

3 · *FISHERIES – Effects of marine protected areas on local fisheries: evidence from empirical studies*

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3.1 Marine protected areas for fisheries

Marine fisheries throughout the world are in serious decline due to over-harvesting (National Research Council, 2001), and management for sustainable fisheries requires effective tactics for limiting exploitation rates. Limitations based on annual stock assessments and total allowable catches calculated from these assessments can be dangerous, and marine protected areas (MPAs) are one tool to limit exploitation rates directly even when total stock size is highly uncertain (Walters, 2000). Contrary to other management strategies usually involving a suite of regulations (e.g., gear and vessel restrictions, minimum sizes, catch and effort limits, prohibited species), MPAs are easier to enforce and can be more easily understood by the public and the fishing industry (Guidetti and Claudet, 2010). Within a fisheries management context the main expected outcome of MPAs is yield enhancement, but MPAs also hold the potential to reduce impacts of fishing on marine ecosystems and to establish undisturbed, reference locations for scientific studies (Fogarty *et al.*, 2000). However, creating an MPA reduces the area that can be fished, thus potentially reducing yield. The question is therefore whether the yield in the area remaining open will increase enough to offset losses from the closed area (Hilborn *et al.*, 2004; Jones, 2008; see also Chapter 4), and whether the displaced fishing effort reduces the sustainability of remaining fishing grounds (Hilborn *et al.*, 2004). These concerns would be lessened if total catch was reduced

by a percentage equivalent to the habitat protected within MPAs (Parrish, 1999).

During recent years there has been a proliferation of studies assessing the efficacy of MPAs in rebuilding exploited populations within their boundaries. Most of them document increases in target species abundance, biomass, individual size, and egg production following the reduction or cessation of fishing (see reviews by Fogarty *et al.*, 2000; Sánchez-Lizaso *et al.*, 2000; National Research Council, 2001; Russ, 2002; Claudet *et al.*, 2008; Molloy *et al.*, 2009; Claudet *et al.*, 2010; see also Chapter 2). However, while there is general agreement about the potential of MPAs as conservation tools, attention to their effects on fisheries has grown particularly controversial (Hilborn *et al.*, 2004). Controversy is nurtured by the above-mentioned concerns of loss of fishing grounds and impacts of effort displacement, although the scarcity of empirical studies rigorously evaluating such effects is also slowing down progress in the research and implementation of MPAs with fishery management objectives (Russ, 2002; Willis *et al.*, 2003a; Hilborn, 2006; see also Chapter 7).

Contributions that may shed light on this issue come from empirical and theoretical studies. Empirical studies generally evaluate the potential fishery effects of spillover (i.e., net export of juveniles and adults: Russ, 2002) from MPAs to adjacent fished areas. Such studies usually collect data from onboard sampling of fisheries catch and effort or landings, experimental fishing, or tag-recapture experiments. Theoretical modeling studies simulate fishery effects of MPAs under different scenarios of MPA configuration, recruitment, connectivity, patterns of adult mobility, and exploitation rates (e.g. Polacheck, 1990; DeMartini, 1993; Acosta, 2002; Walters *et al.*, 2007; Moffitt *et al.*, 2009, to name some that account for movement dynamics; see also the review by Pelletier and Mahevas, 2005 and Chapter 4).

Many difficulties plague empirical studies. In particular, MPAs are commonly too small and too few to affect fisheries in ways that are detectable at the management scale (Russ, 2002). Hence, empirical studies focus on small-scale effects on local fisheries. Model-based studies do not have such constraints but rely on many assumptions and may lead to unrealistic or conflicting conclusions (e.g., Willis *et al.*, 2003a). Furthermore, most published models are neither fitted nor calibrated from real data and thus it is not possible to assess their relevance in real situations (Pelletier *et al.*, 2008). In contrast with the wealth of modeling studies and reviews about the potential applications of MPAs for fisheries, there is a

paucity of empirical studies assessing those effects (Russ, 2002; Pelletier *et al.*, 2008). This has been alleviated in recent years by both growing research efforts and sufficient time for those effects to develop in MPAs.

This chapter reviews and assesses current empirical evidence of MPA effects on fisheries. First, we briefly describe the mechanisms by which MPAs may affect fisheries. Second, we synthesize empirical evidence of those effects, essentially transfer of exploitable biomass from MPAs to fished areas and ensuing changes in catch and effort patterns. Lastly, we discuss this evidence, highlighting the strengths and weaknesses of MPAs for fisheries management based on current empirical knowledge.

3.2 Mechanisms of effects of marine protected area on fisheries

MPAs may affect fisheries if: (1) fishing mortality has been eliminated (i.e., no-take areas) or substantially reduced (e.g. gear-exclusion MPAs); and (2) if fishery restrictions are effectively enforced, and are permanent or have lasted long enough to have developed observable recovery of the exploited species within their boundaries. Recovery in this context means increase in biomass and numbers of individuals when fishing mortality is reduced (Jennings, 2001). Although recovery should ideally be assessed from historical levels, most empirical studies lack baseline data. Well-protected MPAs can then contribute to enhance or sustain nearby fisheries through biomass and egg and larval export.

3.2.1 Biomass export

It is well established that biomass of exploited species is generally higher within well-protected MPAs than on adjacent fished areas (see references above). The ideal free distribution predicts that animals should prefer to move towards areas where density is low relative to available resources if this is beneficial to their fitness (Fretwell and Lucas, 1970). Hence, increases in density of exploited species within MPAs could result in enhancement of yields in neighboring fisheries through density-dependent emigration (Tupper and Juanes 1999; Sánchez-Lizaso *et al.*, 2000; Zeller *et al.*, 2003; Abesamis and Russ, 2005), provided species can move and habitats are continuous. Additionally, random movements (Dugan and Davis, 1993; Rakitin and Kramer, 1996), ontogenetic habitat shifts (Gitschlag, 1986; Tupper, 2007), or migration to foraging or spawning areas (Kelly *et al.*, 2002; Rhoades and Tupper, 2008) from MPAs

could enhance yields in adjacent grounds. Although empirical support for yield enhancement at the stock or fisheries management scale is non-existent, evidence of enhancement at the local scale is growing, as shown below.

3.2.2 Egg and larvae export

Because MPAs can increase the reproductive output of target populations, increased recruitment from egg and larval export is anticipated to produce even larger benefits for fisheries than spillover of adults (Jennings, 2001; Russ, 2002; Moffitt *et al.*, 2009). However, such a recruitment effect is hard to detect due to the high spatial and temporal variability of larval survival and settlement, as well as the large area over which it can occur (Botsford *et al.*, 2009). This has hampered research, and few empirical studies have addressed the recruitment effects of MPAs. One example is Tilney *et al.* (1996) who combined ichthyoplankton sampling and current measurements to show that larvae of commercial Sparidae fishes protected in a South African MPA were exported to adjacent exploited areas. However, along with other similar studies, the location and contribution of those larvae to effective recruitment to fished populations was not determined. Recently, Pelc *et al.* (2009) observed that the density of recruits of an exploited sessile mollusk declined with distance to the boundaries of two MPAs in South Africa. Their results indicated for the first time that larval export can be detected above natural variability of recruitment and that it can enhance recruitment in fished areas up to several kilometers. New methods for individual identification techniques such as larval tagging or parentage analysis have provided new information on self-sustainability or connectivity among MPAs (Jones *et al.*, 2009). Using DNA parentage analysis, Planes *et al.* (2009) demonstrated that within a MPA network, 10% of the recruitment in the peripheral MPAs originated from spawners within a small, centrally located MPA. However, the ability of different genetic techniques to identify natal origins in marine species depends critically upon the levels of genetic structure within and among focal populations and the degree of sampling of potential parents (Saenz-Agudelo *et al.*, 2009). At present, and despite their intuitiveness, empirical evidence of such egg and larval export effects on fisheries remains non-existent and most research is currently focused on unraveling larval dispersal and population connectivity patterns (Jones *et al.*, 2009; Chapter 12). For this reason, we do not cover larval export in this chapter and refer the reader to the

reviews of recruitment effects of MPAs by [Planes *et al.* \(2000\)](#), and of the latest developments in the field of dispersal and connectivity in the context of MPA design by [Jones *et al.* \(2009\)](#), [Bostford *et al.* \(2009\)](#), and [Almany *et al.* \(2009\)](#).

3.3 Evidence of fisheries effects of marine protected areas

Direct evidence of spillover is given by any positive value of the net transfer of exploitable individuals from the MPA to the fished areas ([Russ, 2002](#)). Spillover effects may also be evidenced by spatial or temporal changes in catch and effort patterns in fisheries near MPAs. To evaluate the evidence of spillover on adjacent fisheries in this chapter we have selected studies for which there is evidence of recovery of biomass of the studied species within the MPA boundaries. Design factors – such as habitat type or heterogeneity – are considered only in reference to their potential effects on spillover (e.g., boundary permeability) but not on the overall effectiveness of MPAs to rebuild or sustain greater biomass within their boundaries. For MPAs explicitly created to rebuild the biomass of particular species or groups of species within their boundaries, the biomass rebuilding itself is the fisheries effect assessed. The search for empirical studies focused on the best-studied marine ecosystems and where most existing MPAs are sited: (1) tropical reefs; (2) temperate reefs; and (3) temperate soft-bottom areas.

The search yielded 41 studies, 10 in tropical reefs (Table 3.1), 17 in temperate reefs (Table 3.2), and 17 in temperate soft-bottom MPAs (Table 3.3), the latter being towed-gear-exclusion areas. Although not meant to be exhaustive, this review covers the majority of relevant studies and provides a comprehensive overview of the types of empirical evidence of fisheries effects of MPAs currently available (see Tables 3.1, 3.2, and 3.3). The reef MPAs reviewed are either no-take zones closed to all extractive activities or, more generally, MPAs that contain no-take and buffer zones; in buffer zones some artisanal or recreational fishing may be allowed. Hence, the fisheries studied were commercial or subsistence fisheries adjacent to no-take areas or to buffer areas that did not allow those fisheries. The effects of towed-gear-exclusion MPAs are generally evaluated by means of experimental fishing to assess biomass rebuilding, although in some cases effects of spillover have also been evidenced from commercial fishery data (see Section 3.1). The literature search on peer-reviewed studies in this category excluded those not fully protected against the impact of heavy fishing activities such as small trawlers

Table 3.1 *Coral reef ecosystems: observed effects of marine protected areas on fisheries from empirical studies*

Effect	Observation	Species/group (Reference)	
Spillover effect	YES	Limited evidence (only 1 of 3 species) for spillover of tagged fish biomass from MPAs to open areas	Hogfish (Tupper and Rudd, 2002)
		Limited spillover of several exploited species	Exploited reef fish assemblage (Kaunda-Arara and Rose, 2004) Exploited reef fish assemblage (Francini-Filho and Moura, 2008)
		Net transfer from MPA to fished area due to ontogenetic migration	Lobster (Davis and Dodrill, 1989)
Increased CPUE near MPA	YES	Declining gradient of CPUE with increasing distance from MPA	Unicornfish (Abesamis and Russ, 2005) Exploited reef fish assemblage (Abesamis et al., 2006) Exploited reef fish assemblage (McClanahan and Kaunda-Arara, 1996) Exploited reef fish assemblage (McClanahan and Mangi, 2000) Exploited reef fish assemblage (Rakitin and Kramer, 1996) Hogfish (Tupper and Rudd 2002)
	NO	No relationship between CPUE and distance from MPA	Nassau grouper and white margate (Tupper and Rudd, 2002)
Increased yields (total catch or catch per gear type) near MPA	YES	Higher trap, gillnet, and total catches near MPA	Exploited reef fish assemblage (Alcala et al., 2005 ; Abesamis et al., 2006)
		46% increase in total large trap catches and 90% increase in total small trap catches	Exploited reef fish assemblage (Roberts et al., 2001)
	NO	Lower total catches since implementation of MPA	Exploited reef fish assemblage (McClanahan and Kaunda-Arara, 1996)
Increased seasonal variability of catch rates near MPA	YES	Greater variability of CPUE near MPA than farther away	Exploited reef fish assemblage (Abesamis et al., 2006)

(cont.)

Table 3.1 (*cont.*)

Effect	Observation	Species/group (Reference)
Increased mean size of target species in adjacent exploited areas	YES	Declining gradient of mean size with increasing distance from MPA Unicornfish (Abesamis and Russ, 2005)
	NO	Equal or smaller size of target commercial species Steeper biomass size spectra in protected vs. unprotected areas Exploited reef fish assemblage (Abesamis <i>et al.</i> , 2006)
Effort concentration near MPA boundaries	YES	Effort aggregates near MPA boundaries because of greater catch or value of catch Exploited reef fish assemblage (McClanahan and Mangi, 2000)
	NO	Effort is dispersed to other areas because of high seasonal variability in fish availability Exploited reef fish assemblage (Abesamis <i>et al.</i> , 2006)

CPUE, catch per unit effort.

(e.g., plaice box: Piet and Rijnsdorp, 1998). In reef MPAs, fishes and lobsters are the main focus of study, although a few examine effects on multispecies catches. Effects on mollusk and crab fisheries are addressed solely in soft-bottom towed-gear-exclusion MPAs.

3.3.1 Effects on adjacent fisheries

3.3.1.1 Direct evidence of spillover

Direct measures of fish movement across MPA boundaries are done by conventional tag-and-release studies, with recaptures coming either from the fishery or from experimental fishing (e.g., Rakitin and Kramer, 1996; Rowe, 2001; Kaunda-Arara and Rose, 2004), and by sonic tracking studies (e.g., [Holland *et al.*, 1996](#); [Zeller and Russ, 1998](#); [Wetherbee *et al.*, 2004](#); [Meyer and Holland, 2005](#); [Rhodes and Tupper, 2008](#)). However, net emigration of exploitable individuals from MPAs to fished areas has been demonstrated in only a handful of cases for lobsters and reef fish, or surmised from spatiotemporal patterns of fishery recaptures (Tables 3.1 and 3.2). Spillover studies from tag-recapture experiments in coral reef

Table 3.2 *Temperate reef ecosystems: observed effects of marine protected areas on fisheries from empirical studies*

Effect	Observation	Species/Group (Reference)	
Spillover effect	YES	Mean annual emigration rates 4.2% and 0.8% in 2 MPAs (depending on density difference between MPAs and unfished areas)	Lobster (Rowe, 2001)
		Declining tags/unit effort and proportion tagged in fishery catches with distance from MPA (up to 1.5 km)	Lobster (Goñi <i>et al.</i> , 2006)
		Mean annual emigration rate 6% (years 8–17); immigration assumed negligible	Lobster (Goñi <i>et al.</i> , 2010)
		Emigration rate of 0.18–0.40/year depending on proportion of resident vs. nomadic morphs; immigration assumed negligible	Galjoen (Coracinidae fish) (Attwood and Bennett, 1994)
Contribution of spillover to catch	YES	Mean net contribution of spillover to annual commercial catch in weight = 11% (years 8–17 of protection)	Lobster (Goñi <i>et al.</i> , 2010)
	NO	Effort does not aggregate around MPA because of high cost of traveling and low exploitation near ports	Multispecies reef fish (Wilcox and Pomeroy, 2003)
Increased CPUE near MPA		Pattern of effort aggregation likley due to habitat distribution, rather than spillover	Multispecies reef fish (2 MPAs: Goñi <i>et al.</i> , 2008)
	YES	Declining gradient of CPUE with distance from MPA (up to 2 km)	Lobster (Goñi <i>et al.</i> , 2006) Sparid fish (Millar and Willis, 1999)
		Highest daily CPUE near MPA. Mean CPUE near MPA similar to farther grounds despite effort concentration	Lobster (Kelly <i>et al.</i> , 2002)
		CPUE near MPA increasing over the period of 8–16 years of protection	Multispecies reef fish (Stobart <i>et al.</i> , 2009)
		Experimental CPUE in MPA increasing towards the MPA centre	Lobster (Davidson <i>et al.</i> , 2002)
		Yields near MPA (0.5 km) higher than farther away (1 km)	Seagrass and reef fish (2 MPAs: Forcada <i>et al.</i> , 2009)
	NO	CPUE near boundary not higher than farther away	Lobster (Forcada <i>et al.</i> , 2009)

(cont.)

Table 3.2 (cont.)

Effect	Observation	Species/Group (Reference)	
Increased catch (per unit area) near MPA	YES	Catch (number) per area linearly declining with distance away from MPA	Lobster (Goñi <i>et al.</i> , 2006) Multispecies reef fish (4 MPAs: Goñi <i>et al.</i> , 2008)
		Catch/km of coastline similar near and far from the MPA	Lobster (Kelly <i>et al.</i> , 2002)
	NO	Declining catch/area gradients around MPA attributed to habitat effects	Multispecies reef fish (3 MPAs: Goñi <i>et al.</i> , 2008)
Increased seasonal variability of catch rates near MPA	YES	Greater variability of CPUE near MPA than farther away	Lobster (Kelly <i>et al.</i> , 2002); Lobster (Goñi <i>et al.</i> , 2006)
	NO	Seasonal variability of catch rates near MPA similar to those farther away	Reef fishes (Forcada <i>et al.</i> , 2009)
Increased size of target species in adjacent fishery catch	YES	Greater (and increasing over time) mean size of target species near MPA than farther away (years 8–17)	Lobster (Goñi <i>et al.</i> , 2010)
		Greater proportion of large individuals near the MPA than farther away (mean of 8–16 years)	Multispecies reef fish (Stobart <i>et al.</i> , 2009)
Changes in species composition in adjacent fishery catch	YES	Taxonomic distinctness in catches near MPA intermediate between MPA and farther fished locations (years 8–16)	Multispecies reef fish (Stobart <i>et al.</i> , 2009)
		Multivariate dispersion of catch maximum near MPA border (lowest inside MPA)	Multispecies reef fish (Stobart <i>et al.</i> , 2009)
	NO	Mean species richness and diversity similar near MPA than farther away (lowest inside MPA) (years 8–16)	Multispecies reef fish (Stobart <i>et al.</i> , 2009)
Effort concentration near MPA boundaries	YES	Effort aggregates near MPA boundaries because of greater catch or value of catch	Lobster (Davis and Dodrill, 1989; Kelly <i>et al.</i> , 2002; Goñi <i>et al.</i> , 2006; Parnell <i>et al.</i> , 2007) Multispecies reef fish (6 MPAs: Goñi <i>et al.</i> , 2008) Sparid fish (Willis <i>et al.</i> , 2003b)

Table 3.3 *Temperate soft-bottom ecosystems: observed effects of towed-gear exclusion marine protected areas on fisheries from empirical studies*

Effect	Observation	Species/group (Reference)
Increased CPUE near MPA	YES Declining gradient of CPUE with increasing distance from MPA	Multispecies fish, in particular haddock (Murawski <i>et al.</i> , 2004; Murawski <i>et al.</i> , 2005) Zuwai crab (Yamasaki and Kuwahara, 1989)
	Hypothesized because a reference area faced a parallel increase in abundance of finfish species but with a 1–3 years time lag	Herring, winter flounder, and red fish (Fisher and Frank, 2002)
Increased abundance or production inside the MPA	YES Increased production (kcal/m ² per year) inside vs. outside the MPA	Sea scallop and sea urchin (Hermesen <i>et al.</i> , 2003)
	Increased abundance (number/haul) inside vs. outside the trawl ban areas	Sea scallop (Murawski <i>et al.</i> , 2000) Several elasmobranch species (Rodriguez-Cabello <i>et al.</i> , 2008)
	Heavier adductor muscle tissue and gonads	Great scallop (Kaiser <i>et al.</i> , 2007)
	Larger fish inside the protected area compared to controls for sport fishing	Trophy fish (Blyth-Skyrme <i>et al.</i> , 2006)
	Increased abundance (mean number/tow) in before after comparison within the protected area	Herring, winter flounder, and red fish (Fisher and Frank, 2002)
Increased catch (per unit area) near MPA	YES Increased CPUE (number/pot)	Zuwai crab (Yamasaki and Kuwahara, 1989)
	Increased CPUE, eightfold (experimental trawl survey) comparing period before vs. after protection	Multispecies trawlable assemblage (Pipitone <i>et al.</i> , 2000; Badalamenti <i>et al.</i> , 2008)
	Relative abundance and biomass assessed by annual standardized dredge survey (catch, number/area)	Sea scallop (Murawski <i>et al.</i> , 2005)

(cont.)

Table 3.3 (*cont.*)

Effect	Observation	Species/group (Reference)
Increased seasonal variability of catch rates near MPA	YES Greater variability near MPA than farther away	Groundfish from trawl data (Murawski <i>et al.</i> , 2005)
Increased mean size of target species inside the MPA	YES Greater shell height within closed area than outside closed areas	Sea scallop (Murawski <i>et al.</i> , 2000)
	Increased mean size of two elasmobranch species inside the MPA	Small-spotted catshark and thornback ray (Rodríguez-Cabello <i>et al.</i> , 2008)
	Greater shell length and age within closed area than outside closed areas	Great scallop (Bradshaw <i>et al.</i> , 2001; Beukers-Stewart <i>et al.</i> , 2005)
	Greater size for mature female in protected area vs. control	Red mullet (Fiorentino <i>et al.</i> , 2008)
	NO Equal or smaller size for target commercial species	Hake, monkfish, and red mullet (Badalamenti <i>et al.</i> , 2002)
	Steeper biomass size spectra in protected vs. unprotected areas	Fish trawlable assemblage (Sweeting <i>et al.</i> , 2009)
Effort concentration near MPA boundaries	YES Effort aggregates near MPA boundaries because of greater catch or value of catch	Sea scallop (Murawski <i>et al.</i> , 2005) Multispecies fish (Murawski <i>et al.</i> , 2005)

MPAs show that most reef fishes, even highly vagile species such as carangids, have limited home ranges and low net movement across MPA boundaries (Holland *et al.*, 1996; Rakitin and Kramer, 1996; Abesamis and Russ, 2005; Meyer and Holland, 2005). Emigration rates from isolated reefs tend to be low (Cole *et al.*, 2000; Barrett *et al.*, 2009) but may be much greater where MPAs and fished areas are joined by contiguous reefs (Kaunda-Arara and Rose, 2004). Besides, spillover may be limited when the size of the MPA is large relative to fish mobility (e.g., 50 × 5 km MPA size and median mobility <2 km: Buxton and Allen, 1989). For species that are polymorphic with respect to movement, emigration rates depend on the proportion of nomadic and resident morphs

in the protected population (Attwood and Bennet, 1994) and protection in MPAs could select for individuals with the highest tendency to exhibit residential behavior ([Parsons *et al.*, 2003](#)). The potential of net emigration to offset the loss of fishing grounds in MPAs was addressed by one study. [Goñi *et al.* \(2010\)](#) estimated that 7% of the lobster residing in the Columbretes MPA emigrated every year to the adjacent fishery, where 90% of them were harvested within 1 year. The resulting spillover contributed 31–43% of the annual local catch and, when compared to the proportion of fishing area closed in the MPA, provided a net gain of over 10% of the catch in weight. This benefit was derived primarily from the fact that lobsters grew larger in the MPA before they emigrated.

3.3.1.2 Indirect evidence of spillover

Patterns of catch per unit effort The best evidence of spillover from catch per unit effort (CPUE) patterns is the increase in CPUE over time in fisheries near MPAs. This effect has now been documented for long periods (>1 decade) in the Apo ([Alcala *et al.*, 2005](#)) and Columbretes ([Stobart *et al.*, 2009](#)) MPAs (Tables 3.1 and 3.2). Declining gradients of CPUE with distance from the MPA boundaries (Figure 3.1) also evidence spillover in tropical and temperate MPAs for most, but not all, target species (Tables 3.1 and 3.2) (see Tupper and Rudd, 2002; [Forcada *et al.*, 2009](#)). Biomass spillover has also been observed around towed-gear-exclusion MPAs, as in Georges Bank where CPUE of multispecies, and haddock catch in particular, increased in fisheries adjacent to the MPAs ([Murawski *et al.*, 2005](#)), or as around the Kyoto Prefecture MPA for crab CPUE (Table 3.3) ([Yamasaki and Kuwahara 1989](#)). Also, maximum ([Kelly *et al.*, 2002](#); [Forcada *et al.*, 2009](#)) or highly variable ([Kelly *et al.*, 2002](#); [Murawski *et al.*, 2005](#); [Goñi *et al.*, 2006](#)) CPUEs near MPA boundaries reveal environmentally or seasonally driven fluctuations of spillover.

Catch Whether spillover causing these CPUE patterns translates into greater catches in fisheries adjoining MPAs is less documented. Again, the clearest evidence is the increase of fish catches over time in fisheries around tropical reef MPAs (Tables 3.1 and 3.2) ([Roberts *et al.*, 2001](#); [Alcala *et al.*, 2005](#); [Abesamis *et al.*, 2006](#)). Spillover has also been evidenced by declining gradients of catch/km² of lobster ([Goñi *et al.*, 2006](#)) and of multispecies fish ([Goñi *et al.*, 2008](#)) with distance from several Mediterranean MPAs. Similar catches close to and far from the MPA when effort concentrates and depletes CPUE near the MPA has been interpreted as a sign that spillover offsets the loss of fishing grounds ([Kelly *et al.*, 2002](#)). However, none of the above evidence assures that total catch

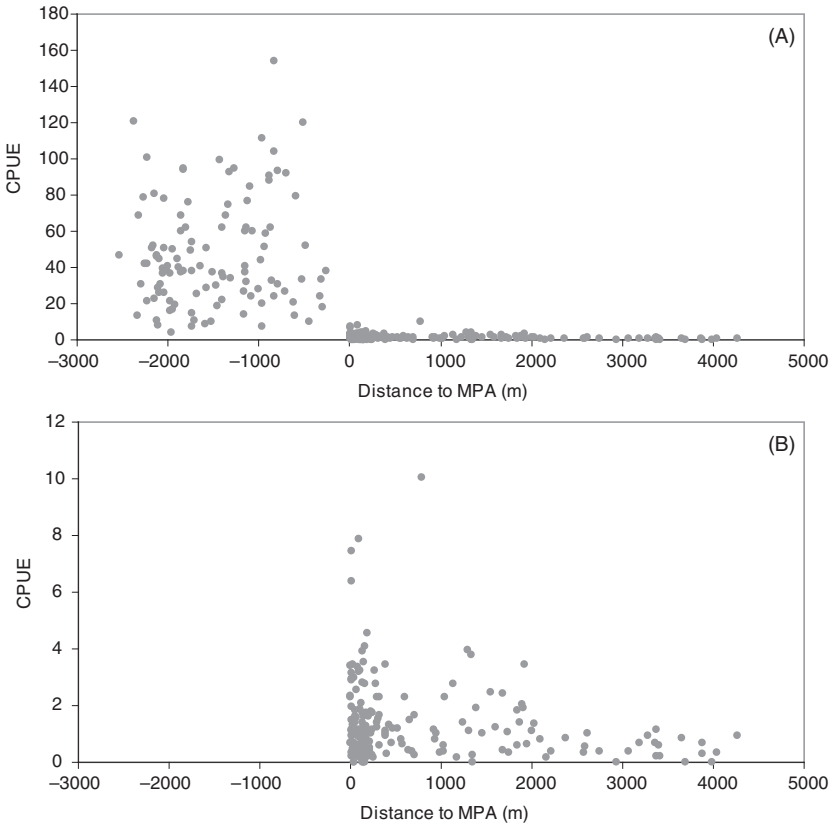


Figure 3.1 Lobster catch per unit effort (CPUE: number of lobsters caught per 600 m of net per day) versus distance from fishing set to the Columbrete Islands Marine Reserve boundary. (A) Commercial and experimental data combined; (B) Commercial fishery data on expanded y -axis scale. Adapted from Goñi *et al.* (2006).

will increase after MPA implementation. In one of the most complete appraisals of fishery changes associated with MPAs, McClanahan and Kaunda-Arara (1996) noted that following the creation of Mombasa Marine Park (Kenya), total catch decreased due to the closure of 65% of fishing grounds and subsequent relocation of fishing effort, whereas CPUE increased by 110%, catch per fisher by 25%, and catch per unit area by 83%.

Mean size in catch As exploited species are allowed to grow larger inside MPAs, spillover may also be evidenced by greater mean sizes of species

in catches of adjacent fisheries (Tables 3.1 and 3.2). Besides, since larger fishes generally have larger home ranges and therefore move greater distances ([Chapman and Kramer, 1999](#)), spillover may primarily involve the emigration of large individuals. This has been evidenced from gradients of decreasing target species size with distance from MPAs in the Philippines ([Abesamis and Russ, 2005](#)), Kenya ([McClanahan and Mangi, 2000](#)), and Spain, where mean size of fish near the boundary were intermediate between larger sizes within the MPA and smaller sizes in farther fishing grounds ([Stobart *et al.*, 2009](#)). However, and in apparent contradiction, larger individuals, which are generally more dominant, have been shown to exclude subordinate fish through interference competition in old Kenyan ([McClanahan *et al.*, 2007](#)) and Philippine MPAs ([Abesamis and Russ 2005](#)). This process has also been inferred for lobster by the mean size of migrants which, despite growing over time, always remained smaller than that of lobster residing within the MPA ([Goñi *et al.*, 2010](#)).

Species diversity in catch Whether spillover can supply a more diverse catch has been poorly studied. As a consequence of spillover processes, [Stobart *et al.* \(2009\)](#) observed that mean taxonomic distinctness of the multispecies fish catch near the Columbretes MPA was lower than within the MPA but higher than in farther fished areas. The multivariate dispersion was highest in catches of the border fishery, indicating that the border was a transitional zone between the MPA and the fishery. Unexpectedly, species richness and diversity indices were lowest inside the MPA and similar among close and far fished areas, indicating the current lack of understanding of the responses of these indices to fishing in temperate areas.

Fishing effort distribution If spillover occurs, fishing effort is expected to concentrate along the MPA boundaries – a phenomenon known as “fishing the line” ([Kellner *et al.*, 2007](#)). Concentration of displaced fishing effort as a response to improved fishing opportunities near MPAs has been well documented in temperate MPAs (Figure 3.2). Aggregation of effort is documented in most reviewed studies and is occurring in all studies on lobster fisheries (Table 3.2). Moreover, effort redistribution upon MPA creation and potential concentration near the boundaries has been observed in the large soft-bottom towed-gear-exclusion zones of Georges Bank, for both the scallop and multispecies groundfish fisheries. Before protection roughly one-third of trawl effort was deployed in the closed areas and after closure 35% occurred within 5 km of the MPA (Table 3.3)

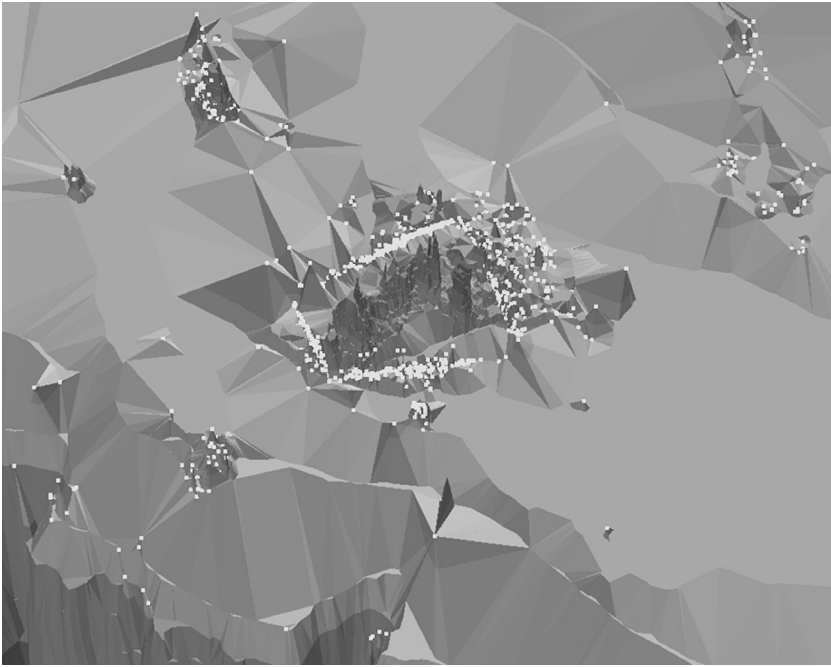


Figure 3.2 Spatial distribution of commercial effort in the regional lobster fishery around the Columbretes Islands Marine Reserve. Each dot represents a fishing set. Adapted from Goñi *et al.*, (2010).

(Murawski *et al.*, 2005). In contrast to temperate MPAs, fishing effort patterns around tropical MPAs have been documented in few studies and with contrasting results (Table 3.1). Effort either aggregated in response to CPUE increases (Mombasa Marine Park: McClanahan and Mangi, 2000), or was lower than in other grounds because of high variability of fish abundance due to seasonal changes in weather conditions (Apo Reserve: Abesamis *et al.*, 2006). Lack of effort aggregation around MPAs may also be due to high costs of traveling or availability of good fishing grounds close to harbor (e.g., Wilcox and Pomeroy, 2003).

Confounding factors It is important to note that fishing effort distribution and the other variables reviewed above related to fishery effects of MPAs do not necessarily respond to the presence of spillover. In particular, habitat distribution may drive or confound observed fishery patterns (e.g., Goñi *et al.*, 2008; Forcada *et al.*, 2009).

3.3.2 Biomass rebuilding within towed-gear-exclusion MPAs

Two towed-gear-exclusion zones aimed at rebuilding the biomass of exploited assemblages have been intensively studied and their ability to attain their objectives evaluated: the Gulf of Castellammare, a bottom-trawl ban of about 200 km² (1990), and the Georges Bank and Southern New England areas (1994) covering over 22 000 km² closed to all mobile fishing gear (Table 3.3). In the Gulf of Castellammare CPUE from experimental trawl surveys carried out before and up to 10 years after the ban showed an overall and stable increase in abundance within the MPA of about eight-fold ([Pipitone *et al.*, 2000](#); [Pipitone *et al.*, 2004](#); [Badalamenti *et al.*, 2008](#)). Interestingly, biomass size spectra showed slopes significantly steeper in the MPA than in control unprotected gulfs ([Sweeting *et al.*, 2009](#)). This unpredictable result was attributed to the exclusion of trawlers, which lacked catch-size selectivity and to the continued, more size-selective fishing by artisanal gears within the MPA. Notably, the biomass increase in the Castellammare MPA was not accompanied by substantial size-related trophodynamic shifts ([Badalamenti *et al.*, 2002](#)) and the exclusion of trawling had limited effect on fish trophic level at size, discounting a large bottom-up influence on fish trophodynamics ([Badalamenti *et al.*, 2008](#); [Fanelli *et al.*, 2009](#); [Fanelli *et al.*, 2010](#)).

In the Georges Bank MPAs the spawning stock of groundfish species increased following the MPA establishment, in particular for the shallow-water sedentary fish assemblages. The biomass and mean size of the scallop *Placopecten magellanicus* grew dramatically (14-fold) during the period 1994–1998 ([Murawski *et al.*, 2000](#)).

Some MPAs closed to trawling have the explicit objective of rebuilding biomass of specific target species. Both the Devon Inshore Potting Agreement areas of about 500 km² ([Kaiser *et al.*, 2007](#)) and the much smaller 2 km² Isle of Man MPA ([Bradshaw *et al.*, 2001](#); [Beukers-Stewart *et al.*, 2005](#)) succeeded in increasing biomass of the scallop *Pecten maximus* (Table 3.3). The Kyoto Prefecture towed gear closure aimed at protecting the crab *Chionectes opilio* accomplished its objective of rebuilding its biomass and mean size ([Yamasaki and Kuwahara, 1989](#)).

Other MPAs where towed gears were excluded with the objective of protecting particular fish species were also successful in increasing their biomass and that of other species. The Emerald/Western Bank closed area (established in 1987) on the Scotian Shelf (~13 700 km²), designed to protect juvenile haddock, also benefited several other species with up to 39-fold increases in abundance (mean number per tow) comparing

pre- and post-closure survey data (1970–2000) (Fisher and Frank, 2002). A similar trend was evident in a reference area (the Brown's Bank) but with a time lag of 1–3 years. The authors concluded that the dynamics of the Brown's Bank finfish community are coupled to the closed area through spillover. MPAs in the Cantabrian Sea closed to trawling for the conservation of elasmobranch species also experienced increases in biomass and mean size of several species, in particular of the small-spotted catshark and the thornback ray (Rodríguez-Cabello *et al.*, 2008).

3.4 Summary and discussion of empirical evidence of effects of marine protected areas on fisheries

Along with the impetus of the creation MPAs worldwide (Allison *et al.*, 1998), empirical research on their effects flourished in the 1980s. For two decades most research focused on assessing the responses of exploited species within their boundaries. By the 2000s it was well established that where properly enforced and where significant fishing restrictions had been in place for sufficient time, biomass of target species increased within MPAs. Subsequently studies investigated relationships between observed biomass responses and species life history traits and MPA characteristics, mainly size and age, combining data from several MPAs (e.g., Halpern, 2003; Russ *et al.*, 2005; McClanahan *et al.*, 2007; Claudet *et al.*, 2008; Claudet *et al.*, 2010). Positive relationships were generally found between the degree of biomass recovery and MPA size and age. Greater or faster biomass responses were associated with sedentary or moderately mobile species, although mobile species could also respond positively to protection within MPAs (Claudet *et al.*, 2010). Attention was then shifted towards investigating the effects of these recoveries within MPAs on fisheries surrounding MPAs. The aims of such empirical studies have so far been modest. Most focused on ascertaining changes in fishing effort and/or catches in commercial or subsistence fisheries near MPAs and only a few measured transfer rates from protected to fished populations.

Our review of MPAs of various types and histories and in various ecosystems indicates that in successful MPAs with permeable boundaries, spillover can induce increases in CPUE of target species in fisheries surrounding MPAs. These increases constitute a yield surplus and fisheries' CPUE tends to be higher, although often more variable due to seasonal processes underlying spillover (MacClanahan and Kaunda-Arara, 1996; Kelly *et al.*, 2002; Goñi *et al.*, 2006). Nevertheless, spillover is species and

habitat specific and when more than one exploited species was studied, not all target species CPUE increased near MPA boundaries (e.g., Tupper and Rudd 2002).

So far, few studies have explored the spatial patterns of total catch following MPA establishment. Because MPA effects on CPUE can be confounded by “leakage,” i.e., overdepletion due to relocation of fishing pressure into a smaller area (Ewers and Rodrigues, 2008), changes in total catch or catch per unit area, rather than CPUE, provide the most unequivocal measure of spillover effects on local fisheries. However, estimating total catch is more data-demanding as both spatial effort and CPUE data are required. Where reported, catches per unit area were higher near the MPAs (but see McClanahan and Kaunda-Arara, 1996). Well-enforced MPAs also lead to increases in species mean size within their boundaries (but see Badalamenti *et al.*, 2002). Consequently, mean size increases in catches outside MPAs can be observed when spillover occurs. Effects of MPAs on multispecies catch size spectra have been more difficult to ascertain, with reports of either a greater (Stobart *et al.*, 2009) or smaller (Sweeting *et al.*, 2009) proportion of large animals (as when a non-size-selective gear is excluded) in comparison to fished areas or to the same areas before protection, respectively. Nevertheless, species size in the catch can be affected by numerous factors, e.g., gear selectivity, catchability, habitat, or species behavior, and may therefore not be straightforward to obtain representative or unequivocal results.

In the medium term, most MPAs attract fishers to their boundaries. Although effort data prior to MPA establishment are usually not available (but see Murawski *et al.*, 2005), at the time of MPA creation fishing effort relocated either homogeneously around available grounds or in preferred areas outside the Columbretes Islands Marine Reserve, coalescing gradually towards its boundaries as biomass in the MPA grew, expanding and contracting thereafter as a function of fishing success on the MPA edge. Once spillover occurs, fishers continue to occupy those desirable fishing spots throughout the season regardless of catch rates. In fishing grounds not associated with MPAs, fishermen are more likely to relocate when fishing depletes the area and catch rates drop. Nevertheless, due to high costs of traveling or catch variability, aggregation of effort near MPA boundaries does not always occur.

Optimal MPA benefits to fisheries are argued for species with intermediate dispersal characteristics. Lobsters and reef fish reviewed here exhibit these dispersal characteristics and most studies reported fisheries effects consistent with spillover. In spite of this, estimates of net transfer

are scarce and range from 0.8% to 7% per year for lobsters (Rowe, 2001; Goñi *et al.*, 2010). Net benefits from spillover to local fishery catches (i.e., accounting for the loss of fishing area due to the MPA) have been estimated at over 10% per year for lobster (Goñi *et al.*, 2010).

Although only a few MPAs have been deliberately located on soft bottoms and generally consist of towed-gear-exclusion zones, they include most of the MPAs specifically designed for fisheries management objectives. The primary goal of existing soft-bottom gear-exclusion MPAs is rebuilding the biomass of exploited fish assemblages or of particular fish or shellfish species. Therefore studies evaluate their performance on those grounds rather than focusing on effects on fisheries outside their boundaries. Our synthesis indicates that all soft-bottom MPAs reviewed succeeded in doing so. Furthermore, since towed gears affect whole assemblages, positive responses on other exploited species have also been documented. With two exceptions reporting spillover, no studies assessed wider fisheries effects. Also, gear-exclusion MPAs may be expected to affect the fisheries allowed within the MPA if they harvest common species, but no study evaluating such effects was found.

Gear-exclusion MPAs have shown potential in replenishing fish and some commercial invertebrate stocks in a matter of years. They are also a useful tool for resolving conflicts among artisanal and industrial fisheries in coastal areas. When large enough with respect to the scale of fisheries and well enforced they offer a good solution for fisheries management of soft bottom areas.

The shortage of fishery data evaluating the effects of towed-gear-exclusion MPAs seems to be attributable mainly to the design features of MPAs on soft bottoms. They are fairly large compared to those on coastal reefs and tend to be located offshore. This implies higher costs to assess fishery effects of these MPAs than that needed for small coastal MPAs on reef areas. Hence, funding for large-scale projects should be encouraged.

3.5 Conclusions and future research directions

Our current empirical understanding of direct effects of MPAs on fisheries is limited by the scale of the process most amenable to study: biomass spillover. Although some studies document positive responses of highly mobile species inside MPAs (e.g., Claudet *et al.*, 2010), biomass increases are favored by the limited movement of organisms. As a result, the spatial extent of observable spillover is generally restricted to the extent of the

movements of low to moderate mobility species, that is, few hundred to thousand meters from MPA boundaries, depending on species mobility, MPA size, and habitat characteristics.

Effects of MPAs on fisheries are difficult to demonstrate empirically. A first reason is that most MPAs are small and were not designed with fisheries management objectives, while MPA success is dependent upon the initial goals laid out (Palumbi, 2001). Except for gear-exclusion zones, which often have explicit fisheries management objectives, most MPAs have been established to meet unspecified conservation benefits, precluding the choice of their location, design, and protected habitats based on specific criteria. Thus, *ad hoc* evaluation of MPA effectiveness should not be used primarily as tests of their benefits, but rather to draw recommendations for the establishment of MPAs in the future. A second explanation is the common lack of fishery data before establishment of MPAs, which often also lack appropriate spatial replication (Palumbi, 2001; [Willis *et al.*, 2003a](#); [Pelletier *et al.*, 2008](#)). Yet non-existence of appropriate replication and of independent control sites is a common issue because MPAs are frequently sited in outstanding areas or protect unique ecosystems. Unfortunately, this problem cannot be solved once MPAs are established, and this dilemma underscores the importance of conducting baseline studies before MPAs are created. In addition, only a few studies encompass evaluations of MPA effects on fisheries over large temporal scales ([Russ *et al.*, 2004](#); [Murawski *et al.*, 2005](#); [Stobart *et al.*, 2009](#); [Goñi *et al.*, 2010](#)). Thus, although there is now a wide consensus on the effectiveness of MPAs for rebuilding density and biomass of target species within their boundaries and on the benefits of spillover at local scales, uncertainty and low predictive power remain due to small sample sizes and possible confounding factors, such as habitat distribution and quality.

Another underlying process on which MPA effects on fisheries depend is connectivity. Understanding and quantifying the dispersal characteristics of populations at different life history stages remains the most difficult and challenging issue in employing MPAs, and is the least developed and the greatest source of uncertainty ([Fogarty *et al.*, 2000](#); [Jones *et al.*, 2009](#)). To effectively enhance or help stabilize fisheries, MPAs must not only preserve minimum viable populations but must also export larvae or harvestable adults to adjacent areas ([Bostford *et al.*, 2009](#)). To do so, MPAs cannot function as islands but must interact with areas open to fishing, or with reserves in the case of networks. Thus, design requirements for MPAs for fisheries management must incorporate information on species life history and ecology, larval sources and sinks, and

metapopulation dynamics. Theoretical models can help in this respect (Walters, 2000).

Even if MPA effects on fisheries evidenced by existing empirical studies seem scanty and limited in their potential to reverse the outlook of fisheries, MPAs can be used as controls to assess the effects of exploitation of marine living resources. They are an essential tool to tease out the complicated interactions between natural and human-induced alteration of these resources (Parrish, 1999). Therefore, more MPAs are needed for proper fisheries research, assessment, and management.

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