Review of Marine Protected Areas as a Tool for Ecosystem Conservation and Fisheries Management



PROTECT
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PROTECT

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Introduction

This report is the result of work carried out by the research project "PROTECT" - or in full "*Marine Protected Areas as a Tool for Ecosystem Conservation and Fisheries Management*" - through support from the 6th Framework Programme under the European Commission and 17 European research institutions, 2005-2008 (see *www.mpa-eu.net*). The overarching goal of the project is to strengthen the decision base regarding potential use, development and management of marine protected areas (MPAs) in Northern Europe as part of an ecosystem-based approach to fisheries management.

Marine protected areas are currently seen as a tool for both fisheries management and marine environmental protection. However, although many potential benefits of MPAs can be identified, little empirical evidence exists to demonstrate the full potential of MPAs in a temperate water setting. This is partly due to insufficient scientific knowledge and tools for MPA selection, design, implementation, monitoring and evaluation. In particular, linkages between fisheries management and marine environmental protection require attention.

The present work is implemetend under Work Package 2 of the project with the objective to collate and review the state of the art of recent knowledge and experiences, and to identify central, often cross-sectorial information gaps regarding MPA development and management.

For MPAs aiming to serve as a tool for integrated ecosystem conservation and fisheries management in European waters, it is important to recognise the multi-faceted and cross-sectorial nature of the issues involved, calling for collaboration and synergies between a range of research and management disciplines.

In this review, a range of different aspects and disciplines are discussed by experts across relevant fields, addressing topics related to MPA selection, development and implementation, legal issues, formulation of ecosystems features, management objectives and 'success criteria', hydrographic-, biological- and economic modelling approaches to evaluate MPA effects, development of monitoring and data collection strategies, as well as social and economic aspects. Each topic is introduced with a review of the key literature, followed by discussions of approaches relevant to the project.

The review serves a dual purpose: as an internal working tool to assist project development and planning, informing and prioritising specific approaches, research questions and activities; and to inform wider management and policy fora of current knowledge, experiences and thinking regarding the use of MPAs.

Three case studies, representing different characteristic ecosystems and management scenarios, serve as focus point for the 'generic' project disciplines. These are: 1) a top-down controlled ecosystem in the Baltic Sea currently experiencing strong fishing pressure on cod, the top-predator species; 2) a 'wasp-waist' ecosystem in the North Sea with substantial fishing pressure on sandeel at the the mid-trophic level, serving as an important food source for sea birds and other animals; and 3) deep water coral ecosystems in the Northeast Atlantic serving as essential habitats for many species, but currently negatively affected by ongoing fishing practices.

The findings and outcomes of the research on frameworks and new tools for MPA development, supported by specific case study examples, are used in further synthesis and formulation of recommendations for management and policies, including the involvement of stakeholders in the MPA development process.

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Executive Summary

This report is the result of work carried out by Work Package 2 (State of the Art of MPAs as management measures) of the EU FP6 project "PROTECT – *Marine protected areas as a tool for ecosystem conservation and fisheries management*", 2005-2008 (further details at *www.mpa-eu.net*).

PROTECT is a research and development project involving 17 European institutions aiming to strengthen the decision base regarding the potential use, selection, development, evaluation and management of marine protected areas (MPAs) in Europe, as part of an ecosystem-based approach to fisheries management.

The use of MPAs as tools for ecosystem conservation and fisheries management is a multi-faceted task that requires integration and synergies between different scientific disciplines. Many of these disciplines are identified and discussed in the review. Project partners have contributed chapters in their specific fields of expertise, and the findings have subsequently been discussed among other experts in the project network. The review has been structured, collated and edited by the WP2 coordinating team.

The chapters and main conclusions of the review are summarised below:

- **Terms & definitions:** In the past decades, many different terms have been applied to the concept of managing geographically defined marine areas for the purpose of conserving the environment and/or managing fisheries. Today, the term "marine protected area", or MPA, is to some extent misinterpreted, and in some cases misused, due to the lack of a common definition and understanding of the term in international and European scientific and management communities, laws and policies as well as the wider public. A selection of terms, definitions and applications of MPAs is presented, including the following working definition in relation to PROTECT: The term MPA is defined as *"...any marine area set aside under legislation or other effective means to protect marine values"* (referring to e.g. conservation, commercial, scientific, recreational, cultural and aesthetic marine values).

- **Legal frameworks for MPAs:** The legal basis underpinning the establishment of MPAs as a tool for ecosystem conservation and fisheries management is described in connection with a number of legal instruments, including international and regional treaties, European Law and policy, as well as a number of "soft law" initiatives.

  There exists a broad range of legal instruments and policy documents that recommend the adoption of MPAs and there is sufficient legal basis within the EU Common Fisheries Policy and European environmental policy to implement an MPA-driven approach. Suggestions are made regarding the legal base on which different types of MPAs might best be established in different geographical regions.

  From a legal perspective, MPAs may only be used as a tool for ecosystem conservation and fisheries management if they are proportionate, science-based, enforceable and specific for each marine area and management objective, consistent with the ecosystem approach, and conform to European and international law. In addition, any measure to implement MPAs for fishery management purposes should be subject to scientific advice and assessment by ICES and STECF.

- **Present and past MPAs used in fisheries management:** A review is presented of the general literature regarding MPAs as fisheries management tools in temperate waters, followed by a presentation of several existing Northern European MPAs (and one North American MPA) based on scientific papers and technical reports. The review concludes that most of the existing North Sea MPAs have had little success in reaching their management objectives. In most cases, it is difficult or impossible to separate effects of management measures and natural variability ocurring during the lifespan of the investigated MPAs.

- **PROTECT case studies:** Three case studies and their current status are introduced and discussed: 1) a *'top-down controlled'* ecosystem represented by cod fisheries and fishing closures in the Baltic; 2) a *'wasp-waist'* ecosystem represented by the sandeel fishery and its implications for seabird colonies that depend on sandeel as an important food resource in western parts of the North Sea; and 3) a deep water coral ecosystem, represented by the *Lophelia pertusa* coral reefs in the northeast Atlantic.

- **Measuring the success of MPAs:** MPAs are established for a wide range of purposes, and there are different considerations involved in determining to what extend a given MPA is reaching its predetermined *goals*. To evaluate performance agaist a predefined MPA *goal*, specific and measurable *objectives* must be defined in terms of what outputs and outcomes are expected. This in turn requires well-defined management plans, pre-defined criteria for MPA *success*, and monitoring of the impact of management actions (see below). The results of these activities should be fed back into the MPA planning process for possible revision of objectives, plans and outcomes, i.e. so-called *adaptive management*.

  Working definitions of *goals* and *objectives* are presented, and a process for defining *success criteria* and related *indicators* of progress is described. A brief description of modelling approaches to evaluate MPA success is provided.

- **MPA monitoring strategies:** Monitoring is defined as repeated observations of one or more parameters according to prearranged schedules in space and time through comparable collection methods. Monitoring programmes should be an integral component of the management of MPAs and fisheries, as they are essential in determining the effectiveness of conservation tools, allowing for adjustment of MPA design and providing information on progress to stakeholders, funding agencies and civil societies. In order to answer key questions about the progress and effectiveness of MPAs, case specific *indicators* and *success criteria* need to be defined and measured on a spatial and/or temporal basis. Procedures for the use of readily available datasets, routine sampling programmes or fishery-dependent data collection should be carefully considered. Furthermore, logistical constraints (equipment, budget, manpower etc.) and scientific knowledge should be taken into account, and thus monitoring programmes for MPAs requires thorough planning. In the review, general principles have been developed as to what kind of monitoring strategy is applicable in the three different PROTECT case studies.

- **Modelling of MPA effects:** Two types of approaches are considered in assessing the ecological and fisheries-related impacts of MPAs: a) *mathematical models* depicting the dynamics of populations, communities or ecosystems, which are generally used for policy screening analyses; and b) *empirical approaches* based on *statistical modelling* of field data used to test effects and provide diagnostics on the ecosystem and their resources. Statistical models lead to the definition of empirical indicators and sampling designs for long-term programmes of experimental monitoring. Mathematical models enable the exploration of issues related to MPA design and its consequences on the dynamics of populations and fisheries; they provide reference points against which system dynamics can be gauged.

  Regardless how remarkable models are as tools to evaluate MPA effects at the scale of fisheries and ecosystems, many are mostly theoretical contributions, and simple models published in well-known journals may have resulted in simplistic prescriptions, e.g. about MPA size requirements. Advanced models are needed to e.g. evaluate the spatial dynamics of population and exploitation at the scale of MPA design, including seasonal dimensions if relevant. Models should account for e.g. mixed fisheries (multispecies multifleet fisheries), fishers' responses to MPA establishment, investigations of MPA designs, including permanent vs. temporary MPAs, partial restrictions of fishing activities, and MPA networks. They should also provide for other management measures, as MPAs are usually not the only management tool used in a given fishery. A model, ISIS-Fish, incorporating most of these features is proposed for application to the PROTECT case studies (see also bio-economic modelling below).

- **Socio-economic evaluation:** There have been few significant real-world attempts to analyse the subsequent economic effects on fishing that arise from the implementation of an MPA. There are even fewer examples in Northern European waters. The majority of socio-economic studies are theoretical. Such studies have been receiving significant interest in the last seven years. Bio-economic models are increasingly being developed to measure the impact of MPAs on biological and economic objectives. Typically in this context, key objectives concerning stock status, fleet profitability and related employment can all be evaluated. Further, institutional objectives relating to the implementation of potential MPA management strategies can be evaluated. From studies published that consider socio-economic effects, there is a difference of opinion between authors on the potential of MPAs. This is largely a result of the specific case study under evaluation and the management strategies considered. That is to say, few “transferable” results have been reported. It is often assumed that there are conservation benefits from MPAs, including an assumption of long-run benefits to the fishery through stock recovery or spill over effects. It has been shown using bioeconomic models (mostly conceptual) that this is certainly not a clear result. However, these have seen little quantification and few real-world studies have explicitly considered the impacts of MPAs on fishermen. The three case studies in PROTECT will all be developed as bioeconomic models and used for analysis.

  Economic valuation of a fishery can be determined using market prices, as in the case of bioeconomic models. However, economic valuation of MPAs with aspects of conservation is difficult because there is no direct market price. This is typically termed a non-use or passive-use value. Stated preference techniques and methodologies are used to estimate such non-use values. The contingent valuation method and choice experiments are the two widely used approaches. They differ in the survey format but offer comparable preferences. In this instance choice experiments are the primary method for estimating non-use value for two case studies in PROTECT: the North Sea sandeel; and the deep water coral. Aspects of conservation relating to birds and corals are addressed.

- **Bio-economic modelling of relevance to deep water corals:** Bio-economic modelling of relevance to deep water corals and their management is presented. Several approaches worth pursuing in relation to bio-economic modelling of deep water corals are introduced and discussed.

Concluding remarks

The present review demonstrates a great deal is known regarding the use and development of MPAs. However, it also reveals substantial knowledge gaps and limitations in the current decision base and instruments to select, develop, implement and evaluate MPAs. This is apparent in the review of past and present MPAs, where none of the reviewed North Sea MPAs have had much success in fulfilling their management objectives. In some cases, effects of MPAs were not detectable due to limitations in available data, monitoring strategies and/or clear, pre-defined management objectives.

Therefore, a holistic framework is required for MPA development, including methods to effectively monitor and evaluate the biological and socio-economic effects that may be attributed to MPA establishment.

In addition, there is a need to better integrate fisheries management and ecosystem conservation objectives in MPA development. Even when it comes to the the use of MPAs in relatively well-researched and well-defined geographic areas, much is still unknown regarding the interrelated processes controlling fisheries and the marine environment.

PROTECT is in the process of unveiling some of these unknowns with the view to strengthen the knowledge and decision-base regarding the use and outcomes of MPAs in ecosystem conservation and fisheries management.

MPA Terms and Definitions

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Over the years many terms have been applied to the concept of managing and protecting marine areas, leaving us with a dense jungle of terms to choose from: *Marine park, reserve, nature reserve, habitat reserve, protected area, national seashore, marine wildlife reserve, wilderness area, maritime park, sanctuary, life refuge, conservation area, no-take area, fisheries closure, fish box, closed area, Special Areas of Conservation (SAC's), multiple-use area, national park*, *species-specific harvest refugia, full reserve*, *refugium, gear or behavioural restrictions, seasonal restrictions, etc.*

MPA Definitions

MPAs have come to mean different things to different people, based primarily on the level of protection provided by the MPA. Some see MPAs as sheltered or reserved areas where little, if any, uses or human disturbance should be permitted. Others see them as specially managed areas designed to enhance ocean use (National Marine Protected Areas Center website 2005).

Below, different examples of MPA definitions are listed. For the purpose of PROTECT, and this report, the following definition is used:

An MPA represents any marine area set aside under legislation or other effective means to protect marine values (***marine values*** referring to e.g. conservation, commercial, scientific, recreational, cultural and aesthetic marine values) (modified from Day & Roff 2000 and the Australian Department of the Environment and Heritage 2005).

The above definition is rather broad and potentially covers almost any area-based marine management measure. However, a broad definiton has several advantages.

For instance, the definition is not sector-dependent, i.e. the specific rationale behind the designation of the MPA is not relevant in relation to the terms used. This aspect is becoming increasingly important, partly because many fisheries management measures have benefits to nature conservation and vice versa, and partly because the management of marine areas is moving rapidly towards a more holistic, ecosystem based approach and away from traditional sector by sector approaches.

By stating that MPAs must be set aside under legislation *or any other effective means*, we do not exclude voluntary agreements such as a code of conduct among fishermen in areas that have conservation values, etc.

The definiton requires management measures to be area specific, thereby excluding the use of the MPA term in relation to other management measures and regulations.

Other existing definitions currently used include:

MPA (**Day & Roff 2000**) = *any marine area set aside under legislation to protect marine values, (Marine values referring to conservation, commercial, species enhancement, scientific importance, historic, recreational, aesthetics, cultural, etc.).*

MPA (**Orton 2000**) = *full ecological protection from human exploitive interests, otherwise the term itself becomes debased. Degrees of restriction of the human use of an oceans area could be encompassed, using another term such as Marine Regulated Area, rather than using, and debasing, the term "protected area".*

MPA (Government of **Australia**, Department of the Environment and Heritage 2005) = *an area of land and/or sea especially dedicated to the protection and maintenance of biological diversity and of natural and associated cultural resources, and managed through legal or other effective means.*

MPA (**Government of Canada** 2005) = *an area of sea that forms part of the internal waters of Canada, the territorial sea of Canada or the exclusive economic zone of Canada and has been designated under this section for special protection for one or more of the following reasons (Canada's Oceans Act (Section 35 (1)))*:

a. *the conservation and protection of commercial and non-commercial fishery resources, including marine mammals, and their habitats;*
b. *the conservation and protection of endangered or threatened marine species, and their habitats;*
c. *the conservation and protection of unique habitats;*
d. *the conservation and protection of marine areas of high biodiversity or biological productivity; and*
e. *the conservation and protection of any other marine resource or habitat as is necessary to fulfill the mandate of the Minister (of Fisheries and Oceans Canada).*

MPA (**IUCN**) = *"Any area of the intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part of all of the enclosed environment"*, World Conservation Union. 1988. *Resolution 17.38 of the 17th General Assembly of the IUCN*. Gland, Switzerland and Cambridge, UK. IUCN refers to 7 categories of protected areas with different objectives. The term "MPA" is intended by IUCN to be a general one, describing areas that are subject to various levels of protection.

> All seven categories could be used for fisheries management purposes even though, in most cases, this would not be the primary objective of the MPA (Ward & Hegerl 2003). IUCN is in the process of addressing the problem of applying these MPA categories to fisheries (IUCN 2004).

MPA (**USA**, National Marine Protected Areas Center 2005) = *any area of the marine environment that has been reserved by Federal, State, territorial, tribal or local laws or regulations to provide lasting protection for part or all of the natural and cultural resources therein* (US Marine Protected Areas Executive Order 13158; US Federal Register 2000).

> In 1996 The United States Congress (1996) decided that *"One of the greatest long-term threats to the viability of commercial and recreational fisheries is the continuing loss of marine, estuarine, and other aquatic habitats. Habitat considerations should receive increased attention for the conservation and management of fishery resources of the United States"* (16 U.S.C. 1801 (A)(9)) .

Congress defined the new term “Essential Fish Habitats” as *"those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity"* (16 U.S.C. 1802(10)).

“Habitat Areas of Particular Concern**”** (HAPC) are subsets of Essential Fish Habitats (50 CFR 600.815(a)(8)). US Congress 1996).

A new US MPA classification system simplifies the currently confusing diversity of terms by focusing on a few key features that together describe aspects of the MPA that most influence impacts and concern to stakeholders, agencies and scientists. Outlines are six design characteristics, and options within them (US National Marine Protected Areas Center 2005).

For more information on applications of the MPA concept in regional and international marine policy, please see the *“Review of the legal framework applacble to marine protected areas as a tool for ecosystem conservation & fisheries management”* (below).

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Review of the legal framework applicable to MPAs as a tool for ecosystem conservation & fisheries management

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This chapter reviews the legal basis in a number of international and European legal instruments underpinning the establishment of Marine Protected Areas (MPAs) as a tool for ecosystem conservation and fisheries management. The global instruments examined include: the 1982 United Nations Law of the Sea Convention (LOS); the 1992 United Nations Conference on Environment and Development (UNCED) and Agenda 21; the 1992 Convention on Biological Diversity; the 1995 United Nations Agreement Relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks; the 1995 FAO Code of Conduct for Responsible Fisheries; and the 2002 World Summit on Sustainable Development (WSSD). The regional instruments considered are: the OSPAR and HELCOM Conventions. European primary and secondary legal instruments reviewed include: the EC Treaties; Council Regulation No 2371/2002; Council Directive 92/43/EEC (the Habitats Directive); and Council Regulation 602/2004 to protect deepwater coral sites in an area north-west of Scotland. Recent policy initiatives such as the Biodiversity Action Plan for Fisheries, the European Marine Strategy and the European Maritime Policy are also mentioned. The chapter concludes by outlining the legal options for establishing Marine Protected Areas as a tool for ecosystem conservation and fisheries management.

Introduction

In recent decades, a “species” approach to fisheries management is no longer considered adequate to conserve the living resources of the sea. In parallel with the move away from traditional management measures there has been a shift towards the adoption of new tools such as the ecosystem(s) approach and the establishment of Marine Protected Areas (MPAs) as a means to protect sensitive marine habitats. There are some suggestions that the latter may be traced back to the establishment in 1935 of a conservation area in Fort Jefferson National Monument Park in Florida that extended seawards from a narrow coastal band.[1] Arguably, at a global level the best known “protected areas” are the Great Barrier Reef Marine Park in Australia and the Sabana-Camanguey archipelago off Cuba which are designated as Particularly Sensitive Sea Areas (PSSA) by the IMO because of their ecological and scientific significance.[2] Irrespective of the origin of the “protected area” concept, there is little doubt but that there is now a plurality of legal instruments and international conference documents that call for the establishment of MPAs both within and beyond national jurisdiction as a means to halt the loss of biodiversity and to protect nursery grounds for commercial fish stocks.

The chapter traces the legal basis of MPAs as a means for ecosystem conservation and fisheries management in a number of international, regional, and European legal instruments. At the outset, it ought to be pointed out that the European Community is an international organisation with legal personality and is thus party in its own right to many international instruments and conference documents that require the establishment of a network of MPAs.[3] A similar obligation arises under a

[1] See, Guidelines for the Identification of Particularly Sensitive Sea Areas adopted November 6, 1991 by the IMO under Resolution A.720 (17) as revised by Annex 2 of IMO Assembly Resolution A.927 (22), November 2001.
[2] On the unsuitability of PSSA designation for fisheries management purposes, see paragraph 2, *infra*.
[3] See, for example, the commitment that arises in the following (discussed below): the 1972 World Conference on Human Environment; the 1982 United Nations Law of the Sea Convention (LOS); the 1992 United Nations Conference on Environment and Development (UNCED) and Agenda 21; the 1992

number of European legal instruments.[4] In this context, it is relevant to note that the Council of Fisheries Ministers has called upon the Community to adopt a coherent approach to MPAs as a means to enhance protection of marine biodiversity and to protect, restore or improve habitats for specific species.[5] This is not a new departure in so far as the provision of special "protected status" to a particular area has been previously tested under the common fisheries policy (CFP) in a number of spatial areas colloquially referred to as Western Waters; the Shetlands Box; the Hake Box and the Plaice box. Indeed, it could be argued that measures aimed at reducing access and fishing effort, as well as restrictions on catch and gear to protect juvenile fish species are effectively applications of the same principles which underpin MPAs from a fisheries management perspective.

Both North Sea sandeel and the eastern Baltic Sea cod fishery are depleted and subject to special conservation measures in line with advice presented by the Scientific, Technical and Economic Committee for Fisheries (STECF).[6] Accordingly, any proposal to establish MPAs in relation to these fisheries will be informed by the general management advice presented by ICES and STECF. This chapter concludes, nonetheless, that there is an adequate legal basis in European fishery conservation and environmental instruments for the establishment of a network of MPAs in sea areas under the sovereignty and jurisdiction of the Member States. Moreover, any such initiative to conserve Baltic Sea cod, North Sea sandeel and deepwater coral by this means is entirely consistent with recent measures to integrate environmental principles into the CFP and to adopting an ecosystem-based approach to fisheries management.

Preliminary Matters

(i). Definitions

Marine Protected Areas (MPAs)
(see previous chapter "MPA Terms and Definitions")

Ecosystem Approach

The Convention on Biological Diversity described an "ecosystem" as "an interaction complex of living communities and the environment, functioning as a largely self-sustaining unit".[7] The ecosystem approach is defined as "a strategy for the integrated management of lands, water and living resources that promotes conservation and sustainable use in an equitable way". ICES have published useful guidance on the application of the ecosystem approach to the management of human activities in the European marine environment that informs the work of PROTECT.[8]

(ii). Caveat

Convention on Biological Diversity; the 1995 United Nations Agreement Relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks; the 1995 FAO Code of Conduct for Responsible Fisheries; the 2001 Reykjavik Declaration; and the 2002 World Summit on Sustainable Development (WSSD).

[4] See, Part III, *infra*.

[5] See, para. 16 *infra*.

[6] Commission Regulation (EC) No 1147/2005 of 15 July 2005 prohibiting fishing for sandeel with certain fishing gears in the North Sea and the Skagerrak OJ L 185 , 16/07/2005 19 - 0019

[7] Ibid.

[8] See, Guidance On The Application Of The Ecosystem Approach To The Management Of Human Activities In The European Marine Environment, September 2004.

This chapter only deals with legal and policy instruments concerning species and habitat protection and does not deal with international agreements aimed at protecting the marine environment from vessel source pollution. Consequently, the use of IMO mechanisms to protect a designated area because of its ecological, socio-economic or economic significance such as Particularly Sensitive Sea Areas (PSSAs) is excluded from the scope of this chapter, as this scheme of protection may not be applied to fishing activity.[9]

The second aspect of this chapter that calls for comment is the distinction that may be made between the protection of a spawning stock for Baltic cod or North Sea sandeel and the protection of a physical structure on the seabed such as carbonate mounds associated with accumulation of deepwater-coral. As will be seen, the application of MPAs as a tool for fishery management purposes is not the same as their application for ecological purposes under the Habitats Directive. This is clearly evident from the Biodiversity Action Plan for Fisheries discussed in paragraph 17 below.

(iii). Geographical Scope

This chapter deals with the law as it applies to sea areas under the sovereignty and jurisdiction of Member States. Many of the conservation and management measurements set out in international agreements apply, *mutatis mutandis*, to these areas. Maritime areas under the sovereignty and jurisdiction of Norway are only mentioned briefly as these areas are not subject to European Community law. This aspect will be assessed separately in a later chapter.

(iv). Structure

This chapter is divided into three parts. Part I review a number of policy initiatives that have been taken at an international level calling for the establishment and management of protected areas. Part II reviews the legal regime for the establishment of protected areas in a number of global and regional instruments. Part III identifies the legal basis for the use of MPAs as a tool for ecosystem conservation and fisheries management in a number of European legal instruments. This is followed by a brief assessment of the adequacy of the existing legal regime as a framework for establishing MPAs to protect deepwater coral, Baltic Sea cod and the North Sea sandeel.

[9] See, Guidelines for the Identification of Particularly Sensitive Sea Areas adopted November 6, 1991 by the IMO under Resolution A.720 (17) as revised by Annex 2 of IMO Assembly Resolution A.927 (22), November 2001.

Part 1: "Soft law" Initiatives

1. General

Several policy initiatives, sometimes referred to as "soft law", have been taken by the international community under the aegis of the United Nations to establish MPAs with a view to protecting the marine environment. Although this approach is endorsed by the United Nations Convention on the Law of the Sea (Article 194(5)) itself, many of these initiatives have sought to link the protection of the marine environment with the concept of sustainable development and may be traced back to the 1972 Declaration of the United Nations Conference on the Human Environment.[10] Some of these initiatives are reviewed here. In general, they demonstrate that the adoption of MPAs as a tool for ecosystem management and fisheries management is a contemporary legal issue.[11]

2. Agenda 21, 1992 United Nations Conference on Environment and Development, Rio de Janeiro 1992

Agenda 21, the action programme adopted by United Nations Conference on Environment and Development (UNCED) at Rio de Janeiro in 1992 places particular emphasis on preserving habitats and other ecologically sensitive areas both within and beyond national jurisdiction. More specifically, the programme calls upon "states to identify marine ecosystems exhibiting high levels of biodiversity and productivity and other critical habitats areas providing necessary limitations on use in these areas, through *inter alia*: designation of protected areas".[12]

3. United Nations FAO Code of Conduct for Responsible Fisheries

The European Community is committed to implementing the FAO Code of Conduct for Responsible Fisheries which sets out principles and international standards of behaviour for responsible practices with a view to ensuring the effective conservation, management and development of living aquatic resources with due respect for the ecosystem and biodiversity.[13] Although the Code is voluntary, it places an express obligation on States and users of living aquatic resources to conserve aquatic ecosystems.[14] The right to fish carries with it the obligation to do so in a responsible manner. Moreover, management measures should not only ensure the conservation of target species but also species in associated ecosystems. According to the Code, management decisions for fisheries should be based on the best scientific evidence available, taking into account traditional knowledge of the resources and their habitat, as well as the relevant environmental, economic and social factors.[15] Furthermore, the Code places an express obligation on States and regional fisheries management organisations to apply a precautionary approach to the conservation, management and exploitation of living aquatic resources in order to protect them and to preserve the aquatic environment.[16] In this regard, the absence of adequate scientific information should not be used as a reason for postponing or failing to take measures to conserve target species, associated or dependent species and non-target species and their environment. One other provision in the Code of Conduct for Responsible Fisheries that is particularly pertinent to the establishment of MPAs is the recommendation that all critical

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[10] UN Doc. A/CONF/48/14/REV.1.
[11] On this point, see, T. Scovazzi, 'Marine Protected Areas on the High Seas: Some Legal and Policy Considerations,' *International Journal of Marine and Coastal Law*, Vol. 19, 2004, pp. 1-17
[12] Agenda 21, paragraph 17.68
[13] For further details of the Code and developments in implementation, see, ww.fao.org/fi/agreem/codecond/codecon.asp. A useful introduction to the Code is provided by W. R. Edeson, "The Code of conduct for Responsible Fisheries, An Introduction", (1996) 11(2) *IJMCL*, 233-238
[14] FAO Code of Conduct for Responsible Fisheries, Article 6.1
[15] Article 6.4, *id*.
[16] Article 6.5, *id*.

fisheries habitats in marine ecosystems, such as reefs, nursery and spawning areas, should be protected and rehabilitated as far as possible and where necessary.[17] In this context, fisheries management should be concerned with the whole stock unit over its entire area of distribution and take into account previously agreed management measures established and applied in the same region, all removals and the biological unity and other biological characteristics of the stock.[18] Elsewhere, the Code calls upon parties to develop and apply selective and environmentally safe fishing gear and practices in order to maintain biodiversity.[19] Moreover, in cases where proper selective and environmentally safe fishing gear and practices exist, they should be recognised and accorded a priority in establishing conservation and management measures for fisheries.[20]

4. Declaration of the World Summit for Sustainable Development, Johannesburg 2002

In 2002, a World Summit to review the implementation of UNCED was held in Johannesburg, South Africa. The purpose of the summit was to review the global commitment to sustainable development enunciated in UNCED and by Agenda 21. The summit adopted two documents: the Declaration of the World Summit for Sustainable Development (the *Johaneesburg Declaration*) and the Plan of Implementation. The *Johannesburg Declaration* calls upon states to implement an ecosystem approach to fisheries management by 2010. Moreover, it set down the following objectives for marine resource management:

- the establishment of……network of marine protected areas by 2012;
- restoration of fisheries by 2015;
- drop in rate of species extinction by 2010.

Ocean issues are dealt with in Part IV of the Plan of Implementation which calls upon states and international organisations to "develop …diverse approaches and tools including…. the establishment of *marine protected areas* consistent with international law and based on scientific evidence, …*and time/area closures for the protection of nursery grounds and periods*…" (emphasis added).[21]

5. United Nation's General Assembly Resolution No. A/57/L.48

Since the adoption of the Declaration of the World Summit for Sustainable Development concluded in 2002, the United Nations General Assembly adopted a resolution that deals specifically with the marine environment, marine resources and sustainable development. [22] In particular, this resolution called upon States:

- To cooperate and to take measures to implement Part XII of the Law of the Sea Convention to protect the environment and living resources;
- Endorsed the need for a…network of <u>marine protected areas</u> by 2012;
- Highlighted requirement for international programmes to halt the loss of marine biodiversity;
- Called for the protection of coral reefs;
- Called for urgent ………action to improve the management of …underwater features.

Clearly, the resolution endorses the establishment of a network of MPAs by 2012.

6. United Nations Open-Ended Informal Consultative Process on Oceans and the Law of the Sea (UNICPOLOS)

[17] Article 6.8, *id.*
[18] Article 7.3.1
[19] Article 6.6, *id.*
[20] *Ibid*
[21] WSSD Plan of Implementation, para. 31
[22] UNGA Resolution No. A/57/L.48 of 10th December 2002

In 2000, the United Nations established the Open-Ended Informal Consultative Process (ICP) to assist the General Assembly in their annual review of developments concerning the oceans and the law of the sea. One of the functions of the ICP is to identify areas where there is a requirement for enhanced international co-ordination and co-operation at the inter-agency level. One of the principal issues discussed at the fourth meeting of the United Nations Open-Ended Informal Consultative Process on Oceans and the Law of the Sea (UNICPOLOS) in 2003 were the means available to “protect vulnerable marine ecosystems”. At the meeting, many delegations including the delegation representing the EU expressed support for the establishment of MPAs as a management tool to implement the ecosystem approach both within and beyond national jurisdiction.[23]

7. Bergen Declaration

In March 2002, the Fifth Conference on the Protection of the North Sea adopted the Bergen Declaration aimed at establishing, *inter alia*: an integrated ecosystem approach to the management of human activities affecting the North Sea. At the conference, North Sea Ministers agreed that by 2010 relevant areas of the North Sea will be designated as MPAs belonging to a network of well-managed sites, safeguarding threatened and declining species, habitats and ecosystem functions, as well as areas which best represent the range of ecological and other relevant character in the OSPAR area.[24]

Moreover, the Ministers agreed that fisheries policies and management should move towards the incorporation of ecosystem considerations in a strategic context.[25] While the transition towards a full ecosystem approach to fisheries management should be progressive and concomitant with the enhancement of scientific knowledge, the Ministers expressed the view that the current state of scientific knowledge, coupled with a sound application of the precautionary principle, allows the immediate setting of certain environmental protection measures. In addition, the Ministers requested the competent authorities to identify additional areas to be closed permanently or temporarily to fishing activities for the protection of juvenile fish.[26] Such closures should then be implemented and regularly assessed to ensure that they are effective for stock recovery. The Ministers also endorsed the implementation of environmental measures into the principles, objectives and operational measures underpinning the CFP.[27]

[23] UN, Report of the Work of the United Nations Open-Ended Informal Consultative Process on Oceans and the Law of the Sea, June 2003, paragraph 104
[24] Bergen Declaration, Fifth Conference on the Protection of the North Sea, paragraph 7
[25] Bergen Declaration, Fifth Conference on the Protection of the North Sea, paragraph 19
[26] Bergen Declaration, Fifth Conference on the Protection of the North Sea, paragraph 24
[27] Bergen Declaration, Fifth Conference on the Protection of the North Sea, paragraph 36

Part 2: International and regional treaties

8. General

The international legal regime for marine areas within and beyond the jurisdiction of the Member States is made up of a number of global and regional legal instruments. Some of these are further examined here. The regional instruments are the OSPAR and HELCOM Conventions.

9. 1982 United Nations Convention on the Law of the Sea

The 1982 United Nations Convention on the Law of the Sea provides the jurisdictional framework for the implementation of MPAs in sea areas under the sovereignty and jurisdiction of the Member States. Aside from Estonia, all Member States and the European Union are party to this Convention which sets out the framework for all aspects of ocean use such as navigation, environmental protection, marine scientific research, economic activities, marine resource use, capacity building and dispute settlement. Significantly, under the Convention, States have a duty to protect and preserve the marine environment and to exploit natural resources in accordance with this duty.[28] These obligations apply to all sea areas under the sovereignty and jurisdiction of the Member States. Measures taken by parties to the Convention *must* include those necessary to protect and preserve rare and fragile ecosystems as well as the habitat of depleted, threatened or endangered species and other forms of marine life.[29] The Convention also requires States to take into account the best available scientific information to ensure the proper management and conservation of marine living resources.[30]

The scope for taking measures to protect Baltic Sea cod, North Sea sandeel and deepwater coral will be tailored by the maritime jurisdictional zones codified by UNCLOS and implemented by the CFP is shown on Figure 1 below. The maritime jurisdiction zones include:

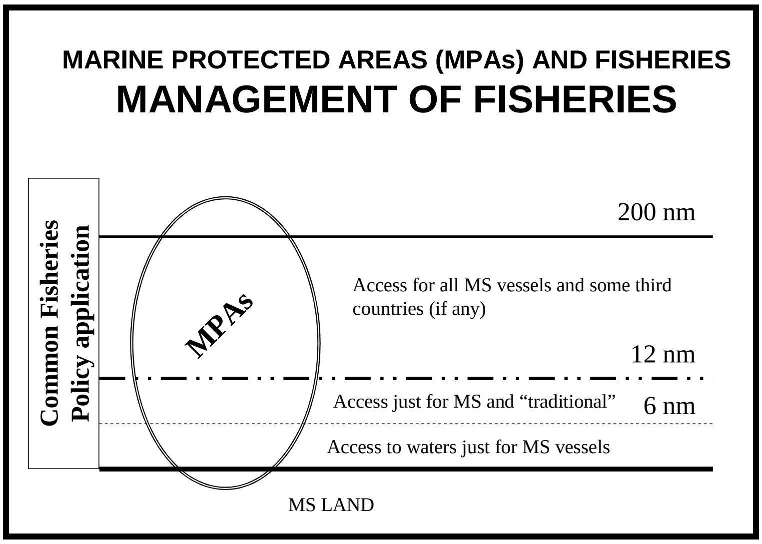
- *Internal waters*: coastal states enjoy full sovereignty.
- *Territorial sea* (out to a maximum of 12 nautical miles from the baseline/low tide mark): coastal states exercise full sovereignty under international law subject to the right of vessels to exercise their right of innocent passage. There is limited scope under the CFP for the adoption of nation measures (discussed in Part III below).
- *Exclusive Economic Zone* (EEZ) (maximum 200 nautical miles from the baseline) coastal states have sovereign rights over natural resources and certain economic activities, and jurisdiction over environmental protection, subject to the rights of other states to freedom of navigation, overflight, laying of submarine cables and pipelines. Fisheries measures are implemented by means of the CFP.
- *Continental shelf* (may extend beyond 200 nautical miles) coastal states have sovereign rights for exploring or exploitation of natural resources of the seabed and subsoil. The CFP has no application to the non-living natural resources of the continental shelf such as carbonated mounds or non-living coral structures.
- *High seas* (areas beyond national jurisdiction) all states enjoy traditional high seas freedoms, subject to other international agreements and duties to protect marine environment and conserve living marine resources. The CFP applies to the activities of fishing vessels flying the flag of a Member State.

In summary, although the European Community has competence to adopt conservation measures for living aquatic resources, the implementation of specific

[28] 1982 UN LOS Convention, Article 192, 193
[29] 1982 UN LOS Convention, Article 194 (5)
[30] 1982 UN LOS Convention, Article 161

measures under the CFP must adhere to the normative jurisdictional framework set out by the 1982 United Nations Convention on the Law of the Sea.

Figure 1: Application of MPAs through the medium of the CFP and international law.



10.Convention on Biological Diversity

The Convention on Biological Diversity (CBD) adopted in 1992 aims to conserve biodiversity. CBD calls upon States Parties to establish national conservation strategies and to establish a system of *protected areas* or areas where special measures need to be taken to conserve biological diversity.[31] This obligation applies to both marine and terrestrial areas. Parties are expected to regulate activities under their jurisdiction that may have a significant adverse effect on biodiversity. In recognition of the special conservation requirements of the marine environment, in 1995 the second Conference of Parties to the CBD adopted the Jakarta Mandate. The Jakarta Mandate lays out a strategy for protection of coastal and marine biological diversity, including the establishment of representative systems of marine and coastal protected areas, within the context of national programmes for integrated coastal area management. At the seventh meeting of the Conference of the Parties to the CBD in Kuala Lumpur, Malaysia, in February 2004, Contracting parties agreed to achieve the "establishment and maintenance by 2012 for marine areas, of comprehensive, effectively managed and ecologically representative national and regional systems of protected areas." The seventh meeting of the Conference of the Parties also established the Ad Hoc Open-ended Working Group on Protected Areas with a mandate to support this objective. A meeting of this working group took place in Montecatini, Italy, 13-17 June 2005.[32]

The European Community has taken a strong position regarding the halting of biodiversity loss, ensuring the conservational and sustainable use of marine biodiversity, as well as the creation of a global network of marine protected areas by 2012. The implementation of the proposed Directive Establishing a Framework for Community Action in the field of Marine Environmental Policy (discussed below in Part III) will contribute to the achievement of the objectives agreed at the Kuala Lumpur meeting including the establishment and maintenance of ecologically representative national and regional systems of marine protected areas by 2012.

[31] Convention on Biological Diversity, Article 5
[32] See, UNEP/CBD/WG-PA/1/2, Ad Hoc Open-Ended Working, Group On Protected Areas, Montecatini, Italy, 13-17 June 2005.

11. United Nations Agreement for the implementation of the provisions of UNCLOS Of 10 December relating to the Conservation And Management of Straddling Fish Stocks and Highly Migratory Fish Stocks (Straddling Fish Stocks Agreement)

The European Union and the Member States are party to the Straddling Fish Stocks Agreement. Although this Agreement is not applicable to Batic Sea cod or North Sea sandeel, it is relevant to adoption of conservation measures for the deepwater coral sites that straddle the Irish exclusive fishery zone and the outer continental shelf. Significantly, this Agreement aims to protect biodiversity and to reduce fishing impacts on associated and dependent species.[33] Moreover, the agreement endorses the adoption of conservation measures and calls for the application of the precautionary principle.[34] Any measures to establish MPAs to protect deepwater coral sites will have to be consistent with this Agreement and with measures adopted by regional fisheries management organisations such as NEAFC.

12.OSPAR Convention

Another Convention influencing the development of EC law to protect the marine environment is the Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR Convention).[35] Among the objectives of this Convention is the provision of a legal framework for concerted action at all levels to manage human activities in such a manner that the marine ecosystem will continue to sustain the legitimate uses of the sea and meet the needs of present and future generations.[36] While the OSPAR Convention is ostensibly focused on marine pollution it contains important provisions in Annex V aimed at the protection and conservation of the ecosystems and biological diversity of the maritime area. Both Article 4 of Annex V and the penultimate recital of the Convention stipulate that measures pertaining to the management of fisheries shall not be adopted under the Convention but shall be referred to the attention of the authority or international body competent for such issues. Thus, questions pertaining to the management of the North Sea sandeel fishery and the deepwater coral sites that impinge upon the activities of fishing vessels flying the flag of Member States of the EC must be taken under the instruments constituting the CFP.

There are a number of recent developments within the OSPAR framework that are relevant to PROTECT. At a meeting in Sintra in Portugal in 1998, the European Commission and the members of the OSPAR Commission bound themselves to implement a strategy on the protection and conservation of the ecosystems and biological diversity of the maritime area and in so doing promote the establishment of a network of MPAs. Subsequently, the Environmental Ministers of OSPAR and HELCOM Contracting Parties expressed their support for implementing the Declaration of the World Summit for Sustainable Development at their meeting in Bremen in June 2003. Contracting Parties also entered into a commitment to

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[33] United Nations Agreement for the implementation of the provisions of UNCLOS Of 10 December relating to the Conservation And Management of Straddling Fish Stocks and Highly Migratory Fish Stocks, Article 5

[34] United Nations Agreement for the implementation of the provisions of UNCLOS Of 10 December relating to the Conservation And Management of Straddling Fish Stocks and Highly Migratory Fish Stocks, Article 6

[35] The OSPAR Convention came into force in 1998. OSPAR refers to Oslo and Paris, the cities in which previous conventions to the 1992 Convention were adopted. The Convention maritime areas are those parts of the Atlantic and Arctic Oceans and their dependent seas as defined in Article 1 of the Convention. Within that particular area the Convention applies to the internal waters and the territorial seas of the Contracting Parties, the sea beyond and adjacent to the territorial sea under the jurisdiction of the coastal state to the extent recognised by international law, and the high seas, including the sea of all those waters and its sub-soil.

[36] Third recital of the Convention.

establish a network of MPAs and to ensure that by 2010 it is an ecologically coherent network of well managed marine protected areas which will:

(a). Protect, conserve and restore species, habitats and ecological processes which have been adversely affected by human activities;
(b). Prevent degradation of, and damage to, species habitats and ecological processes, following the precautionary principle;
(c). Protect and conserve areas that best represent the range of species, habitats and ecological processes in the maritime area.

The OSPAR network of MPAs is complementary to the NATURA 2000 network (discussed below) and will be completed by 2005. While the OSPAR Commission has neither competence to adopt and implement management measures for fisheries it can, nonetheless, bring issues related to the objectives of the Convention to the attention of the European Commission and to other relevant fisheries management bodies. Deepwater coral reefs are included in a list of endangered species under the OSPAR Convention. Significantly, a number mounds (Logathchev, Western Porcupine Bank Mounds, Hovland Mounds, and Belgica Mounds) associated wither deepwater coral species in sea-areas are also under consideration for designation as joint NATURA 2000/OSPAR MPAs. Similarly, one coral reef in nominated in the Norwegian EEZ and further designations are expected in 2007 when the first phase of Norway's national plan for MPAs is implemented.

13.HELCOM Convention

The Convention on the Protection of the Marine Environment of the Baltic Sea Area (HELCOM Convention) aims to protect the marine environment of the Baltic Sea from all sources of pollution and to restore and safeguard its ecological balance through intergovernmental co-operation between Denmark, Estonia, the European Community, Finland, Germany, Latvia, Lithuania, Poland, Russia and Sweden. In 1994, 62 Baltic Sea Protected Areas (BSPAs) were designated under HELCOM Recommendation 15/5. The Joint OSPAR-HELCOM Ministerial Meeting held in Bremen in 2003 adopted a joint work programme on MPAs in the North-East Atlantic and Baltic. At the time of writing, the network of marine and coastal Baltic Sea Protected Areas is not fully implemented and many Contracting Parties have not designated the boundaries of Baltic Sea Protected Areas (BSPAs) or prepare management plans, and few concrete steps have been taken to include the 24 proposed offshore BSPAs in the network. With the exception of Russia, all HELCOM Contracting Parties are Members States of the European Union and any measures to implement MPAs to protect Baltic Sea cod will have to be implemented by means of the CFP. Significantly, a recent communication published by the European Commission notes that: "The marine ecology of the Baltic region is estimated to have "*crashed*" and to be "*locked in*" to permanent eutrophication.[37]

[37]Commission Staff Working Document, Impact Assessment - Thematic Strategy on the Protection and Conservation of the Marine Environment, SEC (2005), (copy with the author)

Part 3: European law and policy

14. General

The European legal regime for establishing MPAs in seas under the jurisdiction and sovereignty of the Member States may be traced back to the EC treaties and to a number of instruments adopted under the CFP and the environmental policy. In addition, MPAs are relevant to the implementation of a number of policy initiatives such as: the Biodiversity Action Plan for Fisheries; the European Marine Strategy; and the European Maritime Policy.

The European Union has almost exclusive competence to regulate fisheries through the medium of European community law. Moreover, closed areas or restricted access zones have been used as a tool for fishery management since the early 1980s. Seasonal bans on fishing, for example, have been used to protect Western mackerel, cod in the German Bight and North Sea herring and sprat. Article 4 of Regulation (EEC) No 3760/92 (since repealed) provided a legal basis for various management tools to protect marine biodiversity including the establishment of zones in which fishing is prohibited or restricted, closed areas or no-take zones. In addition, many measures were adopted as technical conservation measures for the protection of juvenile marine organisms.[38] More recently, there is a trend in European law towards the adoption of measures that minimise the impact of fishing activities on marine ecosystems. This trend is traced here.

15.EC Treaties

The EC is committed to the sustainable development of economic activities and a high level of protection and improvement of the quality of the environment.[39] The EC Treaties also oblige the Community to adopt a common policy for fisheries.[40] While there is no express legal basis in the treaties which obliges the European Commission to establish MPAs as a tool for ecosystem conservation and fisheries management, the European Union is committed to the preservation, protection and improvement of the quality of the marine environment. This commitment has a solid legal basis in the EC Treaty, which states:

> “Environmental protection requirements must be integrated into the definition and implementation of the Community policies and activities.....in particular with a view to promoting sustainable development”[41]

This EC Treaty obligation to integrate environmental considerations into the elaboration and implementation of Community policies is based on the conceptual premise that environmental policy requires specific measures in sectoral policies such as fisheries in order to achieve the global objectives of environmental protection and sustainable development. Elsewhere, the EC Treaty states in the substantive provisions dealing with the environment that the Community policy on the environment shall contribute to prudent and rational use of natural resources.[42] Furthermore, that:

> “Community policies on the environment shall aim at a high level of protection taking into account the diversity of the situation in the various regions of the Community. It shall be based on the precautionary principle

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[38] Council Regulation (EC) No. 850/98 of 30 March 1998, OJ L 125, 27.4. 1998 as amended.
[39] EC Treaty, Article 2
[40] EC Treaty, Article 3
[41] EC Treaty, Article 6
[42] EC Treaty, Article 174(2)

> and on the principles that preventative action should be taken, that environmental damage should as a priority be rectified at source…".[43]

Although the principles enunciated in this EC Treaty provision are generally considered to lack legal clarity and are seen as statements of political intent,[44] they do offer useful guidance which suggests that any measures or tools such as MPAs which are aimed at protecting the marine environment are fully consistent with European law and act as an embodiment of the precautionary and preventative action principles.

16. Integrations of Environmental Considerations into the CFP

The CFP has an environmental dimension since its inception in 1982. Indeed, measures to reduce bycatch of cetaceans and to implement the UN General Assembly Resolution on a moratorium on driftnet fishing, as well the restrictions on North Sea sandeel fisheries to safeguard seabirds, are clearly aimed at minimising the impact of fishing activity on the marine environment.

A major step to integrate environmental concerns into EU policies was taken at the Cardiff summit meeting in 1998. A description of the measures adopted under the CFP is provided in the Communication "Fisheries management and Nature Conservation in the marine environment".[45]

The Green Paper on the Future of CFP published in 2001 notes, nonetheless, *inter alia*, that:

- The CFP should do more to integrate the environmental dimension into policy making in a proactive manner.

- There is a lack of knowledge about the functioning of marine ecosystems and the side effects of fishing that exacerbate the environmental shortcomings of the CFP.[46]

The European Commission subsequently published a Communication on a strategy for the integration of environmental protection requirements into CFP which recommends: the adoption of an ecosystem approach to fishery management; the embracing of the environmental principles in the EC Treaties (discussed above), and: the implementation of the Biodiversity Action Plan for Fisheries.[47]

17. Biodiversity Action Plan for Fisheries

In 2001, the European Commission published a Communication to the Council and the European Parliament: Biodiversity Action Plan for Fisheries.[48] The Communication is part of the European Community's Biodivesity Strategy and fulfils the European Community's legal obligations under Article 6 of the CBD. The Strategy defines a two-step process. The first step is the elaboration of general policy orientation. The second entails the development and implementation of Action Plans and other measures. One of the priorities of the Action Plan for Fisheries is to reduce the impact of fisheries activities on non-target species and on

[43] EC Treaty, Article 174(2)
[44] See L. Kramer, *EC Environmental Law*, (London, Sweet & Maxwell, 4th ed., 2000), pp. 9-20
[45] COM (1999) 363
[46] COM (2001) 135 final, Brussels, 20.3.2001
[47] Communication from the Commission setting out a Community Action Plan to integrate environmental protection requirements into the Common Fisheries Policy, OM/2002/0186
[48] COM/2001/0162 final

marine and coastal ecosystems. The Action Plan for Fisheries notes that most of the concerns about fisheries impacting upon biological diversity have centred on the effect of over-fishing and the physical impact of fishing gear on habitat. One of the overall objectives of the Action Plan for Fisheries is to define and identify, within the current legislative framework, coherent measures that lead to the preservation or rehabilitation of biodiversity where it is perceived as being under threat due to fishing.

In the context of PROTECT, it is significant to note that paragraphs 45-52 of the Action Plan for Fisheries states the following:

> “45. Closed areas or "no-take zones" have been used for a long time within fisheries management both within the EU and elsewhere. It is important to recognise what is the intended purpose of such closures, as they will differ depending on whether the closure is for traditional fisheries management purposes or for ecological purposes. Within the fisheries management ambit, closures are used primarily for the following purposes:
>
> - in emergency situations, to prevent high fishing mortalities being exerted when fish are highly vulnerable because of forming dense aggregations.
> - to enhance protection of juvenile fish when gear selection do not provide enough protection.
> - only means to protect local spawning from depletion or extinction
>
> 46. In such situations it is believed that closures are effective although the relevant scientific evidence is only weakly supportive. This would also apply to non-target or by-catch species.
>
> 47. The experience gained with closures is that the effects are very difficult to evaluate and "no-take zones" are not a panacea to all fisheries management and ecological problems. Closures are less effective in reducing the overall fishing pressure than effort reductions because the effect can be to redistribute fishing effort to areas or time periods that are still open. To overcome such effects the closed areas have to cover a very large portion of the distribution of stocks they are intended to protect, which calls into question whether the use of other management tools (lower TAC, improved selection etc.) or combinations thereof would be more effective and less discriminatory towards those fishermen close to the closed area.
>
> 48. There is less experience with closures applied for ecological purpose in the marine environment although several closures have been in place for many years. Some of these were intended to protect single stocks, but there have also been extended closures in place around some marine installations, such as oil and gas, where fishing is prohibited.
>
> 49. It is important to note that compared to terrestrial organisms, marine organisms are relatively more mobile and closures might therefore be more appropriate in regards of protection of sensitive or representative habitats such as coral reefs and important feeding areas for seabirds during breeding seasons.
>
> 50. It is however generally perceived that if closed areas are well defined, they can be a useful additional tool to enhance protection of stocks and of sensitive habitats. *The plan therefore proposes use of closed areas for the protection of fish and habitats but it will be necessary to define clearly the objectives and to justify the biological basis for any such closure*s. Equally important is to promote research to assess and monitor the effects and pilot

studies therefore need to be initiated as an integrated part of this action. (emphasis added).

51. It is widely perceived that the high exploitation pressure on commercially important fish stocks has more widespread effects, leading to diminished food webs of decreased complexity and, generally speaking, less "biodiverse" ecosystems. Marine habitats are also affected. Although the reversibility of these effects may be questioned in cases of large alterations from the "pristine" situation, it is generally believed that a decrease in fishing pressure on commercially important fish stocks would contribute in the mid-term to increase the overall biodiversity of the marine ecosystems.

52. In some cases, however, the effect of fishing operations on the environment may be considered as positive effects on some populations or resulting in increased productivity. For example, high rates of discarding fish in some areas has led to increases in populations of scavenging seabird species. The reduction in abundance of dominant predatory fish by fishing may allow an increase in abundance of prey fish species. Additionally mild physical disturbance can enhance biodiversity and ecosystem productivity. These effects may be considered positive as long as fishing has not been so severe that the populations lose their ability to recover. It should be borne in mind, therefore, that the effects of changes in fishing practices and distribution should be considered fully, without prejudging the positive or negative implications."

In conclusion, the Action Plan for Fisheries points out that:

- Temporal or spatial closures to enhance survival of juveniles or spawning concentration, including sub-populations or to enhance survival of local populations in order to maintain genetic diversity are considered technical measures with the objective of improving the conservation and sustainable use of commercially exploited fish stocks.

- Temporal or spatial closures to enhance protection of species or habitats, including "no-take" zones are considered technical measures with the objective of reducing the impact on non-target species and habitat.

18. Council Regulation No 2371/2002 (the Basic Fishery Management Regulation)

The Council of Fisheries Ministers agreed a new management regulation for fisheries in December 2002.[49] Council Regulation No 2371/2002 (the Basic Fishery Management Regulation) is a framework regulation and comprises of 36 articles and is supported by a large corpus of implementation regulations that prescribes more detailed rules for various aspects of the policy. The geographical scope of regulation extends to all Community waters and covers the activities of fishing vessels flying the flag of a Member State of the European Union. With respect to adopting MPAs as a tool for ecosystem conservation and fishery management, the Basic Fishery Management Regulation provides a legal basis for the adoption of measures concerning: conservation, management and exploitation of living aquatic resources; limitation of the environmental impact of fishing; and conditions of access to waters and resources.[50] Importantly, the Basic Fishery Management Regulation embraces a number of environmental principles such as the adoption of both a precautionary approach to the protection of the environment and an ecosystems approach to

[49] Council Regulation (EC) No 2371/2002 of 20 December 2002 on the conservation and sustainable exploitation of fisheries resources under the CFP, OJ L 358/59 of 31.12.2002.
[50] *Ibid*. Article 1

fisheries management.[51] Moreover, the regulation states that the policy is guided by the principles of good governance which require a clear definition of responsibilities at the Community, national and local levels; a decision-making process based on sound scientific advice which delivers timely results; broad involvement of stakeholders at all stages of the policy from conception to implementation; and a requirement that the policy is consistent with other Community policies, in particular with environmental, social, regional, development, health and consumer protection policies. [52]

One of the features of the Basic Fishery Management is the move towards a broader and more flexible regime for the management for European fisheries. In particular the Basic Management Regulation places considerable emphasis on a long-term management approach, the adoption of recovery plans for fishery stocks that are in crisis, and in a major departure from the previous regime, allows for the adoption of recovery plans by the European Commission and the Member States in cases where there is evidence of a serious threat to the conservation of living aquatic resources or to the marine ecosystem as a result of fishing activities.[53] The latter power has been utilized to protect deepwater coral reefs near the Darwin Mounds in the United Kingdom in August 2003.[54] This allowed time for the European Commission to prepare a regulation for the permanent protection of the reefs (discussed below). Similarly, the revised regulation provides a clear legal basis for the adoption of recovery plans to allow specific fish stocks to recover from over exploitation such as cod in the North Sea and Irish Sea. In line with the policy during the period 1982-2002, Member States retain the power under the Basic Fisheries Management Regulation to adopt measures in waters up to 12 nautical miles applicable to all fishing vessels provided that such measures are non-discriminatory and prior consultation with the European Commission, other Member States and the Regional Fishery Advisory Councils has taken place.[55] Moreover, the European Community must not have taken measures specifically addressing conservation.

19.Technical Conservation Measures

Council Regulation No 850/98 (referred to as the Technical Conservation Regulation) as amended prescribes the technical measures for the conservation of fishery resources and for the protection of juveniles of marine organisms.[56] This regulation is amended from time to time on the basis of scientific evidence to minimise the impact of fishing activities on marine ecosystems and to protect spawning stocks. There is also a special technical conservation regulation for the Baltic Sea: Council Regulation (EC) No 88/98 of 18 December 1997 which lays down certain technical measures for the conservation of fishery resources in waters of the Baltic Sea, the Belts and the Sound.[57] This regulation is also subject to amendment. Both Technical Conservation Regulations offer the best legal means to implement an MPA as a tool for ecosystems conservation and fisheries management.

20. Council Regulation No 602/2004 protecting deepwater coral reefs from the effects of trawling in an area north west of Scotland

The Technical Conservation Regulation (No. 850/98) was amended in 2004 to prohibit the use of bottom trawls or any similar towed nets operating in contact with

[51] *Id.*, Article 2(1)
[52] *Id.*, Article 2(2)
[53] *Id.* Articles 7 and 8
[54] Commission Regulation (EC) 1475/2003 of 20 August on the protection of deepwater coral reefs from the effects of trawling in an area north west of Scotland, OJ L 211/14, 21, 08, 2003
[55] Council Regulation (EC) No 2371/2002, Article 9
[56] Council Regulation (EC) No 850/98 of 30 March 1998 for the conservation of fishery resources through technical measures for the protection of juveniles of marine organisms, OJ L 125, 27.4.1998, p.1
[57] OJ L 009 , 15.01.1998, p. 0001-0016

the bottom of the sea in an area adjacent to the Darwin Mounds.[58] This area is within the 200 nautical mile fishery limits of the United Kingdom. According to ICES Reports aggregations of deepwater corals *Lophilia pertusa* have been mapped in this area and although they appear to be in good conservation status they appeared to show damage from bottom trawl operations. Moreover, scientific reports prove that these aggregations constitute habitats that host important and diverse biological communities. The prohibition on the use of bottom trawls and similar gear in the area surrounding the Darwin Mounds is justified on the grounds that reef recovery from trawl damage is either impossible or very difficult and slow.[59]

21. Council Directive 92/43/EEC (the Habitats Directive)

Although not applicable to Baltic Sea cod and North Sea sandeel, one legal instrument which is relevant to the protection of deepwater coral is the Habitats Directive.[60] This instrument aims to maintain biodiversity and contribute to the general objective of sustainable development by preserving and restoring natural habitats as well as wild fauna and flora.[61] Under the Directive, Member States are obliged to establish a comprehensive network of Special Areas of Conservation (SACs) for endangered and vulnerable species and habitats. The nature network established by the Habitats Directive in conjunction with the Birds Directive is known as NATURA 2000 and consists of sites of international importance. SACs are generally designated by Member States but there is also provision for EC designation in exceptional circumstances where a site hosts a priority natural habitat type or priority species. The Annexes of the Directive list the broad categories of natural habitat types and the specific animal and plant species of Community interest. The Habitats Directive is applied to sea areas under the sovereignty and jurisdiction of the Member States as is evident from the Communication from the Commission (*Fisheries Management and Nature Conservation in the Marine Environment)*.[62] This interpretation is supported by the decision of the High Court in the United Kingdom which concluded that the geographical scope of application of the Habitats Directive is not limited to the territorial sea but “applies to the United Kingdom’s continental shelf and to the superjacent waters up to a limit of 200 nautical miles from the baselines from which the territorial sea is measured”.[63] The decision of the High Court in the United Kingdom is consistent with the findings of the European Court of Justice in several fisheries cases that have held that the scope of Community law extends as far as the rule making authority remit of Member States under public international law.[64]

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[58] Council Regulation (EC) No 602/2004 of 22 March 2004 amending Regulation (EC) No 850/98 as the regards the protection of deepwater coral reefs from the effects of trawling in an area north west of Scotland, OJ L 097, 1.4.2004, p. 0030-0031

[59] Recital 6, Preamble, Council Regulation (EC) No 602/2004, OJ L 097, 1.4.2004, p. 0030-0031

[60] Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora, Official Journal L 206 , 22/07/1992 p. 0007 – 0050; as last amended by Council Directive 97/62/EC of 27 October 1997, Official Journal L 305 , 08/11/1997 pp. 0042 – 0065. For a discussion of the application of the directive to deepwater coral, see, R. Long, A. Grehan, "Marine Habitat protection in a coastal Member State of the European Union: the case of deep-water coral conservation in Ireland", *International Journal Marine and Coastal Law*, 2002, Vol 17, No. 2, pp. 241-269

[61] Council Directive 92/43/EEC, Article 2

[62] COM(1999), 363 final, Brussels 14.07.1999. The European Commission states that:

> “The provisions of the Habitats Directive automatically apply to the marine habitats and marine species located in territorial waters (maximum 12 miles). However, if a Member State exerts its sovereign rights in an exclusive economic zone of 200 nautical miles (for example, the granting of an operating licence for a drilling platform), it thereby considers itself competent to enforce national laws in that area, and consequently the Commission considers in this case that the “Habitats Directive” also applies, in that Community legislation is an integral part of national legislation”.

[63] *The Queen* v. *The Secretary of State for Trade and Industry ex parte Greenpeace Limited*, High Court of Justice Queen’s Bench Division, 5th November 1999.

[64] Joined Cases 3,4 and 6/76, *Kramer* [1976] ECR 1279, paragraphs 11-14; Case 61/77, *Commission* v. *Ireland* [1978] ECR 417.

There is also general consensus that *Lophilia pertusa* is a reef-forming coral and comes within the definition of “reefs” in the *Interpretation Manual of European Union Habitats* published by the European Commission. Both Ireland and the United Kingdom are in the process of designating deepwater coral sites and have taken steps to apply the Habitats Directive in the sea areas under their sovereignty and jurisdiction. According to the EU Habitats Directive, management of MPAs should aim at assuring that the activities taking place inside these areas do not imply unacceptable levels of disturbance or deterioration of the ecological features present at the protected marine sites. In this context, as noted by the Council of European Environment Ministers: “The Habitats and Birds Directives, and specially the associated network of protected sites in the marine environment "NATURA 2000", constitute a key element for the protection of the marine ecosystem which may have consequences on fisheries. Member States are encouraged to continue their work towards the full implementation of these directives in their exclusive economic zones.” In this context, measures to protect deepwater coral sites from fishing activity will have to be taken through the medium of the CFP. The application of MPAs as a tool for ecosystem conservation & fisheries management is illustrated in Figure 2 below.

LEGAL BASIS FOR MPAs

FISHERIES

Technical conservation measures to ensure the protection of marine biological resources and the balanced exploitation of fishery resources. This may include the limitation of fishing within certain areas and periods and with certain gears.

NATURE PROTECTION

Under Habitats and Birds Directives, EU Member States have to propose the protection of areas containing species or habitats of community importance.

Suitable for Deepwater Coral

Fishing measures for the management of MPAs necessitates their adoption within the framework of the CFP, allowing them to be applied by or enforced against any fishing vessel in sea areas under the sovereignty and jurisdiction of the Member States

Figure 2 Application of MPAs as a tool for Ecosystem Conservation & Fisheries Management

22. European Marine Strategy

The European 6th Environment Action Programme included a commitment to develop a Thematic Strategy for the protection and conservation of the marine environment with the overall aim *“to promote sustainable use of the seas and conserve marine ecosystems”*. While the Strategy is primarily focused on the protection of the regional seas bordered by EU countries, it also takes into account the international dimension in recognition of the importance of reducing the EU’s footprint in marine areas in other parts of the world, including the High Seas. Europe’s marine biodiversity is decreasing and continues to be altered. Marine habitats are being destroyed, degraded and disturbed by a range of human impacts including fishing activity.[65] Against this background, the objective of the Strategy is to protect Europe's oceans and seas and ensure that human activities are carried out in a sustainable manner so that current and future generations enjoy and benefit from biologically diverse and dynamic oceans and seas that are safe, clean and productive. Significantly, the strategy advocates an ecosystem-based approach and the European Commission have recently brought forward a draft Framework Directive for the protection of the marine environment. The Draft Directive envisages the establishment of European Marine Regions and the implementation of strategies at a regional level. Where the European Union has legal competence, action to implement the strategies will be implemented through the medium of European Community law such as the CFP. Significantly, the Habitats and Birds Directives will be used to protect and conserve marine biodiversity. In particular, the Strategy will foster efforts to set up EU marine protected areas through the NATURA 2000 network. In addition, the ecosystem-based approach is fully in line with the EU's biodiversity policy and will contribute to the EU's objective to halt the loss of biodiversity in Europe by 2010. Implementing the Strategy will enable the EU to fulfil obligations contracted under relevant international agreements and will improve the EU's contribution to globally agreed goals and targets. The Strategy will be reviewed in 2010 and feed into the final evaluation of the 6th Environmental Action Programme.

23.European Maritime Policy

The European Commission published a Communication on the future EU Maritime Policy on the 2nd March 2005.[66] The Marine Strategy will deliver the environmental pillar of the Maritime Policy that will be elaborated by the European Commission in a Green Paper in 2006.

[65] See, OSPAR Commission, "QSR2000" (published in 2000).
[66] Communication of 2 March 2005, entitled "*Towards a future Maritime Policy for the Union: a European vision for oceans and seas*".

Part 4: Tentative conclusions

Although it may be early to assess the adequacy of the existing legal regime for establishing MPAs in sea areas under the sovereignty and jurisdiction of the Member States, a number of tentative conclusions are evident and may be further refined in light of the findings of the scientific studies. These are as follows:

1. There are a broad range of international and European legal instruments and policy documents that recommend the adoption of MPAs as a tool for ecosystem conservation and fishery management.

2. There is sufficient legal basis within the CFP and European environmental policy to implement an MPA driven approach.

3. Any measures to implement MPAs for fishery management purposes must be subject to scientific advice and assessment by ICES and STECF.

4. In the Atlantic and North Sea, specific measures may be adopted by amending Council Regulation (EC) No 850/98 of 30 March 1998 for the conservation of fishery resources through technical measures for the protection of juveniles of marine organisms.

5. In the case of the Baltic Sea, specific measures could be adopted by amending Council Regulation (EC) No 88/98 of 18 December 1997 which lays down certain technical measures for the conservation of fishery resources in waters of the Baltic Sea, the Belts and the Sound.

6. Protection of deepwater coral requires designation as a Special Area of Conservation under the Habitats Directive and by means of a technical conservation measure under the CFP.

From a legal perspective, MPAs may only be used as a tool for ecosystem conservation and fisheries management if they are: proportionate; based on scientific evidence; enforceable; specific for each marine area and objective; consistent with the ecosystems approach, and; conform to European and international law.

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Present and past MPAs used in fisheries management in the Northeast Atlantic

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DIFRES

The following chapter is a presentation of several past and existing Northern European MPAs (and one North American MPA) based on scientific papers, reports etc. Emphasis has been placed on well-established MPAs that have been established as fisheries management tools – the so-called fish boxes or fisheries closures. In order to provide a simple overview, the different aspects of the respective MPAs are described in sub-categories. The chapter begins with a review of the general literature regarding MPAs as fisheries management tools in temperate seas.

Fisheries MPAs in general

Introduction

Harvest over-capacity has through the years been combined with habitat damage, inappropriate fishing techniques, lack of enforcement, lacking management adaptivity, technological developments, and allocation issues (Murawski et al. 2000). A comparison of Catch per Unit Effort in the periods 1906-1909 and 1990-1995 shows general, large reductions in stock densities of 18 of 19 examined species (Rijnsdorp et al. 1996). Regulation of gears, effort and catch has so far not succeeded in stopping overfishing of North Sea stocks (FSBI 2001). Meanwhile, the seabed, the benthic habitats and e.g. deep-water coral reefs continue to be disturbed or destroyed by the use of mobile fishing gear, which has been compared with the clear-cutting of forests (Watling & Norse 1998).

Increasingly, humanity is coming to the realisation that no single stock, meta-population or even ecosystem can be considered in isolation (Russ & Zeller 2003). Focus is moving to action directed towards reshaping the relations between human beings and the environment and establishing a system of governance that is able to implement policies that move towards sustainability on international, regional, national and local scales (Crean & Wisher 2000). In the past few years, marine protected areas (MPAs) have been proposed to have an ever-increasing role to play in such policies as an overarching tool for the conservation of ecosystems and the management of fisheries in Northern European seas.

MPA effects

In addition to recovering stocks of target species, other potential key fishery management benefits claimed for MPAs include the development of natural age structures of exploited species, protection of genetic variability, restoration of ecosystem integrity, more predictable and higher catches and insurance against management failure (Bohnsack 1996). MPAs/fisheries closures allow the average size of fish to recover from size selective fishing (FSBI 2001), and greater fish size leads to greater fecundity (Bohnsack 1990). There is better fertilisation efficiency at higher fish densities and improvements in spawning habitats (Dugan & Davis 1993). In addition, MPAs provide a buffer against unpredictable events (FSBI 2001) and may bring marine areas back to a “natural” state. Recovery within an MPA may be slow or none, depending on factors such as the mobility of target species relative to the size of the MPA (Corten 1990) and the level of protection implemented within the MPA. In practice, the benefits of closed areas will usually be very difficult to measure in the sea against the large natural variation seen in fish stocks (Horwood et al. 1998).

MPA effects rely on a number of factors (FSBI 2001):

i) Proportion of fish stock within its boundaries,
ii) Biological characteristics of the protected fish species,
iii) Spatial distribution and magnitude of fishing effort outside the MPA,
iv) Relative “catchability” of fish outside MPA,
v) Other fisheries management systems in place.

Predictability of effects of MPAs also rely on knowledge of oceanographic features and their effects on mortality, recruitment, migration, mobility, etc.

Many of our existing fishery management tools (such as MPAs) have conservation value (Roberts et al. 2005; Nilsson 2005). Gear regulations, in tandem with spatial planning, may provide the key to protecting sensitive habitat types (RCEP 2005). In any case, MPAs will only be successful if we set them up in the right way and for the right reasons (Sale et al. 2005).

MPA design

MPAs should include critical adult habitat and should be sufficiently large to support breeding populations with a stable age structure. Juvenile habitat should be included for species that utilise different habitats as juveniles, especially when juveniles are vulnerable to fishing mortality (Bohnsack 1990). Modelling predicts that large MPAs will increase resilience to overexploitation by keeping the spawner biomass and recruitment success at higher levels than in non-protected areas (Guenette and Pitcher 1999). According to Guenette & Pitcher (1999), MPAs ranging in size between 50 and 75% of stock area are necessary to optimise yields. However, in the case of Georges Bank (NE USA), groundfish species are recovering following mobile gear closures of just 4.000-7.000 km2 (Roberts et al 2005). According to Sale et al. (2005), efforts to prescribe the correct percentage of sea area to protect to sustain a fishery have limited scientific support, and attempts to specify a universal proportion for protection seem naïve.

Hastings & Botsford (2003) compare and contrast the design of networks of MPAs for biodiversity conservation and for increasing fishery yields. They conclude that for biodiversity purposes MPAs should be as large as possible. In contrast, the fisheries goal of maximising yield requires maximising fish larval transport outside of MPAs (i.e. “spillover”), which means that MPAs must be as small as practically possible. These conclusions, however, are based on several simplifications and assumptions that do not reflect the actual behaviour of marine ecosystems. Among these is the assumption that all larvae are mobile and all adults are sessile.

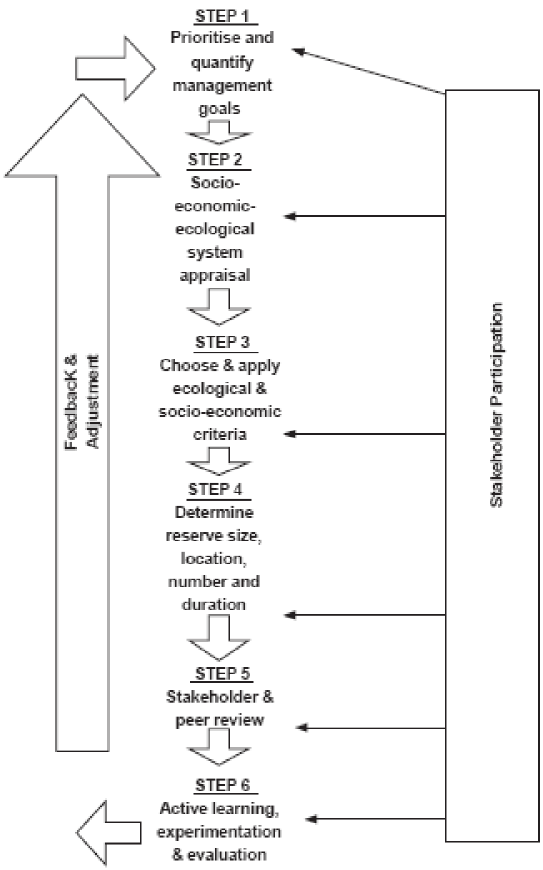
In addition, MPA size cannot be based on biology alone. The process must also be seen from a social point of view, where administrative practicality is also important.

Rotational fisheries closures have also been mentioned as an option, where areas are rotationally opened and closed to fisheries in order to allow stocks to recover. However, these must be large enough for fish stocks to recover within them, and benefits may quickly disappear when areas are reopened (Bohnsack 1996), as was seen in the reopening of the North Sea Cod box (ICES 2004a) (see review of North Sea MPAs).

In summary, the value of trying to develop general rules of thumb for design of MPAs may be limited. Considerable research on the behavior of both fish and fishers in a given fishery will be required before MPAs can be designed effectively for that fishery (Holland 2000).

MPA Management

MPAs often have the advantage that their purpose is very apparent and clear-cut. Fishers, for example, can see that the goal of a fishery closure is to protect and conserve a specific stock. On the other hand, this narrowness means that while such MPAs play an important role in fishery conservation, and may well provide indirect ecosystem benefits, they do not provide comprehensive ecosystem conservation, and do not reflect a full ecosystem approach (Charles 2001). MPAs are a powerful management tool, but work best if they are a supplement, not a substitute for other instruments (Roberts et al. 2005). For instance, area closures are not enough for e.g. migratory species (Lauck et al 1998).



Six steps for active adaptive management of MPAs for fisheries purposes (Grafton & Kompas 2005).

According to Sale et al. (2005) and Grafton & Kompas (2005), the best way to proceed with MPAs is to use the existing science in adaptive management approaches for the design and implementation of MPAs. Active adaptive management is a process to improve management given uncertainties. Ideally, MPAs should be designed on the basis of best available science. Their effects should be evaluated, and the results integrated into improved management practice, i.e. adaptive management (Sale et al. 2005).

However, criteria other than scientific ones are also important. Socio-economic data is needed to predict, for instance, what the impact is on a fishing community of the establishment of an MPA of a particular design, and how does that community's response change fishing effort in the remaining fishable area (Sale et al. 2005)? Arbitrary designation of areas as MPAs may result in significant reductions in fisheries revenues and may have quite unequal and unintended impacts on different groups of fishers (Holland 2000). In reality, MPAs are proposed, designed, legally codified, implemented and managed through socio-economically complex and largely political processes (Sale et al. 2005).

Grafton & Kompas (2005) propose a six-step process for establishing and adaptively managing MPAs for fishery purposes (see figure). The key point is that it involves a process of active learning, planning, evaluation and judgment about the socio-economic-ecological environment and the effects of key decision variables.

Jamieson & Levings (2001) hypothesise that, in balancing options regarding the closure of marine areas, assumption by managers that if area allocations are approximately equal within an arbitrary accepted harvest level, they are probably fair. However, while this assumption may generally apply to opportunities for monetary wealth, its applications to biological systems is not based on science and it is not likely to ensure sustainability over the long term.

Mistakes and caveats

Common to most of the existing North Sea MPAs (see review of North Sea MPAs) is that they have not had much success in reaching their management objectives. In most cases it is difficult or impossible to separate effects of management and natural variations ocurring throughout the lifespan of the MPA. However, there are some fair generalisations that can be made.

First of all, none of the existing North Sea MPAs are protected *per se* (FSBI 2001; see individual MPA reviews below). In e.g. the Plaice box and other established fisheries closures in the North Sea, smaller, less powerful vessels (including beam trawlers), as well as vessels targetting other fisheries than the species being managed (e.g. *Crangon* shrimpers in the Plaice box), are still permitted to fish. The Royal Commission on Environmental Pollution (RCEP 2005) recommended that the UK government should review the activities and impact of smaller vessels that do not fall under the full set of fishing controls in marine areas. This is because they were aware that small vessels of sizeable capacity had been built to benefit from set cut off points established in connection with MPAs/fisheries closures. Combined with a ubiquitous lack of recorded reference data prior to the closure, these issues leave us in the current situation, where we are unable to determine whether or not the use of fisheries closures/MPAs is indeed a successful strategy.

Fisheries managers must guard against overfishing of other stocks toward which effort maybe redirected as a result of area closures. Furthermore, the concentration of fishing effort in smaller areas open to fishing could lead to habitat degradation in those areas (Holland 2000). The latter was one of the unintended results of the establishment of the North Sea Cod box (see review of North Sea MPAs).

It is premature to assume that MPAs/fisheries closures are invariably effective in fisheries management, because there are relatively few empirical studies, many of which are poorly designed, and even the reported increases in density within MPAs can be slight (Sale et al. 2005). To illustrate this, empirical studies of MPAs published before 2002 were outnumbered by theoretical papers and reviews (44% vs. 56% of 205 total) (Willis et al. 2003), and we may soon be in the unusual situation of being faced with a greater number of reviews than there is reviewable material (Willis et al. 2003).

Science gaps

Theoretical studies have focused on the mechanisms of spillover (*i.e. the enhancement of production of a fishery species, within the fished locations surrounding one or more no-take reserves, owing to the net movement of juveniles and adults out of the reserve (Sale et al. 2005)*), and recruitment subsidy (*i.e. the enhancement of production of a fishery species, within the fished locations surrounding one or more no-take reserves, owing to the net export from the reserve of pelagic larvae. (Sale et al. 2005)*), but these theories are rarely tested

(Willis et al. 2003). In addition, there are many fishery species about which we need more basic ecological information before implementing MPAs to help manage them (Sale et al. 2005). Accurate assessment of closed areas in most sea areas is fraught by lack of control areas and monitoring programmes to study their effects (e.g. Hoffmann & Dolmer 2000; Jennings & Kaiser 1998). Benchmarks are needed, not only for assessment of the status of ecosystems but also to assist in development of coastal management and habitat plans in the context of biodiversity conservation. Above all, there is a need for research manipulations that will empirically test the efficacy of MPAs as fishery management tools (Sale et al 2005). "Before After Control Impact Pairs" (BACIP) is a popular sampling design that enables the unambiguous testing of effects on an ecological system owing to a particular impact, such as creation of an MPA (Sale et al 2005). None of the existing North Sea fisheries closures have been implemeted in a manner that allows scientific testing and monitoring of their effects. According to Horwood et al. (1998), the much needed protection of European commercial fish and their fisheries cannot in general be achieved through technical measures, such as closed areas and mesh sizes, alone. Their conservation requires permanent reductions in fishing mortality, which means that less fish must be killed. Pastoors et al. (2000) caution that MPAs, although intuitively attractive, may be a less straightforward management tool.

Sale et al. (2005) identify five crucial gaps in the ecological science of MPAs:

1. Distance and direction in which marine larvae disperse is a primary ecological issue because it directly determines three key things:
    - Size of planned MPA, as it determines rates of self-recruitment
    - Placement and spacing of a network of MPAs and the persistence of target populations through recruitment among them
    - Sizes, spacing and placement of MPAs and the ability to maximize potential fishery benefits on neighbouring fishing grounds through recruitment subsidy, i.e. the enhancement of production of a fishery species, within the fished locations surrounding one or more MPAs, owing to the net export from the MPA of pelagic larvae (Sale et al 2005) (see also e.g. Botsford et al. 2003).
2. We know more about the patterns of movement during juvenile and adult phases of fish, but even here there are serious gaps. In addition, some species might be too mobile for management using MPAs to be practical.
3. Knowledge of ecosystem impacts of fishing is important. The lack of fishing in an MPA may lead to unpredictable changes in community structure.
4. We lack adequate knowledge of the behaviour of water masses.
5. We have remarkably few well designed studies of MPAs that can rigorously demonstrate that they have sustained or enhanced fishery yield in surrounding areas.

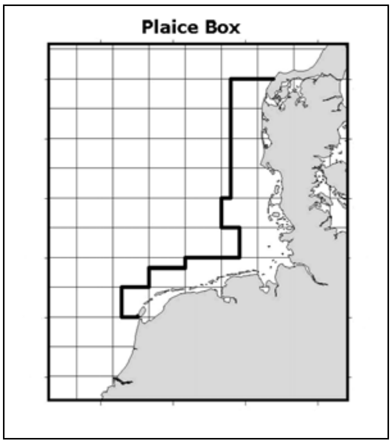
These gaps prevent the development of an explicit science for MPA design, one that can generate quantitative criteria for use in planning of MPA networks (Sale et al 2005).

Concluding remarks

PROTECT will contribute to the identification and filling of science gaps, so we in future may avoid mistakes of the past and make qualified management decisions regarding the use of MPAs for the conservation of marine ecosystems and the management of fisheries.

The North Sea Plaice Box

Map



Approximately 30 nm wide extension of the 12 nm zone stretching from Den Helder in the Netherlands to Hanstholm in Denmark.
Map Source: http://europa.eu.int/comm/fisheries/news_corner/press/inf05_46_en.htm

Purpose of establishment

North Sea Plaice is mainly taken in a mixed flatfish fishery by beam trawlers in the southern and south-eastern North Sea. Minimum mesh size of 80 mm results in much discard (ICES 1987). In addition, the survival of discarded plaice is very poor (unpublished RIVO report 1985 in ICES 1987).

The Plaice box was established to reduce discards of undersized plaice and sole in their main nursery grounds (FSBI 2001; Marchal et al. 2002).

The scientific basis of the current closure is the notion that by a reduction in fishing effort in areas with a high abundance of undersized plaice, discard mortality rates will be reduced so a larger proportion of each cohort of 0-group fish will recruit to the fisheries and to the adult population (Rijnsdorp 1998). Scientific basis developed in 1987 by ICES North Sea Flatfish Working Group (ICES 1987). ICES (1987) advised closure to reduce discard rate.

- A similar proposal was discussed in 1912 by the ICES Plaice Committee. The Plaice Committee agreed that the plaice stock in the North Sea had suffered a decrease in larger-sized fish since the advent of the steam trawling fishery. It agreed that closing nursery grounds would preserve smaller plaice for capture after they grew to a more valuable size, but recognised the political difficulties of such a measure internationally (Rozwadowski 2002).

Selection methodology and design

North Sea Beam trawlers mainly target sole and plaice. Undersized juveniles of these species mainly reside in shallow coastal waters along the continental coast (Piet et al. 1998).

Distribution maps showed that discarding was concentrated on age group 2 and 3 in rectangles along the Frisian Islands, German Bight and up along the Danish coast (ICES 1987). It was expected that a closure of the areas with the highest densities of young plaice would have the highest impact (ICES 1987).

The Plaice box is not part of a designed network of closed areas (ICES 2004a).

Implementation process and legal aspects

The Plaice box is based on EEC Council Regulation No 4193/88 (FSBI 2001).

At first Dutch trawling in the Plaice box area was restricted to small vessels harvesting according to gear and catch restrictions (Piet et al. 1998; Piet and Rijnsdorp 1998) to directly and indirectly reduce numbers of juvenile plaice and sole caught and increase predicted plaice recruitment by 25% (ICES 1994) and sole recruitment by 11% (Rijnsdorp et al. 1998). Failure to meet predicted recruitment levels, possibly due to increased legal fishing within the box (Pastoors et al. 2000) led to regulations being extended to the fourth quarter in 1994 and from 1995 the whole year. In summary, the Dutch trawling effort in the Plaice box was reduced in several phases:

At first the area was closed only during 2nd and 3rd quarters, but in 1994 the closure was extended to 4th quarter. Since 1995 the Plaice box has been closed for all but the exemption fleet all year (Piet et al. 1998; Marchal et al 2002).

Period	Regulation	Period
-1989	No specific regulation	
1989-1993	No fishing within the box by bottom trawlers with vessels larger than 300hp. (Not applicable to exemption fleets)	2nd and 3rd Quarters
1994		2nd, 3rd and 4th Quarters
1995-present		Year round

No fishing is allowed inside the Plaice box within 12 nm of the coast by vessels exceeding 8m overall using beam and otter trawls (Council regulation (EEC) No. 3094/86) (Piet & Rijnsdorp 1998).

Inside the Plaice box (beyond 12 nm from the coast), no fishing is allowed by beam trawlers and otter trawlers exceeding 24 meters and 300 Hp (“Eurocutters”). Fishing by other vessels (exemption fleet) is permitted provided that they are (Piet & Rijnsdorp 1998):

- on an authorised list and that vessel length is less than 24 m and engine power does not exceed 300 Hp, even if fishing with beam trawls
- not on a list but fishing for shrimp (incl. beam trawl)
- not on a list but fishing with other trawls using 100mm mesh, even if engine power exceeds 300 Hp, provided that at least 5% of the catch is sole, and no more than 10% of the catch is composed of cod, haddock and saithe.

Ecosystem effects

Effects of the Plaice box were predicted based on estimated changes in yield per recruit and spawning stock biomass (SSB) per recruit under various fishing patterns (ICES 1987). Using quarterly data on distribution of age groups and distribution of undersized plaice per ICES rectangle, expected gain in recruitment to the fishery was calculated for various scenarios, under the following assumptions (ICES 1987):

- quarterly spatial distribution of each age group was fixed and not affected by changes in fishing patterns or growth
- a constant growth rate, independent of density
- all effort was expelled from the box

<u>Predicted effects of the Plaice box</u> (ICES 1987):

- For a cohort of plaice, proportion surviving could increase by ca. 25% if box closed for all discarding fleets in 2nd and 3rd quarters; and almost ca. 35% if closed all year.
- General enhancement of sole predicted, but to a lesser extent than plaice, due to generally lower discards in sole (Rijnsdorp and van Beek 1991).

To measure the effects of the Plaice box according to its objectives, the question that must be asked is: Has the cumulative discard mortality until the time when the cohort reaches the minimal landing size decreased? (ICES 1999)

<u>Actual effects</u>: According to various references, the reduction in Dutch beam trawl effort to around 6% (e.g. Pastoors et al. 2000) of the original level led to:

- Reduction in overall juvenile discard (Pastoors et al. 2000).

- No real signs of improvement (ICES 1999).
- No change in species composition (Piet and Rijnsdorp 1998).
- Increase in abundance of commercial fish species within marketable size range (Piet and Rijnsdorp 1998).
- Positive effects (in 1994) were probably reduced by low growth rate, exemption fishing fleet and increased fishery in 4th quarter (ICES 1994 in ICES 1999).
- Increase in species richness due to influx of southerly species and decrease in relative abundance of plaice, within and outside Plaice box (FSBI 2001).
- SSB (spawning stock biomass) and yield have decreased since initial claims of increase (Pastoors et al. 2000).
- If the Plaice box were removed, long term standing landings and SSB would decline 8 and 9% respectively (Horwood 2000).
- Beam trawl discards remain very high inside and outside of the Box. Discard is higher inside the Box than outside, usually made up of mostly age 2 plaice (18-27 cm). Shrimp fisheries also appear to have high discards (ICES 1999).

Overall, it is not possible to determine the extent to which the Plaice box has contributed to the apparent increase in the fishing efficiency of some of the exemption fleets fishing on the grounds where management has been implemented (Marchal et al. 2002).

<u>Documented effects of the Plaice box on invertebrate fauna:</u>

- Data of by-catch of benthic invertebrates of two beam trawl surveys showed significant effects of closure. Closing the box in 2nd and 3rd quarter caused an increase in abundance of several benthic invertebrate species followed by a decline when the Plaice box was closed year-round. Perhaps the most abundant were scavengers and predators for which the deleterious effect of additional mortality is overruled by a decreased competition for food and risk of predation.
- Same shift to opportunistic species (mainly polychaetes) adapted to disturbed habitats has been observed in Dogger Bank, Wadden Sea and German Bight (Kröncke 1990, 1995 in ICES 1999).

In addition, the Plaice box is important for breeding Sandwich tern populations and for red and black throated divers, red-necked grebe, common scoter, little gull and common gull (Skov et al. 1995).

Effects on fisheries effort / benefits

Larger beam trawlers (>300Hp) continued to fish in the Box especially in 4th quarter in the period 1989-1994 (Marchal et al. 2002). Surplus effort was probably intensified outside box (Piet et al. 1998). 1993-1996 confirms heavy exploitation just outside the Plaice box by large vessels as well as inside the Plaice box during open months (Rijnsdorp et al. 1998).

The Dutch beam trawl effort was reduced to 40% between 1989 and 1993 (Pastoors et al. 2000). After 1994-1995, beam trawl effort decreased to 6% of original levels (Pastoors et al. 2000). Thus, the year-round closure resulted in a 94% reduction in effort of large Dutch beam trawlers. Most of the effort, however, was displaced to areas just outside the Plaice box (ICES 2004a).

The Plaice box has been an effective measure to exclude large beam trawlers (Pastoors et al. 2000). Reduction in beam trawl effort implies that discard mortality rate is decreased.

Fishing with small vessels continued in the Plaice box, in fact the exemption fleet increased in capacity (Pastoors et al. 2000). Fishing effort of *exemption* beam trawlers (max. 24 m and 300 Hp) increasedby 90% between 1989 and 1994 (Grift et al. 2004). For instance, the main effort build up of the Dutch shrimpers from 1989 to 1993 took place inside the Box, caused by an increase in small roundfish vessels switching to fishing for *Crangon* (brown shrimp) (ICES 1994). It decreased again by 45% between 1994 and 1998 (Grift et al. 2004). Simultaneously, stricter enforcement of engine power limitations in the German area brought effort down as well as reduced catch rates. An increase in Danish gillnet fishing efforts took place between 1989 and 1994 (Grift et al. 2004).

Landings per unit effort for Plaice decreased by more than 50% in the Box, but percentage of

Plaice discards (% of numbers caught) in the beam trawl fishery increased from 77% between 1976 and 1990 to 87% between 1999 and 2003, both in terms of numbers and in biomass (Grift et al. 2004).

Socio-economic effects on fisheries and other stakeholders

German otter trawlers found that the Plaice box measures were too restrictive (Dahm et al. 1996). Among most interviewed fishers (Venema 2001) there is much incongruence in the perception of the Plaice box and its effects. The perception of fishers is naturally dependent on whether or not and/or how their fisheries are affected by the establishment of the closure. Thus, the Plaice box has a varying degree of support from fishers, ranging from "waste of time"/"big mistake" to "undivided support" (Venema 2001).

- "There is generally a lack of communication between authorities, biologists and the fishers. No one has attempted to communicate with fishers" (quote from a fisherman in Venema 2001).
- A fisherman with a vessel greater than 300 hp states in Venema (2001): "There is no control over number of Eurocutters (≤ 300hp) fishing in the Box. Their numbers are increasing within the Plaice box. A high number of Eurocutters of 300hp are just as destructive as beam trawlers greater than 300 hp. A trend among dutch fishers is to sell larger vessels and buy Eurocutters. Eurocutters are exempt of logbooks, but shouldn't be."

Registered engine power cannot in general be considered totally reliable (COM 2001). According to an interviewed fisher, new engines of e.g. 2000 hp can have a much greater power than an old engine of 2500 hp (Venema 2001).

"…there is a perception by the local industry that the Box provides some socio-economic benefit even if there is little evidence for this" (Anon 2005).

Lessons learned

There is no direct evidence that the Plaice box has had a positive effect on recruitment. Since the Plaice box was established in 1989 recruitment has shown a negative trend for the southern North Sea, i.e. SSB (spawning stock biomass) and yield are down by 60% (Grift et al. 2004).

The effects of discard reduction may have been offset by ecosystem changes in the North Sea ecosystem around the time of the establishment of the Plaice box (Rijnsdorp 1998; Pastoors et al. 2000) (changes in species abundance and composition in southern NS and reduced growth rates for plaice (ICES 1999; Pastoors et al 2000; Jennings & Kaiser 1998)) and/or relatively low number of pre-recruit plaice in early 1990's (Pastoors et al. 2000).

A shift in the distribution of juvenile plaice has also been suggested as an explanation (Rijnsdorp 1998). For instance, juvenile plaice usually avoided deeper waters because of predation by cod. As cod stocks are lower now than in previous times, there is less reason for juvenile plaice to avoid predation, i.e. they may leave the Plaice box and swim into deeper waters. Alternatively, a decrease in the abundance of older plaice may have led to less competition for food in deeper areas, i.e. smaller plaice may swim into deeper waters to forage (Rijnsdorp 1998).

The expulsion of Dutch beam trawlers has been blamed for the drop in ecosystem productivity. However, the literature shows (e.g. Schratzberger & Jennings 2002; Schratzberger et al. 2002) no positive effects of bottom trawling on ecosystem productivity.

There is no single parameter from which the ecological effect of the box can be measured. The Plaice box management measure was not set up as experimental design, with a control area, that would have allowed statistically sound comparisons and conclusions (Grift et al. 2004). Effects of MPA on size structure have been shown, but closure effects are in this case impossible to separate from natural changes (ICES 1999; FSBI 2001).

The Plaice box is not a closed area: There are still beam trawlers ≤ 300hp, a *Crangon* (shrimp) fleet and otter trawls operating in the Plaice box. In 2003, still *7%* (6.695 tonnes) of the total plaice landings from North Sea came from the Plaice box (Grift et al. 2004).

Data is lacking in many cases on the spatial distribution of fleets. Data shows that there is still a substantial amount of trawlers exceeding 300hp fishing in the 12 nm zone and in the Plaice box after its full implementation in 1995 (Marchal et al. 2002).

Expected gains were reduced by the increasing amount of effort exerted by small vessels and larger trawlers in the 4th quarter within the Plaice box since its closure in 1989 (ICES 1994).

Using quarterly data on distribution of age groups and distribution of the proportion of undersized plaice per ICES rectangle, expected gain in recruitment to the fishery due to the establishment of the Plaice box was calculated for various scenarios, under the following assumptions (ICES 1987; Pastoors et al. 2000), both of which are quite unrealistic (Ed.):

- quarterly spatial distribution of each age group was fixed and not affected by changes in fishing patterns or growth
- constant growth rate, independent of density

The best way to make the Box effective would be to prohibit all demersal trawling in the area, regardless of gear and engine power (Anon 2005). Closure of the whole box to all vessels on a year-round basis would provide greater fisheries benefits (landings and SSB would increase by 24 and 29% respectively (Horwood 2000). Many young plaice die when discarded from e.g. permitted *Crangon*-shrimpers. Total closure would potentially also lead to increased recruitment rates in sole, which also suffer high discard levels (ICES 1999).

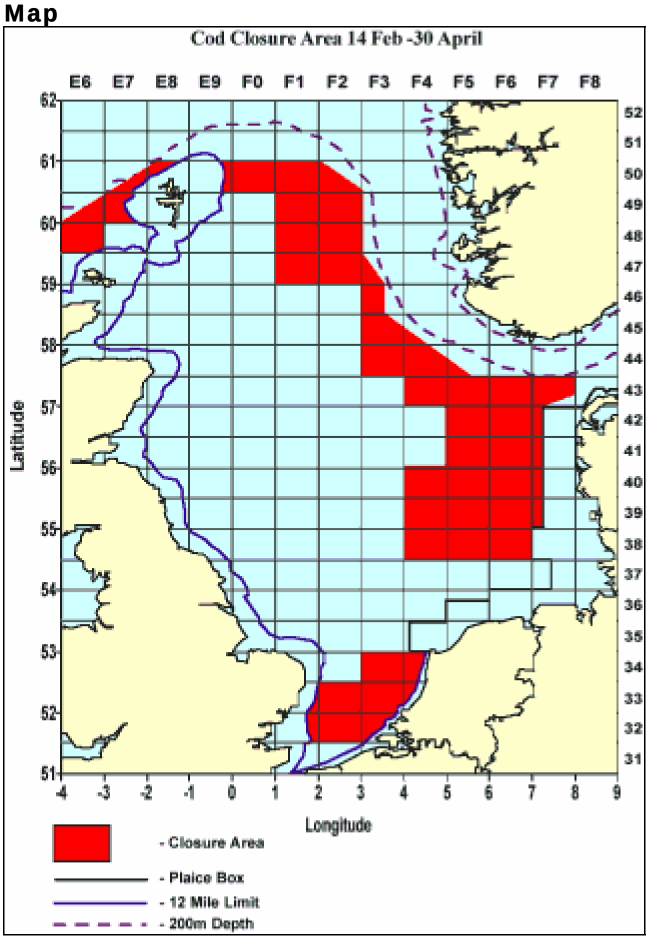
Additional recommendations regarding the Plaice box (Grift et al. 2004):

- Specific aims and objectives of the closure should be considered and well defined
- Relevant, measurable criteria should be considered/developed
- A research programme should be established to monitor effects over a predetermined time scale
- The Plaice box should be established in an experimental setup, which allows for the separation of autonomous developments and the closure (with or without fishing) effects, for example a control area which differs from the treatment area only in terms of fishing intensity.

According to the European Parliament (1999), "a recurrent theme in the EU has been the weakening of conservation policy for particular national social and economic interests, and the Plaice box is a good example of this".

The North Sea Cod Box

Map



Map source: www.cefas.co.uk/fsmi/roundfish.htm

Area of more than 40.000 square miles, almost a fifth of the North Sea, that in 2001 was closed to fisheries likely to catch cod for 75 days (Dinmore et al. 2003).

Purpose of establishment

The cod stock in the North Sea was considered by ICES to be outside of safe biological limits and at serious risk of collapse (ICES 2001; Cook et al. 1997).

The immediate requirement was to allow as many cod to spawn in the period mid-February to end April 2001 (ICES 2004a).

EU Council asked the Commission of the European Communities to establish a plan to protect the cod stock during spawning season and to stop misreporting and discarding of cod in all fisheries. This plan was called the Cod Recovery Plan and included:

- Closed areas
- Technical measures
- Comprehensive proposals for longer-term measures

The North Sea beam trawl fishery doesn't primarily target cod, but cod are taken as a significant and valuable by-catch and vessels fish in many cod spawning areas.

In 1993 the EU had investigated possible effects of closing cod areas. It was concluded that, due to a limited understanding of fish movements and fleet behaviour, a closure would do very little for cod, even if they were very large areas. (Horwood 2000).

Selection methodology and design

The closed area was part of the Cod Recovery Plan and was not designed as part of a larger network of closed areas (ICES 2004a).

Implementation process and legal aspects

In November 2000, ICES indicate that cod stock in North Sea area IV is in serious risk of collapse (ICES 2004a). The Council meets in December 2000, where the Commission and Council note an urgent requirement to establish a recovery plan for the North Sea cod stock, termed the "North Sea cod recovery plan" (ICES 2004a). An Agreed Record was signed January 24 2001 by EU and Norway, indicating the management measures which should take place (ICES 2004a).

It was decided that it was urgent that a closed area be established. However, the North Sea Cod box took months to implement (ICES 2004a).

Commission Regulation (EC)No 259/2001 of 7 February 2001 establishes measures for the recovery of the stock of cod in the North Sea (ICES sub-area IV) and associated conditions for the control of activities of fishing vessels.

- However, fishing for sand eel and pelagic species were allowed in the Cod box. It was decided that observers should be placed on board vessels fishing for these species.

Ecosystem effects

The closure probably had a negative impact on the rate of discarding of vulnerable components of the ecosystem (e.g. elasmobranchs or long-lived benthic species) due to an increase in trawling activities in areas that are not normally fished (ICES 2004a).

No data exists that allows an evaluation of changes outside the closure (ICES 2004a; Rijnsdorp et al. 2001). The closure may even have been counter-effective for cod, commercial species and benthic ecosystems (Rijnsdorp et al. 2001).

In addition to overfishing, the North Sea cod stock is threatened by a decline in the production of young cod that has paralleled warming of the North Sea over the past ten years. Possible persistence of adverse warm conditions combined with a diminished stock endangers the long-term sustainability of cod in the North Sea. To decrease risk of collapse, fishing pressure must be reduced (O'Brien et al. 2000).

Effects on fisheries effort/benefits

Fishing activities were monitored using Vessel Monitoring Systems (VMS) and the biota (demersal fish and benthos) during several bottom surveys (ICES 2004a). VMS was very effective in enforcement. During the period target effort was reduced by (probably) 100% within the Cod box (ICES 2004a).

Beam trawl fisheries were affected. Beam trawlers in the area target sole, plaice, dab, turbot and brill, but they also catch roundfish such as cod as by-catch (Rijnsdorp et al. 2001).

Eurocutters (beam trawlers up to 300 hp) were not directly affected by the area closure, since they may fish in the 12 nm-zone. These smaller vessels may even have benefited from reduced catches in the Cod box, since sole within the closure migrate to shallow coastal areas within the 12nm-zone to spawn in spring (Rijnsdorp et al. 2001).

Discard information shows that plaice discards were about 78% in the box area (ICES 2004a). Adjacent to the box area the discards were 31% before closure but 74% in the period 1999-2000 for focal species. For commercial species there was a minor increase in discards from 12% to 19% (ICES 2004a).

Displaced beam trawlers continued fishing throughout the closure, but in other fishing grounds (Rijnsdorp et al. 2001). Beam trawl effort mainly moved to the area "Open North". Some of the beam trawling effort was displaced to areas that had never been beam trawled before (Rijnsdorp et al. 2001; ICES 2004a), and recovery of benthic communities in these areas was expected to take more than 10 years (ICES 2003). Environmental effects of trawling on diversity, biomass and production of benthic communities are expected to be greater in these previously untrawled and infrequently trawled areas than in the normal fishing grounds (ICES 2003; Frid et al. 2005).

No data exists that allows an evaluation of changes outside the Cod box (ICES 2004a). However, no beneficial effects of the closure on cod are registered (Rijnsdorp et al. 2001; ICES 2004a).

Catches of commercial species within the Cod box were higher after re-opening but returned to normal after 2-3 weeks (ICES 2004a).

Socio-economic effects on fisheries and other stakeholders

- no data

Lessons learned

INSIDE the Cod box: Closed areas only partially overlapped with known spawning grounds (Rijnsdorp et al. 2001; ICES 2004a). In the southern grounds, peak spawning takes place from weeks 4-7 and probably somewhat later further north. The Cod box was closed weeks 8-17 so it probably only protected the second part of the spawning season (ICES 2004a; Rijnsdorp et al. 2001).

The aim of the emergency closure was to reduce fishing mortality on spawning cod, but the wider consequences of this closure were not considered at the outset (Frid et al. 2005)

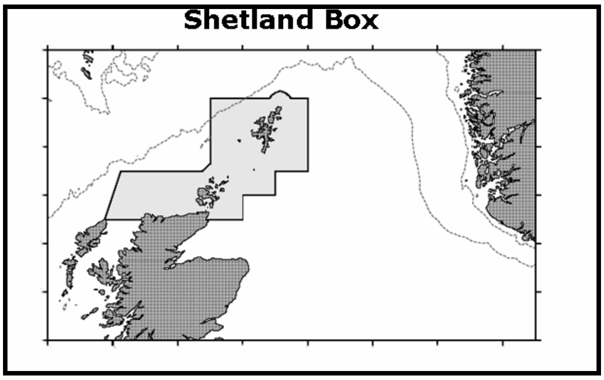
Closure did not meet objectives. Inappropriate timing and positioning of the area resulted in that no positive effects of the closure were achieved (ICES 2004a).

There was no overall effort reduction during closure, only displacement of fishing effort (Rijnsdorp et al. 2001).

The Cod closure was rather poorly designed, did not consider side effects on the level of discarding in demersal stocks, and did not consider the wider ecosystem implications (Rijnsdorp et al. 2001).

The Shetland Box

Map and description



Map Source: http://europa.eu.int/comm/fisheries/news_corner/press/inf05_46_en.htm

Area around the north of Scotland, Orkney and Shetland. Commercially important demersal species in the Box area are: cod, haddock, whiting, saithe, and anglerfish (Kunzlik 2001).

The purpose of establishment

Established in 1983 to protect "species of special importance…which are biologically sensitive by reason of their exploitation characteristics." (NAFC 2004)

The Shetland box played an important role in attempts to achieve a balance between the different fleets and fishing communities.

Selection methodology and design

In principle the main criterion was to grant preference to local fishing vessels (Crean & Wisher 2000).

Implementation process and legal aspects

The legal basis of the Shetland box is Council Regulation (EC) No. 2371/2002 of December 2002 on the conservation and sustainable exploitation of fisheries resources under the Common Fisheries Policy (CFP). The number and nationality of large demersal vessels fishing at any one time is restricted by a CFP licensing scheme (Council Regulation (EC) No. 2371/2002) (NAFC 2004).

Vessels more than 26 m fishing for other than blue whiting and Norway pout are only allowed inside with a license from the European Commission. Allocations (below) are based on track records prior to partial closure (North Atlantic Fisheries College. Vessels without licenses may only enter if less than 26 m, unless they fish only for blue whiting and Norway Pout. There are 128 licenses: 62 to UK, 52 to France, 12 to Germany, 2 to Belgium (NAFC 2004).

The exemption of Blue whiting and Norway pout is to clarify what is covered by "fishing for demersal species". This is because these species are usually caught using different techniques closer to those used in pelagic fisheries, and the species are covered by other regulations, among others the Norway Pout Box (COM 2002).

Ecosystem effects

On the basis of fisheries sensitivity maps (Coull et al. 1998 in NAFC 2004) the Shetland Box is suggested to have relatively important, disproportionate concentrations of spawning and nursery grounds for 9 of 13 species for which maps were available. There appears to be a case for retaining (or strengthening) current management arrangements (NAFC 2004).

Shetland box contains a disproportionate concentration of mature haddock and whiting, young anglerfish and, to a lesser extent, young haddock than neighbouring waters. It indicates that the area is important in the distribution of these fish at a time when the abundance of the principal gadoid fish stocks is known to be generally reduced (Kunzlik 2001).

However, the vulnerability of stocks and importance of areas rely on a qualitative view of data. They reflect differing impacts on species, which also vary in age. Nevertheless, taken together, they support the argument that the region of the Shetland Box is of conservation importance to the species concerned (Kunzlik 2001).

Effects on fisheries effort / benefits

For light trawlers, annual Landings Per Unit Effort (LPUE) when fishing in the Shetland Box are consistently higher than when fishing outside the box (Anon 2005).

Demersal fish stocks of importance to the region are shown to have declined generally in abundance since initial EEC Regulation was adopted in 1983, especially for cod, whiting and haddock (Kunzlik 2001).

Spawning Stock Biomass (SSB) estimates for 1999 are substantially below that of 1983 for cod, haddock, and whiting, and close to its 1983 value for saithe. *Cod*= continuous decline. *Whiting*=stability throughout 80's then continuous decline. *Haddock and saithe*=current stock estimates indicate upturn following lowest observations in 1990's (ICES 2001).

Socio-economic effects on fisheries and other stakeholders

There is a heavy economic dependency of the area's local communities on fishing. They are still dependent on fishing In 1998 33% of Shetland economic turnover was from fishery and appr. 20% of active population is employed in the fishing industry (DEFRA 2002). The Box is a statement of the importance of fishing to the islands (Crean 2000).

Some say that Shetland Box was established to protect northern Scottish fishing communities (NAFC 2004). Some say that the Shetland Box has nothing to do with fisheries, but rather is a compensation to the UK for accepting conservation elements of the Common Fisheries Policy (Holden 1994). The general view among interviewed fishers (NAFC 2004) is that the retention of the Shetland Box could be acceptable if a sufficiently compelling case was made for its conservation benefits.

Discussions from representatives of fishing communities from Member States with and without access revealed support for *non-discriminatory* measures to conserve fish stocks. They were, however, unconvinced of the positive effects of the Shetland Box. They say it must be proven better AND be non-discriminatory (NAFC 2004).

Interviewed Shetlands fishers: The Box, as it is constructed, is viewed as relatively unimportant with regard to excluding outsiders and, therefore, its potential to lessen exploitation pressure upon fisheries resources (Crean 2000; Crean and Wisher 2000).

A strong majority of Shetland fishermen believe that local fishermen do not have enough say in management of coastal fisheries resources and that fishermen's knowledge was not used to help formulate fisheries management regulations (Crean & Wisher 2000). In addition, they believe that fisheries regulations in force do not suit local conditions.

Lessons learned

It seems unlikely that the management regime for the box has ever effectively restricted the level of fishing effort. There is no evidence of unsatisfied demand for licences or for access to the Box. Vast majority of vessels are too small to require a license in any case (NAFC 2004).

To keep the Box, it must be based on future potential and not the past record (NAFC 2004).

If the Box is renewed it will be necessary to develop new management regime that is not overtly discriminatory. (NAFC 2004).

Value of the Shetland Box to Shetland itself is largely, if not entirely, symbolic. Not to say that it is not an important area in biological conservation terms or as a potential conservation tool (NAFC

2004).

Key interviewed informants of the Shetland Islands can be said to have the following points of view, among others (Crean & Wisher 2000):

- diminished capacity of the centre to exert control
- marginalisation of local knowledge/views
- inadequate penalising of rule breakers

No system was ever established to monitor the Shetland Box or to collect the data that would be needed to demonstrate its effectiveness (NAFC 2004).

The Norway Pout Box

Map

Norway pout box was introduced in 1986. Its size is 95.000 km^2 or appr. 30.000 square nautical miles and it overlaps with the Shetland (or North of Scotland) Box.

Purpose of establishment

According to EC Regulation No 3094/86, the purpose of the Norway pout box is to reduce levels of fishing mortality on juvenile gadoids such as haddock and whiting in the Norway pout fishery, and hence increase the recruitment of these species to the stock biomass for sustainability and for future fisheries (Anon 1986).

Selection methodology and design

The Norway pout box was designed by an expert committee.

EC Regulation No 3094/86 defines the boundaries of the Norway pout box (Anon. 1986).

Implementation process and legal aspects

Norway Pout is regulated by minimum mesh size, the Norway pout box and by-catch regulations to protect other species (ICES 2004).

UK Government ratifies statutory instrument setting up area closure of the Norway Pout fishery in Feb 1977.

Dates	Extent of Box			
	Northern Boundary	Eastern Boundary	Southern Boundary	Western Boundary
21 Feb – Mar 77	60°N	0°	56°N	4°W
1 Apr – 31 Aug 77	None	None	None	None
1 Sept – 15 Oct 77	60°N	0°	56°N	4°W
16 Oct 77–30 Sept 78	60°N	0°	56°N	4°W
1 Oct 78 – present	60°N	2°E median	56°N	4°W

(Table: Modified from ICES 1979)

Restrictions on fishing for Norway pout with small meshed trawls to protect other roundfish: The Norway pout box is a defined area in the Northern North Sea, east of Shetland. Retention of Norway Pout on board a vessel inside the Box (exceeding a 5% by-catch level) is considered to be an offence. This regulation is to prevent the capture of juvenile haddock (which are abundant within the Box) by vessels that use 16mm nets, which are allowed for Norway Pout elsewhere (European Parliament 1999).

In 2005 the fishery was closed, and there has been no directed effort for Norway pout in the first two quarters of 2005, except for a very small Danish trial fishery in the 2nd quarter of the year in the North Sea (ICES in press).

Ecosystem effects

Since the establishment of the Norway pout box no studies have been carried out on neither the effects of more selective fisheries technology and changed fleet behaviour, nor does the data exist that enables an evaluation of the Box and an analysis of the consequences of a partial or total reopening of the Box (Anon 1987).

Analyses of catch and bycatch data in the Danish Norway pout fishery inside and outside the Box 1975-1986:

- o The conclusion was that bycatch of each age group of whiting, haddock and herring depends on location, quarter, year class strength and year within the study period (Anon 1987).

Bycatch of whiting and haddock dominated in the Norway Pout fishery. Bycatch was shown to be correlated with introduced technical measures, including the Norway pout box and the introduction of the Common Fisheries Policy in 1983. However, changes in bycatch were shown to be linked to differences in yearly and seasonal distribution of Norway pout. Thus, it is difficult to separate area and seasonal effects. In addition, technological development in the industrial fisheries in this decade was not evaluated (Anon 1987).

A monitoring programme has been established in 2005.

Effects on fisheries effort / benefits

Fishing began in Northern North Sea using light high headline demersal trawl in the late 50's. In the mid 70's the maximum catch was 736.000 tons in 1974. Rapid increases in catch of Norway pout led to ICES establishing a work group on Norway Pout and sand eel in the North Sea. At meetings in 1977 and 1978 the ICES Advisory Committee found no clear need for any regulations on the exploitation of Norway pout (ICES 1979).

Norway pout is caught (for fish meal and fish oil) in small meshed trawls (16-31mm) in a mixed fishery with blue whiting. The blue whiting component in the catches has been relatively low in recent years, and the Norway pout fishery has become cleaner.

In addition to the directed Norway pout fishery, the species is also taken as by-catch in the blue whiting fishery.

The Norway pout TAC in the North Sea shared between Norway and EU (mainly Denmark). Official landings of Norway pout in ICES area Via (northern North Sea) has fluctuated between 2.000 and 14.000 tonnes for the last 10 years with an average of 7.700 tones (SWG 2005). In 2004 the proportion of the official landings of Norway pout landed by Norway in the North Sea was approximately 40%, while the EU (mainly Denmark) landed the remaining 60% (SWG 2005).

In 2005 the fishery was closed, and there has been no directed effort for Norway pout in the first two quarters of 2005, except for a very small Danish trial fishery in the 2nd quarter of the year in the North Sea (ICES in press).

The effects of the Norway pout box are unknown and not yet thoroughly evaluated. Earlier attempts have proven it impossible to differentiate the effects of the box from the effects of e.g. technological advances and selectivity of gear (Anon. 1987). The scientific basis for an evaluation of the effect of the box and the consequences of reopening the box does not exist (Hoffmann et al. 2004; Anon. 1987).

Since the establishment of the box there have been great changes in the industrial fisheries and stocks in the North Sea, i.e. a general reduction in by-catch of roundfish, including the Norway pout fishery. Reduction in bycatch exceeds decline in stock sizes of roundfish. This is partly due to altered behaviour of the fishery, which is related to higher levels of control and enforcement (DIFRES 2001).

Historically relevant studies relating to evaluations of the Norway pout Box include the EU Project "The consequences of increased North Sea herring, haddock and whiting abundances for the fishery for Norway pout in the North Sea" (Anon 1987).

Socio-economic effects on fisheries and other stakeholders

According to Raakjær Nielsen and Mathiesen (2002), Norway pout fishery was accused of having large by-catches of whiting and haddock. Danish fishers say this is more a question of political dispute over territorial fishing rights and not a measure to protect fish.

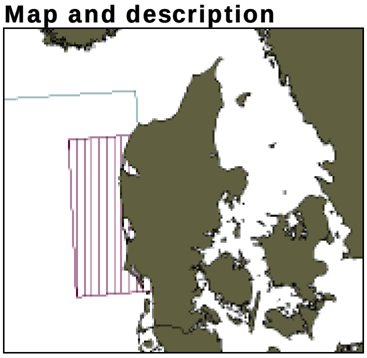
According to Raakjær Nielsen and Mathiesen (2002), the Spanish minister of fisheries in the debate on the reform of the Common Fisheries Policy proposed by the EC accused the Danish industrial fisheries of being unsustainable, all knowing that the hidden agenda was to get focus away from a huge reduction of the capacity of the Spanish fleet and for Spain to get access to the North Sea.

Lessons learned

Since the establishment of the Norway pout box no studies have been carried out on neither the effects of more selective fisheries technology and changed fleet behaviour, nor does the data exist that enables an evaluation of the Box and an analysis of the consequences of a partial or total reopening of the Box (Anon 1987).

The Sprat Closed Area / Box

Map and description



Map Source: Hoffmann et al. 2004.

Purpose of establishment

Although it is called the Sprat Closed Area, it was actually established to reduce mortality of juvenile (0-group) herring (*Clupea harengus*). Establishment of Sprat Box was expected to lead to a significant decrease in the levels of by-catch of juvenile (especially 0-group) herring in the entire ICES IVb-area (Hoffmann et al. 2004).

Selection methodology and design

Much sprat fishery in the box area led to a very large by-catch of juvenile herring. Random sampling showed that 90% of the herring by-catch took place within the current Sprat Box (Hoffmann et al. 2004).

Implementation process and legal aspects

Annual closure to industrial fishery from 1st July to 31st October (Hoffmann et al. 2004).

COUNCIL REGULATION (EC) No 850/98 of 30 March 1998 for the conservation of fishery resources through technical measures for the protection of juveniles of marine organisms

Article 21 Restrictions on fishing for sprat to protect herring

1. The retention on board of sprat which are caught within the geographical areas and during the periods mentioned below shall be prohibited:

...

(c) from 1 July to 31 October, within the geographical area bounded by the following coordinates:
- the west coast of Denmark at latitude 55° 30' N,
- latitude 55° 30' N, longitude 7° 00' E,
- latitude 57° 00' N, longitude 7° 00' E,
- the west coast of Denmark at latitude 57° 00' N.

2. However, vessels may retain on board quantities of sprat from any of the areas described, provided they do not exceed 5 % of the total live weight of the marine organisms on board which have been caught in each separate area during any of the periods specified. (Anon. 1998)

Ecosystem effects

Why the increase in 0-group herring by-catch in the 1990's after a drastic decrease in 1984? Hoffmann et al. (2004) present several hypotheses that have been discussed in various ICES working groups:

- Large cohorts of herring with more widespread distribution outside Box. However there is no consistent connection between *recruitment strength* and fisheries mortality of 0-group herring.
- Overall conclusion: No clear connection between establishment of Box and fisheries mortality of 0-group herring from the Box's establishment to 1996.

From 1996 there has been a reduction of by-catch. This coincides with the introduction of a limitation of herring by-catch in industrial fisheries.

Effects on fisheries effort / benefits

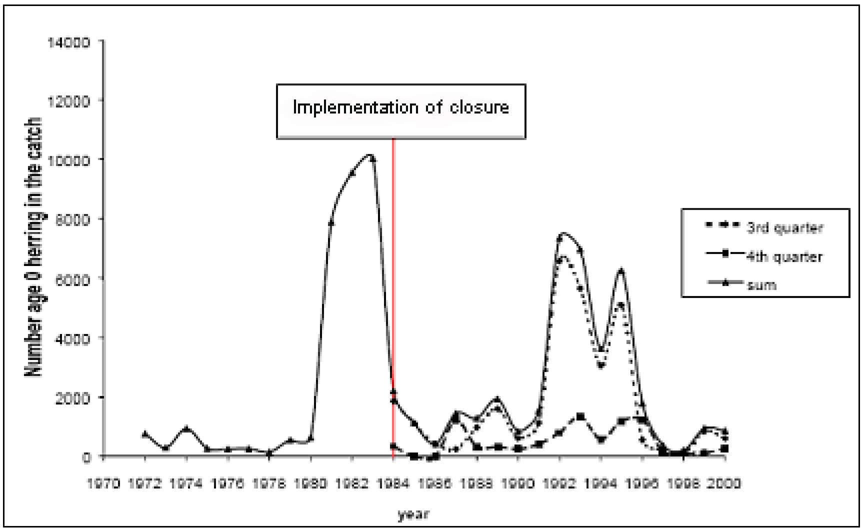


Figure modified from Hoffmann et al. 2004.

The figure clearly indicates that the expected decrease in 0-group herring by-catch could be detected directly after the establishment of the box in 1984. However, in the 1990's the by-catch of 0- group herring in the industrial fishery increases, especially in the 3rd quarter. In 1996 0-group herring by-catch decreases once again and continues to do so (Hoffmann et al. 2004).

Socio-economic effects on fisheries and other stakeholders

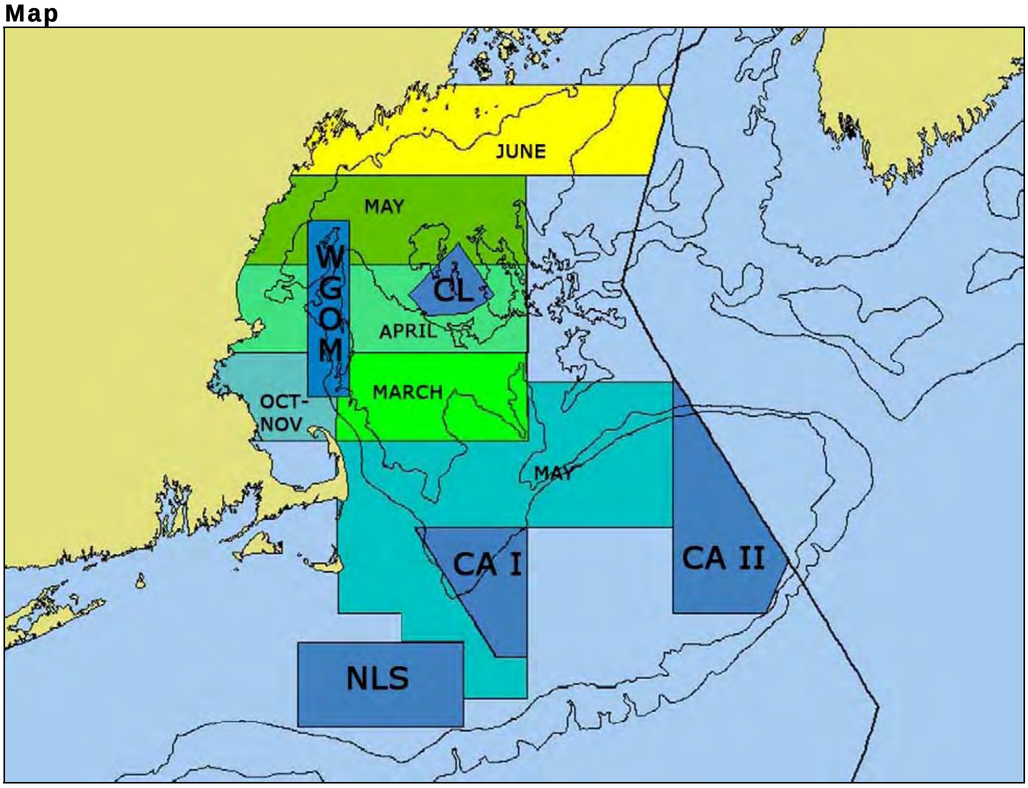
- no data

Lessons learned

In order to study the effects of the Box, we need more knowledge on the distribution of juvenile herring in the North Sea as well as better analyses of the composition of catches in industrial fisheries (Hoffmann et al. 2004).

Georges Bank Closures, NE US Atlantic

Map



Year-round and seasonal closed areas for groundfish protection off the northeast USA. Coding is: CA-I= closed area I,
CA-II =closed area II, NLS= Nantucket Lightship, WGOM= Western Gulf of Maine, CL= Cashes Ledge. Seasonal closure
boundaries are partially obscured by various months (Modified from Murawski et al. 2005).

Georges Bank is a shallow (3 to 150m depth) extension of the NE U.S. Atlantic continental shelf east of New England. Georges Bank covers approximately 40.000 km^2 (Collie et al. 1997). A mosaic of current closed areas consisting of more than 20.000 sq km. Georges Bank is one of the largest closed area systems in effect (Fogarty & Murawski 2004) and includes most of the productive fishing grounds for New England groundfish species (Murawski et al. 2005).

Purpose of establishment

Georges Bank had legendary fish stocks until the mid 20th century, where stocks declined steeply (Fogarty & Murawski 2004) and changes in fish community structure occurred, largely as a consequence of highly species-specific harvesting patterns driven by market considerations (Hall 2002).

Federal regulations established a number of year-round fishery closures on Georges Bank and adjacent areas in 1994 to help conserve and rebuild depleted stocks of flounders, gadoids, and other species regulated under the USA Magnuson-Stevens Fishery Conservation and Management Act (Murawski et al. 2000). All bottom tending fishing gear capable of catching demersal fishes were excluded, i.e. the closures were not designated specifically for habitat protection (Lindholm et al. 2004).

Selection methodology and design

In addition to the year-round closures there are seasonal or “rolling” closures that have been part of a groundfish management plan since the 1990s. These have multiple objectives, but are mainly implemented to limit exploitation on populations of Atlantic cod and harbour porpoise, which are taken as bycatch in demersal gillnets in the Gulf of Maine (Murawski et al. 2005)

The closed areas are good fishing grounds, including part of the scallop grounds of the region and important spawning grounds for, among others, cod, haddock and yellowtail flounder (Hall 2002). Sand/gravel areas that may be important to juvenile survivorship were also included (Hermsen et al. 2003).

Based on the centre points of various 10' squares, Murawski et al. (2005) calculated that 31% of the total trawl-fishing days at sea expended in New England waters during 1991-1993 were located within the "footprints" of the five year-round closed areas.

On Georges Bank, a key factor in larval dispersal is a well-established clockwise circulation pattern, or gyre, resulting from factors including local tidal forces and seafloor topography. The gyre creates a conduit that may allow eggs and larvae to self-seed closed areas, cross-seed other closed areas, and transport larvae to open areas. Analyses for scallop larvae indicate that the closed areas on Georges Bank can be self-sustaining and also contribute to recruitment into other areas (Fogarty & Murawski 2004).

Implementation process and legal aspects

Controls on mesh sizes, minimum fish sizes and seasonal closures failed to conserve stocks because there was no direct control on fishing effort.

Imposition in 1994 of year-round and seasonal groundfish closed areas off the NE USA (including Georges Bank) (evolved from seasonal closures in the 1970s)(Murawski et al. (2005):

- Five year-round closures: 3 southern areas Georges Bank Closed Areas I and II and Nantucket Lightship Area in Southern New England were closed year-round in 1994.
- Two additional areas were added in 1996 and 1998, respectively.
- In addition, nearshore, seasonal or "rolling" closures have been a part of the groundfish management plan since the 1990's.

Fishing by trawlers is not permitted in the closed areas. Since closure, the only gears that have been allowed in the reserves include lobster
traps, midwater trawls (for Atlantic herring), and some limited dredge fishing for sea scallops (Murawski et al. 2005; Fogarty & Murawski 2004).

Together with the establishment of closed areas, NOAA restricted numbers of days at sea (Fogarty & Murawski 2004

Ecosystem effects

The year-round closures have generated build-up of some, but not most, of the groundfish stocks within the boundaries of the closed areas (Murawski et al. 2005).

There is limited evidence for ''spill-over'' of biomass of harvestable sized animals from closed to open areas, for haddock, and yellowtail flounder, and a few other species (Murawski et al., 2004). The most compelling biological effects of the year-round closures on Georges Bank (Figure 1) have been for sessile animals, and in particular for populations of sea scallop (Murawski et al. 2005).

It is not easy to separate the effects of the fishery closure from reduction in days at sea. However, closures play an important role in the overall increase in abundance of stocks within the closed areas (Fogarty & Murawski 2004; Murawski et al. 2005):

- The biomass (total population weight) of a number of commercially important fish species has sharply increased, due to both an increase in the average size of individuals and, for some species, an increase in the number of young surviing to harvestable size.
- Some non-commercial species such as sculpin increased in biomass.
- Since 1993, haddock biomass has increased approximately eight-fold. 2005 stock assessments indicate that haddock will recover to near record levels in the next few years (Committee on Resources 2005).
- Yellowtail flounder populations have increased by over 800% since the establishment of the year-round closures.
- Scallop biomass increased 14-fold by 2001.

Georges Bank cod abundance is only 18 % more in 2005 than in 1994, while Gulf of Maine cod is about 50% more abundant than in 1994. Both stocks, however, declined in recent years (Committee on Resources 2005). However, the number of older fish in each stock has increased, and recent year classes of young fish are also increasing (Committee on Resources 2005).

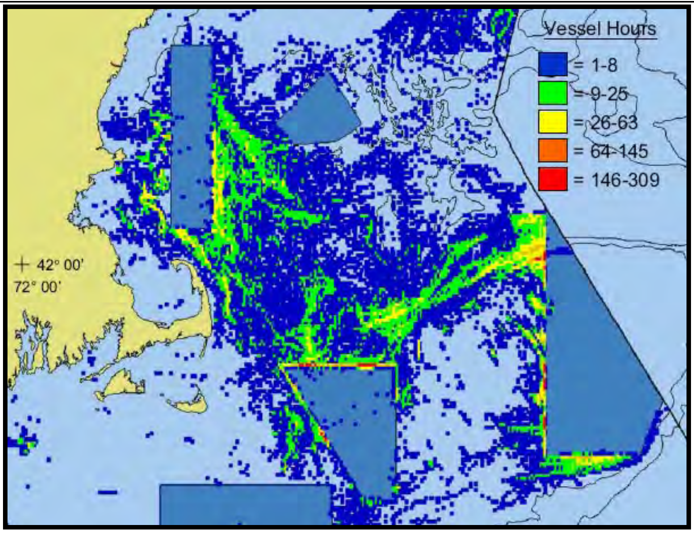
Benthic organisms and community structures re-emerged when areas were closed to trawling disturbance (e.g. Hermsen et al. 2003; Fogarty & Murawski 2004).

Effects on fisheries effort / benefits

Apparent spill-over of animals outside of the year-round closures is driven by a few valuable species (e.g. haddock and yellowtail flounder), and this differentially attracts some effort to the boundaries of three of the five closed areas (Murawski et al. 2005), i.e. large trawlers concentrate effort around the edges of closures (see figure) (Fogarty & Murawski 2004; Murawski et al. 2005).

Analyses confirm that large-scale year-round closed areas, in effect now for more than a decade, affect the abundance and spatial distribution of some target species, and the allocation of trawling effort (see figure below).

In 2001-2003 about 10% of effort targeting groundfish was deployed within 1 km of the MPA boundaries, and about 25% within 5 km. In addition, average revenue per hour trawled was about twice as high within 4 km of the boundary, than for more distant catches, but the catch variability was greater nearer closed area boundaries (Murawski et al. 2005).



Otter trawl fishing vessel effort off the northeast USA, 2003. Data were obtained from vessels using VMS (vessel monitoring systems) using satellite tracking. Locations are plotted only for vessel speeds ≤ 3.5 kn. Data are aggregated to 1' square (Source: Murawski et al. 2005).

Seasonal closed areas attracted more fishing effort after opening than prior to closure even while average CPUE was the same or lower (Murawski et al. 2005).

US part of Georges Bank, Gulf of Maine: closed areas have played a role in increased cod SSB, but so has increased mesh size, decreased vessel days at sea, quotas, etc. (FSBI 2001).

In the case of Georges Bank cod, fishing mortality has been cut in half since 2001 (Committee on Resources 2005).

The scallop fishery continues to generate increasing economic benefits to the US, providing a larger supply of scallops for consumers and higher revenues for fishermen at lower costs (Committee on Resources 2005). Landings from the sea scallop fishery in the Northeast increased to over 50 million pounds in 2003 and reached 60 million pounds in 2004, surpassing observed historic levels. In 1998 only 12 million pounds worth $87 million were landed, increasing steadily to over $300 million in 2004. The scallop fishery is a limited access fishery that has operated under a fishing permit moratorium since 1994 (NEFMC 2006; Committee on Resources 2005).

Lessons learned

Analyses confirm that large-scale year-round closed areas, in effect now for more than a decade, affect the abundance and spatial distribution of some target species, and the allocation of trawling effort (Murawski et al. 2005).

Closed areas benefited some species but not others. Spillover was observed for haddock, yellowtail and winter flounders (Fogarty & Murawski 2004). Large increases in sea scallop was an unintended effect.

The effects of a closed area will depend on factors such as seasonal movement patterns of fish and locations relative to fishing ports that will almost certainly vary from one fishery to the next (Holland 2000).

A variety of groundfish species are recovering and other benefits (e.g. abundant sea scallops) have followed fisheries closures of 4.000-7.000 km^2 , equivalent to square MPAs with a perimeter of appr. 60-80 km (Murawski et al. 2000; Roberts et al. 2005).

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PROTECT Case Study descriptions

Biological and ecological aspects of *Lophelia pertusa* relevant to the establishment of marine protected areas

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Introduction

Europe is home to some of the best examples of the cold-water coral habitat in the world. The principal framework constructing species, *Lophelia pertusa*, together with *Madrepora oculata*, form spectacular reefs along areas of the continental margins of Norway and Ireland. Intensive research of the reefs at many of these locations has begun to shed light on the biology of the species and the dynamics of the ecosystem as a whole. Much work still has to be done to understand the cold-water coral biotope in all it's variations. New information is constantly coming on stream, and new reef areas are being discovered every year (for example, see: ACES, 2003; Freiwald *et al.,* 2004; Freiwald and Roberts, 2005; ICES 2002, 2003, 2004, 2005). The EU 6FP Integrated Project, 'Hotspot Ecosystem Research on the Margins of European Seas' or HERMES (cf. http://www.eu-hermes.net/), will focus a major research effort on cold-water corals over the coming 4 years.

Access to new technologies, particularly Remotely Operated Vehicles, has enabled high resolution mapping and *in situ* sampling of the corals and associated fauna . Sadly, off Norway and Ireland destruction of coral habitat is already much in evidence. Trawling is estimated to have damaged 30 to 50% of known reefs off Norway (Fosså *et al*., 2002) and significantly impacted coral locations off the west coasts of Scotland (Wheeler *et al*., 2000) and Ireland (Hall Spencer *et al*., 2002; Grehan *et al.,* 2005). The degree to which coral habitat destruction impacts on the success of local fish stocks and thus fisheries has not been quantified, however, the well documented decline of the redfish longline fishery off Norway is a case in point (Fosså *et al*., 2002).

Concerns over further damage to corals has lead to the designation of a number of marine protected areas in Norway (Fosså *et al*., 2002) and prompted calls for the rapid implementation of European Union environmental regulations, specifically the EU Habitats Directive (Long and Grehan, 2002). Ireland along with other Member States such as the UK and Sweden are in the process of designating a number of sites as Special Areas of Conservation to protect deep-water coral reefs under the Habitats Directive.

The aim of this paper is to provide an overview of current information pertinent to the designation and management of marine protected areas.

Distribution of *Lophelia pertusa*

The azooanthelate scleractinian, *Lophelia pertusa*, is the predominant reef forming species in European waters. It is widely distributed globally (Fig. 1), but to date the best reef examples are found in European waters. This almost certainly reflects the intensity of research carried out in European waters in recent years, as many areas remain unexplored. *Lophelia pertusa* is found along the European margin at depths well below the photic zone except in some fjordic systems in Norway. *Lophelia* occurs where temperatures fall in the range of 4 to 13° and salinities are above 32 S.

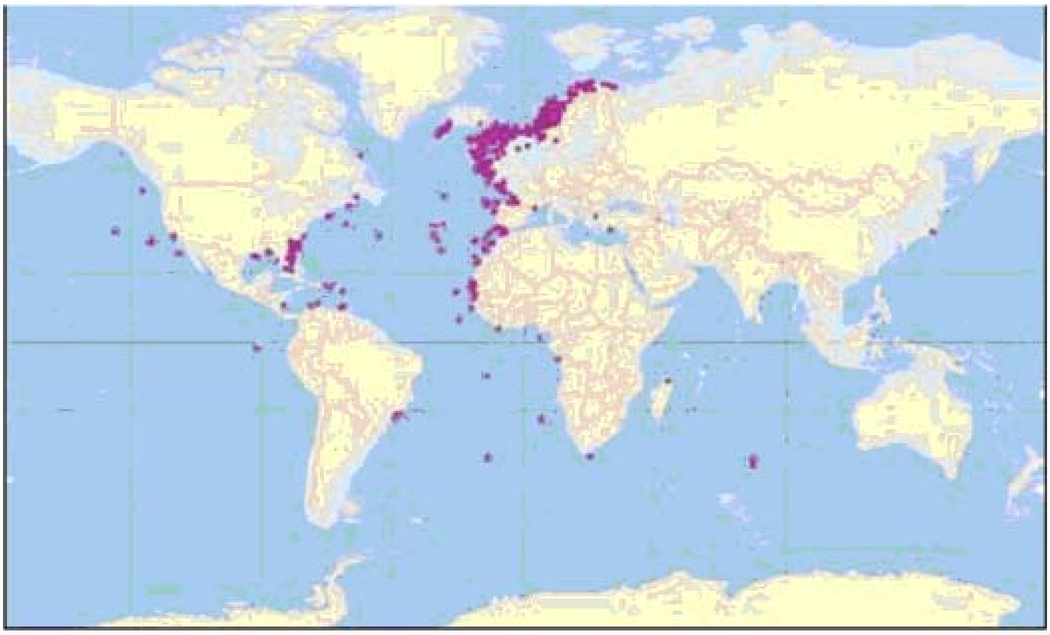

Figure 1 The global distribution of *Lophelia pertusa* (from Freiwald *et al.,* 2004).

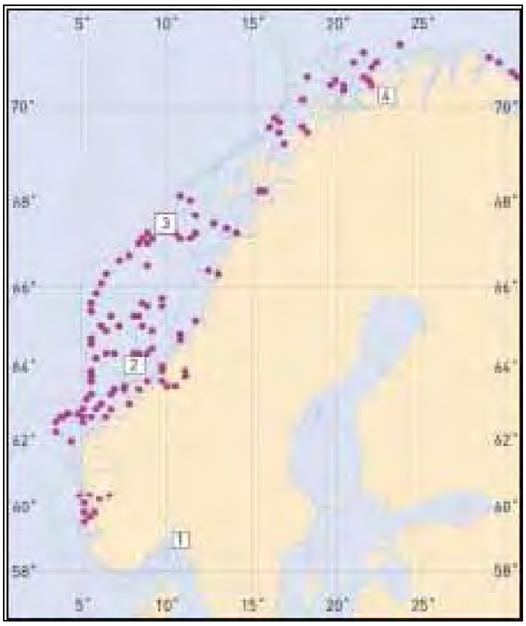

Fig. 2. *Lophelia* reefs on the Norwegian continental shelf and fjords. Boxes indicate locations of major reefs. 1: The eastern Skagerrak reefs. 2: The Sula Ridge. 3. The Rost Reef. 4. The Stjernsund Reef (from Freiwald et al., 2004).

To date, the highest density of *Lophelia* reefs, globally, are found in Norwegian waters (Fig. 2), generally at depths between 40 and 400 m (Fosså *et al*., 2002).

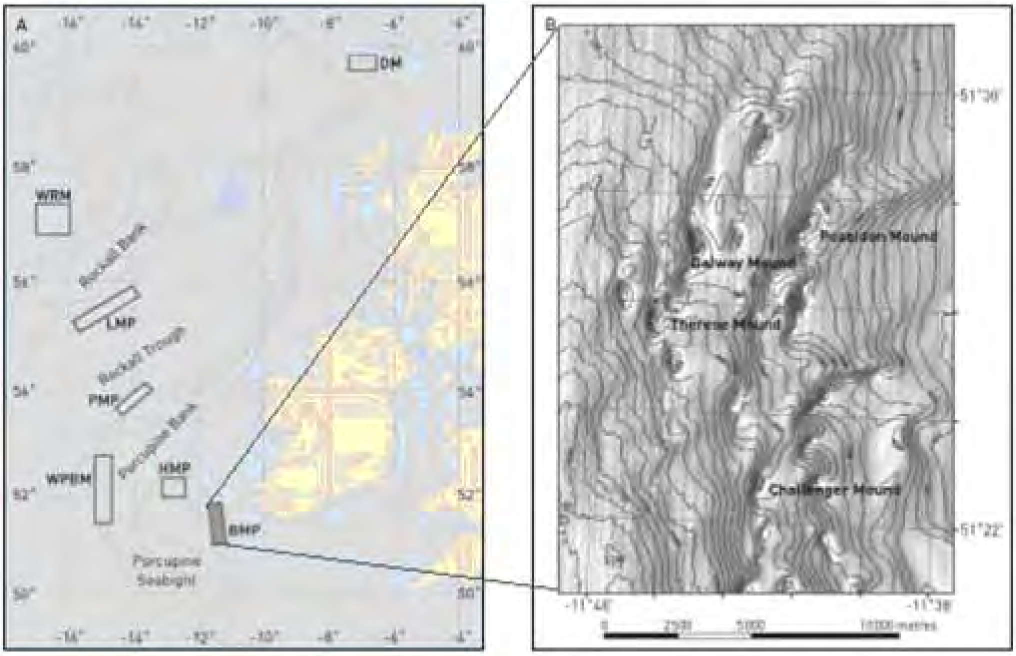

Fig. 3. (A) Major carbonate mound provinces off Ireland and the United Kingdom. BMP : Belgica Mound Province. DM: Darwin Mounds. HMP: Hovland Mound Province. LMP: Logachev Mound Province. PMP: Pelagia Mound Province. WRM: Western Rockall Mounds. WPBM: Western Porcupine Bank Mounds. (B) Shaded multibeam map of the Belgica Mound Province off Ireland. (from Freiwald et al., 2000).

Off the west coast of Ireland, *Lophelia* reefs are found between 500 and 1200m. The best developed reefs occur near or on the summits of giant carbonate mounds which rise up to 300 m above the seafloor. Carbonate mounds occur in clusters (called provinces) along the Irish continental margin (Fig. 3). Corals are also found covering mini-mounds, less than 5 m in height Fig. 4) in some areas.

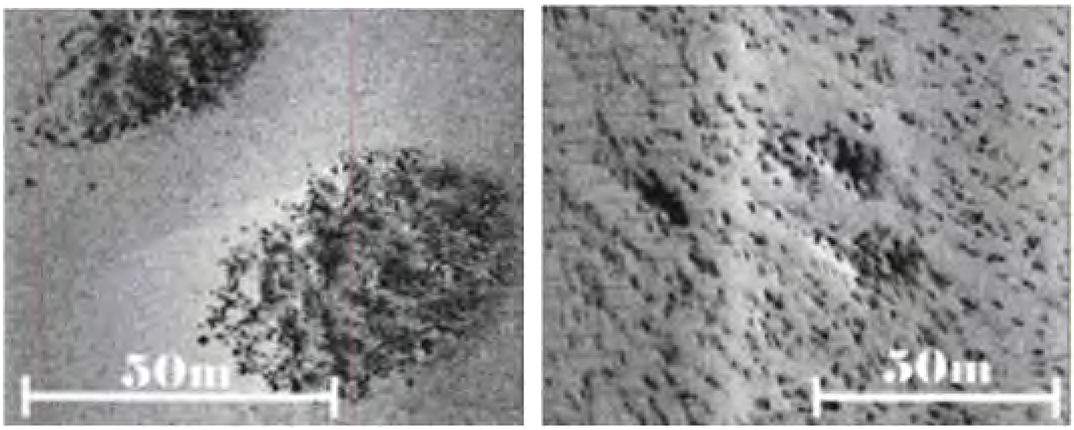

Fig. 4. A towed 410 KHz side-scan image of mini mounds approximately 5 m high showing dark speckled areas which correspond to live and dead coral colonies (from Wheeler et al. 2005).

Biology and Ecology

Coral Appearance

Corals need a hard substrate for settlement. Growth produces an individual bush like colony consisting of a hard calcium carbonate skeleton with numerous calyx which protect the delicate living coral polyps. Growth takes place by asexual budding which gives rise to a typically dense anastomosing branched structure (Fig. 5). Over time the coral colonies reach a critical size of about 1-2m in diameter, above which, the coral branches start to break off under their own weight and due to structural weakening by the activities of bio-eroders. These branches provide new substrate for settlement so that overtime the ground between individual colonies is filled in and reefs are formed. Living coral is typically white or pink with orange polyps (Fig. 5, 6) while dead coral is rendered brown due to colonisation by bacteria and fungi.

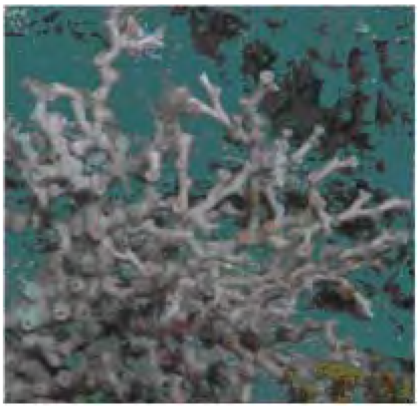

Fig. 5. Typical appearance of single colony (c. IFREMER).

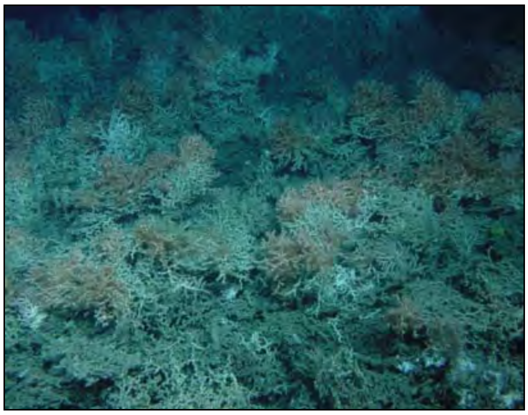

Fig. 6. Typical appearance of *Lophelia pertusa* reefs off Ireland. Image c. IFREMER.

Corals are found in areas of strong currents which provide a plentiful supply of food and also prevent clogging of the polyps by sedimentation. Reefs typically consist of living coral atop a dead coral framework (Fig. 6).

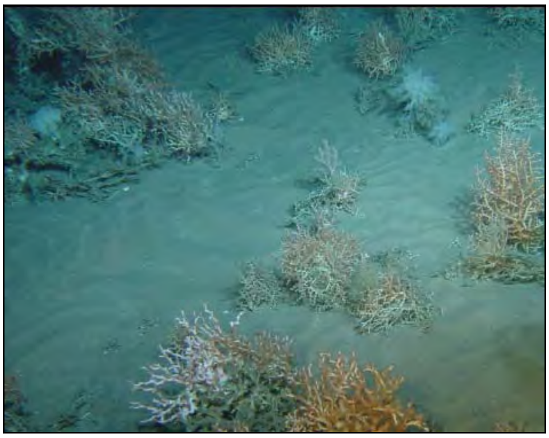

Fig. 7. Corals colonies on open sediments. (c. IFREMER).

The zone beneath living coral areas is often strewn with coarse to fine coral debris. Corals also occur as individual colonies on sand (Fig. 7), attached to drop-stones, and on other hard ground.

Feeding, growth, longevity

Cold-water corals, lacking zooan-thelae, actively feed on zooplankton and suspended particulate organic matter although this aspect of their biology is poorly studied. Growth rates are typically in the range of 5 to 25 mm per year (Freiwald *et al.,* 2004) indicating that habitat restoration if required will be an extremely slow process if possible at all.

Reproduction - larval dispersion capability

An important consideration in marine protected area design is whether a single area or a network are required to ensure an adequate reservoir of genetic material and reproductively viable individuals supplying larvae for reseeding purposes.

In the case of *Lophelia*, only preliminary information is available. According to Waller (2005), *L. pertusa* in the N.E. Atlantic maintains separate sexes (gonochoric), fertilization is external during winter months and resulting larvae are typically lecithotrophic. The genetic structure of *L. pertusa* has been studied by Le Goff and Rogers (2002), Le Goff-Vitry *et al.,* (2004) and Le Goff-Vitry and Rogers (2005). *Lophelia pertusa* does not form one panmictic population. There is significant genetic differentiation between subpopulations in fjords and those offshore. Along the continental margin, the genetic differentiation is moderate, suggesting sporadic, but not continuous, gene flow through larval dispersal over long periods of time. Significant degrees of inbreeding were detected at several sites indicating substantial proportions of self-recruitment within these subpopulations.

Associated fauna

A total of 1317 species were identified by the ACES consortium (Gage and Roberts, 2003), increasing the species inventory recorded on *Lophelia pertusa* reefs in the NE Atlantic from the 886 listed by Rogers (1999). The vast majority of these species are facultative associates of *L. pertusa* .To date there has been little detailed analysis of obligate associates of cold-water corals. However, the presence of coral rubble does appear to significantly enhance local faunal diversity (Gage and Roberts, 2003). Preliminary results indicate discernible variation in the composition of the coral associated fauna assemblages both at local and regional scales.

The nature of the functional relationship between coral assemblages and fish species has yet to be deciphered. It seems likely that cold-water coral habitat benefits fish stocks through increased food web complexity and the provision of refugia offering protection to spawning fish and nursery areas for juveniles. One study by Fosså *et al.,* (2002) shows higher abundance of redfish *Sebastes* spp. over *Lophelia* reefs than in adjacent non-reef areas.

Sensitivity and Vulnerability

Once protected areas have been delineated, the degree of management required to maintain the favourable ecological status of the habitat will require an assessment of the sensitivity and vulnerability of the habitat to impacts and the habitats potential for recovery after damage.

A variety of approaches to sensitivity coding and mapping have been developed (e.g. Holt *et al.,* 1995; Holt *et al.,* 1997; Cooke and MacMath, 1998; Hiscock, 1999), often with the development of oil spill contingency plans in mind (Dicks and Wright, 1989; Gundlach and Hayes, 1978; Michel and Dahlin, 1993; Anderson and Moore, 1997)
Accurate sensitivity coding is difficult for many species due to a lack of information concerning their specific response to impacts. This is made even more complicated when sensitivity coding is attempted for communities and biotopes. Given these constraints it is imperative that a clear understanding of the assumptions or limitations inherent in the assessment of sensitivity and recoverability is clearly understood prior to the application of such coding as a tool to improve decision making related to environmental protection and management issues (cf. www.marlin.co.uk).

The assessment process involves judging the sensitivity of a species or biotope to a change in an environmental factor caused by an external activity. The rationale then assesses the likely recoverability of the species or biotope following cessation of the activity. In addition, the likely effect on species richness of a change in an environmental factor is assessed for biotopes.

The following key definitions are used:
1. 'Biotope' the physical 'habitat' with its biological 'community'; a term which refers to the combination of physical environment (habitat) and its distinctive assemblage of conspicuous species. Marine Nature Conservation Review used the biotope concept to enable description and comparison.
2. 'Factor' a component of the physical, chemical, ecological or human environment that may be influenced by a natural events or anthropogenic activity. Therefore, activities effect the environment by perturbation of these factors.
3. 'Recoverability' is the ability of a habitat, community or species to return to a viable state which is at least close to that which existed before the development, activity or event took place. Recovery may be because of re-growth (in the case of damaged species capable of regrowing from remaining tissue), re-colonization by migration or larval settlement from undamaged populations or may require re-establishment of viability where, for instance, reproductive organs or propagules have been damaged by the event. Recovery can be partial or complete.
4. 'Sensitivity' is the intolerance of a habitat, community or individual (or individual colony) of a species to damage, or death, from an external factor. Sensitivity is assessed in terms of specific environmental perturbations.
5. 'Vulnerability' expresses the likelihood that a habitat, community or individual (or individual colony) of a species will be exposed to an external factor to which it is sensitive. Degree of 'vulnerability' therefore indicates the likely severity of damage should the factor occur at a defined intensity and/or frequency.

Sensitivity can only be estimated (assessed) in response to a change in a specific environmental factor and to the magnitude, duration, or frequency of that change (Hiscock et al., 1998). Standard benchmarks should be used as objective means to rank different levels of change in an environmental factor and to ensure species sensitivity is assessed with respect to the same level of change or perturbation.

The chosen benchmark levels of change in environmental factors are likely to affect different marine species to different degrees. Therefore, the benchmarks are considered precautionary in nature (*sensu* ' the precautionary approach'). Activities that result in incremental long-term change, such as climate change, are difficult to assess since the given level of change varies with time.
More information about the benchmarks used here can be found on the Marine Life Information Network website (www.marlin.ac.uk).

The sensitivity of Lophelia reefs to perturbation

The *MarLIN* assessment of the sensitivity of the *Lophelia* reefs (Tyler-Walters, 2003) can be viewed on the *MarLIN* website (www.marlin.ac.uk).

The review highlights a number of factors which could be expected to have a major impact on *Lophelia* reefs.

Physical Factors

- Substratum Loss

Sensitivity	Recoverability	Species Richness	Evidence/Confidence
High	Very Low	Major decline	High
Remarks: Removal of substratum would result in removal of living coral and dead coral debris, resulting in the destruction of the reef and loss of the biotope with recovery requiring several hundreds to thousands of years.			

- Increase in Temperature

Sensitivity	Recoverability	Species Richness	Evidence/Confidence
High	Very Low	Major decline	Low
Remarks: The requirement of *Lophelia* for oceanic waters suggests that *Lophelia* is probably sensitive to salinity and temperature change. The long term effects of climate change on deep-water currents could have far ranging effects. Therefore, a sensitivity of high has been recorded.			

- Decrease in Temperature

Sensitivity	Recoverability	Species Richness	Evidence/Confidence
High	Very Low	Major decline	Low
Remarks: In a recent study, Roberts *et al.* (2003) noted a strong correlation between the occurrence of *Lophelia* and temperature. With a single exception, *Lophelia* had not been recorded in waters colder than 4 °C and was absent from depths of greater than 500 m in the Faroe-Shetland Channel, presumably due to the influence of cold Nordic waters. Therefore, a sensitivity of high has been recorded.			

Abrasion and Physical Disturbance

Sensitivity	Recoverability	Species Richness	Evidence/Confidence
High	Very Low	Major decline	High
Remarks: Although *Lophelia* reefs occur at depth, they are likely to be subject to physical disturbance due to anchorage or positioning of offshore structures on the seabed but especially due to deep-sea trawling. Rogers (1999) suggested that trawling gear would break up the structure of the reef, fragment the reefs, and potentially result in complete disintegration of the coral matrix, and loss of the associated species. In a recent survey, Fosså *et al.* (2002) documented and photographed the damage caused to west Norwegian *Lophelia* reefs by trawling activity. Mechanical damage by fishing gear would also damage or kill the associated epifaunal species, and potentially turn over the coral rubble field, and modify the substratum (Rogers, 1999; Fosså *et al.*, 2002).. However, damage by long-line or gill net fisheries is probably of limited extent compared to bottom trawling. Overall, there seems to be significant evidence of damage to *Lophelia* and other cold-water coral reefs due to deep-sea trawling, and an overall sensitivity of high has been recorded. Recovery would probably take several hundreds to thousands of years.			

Chemical Factors

Factors considered here, were the likely impacts of exposure to contaminants such as synthetic compounds, heavy metals, hydrocarbons and radionuclides as well as changes in oxygenation, salinity and nutrient levels. *Lophelia* reefs were not thought to be highly sensitivity to any of these factors at the levels and duration of exposure specified in the benchmarks, although in all cases, assessment was hindered by lack of specific information relating to the response of *Lophelia* to chemical exposure.

Biological Factors

Factors considered here were the effect extraction of key or important species would have on the biotope. *Lophelia* reefs were considered to be sensitive to extraction activities.

- Extraction of important species .

Sensitivity	Recoverability	Species Richness	Evidence/Confidence
High	Very Low	Major decline	Low
Remarks: A sensitivity of high has been recorded due to evidence of documented damage to reefs by deep-sea trawling. Recovery would probably take several hundreds to thousands of years.			

Conclusions

Cold-water coral reefs predominantly formed by *Lophelia pertusa*, support a rich and diverse associated fauna. The cold-water coral biotope is likely be an essential habitat for some commercially important fish species during at least part of their life-cycle, although more work needs to be done to establish this relationship.

Corals are slow-growing, may reproduce sporadically and produce larvae with relatively limited powers of dispersion. The potential for habitat restoration after damage is therefore limited, indicating that conservation of pristine coral areas is desirable.

Genetic studies point to substantial proportions of self-recruitment and inbreeding in coral populations at local scales. Coral faunal assemblages also differ in composition and dominance at local and regional scales. This indicates that a network of marine protected areas rather than a single large conservation area will be required to conserve representive examples of *Lophelia* reefs in all it's manifestations.

At present the major activities likely to impact *Lophelia* reefs are: i) deep-sea fishing, particularly trawling, ii) oil and gas exploration, iii) bio-prospecting, iv) neighbouring aggregate extraction, v) scientific research, vi) laying of telecommunications cables and oil and gas pipelines. Climate change resulting in alteration of mass water circulation leading to temperature and salinity changes may in the future have catastrophic results for *Lophelia* reefs. This is supported by the punctual disappearance of *Lophelia* from the stratigraphic record in gravity cores taken from carbonate mounds in the Porcupine Seabight related to past climate change events , i.e. glaciation related changes in ocean circulation and mass water characteristics (Haas *et al.,* 2000).

The human activities listed above will primarily cause physical disturbance to reefs while climate change will cause temperature and salinity fluctuations. The *MarLIN* sensitivity coding highlights the sensitivity of *Lophelia* reefs to these types of impacts. While nothing can be done to manage climate change effects on the local scale, management action can be taken to prevent or mitigate the anthropogenic related physical damage to reefs. The activities which require immediate regulation in the vicinity of coral reefs are: i) deep-sea trawling, ii) scientific research, and iii) oil and gas exploration.

European cold-water coral habitat are of global importance and therefore warrant our best efforts to conserve pristine examples before we are obliged to attempt expensive and futile attempts at habitat restoration in areas already impacted.

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Baltic cod fishery closures (Baltic Cod boxes)

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Background

It is relatively well documented that the drastic decline of the eastern Baltic cod stock in the recent two decades has largely been caused by a combination of high fishing pressure and environmentally driven recruitment failure (e.g. MacKenzie et al. 2000; Köster et al. 2003). Decreased predation pressure by the cod stock, in combination with high reproductive success and relatively low fishing mortality, resulted in the second half of the 1990s in a drastically enlarged sprat stock in the Central Baltic Sea. Sprat predation is an important source of egg mortality for Baltic cod, eventually influencing its recruitment (Köster and Möllmann 2000). Moreover, the present sprat-dominated regime has had major ‘negative’ implications on lower trophic levels (e.g. Möllmann and Köster 2002). The reduced availability of meso- and macrozooplankton has negatively affected the condition, growth and potential recruitment of Central Baltic herring (e.g. Cardinale and Arrhenius 2000; Möllmann et al. 2003). The re-establishment of a more abundant cod stock in the Central Baltic could lead to a more stable ecosystem structure and more sustainable as well as economically sound fisheries.

Closures enforced in mid 1990s (“historical closures”)

In view of a rapid decline of the eastern Baltic cod stock in early 1990’s, two types of “Closures” were enforced in mid 1990s by the International Baltic Sea Fishery Commission (IBSFC) to preserve this stock. These closures were:

- A summer ban on targeted cod fishing was introduced in 1995 and is presently enforced from 15th April to 31st August (*note that the ban was shorter when established in 1995; since then the dates have had some variation*).

- A “spawning closure” for all fisheries from 15th May to 31st August in a relatively small area east of the island of Bornholm (in the Bornholm Basin).

Effects of closures - lessons learned

There is no published information on selection criteria, methodology and design principles of these closures. ICES Baltic Fisheries Assessment Working Group (ICES 1999) assessed the effects of these closures and concluded that the introduction of the summer ban had no significant positive impacts on the Baltic cod stock; this is mainly because the main cod catches in the Baltic Sea are taken from September to April, with in particular the trawl fishery exploiting pre-spawning concentrations of cod in late winter and spring. Similarly, the Working Group concluded that the relatively little “spawning” closure area east of Bornholm to protect the spawning stock has had little effect on stock (ICES 1999).

Clearly, a closure located in one small area is of limited use in enhancing spawning opportunities for a mobile fish such as cod because the reduction in catches is relatively easily compensated by increased catches in neighbouring areas and/or other seasons. It is noteworthy that in 2004 the ICES Study Group on Closed Spawning Areas of Eastern Baltic Cod (ICES 2004) stated that the closed area in the Bornholm Deep enforced in 1995-2003 was not large enough to ensure adequate coverage of potential areas with favourable hydrographic conditions. The

Group also stated that the extension of the closed area in the Bornholm Deep in 2004 is not likely to significantly increase the egg production (i.e., eggs surviving) because the spatial extension covers mainly the eastern slopes where under normal circumstances the hydrographic conditions are not favourable for egg survival and egg density is not particularly high.

Stricter closures enforced in 2005

Due to the lack of recovery of Baltic cod stocks and due to serious risk of stock collapse, new closures were enforced from 1.1.2005 by the EU (these closures are not binding for Russia). These closures were enforced mainly to reduce the overall

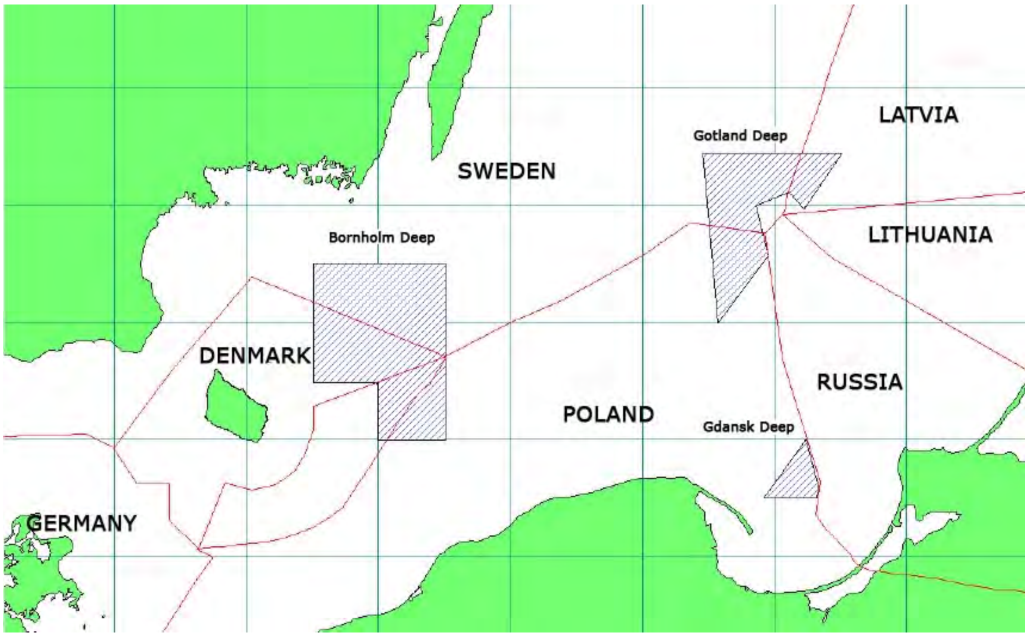

Figure 1. Three closed areas (Borholm, Gdansk and Gotland Deeps) for targeted cod fisheries in 2005 and 2006 (EU fleet). Map modified from Fiskeridirektoratet, www.fd.dk.

fishing mortality of Baltic cod but they also aimed to protect the spawning.

- Extended summer ban: Fishing for cod prohibited in Sub-divisions 25-32 (Central Baltic) from 1st May to 15th September.
- Spring ban (a new measure): Fishing for cod prohibited in Sub-divisions 22-24 (Western Baltic) from 1st March to 30th April.
- All cod fishing prohibited within three historical spawning areas in the Central Baltic (Fig. 1) for the entire year (EU fleet).

New regulations in 2006

New EU regulations relating to the three year-round closures (Fig. 1) were implemented on 1 January 2006. From the beginning of 2006 the areas are only closed during the spawning season of Baltic cod in the areas, i.e. from May 1 to October 31 2006.

In 2005 the three areas were totally closed to alle fisheries. In 2006, however, fishing for salmon with hooks or nets with mesh sizes larger than 157 mm is permitted year-round. In addition, vessels of lengths less than 12 meters using bottom nets with mesh sizes exceeding 110 mm are permitted to fish year-round, provided that bycatch of cod is less than 10% (www.danmarks-fiskeriforening.dk).

Effectiveness of the expanded closures

There is not much information of the efficiency and potential stock implications of the closures enforced from 1.1.2005. However, the assessment made by the ICES Study Group on Closed Spawning Areas of Eastern Baltic Cod (ICES 2004) helps us to predict some of the potential effects.

The ICES Study Group considered that an extended summer ban is an appropriate management measure in particular in the situation when there are improved spawning conditions. An appropriately timed fishery ban protects spawning without redirecting fishing effort towards juvenile cod. The Study Group, however, did not make any conclusions whether an extended summer ban would significantly help to recover the stock.

Regarding closed areas on the potential spawning areas, the Study Group states that the Bornholm Deep has been an important spawning area in all years whereas the Gdansk Deep and in particular the Gotland Deep have been important only in years where the salinity and oxygen conditions have allowed successful spawning, egg fertilisation and egg development, and when the spatial distribution of cod stock has included these areas (this has been the case in years with a large cod stock). Hence, a closure located in the deep water areas of the Bornholm Deep may help to protect the spawning fish and ensure undisturbed spawning. On the other hand, closures located in the more eastern part of the Central Baltic, for instance in the Gdansk Deep and in particular in the Gotland Deep, may have only a limited protection value at the current stock and hydrographic situation.

The Study Groups concluded that any closed area implemented to secure undisturbed cod spawning should cover areas and times of high egg survival, and should be large enough to cover the natural spatial variability of hydrological conditions. The Group, however, also stressed that even favourable hydrographic conditions and high egg production do not guarantee successful reproduction. The reproductive success of Baltic cod depends on many other processes that are affecting early life stages, such as egg and fry predation by clupeids, food availability, cannibalism by adult cod (e.g. Tomkiewicz et al. 1998; Uzars and Plikshs 2000; Hinrichsen et al. 2002a, 2002b; Kraus et al. 2002).

The Study Group further stated that mature cod appear to concentrate in areas of favourable hydrographic conditions for spawning; this implies a spawning migration into the Bornholm Basin when hydrographic conditions are unfavourable in the eastern spawning areas. However, the extent and eventual driving forces of these migrations are not yet clear.

The main spawning time of cod in the Central Baltic is currently from June to August, i.e. in the summer months. The Study Group states that very recently there may have been a slight shift back towards spring spawning (spawning is starting in May). A further shift in spawning time to earlier months of the year would have substantial implications for the design requirements of a closure. Pre-spawning concentrations of cod would start to gather earlier, increasing the catchability of cod in spring months in both the targeted fishery as well in the pelagic fishery (as by-catch).

The fact that the three new closures are enforced year-around, and not only the spring and summer months, was not considered by the Study Group. Neither did the Study Group assess any potential fisheries impacts (socio-economic effects) of these closures. Wider ecosystem effects have not been assessed yet. No information exists about the level of enforcement.

Concluding remarks

The poor status of the cod stock suggests that the present management regime is incapable of facilitating stock recovery. Thus, there is a need for more effective management tools, closures (or MPAs) being one obvious candidate. To be effective in reducing the overall fishing mortality on cod, closure(s) should be designed by taking into account the distribution and migration patterns of cod as well as the adaptive responses of fishing fleets. Baltic cod use separate locations and habitats for spawning, larval development, juvenile and adult feeding. Such complex life history requires a successful temporal and spatial linkage between these locations to integrate the whole life-cycle and produce abundant generations. Clearly, there are many open questions that the Baltic Case Study has to tackle and explore.

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Closure of the sandeel fishery in the Firth of Forth area

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DIFRES

During the last decade a sandeel fishery developed in the north-west North Sea, off the Firth of Forth. The landings from this fishery peaked at over 100,000t in 1993 and then subsequently fell. The Firth of Forth area is important for breeding seabirds and the removal of such large quantities of sandeels within their foraging range soon became a matter of concern. The U.K. called for a moratorium on sandeel fishing adjacent to seabird colonies along the U.K. coast and in response the EU requested advice from ICES. An ICES Study Group, was convened in 1999 in response to this request with two terms of reference (ICES 1999):

a) assess whether removal of sandeel by fisheries has a measurable effect on sandeel predators such as seabirds, marine mammals, and other fish species.

b) assess whether establishment of closed areas and seasons for sandeel fisheries could ameliorate any effects. Identify possible seasons/areas as specifically as possible.

The study group noted that there was suggestion of a negative effect of the Firth of Forth fishery on the sandeel stock in 1993 (subsequently published in Rindorf et al., 2000), which coincided with a particularly low breeding success of seabirds, especially kittiwakes. The study group concluded that there were two reasons for continued concern about this area. First, sandeels supported a number of potentially sensitive seabird colonies (Lloyd et al., 1991). Second, work on stock structure indicated that sandeels in this area are reproductively isolated from the main fished aggregations in the North Sea (area 3 in Wright et al. 1998). Consequently, as sandeel assessments are only conducted for the North Sea there was no reliable information on the state of the sandeel aggregation near the Firth of Forth. Given available information the study group proposed that kittiwake breeding success was the best practical indicator of sandeel availability at least to seabirds. Simulations using plausible values for population parameters of kittiwakes in the North Sea have indicated that kittiwake populations will decline with a breeding success of 0.5 fledged chicks per well-built nest, and increase with breeding success greater than 0.7 fledged chicks per well-built nest (Thompson et al. 1999). The Study Group therefore recommended using these values as thresholds to close and re-open, respectively, the sandeel fishery near the Firth of Forth. As breeding success of kittiwakes had declined to less than 0.5 fledged chicks per well-built nest the study group recommended that the sandeel fishery west of 1^{o} W near the Firth of Forth be closed. It was further recommended that during the period of closure a very limited commercial monitoring fishery should be conducted in order to maintain a time series of commercial CPUE and biological sampling data on sandeels in this area. ICES Advisory committees accepted the advice from the study group.

The EU agreed with ICES advice to close the fishery whilst maintaining a commercial monitoring. A 3 year closure, from 2000 to 2002, was decided and the Commission was requested to produce annual reports to the Council on the effects of the restrictions in the sandeel fishery in the Firth of Forth area. On the basis of these reports the commission can propose appropriate amendments to the limitations on the sandeel fishery in the area. The wording of the Act is stated in article 29a of: "Council Regulation (EC) no 850/98 of 30 March 1998 for the conservation of fishery resources through technical measures for the protection of juveniles of marine organisms". The area that was closed to the fishery for sandeels is shown in Figure 1. The regulation included a monitoring fishery where selected Danish sandeel vessels were allowed 10 fishing days in May and 10 days in June for the collection of information relevant to monitor sandeel population development following the closure. In 2003 the closing of the sandeel fishery off the Firth of Forth was prolonged until 2006, when an evaluation of the management measures will performed by the Commission. In the second period the number of fishing days was extended form 20 to 40.

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Shetland sandeel case study

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History of the Shetland sandeel stock

Around Shetland, sandeels are fished commercially on a number of small inshore grounds within 10 km of the coast (Fig. 1). The fishery at Shetland started in the early 1970s and peaked in 1982 when 52,000t were landed (Fig 2). Subsequently fishing effort declined as price differences made other fisheries more attractive. The fishery is seasonal, usually commencing in April or May when the catches consist of 1-group and older fish, with 0-group appearing in the catches from June onwards and forming the large majority of the catch by the end of the season, typically in August or September. These changes reflect the relative availability of these age-classes in the water column (Reeves, 1994). Up until 1988 the fishery was unrestricted, although a voluntary restriction on the permissible proportion of 0-group in the catch was introduced by the Shetland Fishermen's Association in 1987 (Goodlad, 1989).

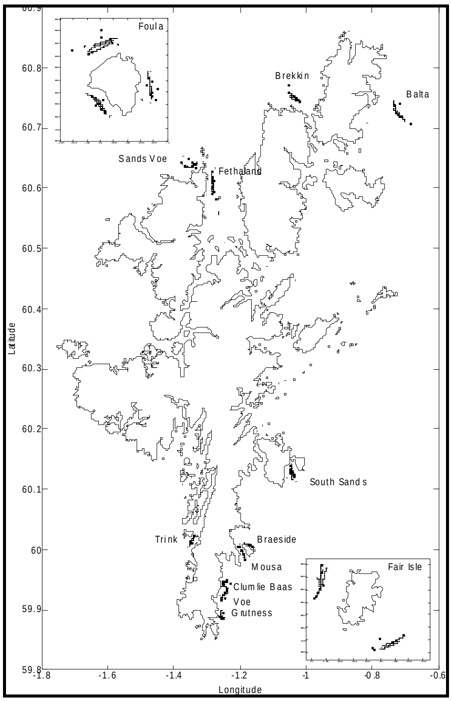

Figure 1 Distribution of fishing grounds in the Shetland sandeel stock.

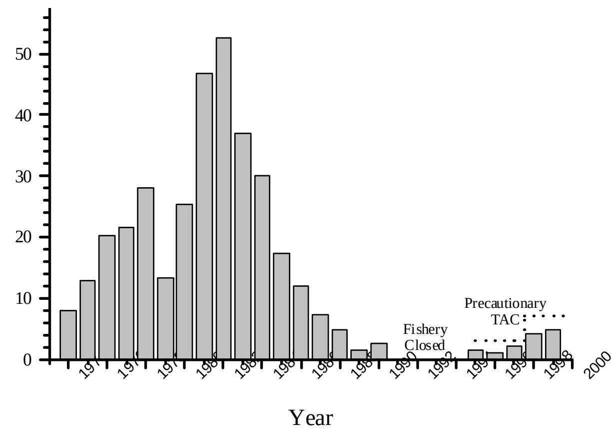

Fig. 2. Commercial sandeel catches in Shetland waters, 1974 to 1999. The fishery was closed from 1991 to 1994. Precautionary annual TACs set since 1995 are also shown

Shetland is home to some internationally and nationally important concentrations of breeding seabirds. During the 1980s there was a substantial reduction in the breeding success of a number of seabird species beginning with Arctic Tern (*Sterna paradisea*), from around 1984. It was clear that the poor breeding success of sandeels was largely due to the low availability of sandeels, particularly 0-group sandeels (Monaghan *et al*, 1989). At the time it was not clear what caused the low availability of sandeels, and the fishery was widely implicated as the cause.

At the time of the collapse in recruitment the sandeel aggregations at Shetland were regarded as a unit stock. Assessments of this stock showed a clear decrease in recruitment after 1982 consistent with the poor seabird breeding success over this period. However, as the decline in recruitment preceded a decline in spawning biomass, and fishing effort was decreasing in the fishery, the view of fishery scientists at the time was that the fishery was unlikely to be the cause of the recruitment decline. As a result, no management measures were implemented until 1989 when the fishery was closed from 1 July. This closure was introduced to reduce fishing mortality on the stock to help protect its spawning biomass at a time when it had been diminished by successive low recruitment. This seasonal closure was implemented again in 1990.

Following an increase in the abundance of the Shetland sandeel stock in the early 1990s the fishery was reopened in 1995 for a three year trial period and has remained open to the present time. When the fishery was reopened a new three-year management plan was agreed. It was intended that this would form the basis of a continuing triennial management programme based on an agreed consensus of local stakeholders and subject to regular review. At the time this was regarded as a significant step forward as it was first time in the UK that environmental organisations, fishermen and fisheries scientists had reached a consensus on management measures to permit a sustainable industrial fishery whilst protecting breeding seabirds, sandeel stocks and other wildlife. The partners in this initiative were the Shetland Fishermen's Association (SFA), the Royal Society for the Protection of Birds (RSPB), Scottish Natural Heritage (SNH), and the SOAEFD (now FRS) Marine Laboratory in Aberdeen (MLA). The agreement was brokered by the Scottish Office Agriculture, Environment and Fisheries Department (now the Scottish Executive Rural Affairs Department - SERAD). The management agreement is underpinned by scientific advice provided by ICES on the state of the Shetland sandeel stock.

The management plan for the years 1995 to 1997 included a precautionary TAC of 3,000 tonnes per annum, and a fixed fishing season of April to June. The fishery was restricted to vessels of under 20 metres in length, all of which had to obtain a specific sandeel licence. A small fishery took place annually under this management plan although catches never actually reached the 3,000 tonne TAC. There was dissatisfaction with the management plan, however, both from local fishermen and from conservation organisations. The fishermen wanted a larger TAC to be set, and also wanted to be allowed to continue fishing later in the year. The conservation organisations were reluctant for the TAC to be increased and wanted the fishery closed during the main sea-bird breeding season (June and July) to avoid any possible conflicts between the fishery and sea-birds.

Following the first triennial review a revised management plan was agreed in march 1998 to cover the fishery during the period 1998 to 2000. The revised management package had four main measures: an increase in the TAC from 3,000 to 7,000 tonnes a seasonal closure during June and July, reopening on 1st August, provided sufficient TAC remained uncaught, a raising of the 20 metre length restriction if it became apparent that the full TAC was unlikely to be taken and management of the fishery under this plan was delegated to the Shetland Fish Producer's Organisation (SFPO). Overall, the revised plan was seen as a satisfactory balance between the interests of all concerned parties.

Despite the precautionary TAC and local management measures the stock collapsed again in the last five years, beginning with very low recruitment in 2000 (Figure 3, next page). In 2003, after a third year of low recruitment FRS and ICES did advise that the local management regime of a 3 year TAC should be re-examined but no action was taken. Overall, it appears that despite a highly precautionary approach to management the stock is liable to collapse.

Understanding of the Shetland sandeel stock

Concern over the continuing breeding failure of Shetland seabird led to meetings in

Aberdeen in September 1988 and Lerwick in October 1988 (Heubeck 1989) to discuss the problem and identify research priorities. These resulted in a directed research project on the biology of sandeels in the vicinity of seabird colonies at Shetland, which started in 1990 (Wright and Bailey, 1993). A key result from this project was that the sandeel aggregations around Shetland appeared to be part of a larger, more widely distributed complex of aggregations. This hypothesis of a sandeel metapopulation has since been supported by further research (Wright, 1996; Proctor et al., 1998). Spawning aggregations around Orkney are much more productive than those at Shetland (Wright & Bailey, 1996) and larvae and juveniles from this area are frequently transported into Shetland waters. As such the Shetland fishing grounds may be a net sink within the larger meta-population. Evidence from observations on 0-group distributions and plankton (Wright, 1996) together with model simulations of larval transport (Proctor et al., 1998) indicated that sea circulation was unfavourable to the transport of young sandeels into Shetland waters during the period of low recruitment in the 1980s. This trend was reversed in 1991. As such recruitment and hence stock abundance in Shetland waters appears largely dependent on oceanographic conditions. This may explain why, after a few years of good recruitment, there was a protracted period of low recruitment leading to a second collapse in the last few years. As such management actions at the scale of the Shetland stock region may be relatively unimportant to the local sustainability of the Shetland 'stock'.

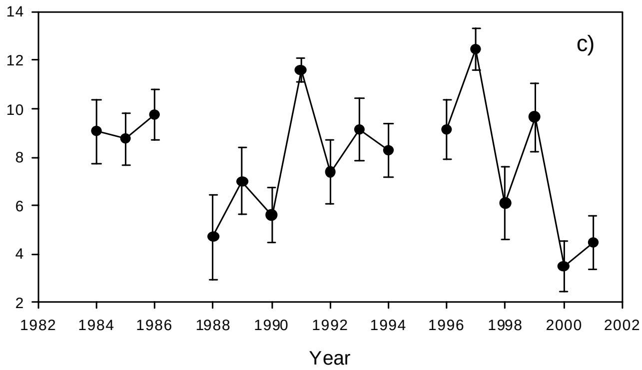

Figure 3 Changes in 0-group sandeel abundance in Shetland waters between 1984 and 2001. Values for ln mean abundance (±SE) derived from 30 min research bottom trawl hauls from 16 fishing grounds surveyed in July-August.

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How do we judge whether an MPA has been 'successful' or not?

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Of the 1,306 MPAs surveyed world-wide by Kelleher et al. (1995) only 31% were thought to be fully achieving their management objectives. How do we judge whether or not an MPA has been 'successful'? What do we need to know in order to assess MPA performance?

Marine protected areas are established for a wide range of purposes, including protecting marine species and habitats, conserving marine biodiversity, restoring fisheries stocks, managing tourism activities, and minimizing conflicts among resource users. To achieve these goals, specific and measurable objectives must be defined in terms of what outputs and outcomes are being sought. This in-turn requires that well-defined management plans be developed, <u>measures of MPA success be identified and defined in advance</u>, impacts of management actions be monitored and evaluated, and that the results of these activities be fed back into the planning process to revise objectives, plans and outcomes (Pomeroy et al. 2004), i.e. 'adaptive management'.

The process of goal setting is closely linked to stakeholder expectations, MPA design, and the establishment of criteria to evaluate the progress made in meeting those objectives (Agardy, 2000). If goals are not well articulated, it is difficult to define criteria to measure progress or to identify and quantify the indicators of progress (Kay & Alder, 1999).

A **'goal'** is a broad statement of what the MPA is ultimately trying to achieve, i.e. why was the MPA created and what are the main aspirations

An **'objective'** is a more specific measurable statement of what must be accomplished to attain the related goal. Attaining a goal is typically associated with the achievement of two or more corresponding objectives. A useful objective (Margolius & Salafsky, 1998) is one that is:

- specific and easily understood,
- written in terms of what will be accomplished, not how to go about it,
- realistically achievable,
- defined within a limited time period, and
- achieved by being measured and validated.

Monitoring is an integral component of marine area management; it provides the data required to evaluate changes in marine ecosystems as a result of the implementation process. These evaluations are essential for determining effectiveness, improving design, and providing progress reports to stakeholders (Houde *et al.* 2001).

Some of the key questions that should be addressed through monitoring include: (1) Does the MPA regime meet its goals and why or why not? (2) Have there been unanticipated consequences? (3) Are the size and location of reserves within the MPA optimal?
The management criteria and monitoring systems put in place for an MPA are case specific. However, analysis of the effects of any MPA is likely to require certain

fundamental knowledge of fisheries and ecosystems independent of the specific case. Common data-types are likely to be a common feature (e.g. baseline information from before and after the MPA was established), and such data can be analysed using a standardized suite of methodologies.

Four categories of information may be included in a monitoring program: (1) structure of marine communities (abundance, age structure, species diversity, and spatial distribution); (2) habitat maintenance or recovery; (3) indicators of water quality or environmental degradation (e.g., pollutants, nutrient levels, siltation); and (4) socioeconomic attributes and impacts (Houde *et al.* 2001).

There are two approaches to analysing the impacts of MPAs on living resources (Houde *et al.* 2001). In the first approach, changes within the MPA are evaluated temporally such that conditions are documented before the implementation and then compared to conditions following implementation (**before vs. after**). A limitation of this approach is that environmental variation in the years before and after the establishment of the MPA may obscure trends resulting from protection. For instance, variable recruitment in a fishery due to a change in oceanic conditions may affect, either positively or negatively, the apparent recovery of a stock after closure of an area. In Kenyan reefs, a twofold increase in fish abundance was observed in surveys of both unprotected and protected sites (McClanahan, 1995); hence, the change was independent of the MPA. A further example is provided by the North Sea ‘Plaice box’ (see chapter on “past and present MPAs”).

System ‘carrying capacity’ will vary with temperature (e.g. productivity) and habitat type, but is often thought of as the total biomass of all components within the ‘virgin’ ecosystem (see Jennings & Blanchard 2004). Knowledge about the ‘virgin’ state of an ecosystem is often poor. However, such information is often required in order to establish baselines against which current or future levels of impact can be compared (Steel & Schumacher 2000), without suffering the problem of ‘shifting baseline syndrome’, i.e. when a baseline is set with a short-term perspective and represents an increasingly exploited state over time (see Pauly 1995). Jennings & Blanchard (2004) point out that the unexploited biomass of a community (the ‘carrying capacity’) is not necessarily the same as the historically observed state, because climate has also changed over time. Indeed it is unlikely that ecosystems today would always revert to historic levels if fishing were stopped, either because phase-shifts have occurred or because the environment is fundamentally different from that existing prior to human exploitation.

In the second approach, changes in the MPA are evaluated spatially such that conditions inside the MPA are compared to conditions in a similar area outside (**inside vs. outside**). The limitation of this approach is that MPAs often encompass unique habitats and are set up because the area is distinctive or ‘special’ in the first place; hence, there are few situations in which comparison areas accurately represent the features found within the MPA. A further alternative would be to use a ‘spectrum’ of sites with different (quantified) levels of fishing pressure, to look for trends and correlations rather than a simple ‘pairwise’ comparison (inside vs. outside). This approach has been adopted in the North Sea, Fiji, and in the Seychelles by Jennings et al. (2001, 1995) and Jennings & Polunin (1996).

The ideal experimental design, to test conclusively whether MPAs have a particular ecological effect relative to their original goals, would involve monitoring regimes at multiple localities that include surveys before and after MPA establishment. Ideally, survey methods should be rigorous enough to detect a 10-25% change in biomass, density, or species numbers (Pomeroy et al. 2004). In many cases, however, such quantitative rigor is difficult to achieve.

A recent paper by Maxwell & Jennings (2005) set out to explore the power of a large-scale annual monitoring programme (the English North Sea bottom trawl survey) to detect decline and/or recovery of species that are vulnerable to fishing. Even though this survey was one of the largest and best resourced trawl surveys in the north-east Atlantic, the power to detect declines in abundance of vulnerable and rare species (elasmobranchs, cod etc.) on time scales of <10 years was low. Furthermore, the study showed that if conservation measures were effective, and vulnerable populations recovered at maximum potential rate, 5-10 years of monitoring would often be required to detect recovery.

Unfortunately, many surveys and monitoring schemes are established with no prior assessment of power, and others are used to study species that were not their original focus. This is increasingly the case given the recent focus on the integration of conservation concerns into fisheries management. Fisheries surveys are often the only source of time-series distribution and abundance data for species in offshore waters (Maxwell & Jennings 2005). Nicholson & Jennings (2004) tested the power of the North Sea International Bottom Trawl Survey (IBTS) to detect trends in six community metrics: (mean length, mean weight, mean maximum length, mean maximum, weight, slope of the biomass size spectrum, and mean trophic level). The authors demonstrated that the power of the trawl survey to detect trends at the community level is generally poor. While community metrics do provide good long-term indicators of changes in fish community structure, it is argued that they are unlikely to provide an appropriate tool to support short-term management decisions, for example to judge the success of MPAs. Similar concerns have been raised by Nicholson & Fryer (1992), Fryer & Nicholson (1993) and Gerrodette (1987).

It is important to note that different species will respond to protection in different ways, and at differing rates. Comparisons of 'before vs. after' and 'inside vs. outside', need to take such factors into account. Small species typically have higher growth rates, mature earlier, and have higher intrinsic rates of population increase (Jennings et al. 1999). Hence we would anticipate a more rapid response to protection in these species. Badalamenti et al. (2002) examined the response of three fish species following a trawl ban in the Gulf of Castellamare, Sicily. The largest and most sedentary of the three fish species (*Lophius budegassa*) exhibited the smallest numerical increase following the trawl ban. This species is known to mature later and at a greater size in comparison with *Mullus barbatus* and *Merluccius merluccius*, which exhibited remarkable numerical increases once protected from fishing (within 5 years).

Sometimes 'outside vs. inside' type comparisons do not yield significant differences because the species concerned are highly migratory and frequently cross the MPA boundaries. Differences are more likely to occur where species are less motile and site-attached (e.g Russ and Alcala 1998; Murawski et al. 2000).

The changing role of research and monitoring programmes

The most useful input of science in the planning phase of an MPA is to help define management issues, why there are problems, and how they should be addressed. The first task of natural scientists is to supply objective data to support or challenge perceptions of resource depletion/degradation or risk. A key role of science is to isolate the causes of the problem and help eradicate misconceptions and prejudices, so that management can then focus on real solutions. Baselines and monitoring of natural conditions should be in place before the implementation stage, so that an assessment can be made of whether the programme's objectives are being met or not. In theory, many technologies, e.g. GIS and remote sensing, are available at the planning phase, but their use is likely to be limited by a lack of time, money and data availability (Pomeroy et al. 2004).

As the MPA programme matures, the role of science evolves from identifying issues to developing the technologies needed to support management and to understanding the results of research, and monitoring. Reporting on success in management is very important; so is reporting on setbacks and failures. The results from monitoring should be used to adapt management, so that management actions have the intended effects in the long-term. Typically such work requires a long-term commitment to data collection, management and analysis. Ideally, monitoring and research should be supported by long-term funding as part of the core management of the MPA. Often a data set extending over many decades is needed to understand the significance of human impacts as compared to the natural impacts and processes which underpin the functioning of an ecosystem. In the interim, caution should be applied in interpreting results (Pomeroy et al. 2004).

It is important to continually update and refine the management programme on the basis of the results of monitoring. This step has been omitted or performed superficially in most MPAs. Yet, if MPAs are to be ecologically and socially sustainable, almost continuous evaluation and learning is essential. Evaluation must address two broad questions:

a) What has been accomplished by the MPA and learned from its successes and failures?
b) How has the context (e.g. environment, governance) changed since the programme was initiated?

A meaningful evaluation can be conducted only if the MPA objectives were stated in clear terms and if indicators for assessing progress were identified in the planning phase, and monitored afterwards. Baseline data are essential. Many evaluations yield ambiguous results because these preconditions for assessing performance do not exist. Natural and social scientists have important roles to play in evaluation. In particular, they should assess the relevance, reliability and cost-effectiveness of scientific information generated by research and monitoring, and advise on the suitability of control data (Pomeroy et al. 2004).

Few methods have been developed to evaluate the effectiveness of MPA management (Kelleher, Bleakley, & Wells, 1995; Alder, 1996; Hockey & Branch, 1997). Most of these studies investigated whether designated MPAs were transformed from "paper parks" to functional management systems. For example, Hockey and Branch (1997) proposed broad criteria to measure the scientific, practical, socioeconomic, and legal performance of MPAs against the management objectives. Some of their criteria are difficult to score because they included several factors such as education, recreation, tourism, and research in a single criterion (Alder et al 2002).

Choosing and using indicators

There are hundreds to thousands of potential indicators of ecosystem status that can be used for management. They range in complexity from single-species indicators to 'emergent properties' of ecosystem models (Rice 2003).

To be useful for management, indicators should be:
- Relatively easy to understand by non-scientists and other users;
- Sensitive to a manageable human activity;
- Relatively tightly linked in space and time to that activity;
- Easily and accurately measured, with a low error rate;
- Measurable over the area where they may be used,
- and based on existing time-series data to help set reference points

In 2000 the IUCN together with the World Wide Fund for Nature, formed the *MPA Management Effectiveness Initiative (MPA-MEI)*. This programme had four main objectives:

1. to develop a set of natural and socio-economic indicators to evaluate MPA management effectiveness,
2. to develop a process for conducting an MPA evaluation – in the form of an easy-to-use guidebook,
3. to ground-truth and field-test the guidebook and indicator methods, and
4. to encourage uptake.

The MPA-MEI programme conducted a survey of MPA goals and objectives from around the world, and categorized these into three broad types: biophysical, socio-economic and governance. 130 'indicators' were investigated and mapped to relevant MPA goals and objectives. Operational descriptions and definitions were subsequently provided for 44 indicators as well as a detailed narrative of methods of measurement and guidance on analysis/interpretation of results (Pomeroy et al. 2004;
see www.effectivempa.noaa.gov/guidebook/guidebook.html).

Pomeroy et al. (2004) provide a useful tool which could be applied within PROTECT for devising hypothetical monitoring programmes under each of the three case-studies and matching indicators, to the aims and objectives of the MPAs concerned. Biophysical (natural) goals of MPAs are considered to fall into 5 broad and distinct categories. Those associated with maintaining/protecting resources and hence yields in the future, MPAs aimed at protecting individual species, MPAs aimed at maintaining/protecting vulnerable habitats and those established with the aim of restoring already degraded areas. The three case-studies being considered under PROTECT fall within this overall framework (one focuses on an MPA to protect/maintain seabirds, one focuses on an MPA to protect vulnerable deep sea habitats, on focuses on an MPA to potentially increase/restore fishery yields in the Baltic).

Not all indicators will be appropriate for use in every MPA and case-study. Some indicators require a higher level of skill, labour, financing and time to measure than others.

In PROTECT we are mainly concerned with biophysical indicators since these are the ones of primary interest to scientists. Regardless of their many social benefits, MPAs are ultimately a tool for conserving or restoring the biophysical conditions of oceans and coasts. In most cases the link between the biological state of the marine environment and the livelihoods, income and food security of the people who use and depend upon the resource is explicit. It then follows that beyond characterizing natural systems, the measurements of biophysical indicators can also be useful when viewed in the context of the socio-economic and governance conditions that operate in and around the MPA (Pomeroy et al. 2004).

On the other hand, experience shows that social, cultural, economic and political factors can shape the development, management and performance of MPAs more than biological or physical factors (Fiske 1992, Kelleher & Recchia 1998). Understanding the socio-economic context of stakeholders involved with and/or influenced by the MPA is essential for assessing, predicting and managing MPAs. The use of socio-economic indicators allows MPA managers to: (a) incorporate and monitor stakeholder group concerns and interests into the management process; (b) determine the impacts of management decisions on the stakeholders; and (c) demonstrate the value of the MPA to the public and decision-makers (Pomeroy et al. 2004).

Modelling approaches to judge the 'success' of MPAs

Monitoring and evaluation of MPA management performance is beginning to receive attention and several analytical approaches are emerging, from complex strategic comparisons of MPAs using multidimensional scaling (e.g. Alder et al 2002) to park-specific programs (e.g. Hockings, 2000).

A recent study by Alder et al. (2002) considered management effectiveness in 20 MPAs located in different regions of the World. This work was based on an ordination method known as 'Rapfish'. The development of the Rapfish approach is detailed in Pitcher et al. (1998), and it has now been used elsewhere to evaluate the sustainability of fisheries throughout the North Atlantic (Alder et al., 2000).

Rapfish uses a multidisciplinary appraisal technique based on a number of easy-to-score attributes (Pitcher & Preikshot, 2001). The attributes within five evaluation fields (ecological, economic, social, technological, and ethical) are chosen and defined to reflect the notion of sustainability. Rapfish was modified for MPA use based on the following considerations: Any measure of management effectiveness must be pragmatic so that policy and decision makers can readily understand what is being measured and apply its relevance in MPA management. Similarly, the cost of collecting and analyzing the information needed to evaluate management effectiveness must be small compared to the market and non-market value of the MPA and the cost of managing the area.

Twenty-two MPA managers and researchers tested the approach by scoring MPAs in which they were presently or recently working. These managers and researchers were considered experts in the areas they scored, and they based their scores on reports or studies with which they were familiar. The analysis provided an overall comparison of northern and southern hemisphere MPAs based on the average score in each evaluation field. In this particular case (Figure 1), northern hemisphere MPAs scored better for ecosystem management objectives compared to southern hemisphere MPAs. Southern hemisphere MPAs, however, scored better for meeting social objectives than northern hemisphere areas (Alder et al. 2002).

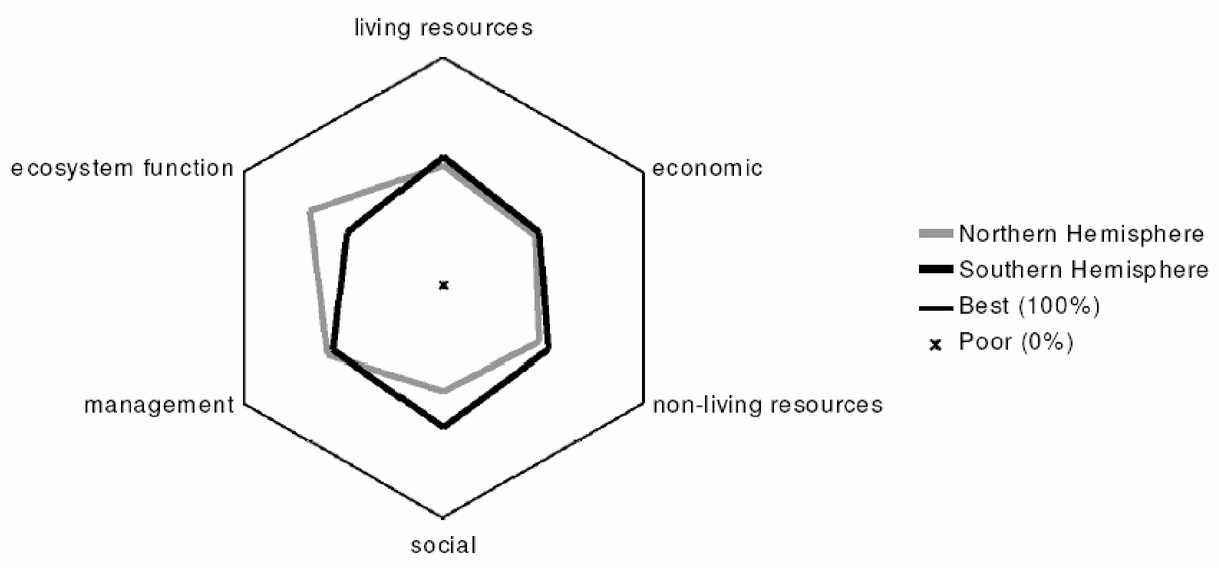

Figure 1. Composite kite diagram of the average scores in each evaluation field for all MPAs evaluated, grouped by northern and southern hemispheres (from Alder et al. 2002).

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MPA monitoring strategies

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In temperate waters relatively few examples of the successful implementation of Marine Protected areas (MPAs), in terms of a partial or total protection from fishing or other adverse influences, exist (Willis et al., 2003, Murawski et al., 2000). In these systems some are temporal closures of areas for targeted fishing in the North Sea, e.g., the Norway pout box, the herring box or the Plaice box. However, in none of the Marine Nature Reserves or Areas of Conservation in the North Sea all extractive activities are prohibited. So far, evidence for positive effects on targeted stocks by closed areas are sparse with much more information existing regarding tropical ecosystems, where mainly sedentary species are conserved (e.g. Duval et al., 2004; Maliao et al., 2004, Pomeroy et al., 2004). In order to achieve a beneficial effect on reproductive output of highly mobile and migrating fish stocks, the size of MPAs needs to be large (Bohnsack, 2000; Parrish, 1999; Walters, 2000) making it difficult to find a compromise solution to make the MPA acceptable to stakeholders.

In recent years MPAs were increasingly considered as management tool for fisheries rather than as a pure conservation tool for species, biodiversity or regional ecosystems (Houde, 2001; ICES, 2004; Nowlis, 2000; Parrish, 1999; Roberts et al., 2005). To evaluate changes in stock development and ecosystem structure as a result of the implementation of protected zones, monitoring programmes are an integral component of MPA and fisheries management. In addition, they are essential to determine the efficacy of conservation tools, to improve MPA design and provide progress information to stakeholders, funding agencies and civil societies.

Research and monitoring in and around reserves will benefit by (1) increasing the understanding of the effects of reserves in ecological and socio-economic terms; (2) improving the knowledge base of complex marine ecosystems and of the ways human activities affect these systems; and (3) developing and applying marine management methods, which achieve the specific goals cost-effectively (Houde, 2001).

What is monitoring?

Monitoring is the “process of repeated observation, for specified purposes, of one or more elements of the environment, according to prearranged schedules in space and time and using comparable collection methods” (Meijers, 1986). In other words, data and information about MPAs and surrounding areas are gathered, partly on a regular basis, over an extended period of time. Through monitoring, some key questions about the effectiveness of MPAs should be addressed (Houde, 2001):

- Does the MPA system meet its goals and why or why not?
- Are there unanticipated consequences?
- Are the size and location of reserves within the MPA optimal?

To answer these questions case specific indicators and success criteria need to be defined that fulfil several conditions, but should be first easily and accurately measurable (previous chapter).

Monitoring is an essential component in the planning and implementation process of MPAs. It provides the data that feeds back into the management cycle enabling the evaluation of the status and development of protected zones (Fig. 1). Four different aspects are addressed in monitoring programmes: (1) condition of the biological resources of the site/ system; (2) condition of the cultural resources of the site/ system; (3) socio-economic aspects; (4) impacts of the site/ system's management on local communities (Hockings et al., 2000). PROTECT will focus on biophysical monitoring of MPAs, i.e. the assessment of attributes measuring the effectiveness of MPAs to protect sensitive species, habitats and ecosystems from primarily the adverse affect of fishing. Over a longer period, MPAs are supposed to assist in the sustainable harvesting of economically valuable species and a successful implementation would therefore be additionally measurable in improved economics, e.g. by higher CPUEs and economic yield. Biophysical indicators can also serve as socio-economic measure. As an example, fish can be thought of in financial terms.

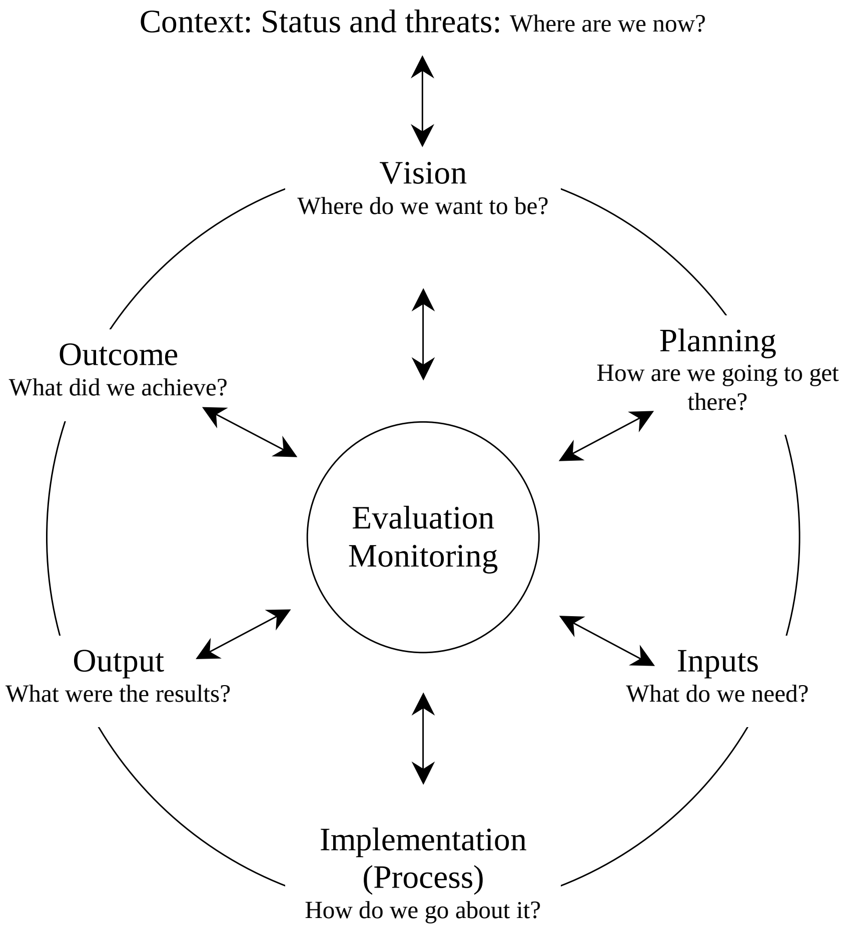

Figure 1: The management cycle (redrawn from Hockings et al., 2000)

Developing MPA monitoring programmes

The development of a monitoring programme for outcome evaluation of MPAs follows a stepwise process (Fig. 2): The objectives that should be achieved by the establishment of the protected area determine the indicators and success criteria (General objectives for management are specified in the IUCN *Guidelines for Protected Area Management Categories* (IUCN, 1994)). For these indicators data needs are defined and reviewed for efficiency. This information forms the basis for the design of a monitoring programme that should be implemented in cooperation with managers and stakeholders. The results of a long-term monitoring programme need to be periodically assessed and the status of the MPA should be made available to researchers, managers, stakeholders and the public by regular reporting and publications (Hockings, 1998, Jones, 2000).

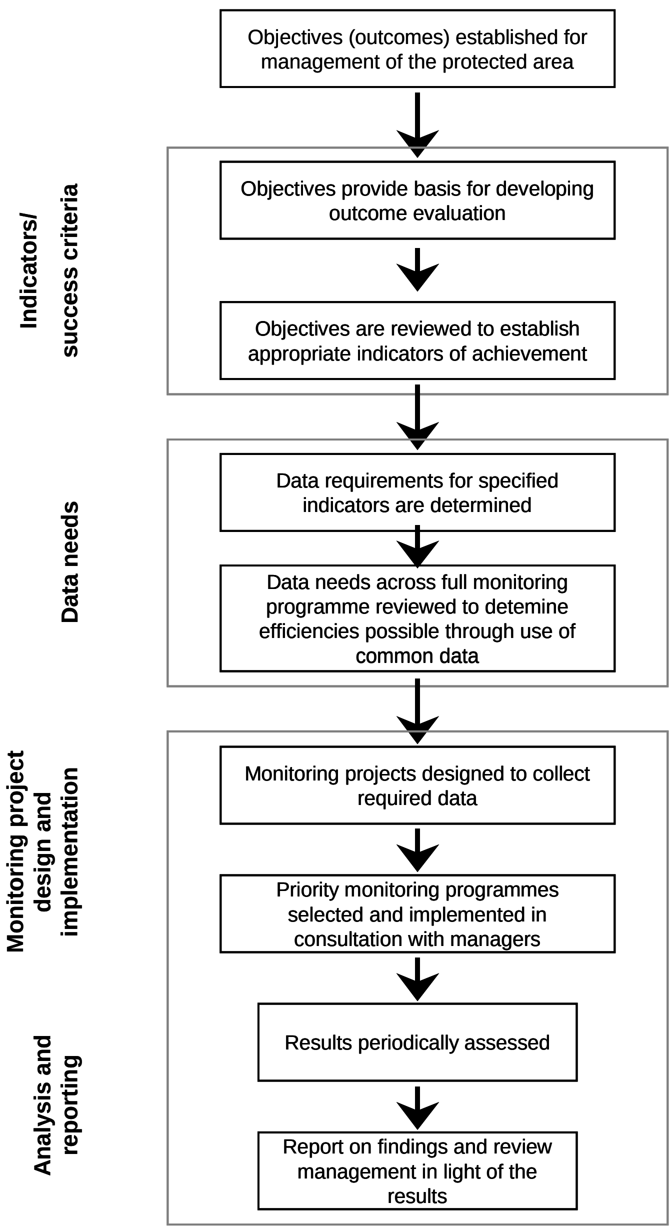

Figure 2: Process used to develop an outcome-monitoring programme (adapted from Hockings (1998) and Jones (2000).

Approaches to MPA monitoring

Although monitoring programmes need to be flexible and have to accommodate to different needs, there are general principles that apply to the way in which all monitoring activities of protected areas should be conducted. Within the programme design issues, the appropriateness of management systems and processes, and the achievement of protected area objectives are evaluated (Hockings et al., 2000).

Empirical approaches to impact assessment – Monitoring programmes entail the measurement of certain ecosystem properties and characteristics as well as socio-economic conditions. The effectiveness of MPAs for ecosystems and/or living resources can be primarily evaluated in two different ways (Houde, 2001): In the first approach, changes of key parameters within the reserve (and ideally in adjacent areas) are measured repeatedly over time and conditions are documented before and after the implementation of a closure. In the second approach, changes are evaluated spatially such that conditions inside the reserve are compared to conditions in similar habitats beyond it. Both approaches have limitations, although the use of a spatial evaluation is specifically difficult, as biophysical conditions within the reserve are typically not mimicked in reference areas. Therefore, the first approach has been more widely applied (see Wilkinson et al., 2003). A design to evaluate impact from empirical studies considering both approaches has been popularised by Stewart-Oaten et al. (1986) and is referred to as the Before-After-Control-Impact paired model (BACIP). Similar to the suggestions by Houde (2001) sampling needs to be initiated prior to the establishment of the reserve. However, the impact area is not only investigated before and after the implementation of the MPA, but is also paired to a control site. In a BACIP design, sites are measured repeatedly and each site pair-by-time combination is treated as a unit (Smith, 2002).

Measuring of indicators – Attributes included in a monitoring programme will depend on the goals established for the MPA and the main environmental variability the ecosystem experiences. In general, the monitoring strategy and criteria measured should follow principles of simplicity, low cost, and general applicability that allow comparisons to other areas. In the temporal evaluation of changes, a regular sampling that is sensitive to life history characteristics of monitored species follows the development from the initial status of the MPA (fish stock structure and size, ecosystem structure, socio-economic elements of fishery) to e.g. a recovery of the target species or communities. In the past, in reserves focussing on enhancing or preserving commercially important species, the assessment of the target populations has been the major or exclusive objective of monitoring programmes (e.g. Keogh et al., 1993; Attwood et al., 1997; Jennings and Polunin, 1997). This, however, has to be supplemented by further, fishery-independent studies, including aspects of population structure, biodiversity, species specific abundances, distribution patterns, and species interactions. The need for extended monitoring is indicated by studies showing strong ecological shifts after the protection of specific organisms (e.g. Babcock et al., 1999).

Furthermore, environmental influences as well as other management tools (e.g. fishing effort reduction, gear changes etc.) have to be taken into account when the MPA is evaluated in terms of its usefulness to fish stock protection and recovery. To assess the impact of fisheries on communities, long-term and large-scale studies covering also unfished areas are necessary. The existence of unexploited control areas provide a benchmark against which the impacts of extractive activities can be evaluated. Such reference areas can additionally assist in the understanding of long-term changes and natural-environmental variability. As an example, environmental conditions and the fluctuation of major predator and prey species play a significant role in the recruitment variability of several species as e.g. cod in the Baltic (Köster et al., 2001a, b), and basic knowledge about the ecosystem dynamic is therefore essential.

Fishery-related monitoring strategies

Fishery reserves are implemented to improve the management and to rebuild stocks of commercially important fish species that are otherwise threatened by intensive fishing (Bohnsack, 1998; Mosquera et al., 2000; Roberts, 2005). They provide buffers against the uncertainties of fisheries data, implementation of regulations and ecosystem dynamics (Perry et al., 1999). As for all kind of MPAs the intensity of monitoring will depend on many factors related to the objectives of the reserve and the logistical constraints. However, fishery-dependent data and other monitoring programmes related to fisheries management can supplement the information from targeted monitoring programmes (Tab. 2). The monitoring of catch values, amounts of effort directed at target species, and costs of fishing in the immediate vicinity of protected areas will additionally provide the knowledge to determine the social and economic benefits and costs that emerge from the implementation of reserves.

Fishery-dependent monitoring methods – Fisheries displacement is one important measure in MPA monitoring programmes. Although impacts on the ecosystem due to spatially displaced trawling effort are supposed to be low for small reserves, they become much more significant when large portions of habitat are closed (Bohnsack, 2000). So far there is only limited evidence of a “spill-over” of biomass of harvestable sized demersal fish species from closed to open areas, but recent studies found a significant density effect and concurrent to this a fishing effort concentration in the immediate vicinity to protected areas in the Georges Bank region (Murawski et al., 2005). Monitoring of MPAs should, thus, investigate in how far a spill-over of biomass from the closed area exists and if fishing effort has become displaced, i.e. if it concentrates at the boundaries of the MPA. Effort distribution should be not only studied after the set-up of the MPA, to assess effort displacement and the resulting influences on the ecosystem, but also prior to the implementation of the reserve, to determine how human activities might get influenced in the future.

Fishery-dependent data on catch (methods, catch composition, discard rates, biological characteristics of the catch) and effort can be collected from log-book surveillance, dockside monitoring, at-sea observers, satellite tracking, aerial surveillance, and patrol vessels. The European Community has established a framework for the collection and management of the data needed to conduct the common fisheries policy (EC No. 1543/2000, 1639/2001), which should lead to a better co-ordination and co-operation of data collection activities in Member States. The framework defines minimum requirements of programmemes (sampling intensity, precision levels) and facilitates the transmission of data to international organisations. Mandatory data collected under various control regulations mainly comprise fishing capacity, effort and catch, whereas other data like discard, CPUE, length and age composition or biological parameters are not systematically measured. Until recently, the quantity and quality of data collected by the Member States was regulated, but explicit methods, how these goals should be accomplished, were rarely defined. Only since January 2004 obligations to meet the provisions are effective and amendments have been made (EC No. 1581/2004) that give the legal background e.g. for the acceptance of on-board observers on all kinds of fishing vessels to control fisheries and allow unbiased sampling.

Paper-logbooks are carried by EU fishing vessels to record details of catch, effort and landings. As the processing of data is time-consuming, the feasibility of on-board electronic logbooks is currently tested in the on-going EU-Project SHEEL. In the future, this might give us the opportunity to use logbook information directly and it will enable a real-time assessment of fisheries activities nearby MPAs. So far, the variability of logbook data due to misreporting, suspect positions and lack of independent verification usually requires that fishing locations need to be aggregated into larger areas of traditionally 10' squares. As the concentration of fishing activity adjacent to MPAs can be very localized and might occur only within a distance of less than one up to five kilometres (Murawski et al., 2005), other more accurate methods need to supplement or even replace the logbook reporting.

Observer data usually give a much finer spatial resolution of fishing activity compared to self-reported logbooks. Furthermore, they provide the information base from which to undertake assessments of the effects of fishing and the changes in the biological characteristics inside and outside the boundaries of (partially) closed areas. Many countries routinely require vessels to carry independent observers (see e.g. NAFO observer programme, NAFO FC Doc. 05/1 Serial No. N5070) but the EU was until recently an exception to this. In the future, at-sea observers on fishing vessels targeting stocks outside closed areas should assess catch information including the bycatch and discard rates. These activities laid down in Commission Regulation No. 1581/2004 would be not only essential for a successful fisheries management but also for the monitoring of MPAs implemented within European waters. Dockside checks of catches would supplement the available information and would include smaller vessels that are not yet or only seldom controlled.

Precise vessel locations that can be integrated with logbook and observer records are measurable by the satellite-based Vessel Monitoring System (VMS). VMS-data indicating trawling effort are readily available (Mills et al., 2004), but they are still limited to certain vessel categories. Since January 2000, the European Community requires fishing vessels over 24 metres overall length to be included in a VMS programme, and this became expanded to vessels exceeding 15 metres in January 2005 (EC No 2371/2002). However, fishing vessels operating exclusively inside the baselines of Member States are not subject to this requirement. Therefore, for near-shore MPAs other methods like aerial surveillance still need to be used to monitor fishing activity.

Fishery-independent surveys – Fishery-dependent data, such as observer data can provide information on catch composition and species density gradients within open areas but lack comparable information from the MPA. In particular, density differentials between open and closed areas remain unknown. Fishery-independent scientific surveys thus need to supplement the information from commercial fisheries. For evaluation, sampling needs to be initiated at the proposed MPA site and an independent control site prior to the establishment of the reserve. The effectiveness of protection is then testable using the previously mentioned paired Before-After-Control-Impact (BACIP) design.

Each sampling programme will have a very specific design related to the most urgent questions that should be answered and often can cover only small spatial or temporal scales. Their realization and laboratory analyses are cost- and labour-intensive and therefore, the intensity of such surveys is generally limited. Where applicable, established sampling programmes for fisheries management like egg, larval fish and trawl surveys (e.g. International Bottom Trawl Survey (IBTS) in the North Sea, Baltic International Trawl Survey (BITS) in the Baltic) can give additional information to monitoring programmes and extend the datasets collected. Oceanographic, fishery, and biological data are for example partly accessible through the ICES data centre (http://www.ices.dk/datacentre/data_intro.asp) and time-series of different aspects of the ecosystem are available by various research institutions, partly participating in the PROTECT-project.

Table. 1: Features of fishery-dependent and -independent monitoring of MPAs (including suggestions by Houde, 2001)

Type of monitoring	**Potential indicators**	**Data collection**
Fishery-dependent strategy	Stock structure (size and age structure), catch composition	Fisheries statistics, at-sea observers, dockside checks of landings
	CPUE	Fisheries statistics, at-sea observers
	Distribution of target and key species	Fisheries statistics (CPUE), at-sea observers
	Effort distribution/ displacement	Vessel Monitoring Systems, aerial surveillance, log book analysis
	Other fishery-related variables (see Pope, 1988; Shepherd, 1988)	Fisheries statistics, at-sea observers, dockside checks of landings
Fishery-independent strategy	Ecosystem structure	Biological sampling at different trophic levels (benthos and pelagic)
	Biodiversity	Biological sampling of communities
	Status of target species as well as key predator and prey species	Measurements of abundance, size and age of relevant species
	Larval retention and dispersal	Measurements of oceanic current patterns, modelling studies (passive particle tracking), measurement of larval dispersal
	Genetic connectivity among populations, stock monitoring	Molecular analysis including early life stages, tracing where a fish was caught by DNA-analysis
	Emigration or immigration into reserves	Abundance and recruitment measurements in protected and non-protected areas; tagging and drift experiments, studies on population mixture and segregation
	Habitat distribution and complexity	Measurement of biological (e.g. coral density) and physical (e.g. water quality, T, S) environmental conditions
	Water quality	Physical measurements

Resource needs

Monitoring programmes for MPAs require a thorough planning phase, where resources needed for measuring success criteria and evaluating the effectiveness of the MPA should be estimated. The work plan needs to be adapted to logistical constrains and its budget should be calculated. If possible, MPA monitoring should be included in existing programmes, using current knowledge, expertise and technical equipment. Table 2 gives a coarse outline what aspects need to be taken into account during the planning phase of the programme (partly following suggestions by Pomeroy et al., 2004).

Table. 2: Resources potentially required for monitoring programmes

Resource needs	**Specifications**
Background knowledge	- ecosystem dynamics - environmental variability - systematics of species - recent and historic fish stock development
Available datasets	- Commercial fishing data (landing statistics, CPUE etc.) - Existing surveys (e.g. IBTS) - Fishing effort distribution (e.g. by Vessel Monitoring Systems)
Equipment	- biological sampling gears (e.g. plankton nets, trawls, *in-situ* video systems) - physical probes - software/ analytical tools (e.g. GIS)
Infrastructure	- ship time - location (head office, regional office)
Human resources	- scientific personnel providing ecological, taxonomic, fisheries or socio-economic expertise, involvement of research institutions, training of staff - technical personnel to operate gears and collect data - stakeholder involvement - volunteers and amateur naturalists (providing e.g. data on species counts from amateur fishing)
Budget	- costs for evaluation time (e.g. personnel costs) - training, consultant costs - cost for equipment and infrastructure

Example: Distribution of monitoring effort in PROTECT case studies
Three case studies of different MPA designs in temperate waters will be investigated within Protect. The focal species, objectives and other characteristics of each case study require different monitoring strategies to evaluate the effectiveness. Specifically, the monitoring effort within the MPA and adjacent areas will be distributed differently (Fig. 3). In the Central Baltic Sea, where spawning aggregations and/or nursery areas of cod will be protected, effectiveness will be measurable by the stock structure of cod and the development of the ecosystem in the entire area and less within the protected zone. The monitoring programme thus needs to cover the core distribution area of the population. In contrast to this, the success of a marine reserve for deep-water corals will be mainly measurable within the reserve and hardly in surrounding areas, where fishing activities will be allowed. Each species has a different diffusion potential with respect to a closure. A spill-over from protected deep-water coral habitats can only happen by larval stages which means that fishing effort in respect to the focal species need not to be monitored in adjacent areas of the MPA. For mobile fish species other monitoring strategies need to be developed. Adult sandeels are resident to certain banks and a spill-over is thus only expected in the immediate vicinity to the MPA, which makes a high spatial resolution of fishery-dependent measures necessary. Cod is a highly mobile fish species and, in the scenario of protecting spawning aggregations, the

fishing activities along the boundaries of the seasonal closure can be expected to increase considerably, which means that the surveillance of fishing effort would be one very important aspect in the development of monitoring strategies.

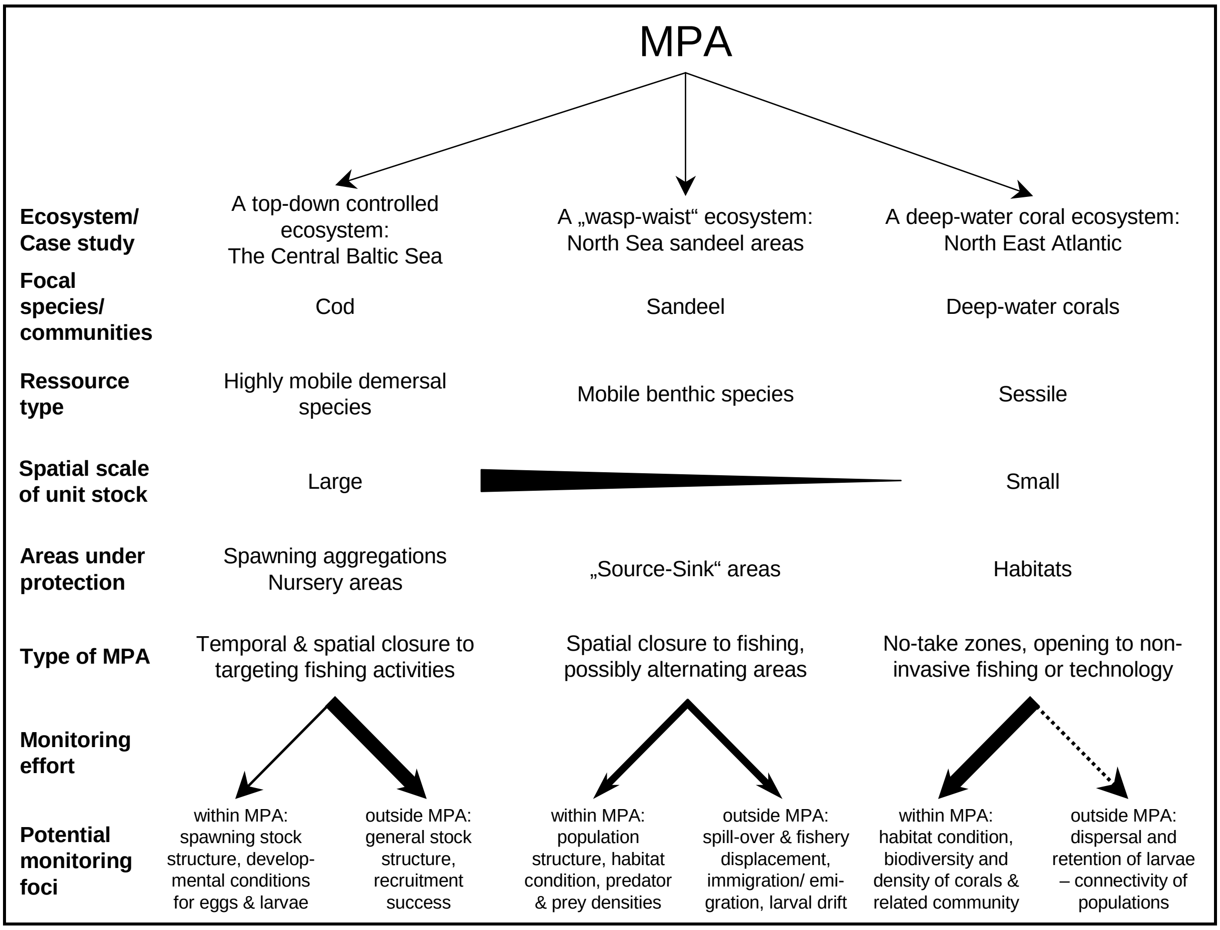

Fig. 3: Monitoring effort necessary within the MPA and in adjacent waters, considering three different case studies of the EU-project PROTECT as an example.

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Modelling: current approaches for assessing MPA effects

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IFREMER

Two kinds of approaches have been envisaged to assess ecological and fisheries-related impacts of MPAs: mathematical models depicting the dynamics of populations, communities or ecosystems that are generally used for policy screening analyses; and empirical approaches based on statistical modelling of field data that are used to test effects and provide diagnostics about the ecosystem and resources. Statistical models lead to defining empirical indicators and sampling designs for long-term programmes of experimental monitoring. Mathematical models enable to explore issues related to MPA design and its consequences on the dynamics of populations and fisheries; they provide reference points against which system dynamics can be gauged.

1. Assessing MPA effects from mathematical modelling approaches

A number of dynamic models of fisheries and exploited populations have been developed in the last decade to evaluate ecological and fisheries-related effects of MPA (Gerber et al. 2003). An extensive review of these models may be found in Pelletier and Mahévas (revised). Here, we propose a typology of existing models illustrated by a limited number of examples (Table 1). The typology is intended to provide an overview of existing models while avoiding a tedious description of all references.

Models were classified into four types ranging from simple to complex models (Table 1). Non spatial single species models often rely on logistic growth and assume instantaneous dispersion of fish over the entire fishery area. They are used to investigate permanent no-take reserves covering a fraction of the fishery area (Hastings and Botsford 1999; Pezzey et al. 2000, and others). Source-sink models make assumptions about larval dispersion schemes, considering a local dynamics in each patch. These models enable exploring no-take reserve designs in terms of number, size and location in source vs. sink patches. From the literature, these two kinds of models mostly appear as heuristic tools that yield general ideas about MPA effects. They are of limited interest for policy analysis because they rely on simplistic assumptions and are too aggregated. Similarly, single species models ignoring population demographic structure provide little insight about the performance of MPA aimed at protecting juveniles or spawners. Spatially-explicit demographic models depict growth, reproduction and fishing and natural mortalities, as well as fish movement.

MPA designs investigated include number, size and location of MPA, and possibly temporary closures in the case of models expliciting intrannual dynamics. Spatially-explicit fisheries models include additional detail concerning exploitation, in particular effort may be described in terms of gears, number and characteristics of vessels, and fleet dynamics (Table 1). These models are appropriate for investigating MPA designs other than permanent no-take zones, for appraising the impact of MPA upon population structure, and account for restoration through enhanced reproduction and recruitment. MPA designs aimed at protecting sensitive stages of the population, possibly on a seasonal basis, may be investigated, although there are few such published examples. Spatially-explicit fisheries models permit in addition to study more elaborated policies including MPA targeting particular fishing activities or gears, and other regulatory measures such as effort controls. They are also needed for exploring mixed fisheries issues, e.g. technical interactions and discards. Trophodynamic models describing the state or dynamics of communities or ecosystems based on trophic interactions have been used in a few instances for exploring the performance of permanent no-take zones (Table 1). The complexity of trophic interactions makes it difficult to further refine the model; 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).

In the light of defining indicators for MPA effects, it is important to keep in mind these differences between model objective and model ability to assess MPA effects. Heuristic models should be distinguished from policy screening tools that can form the basis for decision support systems. Metrics[67] and indicators should thus be devised depending on model objectives. Metrics common to all models are total catch, and, to the exception of metapopulation models, total abundance and biomass (Table 1). The other metrics depend on model state variables and model assumptions, and on the level of detail in the model.

For a given metric, results depend on model attributes; they may even be contradictory (not detailed here, see Pelletier and Mahévas (revised) for examples). 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 results. The definition of a metric must include the specification of the model from which it is calculated. Evaluating the precision and accuracy of each metric in Table 1, and its sensitivity to MPA effects 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.

It should also be noted that most published models correspond to theoretical exercises which are not fitted nor even calibrated from real data.

2. Assessing MPA effects from the analysis of field data

Many studies have assessed the impact of reserves on fish populations and on marine organisms (see e.g. reviews in Roberts and Polunin 1991, Garcìa-Charton and Pérez-Ruzafa 1999; Russ 2002, Halpern 2003). The majority of these studies pertain to coral reef ecosystems (but see Sanchez-Lizaso et al. 2000, for a review focused on Mediterranean ecosystems) and no-take reserves aimed at preserving natural heritage, marine ecosystems and biodiversity. 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 date (Russ and Alcala 1996; Edgar and Barrett 1997, 1999; Babcock et al. 1999; Chiappone et al. 2000; Willis et al. 2003, and many others). 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, and mostly through multivariate descriptive methods which

[67] By indicator, we mean a function of observations or the output of a model, which value indicates the present state and/or dynamics of the system of interest (FAO 1999). An indicator is linked to a manager or research question, and should meet desirable performance criteria in this respect. In order to stress the importance of validating indicators through performance criteria, we define a metric as a biological response at a given scale, while an indicator is a metric displaying desirable performance in terms of MPA assessment.

do not provide for statistical inference (Garcia-Rubies and Zabala 1990; Dufour et al. 1995; Russ and Alcala 1998; Paddack and Estes 2000).

The examination of the literature reveals that significant differences are obtained for some particular species (Bell 1983 and Paddack and Estes 2000, among others), taxonomic families (e.g. Alcala 1988, 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). In many cases however, 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).

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, for overall trends and for four functional groups including herbivores, planktivores/invertebrate eaters, carnivores and invertebrates. However, the statistical significance of the results was not taken into account in this approach, descriptive analyses being treated in the same way as inferential ones.

A different insight on the empirical assessment of MPA is provided by Pelletier et al. (2005) who reviewed 94 published empirical studies in order to identify and evaluate indicators of MPA effects. Effects were listed from review papers on MPA and were ranged according to the time horizon at which the effect is expected to be detectable (Table 2). The metrics used for assessing ecological effects of MPA were listed from the literature and scored with respect to each effect according to relevance and effectiveness criteria (Nicholson and Fryer 2002). The relevance of an indicator illustrates the link between the indicator and the effect it is supposed to indicate; it was evaluated through the number of times the metric was used for assessing an effect in the literature, assuming that the more often it was used, the stronger the link between the metric and the effect. The effectiveness of an indicator gathers the concept of statistical power, precision, variability, sensitiveness and the existence of reference values or thresholds against which the indicator can be tested. The effectiveness of a potential indicator for a given effect was calculated as the ratio of the number of times it gave a significant result (whether positive or negative) divided by the number of times it was used across all studies based on inferential statistical analysis. Note that the significance of a result is mostly tied to the statistical power of the analysis, which in turn depends on system variability, metric sensitivity, and experimental design. This definition of effectiveness did not account for the existence of reference values or thresholds, since empirical assessments did not provide reference values (but see subsection "Reference points" below).

Results show that several effects are not really studied up to now (Figure 1), mostly long-term effects, but also effects linked to trophic interactions, density-dependent changes and protection of endangered species. Effects at community level are less well studied than effects at population level. Habitat-related effects are rarely investigated, although examples become more frequent now. Relatively few metrics were deemed relevant, and this was mainly for short-term effects (Table 2). Furthermore, no relevant metrics could be identified for several effects studied. Results indicate that many metrics were contemplated for studying MPA effects, but that overall, few of them were found to be relevant and effective. A number of metrics gave non significant results in more that 50% of the studies where they were used (Table 2). These findings constrast with Halpern's (2003) results, mainly because statistical significance was accounted for. But they corroborate and systematize Russ's (2002) considerations about the lack of significance found in many studies.

The lack of statistical significance may be explained by several weaknesses encountered in many studies. The first one pertains to the lack of initial evaluation. In a large number of studies, the initial state of the fish community was not assessed before establishing the reserve. Abundance and other biological variables inside the reserve were compared to those in a reference zone, i.e. from a Control-Impact design (e.g. Letourneur 1996, Harmelin et al. 1995). Spatial and temporal heterogeneities of ecosystems lead to confusion of protection effects with environmental effects such as those linked with habitat structure (Garcìa-Charton and Perez-Ruzafa 1999), and make it necessary to rely on designs including measurements before and after establishment of the reserve, inside and outside of the reserve (Before-After Control-Impact (BACI) designs (Underwood 1994)). Although such designs become more frequent, there are still few published examples (e.g. Edgar and Barrett 1999).

A second issue relates to habitat effects. Habitat is determinant to explain the spatial distribution and structure of fish communities (Sale 1998) and should thus be accounted for when comparing biological responses in distinct zones. Relatively few reserve assessments have explicitly considered habitat. In several instances, differences in densities were tested by habitat type (e.g. Letourneur 1996). Paddack and Estes (2000) compared fish assemblages between sites while accounting for substratum 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), or some other definition of habitat (McCormick and Choat 1987, Castilla and Bustamante 1989, Garcìa-Charton et al. 2004).

The third issue relates to the diagnostic of reserve effects. Direct effects are in general assessed by comparing densities, biomasses, mean size or diversity indices, between the reserve and the exploited 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 particular species to reserve protection. But they do not provide a synoptic view of the impact of the reserve, and do not allow to compare the sensitivities of different fish community components to reserve status. Assessing reserve impact 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).

3. Perspectives : Improving methodology for MPA assessment

Concerning empirical assessment of MPA effects, experimental design is a major area of improvement. Because many different processes operate simultaneously to generate spatial and temporal variability in populations, assessing effects requires multifactorial sampling designs. Beyond BACI designs provide such a framework (Underwood 1994). Causal inference is possible with these designs if data are sampled at several dates before and after MPA establishment, both in the MPA and in several control locations. Multiple controls are necessary to avoid confounding natural spatial variability with MPA effects, or missing important effects 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.

There are other issues pertaining to sampling design and monitoring. For instance, MPAs are more and more envisaged under the form of reserve networks, which implies both local and regional scaling for sampling designs. Sampling designs and resulting performance of indicators in terms of precision and statistical power may be investigated and optimized using simulations and comparative approaches across case studies (Benedetti-Cecchi 2001; Sala et al. 2002). Also, the choice of an observation technique bears consequences on the precision and accuracy of indicators, and on monitoring costs (Willis et al. 2000).

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 as the same time as fish communities (García-Charton et al. 2000; Stewart-Oaten and Bence 2001). Information on habitat and more generally the different components of spatial variability should be introduced in models, thereby reducing variability (García-Charton et al. 2004; Ferraris et al. 2005). The latter found that habitat explained a substantial part of density variation when assessing the impact of fishing in a reserve.

A third area of improvement lies in more holistic approaches to evaluate MPA effects. Effects are mostly evaluated from univariate approaches, i.e. for a single species or species group, and for a single variable (e.g. density or biomass) at the time. This way, effects cannot be compared across species or species groups, and across variables. Consequently, an overall diagnostic about MPA effects may not be established, and the performance of metrics cannot be compared. Several approaches were recently proposed to overcome these problems. Hence, Ferraris et al. (2005) simultaneously tested for the effect of fishing in a former reserve, upon the densities of all species groups within a single model. This study was further pursued by Amand et al. (2004) who jointly modelled biological metrics such density, mean size, species richness and biomass as a function of factors linked to reserve status and habitat, in order to rank the sensitivities of these metrics to the effect of fishing in a former reserve. Note that these methods could be easily transposed to the assessment of MPA effects. Claudet et al. (2004) and Claudet et al. (unpublished findings[68]) used non-parametric multivariate analysis of variance (Anderson 2001) and multivariate regression trees to model density, mean sizes and diversity indices in order to estimate the impact of a no-take reserve on the fish assemblage.

Concerning modelling approaches, the use of mathematical models to evaluate the impact of MPA has recently been challenged by Willis et al. (2003). According to these authors, "theoretical models are useful in developing our ideas, but they are just that: ideas". Many models are indeed theoretical contributions, and that simple models published in well-known journals may have resulted in simplistic prescriptions, e.g. about the reserve size needed to protect fisheries resources. However, models are remarkable tools to evaluate MPA consequences at the scale of fisheries and ecosystems. In this respect, the main area of improvement for modelling 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 if relevant (e.g. for temporary MPAs). Models should account for mixed fisheries (multispecies multifleet fisheries), and for fishers' response to MPA. Models 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 MPA are not the only management tool used in a given fishery. Several of these points were already raised in a SCOR symposium (Sumaila et al. 2000). Pelletier et al. (2001) and Mahévas and Pelletier (2004) proposed ISIS-Fish, a model that incorporates most of these features. The model was applied to the Bay of Biscay mixed fishery by Drouineau et al. (in revision).

[68] submitted for publication in Environmental Conservation

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 is necessary, and these aspects may be poorly known, but the need for a better appraisal of the spatial dynamics of exploitation should be more emphasized. Conventional fisheries statistics provide some information, but their spatial resolution is generally poor. Additional information must be obtained through fishers interviews, and recent research projects[69] focus on these questions.

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

[69] e.g. TEChnical and TACtical Adaptations of Important European Fleets (TECTAC), EC project n° Q5RS-2002-01291

Table 1. Mathematical models used for MPA assessment.

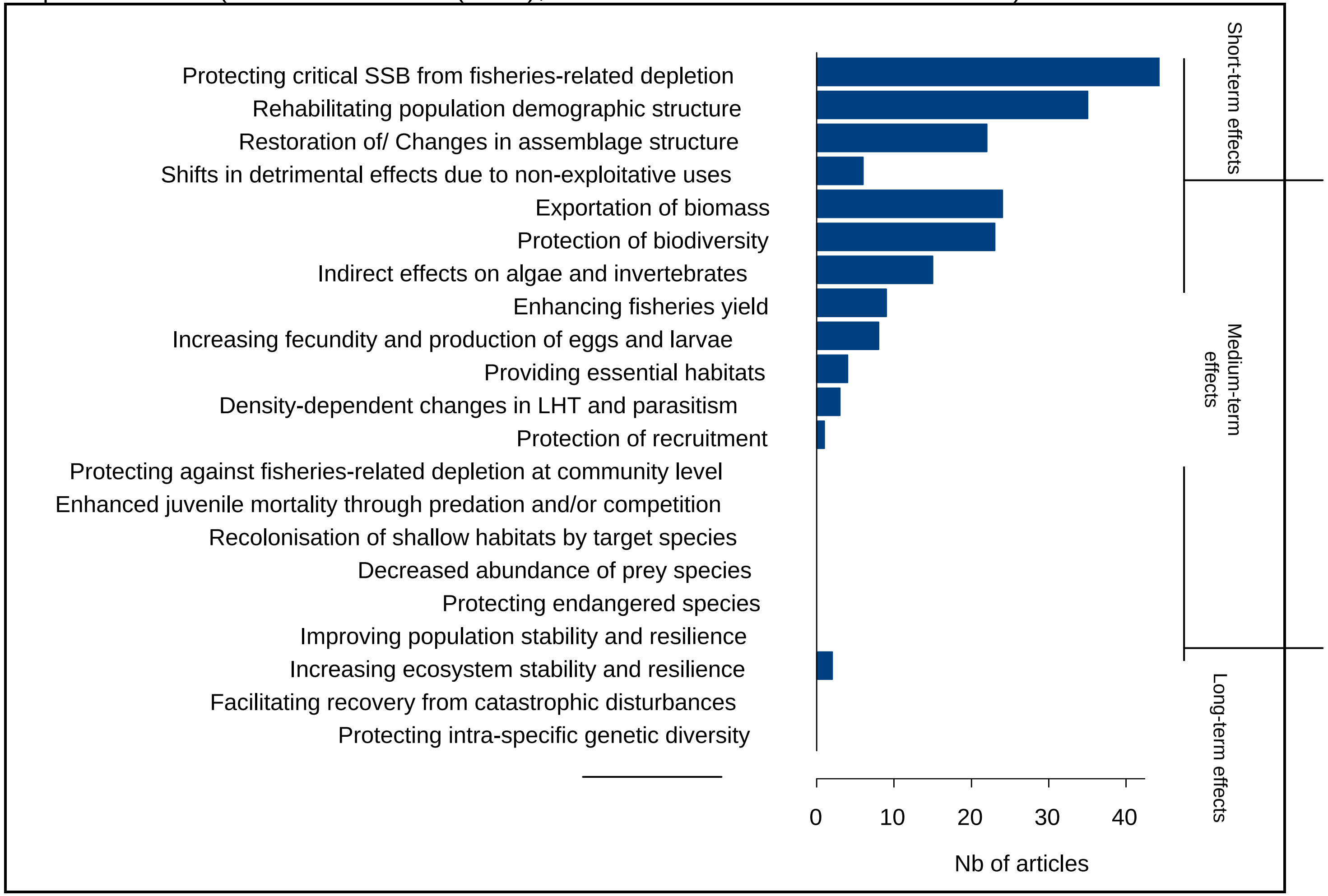
Model type	Representative examples	Main assumptions	Metrics studied in corresponding papers	MPA designs investigated
Non spatial single species model	Lauck et al. 1998; Hastings and Botsford 1999; Mangel 1998, 2000; Pezzey et al. 2000	Dynamic pool and stock-recruitment models assuming instantaneous dispersion of fish and uniform fishing mortality	total catch; total biomass; total abundance; current biomass/virgin biomass; risk of population collapse	-only no-take permanent reserves -reserve size
	Man 1995	Metapopulation models assuming instantaneous dispersion of fish and local extinction rate	proportion of occupied patches total catch	
Spatially-explicit single species model	Brown and Roughgarden 1997; Tuck and Possingham 2000; Crowder et al. 2000; Sanchirico and Wilen 2001	Source-sink models : subpopulations connected through larval and adult dispersion, local dynamics in each patch (e.g. logistic)	total catch total abundance abundance per subpopulation	-only no-take permanent reserves -number and size of reserves -location in source versus sink populations
	Beverton and Holt 1957 ; Polacheck 1990 ; Walters et al. 1993 ; Pelletier and Magal 1996 ; Sladek Nowlis and Roberts 1999; Guénette et al. 2000, and many others	Stage-structured demographic models with explicit growth, mortalities, reproduction, fish movement (dispersion or migration), and (rarely) seasonal dynamics	catch, abundance and biomass (total or per patch[70]); spawner abundance and biomass; asymptotic growth rate stable age distribution; yield (equilibrium and short-term); risk of population collapse; catch variation	-mostly no-take permanent reserve -number, size, spacing and location of reserves -temporary closures
Spatially-explicit fisheries model	Walters and Bonfil 1999 ; Holland 2000, 2002 ; Pelletier et al. 2001 ; Mahévas and Pelletier 2004	Stage-structured demographic population models with parameterization of fishing effort[71], multiple fleets, multiple species, (rarely) economic aspects, and (rarely) explicit fishers' response to MPA	same as above per species or overall catch per fleet effort-related metrics economic metrics	Any type of MPA : -number and location or spacing of MPA -timing of MPA -targetted activities
Trophodynamic models	Walters 2000; Watson et al. 2000; Shin and Cury 2001; Beattie et al. 2002	Models structured according to size or trophic component	catch or biomass (total or per component); size or biomass spectra	-only no-take reserves -reserve size

[70] A patch denotes a zone defined in the model.
[71] In such models, fishing effort may be described in terms of number of vessels, number of gears, gear selectivity. In mixed fisheries models, several fishing activities (fleets) may be considered with distinct attributes and behaviours.

Table 2. Relevant empirical metrics and their effectiveness. A metric is regarded relevant for a given effect when used to study this effect in more than 5 published studies. Effectiveness was measured through the proportion of significant results in published studies, but only for effects that were tested in more than 5 studies. The effectiveness of metrics related to biomass exportation was not calculated, because these metrics were not used in formal tests of the effect. Density profiles were generally studied through multivariate techniques.

Time scale	Effects (overall nb. of metrics used in studies)	Relevant empirical indicators (effectiveness in %)
Short-term effects	Protecting critical spawning stock biomass (39 metrics)	total biomass (85) biomass per family (72) biomass per species or genus (39) total density (56) total density over fishable species (82) density per family (50) density per trophic group (95) density per species or genus (41) density per species stage (67) CPUE per species (40) size distribution of species (95)
	Rehabilitating population demographic structure (20 metrics)	biomass per species or genus (35) mean size per species or genus (38) size distribution of species (56)
	Restoration of assemblage structure (10 metrics)	density profile per species (67) species richness per family (41)
Medium-term effects	Exportation of biomass (11metrics)	movement patterns home range site fidelity
	Protecting biodiversity (9 metrics)	total species richness (59)
	Indirect effects on algae and invertebrates (10 metrics)	density per species or genus (39) benthic cover (68)

Figure 1. Ecological and fisheries-related effects expected from MPAs and the number of times they were studied in published empirical studies (see Pelletier et al. (2005), Tables 2 to 4 for the list of references).



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Socio-economic evaluation: a review

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Introduction

Marine protected areas (MPAs) are sometimes seen as a panacea for marine problems (e.g. Kaiser, 2005). However, the concepts behind the success or failure of MPAs are different depending on the discipline being studied (e.g. ecology, fisheries science, fisheries economics) or the interest group being represented (e.g. environmental groups, fishermen, society).[1] There is a long history of MPAs in fisheries management in Europe. The implementation of these areas has typically been decided using biological (i.e. fisheries science) advice and much of the evaluation has targeted the biological impact of MPAs, ignoring the socio-economic impacts.

In fact, there have been few significant real-world attempts to analyse the subsequent economic effects to fishing that arise from the implementation of an MPA. The vast majority of socio-economic studies are theoretical and largely limited to the last seven years. Among these, bio-economic models are notable, being increasingly used to measure the impact of MPAs on both biological and economic indicators (Carter, 2003), but they are not the only socio-economic approach available as will be highlighted hereafter. It is worth noting that the development of economic indicators for MPAs, particularly with respect to fishing activity, is not well developed (Dalton, 2004; Pelletier, 2005). This is in stark contrast to biological indicators. Therefore, unlike biological impacts, economic impacts are evaluated at best on a case-by-case basis.

From the socio-economic studies published, there is a wide difference of opinion between authors on the potential of marine protected areas. There is often an assumption of long-run benefits to the fishery through stock recovery or spill over effects (Johnson et al., 1999). However, these have seen little quantification and few studies have explicitly considered the impacts of MPAs on fishermen. Of the work that has been undertaken, there is a body exhibiting scepticism as to the role that marine reserves can play in fisheries management (Conrad, 1999; Hannesson, 1998), with Carter (2003) elucidating the broad finding of modelling work to date that MPAs by themselves are not likely to increase aggregate fisher welfare in a fishery characterized by (regulated or restricted) open access. MPAs need to be used in combination with effort controls and/or other management measures to avoid the dissipation of benefits (Shepherd, 1993; Hannesson, 1998; Horwood *et al*. 1998; Anderson, 2000). Not all studies, however, lack optimism. Holland (2000), utilising a multispecies bioeconomic model, found that the impacts of MPAs vary across species, with some experiencing increasing yields and some decreasing. More specifically, Holland and Brazee (1996) conclude from their bioeconomic modelling that marine reserves can probably sustain or perhaps increase yields for moderate to heavily fished fisheries, but will probably not improve yields for lightly fished fisheries. The value of the reserve, however, has been found to be highly dependent on the level of fishing effort, as has optimal size, with natural increase in stock being important in a lightly fished fishery, while outflow from the reserve is more important and optimal MPA size larger in a heavily fished fishery. The design of the MPA and its management relative to the species and fisheries targeted is evidently critical, as is the nature and scale of activities other than fishing in the

[1] In the case of fisheries management an MPA is the same as a fishing exclusion zone for at least one type of (if not all) fishing method.

vicinity of the MPA. For example, there is a wide body of literature addressing the negative impacts of tourism and recreation on the value of MPAs (Sala, *et al*.,1996; Zabala,1996; Badalamenti, *et al*., 2000; Francour, *et al*., 2001; Salmona and Verardi, 2001; Himes, 2002). Although only a sample of the socio-economic studies undertaken on MPAs, one observation common to all is the predominance of hypothetical studies and lack of real-world studies to validate such observations.

The largely hypothetical nature of MPA studies so far reflects the complexity of the challenge posed by MPAs and data paucity. The existence of knowledge on the migration rates, or general movement of fish species is one particular challenge, fisher behaviour (i.e. reaction to an imposed MPA) is another. One of the few studies to consider this latter aspect has recently been undertaken to evaluate the economic impact of three alternative fishing exclusion zones in the North Sea, specifically to assist in stock recovery (Pascoe and Mardle, 2005). In terms of practicability, the main difficulty with the design of MPAs arises from the multi-species element. 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.

Socio-economic effects and influences

The use of MPAs has been prompted by, inter alia, a long-term decline in catch and CPUE, associated with increasing fishing effort, along with an anticipated benefit of a mitigation or reversal of this impact. MPAs are attributed as holding the potential to build up fish biomass and facilitate growth and reproduction of a number of over-fished or otherwise endangered species, thus creating positive economic as well as environmental benefits (Pipitone, *et al.*, 2000; Nowlis, 2000). These economic benefits are often regarded as benefits to local fishers arising from the contribution MPAs make to:

- decreasing by-catch and the landings of over-exploited and depleted fish stocks
- reducing fishing mortality on fishes that have not yet reached the age of sexual maturity (growth overfishing)
- increasing the life-span and size reached by individuals within the stock
- increasing the reproductive potential of stocks
- reducing the potential for stock collapse
- protecting natural habitats from fishing gear impacts or assisting in their recovery
- maintaining the integrity and biodiversity of the ecosystems supporting fisheries (e.g. posidonia beds)
- mitigating user conflicts
- increasing awareness of fisheries and their associated issues through education
- supporting existing legal provisions (e.g. illegal trawling areas, not complied with)
- supporting categories of fisheries (e.g. artisanal fleets)
- knowledge and research

There is also an accompanying assumption of a long-run dimension to these benefits, through recovery and spill-over effects (Johnson, *et al.,* 1999).

Unfortunately, and especially in comparison to the Caribbean and the South Pacific, very few articles have been written cataloguing, describing and analysing the economic impact of MPAs in Europe or the northern hemisphere. Therefore, the nature of the benefits and costs are poorly defined and quantified. However, by drawing on the experience of a wide range of studies, it is evident that the potential effects are broader and can be experienced on a shorter time scale. The following table indicates the range of economic values affected by MPAs with examples of their effects (Table 1).

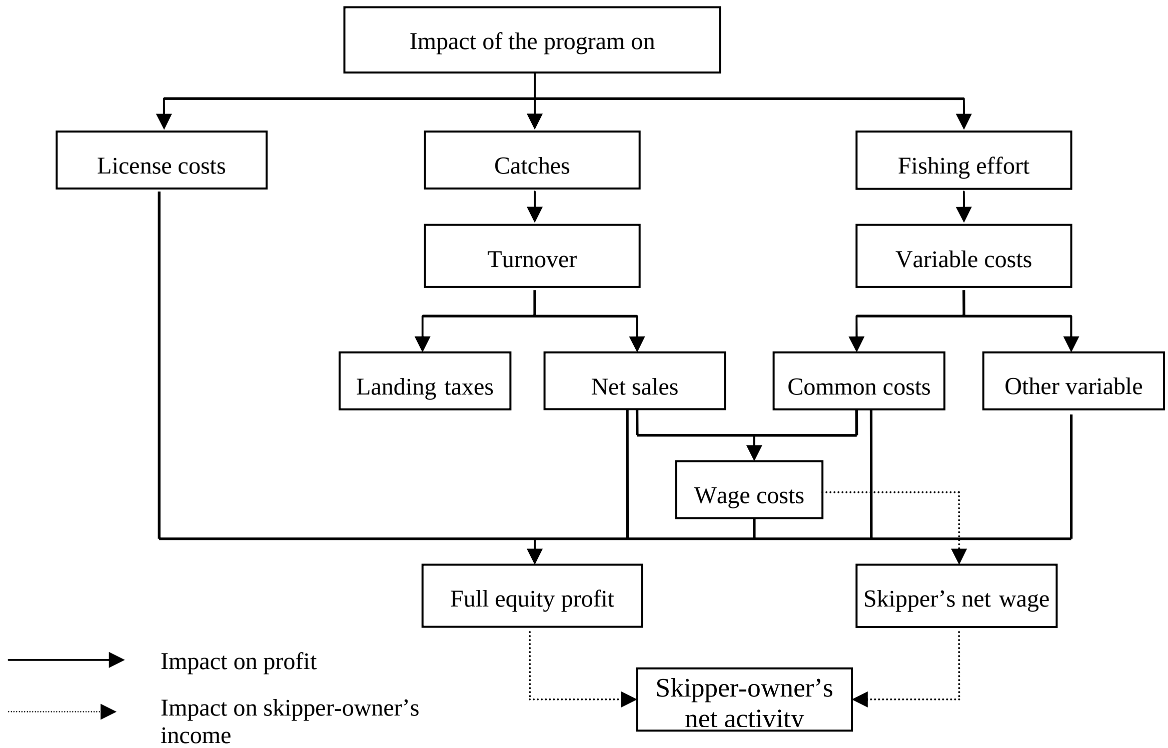
In terms of direct and indirect use value and the impact on fishing fleets, MPAs have the potential to affect various elements of their economic performance as shown in Figure 1.

Table 1. Economic values affected by MPAs

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 (existence value) or availability to future generations (bequest value)
Example: Extractive uses (e.g. commercial and recreational 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

MPAs hopefully contribute to increases in harvestable fish stocks and thereby catches through the flow of natural increase in stocks in the regulated zone(s) and the flow of net transfers from these zones, albeit that the relative levels of fishing effort in the two areas and predator-prey relationships will determine the outcome in this regard. The effects are felt not only within the zone but also downstream in both commercial and recreational fishing operations. For commercial fisheries, and any charter boat ventures, changes in catches will have consequences for turnover, landing taxes and net sales (incorporating price variation due to any impact of the MPA or external factors on supplies to the market). For any exempted vessels permitted in the zone(s), the MPA can potentially reduce the fishing effort required to maintain catch levels, while providing incentives to increase effort to take advantage of enhanced opportunities, potential profits and reduced competition (Bailey, 1997). Variable costs, such as fuel, will tend to decrease or increase, respectively. Reductions in conflict can likewise reduce gear replacement costs. These factors potentially contribute to earnings and profit levels, and the maintenance of employment, although stability does not necessarily improve with MPA implementation. For vessels excluded from the zone, inter alia, variable costs may well increase given the need to travel further afield. Additional investment in gear and boats may also prove necessary to diversify fishing operations. Displaced vessels have the option of targeting stocks at the edge of the zone(s) to gain from net transfers from the reserve, targeting alternative stocks or species and raising issues of effort displacement, or laying up. Any socio-economic assessment of MPA implementation and performance should take onboard the consequences and actions of excluded vessels along with those of exempted vessels, with both a short to medium term and long-run perspective.

Figure 1. The impact of MPA on the economic performance of the fleet (source:



CEDEM)

In socio-economic terms as table 1 indicates, there are other aspects of costs and benefits to MPA deployment, in addition to those experienced through upstream fishing operations, including: the costs of MPA implementation, monitoring and enforcement (relative to management alternatives); the benefits and costs transferred from fishing into the wider community through, inter alia, downstream support services, markets and the purchasing power of fishers' salaries and profits, the 'multiplier effect'; and the benefits and costs associated with the take-up of opportunities for diversification by fishers and the wider community into recreation and tourism. Looking to option and passive use values (table 1) there are further dimensions to the socio-economic assessment of MPAs, the value to society of maintaining opportunities for exploitation into the future and the preservation of habitats and maintenance of diversity as an end in itself. Albeit not readily monetised, techniques are available for attaching value there-to, facilitating the approximation of the socio-economic value of MPAs to society (as are hereafter discussed).

Unfortunately, previous studies have shown that the socio-economic value of MPAs, whether positive or negative, is not a general question nor readily anticipated or explained. It has been shown to be dependent on the scenario under analysis with multiple factors affecting it, including the: structure and socio-economic characteristics of the fishery, its location and physical characteristics, nature and state of the stocks, fishing method(s) employed, and management policy applied (including the nature and size of any MPA used) (Man, *et al.*, 1995; Pezzey, *et al.,* 2000). Even in situations sharing common characteristics, these factors can result in very different outcomes. The complexity of multi-species interactions and the complex relationship between fishing effort, gear effects and species biology are just a few of the confounding influences in the drive to an effective MPA from a socio-economic perspective. For example, MPAs have been linked with increasing fishing costs, over-capitalisation, the shortening of fishing seasons and the achieval of the same conservation effects as an optimum fishing strategy but with smaller catches (Hannesson, 1998), posing socio-economic consequences for fishers. When net transfer rates are low MPAs have also been shown not to mitigate against losses in economic rent to the fishery, albeit providing stock protection (Conrad, 1999). Further, shifts in relative species abundance have in cases favoured low-value species at the expense of higher-value species (Relini, *et al*., 1996; Pipitone, *et al*., 2000). It is evident that there is potential for MPAs to not only have benefits, but also undesirable side-effects and unanticipated costs and for their benefits to be dissipated by external forces.

One category of external forces to have been felt widely in the northern hemisphere, especially Europe, is the growth of tourism and recreation, both as a benefit and cost. Recreational fishing opportunities, diving, the development of visitor centres, aquaria and educational and research opportunities offer diversity, income generation and education for communities, yet at the same time can give rise to significant potential damage to both resources and habitats undermining gains achieved from the exclusion of fishing activities (Sala, *et al.,* 1996; Zabala, 1996; Pozo, 1998; Badalamenti, *et al.,* 2000; Francour, *et al.,* 2001; Salmona and Verardi, 2001). Recreational activities, along with commercial fisheries, may well require management to avoid undermining any economic benefit from implementing MPAs (Badalamenti, *et al.,* 2000). The socio-economic contribution of recreation and tourism to the local and national economy does not necessarily compensate for the reduction in value caused in other respects (Francour, *et al.,* 2001). Such issues will be of particular pertinence to the sand eel case study.

MPAs can represent a valuable tool of management in certain circumstances, although their benefits are rarely felt uniformly across species, segments of the fishery or across geographical sub-areas.

Methodology of socio-economic assessment

The nature and purpose of socio-economic assessment

Three generic types of socio-economic assessment can be identified, distinguished according to purpose (Table 2) of describing the methodology of the socio-economic assessment of MPAs in PROTECT. These are: (i) profiling, (ii) impact analysis, and (iii) benefit assessment. The principle approaches to be used for the case studies fall within benefit assessment, but draw from profiling and impact analysis in terms of inputs and to cater for data paucity. Bioeconomic models, drawing from impact analysis and benefit assessment, are being utilised for all three case studies, with greater complexity and empirical underpinning for the North Sea sandeel case study and Baltic Sea cod case study where knowledge and data availability is greater. To take on board the non-market based value of MPAs, choice experiments (within the benefit assessment category) are being utilised for those case studies where there is a marked non-market component: the North Sea sandeel and deep water coral case studies.

Profiling:

Profiling aims to provide basic empirical information on the socio-economic characteristics of a MPAs 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 MPA), 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 is potentially 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. The modelling within PROTECT will make use of profiling in several respects.

Table 2. Socio-economic assessment of MPAs

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 MPA 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 MPA 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
		Bioeconomic modelling
Benefit assessment	To determine the net economic value to society of a proposed MPA in relation to alternative management options, and to identify optimal MPA configuration (i.e. size, location, etc.)	Cost-effectiveness analysis Cost-benefit analysis Contingent valuation & choice-experiments Trade-off analysis Multi-criteria methods

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, it can be used to measure the effects of establishing an MPA 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 is a value judgement, and these inevitably differ between stakeholders. Given that the economic impacts of MPAs are in principle quite diverse, the range of applicable techniques for examining them is also potentially large. Those listed in the Table are: input-output analysis, which can be used to trace through the direct and indirect effects of a MPA on economic activity and hence to derive multiplier effects; demand analysis, which is relevant to identifying the market impact of changes on fish landings which may result from the imposition of an MPA; financial analysis, which is appropriate where a MPA 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 MPA is perceived by fishermen, recreationists, conservation groups and others. All these involve the testing of particular hypotheses about the effects of a MPA. This is what mainly distinguishes them from straightforward descriptive profiling. Impact studies of exclusion zones in Europe include those by Whitmarsh *et al*. (2002), Leeworthy and Wiley (2000) and Suman *et al.* (2000). Several of these techniques will be undertaken in the process of undertaking the socio-economic evaluation of the three case studies.

Benefit assessment:

Benefit assessment attempts to measure the net economic value of an MPA in terms of the flow of benefits, 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, e.g. whether or not to establish a MPA and/or whether a MPA is the best (i.e. most economically efficient) management option (scenarios being considered in the case studies). 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, through contingent valuation methods or as planned here, choice experiments. 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, increased tourism through larger seabird populations in the sandeel case study 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 (Wattage *et al*., 2003), with a parallel also being evident for bio-economic modelling in the form of multi-criteria methods (Brown, *et al.,* 2001a; 2001b). Within the scope of PROTECT, the possibility of incorporating the findings of the bioeconomic analysis and choice experiments into a CBA framework is being looked in to.

Bio-economic modelling:

Bio-economic models of commercial fisheries have a role in both impact analysis and benefit assessment. Specifically, (a) a bio-economic model can be used as an engine for generating hypotheses concerning the effects of MPAs which can then be tested against field observations, (b) empirically estimated bio-economic models can be used to simulate the behaviour of a fishery under a variety of 'what-if ?' scenarios (such as changes in the size of a MPA), or else to identify the circumstances under which the performance of a fishery system would be optimised. Bioeconomic models have been utilised in a number of MPA studies globally (refer to Table 2 and the outputs of the VALFEZ project (Pickering 2003)), with a variety of key features and assumptions reflecting the scenario in question, data availability and characteristics of the species involved and dependent fisheries. The same factors will determine the models used within PROTECT for each case study.

Conclusion

A common feature of the PROTECT case studies is that they all deal with the *direct use value* of the marine resources, typically in the form of 'consumptive' outputs (e.g. fish) but in some instance 'non-consumptive' benefits (ecotourism linked to seal watching). Methodology for studies of this nature tends to be straightforward, as the goods and services are commercially traded, such that market prices can in general be used as the yardstick for measuring societal benefits. Within PROTECT bioeconomic modelling is the principal approach being used for the determination of direct use value in the three case studies, supplemented as appropriate by other relevant approaches.

The difficulty arises when the commodity in hand is un-priced, with no commercially traded and priced dimension. MPAs can have a major portion of their value as indirect use value (i.e. functional benefits) or non-use value (i.e. existence and bequest value), associated with conservation, even though the potential significance of these sources of benefit may not be recognised or alluded to. For these latter forms of value the only available methods are contingent valuation method (CVM) (Wattage, *et al.,* 2002) and/or choice experiments (CE). Each method has its own distinct advantages and disadvantages, which are well rehearsed in the literature. Given the opportunity to elicit a deeper understanding of different levels of management attributes and aspects of an MPA, in this instance choice experiments is the method of choice (Wattage et al., 2005). Choice experiments will be targeted at the two case studies with a significant unpriced element: the North Sea sand eel case study and the deep water coral case study.

Table 2 MPA bioeconomic modelling publications (Source: Mardle, 2004).

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	• Schaefer –Gordon 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

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Review of bio-economic modelling work of relevance for deep-water coral management

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Abstract
This presentation is a review of the bioeconomic modelling relevant for deep-water coral management, written for a general readership. As most of the biological knowledge of deep water coral is relatively slim and recent, the bioeconomic modelling on this resource is non-existent. However, there is much research on related issues that have relevance for the bioeconomic modelling of deep-water coral, and that is what is presented here.

The review presents bioeconomic modelling of habitat and resource use generally, bioeconomic modelling of coral reefs specifically, bioeconomic valuation; the so called production function approach**,** as well as a short overview of the bioeconomic analysis of habitat management options.

Introduction

In recent years increasing attention has been directed towards the habitat effects of fishing activities. This interest is a result of the large decline in commercial fish stocks and the questions connected to this decline. Different types of fishing activity have been shown to have a large variety of impacts upon marine habitats (Auster and Langton 1999, NRC 2002), and recently also on deep water coral habitats (Fosså et al. 2002). Biological research furthermore indicates that loss or changes in habitat affect species of commercial interest (Lindholm et al. 2001), hence fishing vessels create so called externalities for both own and other vessels' activities. The economic consequences of these activities, even in excess of the more non-market type of valuations of habitat loss or change, are only partially understood. In the following we aim to present an overview of the relevant bioeconomic modelling with regard to the management of fisheries in connection with deep-water coral (DWC) habitats.

Writing an overview of bio-economic modelling of deep-water coral could be a very quick affair if done in an overly critical manner. I.e. there is no, as far as the author is aware of, published bioeconomic modelling dealing specifically with DWC. However, there are several avenues of bioeconomic research that could be relevant for the modelling of DWC, and that is what is presented here.

In the following we will present the literature on bioeconomic modelling of habitat and resource use generally, bioeconomic modelling of coral reefs specifically, bioeconomic valuation; the so called production function approach**,** as well as a short overview of the bioeconomic analysis of habitat management options.

Bioeconomic habitat modelling

There is very little bioeconomic modelling work that explicitly takes into account the interaction between marine habitats and fisheries. And yet the connection between the two is increasingly being made with regard to management (see for instance Gass (2003) on "essential fish habitat" (EFH) conservation of "habitat areas of particular concern" (HAPC) in the US Magnusen-Stevens Act).

There exists some bioeconomic modelling of fisheries and environmental influences (see Knowler (2002) and Upton and Sutinen (2003) for overviews), of which some approaches could be modified to look at specific aspects of fisheries and deep-water coral habitats. These are presented below, organised according to whether the habitat effects are exogenous to the model or a function of the controls in the system studied.

Exogenous environmental effects

Bell (1972) presents an early bioeconomic model including a habitat variable, namely water temperature, in an empirical model of a fishery. However, outside of climate change issues this is not a variable usually subject to management control. Ellis and Fischer (1987), present a standard Cobb-Douglas harvesting function which depends on effort and environmental quality, rather than the usual stock size. Effort is the control variable, while environmental quality is exogenously determined. I.e. in our context this would mean harvesting depends on the coral habitat (its size or quality) rather than stock size of the commercially important species. Though the idea of harvests depending on habitat qualities seems relevant, the total exclusion of stock size playing a role seems unrealistic for most species.

Kahn (1987) studies how an exogenous environmental quality, which influences the intrinsic growth rate or the carrying capacity of a target species, affects optimal strategies. Schnier (2005) expands upon this approach allowing the intrinsic growth rate and the carrying capacity to vary over a distribution. Lynne et al. (1981) and Barbier and Strand (1998) both study habitat loss via the carrying capacity as a linear function of mangrove area. Freeman (1991) introduces environmental quality into the cost function of harvesting as well as the growth function of the resource in question, hence combining the Ellis and Fischer (1987) and the Kahn (1987) approaches, and setting it in a dynamic model. However, he only studies environmental quality as exogenous to the model, and not directly affected by human behaviour. Skonhoft (1995, 1999) presents a dynamic terrestrial habitat-harvesting model, where habitat enters as a function of the carrying capacity of the species. Land can either be used for agricultural activities, or be left as habitat for the species harvested, hence habitat has an opportunity cost. Again, however, habitat is exogenously decided, and is not affected by harvesting.

Endogenous environmental effects

Very little research is done on endogenous environmental effects. I.e. limited attention has been given to the fact that fishing effort is a control variable in many policy situations, and the choice of policy affects effort levels and forms, which again impact in various ways upon habitat.

Upton and Sutinen (2003) design a bioeconomic model where one vessel group's fishing effort impacts on the habitat of their targeted species, or the habitat of a targeted species of another vessel group. Habitat growth is modelled as logistic, with growth reduced by habitat damage as a function of effort level and type, as well as habitat amount. The resulting habitat amount enters into both the carrying capacity and the intrinsic growth rate.

In a model that diverges from the above in the sense that it describes the interaction between the use of a renewable and a non-renewable resource, Swallow (1990) designs a model where the non-renewable resource could be a habitat that enters into the growth function of the renewable resource. Both the renewable and the non-renewable resource are utilised, and hence the management chosen for either resource affects the results. The author shows how failure to recognise the resource interactions leads to too fast utilisation of the non-renewable resource. This is clearly a relevant model for the DWC case, where instead of there being parallel use of the renewable and non-renewable resource, the latter is indirectly utilised when the former is harvested.

Bioeconomic models of coral reefs

We have not been able to find any bioeconomic models that explicitly study deep water coral (it is even hard to find purely ecological models that do that), however there is some bioeconomic work on shallow-water coral reefs. Crepin (2004) presents a tropical coral environment, and her focus is on the interaction between coral, fish and algae blooms. The paper is purely theoretical and focuses on threshold effects and shifts in the ecosystem.

Bioeconomic valuation – the production function approach

Due to the concern that coastal wetlands are increasingly disappearing, a large amount of research has been taking place on the economic value of coastal environments as support to neighbouring marine fisheries (see Barbier 2000 for an overview).

Barbier (2000) categorises the functions of mangroves for humankind into direct and indirect use values, as well as non-use values. A similar set up can be made for deep-water coral, as illustrated in Figure 1.

Economic values

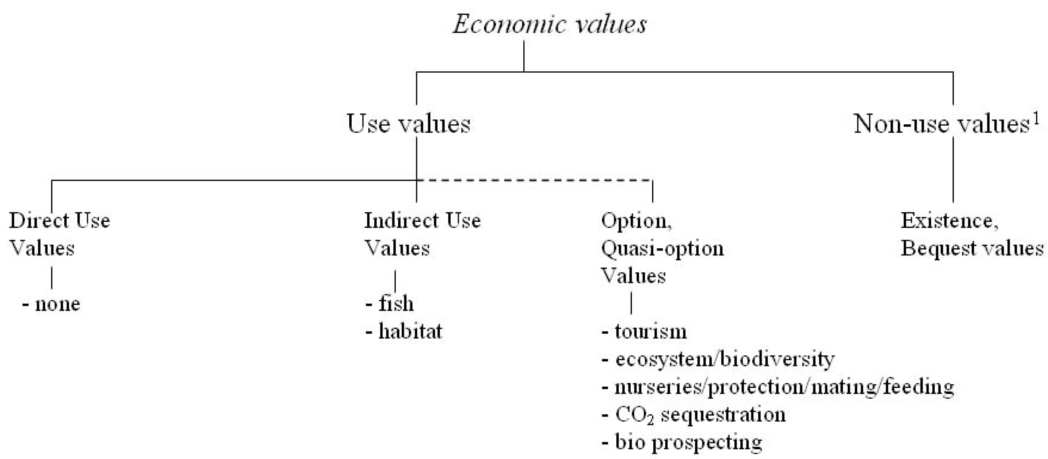

Figure 2 Classification of economic values of deep-water coral (based on Barbier, 2000).

In Figure 1 we observe how use values are either direct or indirect, as well as option or quasi-option values. Direct use values of DWC are as of today non-existent. The indirect use values are based on the coral reefs as areas where some fishing gear type users target higher concentrations of fish (Husebø et al. 2002), and in the context that DWC coral reefs function as habitats for commercial fish species, or for species of import for the former. The question of whether DWC is an essential habitat or ecosystem for any species remains unclear, and though there is evidence that may support DWC being nurseries for commercial species, this evidence is inconclusive (Fosså et al. 2002), and hence these values are to be found as option or quasi-option values, as they may be shown to exist with further research. The same expectation exists with regard to DWC being an important supplier of ongoing CO_2 absorbtion (Freiwald *et al.*, 2004), and a future supplier for bioprospecting (Freiwald et al. 2004). Submarine tourism is expected to be a future area of economic activity, but is as of yet a quasi-option value for DWC. As regards non-use values it is clear from NGO involvement that DWC has both existence and bequest values. I.e. the general public sees value in the existence of deep-water coral reefs, despite never actually being able to observe them in nature, and also sees the value in preserving this habitat for the benefit of future generations.

Though valuation of natural habitats is a relatively recent area of economic research, a number of papers on mangrove-fishery linkages have appeared (Barbier and Strand 1998, Barbier, Strand and Sathirathai 2002), using what has been termed the *production function approach*. This research gives a broad understanding of the economic connections between fisheries and mangroves. The production function approach is a good example of a method for understanding the values connected to habitat, despite limited knowledge of the biological interactions.

The general approach consists of a two-step procedure, with the first step being the characterisation of the interaction between the habitat in question and the economic activity, i.e. the indirect use as portrayed in Figure 1. Hence the habitat effects are modelled as described above in the bioeconomic habitat models. The second step involves the valuation of the impact of habitat change or loss upon the economic activity, by statistically determining a relationship between for instance harvest changes and habitat changes (e.g. see Barbier and Strand 1998).

Bioeconomic analysis of habitat management options

The management of fisheries is usually divided into *direct* and *indirect* control methods.

Direct controls

Direct controls are divided into *input* controls and *output* controls, where inputs controls are the limiting of effort put into the fishery, such as gear of time limitations in fishing. Output controls consist of limiting the outputs from the fishery, i.e. quota limitations. It is clear that these kind of controls could also be implemented with regard to deep water coral conservation, but a clear spatial aspect would be necessary as well. In the bioeconomic literature there is much reference to different gear types and their interaction with target (Armstrong 1999, Armstrong and Sumaila, 2001) and bycatch species (see references in Herrera 2005). Especially the issue of bycatch species would be relevant with regard to the effects of fisheries upon DWC, where the bycatch species would be the DWC, in the shape of a non-renewable resource, and the different types of gear would result in different degrees of extraction/depletion of DWC. Choice of management option would then determine the final harvesting and habitat situation.

Marine reserves

What is sometimes seen as the failure of fisheries management to date, has lead to an increasingly growing interest in closed areas or marine reserves as a way to manage the oceans. This is in a sense a return to direct input controls, but in this case a much more encompassing input control than previously. Much of the bioeconomic literature on marine reserves (for some general overviews of this literature, see Armstrong 2004, Farrow 1996) has been preoccupied with answering the sometimes overly optimistic biological literature within the field. Economists have pointed to the fact that much of the biological literature makes very simplifying and strong assumptions with regard to human behaviour in the face of marine reserve implementation (Hannesson 1998, Smith and Wilen 2003), resulting in reserves having more positive effects than otherwise. However, most of the bioeconomic models applied make simplifying assumptions with regard to the ecosystem modelling, seldom taking into account more than one species (here Boncour et al. (2002) and Reithe (forthcoming) are exceptions), and only sporadically taking into account habitats. In the following we will present the few works that do take into account habitat issues and marine reserves, and finally discuss how these may be accommodated in order to study DWC and marine reserves.

In their general marine reserve model presentation, Sanchirico and Wilen (1999, 2001) allow for heterogeneity in habitat, but all analysis assumes that the habitats have the same quality. Schnier (2005), as described above, models heterogeneity of habitats, but assumes that this heterogeneity is independent of the marine reserve implementation itself. Armstrong and Skonhoft (forthcoming) also analyse independent implicit heterogeneity of habitat via the use of varying densities of the targeted species inside and outside the reserve. Rodwell *et al*. (2003) study positive and negative fisheries effects as a function of time with a reserve in place; the positive effects emanating from reduced natural mortality and the negative being reduction in spatial movement out of the reserve, both due to improved habitat within the reserve. Upton and Sutinen (2003), described above, apply their fisheries affected habitat model to a marine reserve management model, studying changes in carrying capacity and intrinsic growth rates. Armstrong (2005) analyses habitat effects upon marine reserves, assuming that the size of the reserve affects the carrying capacity of the resource in question.

Indirect controls

Indirect controls are the use of economic incentive instruments such as taxes or subsidies, or transferable quotas to manage fisheries in an efficient manner. These are not instruments that are often discussed with regard to habitats, but there is however a paper by Holland and Schnier (forthcoming) which describes a bioeconomic simulation model of transferable habitat quotas. Habitat is assumed to regenerate at a given rate (which could be set close to zero for DWC), and decline linearly with harvesting. Transferable habitat impact units (HIU) would be allocated, which allow the fishers to allocate their fishing activity efficiently according to harvesting as well as habitat effects. The authors show how the transferable habitat impact model allows for greater efficiency than a marine protected area management system, but that depending on the degree of diffusion between different areas, the MPA may give higher habitat protection.

Conclusion

From the existing bioeconomic work there seems to be several interesting approaches worth pursuing with regard to the bioeconomic modelling of deep-water coral reef management. These are summarised shortly in the below.

1. Design a bioeconomic model with renewable and non-renewable interaction, where the non-renewable resource (DWC) enters into the growth function of the renewable resource (commercial fish species). Non-use values of the DWC could also be included in the model description.
2. Apply a production function approach to see if one can determine an interaction between coral coverage and fisheries, based on the model in 1 and data on coral decline in a specific area.
3. Study management modelling using gear restrictions/marine reserves/transferable habitat quotas, applied to the model in 1.
4. Design an applied model using a specific fishery in the proximity of DWC; test and simulate management options.

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