.

Effects of Fish on Other Fish

The pronounced increase of large predator biomass within the no-take protected zones (Russ and Alcala, 1996) must negatively affect prey fish populations, but there is little direct evidence for this effect within

Mediterranean MPAs, although the phenomenon has been reported elsewhere. Micheli *et al.* (2004) found, through a meta-analysis of the results of up to 20 studies and 31 sites, that 19% of fish species were negatively affected by the protection. The most affected fish were small benthic species such as Blennidae, Gobiidae, Pomacentridae, Atherinidae and Apogonidae.

In the Mediterranean, Macpherson (1994) showed a lower density of small gobies and blennies in the Medes Islands MPA in comparison with two unprotected areas. Sasal *et al.* (1996) observed an increase in size of male and female *Gobius bucchichi* within the no-take zone of Banyuls MPA. The authors explained this finding on the basis of increased predation on the smaller specimens, as well as the result of an increased competition for refugia. Garcia-Rubies (1999) also observed an increase in the number of mid- and large-sized *Diplodus sargus* inside the Medes Islands MPA but also a decrease in the number of young of a year (YOYs), which could be attributed to a rise in predation pressure on small individuals during the first year in the adults' habitat. However, the predation on settlers of *Diplodus* spp. was not related to protection level, since most of the predators of settlers were themselves small species not affected by protection (Macpherson *et al.*, 1997).

Effects of Fish on Invertebrates

Sea Urchins The first studies on the consequences of increasing fish abundance showed a direct effect on sea urchin populations in MPAs (Sala *et al.*, 1998). Thus, sea urchin density in the Medes Islands MPA was four times lower than in the unprotected area, and the results of a tethering experiment demonstrated that fish predation on sea urchins was five times higher in the no-take zone. Other effects of the increased predation inside the MPA were lower sea urchin mean size, changes in size–frequency distribution, and changes in the behaviour of the sea

urchins. In a more recent study based on a long time series, carried out in the same Medes Islands MPA, Hereu *et al.* (2012) demonstrated that the relationship between fish and sea urchins is not as direct as it seemed in previous snapshot studies. In fact, these authors did not find significant long-term differences in the sea urchin density between the protected and unprotected areas. The most obvious result was that inter-annual variations in sea urchin abundance were less pronounced within the MPA than in open areas. But even the increase of predation pressure inside the MPA cannot cancel out the effects of episodic massive recruitment.

Although the effects of higher rates of urchin predation within MPAs were not as marked as suggested by short-term studies, there are more subtle results which demonstrate that fishes play an important role in regulating sea urchin populations in Mediterranean MPAs. For instance, there is a positive correlation between the abundance of juvenile and adult sea urchins inside the MPA, suggesting that the survival of juveniles is density-dependent and is facilitated by the presence of adults. Outside the MPA, no such relationship was observed since juvenile urchins can move more freely on open surfaces due to the reduced predation risk (Hereu *et al.*, 2005). One can conclude that although the increasing predation rate by fish does not have a determinant effect on total sea urchin abundance, it can buffer extreme variations in recruitment. Moreover, the total displacement and home range of sea urchins were significantly lower inside the MPA.

Spiny Lobsters As a rule, populations of exploited species recover inside MPAs, but there are some exceptions in which the trend is not so evident. In some cases, the population not only does not recover but actually tends to decline. This may be due to

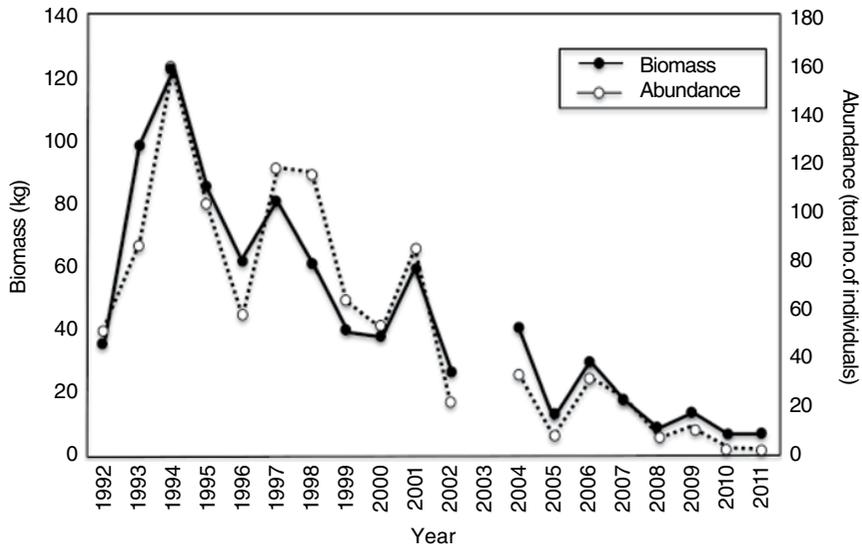


Figure 2.7 Temporal trends of abundance and biomass of the spiny lobster *Palinurus elephas* in the Medes Islands MPA over a period of 20 years. *Source:* Redrawn from Díaz (2013).

different factors: (i) protected areas are too small for those species with a large home range that regularly exceeds the boundaries of the protected area; (ii) an excessive fishing pressure outside the protected area can affect the populations inside; and (iii) the protection favours species that can damage others due to an increasing predation pressure. This seems to be the case for the spiny lobster *Palinurus elephas* in the Medes Islands MPA (Díaz *et al.*, 2005) where, according to a tethering experiment, juvenile mortality is much higher within the MPA (41%) than outside (17%). The increased juvenile mortality results in a progressive decline of the lobster population inside the Medes Islands MPA (Figure 2.7), contrasting with the trends of other highly vulnerable species (García-Rubies *et al.*, 2013). In this particular case, increased predation pressure acts along with the home range of the lobster (Giacalone *et al.*, 2006) which is larger than the protected area of the no-take zone (93 ha), high fishing pressure in the area surrounding the MPA, and the high catchability of this species.

On the other hand, opposite trends can be observed in larger MPAs, such as Columbretes Island MPA, where the lobster population has shown a recovery trend according to what might be expected in a protected area (Díaz *et al.*, 2011) in spite of a high predatory fish biomass (Goñi *et al.*, 2006).

Trophic Cascades

A trophic cascade is an indirect effect of predators, not only upon prey populations, but also on the whole food web, involving more than two trophic levels. The simplest model of a trophic cascade would include a predator (e.g. carnivore), a prey (e.g. herbivore) and a primary producer. A trophic cascade is a top-down process in which variations in predator abundance can affect the structure of the whole community (Babcock *et al.*, 1999), that is, once predator populations exceed a certain threshold, predation can control prey populations and their effects on the community. However, when predation is weak, other factors become more important in structuring communities. The best documented trophic

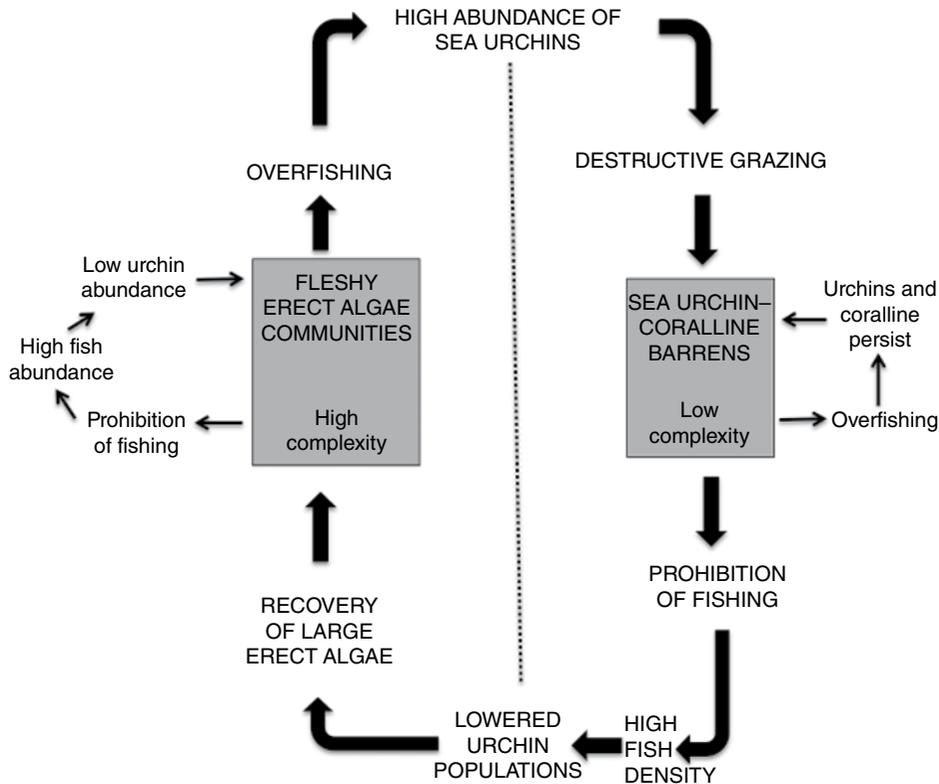


Figure 2.8 Classical model of a trophic cascade due to overfishing in rocky infralittoral algal assemblages. Recent studies have demonstrated that factors other than fishing are important in regulating sea urchin densities in the Mediterranean Sea. *Source:* Redrawn from Sala *et al.* (1998).

cascade in the Mediterranean Sea is the relationship between fishes, sea urchins and macroalgae on sublittoral rocky bottoms (Sala *et al.*, 1998) (Figure 2.8).

It has been noted that in the Mediterranean Sea many localities suffered a progressive shift from macroalgal canopies to barrens as a result of overgrazing by sea urchins, with a concurrent large loss of species diversity and changes in community structure (Ling *et al.*, 2015). Lack of sea urchin predators due to overfishing was claimed to be the main factor responsible for barren formation, so it was expected that recovery of sea urchin predators inside the MPAs would ultimately prevent the overgrazing by urchins and thus arrest barren formation

through a trophic cascade, or even lead to the recovery of (former) macroalgal canopies. The first studies showed that the relationship between fishes, sea urchins and algae seemed to be quite straightforward: high predator abundance led to an increasing predation rate on sea urchins, keeping urchins in densities low enough to prevent barren formation or even to allow a recovery of macroalgal forests. Each step seemed to fit perfectly in the model: predatory fish were more abundant in the MPA than outside (Garcia-Rubies and Zabala, 1990), the predation rate increased five times inside the MPA, and the density of sea urchins was significantly lower within the MPA (Sala and Zabala, 1996).

This process is a clear paradigmatic example of a discontinuous catastrophic regime shift that meets all the requirements of such phenomena, namely: there is an abrupt change from one state to the other that occurs once the sea urchins exceed a certain threshold of density or biomass; the new state is very stable and persists over time as sea urchin biomass is maintained after the shift, preventing any reversion towards the previous state; hysteresis occurs since for a reversion to occur, the sea urchin biomass must be, at least, one order of magnitude less than that which caused the shift. An obvious temporal asymmetry is also introduced since the shift back to the original state takes much longer than the shift forward (Ling *et al.*, 2015).

Sea urchin biomass can be maintained after the establishment of a barren at the cost of sea urchins having to eat encrusting algae and sessile invertebrates. They also eat any newly settled macroalgae, thus preventing new re-colonization, although this has not yet been shown for the Mediterranean Sea. In addition, once the barren has been established there are many negative feedbacks tending to reinforce this alternative phase and preventing the shift backwards. It has been shown that overfishing of predators of sea urchins is a key factor (Guidetti, 2005) both in the development of barrens and in preventing reversion to the original state. In the case of the Mediterranean Sea, it has been argued that the overexploitation of fish that feed on sea urchins is the main cause of the formation of barrens, which can also be favoured by destructive fishing methods (Guidetti, 2011).

Marine Protected Areas are useful in maintaining high densities of sea urchin predators (mainly the sea bream *Diplodus sargus* and *D. vulgaris*; Guidetti, 2006). In a broad study Guidetti and Sala (2007) found that a minimum of 15 adult sea bream per 100m² were necessary to reduce inter-annual variation in sea urchin densities by

preventing peaks occurring due to abnormally high annual recruitments (Cardona *et al.*, 2007, 2013). Hereu *et al.* (2012) came to a similar conclusion after analysing the longest series of data on sea urchin density from the Mediterranean. While the average density of sea urchins in the Medes Islands marine reserve was similar to that found in the fished area near the MPA, annual variations within the MPA were much smaller. The peaks due to recruitment events leading to abnormally high sea urchin densities were suppressed in the marine reserve, reducing the risk of barren formation and maintaining the presence of communities of macroalgae (Figure 2.9).

The MPA as a Touchstone: Estimation of the Degree of Exploitation Outside MPAs

Even though the overall performance of no-take, well-enforced Mediterranean MPAs is positive in recovering exploited populations (Sala *et al.*, 2012), it must be pointed out that in 2012, only 0.012% of the total Mediterranean Sea surface was known to enjoy this highly protected status (Gabrié *et al.*, 2012). This very limited area prevents any significant effect at the scale of the whole Mediterranean, although local effects can be important, even in increasing fishing yields (Goñi *et al.*, 2006). In the current situation, protected areas are nothing but a small exception, so perhaps the best value of MPAs is as the best benchmarks available for assessing the resources exploited in the other 99.88% or so of the Mediterranean. Up to now only McClanahan *et al.* (2011), in the Indian Ocean, and Sala *et al.* (2012), in the Mediterranean, have established such baseline data.

Results obtained in well-protected Mediterranean MPAs are a good baseline reference for the state of exploitation of non-protected areas. According to Worm *et al.* (2009), an exploited stock whose

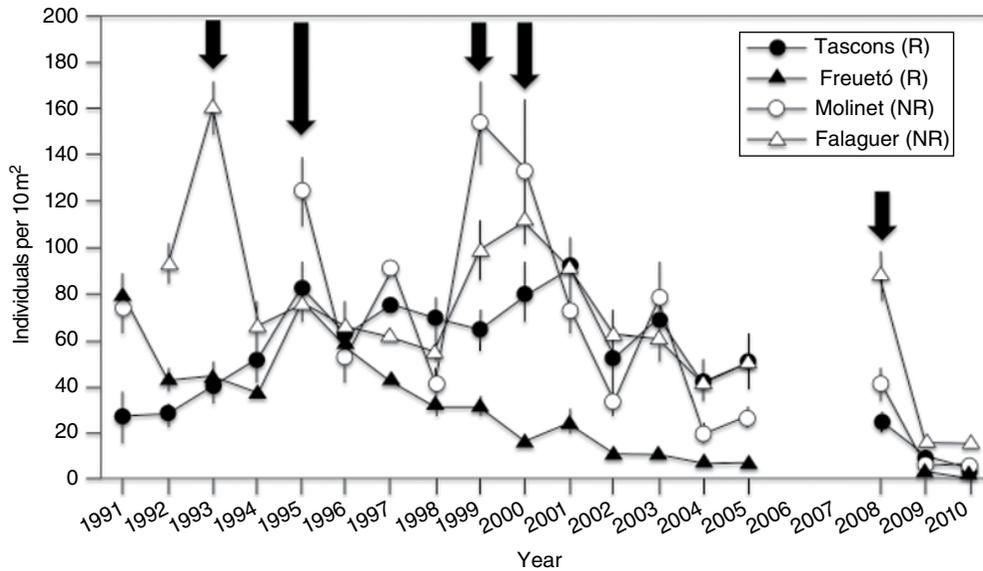


Figure 2.9 Mean density (individuals per 10 m² ± SE) of sea urchin *Paracentrotus lividus* (>1 cm diameter) inside and outside the Medes Islands MPA on boulder substrates. Arrows indicate abnormally high peaks of annual recruitment. R, reserve; NR, non-reserve. *Source:* Redrawn from Hereu *et al.* (2012).

numbers do not reach 10% of the unexploited biomass (pristine biomass, according to McClanahan *et al.*, 2007) can be considered to be practically in ecological collapse; that, for instance, was the case of three species (*Epinephelus marginatus*, *Dicentrarchus labrax* and *Sciaena umbra*) out of six analysed by Garcia-Rubies *et al.* (2013).

Although the differences between MPAs and fished areas are basically the result of a top-down process (fishing vs. no fishing) there are several environmental factors that can favour and enhance these differences. The Medes Islands MPA (Garcia-Rubies *et al.*, 2013) and other especially favoured or 'hotspot' Mediterranean MPAs (Sala *et al.*, 2012) approach what Coll *et al.* (2012) described as the ideal MPA, that is, a rocky outcrop in the open sea, totally exposed to winds and currents with a pronounced nearby slope and a highly complex (rugose) bottom composed of big rocky blocks. A surrounding boundary of sedimentary

bottoms, preventing spillover (Garcia-Rubies *et al.*, 2013), will further help to get an elevated rocky reef fish biomass.

Not all the Mediterranean rocky coast shows such favourable conditions, but Coll *et al.* (2012) proposed a model that allows prediction of maximum biomass of fish if the environmental conditions are known. This predictive biomass model was applied in 28 exploited sites ($N=260$ transects) in the littoral of the Balearic archipelago. A mean value of expected total biomass (Bt_e) was obtained for each site, assuming that the sites had the maximum protection level. In short, Bt_e is the projected value closest to the potential carrying capacity (K) of each site, as a proxy of pristine biomass according to McClanahan *et al.* (2007). The difference between Bt_e and the mean biomass observed at each site (Bt_o) gives the degree of exploitation of that site as well as its potential for recovery. Moreover, one can establish a range of Bt_e values at each site to determine whether the resources are within the limits

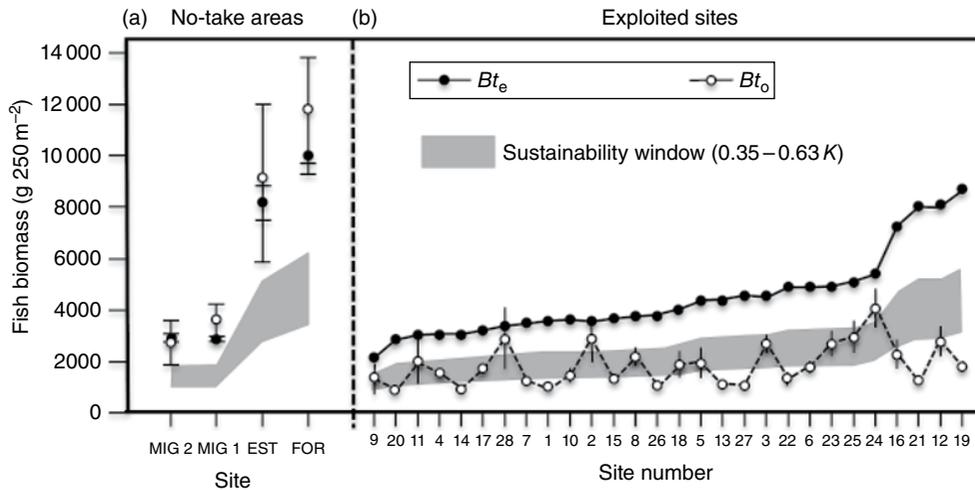


Figure 2.10 (a) Observed and expected total biomass values (Bt_o and Bt_e , respectively) in two no-take sites from two marine reserves where the model was tested: Mitgjorn Marine Reserve (MIG 1 and MIG 2) and Cabrera National Park (EST and FOR). (b) Bt_o in increasing order of Bt_e values from 28 exploited sites of the Balearic Islands; the grey band shows the biomass range window within the exploitation status of each site that could be considered sustainable ($0.35K < Bt < 0.63K$, K being the value of carrying capacity). Source: Redrawn from Coll *et al.* (2013).

of sustainability. The maximum and minimum of these limits were set to 63% and 35% of Bt_e taking into account both traditional target reference points (TRP) from maximum sustainable yield and the concept of multiple maximum sustainable yield (MMSY) for fish assemblages (Worm *et al.*, 2009), respectively.

Applying the model to exploited areas showed that 46% of them are within the limits of sustainability, while 43% are clearly overexploited and only 11% could be considered underexploited. The differences between no-take and open areas are evident (Figure 2.10). It is interesting to see that the areas that can potentially host the greatest biomasses are also the ones that show the highest differences from the expected biomass values. The fact that higher quality habitats that might support populations of highly vulnerable sedentary fish species with limited home ranges, high trophic status and long lifespan are especially affected by exploitation (i.e. *Epinephelus* spp., *Sciaena*

umbra, *Phycis phycis*), or are recruitment limited (such as *Diplodus* spp.), could be the cause.

The predictive biomass model described above has been developed for the littoral of the Balearic archipelago and it encompasses only the total biomass of 13 commercial species in a narrow range of depths (3–15 m). It would be desirable to know the K values for each fish species and also to improve the sampling methodology to determine the habitat requirements at a specific level. However, two main results show the applicability of the predictive biomass model. First the correlation between observed and predicted values for the 32 sites studied ($R=0.60$; $p<0.001$) suggests that the environmental variables included in the model are limiting factors for fish biomass, and that these factors are also good descriptors of the variation in fish biomass throughout the entire geographical area of the Balearic Islands, representing characteristic rocky habitats of the western Mediterranean.

The second key result of this study is the high predictive capacity of the model when no-take areas are tested as controls. The expected mean values are almost identical to the observed values and always fall within their margins of error.

Spillover and Larval Export

Adults and Juveniles

Marine Protected Areas can benefit neighbouring populations through 'spillover', that is, the net export of adult or juvenile individuals to non-protected areas (Russ *et al.*, 2004; Dudley and Hockings, this volume). There is some direct evidence of spillover in the Mediterranean MPAs. The study on spiny lobster in the Columbretes Islands MPA was the first to assess spillover from tag-recapture data (Goñi *et al.*, 2006, 2010). In this case it was estimated that 7% of the lobsters residing in the MPA emigrated every year to the adjacent fishing grounds, providing a net gain of over 10% of the catch in weight. A further study on spiny lobster using tag-recapture data was done by Follesa *et al.* (2011) to assess spillover in the Su Pallosu MPA in Sardinia. The authors found a clear gradient of catch per unit effort (CPUE) from the MPA boundaries towards fishing grounds, with a maximum located within about 6 km from the marine reserve boundary. The study by La Mesa *et al.* (2012) is the only one in the Mediterranean addressing, albeit indirectly, spillover using radio-tracking techniques. It showed that the home range of the parrotfish *Sparisoma cretense* extended beyond the boundaries of the no-take area of the Portofino MPA, and therefore that spillover was possible. Other studies have assessed density/biomass gradients across no-take area boundaries as a way to infer spillover, for example Guidetti (2007), Harmelin-Vivien *et al.* (2008), Forcada *et al.* (2008), La Mesa *et al.* (2011) and García-Rubies *et al.* (2013).

Eggs and Larvae

Spillover can be relevant in those overexploited species where reproduction occurs mostly in MPAs, for example the dusky grouper (Zabala *et al.*, 1997). Most marine organisms have complex life histories that include pelagic eggs and larval stages. These stages influence dispersal capabilities, affecting not only the geographical distribution of the species but also settlement rates and patterns of recruitment to the adult population, with the resulting effects on community structure. Recent studies suggest that the extent of dispersal between populations is more complex than previously assumed, with water flow dynamics and oceanographic discontinuities being important factors determining the population connectivity and settlement intensity (Cowen *et al.*, 2006; Galarza *et al.*, 2009; Schunter *et al.*, 2011a, 2011b, 2014).

In the Mediterranean Sea the large-scale circulation is superimposed on small-scale dynamics (see Boero, this volume). More than 500 canyons act as auxiliary engines to the main cold engines in the Western Mediterranean and the Adriatic Sea. The shape of the coast also generates gyres and eddies that concentrate nutrients and propagules. The portions of marine systems where production phenomena are generated by the interaction of physical, chemical, biological and ecological processes have been termed 'cells of ecosystem functioning' (Boero, 2015). Each one of these cells has its own specific characteristics and can be considered as relatively isolated from other cells due to physical boundaries (fronts) that reduce exchange of propagules. The cells are thus to be considered as the true biogeographical regions, each one with its own biological features. This is why Boero (this volume) suggested that each of these cells should contain, at least, an MPA network in order to preserve the main habitats of each cell of ecosystem functioning.

Populations that are not separated by evident oceanographic boundaries can also show a strong genetic isolation, usually related with a very low larval dispersal capability of the species and a sessile adult life (Duran *et al.*, 2004; Carreras-Carbonell *et al.*, 2006). This genetic isolation has been observed in many sessile organisms that constitute a fundamental part of the Mediterranean ecosystems (Uriz and Turon, 2012).

In addition, several studies have demonstrated that fish populations are not always open and that the proportion of larvae that may return to their natal population (self-recruitment) is very high (Galindo *et al.*, 2006; Almany *et al.*, 2007; Carreras-Carbonell *et al.*, 2007; Schunter *et al.*, 2014). These studies suggest that the extent of dispersal between populations is lower than currently assumed, as predicted by Cowen *et al.* (2006), affecting the connectivity among populations and having important implications for marine conservation policies.

Studies on the degree of self-recruitment in Mediterranean littoral fishes suggest that the self-recruited juveniles have lower probabilities of survivorship compared to juveniles from other localities (Carreras-Carbonell *et al.*, 2007; Planes *et al.*, 2009; Félix-Hackradt *et al.*, 2013). Therefore, the degree of connectivity among populations can also influence the spillover role of protected areas, the assessment of numerous fishery-exploited species and, in general, the management of marine ecosystems (Palumbi, 2004).

This scenario, however, is very different for species with very limited larval dispersal capabilities (e.g. sponges, ascidians, bryozoans and numerous algae). These species show a fine-scale genetic structure with genetic differences at distances in the range of metres (Duran *et al.*, 2004; Calderón *et al.*, 2007; Mokhtar-Jamaï *et al.*, 2011). This genetic structure may be common in invertebrates with lecithotrophic larvae.

Such invertebrates constitute an essential component of coastal rocky Mediterranean ecosystems, suggesting that the isolation in these species can have a strong effect in the dynamics of their populations. No spillover is expected for these organisms.

The approach of inferring population connectivity through genetic differentiation of locations is effective for the identification of major biogeographical or oceanographic barriers (Planes *et al.*, 2009) and allows for the measurement of gene flow across evolutionary timescales. Unfortunately, larval trajectories remain largely unknown and understanding present-day dispersal patterns is still a great challenge. Parentage analysis permits the direct estimation of connectivity, as the detection of parent-offspring pairs allows in many cases the movement of the offspring to be reconstructed, providing direct evidence of dispersal (e.g. Almany *et al.*, 2007; Planes *et al.*, 2009; Saenz-Agudelo *et al.*, 2011; Berumen *et al.*, 2012). At present, only one study (Schunter *et al.*, 2014) has used this direct measure of connectivity in the Mediterranean: it demonstrated a limited connectivity, with a decrease in dispersal success over 1 km distance and approximately 15% of the collected juveniles of *Tripterygion delaisi* identified as self-recruits; sibship reconstruction analysis found that full siblings in general did not recruit together to the same location, and that the distance between recruitment locations was more extensive (about 11.5 km).

The main conclusions from these results are that most coastal species in the Western Mediterranean have a reduced level of connectivity among populations, and it is likely that similar patterns of connectivity are present in the Eastern Mediterranean too, as well as the Black Sea (Öztürk *et al.*, this volume). As a result, islands (e.g. Balearic Islands, Ionian Islands, Aegean Islands) are mostly isolated from the continental coast

and from each other, especially for littoral species, although a different situation may operate for species inhabiting deeper ecosystems. Therefore, the vulnerability of these island ecosystems is higher than for the continental coast. These results emphasize the necessity of establishment of MPA networks among islands and between the continent and the islands. Therefore the areas separated by oceanographic discontinuities (Schiavina *et al.*, 2014) should be considered as separate management units, requiring, at least, an MPA network for each unit (Boero, 2015). Finally, considering that many structural and/or key species (e.g. *Paramuricea clavata*, some sponges, some arborescent algae) have limited gene flow between populations, the distance between MPAs in the network should be established keeping these considerations in mind.

Management Implications

Anthropogenic factors that threaten marine ecosystems are many, complex and often act cumulatively or synergistically (Spalding *et al.*, 2013). Some are difficult to address, or virtually impossible to reverse in the short term (e.g. the effects of climate change), while others require management measures based on different approaches at different spatial scales. One of these management measures is the establishment of MPAs.

The IUCN defines an MPA 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' (Laffoley, 2008). This is a very loose definition that can lead to many interpretations. The IUCN also defines up to seven categories of MPAs but only one includes a no-take zone and limited public access and can be considered a marine reserve.

There are various initiatives for the protection of Mediterranean ecosystems through MPAs. Many international organizations have proposed the creation of networks of MPAs for preserving Mediterranean ecosystems. Environmental NGOs (including WWF, ACCOBAMS, OCEANA and MedPAN) and scientific organizations (CIESM) have proposed various areas of special conservation interest including not only coastal habitats at a regional or national level, but also large transnational MPAs in offshore or deep-sea ecosystems (see Micheli *et al.*, 2013). The EU is also interested in the use of MPAs as a management measure in the Mediterranean Sea. Consequently, it has funded some international research projects, from the mid-1990s to the present (e.g. ECOMARE, BIOMEX, EMPAFISH, PERSEUS, CoCoNet), as well as others, such as LIFE projects. These projects have been instrumental in promoting international collaboration, including scientific publications, reports, websites, models and other tools, information sites for the general public, as well as training of personnel who can continue to contribute to increasing knowledge of the marine ecosystem and the role of MPAs. In general, these products have shown the utility of MPAs in increasing the biomass of natural resources, including increasing fisheries yields around MPAs, and conservation of littoral habitats.

However, the ecological benefits of protection contrast with the sad reality that most of the beneficial results shown by the Mediterranean MPAs come from no-take zones (i.e. true marine reserves) that represent only 0.012% of the total sea surface (Gabrié *et al.*, 2012). Although there is an increasing number of MPAs in the Mediterranean and around the world (Gabrié *et al.*, 2012; Costello and Ballantine, 2015), only a few can be considered as really effective. Most MPAs are nothing but a false

image of protection. Currently, the main flaws of many Mediterranean MPAs include:

- 1) Few MPAs really work as protected areas due to poor or no management and lack of effective surveillance and enforcement, and, in some cases, no implementation of the management plans (assuming these exist).
- 2) Many MPAs are the result of political opportunism or spatial considerations that have little to do with ecology; that is, most MPAs have been established where and when it was opportunistically possible for mostly non-scientific reasons, such as in areas where there is likely to be least negative reaction from stakeholders.
- 3) Most MPAs seem to be ‘cure-all’ remedies aiming at the conservation of ‘biodiversity’ and at the same time favouring artisanal fisheries and sustainable use of resources. These all-purpose MPAs may sound good, but rarely incorporate adequate management measures and the ecological conditions to achieve all these high goals.
- 4) For many MPAs there is a lack of representation and of information on what is to be protected (no lists of species, no habitat mapping and no baselines which are necessary to test the effectiveness of protection).
- 5) Most MPAs lack long-term monitoring and adaptive management based on the monitoring results.
- 6) Some MPAs show serious deviations from the original objectives due to a bias towards economic interests (e.g. tourism).
- 7) Most MPAs are located in the western basin of the Mediterranean Sea; only a small number of MPAs have been established along the North African coast and in the eastern Mediterranean basin.

The number of EU-funded projects aimed at testing the effects of MPAs contrast with the caution shown by the EU when it comes to

the effective protection of the Mediterranean through the use of MPAs. The Natura 2000 initiative can be considered only a hesitant approach to the issue of protecting the coastline and inshore waters; the guidelines for the establishment of the Natura 2000 network are based on the Birds Directive (2009/147/EC) and Habitats Directive (92/43/EEC). There is no question that birds are an important component of marine ecosystems, but they cannot be fundamental in the design of protected areas aiming to protect marine (mostly ‘submarine’) habitats. On the other hand, the Habitats Directive includes only 10 marine ‘habitats’ (actually mostly geomorphological units and habitat complexes), including *Posidonia oceanica* meadows and ‘reefs’, which are already protected by law (<http://www.europa.eu.int./comm/environment/nature/hab-en.htm>).

Considering Natura 2000 sites to be real MPAs may even prove counterproductive in the Mediterranean since they create a false impression of protection when, in fact, no actual protection is occurring (Agardy *et al.*, 2011). The management of these areas depends on national and regional governments and, in most cases, can be considered negligible or nil (Gabrié *et al.*, 2013). Only 25% of these spaces have some kind of management. In some countries (e.g. France, Spain), the marine Natura 2000 sites have been proposed as actual MPAs to meet the targets set out in the Aichi Biodiversity Targets of the Convention on Biological Diversity (Meinesz and Bienfune, 2015), which allows these countries to meet target No. 11 which states that ‘10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well connected systems of protected areas and other effective area-based conservation’ by 2020 (see <https://www.cbd.int/sp/targets/>).

Spain, for instance, has not hesitated even to integrate some of these spaces in a brand new 'network' of MPAs (the so-called RAMPE: *Spanish Network of Marine Protected Areas*), grouping several areas with very different management levels (from Natura 2000 sites, to well-protected marine reserves). The possible connectivity among MPAs has almost certainly not been taken into account in this Spanish network. In short, this indicates that although the Aichi target No. 11 is officially met, in practice much less than 10% of the area is conserved and properly managed. The situation in other Mediterranean countries may be similar.

Among European countries, the most coherent policy towards MPAs seems to be that of Italy, which does not include the Natura 2000 sites in the Aichi target, but is based on the establishment of 29 well-managed MPAs (Meinesz and Bienfune, 2015), all of them with the same zoning, effective protection (including at least one no-take area) and a similar management regime. Hence the Italian MPAs form effective conservation and geographical networks in which no-take areas are included.

Although the number of Mediterranean MPAs that function as effective protected areas are few, several international organizations are presently promoting the establishment of networks of MPAs. These networks have to meet the minimum requirements of representativeness, effectiveness, replicability and connectivity (IUCN-WCPA, 2008). The first step in creating a conservation MPA network (Beal *et al.*, this volume) is that the component sites have effective management to ensure good protection of threatened habitats and species. In this sense, the initiative to establish a list of *Specially Protected Areas of Mediterranean Importance* (SPAMI) seems much more realistic than to merely consider marine Natura 2000 sites as effective MPAs. At least, the candidate areas to be included in SPAMI meet certain minimum requirements,

including having a legal status and protection, planning and management measures. However, laudable initiatives such as the detailed Mediterranean MPAs Roadmap (http://www.medmpaforum2012.org/sites/default/files/mediterranean_mpa_roadmap.pdf) do not impose the minimum requirements of management and effective enforcement for the MPAs to be integrated into the networks, and include Natura 2000 zones as MPAs. Assembling 'paper MPAs' into a nominal network will not improve the situation.

It seems that the EU lacks the political will to enforce the minimum requirements that any European MPA should have to be considered a real MPA. In this regard the EU should evaluate the existing and future MPAs based on some criteria such as serious management and means to ensure the effective protection of the ecological and biological features that the MPAs were set up to conserve. The minimum requirements for any coastal MPA to be approved by the EU would be similar to those proposed by Meinesz and Bienfune (2015):

- 1) All the MPAs must include a representative no-take area in which fishing should be absolutely banned and a buffer zone in which a limited amount of fishing could be allowed, excluding the more harmful methods (e.g. spear-fishing) in order to preserve the local artisanal fishery.
- 2) There must be effective enforcement of protection measures with a sufficient number of wardens and the means that allow an efficient surveillance of the protected area.
- 3) There must be clear protection objectives, and avoidance of the misinterpretation and wrong implementation of such objectives that could lead to misuse of protected areas.

Such effective MPAs may serve to protect biologically rich habitats, restore overexploited stocks of target species, resolve user

conflicts and ameliorate degraded areas. Therefore the establishment of MPAs taking into account the above minimum criteria will eventually lead to better management and protection of marine species and habitats, particularly if MPAs form an ecologically coherent network linking cells of ecosystem functioning. However, routine monitoring of reserves is far from common and, in general, the elements conferring effectiveness have in many cases not yet been established. Management of future Mediterranean MPA networks is also challenged by lack of information on habitat distribution, and on how populations are connected between habitats and MPAs through dispersal of pelagic larvae or propagules.

The present overview of the ecological effects and benefits of MPAs gives rise to several issues that have implications for management. In particular, good management of MPAs and MPA networks should take the following into account:

- 1) The carrying capacity (K) for exploited species is a key factor that is indicative of management effectiveness; management actions should focus on attaining K values.
- 2) The magnitude of K values and time to attain them depend on various factors, including the life history of the species, extent of protection and environmental features of the area. Predictive biomass modelling can be used to determine the K values under a particular set of circumstances and hence to monitor the effectiveness of MPAs; therefore collection of the required environmental data for biomass modelling should be built into management plans.
- 3) Since environmental features play a role in determining the extent of population recovery, the physical environment itself must also be managed. In addition, when designing MPA zoning schemes, those areas having the 'best' habitat for the most vulnerable species, or habitats which are more susceptible to adverse effects, should be chosen as no-take zones.
- 4) Activities that are of particular concern (e.g. spear-fishing in the case of large long-lived predatory fish species) may need to be banned even outside no-take zones, while other activities will need to be regulated. The extent of regulation for different activities will depend on the objectives of the MPA.
- 5) Buffer zones can only serve as 'buffers' if they are adequately managed. In the absence of management measures, these zones would be similar to non-protected areas and may even be counterproductive, leading to excessive fishing pressure outside the fully protected areas that can even affect the populations inside. Management plans should include carefully designed zoning schemes to reduce such impacts, for instance through having a set of nested buffer zones, each with a different regulatory regime.
- 6) MPAs should not be managed solely for recovery of top predators since prey species are equally important for maintaining a functioning ecosystem and some prey species are themselves of conservation concern. In addition, prey abundance is itself a factor that can influence the carrying capacity of the system for top predators. Management must therefore take prey species into account and actions that favour a balance between predator and prey populations may be needed; for example, management of the physical habitat should not focus solely on habitats that yield maximum biomass of predators but also habitats that offer shelter to their prey. This should ultimately lead to MPAs having an ecosystem structure and function that is similar to that found in pristine environments.

- 7) MPAs should be large enough to protect substantial portions of populations of sedentary species since MPAs that are much smaller than the home range of a species offer inadequate protection for such species. In such circumstances, management may therefore involve extending the areas of MPAs or linking adjacent MPAs through connected corridors where activities are regulated.
- 8) MPAs should be linked into an ecologically coherent network; 'connectivity' should therefore be built into any management plan. This can be taken into consideration when designing MPA networks to allow connectivity through eggs, larvae and propagules; such connectivity cannot be assumed and must be ascertained through appropriate research. For already existing MPAs, connectivity may be enhanced through establishing 'protected corridors' between the MPAs or having large buffer zones linking neighbouring MPAs, where activities are regulated.
- 9) MPA effectiveness is also linked to the level of enforcement of, and compliance with, the protection measures; setting protection levels on paper may be easy but enforcement is difficult. Compliance is a function of both enforcement and stakeholder education; therefore, management actions should also focus on educating stakeholders.

We hope that these considerations will aid the formulation and implementation of appropriate management plans for Mediterranean MPAs in the near future, which, together with the establishment of a functioning network of MPAs whose design is based on sound scientific data on the distribution of species, habitats and their connectivity patterns, will ultimately serve to improve the health of one of the world's hotspots of marine biodiversity.

References

- Abdulla, A., Gomei, M., Maison, E. and Piante, C. (2008) *Status of Marine Protected Areas in the Mediterranean Sea*. IUCN, Malaga and WWF, France. 152 pp.
- Agardy, T., Notarbartolo Di Sciara, G. and Christie, P. (2011) Mind the gap: addressing the shortcomings of marine protected areas through large scale marine spatial planning. *Marine Policy*, **35** (2), 226–232.
- Almany, G.R., Berumen, M.L., Thorrold, S.R. *et al.* (2007) Local replenishment of coral reef fish populations in a marine reserve. *Science*, **316**, 742–744.
- Babcock, R.C., Kelly, S., Shears, N.T. *et al.* (1999) Changes in community structure in temperate marine reserves. *Marine Ecology Progress Series*, **189**, 125–134.
- Bell, J.D. (1983) Effects of depth and marine reserve fishing restrictions on the structure of a rocky reef fish assemblage in the northwestern Mediterranean Sea. *Journal of Applied Ecology*, **20** (2), 357–369.
- Berumen, M.L., Almany, G.R., Planes, S. *et al.* (2012) Persistence of self recruitment and patterns of larval connectivity in a marine protected area network. *Ecology and Evolution*, **2**, 444–452.
- Boero, F. (2015) The future of the Mediterranean Sea ecosystem: towards a different tomorrow. *Rendiconti Lincei*, **26**, 3–12.
- Botsford, L.W., Castilla, J.C. and Peterson, C.H. (1997) The management of fisheries and marine ecosystems. *Science*, **277**, 509–515.
- Botsford, L.W., Brumbaugh, D.R., Grimes, C. *et al.* (2009) Connectivity, sustainability, and yield: bridging the gap between conventional fisheries management and marine protected areas. *Reviews in Fish Biology and Fisheries*, **19** (1), 69–95.
- Calderón, I., Ortega, N., Duran, S. *et al.* (2007) Finding the relevant scale: clonality and genetic structure in a marine invertebrate (*Crambe crambe*, Porifera). *Molecular Ecology*, **16**, 1799–1810.

- Cardona, L., Sales, M. and Lopez, D. (2007) Changes in fish abundance do not cascade to sea urchins and erect algae in one of the most oligotrophic parts of the Mediterranean. *Estuarine, Coastal and Shelf Science*, **72** (1–2), 273–282.
- Cardona, L., Moranta, J., Reñones, O. and Hereu, B. (2013) Pulses of phytoplanktonic productivity may enhance sea urchin abundance and induce state shifts in Mediterranean rocky reefs. *Estuarine, Coastal and Shelf Science*, **133**, 88–96.
- Carreras-Carbonell, J., Macpherson, E. and Pascual, M. (2006) Population structure within and between subspecies of the Mediterranean triplefin fish *Tripterygion delaisi* revealed by highly polymorphic microsatellite loci. *Molecular Ecology*, **15**, 3527–3539.
- Carreras-Carbonell, J., Macpherson, E. and Pascual, M. (2007) High self-recruitment levels in a Mediterranean littoral fish population revealed by microsatellite markers. *Marine Biology*, **151**, 719–727.
- Cheung, W.L., Pitcher, T.J. and Pauly, D. (2005) A fuzzy logic expert system to estimate intrinsic extinction vulnerabilities of marine fishes to fishing. *Biological Conservation*, **124** (1), 97–111.
- Claudet, J., Pelletier, D., Jouvenel, J.Y. *et al.* (2006) Assessing the effects of a marine protected area (MPA) on a reef fish assemblage in a northwestern Mediterranean marine reserve: identifying community-based indicators. *Biological Conservation*, **130** (3), 349–369.
- Claudet, J., Osenberg, C.W., Benedetti-Cecchi, L. *et al.* (2008) Marine reserves: size and age do matter. *Ecology Letters*, **11** (5), 481–489.
- Claudet, J., Garcia-Charton, J.A. and Lenfant, P. (2011) Combined effects of levels of protection and environmental variables at different spatial resolutions on fish assemblages in a marine protected area. *Conservation Biology*, **25** (1), 105–114.
- Coll, J., Linde, M., Garcia-Rubies, A. *et al.* (2004) Spear fishing in the Balearic Islands (west central Mediterranean): species affected and catch evolution during the period 1975–2001. *Fisheries Research*, **70**, 97–111.
- Coll, J., Garcia-Rubies, A., Morey, G. and Grau, A.M. (2012) The carrying capacity and the effects of protection level in three marine protected areas in the Balearic Islands (NW Mediterranean). *Scientia Marina*, **76** (4), 809–826.
- Coll, J., Garcia-Rubies, A., Morey, G. *et al.* (2013) Using no-take marine reserves as a tool for evaluating rocky-reef fish resources in the western Mediterranean. *ICES Journal of Marine Science*, **70** (3), 578–590.
- Coll, M., Piroddi, C., Steenbeek, J. *et al.* (2010) The biodiversity of the Mediterranean Sea: estimates, patterns, and threats. *PLoS ONE*, **5** (8), e11842. doi:10.1371/journal.pone.0011842
- Costello, M.J. and Ballantine, B. (2015) Biodiversity conservation should focus on no-take Marine Reserves: 94% of Marine Protected Areas allow fishing. *Trends in Ecology and Evolution*, **30** (9), 507–509.
- Côté, I.M., Mosqueira, G. and Reynolds, J.D. (2001) Effects of marine reserves characteristics on the protection of fish populations: a meta-analysis. *Journal of Fish Biology*, **59** (Supplement A), 178–189.
- Cowen, R.K., Paris, C.B. and Srinivasan, A. (2006) Scaling of connectivity in marine populations. *Science*, **311**, 522–527.
- Denny, C. and Babcock, R. (2004) Do partial marine reserves protect reef fish assemblages? *Biological Conservation*, **116** (1), 119–129.
- Di Franco, A., Milazzo, M., Baiata, P. *et al.* (2009) Scuba diver behaviour and its effects on the biota of a Mediterranean marine protected area. *Environmental Conservation*, **36** (1), 32–40.
- Díaz, D. (2013) Els grans decàpodes de la costa del Montgrí i les Illes Medes, in 'Els fons marins de les Illes Medes i el Montgrí: quatre dècades de recerca' (eds B. Hereu and X. Quintana). *Recerca i Territori*, **4**, 139–154.

- Díaz, D., Zabala, M., Linares, C. *et al.* (2005) Increased predation of juvenile European spiny lobster (*Palinurus elephas*) in a marine protected area. *New Zealand Journal of Marine and Freshwater Research*, **39** (2), 447–453.
- Díaz, D., Mallol, S., Parma, A.M. and Goñi, R. (2011) Decadal trend in lobster reproductive output from a temperate marine protected area. *Marine Ecology Progress Series*, **433**, 149–157.
- Duran, S., Pascual, M., Estoup, A. and Turon, X. (2004) Strong population structure in the marine sponge *Crambe crambe* (Poecilosclerida) as revealed by microsatellite markers. *Molecular Ecology*, **13**, 511–522.
- Durant, S.M. and Harwood, J. (1992) Assessment of monitoring and management strategies for local populations of the Mediterranean monk seal *Monachus monachus*. *Biological Conservation*, **61** (2), 81–92.
- Félix-Hackradt, F.C., Hackradt, C.W., Pérez-Ruzafa, A. and García-Charton, J.A. (2013) Discordant patterns of genetic connectivity between two sympatric species, *Mullus barbatus* (Linnaeus, 1758) and *Mullus surmuletus* (Linnaeus, 1758), in south-western Mediterranean Sea. *Marine Environmental Research*, **92**, 23–34.
- Fenberg, P.B., Caselle, J.E., Claudet, J. *et al.* (2012) The science of European marine reserves: status, efficacy, and future needs. *Marine Policy*, **36** (5), 1012–1021.
- Ferretti, F., Myers, R.A., Serena, F. and Lotze, H.K. (2008) Loss of large predatory sharks from the Mediterranean Sea. *Conservation Biology*, **22** (4), 952–964.
- Follesa, M.C., Cannas, R., Cau, A. *et al.* (2011) Spillover effects of a Mediterranean marine protected area on the European spiny lobster *Palinurus elephas* (Fabricius, 1787) resource. *Aquatic Conservation of Marine and Freshwater Ecosystems*, **21** (6), 564–572.
- Forcada, A., Bayle Sempere, J., Valle, C. and Sánchez-Pérez, P. (2008) Habitat continuity effects on gradients of fish biomass across marine protected area boundaries. *Marine Environmental Research*, **66** (5), 536–547.
- Gabrié, C., Lagabrielle, E., Bissery, C. *et al.* (2012) *The Status of Marine Protected Areas in the Mediterranean Sea*. MedPAN, Marseilles and RAC/SPA, Tunis. MedPAN Collection. 256 pp.
- Galarza, J., Carreras-Carbonell, J., Macpherson, E. *et al.* (2009) The influence of oceanographic fronts and early-life history traits on connectivity among fish populations: a multi-species approach. *Proceedings of the National Academy of Sciences*, **106**, 1473–1478.
- Galindo, H.M., Olson, D.B. and Palumbi, S.R. (2006) Seascape genetics: a coupled oceanographic–genetic model predicts population structure of Caribbean corals. *Current Biology*, **16**, 1622–1626.
- García-Charton, J.A.A. and Pérez-Ruzafa, Á. (1999) Ecological heterogeneity and the evaluation of the effects of marine reserves. *Fisheries Research*, **42** (1), 1–20.
- García-Charton, J.A., Pérez-Ruzafa, Á., Marcos, C. *et al.* (2008) Effectiveness of European Atlanto-Mediterranean MPAs: do they accomplish the expected effects on populations, communities and ecosystems? *Journal for Nature Conservation*, **16** (4), 193–221.
- García-Rubies, A. (1999) Effects of fishing on community structure and on selected populations of Mediterranean coastal reef fish. *Naturalista Siciliano*, **23** (Supplement), 59–81.
- García-Rubies, A. and Zabala, M. (1990) Effects of total fishing prohibition on the rocky fish assemblages of Medes Islands marine reserve (NW Mediterranean). *Scientia Marina*, **54**, 317–328.
- García-Rubies, A., Hereu, B. and Zabala, M. (2013) Long-term recovery patterns and limited spillover of large predatory fish in a Mediterranean MPA. *PLoS ONE*, **8** (9), e73922. doi:10.1371/journal.pone.0073922

- Garrabou, J. and Harmelin, J.G. (2002) A 20-year study on life-history traits of a harvested long-lived temperate coral in the NW Mediterranean: insights into conservation and management needs. *Journal of Animal Ecology*, **71** (6), 966–978.
- Giacalone, V.M., D'Anna, G., Pipitone, C. and Badalamenti, F. (2006) Movements and residence time of spiny lobsters *Palinurus elephas* released in a marine protected area: an investigation by ultrasonic telemetry. *Journal of the Marine Biological Association*, **86** (5), 1101–1106.
- Goñi, R., Quetglas, A. and Reñones, O. (2006) Spillover of spiny lobsters *Palinurus elephas* from a marine reserve to an adjoining fishery. *Marine Ecology Progress Series*, **308**, 207–219.
- Goñi, R., Hilborn, R., Díaz, D. *et al.* (2010) Net contribution of spillover from a marine reserve to fishery catches. *Marine Ecology Progress Series*, **400**, 233–243.
- Guidetti, P. (2005) Reserve effect and trophic cascades in Mediterranean sublittoral rocky habitats: a case study at the Marine Protected Area of Torre Guaceto (southern Adriatic Sea). *Biologia Marina Mediterranea*, **12** (1), 99–105.
- Guidetti, P. (2006) Marine reserves re-establish lost predatory interactions and cause community changes in rocky reefs. *Ecological Applications*, **16**, 963–976.
- Guidetti, P. (2007) Potential of marine reserves to cause community-wide changes beyond their boundaries. *Conservation Biology*, **21** (2), 540–545.
- Guidetti, P. (2011) The destructive date-mussel fishery and the persistence of barrens in Mediterranean rocky reefs. *Marine Pollution Bulletin*, **62** (4), 691–695.
- Guidetti, P. and Micheli, F. (2011) Ancient art serving marine conservation. *Frontiers in Ecology and Environment*, **9**, 374–375.
- Guidetti, P. and Sala, E. (2007) Community-wide effects of marine reserves in the Mediterranean Sea. *Marine Ecology Progress Series*, **335**, 43–56.
- Guidetti, P., Milazzo, M., Bussotti, S. *et al.* (2008) Italian Marine Reserve effectiveness: does enforcement matter? *Biological Conservation*, **141**, 699–709.
- Halpern, B.S. and Warner, R.R. (2002) Marine reserves have rapid and lasting effects. *Ecology Letters*, **5** (3), 361–366.
- Harmelin, J-G., Bachet, F. and Garcia, F. (1995) Mediterranean marine reserves: fish indices as tests of protection efficiencies. *Marine Ecology*, **16** (3), 233–250.
- Harmelin-Vivien, M., Le Direach, L., Bayle-Sempere, J. *et al.* (2008) Gradients of abundance across reserve boundaries in six Mediterranean marine protected areas: evidence of spillover? *Biological Conservation*, **141** (7), 1829–1839.
- Hereu, B., Zabala, M., Linares, C. and Sala, E. (2005) The effects of predator abundance and habitat structural complexity on survival of juvenile sea urchins. *Marine Biology*, **146**, 293–299.
- Hereu, B., Linares, C., Sala, E. *et al.* (2012) Multiple processes regulate long-term population dynamics of sea urchins on Mediterranean rocky reefs. *PloS ONE*, **7** (5), e36901. doi:10.1371/journal.pone.0036901
- International Union for Conservation of Nature – World Commission on Protected Areas (IUCN-WCPA) (2008) *Establishing Marine Protected Area Networks: Making It Happen*. IUCN-WCPA, National Oceanic and Atmospheric Administration and The Nature Conservancy, Washington, DC. 118 pp.
- La Mesa, G., Molinari, A., Gambaccini, S. and Tunesi, L. (2011) Spatial pattern of coastal fish assemblages in different habitats in North-western Mediterranean. *Marine Ecology*, **32** (1), 104–114.
- La Mesa, G., Consalvo, I., Annunziatellis, A. and Canese, S. (2012) Movement patterns of the parrotfish *Sparisoma cretense* in a Mediterranean marine protected area. *Marine Environmental Research*, **82**, 59–68.

- Laffoley, D. (2008) *Towards Networks of Marine Protected Areas: The MPA Plan of Action for IUCN's World Commission on Protected Areas*. IUCN-WCPA, Gland, Switzerland. 28 pp.
- Lester, S.E., Halpern, B.S., Grorud-Colvert, K. *et al.* (2009) Biological effects within no-take marine reserves: a global synthesis. *Marine Ecology Progress Series*, **384** (2), 33–46.
- Ling, S., Scheibling, R., Rassweiler, A., Johnson, C. *et al.* (2015) Global regime shift dynamics of catastrophic sea urchin overgrazing. *Philosophical Transactions of the Royal Society B: Biological Sciences*, **370** (1659), 20130269.
- Lloret, J., Zaragoza, N., Caballero, D. and Riera, V. (2008) Biological and socioeconomic implications of recreational boat fishing for the management of fishery resources in the marine reserve of Cap de Creus (NW Mediterranean). *Fisheries Research*, **91** (2–3), 252–259.
- Macpherson, E. (1994) Substrate utilization in a Mediterranean littoral fish community. *Marine Ecology Progress Series*, **114**, 211–218.
- Macpherson, E., Biagi, F., Francour, P. *et al.* (1997) Mortality of juvenile fishes of the genus *Diplodus* in protected and unprotected areas in the western Mediterranean Sea. *Marine Ecology Progress Series*, **160**, 135–147.
- Macpherson, E., García-Rubies, A. and Gordo, A. (2000) Direct estimation of natural mortality rates for littoral marine fishes using populational data from a marine reserve. *Marine Biology*, **137** (5–6), 1067–1076.
- McClanahan, T.R., Graham, N.A., Calnan, J.M. and MacNeil, M.A. (2007) Toward pristine biomass: reef fish recovery in coral reef marine protected areas in Kenya. *Ecological Applications*, **17** (4), 1055–1067.
- McClanahan, T.R., Graham, N.A., MacNeil, M.A. *et al.* (2011) Critical thresholds and tangible targets for ecosystem-based management of coral reef fisheries. *Proceedings of the National Academy of Sciences*, **108** (41), 17230–17233.
- Meinesz, A. and Biefune, A. (2015) 1983–2013: Development of marine protected areas along the French Mediterranean coasts and perspectives for achievement of the Aichi target. *Marine Policy*, **54**, 10–16.
- Micheli, F., Halpern, B.S., Botsford, L.W. and Warner, R.R. (2004) Trajectories and correlates of community change in no-take marine reserves. *Ecological Applications*, **14** (6), 1709–1723.
- Micheli, F., Benedetti-Cecchi, L., Gambaccini, S. *et al.* (2005) Cascading human impacts, marine protected areas, and the structure of Mediterranean reef assemblages. *Ecological Monographs*, **75** (1), 81–102.
- Micheli, F., Levin, N., Giakoumi, S. *et al.* (2013) Setting priorities for regional conservation planning in the Mediterranean Sea. *PLoS ONE*, **8** (4), e259038. doi:10.1371/journal.pone.00259038
- Mokhtar-Jamali, K., Pascua, M., Ledoux, J.B. *et al.* (2011) From global to genetic structuring in the red gorgonian *Paramuricea clavata*: the interplay between oceanographic conditions and limited larval dispersal. *Molecular Ecology*, **20** (16), 3291–3305.
- Molloy, P.P., Reynolds, J.D., Gage, M.J.G. *et al.* (2008) Links between sex change and fish densities in marine protected areas. *Biological Conservation*, **141** (1), 187–197.
- Molloy, P.P., McLean, I.B. and Côté, I.M. (2009) Effects of marine reserve age on fish populations: a global meta-analysis. *Journal of Applied Ecology*, **46** (4), 743–751.
- Olsen, E.M., Johnson, D., Weaver, P. *et al.* (2013) *Achieving Ecologically Coherent MPA Networks in Europe: Science Needs and Priorities* (eds K.E. Larkin and N. McDonough). Marine Board Position Paper 18. European Marine Board, Ostend, Belgium. 88 pp.

- Palumbi, S.R. (2004) Marine reserves and ocean neighborhoods: the spatial scale of marine populations and their management. *Annual Review of Environment and Resources*, **29**, 31–68.
- Pauly, D., Christensen, V., Dalsgaard, J. *et al.* (1998) Fishing down marine food webs. *Science*, **279**, 860–863.
- Planes, S., Jones, G.P. and Thorrold, S.R. (2009) Larval dispersal connects fish populations in a network of marine protected areas. *Proceedings of the National Academy of Sciences*, **106**, 5693–5697.
- Reñones, O., Goñi, R., Pozo, M. *et al.* (1999) Effects of protection on the demographic structure and abundance of *Epinephelus marginatus* (Lowe, 1834): evidence from Cabrera Archipelago National Park (West-central Mediterranean). *Marine Life*, **9** (2), 45–53.
- Russ, G.R. and Alcala, A.C. (1996) Marine reserves: rates and patterns of recovery and decline of large predatory fish. *Ecological Applications*, **6** (3), 947–961.
- Russ, G.R., Alcala, A.C., Maypa, A.P. *et al.* (2004) Marine reserve benefits local fisheries. *Ecological Applications*, **14**, 597–606.
- Saenz-Agudelo, P., Jones, G.P., Thorrold, S.P. and Planes, S. (2011) Connectivity dominates larval replenishment in a coastal reef fish metapopulation. *Proceedings of the Royal Society of London B, Biological Sciences*, rspb20102780.
- Sala, E. and Zabala, M. (1996) Fish predation and the structure of the sea urchin *Paracentrotus lividus* populations in the NW Mediterranean. *Marine Ecology Progress Series*, **140** (1), 71–81.
- Sala, E., Ribes, M., Hereu, B. *et al.* (1998) Temporal variability in abundance of the sea urchins *Paracentrotus lividus* and *Arbacia lixula* in the northwestern Mediterranean: comparison between a marine reserve and an unprotected area. *Marine Ecology Progress Series*, **168**, 135–145.
- Sala, E., Ballesteros, E., Dendrinis, P. *et al.* (2012) The structure of Mediterranean rocky reef ecosystems across environmental and human gradients, and conservation implications. *PLoS ONE*, **7** (2), e32742.
- Sasal, P., Faliex, E. and Morand, S. (1996) Population structure of *Gobius bucchichii* in a Mediterranean marine reserve and in an unprotected area. *Journal of Fish Biology*, **49** (2), 352–356.
- Schiavina, M., Marino, I.A.M., Zane, L. and Melià, P. (2014) Matching oceanography and genetics at the basin scale: seascape connectivity of the Mediterranean shore crab in the Adriatic Sea. *Molecular Ecology*, **23**, 5496–5507.
- Schunter, C., Carreras-Carbonell, J., Planes, S. *et al.* (2011a) Genetic connectivity patterns in an endangered species: the dusky grouper (*Epinephelus marginatus*). *Journal of Experimental Marine Biology and Ecology*, **401**, 126–133.
- Schunter, C., Carreras-Carbonell, J., Macpherson, E. *et al.* (2011b) Matching genetics with oceanography: directional gene flow in a Mediterranean fish species. *Molecular Ecology*, **20**, 5167–5181.
- Schunter, C., Pascual, M., Garza, J.C. *et al.* (2014) Retention and fish larval dispersal potential on a highly connected open coast line. *Proceedings of the Royal Society of London B*, **281** (1785), 20140556.
- Spalding, M.D., Meliane, I., Milam, A. *et al.* (2013) Protecting marine spaces: global targets and changing approaches. *Ocean Yearbook*, **27** (1), 213–248.
- Stelzenmuller, V., Maynou, F. and Martin, P. (2007) Spatial assessment of benefits of a coastal Mediterranean Marine Protected Area. *Biological Conservation*, **136** (4), 571–583.
- Uriz, M.J. and Turon, X. (2012) Sponge ecology in the molecular era. *Advances in Marine Biology*, **61**, 345–410.

- Vacchi, M., Bussotti, S., Guidetti, P. and La Mesa, G. (1998) Study of the coastal fish assemblage in the marine reserve of the Ustica Island (southern Tyrrhenian Sea). *Italian Journal of Zoology*, **65**, 281–286.
- Worm, B., Hilborn, R., Baum, J.K. *et al.* (2009) Rebuilding global fisheries. *Science*, **325**, 578–585.
- Zabala, M., Garcia-Rubies, A., Louisy, P. and Sala, E. (1997) Spawning behaviour of the Mediterranean dusky grouper (*Epinephelus marginatus* (Lowe, 1834) (Pisces, Serranidae) in the Medes Islands Marine Reserve (NW Mediterranean, Spain). *Scientia Marina*, **61** (1), 65–77.