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# Models and indicators for assessing conservation and fisheries-related effects of marine protected areas

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**Abstract:**

Two kinds of approaches have been used for assessing conservation and fisheries-related effects of marine protected areas (MPAs): (i) statistical modelling based on field data and (ii) mathematical modelling quantifying the consequences of MPAs on the dynamics of populations, communities, and fisheries. Statistical models provide a diagnostic on the impact of MPAs on the ecosystem and resources; they are also needed for devising and assessing sampling designs for monitoring programs. Dynamic models enable exploration of the consequences of MPA designs and other management policies. We briefly review how each of these approaches has been implemented up to now in the literature and identify potential indicators of MPA effects that can be obtained from each approach to provide scientific advice for managers. Methodological gaps that impede the assessment of MPA effects and the construction of appropriate indicators are then discussed, and recent developments in this respect are presented. We finally propose ways to reconcile the two approaches based on their complementarity to derive suitable indicators to support decision making. In this respect, we suggest in addition that MPA managers should be associated from the beginning to the design and construction of indicators.

**Résumé:**

Deux sortes d'approches sont utilisées pour évaluer les effets des aires marines protégées (AMP) sur les pêcheries et la conservation biologique : (i) la modélisation statistique de données de terrain et (ii) la modélisation mathématique qui quantifie les conséquences des AMP sur la dynamique des populations, des peuplements et des pêcheries. Les modèles statistiques fournissent un diagnostic sur l'impact des AMP sur les écosystèmes et les ressources; ils sont également nécessaires pour mettre au point et évaluer des protocoles d'échantillonnage pour les programmes de suivi. Les modèles dynamiques permettent d'explorer les conséquences de différentes configurations de AMP et d'autres mesures de gestion. Nous faisons une brève revue bibliographique des applications de chacune de ces approches et identifions des indicateurs potentiels des effets des AMP, tels qu'ils peuvent être obtenus par chaque approche, dans le but de fournir un avis scientifique aux gestionnaires. Nous discutons ensuite des faiblesses méthodologiques qui nuisent à l'évaluation des effets des AMP et à la construction d'indicateurs appropriés et nous présentons des développements récents en la matière. Pour finir, nous proposons des pistes pour réconcilier les deux approches sur la base de leur complémentarité, afin d'en déduire des indicateurs adéquats pour aider à la prise de décisions. Dans cette optique, nous suggérons de plus que les gestionnaires de AMP soient associés dès le départ à la mise au point et à la construction des indicateurs.

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**Keywords:** Marine Protected Areas – indicators - modelling –fisheries management – ecosystem conservation

**Mots-clés:** Aires Marines Protégées – indicateurs – modélisation – gestion des pêcheries – conservation des écosystèmes

# 1. Introduction

Marine Protected Areas (MPAs) are now considered as major tools for biodiversity conservation and for fisheries management. Quantitative targets have been set in several international agendas as to the coverage of a global network of MPAs to be reached in the next ten to fifteen years in order to protect biodiversity (2002 World Summit for Sustainable Development, http://www.johannesburgsummit.org/html/documents/documents.html, Convention for Biological Diversity (CBD), http://www.biodiv.org/defaults.html). Other instruments have been mandated to assist in the designation of sites to be protected (OSPAR Convention for the Protection of the Marine Environment of the North-East Atlantic, http://www.ospar.org/fr/html/welcome.html; European “Habitat Directive”, http://europa.eu.int/comm/environment/nature/nature_conservation/eu_nature_legislation/habitats_directive/index_en.htm; the future Marine Strategy Directive, http://europa.eu.int/comm/environment/water/marine.htm). With respect to fisheries management, the new European Common Fisheries Policy (CFP) recommends the use of MPAs in the form of partial or total restrictions to fishing, and requires that the efficiency of these measures be evaluated through a set of biological, economic and social indicators (http://europa.eu.int/comm/fisheries/greenpaper). Likewise, the Program of Work for Protected Areas of the CBD called on Parties to develop and adopt appropriate methods and standards, criteria and indicators for evaluating management effectiveness and governance by 2008, and to assess at least 30% of their protected areas by 2010. In spite of the general pressure for implementing MPA networks, there remain large uncertainties as to the designation of sites according to biodiversity status, threats and constraints linked with human uses.

Because MPAs aim at "preserving specific areas together with their overlying water, substrate and associated flora, fauna historical and cultural heritage" (Kelleher and Kenchington 1992), they are considered as a central element of ecosystem-based management (Agardy 2000; Browman and Stergiou 2004). Consistently with this definition, MPAs are taken in the present article in the wider sense of an area where fishing and other human activities are restricted, even partially and/or temporarily. Management goals for MPAs are various, including conservation and heritage preservation, education and research, sustainable exploitation and promotion or control of tourism, but biodiversity conservation and fisheries management are the main motivations for MPA establishment (Boersma and Parrish 1999; Salm et al. 2000; Claudet and Pelletier 2004).

The interest of MPAs as a management tool for sustainable fisheries management is still open to debate: some argue that MPAs should not be considered as the “one size fits all” solution to all fisheries problems (Hilborn et al. 2004; Kaiser 2005), whereas others claim that global fishery declines could be reversed by implementing large-scale networks of marine reserves (Gell and Roberts 2003). For some authors, there is no overwhelming evidence on the effectiveness of MPAs for ecosystem conservation, let alone for fisheries management (Russ 2002; Hilborn et al. 2004). Assessing effects of MPA on ecosystems and fisheries in a reliable and unquestionable way is therefore important.

Methodologies for assessing the effectiveness of MPA management are developing and several initiatives are undertaken to evaluate them (Wells and Dahl-Tacconi 2006). In the present article, our interest is twofold. First, we review the methods that can be used to assess progress toward the achievement of ecosystem conservation and fisheries management objectives. Second, we discuss the ability of these methods to produce indicators for this purpose.

We particularly concentrate on fish populations and communities. By fish, we mean all macroinvertebrates and fish species, whether exploited or not. Economic and social effects, i.e. consequences of MPA on fishing and other human activities, will not be dealt within this paper, although we are fully aware of their importance (see e.g. Pelletier et al. 2005 and Perspective section).

Assessment is here understood as the main support to reliable quantitative scientific advice for management and decision-making. A key component of this scientific advice is the provision of indicators. By indicator, we mean a function of observations or of the outputs of a model, which value indicates the present state and/or dynamics of the system of interest (Food and Agriculture Organisation 1999). We are here interested at indicators aimed at testing hypotheses about conservation or fisheries-related effects of MPA. The performance of such indicators mainly lies in their statistical properties and their sensitivity to the question addressed. In order to stress the importance of validating indicators through performance criteria, we distinguish metrics (biological responses observed or calculated at a given scale) and indicators (metrics displaying desirable performances for a given purpose, in this case for testing MPA effects). Although there are a number of references about indicators of fishing effects (Trenkel and Rochet 2003; Daan et al. 2005; Jennings 2005), there are surprisingly few articles quoting indicators of MPA effects (Ohman et al. 1998; Harmelin 1999; Amand et al. 2004). Aside from the peer-reviewed scientific literature, the MPA

Management Effectiveness Initiative (MEI) formed in 2000 by the IUCN World Commission on Protected Areas and by World Wildlife Fund proposed sets of indicators to assess management effectiveness. The main objective of MPA-MEI was “to develop a set of marine-specific natural and social indicators to evaluate MPA management effectiveness”, based on both scientific and practitioner expertise. Results are presented in a guidebook aimed at helping managers and practitioners to better achieve the goals and objectives for which their MPA was created (Pomeroy et al. 2004). The guidebook provides clear insight into the construction of indicators, but it is not intended to propose assessment methods nor means to evaluate indicator performance. Thus, to our knowledge, there is currently no review and comparison of MPA assessment methods and of their ability to provide indicators of MPA effects for informing the decision making process.

In the scientific literature, assessment of conservation and fisheries-related impacts of MPAs is generally achieved via two approaches: (i) empirical approaches based on statistical modelling of field data, used to test effects and provide diagnostics; and (ii) dynamic models of populations, communities or ecosystems, generally used for policy screening and for generating hypotheses to be further tested in the field. These two approaches respectively relate to the two scales, local and system-wide, at which conservation and fisheries-related effects of MPAs operate. After MPA establishment, the ecosystem is first affected within the MPA at a local scale. In the case of Multiple Use MPAs (MUMPA), fisheries-related effects may also occur in the partially protected area. If dispersion and migration patterns bring species outside of the MPA even temporarily, effects within the MPA will in turn affect both ecosystem and fisheries at a larger scale. MPA effects should thus be assessed at each scale (Guidetti 2007). These spatial scales are associated to specific time scales, depending on ecosystem connectivity and response to exploitation.

Empirical assessment methodology is briefly discussed by Russ (2002) and Willis et al. (2003b), among others. Pelletier and Mahévas (2005) reviewed dynamic models used for MPA analysis and discussed model assumptions and their consequences on model outputs. But we could find no reference addressing and discussing together empirical assessments and dynamic modelling of MPA.

In this article, we first provide a brief state-of-the-art about the assessment of MPA effects on fish assemblages and fisheries. Assessment methods are then discussed in the light of their ability to lead to indicators of MPA effects. In a third step, we identify methodological gaps in current approaches, and propose possible ways for improving MPA assessment, in particular regarding the development of indicators tracking progress toward the achievement of management objectives. We finally investigate how the two approaches may benefit each other, with the ultimate aim of providing scientific advice to support decision-making by MPA and fisheries managers.

## 2. Current approaches to assessing MPA effects

As mentioned above, two kinds of approaches have been envisaged to assess conservation and fisheries-related effects of MPA: dynamic models depicting temporal changes in the spatial dynamics and structure of the populations, communities or ecosystems; and empirical approaches based on statistical modelling of field data. The latter should lead to defining empirical indicators and sampling designs for long-term monitoring programmes, whereas the former enable exploring issues related to MPA design and its consequences on the dynamics of populations and fisheries. These can also provide reference points against which system dynamics can be gauged.

## 3. Assessing MPA effects from dynamic models

A number of dynamic models of fisheries and exploited populations have been developed in the last decade to evaluate conservation and fisheries-related effects of MPA (Gerber et al. 2003). In the present article, we propose a classification of existing models based on the extensive review of Pelletier and Mahévas (2005), mainly to serve as a basis for the discussion on indicators (section “Which indicators for MPA assessment?”). Models were classified into five types ranging from simple to complex models. First, non spatial single species models often rely on logistic population growth and assume instantaneous dispersion of fish over the entire fishery area and uniform fishing mortality. They are used to investigate permanent no-take reserves covering a fraction of the ocean (e.g. Lauck et al. 1998; Mangel 1998; Hastings and Botsford 1999). Second, source-sink models make assumptions about larval dispersion schemes, considering local dynamics in each patch; they enable exploring no-take reserve designs in terms of number, size and location in source versus sink patches (e.g. Crowder et al. 2000; Tuck and Possingham 2000; Sanchirico and Wilen 2001). From the literature, these two kinds of models are used as conceptual tools to yield general ideas about MPA effects. A discussion about the shortcomings and advantages of simple models in theoretical ecology may be found in May (2004). In practice, these aggregated models cannot address some issues raised by policy makers regarding the management of real fisheries, such as MPA location and design of MUMPAs,. In the same line of thought, single species models ignoring the demographic structure of the population provide little insight about the effectiveness of MPAs aimed at protecting juveniles or spawners. Third, spatially-explicit demographic models depict growth, reproduction, fishing and natural mortalities, as well as fish movement (e.g. Polacheck 1990; Walters et al. 1993; Pelletier and Magal 1996). MPA designs investigated include number, size, location of MPAs, and possibly temporary closures in the case of models with intra-annual dynamics. Fourth, spatially-explicit fisheries models include additional detail concerning exploitation, in particular effort may be described in terms of gears, vessel number and characteristics, and fleet dynamics (Walters and Bonfil 1999 ; Holland 2000; Pelletier and Mahévas 2005). These models are appropriate for investigating MPA designs other than permanent no-take zones, for appraising the impact of MPA upon population structure, and they account for restoration through enhanced reproduction and recruitment. MPA designs aimed at protecting sensitive stages of populations, possibly on a seasonal basis, may be investigated, although there are few published examples (see papers cited above, and Drouineau et al. 2006). Spatially-explicit fisheries models permit in addition to investigate more elaborated policies including MPAs targeting particular fishing activities or gears, combined with other regulatory measures such as effort controls. They are also needed for exploring mixed (multispecies multifleet) fisheries issues, such as technical interactions and discards. Finally, they may account for fishers' response to policy implementation, which is particularly relevant in the case of MPAs. In general, these models do not address the consequences of MPAs at the community level. In contrast, trophodynamic models describe the state or dynamics of communities or ecosystems based on trophic interactions. Interactions are modelled at the scale of either age- and size specific fish schools (Shin and Cury 2001) or functional groups using biomass flows and mass-balance equations such as in ECOPATH (Christensen and Pauly 1992) and in ECOSPACE, a spatially-explicit and dynamic model that accounts in addition for dispersal and migration (Walters et al. 1999). Such models have been used a few times for exploring the effectiveness of permanent no-take zones (Walters 2000; Watson et al. 2000; Shin and Cury 2001). The complexity of trophic interactions makes it difficult to explicit additional model components such as demographic processes and exploitation; therefore, they cannot be easily used for exploring MPA designs and for comparisons with other management measures (but see Beattie et al. (2002) for trawl exclusion scenarios). Predator-prey models have also been developed lately (Boncoeur et al. 2002; Micheli et al. 2004a; Mangel and Levin 2005).

Simple conceptual models in the first two categories (e.g. Hastings and Botsford 1999, 2003) are aimed at understanding possible consequences of MPA and providing theoretical insight on the dynamics of resources and exploitation under MPA management. They are less demanding in term of input data and model implementation and less prone to problems of structural stability and parameter identifiability. More complex models such as spatially-explicit models, mixed fisheries models, or trophodynamic models allow to consider additional key processes, and are used for policy-screening that can form the basis for decision support systems. However, because they have more parameters and are data-driven, they are more prone to uncertainties and their outputs should be carefully analysed in this respect (see Pelletier and Mahévas (2005) for a discussion). For the purpose of defining indicators for MPA effects, simple conceptual models are helpful for appraising in a simple way consequences of MPA on fisheries and marine populations, while data-driven models will provide quantitative indicators on the dynamics of resources and exploitation in real cases. To be able to provide indicators that can be subsequently used by managers for adaptive management, models should capture the various components of fishing mortality, and the zoning of resources and fishing, so that scenarios including changes in fishing regulations (e.g. concerning gears, zones or vessels) can be evaluated.

Metrics common to all models are total catch, total abundance and total biomass. The other metrics differ depending on model state variables, assumptions, dimensions and parameterization, e.g. is the model spatially-explicit, does it consider age- or size-structure, are interspecific relationships accounted for or not, etc. For instance, total catch calculated from a demographic population model is not the same metric as total catch obtained from a trophodynamic model. Although not surprising, this stresses the fact that the way a metric is estimated or calculated influences its properties. Moderate differences in model assumptions may lead to different and sometimes contradictory results (not detailed here, see Pelletier and Mahévas (2005) for examples). Hence, defining a metric must include

the specification of the model from which it is calculated: assumptions, processes considered and parameter values. Evaluating the precision, accuracy and sensitivity to MPA effects of each metric would require to implement each model and to carry out sensitivity analyses and stochastic simulations. Ideally, the publication of a model should include such evaluations, and comparative analyses between models and metrics may then be achieved. But this is tedious and thus rarely done in practice (see Perspective section, "Possible contributions of dynamic models to empirical analyses").
Finally, it should be noted that most published models are neither fitted nor calibrated from real data (see section 'Improving methodology for MPA assessment'), thereby providing little information about the relevance and robustness of the metrics in real situations.

## 4. Assessing MPA effects from the analysis of field data

Many studies have assessed the impact of MPAs on fish populations and on marine organisms (see e.g. reviews in Russ 2002; Halpern 2003; Micheli et al. 2004b). The majority of these studies pertains to no-take reserves aimed at biological conservation in coral reef ecosystems (see García-Charton et al. 2000, Sanchez-Lizaso et al. 2000, and Planes et al. 2006) for reviews focused on Mediterranean ecosystems). In this subsection, the term reserve is preferred to MPA because published studies concern no-take reserves (unless otherwise specified). Most papers are interested in assessing direct effects of reserves, i.e. restoration of populations and assemblage structure within the reserve, which is commonly achieved by analysing biological responses (such as densities, biomasses, mean sizes, species richness and other diversity indices) to evidence differences between the reserve and a comparable zone. Early references relied on descriptive analyses (e.g. Alcala 1988), while most others use statistical modelling of biological responses. The techniques most often used are parametric and non-parametric univariate tests (Rakitin and Kramer 1996) and univariate general linear models involving design factors such as location and time (e.g. Babcock et al. 1999; Chiappone et al. 2000; Willis et al. 2003a). In such univariate models, tests are carried out separately for each metric, e.g. the density of Serranids or the overall mean size of fish. Changes in assemblage structure are examined more rarely: early papers mostly relied on descriptive multivariate methods which do not allow statistical inference (Dufour et al. 1995; Russ and Alcala 1998; Paddack and Estes 2000), although a few recent papers use multivariate inferential methods (Micheli et al. 2005; Ceccherelli et al. 2006; Claudet et al. 2006)
The examination of the literature reveals that significant differences are obtained for some particular species (e.g. Bell 1983; Paddack and Estes 2000; García-Charton et al. 2004), taxonomic families (e.g. Jennings et al. 1996; Letourneur 1996; Wantiez et al. 1997), or other groups of species, e.g. large predators (Russ and Alcala 1996; Chiappone et al. 2000). Significant differences are more likely observed when the reserve has already been in place for several years (Alcala 1988; Paddack and Estes 2000) and when fishing pressure is high before MPA establishment and outside the MPA (Côté et al. 2001; Micheli et al. 2004b). However, in a number of cases, non-significant results were obtained for a substantial number of species, genera or taxonomic families (e.g. Rakitin and Kramer 1996; Chapman and Kramer 1999; Paddack and Estes 2000), in particular in recently established reserves (Alcala 1988). This lack of significance was also pointed out by Russ (2002) and Willis et al. (2003b).
Halpern (2003) compiled the results from a large number of empirical studies and found that reserves were associated with higher values of biomass, density, mean size and species diversity, in terms of overall trends and for four functional groups including herbivores, planktivores/invertebrate eaters, carnivores and invertebrates. Micheli et al. (2004b) conducted a meta-analysis of published studies of changes in the abundance of fish assemblages within no-take reserves. They evidenced differences in species response according to trophic level, exploitation rate and duration of protection, and showed that a substantial fraction of species were negatively affected by protection, illustrating indirect effects of MPAs. Similar results were obtained by Guidetti and Sala (2007) for Mediterranean fish assemblages.
In another review about the empirical assessment of MPA, Pelletier et al. (2005) identified the metrics used for assessing conservation and fisheries-related effects of MPA, and scored their performance as indicators according to the relevance and effectiveness criteria proposed by Nicholson and Fryer (2002). They showed that several effects were still rarely addressed, mostly long-term effects, but also effects linked to trophic interactions, density-dependent changes and protection of endangered species. Effects at community level were less studied than effects at population level, and habitat-related effects were not often investigated. Many metrics have been contemplated for studying MPA

effects, but overall, few of them appear to be relevant and effective. These findings differ to some extent from Halpern's (2003) results, mainly because the analysis is restricted to inferential assessments, thereby granting more weight to statistical significance. In Halpern's study, descriptive analyses were handled in the same way as inferential ones since meta-analyses allow for use of non inferential results (whereas the statistical significance of the meta-analysis is reported). The results of Pelletier et al. (2005) corroborate and systematize other recent reviews of empirical studies which point out a low level of empirical evidence (based on statistical significance) for MPA effects (Russ 2002; Willis et al. 2003b), probably reflecting in some instances the lack of appropriate controls and poor experimental designs. Note that sometimes there may also be no MPA effects due to MPA recentness, lack of enforcement or poor MPA design. Otherwise, the lack of statistical significance may be explained by several weaknesses encountered in a number of studies. The first weakness pertains to the lack of initial evaluation, as in many studies, the initial state of the fish community was not assessed before MPA establishment. Abundance and other biological variables inside the MPA were compared to those in one or several control zones, i.e. from a Control-Impact design (e.g. Harmelin et al. 1995; Letourneur 1996). Spatial and temporal heterogeneities of ecosystems lead to confusion of protection effects with environmental effects such as the relationships between species and habitat structure (Samoilys 1988; Garcìa-Charton and Perez-Ruzafa 1999, see also below). This makes it necessary to rely on designs including measurements before and after establishment of the MPA, and inside and outside of the MPA, i.e. Before-After Control-Impact (BACI) designs (Osenberg et al. 1994; Underwood 1994). Although such designs become more frequent, there are much less published examples than for After Control Impact (ACI) designs.
A second issue relates to habitat effects. Habitat is determining for explaining the spatial distribution and structure of fish communities (McCoy and Bell 1991; Sale 1998) and should thus be accounted for when comparing biological responses in distinct zones. Relatively few assessments have explicitly considered habitat. In several instances, differences in densities were tested by habitat type (e.g. Letourneur et al. 1997). Paddack and Estes (2000) compared fish assemblages between sites while accounting for substrate composition. Sometimes, an additional factor related to habitat was included in the model, like depth (Bell 1983; Garcìa-Rubies and Zabala 1990; Kelly et al. 2000), reef type (Chapman and Kramer 1999), geographical orientation (Micheli et al. 2005), or some other definition of habitat (McCormick and Choat 1987; Castilla and Bustamante 1989; Ferraris et al. 2005). Recent works by Garcìa-Charton et al. (2004), Micheli et al. (2005) and Claudet (2006) show how habitat effects and protection effects may interact and complicate the evaluation of conservation effects of MPA. These studies indicate the importance of explicitly considering habitat effects to avoid spatial confounding and to increase the strength of statistical inference when assessing MPA effects.
The third issue relates to the diagnostic of MPA effects. Direct effects are in general assessed by comparing densities, biomasses, mean size or diversity indices, between the reserve and the unprotected area. Statistical tests are carried out independently for some species or species groups of interest. These results are helpful for better understanding the response of some particular species to reserve protection, but they do not provide a synoptic view of the impact of the reserve. Besides, they do not allow comparing the sensitivities of different fish community components to protection, thereby impeding the construction of ecosystemic indicators for MPA assessment. Evaluating effects at the fish community level would be more desirable to provide scientific elements for an ecosystem approach to management (Botsford et al. 1997; Jennings and Kaiser 1998) in the context of MPAs.

# 5. Which indicators for MPA assessment ?

## 5.1. Candidate indicators

In the light of the previous section, we compiled a number of metrics that can be considered as candidate indicators for the MPA effects of interest here: protecting critical spawning stock biomass, rehabilitating population demographic structure, restoration of or changes in assemblage structure, exportation of biomass, protecting biodiversity, indirect effects on algae and invertebrates, enhancing fisheries yield, increasing population stability and resilience (Table 1). Model-based indicators are based on outcomes from dynamic models while empirical indicators rely on the analysis of field data.
The properties of model-based indicators depend on model assumptions and on the algorithms used (see previous section). Accordingly, data requirements depend on the model. Most dynamic models involve describing population dynamics and exploitation. Population dynamics requires biological parameters and, in the case of spatially-explicit models, information on the spatial distribution and

movements of organisms. Exploitation is parameterized through effort and catch data regarding the resources at stake. These data may be available in many documented fisheries, although information on spatial dynamics is often lacking. But they will be nonexistent in many countries or for highly multispecific fisheries. This is particularly problematic as many MPA issues arise in coastal areas where exploited species are numerous. The cost of gathering the information needed for parameterizing and calibrating these models is in general high since information has to be collected at the scale of the fishery for both resource and exploitation. To achieve a given precision in model-based indicators, this cost is determined by the sensitivity of model-based indicators to uncertainties in inputs and to changes in exploitation following MPA implementation, which totally depends i) on model structure and model complexity, and ii) on the precision and accuracy of model parameter estimates. The sensitivity of single-species fisheries models to such uncertainties and their implications for assessing alternative management options have long been studied (Pelletier and Laurec 1992; Kell et al. 2006), but in the case of MPA, studies are scarce (Drouineau et al. 2006). We are aware of no such studies for multispecies models.

With regard to empirical indicators, their properties depend on both experimental design and observation techniques. For example, overall fish abundance estimated from Underwater Visual Censuses (UVC) following a given observation design is not the same metric as overall fish abundance obtained from experimental fishing, or from UVC but from another design. The cost of gathering the information needed for calculating empirical indicators depends on the precision required for the indicator, which in turn depends on the size of the effect to be detected, on sample size, and on the sensitivity of the indicator to this effect (i.e. the relevance of the indicator according to Nicholson and Fryer (2002)). There is thus no general rule as to the cost of data collection for empirical indicators, although general qualitative guidelines are given in Pomeroy et al. (2004). Hence, if a given set of information may be used to estimate several indicators, it will thereby result in decreased costs per indicator. For instance, fish density, biomass, and mean size may be estimated from the same set of UVC data. This possibility should be taken into account when selecting empirical indicators. Another option is to prefer indicators based on data less demanding in terms of observation skills or human resources. For instance, Graham et al. (2005) proposed size-spectra at the scale of the fish assemblage as an indicator of the effects of fishing on coral reef fish assemblages, size estimation being an easily trainable technique.

Capturing the various effects of MPA requires several indicators. Regarding fisheries management, metrics based on spatially-explicit stage-structured fisheries models are necessary to appraise the consequences of MPA designs and adjunct fisheries management measures at population level. Abundance, biomass and age distribution indicate population state, while asymptotic growth rate, risk of collapse, and abundance variations reflect its dynamics (see Pelletier and Mahévas (2005) for other metrics of population and catch dynamics). While most dynamic models focus on abundance and catch or yield, yield enhancement is not often studied from field data, and there is still little empirical evidence for overall yield increase following MPA establishment (McClanahan and Mangi 2000; Russ et al. 2004; Goñi et al. 2006). As underlined by Russ (2002), “There is a plethora of reviews on what marine reserves could do as fisheries management tools. Yet there is a distinct paucity of empirical studies demonstrating what they can do.” As for conservation issues, these are mostly tackled through experimental field data. Total biomass, density of fished species, predator density, size distribution per species, and to a lesser extent total species richness were the most effective families of metrics in Pelletier et al. (2005) (Table 1). Several empirical metrics may be relevant for a given effect, and their performance needs to be further assessed. In addition to species richness which depends on the extent of the area surveyed, we suggest that other diversity indices should be more often used. In the case where fishing is not totally prohibited within the MPA (e.g. MUMPA), CPUE provide interesting metrics if their calculation yields accurate abundance indices, which requires good appraisal of fishing effort location and magnitude. As for dynamic models, models accounting for species interactions are necessary to evaluate effects at community level, e.g. trophic cascade effects (Pinnegar et al. 2000). Metrics resulting from these models include catch and biomass levels and profiles over trophic groups.

### 5.2. Scale dependency

Although apparently similar in essence (i.e. mostly based on abundance indices), model-based and empirical indicators differ because they pertain to distinct spatial, temporal and ecological scales (see Hastings (2004) for a discussion on the disconnection between time scales in empirical and theoretical studies). On the one hand, empirical approaches are strongly constrained by observation scale. Most studies rely on UVC or experimental fishing, are conducted locally and provide snapshot observations.

But these data enable observing every species and estimating metrics at community level. On the other hand, dynamic models focus on system-wide effects, whether the system refers to the fishery, the fish community or a single fish population. They encompass a longer time horizon, and may integrate ecological scales from the individual up to the community. They thus provide insight into the sustainability of resources and exploitation and possible yield enhancement through MPA which are not local issues.
MPA effects could be studied from large-scale surveys or joint analyses of large-scale data (e.g. Jennings et al. 2002; Link et al. 2002, Methratta and Link 2006). Increasing the temporal range of the study would require accounting for dynamics in empirical assessments, and adapting sampling designs.

## 6. Reference points

Sainsbury and Sumaila (2002) and others define reference points as desired targets and limits for an indicator. The availability of reference points with threshold and limit values is considered as a desirable property for indicators (Nicholson and Fryer 2002; Rochet and Trenkel 2003).
In the case of dynamic modelling approaches, reference points may be provided through the model. The direction and magnitude of changes in indicators as a function of exploitation controls (e.g. MPA design) may be studied. As such, models provide indicators that are appropriate to assess fisheries-related effects at the scale of the whole fishery system. However, indicators and reference points are entirely dependent on the model.
In the case of empirical assessments, there is no such provision of reference points. In our view, this is not problematic because the role of a reference point is to provide values against which indicators can be gauged in order to provide a diagnostic. In this respect, reference points are provided by values of indicators in control areas, i.e. before MPA establishment and outside the MPA. Although empirical indicators do not allow to anticipate about future changes in the ecosystem and fishery, they may still be appropriate to address local issues in fisheries management, e.g. artisanal fishers operating in the vicinity of the MPA or in a MUMPA.
Finally, MPA can (and should) themselves become in the long-term control areas for the evaluation of population and ecosystem effects of fishing (the role of MPA as a natural laboratory). They then allow for experimental approaches (including actively adaptive management) at the ecosystem scale (Walters 1997; Castilla 2000). However, the ecosystem restored in the MPA may not resemble the pristine ecosystem, due to the irreversibility of changes. Historical data (Jackson et al. 2001) coupled to monitoring before and after MPA establishment may then help to understand the causes of ecological changes at several scales, both retrospectively and within the MPA.

## 7. Improving methodology for MPA assessment

The examination of the current status of MPA assessment and previous discussion about possible indicators of MPA effects reveals that methodologies for assessing conservation and fisheries-related effects of MPA could be improved. This would facilitate estimating appropriate indicators. Concerning empirical assessments of MPA effects, experimental design is a major area of improvement. There are still few examples of properly replicated sampling designs in MPA assessment (Fraschetti et al. 2002). Because many different processes operate simultaneously to generate spatial and temporal variability in populations, assessing these effects requires multifactorial sampling designs. Beyond-BACI designs provide such a framework (Underwood 1992, 1994; Osenberg et al. 1994). Inference is possible with these designs if data are sampled at several times before and after MPA establishment, both within the MPA and in several control locations outside. Multiple controls are necessary to avoid confounding natural spatial variability with MPA effects or missing other consequences of management. The significance of the difference between MPA mean and the mean over control locations is then assessed with reference to the natural variability of the system, estimated by the differences among controls. In contrast, using a single control location may lead to erroneous assessments. Edgar et al. (2004) and Kendall et al. (2004) provide recent examples of baseline assessment before MPA establishment or enlargement. There are also several examples of assessment of MPA effects based on ACI designs with multiple control sites (Garcia-Charton et al. 2004; Micheli et al. 2005; Ceccherelli et al. 2006).

A second area of improvement relates to habitat considerations. Habitat is a crucial source of spatial variability for fish communities (Sale 1998). Ignoring habitat when assessing MPA effects results in increased residual variability and less statistical power. Sampling designs should account for such confounding factors, and ideally habitat should be monitored at the same time as fish communities (García-Charton et al. 2000; Stewart-Oaten and Bence 2001). Information on habitat and more generally on the different components of spatial variability should be introduced in models, thereby reducing variability (García-Charton et al. 2004). In Ferraris et al. (2005), multivariate habitat data collected at the same time as fish counts were incorporated in the model for assessing the impact of fishing in a reserve. This approach is suited for considering habitats at small scale and for taking into account several variables in the definition of habitat. In this case, habitat data can be used for defining habitat types based on multivariate and cluster analyses. Alternatively, the survey of fish and macrofauna can be a priori designed to cross protection factors and habitat factors. This is for instance appropriate when the habitat proxy is monofactorial and at an intermediate scale that is compatible with design and replication requirements (e.g. Micheli et al. 2005).

Concerning dynamic modelling approaches, the usefulness of mathematical models for evaluating MPA effects is sometimes challenged: "theoretical models are useful in developing our ideas, but they are just that: ideas" (Willis et al. 2003b). From the existing literature, we generally agree that many models are theoretical contributions, and that simple models published in well-known journals may have resulted in simplistic prescriptions (e.g. about the minimum size of no-take zones needed to protect fisheries resources), which in turn may hamper progress in marine conservation (Agardy et al. 2003). However, models are remarkable tools to evaluate MPA consequences at the scale of fisheries and ecosystems. To our opinion, the main area of improvement for these approaches lies in the development of models that achieve a trade-off between parsimony and complexity, and are parameterized and calibrated against real data. More specifically, models are needed that explicit the spatial dynamics of population and exploitation at the scale of MPA design, including the seasonal scale when relevant (e.g. for temporary restrictions on fishing). Models should account for mixed fisheries and for fishers' response to MPA. They should allow for thorough investigations of MPA designs including permanent versus temporary MPAs, partial restrictions of fishing activities, and reserve networks. They should also provide for other management measures as MPAs are not the only management tool used in a given fishery. Several of these points were already raised by Sumaila et al. (2000). Pelletier et al. (2001) and Mahévas and Pelletier (2004) proposed a model (http://www.ifremer.fr/isis-fish) that incorporates most of these features. The model was compared with existing models in Pelletier and Mahévas (2005) and applied to the Bay of Biscay mixed fishery by Drouineau et al. (2006) who found that several MPA designs and improvement in trawl selectivity resulted in increased abundances for the modelled resources.

In order to be able to calibrate models against real data, appropriate information is needed at the scale of the ecosystem and fisheries. Knowing the spatial dynamics of population demographic stages, including early stages, is necessary, and some of these aspects are unfortunately still poorly known, but the need for a better appraisal of the spatial dynamics of exploitation should also be emphasized. Conventional fisheries statistics provide information with good spatial and seasonal coverage, but their interpretation may be difficult and spatial resolution may be limited (Verdoit et al. 2003). Additional information can be obtained through vessel monitoring systems (Murawski et al. 2005) and fishers interviews. Recent research projects (http://ec.europa.eu/research/agriculture/projects/qlrt_2001_01291_en.htm, http:// www.mpa-eu.net/, http://www.um.es/empafish) have focused on these questions. In any case, the model should be used in order to account for uncertainties, whether through simulation designs (e.g. Drouineau et al. 2006) or other techniques e.g. risk analysis.

These modelling issues underpin the construction of model-based indicators, as reliable model outputs require models that are grounded with respect to real data.

# 8. Perspectives

## 8.1. The complementarity of empirical and model-based indicators

There is currently a gap between empirical studies and dynamic modelling approaches concerning MPA assessment. The latter may be deemed too theoretical by field ecologists (see previous subsection), whereas field data collected for MPA assessments are generally not used by modellers to calibrate models. Yet, in our view, dynamic models are indispensable to evaluate MPA consequences at the scale of fisheries and ecosystems, and to provide corresponding indicators. The development of more realistic models should reduce this gap. We believe that empirical and model-based approaches to MPA assessment are complementary for several reasons (Table 2).

First, empirical approaches provide indicators that have traditionally focused on biodiversity conservation, the interest for fisheries management being more recent. Biodiversity may be directly observed at local scales from a variety of metrics whereas indicators for resources and fisheries need to account for the dynamics at the fisheries level and for the spatial distribution of fishing effort, which is not a trivial issue for constructing empirical indicators. Therefore, conventional fisheries abundance indices are more often based on fisheries-independent surveys at the scale of the fishery (e.g. bottom trawl surveys used in assessment working groups in the International Council for the Exploration of the Sea (ICES, http://www.ices.dk/iceswork/acfm.asp?topic=workingroups, e.g. Anonymous (2006)). Indices have also been calculated from models integrating population dynamics and fishing effort, e.g. models used for stock assessment in many international fishing agencies (ICES, Inter-American Tropical Tuna Commission). Recently fishery data collected at fine spatial scales have been used to analyse the effects of MPAs on fishing effort and catch (Murawski et al. 2005; Stelzenmüller et al. 2007; Gõni et al., unpubl. data[1]).

Second, empirical approaches mostly provide indicators of the status of fish communities and resources. When inferential, these approaches lead to diagnostics based on statistical testing of hypotheses about indicators. In contrast, approaches relying on dynamic models enable building indicators of the dynamics of resources and fish communities. They generally do not provide for inferential diagnostic but allow comparing the current assessment to alternative situations corresponding to other fishing pressures and other population dynamics. In some instances, the dynamics inherent in the model may lead to the definition of reference points (Mace 1994). Note that in empirical approaches, reference points may also be defined and estimated but in general require a long time series of data to appraise a variety of states of nature over time.

Third, alternative hypotheses about management and states of nature may be explored using dynamic models. Previous workshops with MPA managers (Pelletier 2005) evidenced the importance of testing scenarios for MPA managers: what would happen if the zoning in the MPA were modified, or in the case of changes in controls on fishing activities either within or outside the MPA? Obviously, these questions cannot be addressed through empirical approaches unless under a large-scale manipulation in the framework of adaptive management (Walters 1997). In addition, assessing and comparing the consequences of management measures other than MPA from empirical approaches is not easy neither from ecological or social perspectives (some measures may have harmful consequences), nor from a scientific standpoint (replicates would be needed).

Last, empirical indicators are directly tied to monitoring. Sampling protocols may then be optimised or at least adapted to requirements on indicators' precision and accuracy (Benedetti-Cecchi 2001).

## 8.2. Possible contributions of empirical analyses to dynamic models

Empirical analyses provide ground truth that could be used for validating dynamic models. Hence, abundance indices collected during monitoring programmes could be confronted to abundances calculated from dynamic models to help model calibration. For instance, UVC and catch data obtained from a monitoring programme in a Corsican MPA have been used to calibrate an application of the ISIS-Fish model (D. Pelletier, unpubl. Data, http://www.liteau.ecologie.gouv.fr/rubrique.php?id_rubrique=25). This is achieved by calibrating model outputs with respect to field data by e.g. minimising the discrepancy between model outputs and data through the implementation of numerical algorithms such as the simplex or genetic algorithms (S. Mahévas, unpubl. data).

For this purpose, it is important that model scales are consistent with the scale at which ecological and fisheries data are collected. However, models should also be considered as tools for integrating knowledge and local information may be incorporated in a larger-scale model, thus illustrating the potential for scale transfer through modelling. This scale transfer refers not only to spatial aspects, but also to the integration of information about system components, and to temporal integration.

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[1] Goñi R., Adlerstein S., Álvarez-Berastegui D., Forcada A., Reñones O., Criquet G., Polti S., Cadiou G., Valle C., Lenfant P., Bonhomme P., Pérez-Ruzafa A., Sánchez-Lizaso J.L., García-Charton J.A., Bernard G., Stelzenmüller V., Planes, S

In particular, empirical approaches provide for estimations of variance components and effect sizes, i.e. the magnitude of the MPA effect that is to be detected through monitoring (Underwood 1997). These estimates may be used on dynamic models to quantify spatial and temporal variability.

## 9. Possible contributions of dynamic models to empirical analyses

The spatial and temporal integration of population and exploitation dynamics provided by dynamic models may be interesting for empirical approaches for several reasons. First, as the dynamic models used for MPA assessment are generally fishery-oriented, they can provide estimates of fishing intensity to be incorporated in empirical analyses. Proxies for fishing intensity are more readily obtained from models than from direct measures of fishing effort which raise the problem of effort standardization. The need to account for fishing intensity in empirical assessments of MPA effects has been underlined in several instances (Russ 2002, Micheli et al. 2005), but to our knowledge, no assessment quantitatively accounting for fishing pressure has been published yet. This may also be explained by the fact that the majority of studies deals with binary comparisons of no-take versus unprotected areas. As many MPAs established are indeed MUMPAs with a range of fishing controls following a zoning plan, the need for assessing MPA effects at multiple protection levels will likely increase in the near future.

Second, as they integrate several processes affecting fish populations, both biological and fisheries-related, mathematical models allow investigating key processes of dynamics. This may be useful for building hypotheses about MPA effects and spatio-temporal patterns of resources *at model resolution*; the latter being generally chosen to capture population and exploitation features. Preliminary modelling work may thus facilitate to some extent the design of monitoring programmes, in particular to account for seasonal and spatial variations in resources and fishing. Identifying which specific parameters of dynamic models are sensitive and thus require good estimation would be valuable to set up priorities for empirical studies. Which parameters are important largely depends on the model, so that it is difficult to derive general guidelines on the matter. Conducting sensitivity analysis when exploring different conservation strategies is rarely done in practice. In the case of complex simulation models, sensitivity analysis must rely on simulation experiments involving combinations of the scenarios tested, e.g. a range of MPA designs, and of possible parameter values for the uncertain parameters of interest. Appropriate simulation designs may be constructed to reduce the number of simulations. Statistical analyses adapted to the design are then used to interpret simulation outcomes and conclude to the interest of each scenario while taking into account uncertainties (Saltelli et al. 2000). These techniques were used by Drouineau et al. (2006) to compare an MPA with other fisheries management options in the case of the Bay of Biscay mixed fishery while identifying critical biological parameters.

More generally, the confrontation of empirical analyses and dynamic models is easier if the scales of data collection and model resolution are compatible. Because empirical approaches mostly pertain to local assessment, their integration with models would be facilitated by large-scale surveys and regional approaches. In addition, moving from local assessment to system-wide assessment is particularly needed in the perspective of monitoring MPA networks.

In the light of the previous considerations, framing methodological issues in the perspective of designing valid indicators of MPA effects would be beneficial, as it would guide toward improved statistical assessments and more realistic models, with the view of setting up monitoring programmes. In particular, the issue of shifting baselines (Jackson et al. 2001) could be explicitly handled in collaboration with MPA managers.

Compared to other fisheries management questions which up to now mostly rely on model-based approaches, experiences with empirical assessments of MPAs shed different light into indicator issues. They illustrate the multiple sources of variability inherent in the data that may preclude the detection of fishing or protection effects, which is of primary importance for constructing reliable indicators (Nicholson and Jennings 2004). In particular, they show the consequences for the estimation of fish abundance of several nested sources of spatial variation, in relation with (i) changes in habitat occurring at several scales, from local substrate to essential habitats, and (ii) anthropogenic pressure ranging from variation in fishing effort at the fisher level to regional regulations via MPA zoning.

## 10. MPA management

As mentioned in the introduction, the present article was deliberately oriented toward the identification of indicators for testing hypotheses about MPA effects. This is a first step, as conservation and fisheries-related effects of MPA are not yet fully understood and evidenced, and there is no consensus about what MPA can overall achieve for ecosystems and fisheries.
Our study did not include economic and social effects of MPA on fishing and other human activities. However, previous work reviewing field analyses of economic and social effects of MPA (Pelletier et al. 2005) and bioeconomic models (Pelletier and Mahévas 2005) leads us to think that much of the above considerations remains valid for the assessment of the socio-economic effects of MPA implementation through either mathematical models or empirical evidence. At present, the relative scarcity of empirical studies concerning these effects is problematic for discussing indicators further.
Many MPAs are being established or in project and new monitoring programmes should avoid the flaws already identified. Because indicators are derived from observations, whether directly (empirical indicators) or indirectly (model-based indicators), their information requirements should form the basis for designing monitoring programmes.
Indicators for MPA-based management are not only spatial versions of indicators that may be used for other fisheries management issues. Although models rely on conventional population dynamics, exploitation features and fishers' behaviour, the variety of possible MPA designs make it a complex multidimensional issue compared to other management measures. Empirical approaches illustrate the diversity of ecological effects of MPA, and consequently several indicators are needed. We proposed a list of indicators for several key MPA effects; they could be implemented, and their performance should be further evaluated.
In a second step, these indicators should be used by managers for improving the effectiveness of MPA management strategies, e.g. through adaptation of zoning or regulations, and also for reporting on MPA effectiveness (then referring to the accountability of management, Ehler 2003).
The transition toward indicators for decision support would require integrating them in the light of MPA management objectives while accounting for performance criteria linked to decision making (Figure 1). Management objectives should thus be made explicit by MPA managers and fisheries managers, so that scientists are able to formalize them and propose appropriate indicators (Claudet and Pelletier 2004). Indicators could be scored by managers, and other stakeholders with the help of scientists. Discussions about indicators for MPA assessment have for instance been organized in the Liteau-AMP research project (Pelletier 2005) during several workshops gathering MPA managers and scientists. More generally, involvement of stakeholders is to be developed. This is for instance part of the programme of work of the ongoing European research projects PROTECT (http:// www.mpa-eu.net/) and EMPAFISH (http://www.um.es.empafish/). In other contexts, the collaboration of local communities to MPA monitoring provides additional knowledge and contributes to a better social acceptance of MPA projects (Wells and White 1995; Elliott et al. 2001). Developing indicators for a better assessment of MPA effectiveness thus requires not only methodological contributions but also a clear appraisal of management objectives and of human resources allocated to monitoring.

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Table 1. Synthesis of candidate indicators for conservation and fisheries-related effects expected from MPA establishment. Additional metrics were suggested (in italics) when no metric could be identified from the literature, or when they were obviously underrated from existing literature. Unless specified, model-based metrics may be obtained from several models (see text for discussion). Short-term refers to the year scale (1-3 yrs) and medium-term refers to generation time.

| Time scale | Effects | ***Empirical indicators*** | ***Model-based Indicators*** |
|---|---|---|---|
| Short-term effects | Protecting critical spawning stock biomass | total biomass, biomass per family, total density, density of fishable species, density per trophic group, family, or species stage<br>size distribution of species<br>*biomass per species or genus, density per species or genus, CPUE per species* | biomass (total or per patch)<br>abundance (total, per patch or per subpopulation)<br>spawner abundance and biomass<br>asymptotic growth rate[1]<br>risk of population collapse |
| | Rehabilitating population demographic structure | size distribution of species<br>*mean size per species or genus* | spawner abundance and biomass<br>stable age distribution[1] |
| | Restoration of / Changes in assemblage structure | *biomass per species or genus,*<br>density profile per species<br>*species richness per family* | catch or biomass per community component[2]<br>size or biomass spectra |
| Medium-term effects | Exportation of biomass | movement patterns, home range, site fidelity | abundance (per subpopulation or per patch)<br>catch per patch<br>biomass (per subpopulation or per patch) |
| | Protecting biodiversity | total species richness<br>*other diversity indices* | catch or biomass (total or per component)<br>size or biomass spectra |
| | Indirect effects on algae and invertebrates | benthic cover<br>*density per species or genus* | *abundance of invertebrates*<br>*abundance of algae* |
| | Enhancing fisheries yield | *CPUE per species* | catch (total or per fleet), catch variation<br>equilibrium yield[3], short-term yield<br>effort-related metrics, economic metrics |
| | Increasing population stability and resilience | *density variation*<br>*CPUE variation* | risk of population collapse<br>asymptotic growth rate[1] |

[1] for Leslie models
[2] trophodynamic models
[3] yield per recruit model

Table 2. Complementary aspects of empirical studies and dynamic modelling approaches with regard to the assessment of MPA effects and subsequent provision of indicators.

| Empirical studies | Modelling approaches |
|---|---|
| Local assessment | System-wide integration (fisheries scale/ecosystem scale/coastal management scale) |
| Snapshot information | |
| | Integrate knowledge about system components |
| | Transfer of local knowledge at the scale of the system |
| Formal testing of effects | Quantitative assessment of system dynamics |
| Field validation | Exploration of scenarios (e.g. MPA design) and hypotheses |
| Actual estimates of variance components and effect sizes | Possible projections in the future |
| | Generate hypotheses to be tested from field experiments |
| Control sites | Provision of theoretical reference points |
| Direct link with monitoring | Overall diagnostic on system |

## Figures

Figure 1. Conservation and fisheries-related indicators for decision-making about MPAs. Only ecosystem conservation and fisheries management issues are considered. It illustrates the multiplicity of management objectives within each general objective, the multiplicity of possible controls for management design. Several indicators are thus needed for assessing the performance of MPA for ecosystem conservation and for fisheries management.