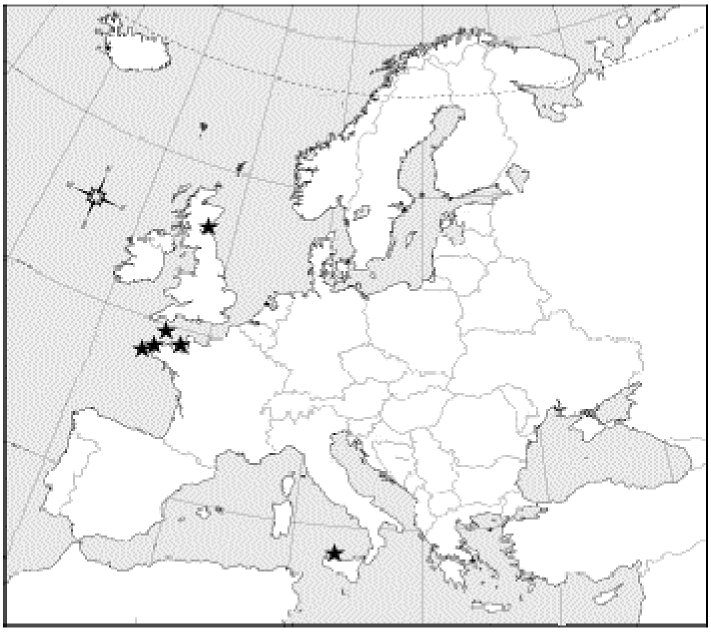
Value of Exclusion Zones as a Fisheries Management Tool in Europe

A strategic evaluation and the development of an analytical framework (QLK5-CT1999-01271)

D3 – Frameworks of Analysis



Evaluation of frameworks of analysis employed in studies of exclusion zones

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1. Introduction

The purpose of this Work Package is to examine the various approaches to analysing fishery exclusion zones (FEZs) and to identify the circumstances in which one approach might be preferred to another. Our concern here is not so much with answering questions about exclusion zones - these being dealt with in later Work Packages - as with articulating the questions themselves and in understanding how in principle they could be addressed. An important theme is the precision with which questions need to be answered, since this will determine the type of information collected and how such information is analysed. The question *' is an exclusion zone likely to improve the condition of this fishery ?'* is less precise and less demanding of data than the question *' by how much will an exclusion zone improve this fishery ?'* since it could in principle be answered by expert judgement rather than quantitative analysis. In practice fisheries managers may be confronted with situations where decisions have to be made quickly, and qualitative answers may be the only thing possible in circumstances where data cannot be obtained in the available time.

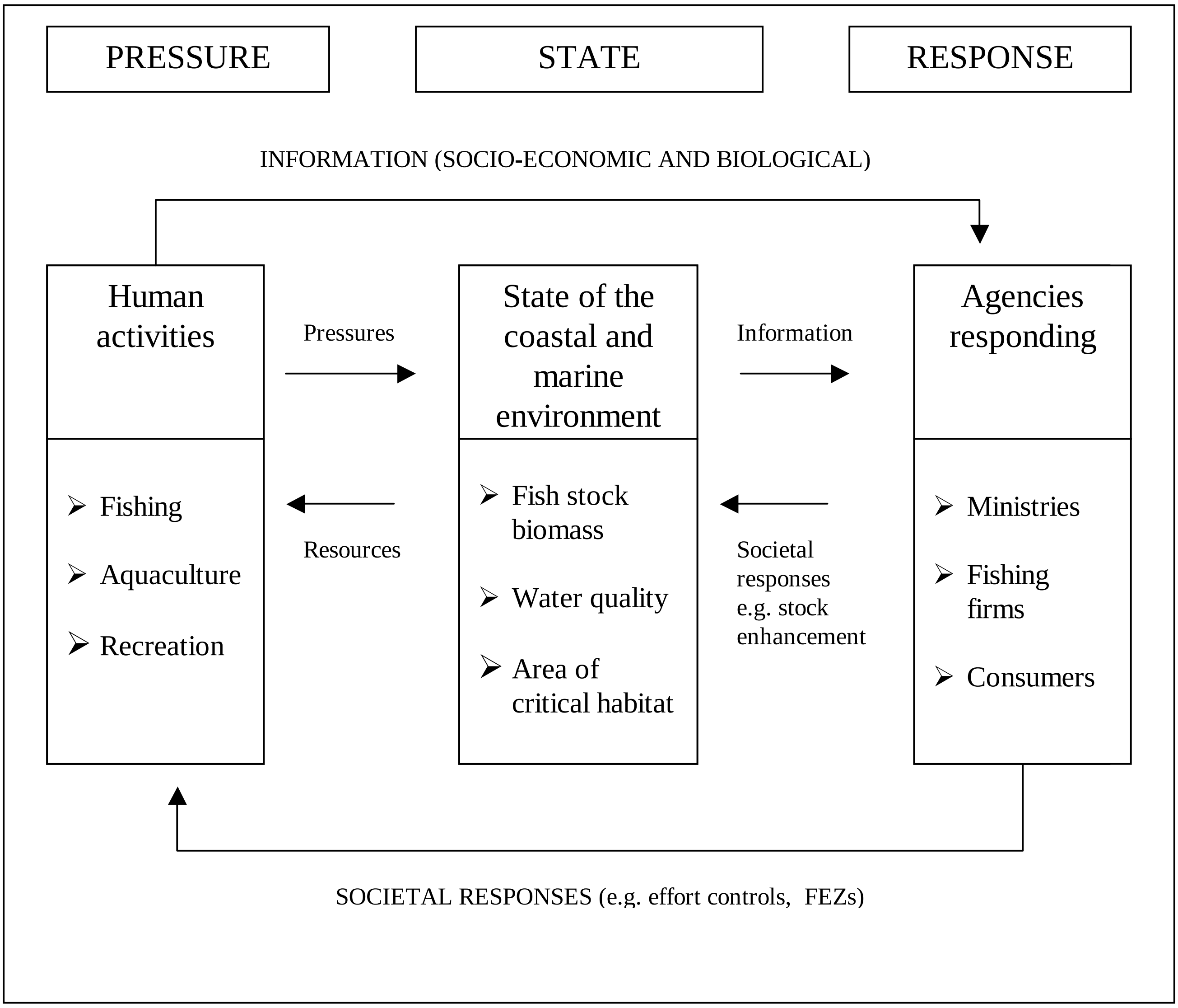
The Work Package will look at FEZs from a number of different perspectives, but its dominant concern is with the information – principally in the form of socio-economic and biological indicators - needed by fisheries managers in order to evaluate the effectiveness of FEZs. To contextualise the discussion we start by outlining a paradigm for understanding the linkages between human activities and the environment, showing how it can be applied to fisheries and marine resources. The Work Package then considers the substantive information requirements of fisheries managers, commencing with socio-economic assessment and moving on to a review of biological assessment and the progress which has been made in the development of mathematical models of FEZs. Bio-economic modelling, which is essentially a specialised type of socio-economic assessment in which explicit account is taken of the interaction between the biological and economic components of the fishing system, is dealt with in the final section.

2. The Pressure-State-Response paradigm

A framework which is commonly used to develop indicators of environmental performance is the pressure-state-response (PSR) paradigm (OECD, 1993), and this can usefully be applied in the present context to the functioning of the coastal and marine environment which supports fisheries. (Figure 1). In general terms the PSR model contends that anthropogenic (i.e. human) activities impose pressure on the environment, inducing a change in its state. Since society is not indifferent to these impacts there will be a response in the form of policies designed to amelioriate the pressure and to mitigate any environmental damage it may have caused. In the case of the coastal and marine environment the pressure might typically arise from a range of activities (e.g. capture fisheries, aquaculture, recreation), impacting on a number of 'state' variables (e.g. fish stock biomass, water quality, area of critical habitat), which in turn will induce a response from government agencies, firms and households. Where the problem is over-exploitation the response by governments may take the form of measures intended to directly offset stock depletion (e.g. stock enhancement or marine ranching) or through attempts to regulate the human activities that cause the over-exploitation (e.g. effort controls, quotas, area closures). For their part, commercial fishing firms may respond by redirecting their harvesting activities to sea areas where fish stock abundance is higher, while consumers may be prompted to buy eco-labelled seafood products which purport to come from 'sustainably managed fisheries'. Clearly, the range of responses by all of these agents together (government, firms, households) is potentially large.

For our purposes the important linkage in Figure 1 concerns information, since it is this which forms the basis of the indicators that the various agents act upon. The form of that information, the mechanism of its collection, its reliability and validity, will all impinge on the actions and decisions that are taken. The relevance of the PSR paradigm in the present context is twofold. Firstly, it shows that the information (= indicators) used by decision-makers should be comprehensive enough to enable the identification of cause-and-effect relationships (e.g. fishing effort → biomass → catch rates → profitability). Without some understanding of causality, it becomes almost impossible to anticipate or predict the likely success of fisheries management measures, whether these be exclusion zones or any other policy instrument. This point is crucial to mathematical modelling approaches to FEZs, which are explicitly build on assumptions regarding underlying cause-and-effect relationships. Secondly, the PSR paradigm implies that the information which flows back to decision-makers will itself be a function of the actions and decisions that are taken. This applies *a fortiori* to control measures imposed on fisheries, where the consequences of those measures induce (as intended) a change in state variables. If the magnitude of these changes can be accurately monitored and recorded, it should help provide fisheries managers with a better understanding of the behaviour of the fisheries system as a whole. This indeed is the rationale behind so-called 'adaptive' management of fisheries, which seeks to reduce uncertainty by actively seeking out information by probing the fisheries system through deliberate changes in management measures (Charles, 2001).

Figure 1: The Pressure-State-Response paradigm adapted to the coastal and marine environment



3. Socio-economic assessment of FEZs

3.1 The nature and purpose of socio-economic assessment

In light of the dictum that fisheries management is 'intended for the benefit of man, not fish' (Burkenroad, 1953) it is important to have a clear idea of the way FEZs can contribute to fisheries management and, more precisely, how we can assess the magnitude of any social and economic gains that may arise. Three generic types of socio-economic assessment can be identified, distinguished according to purpose (Table 1). These are: (i) profiling, (ii) impact analysis, and (iii) benefit assessment

3.1.1 Profiling

Profiling aims to provide basic empirical information on the socio-economic characteristics of a FEZ in respect of (a) the individuals and groups involved (e.g. fishermen, tourists), (b) the use they make of the marine resource (e.g. whether it is a consumptive activity, such as fishing, or a non-consumptive activity such as bird watching), (c) the spatial pattern to their activities (e.g. 'fishing the line' at the outer edge of the FEZ), and (d) the trend in resource use over time. The methods used in this type of assessment range from simple enumeration (e.g. a census of fishermen in a given year), through to more sophisticated multi-variate statistics involving data-reduction methods such as cluster analysis, principal components analysis, factor analysis or multi-dimensional scaling. Where data are sufficiently detailed and cover two or more time periods, it may be possible to construct transition matrices which could be used, for example, to derive the probabilities of vessels moving between different zones or ports. Likewise, given adequate data of the right periodicity, time series analysis may be used to identify empirical regularities (e.g. seasonality) in the pattern of resource use. These latter methods may not only be applied retrospectively but may also be used to make short-run forecasts of future developments in the use of marine resources affected by an exclusion zone. A recent account of MPAs in the Mediterranean by Badalamenti et al (2000) provides an illustration of the profiling approach from a largely descriptive standpoint, while the paper by Alder et al. (2002) demonstrates the use of multi-dimensional scaling in characterising MPAs in terms of particular attributes.

3.1.2 Impact analysis

In general terms the purpose of impact analysis is to trace out the ramifications of a particular event or action for variables which are considered to be particularly important (Field, 1994). In this context, therefore, our concern would be in measuring the effects of establishing a FEZ in terms of variables such as economic activity (i.e output, employment, incomes), markets and prices, the financial performance of affected firms, and the attitudes of individuals and groups who might perceive themselves to be interested stakeholders. What constitutes an 'important' variable will necessarily be a value judgement, and these will differ; local politicians might regard the employment consequences of an FEZ as being of far greater relevance than, say, its effect on fish catches and price. Indeed, given that the economic impacts of FEZs are in principle quite diverse, the range of applicable techniques for examining them is potentially large. Those listed in the Table are: input-output analysis, which can be used to trace through the direct and indirect effects of an FEZ on economic activity and hence to derive multiplier effects; demand analysis, which is relevant in identifying the market impact of changes on fish landings which may result from the imposition of an FEZ; financial analysis, which would appropriate where an FEZ impacts on catch rates and profits of fishing firms; and attitude surveys, where the concern is with assessing the way in which the establishment of a FEZ is perceived by fishermen, recreationists, conservation groups and others. All these illustrate a fundamental point, which is that impact analyses necessarily involve the testing of particular hypotheses about the effects of a FEZ. This is what mainly distinguishes them from straightforward profiling, which is essentially a descriptive characterisation of FEZs not involving hypothesis testing. Impact studies of exclusion zones in Europe include those by Whitmarsh et al. (2002), which analyses the financial performance of artisanal vessels inside a trawl ban area in N.W. Sicily, looking specifically at the extent to which profitability has been effected by increases in demersal stock abundance. In the North American context two noteworthy studies of the Florida Keys National Marine Sanctuary (FKNMS) include those by Leeworthy and Wiley (2000) which calculates the losses to displaced consumptive users (commercial and recreational) following the creation of the Tortugas marine reserve, and by Suman et al (2000) which reports the results of a survey of stakeholder attitudes and how they have been affected by the zoning plan established within the FKNMS.

3.1.3 Benefit assessment

Benefit assessment attempts to measure the net economic value of a FEZ in terms of the flow of benefit which it provides to users (e.g. fishermen, recreationists) and non-users. The distinguishing feature of benefit assessment in its traditional form is that it sets up a normative objective (economic efficiency) by which resource allocation decisions may be evaluated, the purpose being to decide whether a particular course of action is likely to be beneficial or detrimental to society as a whole. In this context the decision might be, for example, whether or not to establish a FEZ or whether a FEZ is the best (i.e. most economically efficient) management option. The standard ways in which this type of economic assessment may be carried out in practice are: firstly, via cost-benefit analysis (CBA), which seeks to establish the relationship between the monetary benefits and costs of a project; and secondly, via cost-effectiveness analysis (CEA), which tries to determine the least-cost way of achieving a given objective given that there may be several options available. Where the benefits of a project do not have a market price attached to them – for example, the bequest or existence value of marine resources associated with the maintenance of biodiversity – it may still be possible to monetise them for inclusion within the arithmetic of CBA. Where this is not feasible then it will be necessary, at the very least, to identify the range of effects engendered by the project and the conflicts or trade-offs between them. The creation of a FEZ may be associated, for example, with increased tourism and higher incomes for the local economy but also degradation of the marine environment due to pressure of visitor numbers. ‘Partial’ benefit assessments of this kind, involving the monetising of some but not all the effects of an exclusion zone, include those by Dixon, Fallon Scura and van't Hof (1993) and Brown et al (2001). In recent years attempts have been made to extend CBA by examining the incidence of costs and benefits for particular socio-economic groups or stakeholders. In so doing the definition of ‘social welfare’ – and hence optimality – takes on a much more fluid meaning, since it thus depends not only on the overall balance of benefits and costs to society as a whole but also on who the winners and losers are. Multi-criteria methods represent one approach to this, an example of their application to MPAs being the study of the Buccoo Reef Marine Park (Tobago, West Indies) by Brown, Tompkins and Adger, 2001 and Brown et al, 2001.

3.2 Problems of monetary valuation

We now look more closely at the problems of ascribing monetary values to the various inputs and outputs that may be affected by a FEZ. Table 2 presents the familiar typology of economic values which can be attributed to the natural environment (Turner and Jones, 1991; Munasinghe,

1993) and which would need to be accounted for in any benefit assessment of FEZs. In practice the importance of the various elements will vary from case to case, if only because FEZs differ in the objectives they are intended to achieve. For example, a FEZ may be purposely established to protect seagrass beds which support the productivity of local fisheries by serving as an important spawning and nursery area for juvenile fish. If the area in question possesses no features or attributes which make it unique, then the significance of the FEZ is likely to be greater for *indirect use value* (commonly associated with ecological support functions) rather than *passive use value* (commonly associated with the 'uniqueness' of an environmental asset). One can, of course, easily imagine situations where the reverse is true and where protection is given to an area principally because of its special and irreplaceable characteristics – the Great Barrier Reef being a case in point (Farrow, 1996).

Below we consider three areas of potential difficulty associated with monetary valuation of FEZs:

(i) What is being valued ?

It needs to be emphasised that the relevant focus here is not on the economic value of the marine resource as such but on the *change in value that the creation of a FEZ would occasion*. This is essentially the point made by Pendleton (1995), who criticises certain economic appraisals of tropical marine parks (notably those by Post (1992) and Dixon, Fallon Scura and van't Hof (1993) of the Bonaire Marine Park in the Caribbean) on the grounds that they value the resource protected rather than the protection provided. The benefit assessment undertaken in these studies, it is claimed, is methodologically flawed. Pendleton argues that the value of protection given by a park or reserve requires a 'with' and 'without' comparison of the flow of economic benefit over time. The idea is illustrated in Figure 2, which shows two generic scenarios: one where an exclusion zone maintains the environmental quality of a marine area, in contrast to the other where the environment becomes degraded and economic benefits decline. The value of protection is the vertical difference between the 'with protection' and 'without protection' lines, discounted appropriately to allow for the time when these net benefits are received. (see below).

Though Pendleton's concern is mainly with coral reef degradation, the issue raised is fundamental to any economic assessment of protected areas: what is required is a knowledge of how the flow of economic benefits would have evolved had the protection not been introduced. This presents a particular challenge in the case of fisheries which are subject to worsening over-exploitation and where FEZs are being considered as a management option, because the 'without protection' baseline scenario is quite likely to be one of progressive deterioration in the stream of economic benefit rather than simply one of no change. Measuring this stream requires an explicit forecast of catches and net revenues based on anticipated trends in fishing effort and stock abundance. It is in circumstances such as these that bio-economic modelling comes into it own, because it offers an analytically rigorous way of establishing a baseline in an exploited fishery against which to compare the effects of different management options, including FEZs.

(ii) Absence of market prices

A basic difficulty in attempting to impute monetary values to the benefits identified in Table 2 is that the environmental assets or the uses they provide typically do not command a market price. This applies to situations where, for example, a FEZ creates recreational opportunities (e.g. yachting, scuba diving) or conservation values (e.g. maintenance of biodiversity) for which individuals would be willing to pay but which in the event they receive free of charge. Even in cases where it is clearly possible to attach a market price to outputs – as in the case of commercially-traded fishery products – the marine resource from which production derives may itself be unpriced by virtue of its public-good characteristics. This is conspicuously true of several types of coastal habitat such as coral reefs, mangroves and seagrasses, all of which are indirectly linked with fisheries production (Spurgeon, 1998).

The absence of market prices means that some other way of imputing monetary values has to be found, which in practice generally involves establishing people's preferences (reflected in their willingness to pay, WTP) for specified benefits derived from marine environmental assets. An extensive literature on environmental valuation now exists, and here we will single out those studies that best illustrate the valuation problem in the context of FEZs. Two such studies are noteworthy, both of which make use of the contingent valuation method (CVM) for eliciting respondent's preferences for unpriced benefits associated with environmental quality. The first example is the previously-cited case of the Bonaire Marine Park (Dixon, Fallon Scura and van't Hof, 1993), where part of the economic appraisal included a contingent valuation survey of visitors to assess their WTP access fees for diving. The results implied that a substantial consumer surplus was received by dive tourists, since the average WTP ($27.40/diver/yr) was well in excess of the amount that they were actually required to pay ($10.00/diver/yr). The second example relates to the Buccoo Reef Marine Park in Tobago, in which the investigators undertook a contingent valuation survey of visitors and residents in order to gauge their WTP to prevent further deterioration in the quality of the reef under a range of different development scenarios. (Brown, Tompkins and Adger, 2001; Brown et al, 2001). The results provided a monetised measure of visitor enjoyment, and revealed *inter alia* that tourist benefits (= total WTP) were substantially greater when respondents were presented with development scenarios in which environmental management of the marine park was improved. The CVM findings also exposed the trade-offs involved with a development strategy of expanded tourism - on the one hand, there would be macro-economic benefits and local employment; on the other, reduced environmental quality and consequently lower total visitor enjoyment.

(iii) The time profile of benefits

A final difficulty relates to the fact that the flow of monetary benefits arising from the introduction of a FEZ is unlikely to be constant over time, at least in the case of fisheries. As Bohnsack (1994) has made clear, the permanent closure of an area to fishing will result in an immediate loss in harvest, and while this may be compensated by increased total production from the adjacent open area the effect will not be immediate. The worse the over-exploitation of the fishery prior to the establishment of the FEZ, and hence the smaller the available spawning stock, the longer the delay before the full benefits are achieved (p. 229). In terms of economic analysis this non-constancy of returns is not in itself a problem, and in theory the trade-off between short-run losses and long-run gains can be evaluated within the framework of conventional cost-benefit analysis by discounting the stream of net benefits over time in order to arrive at a net present value (NPV). Depending on whether the NPV is positive or negative, the FEZ is deemed to be economically worthwhile or not.

In practice, however, such an evaluation will present several problems. To start with, a fairly robust knowledge of the biology of the exploited fish stocks will be required in order to assess its recovery prospects and hence the time profile of catches. This problem magnifies up according to the complexity of the marine ecosystem and the number of species affected. Secondly, the choice of discount rate becomes critical. Discounting effectively penalises economic returns according to when they are expected to accrue, with higher discount rates implying a more severe penalty for any delay in the recovery of a fishery. While in principle it may be possible to determine the socially optimal discount rate for use in project appraisal, in practice it is far from straightforward and remains a controversial issue when inter-generational effects are involved – as indeed they will be in the case of fisheries whose recovery may take several years. However, while the actual value for the discount rate may be in contention, most economists would argue that the rationale for discounting is valid. If this principle is accepted, and a positive discount rate is applied to the stream of benefits, it no longer follows that a FEZ which results in increased catches in the long-run will inevitably compensate for any short-run losses. The argument that a FEZ which succeeds in raising economic returns in a fishery 'must eventually pay for itself' would be correct only in the limiting and unrealistic case of a zero discount rate. (see Annex 1). The choice of discount rate is again significant where an FEZ is only one of several fisheries management strategies for rehabilitating a depleted fishery because the time profile of economic returns is unlikely to be the same for each. The higher the discount rate, the greater the preference in favour of rehabilitation strategies that yield economic improvements to a fishery sooner rather than later. This will militate against choosing FEZs, *ceteris paribus*, if other rehabilitation options (e.g. across-the-board quota reductions) start to work quicker. Indeed, for fisheries that are already well managed via conventional measures (e.g. catch and effort controls), a switch to area closures may produce no significant economic improvements once the changing time profile of benefits has been correctly accounted for (Milon, 2000; NRC, 2001).

Table 1: Socio-economic assessment of exclusion zones

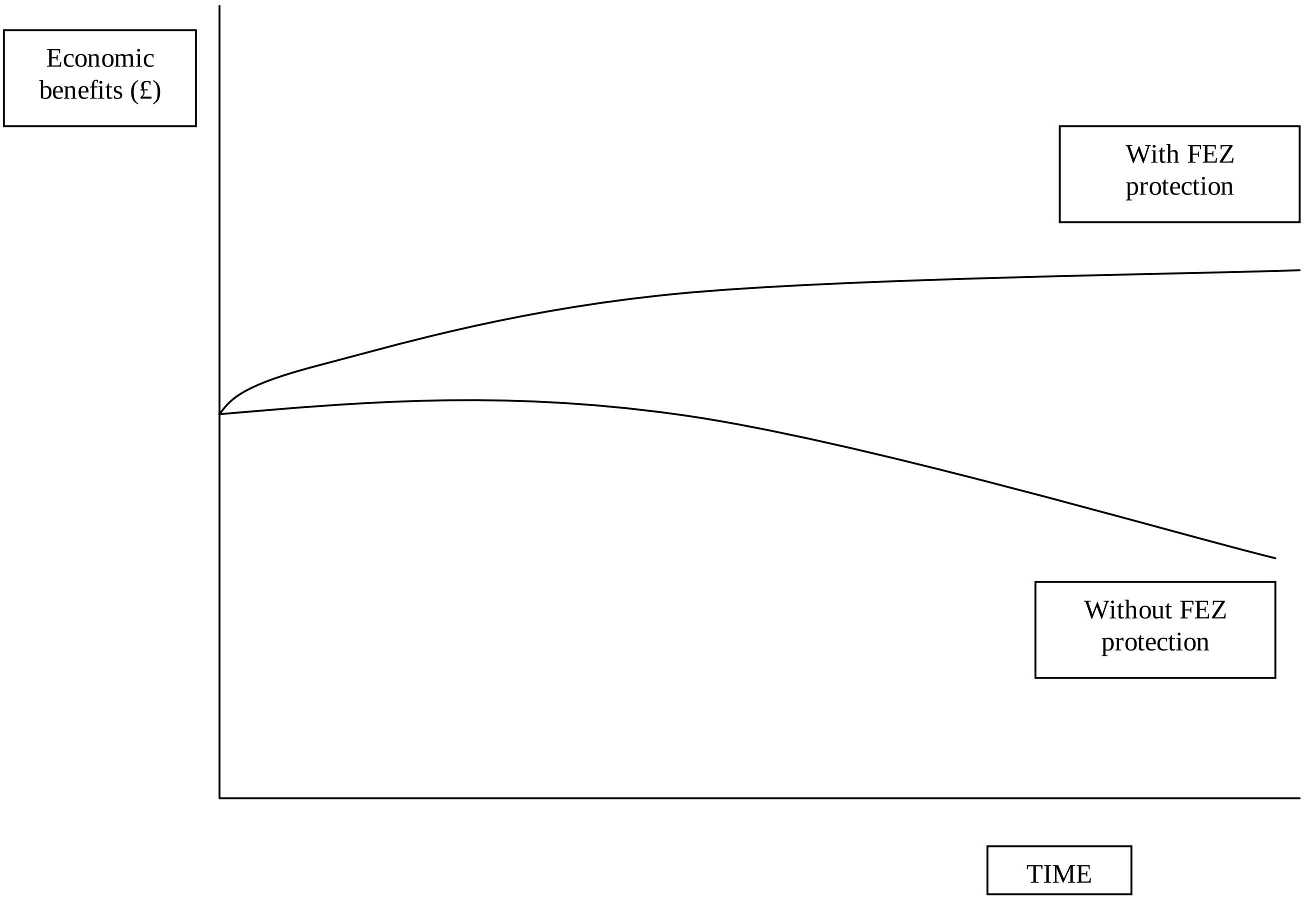
Type of assessment	Purposes	Applicable methods
Profiling	To identify trends and patterns (e.g. spatial clustering) in the use of marine resources affected by a FEZ based on a range of empirical indicators, and to anticipate their future development	Enumeration Data-reduction techniques Transition matrices Time series analysis
Impact analysis	To estimate the actual or potential impact of a FEZ on a given set of economic or social variables, typically economic activity (i.e. output, employment and income), markets and prices, financial performance and community attitudes	Input-output analysis Demand analysis Financial analysis Attitude surveys
Benefit assessment	To determine the net economic value to society of a proposed FEZ in relation to alternative management options, and to identify optimal FEZ configuration (i.e. size, location, etc.)	Cost-effectiveness analysis Cost-benefit analysis Trade-off analysis Multi-criteria methods

Note: Bio-economic models of commercial fisheries have a role in impact analysis and benefit assessment. Specifically, (a) a bio-economic model may be used as an engine for generating hypotheses concerning the effects of FEZs which can then be tested against field observations, (b) empirically estimated bio-economic models may be used to simulate the behaviour of a fishery under a variety of ‘what-if ?’ scenarios (such as changes in the size of a FEZ), or else to identify the circumstances under which the performance of a fishery system would be optimised.

Table 2: Economic values affected by exclusion zones

Total economic value			
Direct use value	**Indirect use value**	**Option value**	**Passive use value**
Outputs from a marine resource in the form of commodities or services that can be consumed directly	Functional benefits that a marine resource provides to support other economic activities	Benefits from possible use of a marine resource at a later date	Benefits from a marine resource from knowledge of its continued existence or availability to future generations
Example: Extractive uses (e.g. commercial fishing) and non-extractive uses (e.g. marine wildlife observation)	Example: Biological support for fish production provided by seagrass, mangrove or coral	Example: 'Insurance value' of maintaining opportunities for fishing or recreation in subsequent years	Example: Preservation of unique habitats or maintenance of biodiversity

Figure 2: Flows of economic benefit over time with and without FEZ protection



4. Biological assessment of FEZs

4.1 Design of empirical studies

The literature on fishery exclusion zones (FEZ) comprises many reviews and theoretical studies, and comparatively few empirical studies of the biological and fishery effects of spatially explicit protection from fishing (Edgar & Barrett, 1999). Most empirical studies of the biological effects of closed areas have focussed on effects within the exclusion zone itself, but a few have addressed impacts on the population in the wider area and on the surrounding fishery.

4.1.1 Within-FEZ effects

Probably the most common approaches have been comparison of population characteristics between open and closed areas at one time, or monitoring of population dynamics within a FEZ over time (Dugan & Davis, 1993). Population density (in numbers or biomass), size distribution and sometimes species diversity have been assessed. Less frequently, derived variables, such as mortality rates and growth rates have been estimated (Ulmestrand, 1996; Pastoors et al, 2000; Sánchez Lizaso et al., 2000). A few studies have examined the range and frequency of movement of fished species in relation to reserve boundaries (e.g. Lockwood, 1988; Goñi et al., 2000; Sánchez Lizaso et al., 2000).

Even when monitoring studies include ‘baseline' data from before the establishment of the exclusion zone (before/after comparison), if there are no 'control' or 'reference' areas, changes in biological variables within the FEZ cannot be attributed unequivocally to protected status, since equivalent changes may have taken place outside the protected area. Similarly, comparisons solely in the spatial domain, contrasting protected and unprotected zones without baseline data (open/closed comparison), can not exclude the possiblity that any differences detected between the zones existed prior to FEZ introduction. A few studies have attempted a before-after, control-impact (BACI) study design, in which replicated samples are taken at two times (before and after implementation of the reserve), in two places (within the reserve and at a ‘control' site) (Edgar & Barrett, 1999; García Charton et al., 2000; Lindegarth et al., 2000). Ideally, several control sites should be examined to minimise the chance that any before/after change in the difference between reserve and unprotected zones is due to spatial differences in temporal variation due to localised factors other than the permission or prohibition of fishing (Rowley, 1994).

Extensions of BACI designs, in which there is repeated sampling over time at different temporal and spatial scales, allow detection of different types of response in time (e.g. pulse response, oscillations, sustained change) and space (e.g. distributional changes) (Underwood, 1991, 1997). The data requirements for a rigorous empirical assessment of actual biological effects within a reserve are therefore considerable, involving repeated sampling in time before and after reserve implementation, in one or more FEZs and in several equivalent, but unprotected areas. As far as we are aware, this type of study has not been done for any marine protected area in the northeast Atlantic (or possibly elsewhere).

4.1.2 Effects beyond the FEZ

In fisheries management, fishery exclusion zones are used in the hope of benefiting a fishery outside the boundaries of the protected area, for example, by reducing overall fishing mortality, by protecting nursery areas to reduce juvenile mortality, or by protecting a portion of the adult population so that they may export larvae or adult biomass to the fishery outside. In the context of fishery management, therefore, it is change in the population dynamics of the whole stock (or stocks) that is of interest, not just that present within the protected area. Such changes may be assessed by standard fishery sampling techniques (e.g. market sampling, catch sampling, trawl surveys, acoustic surveys, larval surveys), methods of analysis (e.g. trend analysis, virtual population analysis, time series methods) and population modelling. However, isolating the effects of closed areas in large fisheries empirically is likely to be problematic. Owing to the reduction in fishing opportunities they cause, closed areas are usually not implemented until the fishery is in crisis. In such circumstances, a number of other recovery measures which affect fishing mortality are likely to be instigated in addition to closed areas, confounding before/after comparisons. Economic processes affecting fishers' behaviour and patterns of fishing effort are also likely to be altered. In addition, it is possible that adequate 'control' areas without closures would not be available for comparison, so that any effects of the reserve would be confounded with more widespread changes. Fishery managers would presumably be unwilling to jeopardise fishermen's livelihoods, or the security of the stock, by conducting controlled experiments in which remedial measures were implemented in some areas and not others, although the importance of experimentation in fishery management has previously been highlighted (Larkin, 1978; Walters & Holling, 1990: both cited by Dugan & Davis, 1993).

With respect to migratory species, even if the fishery is not in crisis, the scale of the geographic range of the stock may be such that it is not possible to identify other equivalent stocks to act as controls without closed areas for comparison. The larger the range of stocks of a given species, the more likely that adjacent stocks will be subject to significantly different conditions. In a large-scale fishery, the best evidence that a particular closed area has had the intended effect on a stock may therefore be a close agreement between the nature and timing of observed changes in population parameters and those predicted from population and economic theory to result solely from closures. This constitutes a different approach from the classical statistical approach underpinning BACI (falsification of a null hypothesis). Ideally, there should be a number of 'competing' models of FEZ effects, whose fit to the data could be compared, and further management decisions considered on the basis of the most likely model (Hilborn & Mangel, 1997). Bayesian methods of taking account of prior information about the system are becoming increasingly popular in this type of approach (e.g. Maunder et al., 2000). Nevertheless, without control areas, evidence of this type will be equivocal in particular cases. More generally, consistent indications from several studies of changes in fishery performance in a particular direction after FEZ introduction could be construed as aggregate evidence on which to base firmer conclusions about the effects of FEZs, even though all of the studies may have been deficient in some aspect of experimental design (Rowley, 1994). However, such evidence has been slow to accumulate (Dugan and Davies, 1993; Rowley, 1994) and does not consistently indicate positive effects (e.g. Latrouite, 1995; Pastoors et al., 2000; Sánchez Lizaso et al., 2000).

4.3 Mathematical models of the biological effects of FEZs

There are only a few field studies addressing the impact of FEZs on adjacent fisheries. This lack of information puts the scientist and/or the manager in front of subjective choices when asked to design a new FEZ. In turn this increases the uncertainty on the outcome of the reserve and reasons of its success or failure. Although some scientists claim for a practical approach to the problem of FEZs (which means building FEZs and studying their effects), theoretic studies using mathematical models and/or computer simulations can shortcut long and expensive field studies by providing a general framework to establish FEZs. While not replacing at all experimental data, this knowledge can be used to optimise the design of a FEZ according to the specificity of the target fish stocks and the related fisheries. Moreover it can be used to plan the monitoring of the impact of the reserve in order to verify if this protection tool is efficient in reaching its goals. Six bibliographic references were identified during the literature review made by IRMA, which seemed to potentially fit the year-round FEZs identified in the Mediterranean (DeMartini, 1993; Guenette and Pitcher, 1999; Lindholm et al., 2001; Mangel, 2000; Polacheck, 1990; Sladek Nowlis and Roberts, 1999). All of them propose techniques for modelling FEZs effects based on the use of simulation models requiring computer intensive calculus. These are based on a mathematical description of the main components of a fishery system (*i.e.*, fish stock(s) and fishing fleet) and their interactions. The magnitude of these values are evaluated at each step and reprocessed as input values during the following step. This computer intensive method allows monitoring the behaviour of the variables along the iterative calculus. Moreover it permits to take into account variable processes such as reproduction or catches by including stochastic terms into the mathematical description of the different components of the model. Each simulation focuses on a particular characteristic of the fishery supposed to be affected by the establishment of a FEZ. Among them, reproductive enhancement of the fish stock and re-population of adjacent fishing grounds were the most frequently considered. In this review, we will not consider the validity of the different modelling approaches used in each article. Instead we will expose the assumptions made to model the components of the fishery and the results that these theoretic studies provide.

4.2.1 Hypotheses of the models

Biological hypotheses

All models reviewed here deal only with one species at a time (=single species stocks). Some authors modelled different ecological categories of fishes (DeMartini, 1993; Sladek Nowlis and Roberts, 1999) in order to foresee the reactions of different species to different reserve designs, but none of them modelled multi-species interactions. Usually the considered stock was subdivided in year classes, but Sladek Nowlis and Roberts (1999) used a size-structured population model arguing that most of the processes under study were size-dependent.
These studies were normally motivated by a particular fishery: Atlantic cod (Guenette and Pitcher, 1999; Lindholm et al., 2001; Polacheck, 1990), rockfishes (*Sebastes* spp.) (Mangel, 2000) and some tropical reef fishes (Sladek Nowlis and Roberts, 1999). As a consequence, life history parameters came from field studies carried out on these natural populations.

<u>Individual growth</u>
The modelling of growth was achieved by three different ways. The first was to use the mean size by year class, based on field data (Guenette and Pitcher, 1999; Polacheck, 1990). The

second was to consider that individual growth follows the Von Bertalanffy growth function (VBGF) (DeMartini, 1993). Finally, Mangel (2000) assumed equal weight for adults and juveniles, arguing that this assumption had no qualitative effect on the results.

Larval features

In cases where larval stages were considered, their features (duration of the larval stage and larval mortality) were taken from the literature reporting experimental data where available. In the lack of non-availability of experimental data, Sladek Nowlis and Roberts (1999) run the model with different guessed values until it had a realistic behaviour. These authors modelled the larval survivorship as a stochastic process normally distributed around a mean value.

Mortality

In all cases but one, natural mortality (M) was represented by a constant and survival was modelled by an exponentially decreasing function of time. In those cases, magnitude of the mortality was taken from the literature on the considered stock. The exception was Lindholm et al. (2001), who assumed density-dependent mortality and modelled it as a function of prey density using three deferent equations. The first one is a simple linear relationship (Eq. 1) and the second a curvilinear (Eq. 2).

$$M = \alpha\, x + z \qquad \textbf{Eq. 1}$$

$$M = \frac{\alpha\, x}{1 + \alpha\, x} z \qquad \textbf{Eq. 2}$$

Where α is a habitat-specific constant,
x a measure of the density of fishes,
z a scaling factor.

The third relationship (Eq. 3) is an S-shaped function with two additional habitat-specific constants.

$$M = \frac{\alpha\, x^2}{1 + c\, x + \beta\, x^2} z \qquad \textbf{Eq. 3}$$

Where c and β are habitat-specific constant.

Sexual maturity and fecundity

In general, the schedule of sexual maturity was based on experimental data that gave the percentage of mature individuals as a function of age (DeMartini, 1993; Guenette and Pitcher, 1999; Polacheck, 1990). By contrast, Sladek Nowlis and Roberts (1999) used a specific threshold length to determine whereas the females of a given size class were involved in reproduction. Mangel (2000) assumed that reproduction occurred for 10% of adults. Fecundity, expressed as number of eggs, was assumed to be an allometric function of female length (Guenette and Pitcher, 1999; Sladek Nowlis and Roberts, 1999). It was assumed by Mangel (2000) that breeders have a fixed number of offspring.

Migration rates between protected and unprotected areas

The migration rates (expressed in (no. of fishes)$^{-1}$) between protected and unprotected areas received much attention in each paper, because the final outcome of a simulated FEZ depends heavily on it.

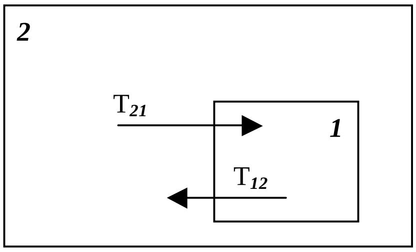


Figure 2: Schematic representation of a FEZ. Areas 1 & 2 represent respectively the reserve area and the fishing ground. The migration rates (T) are symbolized by the two arrows.

Polacheck (1990) supposed that during the simulation a fixed fraction of the fishes in each area moved to the other one. The migration rate is varied between different runs of the model, in order to investigate the effect of this variable on the reserve outcome. In order to analyse the effect of refuge size on a population with intrinsic migration rate, Polacheck (1990) had to link the change in transfer rate (*i.e.*, migration rate) with the refuge size. This author used the equation (Eq. 4) to predict the changes in transfer rate from changes in the refuge size, arguing that the transfer rate out of the reserve (T_{12}) is expected to decrease as the reserve proportion increases:

$$T_{12} = nT_{1s}\left(\frac{R_s}{R_1}\right)^{\frac{1}{2}} \qquad \textbf{Eq. 4}$$

where :

n is the number of edges of the closed area which are not population boundaries,

T_{1s} is the transfer rate for a refuge of a standard size (chosen to be 10% of total area),

R_s is the proportion of the total area contained within the standard size refuge,

R_1 is the proportion of the total area contained within a reserve.

This empirical relation was further re-used in its original formulation by DeMartini (1993) during one set of simulations. (Guenette and Pitcher, 1999) used a transformed version of Polacheck's formula (Eq. 5) that did not take into account the number of edges of the closed area.

$$T_{12} = T_{1s}\left(\frac{R_s}{R_1}\right)^{\frac{1}{2}} \qquad \textbf{Eq. 5}$$

DeMartini (1993) run a second set of simulations involving a density-dependent migration rate. It was varied from time to time according to the density of the sub-population in each area and is expressed as follows:

$$T_{12,t+1} = T_{12,t}[(N_{1,t} / N_{1,0})/(N_{2,t} / N_{2,0})]^x \qquad \textbf{Eq. 6}$$

$$T_{21,t+1} = T_{21,t}[(N_{2,t} / N_{2,0})/(N_{1,t} / N_{1,0})]^x \qquad \textbf{Eq. 7}$$

where :

$N_{i,t}$ is the fish density in area *i* at time *t*,

$N_{i,0}$ is the initial density of fish in area *i*,

x is the power used to scale the ratio of fish densities.

Two values of x were evaluated 0.125 (eighth root) and 0.5 (square root).

In order to investigate the potential re-population effect of a FEZ through larval dispersion, the model by Sladek Nowlis and Roberts (1999) assumed that fishes could cross the reserve borders only in their larval phase. This larval dispersion results in an even density of new fish settlement in both reserve and non-reserve areas.
Mangel (2000) based his simulation on rockfishes (*Sebastes* spp.), which exhibit a territorial behaviour linked to reproduction. "Floaters", *i.e.*, juveniles and adults that have no territories, move around in search of a place to settle. Mangel (2000) described this movement pattern by considering that the FEZ always contained a proportion of the population of floaters equal to the percentage of protected ground. This accounted for a net migration rate between the protected and the unprotected area.

Fishery hypotheses

The simulations were made to model a year-round fishing closure, which implied that one of the areas was totally free from fishing effect. Fishing mortality (*F*) was directly incorporated in the demographic equations (DeMartini, 1993; Mangel, 2000; Polacheck, 1990), or it was expressed as the proportion of fishes caught per year (*u*) by the following equation:

$$u = 1 - e^{-F} \qquad \textbf{Eq. 8}$$

(Guenette and Pitcher, 1999; Sladek Nowlis and Roberts, 1999) Normally fishing mortality was varied between simulations, in order to investigate the influence of fishing pressure on the outcome of a reserve. The age-specific recruitment of fishes was taken from field data (DeMartini, 1993; Polacheck, 1990). The number of recruits was a function of the number of eggs, following either Beverton or Ricker's equation (Guenette and Pitcher, 1999). In Sladek Nowlis and Roberts (1999), the length at recruitment was determined by a fix length threshold.A common assumption was the redistribution of fishing effort in the reduced fishing ground (DeMartini, 1993; Guenette and Pitcher, 1999; Polacheck, 1990), which accounts for the increase in fishing mortality in the non-reserve area after the establishment of a FEZ. The assumption was that the total effort did not change and the displaced effort was uniformly redistributed. The increase in fishing mortality resulting from a increase in fishing pressure was modelled by the following equation (DeMartini, 1993; Polacheck, 1990):

$$F_c = F_2 \frac{N_{1,0}}{N_{1,0} + N_{2,0}} \qquad \textbf{Eq. 9}$$

where

F_c is the fishing mortality rate before the establishment of the FEZ,

F_2 is the fishing mortality rate in area 2 following the establishment of the FEZ,

$N_{i,0}$ is the number of fishes in area *i* at initial time *t*=0.

Guenette and Pitcher (1999) assumed that the fishing mortality varied as a function of the reserve size (R_1) according to the following equation:

$$F_2 = \frac{F_c}{1 - R_1} \qquad \textbf{Eq. 10}$$

Instead of considering the redistribution of fishing effort, Sladek Nowlis and Roberts (1999) varied the fishing mortality independently from the FEZ size. Mangel (2000) used the same approach but added a stochastic component at the fishing effort to account for the variability of catch by the fishing fleet. Lindholm et al. (2001) simulated the effect of fishing through its impact on the complexity of the seafloor which influences natural mortality.

4.2.2 Modelling the demographic variation in fish populations

The model of Beverton and Holt (Beverton and Holt, 1957), which accounts for the demographic variation of fish populations assuming spatial variation in fishing mortality, was the most widely used model (DeMartini, 1993; Lindholm et al., 2001; Polacheck, 1990). Polacheck (1990) described the simplest case of a fishery exclusion zone: the fishing ground is divided in two areas, one of which is closed to fishing. This model took the form of a system of two differential equations, which described the demographic variation of the population in each area as a function of biological parameters (natural mortality, migration rate) and exploitation parameters (fishing mortality):

$$\frac{dN_{1,t}}{dt} = -(M + r_t F_1 + T_{12}) N_{1,t} + T_{21} N_{2,t} \qquad \textbf{Eq. 11}$$

$$\frac{dN_{2,t}}{dt} = -(M + r_t F_2 + T_{21}) N_{2,t} + T_{12} N_{1,t} \qquad \textbf{Eq. 12}$$

where

$N_{i,t}$ is the size of the cohort in area *i* at age *t*,

T_{ij} is the transfer rate from area *i* to *j*,

F_i is the fully recruited fishing mortality rate in area *i*,

r_t is the proportion of individuals of age *t* recruited into the fishery,

M is the natural mortality rate.

(To account for the fishing prohibition in the FEZ, the parameter of fishing mortality (F_1) is set to zero in Eq. 11)

Besides this specific implementation of the model of Beverton and Holt, the other authors built their own model according to their specific hypothesis. The closest is the age-structured model used by Guenette and Pitcher (1999), which take into account the migration rate in and out for both area. It is describe by the following equations:

$$N_{1,t,a} = (N_{1,t-1,a-1} + N_{2,t-1,a-1} \times T_{21} - N_{1,t-1,a-1} \times T_{12}) \times S_{a-1} \quad \textbf{Eq. 13}$$

$$N_{2,t,a} = (N_{2,t-1,a-1} + N_{1,t-1,a-1} \times T_{12} - N_{2,t-1,a-1} \times T_{21}) \times S_{a-1} \times (1 - v_{a-1} \times u) \quad \textbf{Eq. 14}$$

where

$N_{i,t,a}$ is the number of fishes in area i (closed (i=1) and fished (i=2)), at time t, of age a,

S_i is the annual survival rate at age i,

v_i is the vulnerability to the fishery at age i,

u is the annual exploitation rate.

Sladek Nowlis and Roberts (1999) assumed that adult fishes experienced density-dependent survivorship at settlement. Thus instead of surviving at a rate v_i like individuals of other size classes, their survival was weighted by a density-dependent function of the form $e^{-\rho/K}$ where ρ is the population density and K is the carrying capacity, which is arbitrarily (due to the lack of experimental information on carrying capacity for the species under study) set to 1000. The model for adult survival in the reserve was:

$$N_{i,t} = v_{i-1}\ p_{i-1}\ N_{i-1,t-1}\ e^{-N_{i-1,t-1}/K} + v_i(1 - p_{i-1})N_{i,t-1} \quad \textbf{Eq. 15}$$

where

$N_{i,t}$ is the population density of the individuals of size class *i* (only for adults) at time *t*,

v_i is the survival rate of the size class i,

p_i is the probability to grow from one size class to another.

In the paper by Sladek, Nowlis and Roberts (1999), the equation describing the demographic variation in the fished area was identical, but the density of the recruited size class was reduced by a specific proportion in a further stage of the simulation.

Mangel (2000) built a model describing the population dynamics of *Sebastes* spp., whose reproduction behaviour is territorial. As a consequence, the equation accounting for the demographic variation depends on the number of breeding site. The variation of the number of adults in both areas is described by the following two equations:

$$N_a(1,t+1) = B(1,t)\, e^{-M_a(t)} + A\,[W_a(1,t)\, e^{-M_a(t)} + W_a(2,t)\, e^{-M_a(t)-F_a(t)}] + A\, f_j [N_j(1,t)\, e^{-M_j(t)} + N_j(2,t)\, e^{-M_f(t)-F_j(t)}] \quad \textbf{Eq. 16}$$

$$N_a(2,t+1) = B(2,t)\, e^{-M_a(t)-F_a(t)} + (1-A)[W_a(1,t)\, e^{-M_a(t)} + W_a(2,t)\, e^{-M_a(t)-F_a(t)}] + (1-A)\, f_j [N_j(1,t)\, e^{-M_j(t)} + N_j(2,t) \quad e^{-M_j(t)-F_j(t)}] \quad \textbf{Eq. 17}$$

where

$N_a(i,t)$ is the number of adults in area *i* in year *t*,

A is the proportion of individuals that ends up in the reserve,

$B(i,t)$ is the number of breeders in area *i* in year *t*,

$M_a(t)$ and $M_j(t)$ are the adult and juvenile natural mortality in year *t*,

$F_a(t)$ and $F_j(t)$ are the adult and juvenile fishing mortality in year *t*,

f_j is the proportion of juveniles who become adults in a year

$W_a(i,t)$ is the number of adults without breeding territory.

4.2.3 Shape and relative size of the protected and the fished area

Generally an MPA is modelled as a rectangle in which the number of sides in common with the fishing area is chosen according to the purpose of the authors. Some of them used a three-side model to describe shore reserves (DeMartini, 1993). Others used a four-side shape to model offshore reserves (Guenette and Pitcher, 1999; Polacheck, 1990). Lindholm et al. (2001) modelled the protected area as a collection of squares randomly chosen in a lattice of cells. The area of the reserve was normally expressed as a fraction of the total area. It varied from 0 to 100% of the total area in simulations that wanted to describe the effect of FEZ size on fish populations and related fisheries.

4.2.4 Indicator variables used to monitor the performance of a FEZ

FEZs are proposed as an alternative solution to actual management strategies, mostly based on fishing effort and mesh size regulation. Their purpose is to increase the commercial catches in the adjacent areas open to fishing, and to reduce their variability through the protection of large sized fishes, which are known to be the main producers of eggs. As a consequence the authors of the simulations chose the variables that expressed the catches obtained in the unprotected area and the spawning stock biomass (SSB) of the total population to monitor the performance of a FEZ. Apart from total catch, which was employed by all the authors as an indicator variable, Polacheck (1990) proposed to use the ratio of the SSB (realised over the life span of a cohort) to the number of recruits (R) as a monitoring variable, because it was suggested as a measurable management objective. SSB/R and yield per recruit (Y/R) were calculated by standardising the total spawning biomass and yield of the cohort over its life span by the total number of recruits (potential recruits in the protected area and effective recruits in the fished area). The indicator variable in the case of Polacheck (1990) and DeMartini (1993) was the percent SSB/R level relative to the SSB/R level that would be realised if no fishery existed.

Besides these indicator variables, Guenette and Pitcher (1999) monitored the number of years in which the simulated fishery showed low recruitment, *i.e.*, recruitment under an arbitrary threshold. Mangel (2000) used the probability of maintaining a population at a sustainable level, that is the fraction of simulated population that in presence of a reserve showed an abundance of fishes 35% higher than simulated populations without a protected area.

4.2.5 Results

Any FEZ establishment is likely to produce a reduction of total catch, because the Y/R is a decreasing function of the FEZ size (DeMartini, 1993; Polacheck, 1990). In fact at the maximum sustainable yield (MSY) level of exploitation, the yield is 12-50% lower for reserves sizes of 30-70% of the total area (Guenette and Pitcher, 1999). But, according to some of the simulations reviewed, at higher levels of exploitation the no-reserve regime collapsed while the reserve regime was kept at 23% of MSY (Guenette and Pitcher, 1999; Sladek Nowlis and Roberts, 1999). FEZs are likely to have a "buffer" effect on the catches by diminishing their variability over the years (Mangel, 2000; Sladek Nowlis and Roberts, 1999), which may avoid the boom and burst cycles often caused by over-exploitation. Polacheck's model predicts that the presence of a FEZ induces a shift in the age distribution towards older individuals. This favours the SSB/R, which increases as the refuge size increases (DeMartini, 1993; Polacheck, 1990). Closing from 10 to 50% of an area can produce an enhancement of the spawning biomass up to 200% (Polacheck, 1990). Guenette and Pitcher 1999) drew the same conclusion but only for FEZ sizes larger than 30% of the total area.

The movement rate of fishes decreases the benefits of FEZs (DeMartini, 1993; Guenette and Pitcher, 1999; Lindholm et al., 2001; Polacheck, 1990). This is due to the decrease in survival probability. In fact Polacheck (1990) reported that, with a small protected area (10% of the fishing area), a 10-year old individual will have a chance to survive 25 times greater if the transfer rate (T) is 0.10 rather than 1.0. As a consequence, at high migration rates closed areas have very little effect on both spawning biomass and yield. So the FEZ size should be bigger in the case of species with high movement rate. The optimal reserve proportion increases with increasing fishing mortality, and heavily exploited fisheries require particularly large reserves to remain productive (Sladek Nowlis and Roberts, 1999).

4.2.6 Discussion

On an overexploited stock, the implementation of a year-round closure may be a viable option to reduce fishing mortality and increase the spawning biomass. Mobility of fishes is the key parameter: at low movement rates small closed areas can yield substantial benefits. Another key parameter is the size of the closed area: size should be decided according to both management objectives and the biology of target species. A factor that should be kept in mind is that the larger the closed area, the smaller the total catch (at least in the short term), and this is likely to call for complaints from the fishing industry.

From the literature reviewed, it is clear that models often need more data than those actually available, and this is particularly true for regions where an established tradition in marine science and long and reliable time series of data are missing or inadequate. Information on crucial data like migration rates, stock size, fecundity, etc is often unavailable in many exploited areas. This framework is made even more complicated when the multispecies character of most temperate and warm-water fisheries is taken into account. Further, it is worth noting that much of the modelling work concerning FEZs has focussed entirely on no-take zones, with little consideration given to regimes that allow preferential access to some but not all types of fishing vessel. This same point will be re-iterated in the next section, which looks at bioeconomic modelling.

5. Bioeconomic modelling

5.1 From biological to bioeconomic models

The use of mathematical models to assess the usefulness of marine protected areas (MPAs) for stock management was first proposed by Beverton and Holt (1957).[1] Nevertheless, until the late 1980s and early 1990s, little work developed in this area.[2] For the past ten years, and especially in the last five, there has been a remarkable growth of interest in the literature considering the development of mathematical models to attempt to evaluate the potential benefits that MPAs may be able to provide. It is however noticeable that much of this modelling research has concentrated on the biological effects (e.g. DeMartini 1993, and Man *et al.* 1995). Guénette et al. (1998) provide a comprehensive review of these models highlighting some of the main features, including; species population, adult migration, exportation of larvae, yield and protection against overexploitation. The aims of these models have generally been to look at the effects of MPAs towards conservation and environment protection, not specifically for the management of fisheries and the fishing industry.

In fact, there have been few significant attempts to analyse the subsequent economic effects that will undoubtedly exist. It is clear that in overexploited fisheries where MPAs have been proposed as management tools, the importance of the fishing industry dictates that any realistic analysis must include the effects on fishermen from an economic perspective.[3] In recent years, bioeconomic model-based analysis has been growing in the literature. This is not least a result of the many MPA based conferences, workshops and special streams at conferences that have taken place in recent years, such as: the International Symposium in Fisheries Ecology: Essential Fish Habitat and Marine Reserves (November 4-6, 1998, Sarasota, Florida); VALFEZ (April 2000); and the Conference on the Economics of Marine Protected Areas (July 6-7, 2000, Vancouver, Canada).

In the design and evaluation of marine protected areas (MPAs), mathematical models are a useful approach in the toolkit to investigate some of the effects that may result from the implementation of an MPA. Such models offer a formal framework where the effects of alternative measures can be considered, and potential benefits and/or drawbacks can be predicted. However, the validity of using mathematical models for MPA design is based principally on the knowledge and data available for the study under investigation with the accompanying requirements of the MPA. The basic framework of MPA evaluation is given in

[1] See Guénette *et al.* (1998)
[2] See Conover *et al.* (2000) for an overview on the popularity of publications in marine reserves. In five year periods from 1976-1999, they report growth in average papers per year to be 0.2, 0.6, 2.6, 10, 22.2.
[3] This is particularly the case for temperate zone fisheries. It should be noted that many of the models in the literature have been concerned with tropical (often reef-based) fisheries.

figure 3. As shown, modelling forms a section of the evaluation tools that exist, not all of which may be appropriate for the evaluation of MPA for a given area.

Commonly declared benefits that can be expected to result from MPAs are: to improve population sizes and structures for key species; to increase the individual size and abundance of fish inside and outside the MPA; to sustain habitat or non-fish species; and to provide insurance against scientific uncertainty. However, the effectiveness in which these are achieved depends largely on the design and situation of the MPA(s). Other practical considerations include the displacement of fishermen, level of protection given by the MPA and enforcement (specifically of fishing activity). Apart from the last of these, mathematical models can play an invaluable role in the assessment of MPA design. Therefore, some of the key questions that a model is required to provide information towards in the design and evaluation of MPAs are: where should the MPA(s) be sited?; what size should the MPA(s) be?; when will the benefits be visible?; what will be the effects on fishermen displaced?; and, what will be the effects on other fisheries (or areas)?

5.2 Data requirements of bioeconomic models for MPA design

As the name implies, bioeconomic models are generally formed of two distinct components; biological and economic information, even though social and/or regulatory factors may be included. From a biological perspective, there are 2 basic types of model that are developed: (i) age-structured models – requiring weight-at-age and length-at-age data for given species; and (ii) surplus production models – assuming an overall growth function for a species (or group of species).

Ideally, in order to assess marine reserves for a given area, complete spatial knowledge is required for relevant species throughout their lifecycle. The main parameters are therefore age- and/or length-based for population size, distribution and movement (including dispersal and migration) in the area of study, *i.e.* where do they spawn, congregate, migrate to *etc*. The spatial knowledge allows immediate classification of MPA or non-MPA per subarea. Typically, the majority of studies published simply look at an inside area and an outside area with a single species or single global biomass. However, Holland (2000) considers 3 key species (cod, haddock and yellowtail flounder) on Georges Bank split into 16 sub-areas as part of his model. The economics is then built on the catch that can be obtained given a level of fishing activity (designated by type of activity) in a certain area at a certain time.[4] It is also desirable to have a long time-frame of varying fishing activity giving some knowledge to how different levels of fishing activity can affect the structure of stocks.

In reality, the data and knowledge available is limited. Generally, only the key commercial species that have a high level of fishing mortality are strongly researched (see International Council for the Exploration of the Sea (ICES) reports). Furthermore, interactions in multi-species environments due to predation, displacement or other effects are generally also not known accurately. In an excellent Guide to MPAs, Baker (2000) discusses “a theoretical *wish list* of data needs for design of fisheries MPAs”. These can be summarised as:

1. ‘seascape’ ecology – includes the specifics and limiting factors of the living environment, particularly with ‘what happens where’;

[4] Generally, restrictions due to regulation are implicitly included in this.

2. meta-population dynamics – defined as a distinct population of one or several connected species, where environmental variation is under consideration on the underlying population structure, spatially and over time;
3. minimum viable population size – based on age-structure;
4. source and sink dynamics – simplistically, this is where a species spawns to where it lives as an adult, more generally it includes movement of the population throughout its lifecycle (Sanchirico and Wilen (1999 and 2001) consider the modelling aspects of this in more detail);
5. population dynamics – the scale and distribution of a species by age over the area; and
6. habitat protection – which provides links between habitat and healthiness of the stock, e.g. sea-grass, also varieties of habitats and links between them from a physical viewpoint.

As developed in VALFEZ (2000), three of the main design criteria for an MPA that result are form (i.e. size, number and shape), location and temporal scale (e.g. operating seasonally, temporarily or permanently). Integrating the available knowledge and data into a modelling framework to consider these facts, efficacy of proposed MPAs can then be evaluated against the displacement (or removal) of fishing effort, species' stock levels and other environmental consequences.

5.3 Review of bioeconomic models of marine protected areas

Progress in developing bioeconomic models of MPAs is comparatively recent, and most of the important work in this area dates from the mid-1980s. There is a wealth of literature concerning standard bioeconomic and biological models for fisheries, which forms a good basis for development. However, due to the newness of bioeconomic modelling application to MPA, there are still few articles actually published. Table 1 lists the main contributions in the journal literature to date, detailing their applicational areas and key features. Several studies of each type of biological model are given, namely age-structured and surplus production based models. The main objective of the analysis has been to evaluate the optimal size of MPA design.

The spatial nature of the models has been restricted in most cases to inside and outside the MPA, primarily modelling a 'no-take' zone inside. Sanchirico and Wilen (1999 and 2001) considered the potential of incorporating integrated and explicit spatial interaction of biological and economic systems in a model. Primarily, they used equilibrium based logistic growth functions, where biomass is dependent on typical growth relationships, but also to dispersal rates over areas at a given time. Although many of the ideas presented in their study are particularly interesting, to our knowledge they have yet to be applied to a specific case study. Other than applicational papers of bioeconomic modelling to MPA, there have been several theoretical contributions presenting potential bioeconomic models for MPA evaluation. They have generally used hypothetical data to discuss theoretical developments rather than a specific case study (*e.g.* Hannesson 1998; Sanchirico and Wilen 1999 and 2001; and Li 2000). The model developed by Hannesson (1998) is useful as a 'black box' single species generic MPA model. In addition to being of use to specific case studies, possibly with slight modification (*e.g.* Boncoeur *et al.* 2000), this model can be of notable worth in the early stages of the MPA design and evaluation process when only 'sketchy' details are known about parameters. The model can be viewed almost as an overview of the fishery in question. By modifying parameters and investigating alternative scenarios, results obtained can then be used as an indication of how and in what direction the evaluation should proceed. A similar model has been proposed by Li (2000), looking at the optimal size and optimum harvesting through MPAs.

Although attempts have been made in studies to evaluate the effects on profitability in the case study fisheries, only Holland (2000) has incorporated fleet dynamics into the modelling structure. In our opinion therefore, this application deserves specific mention. The model includes spatial fleet dynamics and fishermen activity functions to analyse possible effects on the fishing industry. Table 1 lists the key features of this study. However, a simulation model is developed using a Baranov catch equation, which follows a negative exponential-based distribution, $a\left(1-e^{-Z}\right)$, where *a* is the number of fish susceptible to fishing mortality and *b* is the total mortality of the stock. Catch is considered by cohort and sub-area fished. After each 5 days (length of an average fishing trip) fish migrate according to given rates of diffusion and seasonal movements between sub-areas. It is assumed that there is an equal movement of fish between adjacent sub-areas. Seasonal movements are best-guess estimates based on observed cod movement, *e.g.* cod and haddock move east-west. Recruitment is divided equally between areas. Weight-at-age and per-kilogram prices are then used to calculate total revenue. Fleet dynamics are based on a defined utility function for fishermen activity choice selection. The number of vessels is fixed to 150 and average fishing power of vessels is assumed to be proportional to time fished. As noted previously, it is clear that in overfished fisheries especially, the introduction of MPA(s) will have an obvious effect on the fishing industry in terms of displacement or redundancy. Holland (2000) has produced a framework that attempts to analyse some of these effects.

Further to specific bioeconomic models that have been developed to evaluate MPA, generic packages to analyse effects have also been developed in recent years. Ecosim is one such package, principally developed by Carl Walters at the University of British Columbia, and is part of the Ecospace project initiated in part by the European Commission. It is described in the promotional material as “a dynamic simulation module for predicting results of human and climatic impact on ecosystem components”. Bio-economic consequences of harvest strategies can be evaluated by entering estimates of costs and prices. Equilibrium analysis to study the impact of fishing effort on yield and biomass can also be undertaken for all ecosystem components. Walters (2000) gives a general discussion on using Ecosim for MPA design. An example of Ecosim in use is reported by Pitcher *et al.* (2000). They apply Ecosim to fisheries in the South China Sea using an MPA and artificial reef (AR) system. They find that with an MPA/AR system of 10-20%, significant benefits could be realised within 10 years.

5.4 Discussion

From the modelling analyses published, there is a wide difference of opinion between authors on the potential of marine protected areas. In the review by Guénette *et al.* (1998), the overall conclusion from the models examined, and the authors, is towards the promising potential of reserves. Rodwell *et al.* (2000) and Pezzey *et al.* (2000) emphasise this in concluding that the benefits achieved make them essential in certain areas. However, DeMartini (1993), Conrad (1999) and Hannesson (1998) for example are not optimistic about the role that marine reserves can play in fisheries management. Due to the inherent variability in fisheries, the conclusions of most are that marine reserves could play a useful role if certain conditions are met (or even, can be guaranteed). One clear deficiency in the majority of these studies is the lack of some crucial data: accurate ecological knowledge, migration and/or transfer rates, and economic (especially fleet) data to analyse the effects of a changing environment to the catching fleet and vice-versa. The main aim of bioeconomic models in the field must be to improve the detail and reliability of information produced. This will also enable analysis of some of the short-term effects imposed on the fishing industry as well as the more traditional long-term conservation effects of the stocks.

In terms of practicability, the main difficulty with the design of MPAs arises from the multi-species factor. The majority of fisheries fall into the multi-species category, and very rarely (except with sedentary species such as lobster) are individual species targeted alone. Also, it is uncommon to find two species that spawn, move and generally exist in the same areas at the same time. It is therefore unsurprising that in the multispecies bioeconomic model developed by Holland (2000), the impacts vary across species, with some experiencing increasing yields and some decreasing. As noted by Pezzey *et al.* (2000), an outcome of their model analysis is that reserves' potential for protecting biodiversity and increasing tourism are far clearer than for increasing catches. More specifically, Holland and Brazee (1996) conclude from their bioeconomic model that marine reserves can probably sustain or increase yields for moderate to heavily fished fisheries, but will probably not improve yields for lightly fished fisheries.

The main design aim of models to date has been to establish the optimal size of an MPA. The majority of these studies have concluded that this size needs to be around 50% of the area viewed as the stock domain. For example, Man et al. (1995) using a metapopulation model found that the closed area must be around 50% of the habitat to be effective. Also Pezzey *et al.* (2000) note in their coral reef analysis that the reserve size should be about 50% of the original fishing area. In the models developed by Hannesson (1998) and Sumaila (1998), a range of migration (or net transfer) rates were used to investigate the size of reserve required considering a single species. Hannesson (1998) concluded that in tests the conservation effects of MPAs are critically dependent on the size of reserve and migration rate, where a high rate implies a large reserve. He further noted, by comparing 3 scenarios of open-access fishing with and without a reserve, that the reserve will increase fishing costs and overcapitalisation, and for seasonal fisheries will shorten the season. Moreover, a reserve of appropriate size would achieve the same conservation effects as an optimum fishing strategy but with smaller catch. Similarly, by considering the Barents Sea cod stock, Sumaila (1998) concluded that reserves can be bioeconomically beneficial when net transfer rates are high and the reserve size is large. However when net transfer rates are low reserves do not mitigate against losses in rent achieved from the fishery, although they are beneficial for stock protection.[5] In a case study of the Mombassa Marine National Park, Rodwell *et al.* (2000) found that the optimal area of the reserve is 15-25% of original area if less than 40% of the biomass is exploited. They note that larger reserves are required if fishing effort cannot be controlled. A further benefit of MPAs as noted by Conrad (1999) is that reserves could help reduce biomass variance, although in the example presented reserves less than 30% of the overall fishing grounds produced higher than average variance than in the "no sanctuary" case.

Much of the bioeconomic modelling work undertaken has concentrated solely on no-take zones, a situation which parallels that relating to biological models. This emphasis is misplaced, since in EU waters especially there are many defined MPA and almost all allow some degree of fishing. The pure 'no take' zone is the exception rather than the norm. The inclusion of economic

[5] In this study (Sumaila 1998), a range of net transfer rates are discussed but no attempt is made to identify an exact rate.

aspects depends on the aims of the MPA, whether for fisheries management or purely conservation. As the majority of the research published has concentrated on biological-based models, the analysis to date has therefore tended towards the effects of MPAs on conservation.

Bioeconomic models can certainly offer useful information to assist in the design and evaluation of MPAs, to highlight some of the benefits and drawbacks that may accompany them. There is a range of opinion from the models produced until now about the role that MPAs can play in fisheries management. However, it is generally agreed that under certain circumstances, particularly with less migratory (or even sedentary) species, that MPAs can be a useful management tool. In the type of fishery modelled by Rodwell *et al.* (2000), they conclude that reserves are essential for the management of fishing effort directed at the stock. Conrad (1999) is less positive by stating that MPAs provide some basis as a "hedge" for management.

Finally, there are obvious links to the standard bioeconomic modelling literature, which is well established, and MPA bioeconomic modelling. With the growing interest in using bioeconomic models to design and evaluate MPAs, the undoubted role that they can play will be made clearer with application.

Figure 3: **Framework of evaluation for key stages in MPA design**

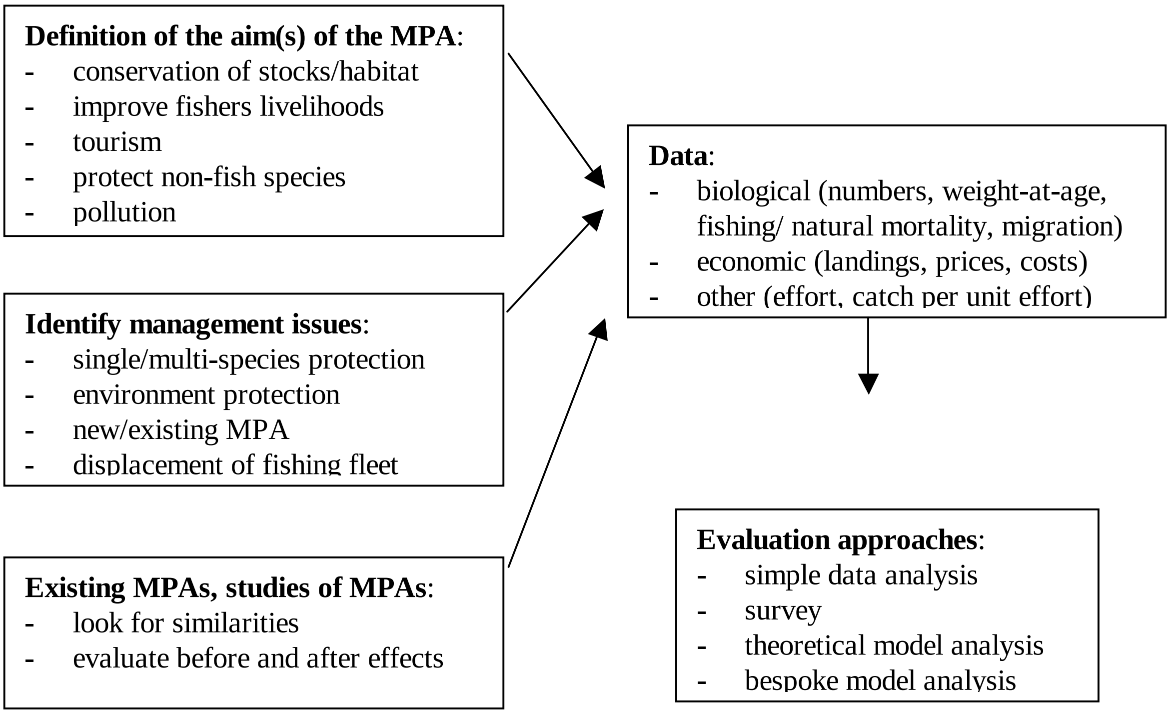


Table 3: **Key publications on bioeconomic modelling of MPAs**

Author(s)	*Study area*	*Species*	*Type of MPA*	*Key features/assumptions:*
Age-structured models:				
Holland (2000)	Georges Bank (north-west Atlantic)	Cod, haddock, yellowtail flounder	Permanent, fisheries management	• spatial model with fixed level of nominal fishing effort • fish move between areas within zones but not between zones • von Bertalanffy weight-at-age functions estimated from data • same biological model used for all species (different parameters) • fishermen activity included
Sumaila (1998)	Barents Sea (north-east Atlantic)	Cod	Fisheries management	• stock & recruits are evenly distributed and randomly dispersed (constant density) • dynamic simulation model looking at optimal size development • population is split into protected and unprotected • net movement from protected to unprotected area (*net transfer rate*)
Holland and Brazee (1996)	Gulf of Mexico	Red Snapper (reef fishery)	Fisheries management	• dynamic model considering the equilibrium position and path to it • effort fixed • optimal reserve size • sensitivity to assumptions measured
Surplus production (or logistic growth) models:				
Boncoeur *et al.* (2000)	Iroise Sea (north-west France)	Fish stock Vs. seals	Ecotourism (reserve and fishing area)	• predator/prey bioeconomic model • seal watching is a commercial activity (boat tours) • uses a plurispecies extension to the Hannesson (1998) model • fishing effort is limited by licenses
Pezzey *et al.* (2000)	Coral reef fishery		Fisheries management	• Gordon-Schaefer based equilibrium model • 4 general tropical fisheries are investigated
Conrad (2000)	Offshore fishing grounds	Fish stock	Fisheries management	• 2 models: deterministic (optimally managed Vs open access and reserves of varying sizes with optimally managed general access parts – present value calculated) and a stochastic recruitment model with a linear TAC policy • No fleet dynamics or specific case study
Rodwell *et al.* (2000)	Mombasa Marine National Park	Global biomass	Permanent	• spawner-recruit model, no fleet dynamics or stochastics • larval dispersal and retention with zero, moderate and high adult migration • basic reserve area position is assumed with a single stock
Hannesson (1998)	n.a.	Single species	Permanent	• equilibrium models – no specific case study applied • 3 comparisons: open access all, open access outside, and optimum fishing
Li (2000)	n.a.	Single species	Fisheries management	• Reserve size included as a control variable • Cooperative harvesting behaviour • Optimal harvesting fishery modelled as a perpetual annuity investment
Sanchirico and Wilen (1999)	n.a.	Single species	Fisheries management	• Population structure incorporates biological notions of spatial patchiness, heterogeneity and interconnectedness • An ‘economic gradient’ operates to reallocate effort and a ‘biological gradient’ operates to reallocate biomass across space
Sanchirico and Wilen (2001)	n.a.	Single species	Fisheries management	• Spatial bioeconomic model used to simulate effect of marine reserve creation under different ecological structures • Identifies circumstances giving rise to a ‘double payoff’ of both increased biomass and harvest in a spatial system

6. Summary and conclusions

The Work Package has examined various approaches to analysing fisheries exclusion zones (FEZs) and, in the process, has attempted to articulate the main questions which are commonly raised by fishery managers and others. The information needed to answer questions concerning FEZs typically takes the form of empirical measures (often expressed or interpreted as performance indicators), and the significance of these measures was discussed within the context of the pressure-state-response (PSR) paradigm. This offers a useful analytical framework for exposing the linkages between the anthropogenic activities which impact on the coastal and marine environment (e.g. commercial fishing) and feedback in the form of policy responses (e.g. control measures, including FEZs).

Detailed consideration was then given to the way in which data concerning FEZs could be generated from socio-economic and biological assessments, as well as from bioeconomic modelling. Three generic types of socio-economic assessment were distinguished (profiling, impact analysis and benefit assessment) and the problems of ascribing monetary values to the various inputs and outputs affected by FEZs were discussed. These included: firstly, the difficulties of measuring incremental (i.e. marginal) values impacted by a FEZ; secondly, the absence of market prices attaching to environmental services; and lastly, the non-constancy of monetary benefits over time and hence the need to discount net returns according to how far into the future they were likely to be received. Biological assessment of FEZs was examined from several aspects. Most empirical studies of the biological effects of exclusion zones have been ‘within FEZ’ investigations, while relatively few have explored the effects on the fishery as a whole. Mathematical models of the biological effects of FEZs were shown to hold considerable potential, but the evidence suggests that such models typically require more data than is actually available. The same problem applies *a fortiori* to bioeconomic models of FEZs, which require data on both the biological and the economic components of the fishery system. Some bioeconomic modelling studies have been purely theoretical exercises, but in a few cases attempts have been made to calibrate the models against field data and to apply them to real-world fisheries where exclusion zones are in operation. Bioeconomic modelling represents a useful, and undoubtedly challenging, approach to investigating the costs and benefits of FEZs. However, if this work is to be of greater use to EU fisheries policy in the future, its focus should arguably be broadened to include not simply pure ‘no take’ zones but also partial exclusion zones that permit some degree of fishing – these being more representative of what happens in EU waters.

Annex 1: The significance of the discount rate in evaluating the economic worth of FEZs

Evaluating the economic worth of a proposed FEZ requires a methodology to allow for the fact that the costs and benefits will typically arise over an extended period of time. The procedure used in economic appraisal is to discount costs and benefits over a specified number of years using a pre-determined interest (= discount) rate in order to establish whether the sum of the net discounted returns is greater or less than zero. This is termed the *net present value* (NPV) criterion, where NPV is based on the summation of the series:

$$NPV = (B_0 - C_0) / (1 + r)^0 + (B_1 - C_1) / (1 + r)^1 + \ldots\ldots + (B_n - C_n) / (1 + r)^n$$

Where:	$B_0 \ldots B_n$	= benefits expected in each year 0 to n
	$C_0 \ldots C_n$	= costs incurred in each year 0 to n
	r	= discount rate

If NPV is positive is implies that the returns from the FEZ exceed the opportunity cost of capital, and the FEZ should therefore be introduced. Strict interpretation of this approach demands that all quantifiable impacts of the FEZ (including externalities) be monetised and incorporated within the stream of benefits and costs. The NPV thus calculated provides an absolute measure of the net gains to society as a whole.

To illustrate the importance of discounting, consider a fishery currently regulated through a system of vessel quotas and licences. Dissatisfaction with its economic performance has led to the proposal to spatially partition the fishery into a closed 'no-take' zone and an open area within which boat numbers would be limited and catches controlled. Biological assessment has suggested that catches from the fishery are likely to fall sharply following the partitioning of the fishery, and while there will be some cost savings as a result of effort withdrawal the net result in the short run will be a fall in economic rent. However, stock recovery within the no-take zone is expected to lead to biomass export across the reserve boundary, raising harvest opportunities within the open area. Effort and catch are assumed to be regulated such that a substantial proportion of the potential economic rent is captured, and this is calculated to be higher in the long run than that currently being earned under the existing regime. This is shown in the Table below.

The partitioning of the fishery is thus expected to yield an increase in the annual flow of net benefit, but not without a significant delay and at the cost of a short run fall relative to the status quo represented by the current regime. Whether the area closure strategy is economically worthwhile or not depends *inter alia* on the time value of money as measured by the discount rate. In the hypothetical example given in the Table we consider discount rates of 3%, 6% and 10% and calculate NPV over a 30 year time horizon. At the 3% rate the NPV is positive, implying that area closure is the preferred alternative. As the discount rate increases, however, a greater weight is attached to the short run over the long run, with the result that at a 6% discount rate or higher the economic advantages of the area closure strategy are effectively wiped out.

Annex Table 1: Illustrative example of the influence of discounting on the net benefits from a fishery

Years	Economic rent under alternative management scenarios					
	Current regime	Area closure	Net benefit	Net benefits discounted at:		
				3%	6%	10%
0	80	20	-60	-60.0	-60.0	-60.0
1	80	25	-55	-53.4	-51.9	-50.0
2	80	30	-50	-47.1	-44.5	-41.3
3	80	45	-35	-32.0	-29.4	-26.3
4	80	65	-15	-13.3	-11.9	-10.2
5	80	85	5	4.3	3.7	3.1
6	80	100	20	16.7	14.1	11.3
7	80	100	20	16.3	13.3	10.3
8	80	100	20	15.8	12.5	9.3
9	80	100	20	15.3	11.8	8.5
10	80	100	20	14.9	11.2	7.7
11	80	100	20	14.4	10.5	7.0
12	80	100	20	14.0	9.9	6.4
13	80	100	20	13.6	9.4	5.8
14	80	100	20	13.2	8.8	5.3
15	80	100	20	12.8	8.3	4.8
16	80	100	20	12.5	7.9	4.4
17	80	100	20	12.1	7.4	4.0
18	80	100	20	11.7	7.0	3.6
19	80	100	20	11.4	6.6	3.3
20	80	100	20	11.1	6.2	3.0
21	80	100	20	10.8	5.9	2.7
22	80	100	20	10.4	5.6	2.5
23	80	100	20	10.1	5.2	2.2
24	80	100	20	9.8	4.9	2.0
25	80	100	20	9.6	4.7	1.8
26	80	100	20	9.3	4.4	1.7
27	80	100	20	9.0	4.1	1.5
28	80	100	20	8.7	3.9	1.4
29	80	100	20	8.5	3.7	1.3
30	80	100	20	8.2	3.5	1.1
NPV				98.8	-2.9	-72.0

Annex 2: The Ecopath/Ecosim approach

Ecopath with Ecosim (Anon., 2002a, 2002b) is a simulation environment (available as a freeware download from http://www.ecopath.org) that allows, among other things, to address the implications of particular fishery activities on marine ecosystems (Jennings *et al.*, 2000; Pauly *et al.*, 2000). The simulated ecosystem is composed of different groups of organisms interacting through trophic relationships. The flows of matter between each component are quantified by mass-balanced equations that describe the production and consumption processes.

Simulation: method and results

While the models described in the section on biological modeling deal with single species, or with a few ecological categories at most, Ecopath/Sim uses an ecosystem approach. All groups of organisms are split in two sub-pools: one is protected from fishing activity while the other suffers fishing mortality. The initial biomass of each group is distributed in the sub-pools proportionally to the relative size of the MPA. A rate of transfer between each sub-pool is set and can be varied between different runs of the simulation. Fishing mortality is held constant between simulations in order to model a redistribution of the fishing effort following the establishment of an MPA. The capability of Ecopath/Sim to build spatially explicit models has enabled Watson *et al.* (2000) to investigate the impact of MPAs on fishery activity in shallow water areas of the Gulf of Thailand. We have taken Watson's paper as a good example of application of the Ecopath/Sim principles.

According to the simulation, from the fishery point of view the establishment of an MPA results in a loss on the short time scale (1-3 years): trends in simulated catches (all groups of organisms considered) are always negative. They vary proportionally to the size of the protected area: the bigger it is the larger is the loss for the fleet. In the long term (10 years), MPAs provide an increase of catches. Its magnitude depends on the size of the MPA, with an optimal size for the closed area lying between 10% and 15% of the fishing area. The transfer rate between sub-pools of organisms is a critical parameter that modifies the efficiency of an MPA. For high transfer rates, the maximum biomass is reached with larger MPA size than for organisms with a low transfer rate. Instead catches reach an optimum level for a determinate size of the MPA (10% to 30%) independently of the transfer rate of the organisms: too large MPAs impede fishers to fully exploit the enhanced biomass.

Conclusions

The Ecopath/Sim modeling approach differs from single species assessment by providing a framework to study the impact of fishing activities on marine ecosystems. Its application to the evaluation of MPAs as a management tool for fisheries provides similar overall conclusions than those achieved using single species simulation. But Ecopath/Sim can give detailed insight into the response of each particular group of organisms: this approach can provide us with a larger view of the effect of the implementation of MPAs.

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